Article

Effect of Organic Farming and Agricultural Abandonment on Beneficial Arthropod Communities Associated with Olive Groves in Western Spain: Implications for Bactrocera oleae Management

Víctor de Paz *, Estefanía Tobajas, Natalia Rosas-Ramos , José Tormos , Josep Daniel Asís and Laura Baños-Picón

Department of Biology Animal, Ecology, Parasitology, Edaphology and Agricultural Chemistry, Faculty of Pharmacy, Campus Miguel de Unamuno s/n, Universidad de Salamanca, 37007 Salamanca, Spain; estefaniatob@usal.es (E.T.); nataliarosasr@usal.es (N.R.-R.); tormos@usal.es (J.T.); asis@usal.es (J.D.A.); lbanos@usal.es (L.B.-P.)

* Correspondence: victordepaz177@usal.es

Simple Summary: Olive cultivation has been extremely relevant in the Mediterranean area for centuries, creating traditional landscapes with high cultural and biodiversity values. However, in recent decades, these landscapes have been affected by two processes. On the one hand, the most productive areas have undergone significant intensification, with greater input of agrochemicals and a much higher tree density; on the other hand, marginal areas, with lower production, are being progressively abandoned. While more attention has been paid to intensification effects, few studies have considered the consequences of olive grove abandonment. In our study, we analyzed how abandonment and management regimes (organic or traditional) affected the main olive pest (Bactrocera oleae) and different groups of natural enemies in olive groves established near the border between Spain and Portugal. Our results showed that abandoned and managed olive groves had different, but similarly rich and diverse, communities of natural enemies, highlighting the complementary role that these two habitats play at the landscape scale. Moreover, abandoned groves may not be acting as a reservoir for the olive fly. To prevent land abandonment from continuing, measures such as organic farming or agritourism, which have been implemented in the studied area, could be effective.

Abstract: Agricultural abandonment and intensification are among the main land-use changes in Europe. Along with these processes, different proposals have been developed to counteract the negative effects derived from agricultural intensification, including organic management. In this context, we aimed to determine how organic management and farmland abandonment affect Bactrocera oleae and its main groups of natural enemies: hymenopteran parasitoids, spiders, ants, carabids, and staphylinids. Between May and October 2018, four samplings were carried out in nine olive groves (three under organic management, three under traditional management, and three abandoned) in a rural area on the border between Spain and Portugal (Salamanca, Western Spain). Our results suggested differences between the natural enemy community composition of abandoned and organic groves, with slightly higher levels of richness and abundance in abandoned groves. We found no differences between organic and traditional groves. The managed olive groves sustained a different natural enemy community but were similarly rich and diverse compared with the more complex abandoned groves, with the latter not acting as a reservoir of B. oleae in our study area. Both systems may provide complementary habitats; however, further abandonment could cause a reduction in heterogeneity at the landscape scale and, consequently, a biodiversity loss.

Keywords: abandonment; organic farming; traditional olive groves; Bactrocera oleae; parasitoids; spiders; staphylinids
1. Introduction

Land-use change is the main driver of biodiversity loss worldwide, with the expansion and intensification of agriculture—characterized by an increased input of synthetic pesticides, herbicides, and fertilizers—being the prime cause of insect population declines [1–3]. Biodiversity loss can negatively affect arthropod-mediated ecosystem services, such as pest control by natural enemies [4,5]. To mitigate the negative effects of agricultural intensification, different strategies have been developed, including organic management, in which synthetic pesticides, herbicides, and inorganic fertilizers are avoided [6]. Although there has been considerable controversy regarding the ability of organic management to support greater biodiversity, it has been consistently demonstrated that organic farming increases the richness and abundance of arthropods and other groups [4,6,7]. However, the positive effect of organic farming is highly dependent on the taxon and crop evaluated, and it is greater in annual crops and intensified landscapes [6,7]. Therefore, the landscape context modulates local biodiversity responses to organic farming, as predicted by the intermediate landscape complexity hypothesis [8]. This hypothesis states that in both cleared (<1% of non-crop habitat) and complex landscapes (>20% of non-crop habitat), only small positive responses to local agri-environmental management (such as organic farming) can be expected because of poor species pools and high immigration from semi-natural habitats, respectively, and that only simple landscapes, with intermediate levels of complexity, respond positively to agri-environmental schemes [9]. Moreover, increased biodiversity in organic farming may not happen in permanent crops, such as fruit orchards or vineyards, that present low disturbance levels, and the further reduction in the disturbance intensity of organic farming might not result in greater biodiversity [10]. This is explained by the intermediate disturbance hypothesis, which predicts higher diversity levels at intermediate levels of disturbance, with both slightly and highly disturbed systems harboring less biodiversity [11].

While agricultural intensification usually takes place on fertile soils, in marginal areas with traditional non-intensive farming systems, there is an ongoing process of farmland abandonment [12]. The consequences of the abandonment of traditional crops vary depending on the geographical region, scale, and taxa [13]. In Europe, traditional farming systems have existed for centuries and constitute high-nature-value systems [14], characterized by a low input of pesticides and synthetic fertilizers, with low levels of mechanization and high associated biodiversity, which are currently threatened by land abandonment [15–17]. In fact, Queiroz et al. [17], reviewing farmland abandonment effects on biodiversity, found that in Europe, most studies revealed negative effects. However, the majority of the studies that assessed farmland abandonment effects on arthropods were carried out in grasslands and annual crops in Central and Northern Europe.

In the Mediterranean Basin, a review by Plieninger et al. [18] revealed slightly positive effects of farmland abandonment on biodiversity, although their results were highly heterogeneous, and only four cases related to arthropods in permanent crops were included. They also found a decline in species richness after an abandonment period of fifty or more years that, they suggest, “may indicate that exclusion processes eventually follow colonization processes in many of the case studies.” Actually, many studies found higher biodiversity levels in the early stages of farmland abandonment, which tend to decrease as plant succession progresses [19–22]. In the early stages of farmland abandonment, plant diversity reaches its maximum, with herbaceous plants and scrubs coexisting, which results in habitats with high vegetation complexity [19,23]. Vegetation complexity, measured as structural and chemical complexity, enhances arthropod abundance and diversity by providing more microhabitats and resources, as well as diverse plant volatiles [20,24]. In the late stages of farmland abandonment, scrub and tree species of the surrounding vegetation become dominant, excluding open-habitat species, and reduce heterogeneity at a landscape scale, resulting in reduced biodiversity levels [13].

More work is needed to disentangle the effects of farmland abandonment on arthropod communities, especially in permanent crops, which are underrepresented in the bibliogra-
phy, as well as in the Mediterranean Basin considering that it is one of the world’s diversity hotspots [25] and also an area where land abandonment is prevalent [26]. In this region, landscapes have been shaped by humans for millennia, creating different cultural landscapes that form the identity of the Mediterranean [27]. Such is the case of olive (Olea europaea L.)-dominated landscapes, where olive cultivation has been taking place since the Roman Age [28]. Within the Mediterranean region, Spain, with a production of almost 1.8 tons of olive oil and 10 tons of olives, is the largest producer [29]. Olive grove yields can be affected by a variety of pests, with the olive fruit fly, Bactrocera oleae (Rossi, 1790), being the most relevant [30]. Bactrocera oleae is attacked by a variety of natural enemies and it produces between three and five generations per year in the Mediterranean area, starting in early spring [31,32]. The adult flies oviposit in olives, where the larvae are vulnerable to hymenopteran parasitoids, and after completing their development, the larvae leave the fruits to pupate in the ground, where they are again exposed to generalist predators, mainly spiders, carabids and staphylinids, and ants [33,34].

Despite the importance of the olive groves in the Mediterranean region and specifically in Spain, studies that evaluated the effect of the abandonment of olive groves on the associated arthropod biodiversity are scarce (but see [35] for bees, [36] for isopods, and [37] for butterflies), and to our knowledge, no studies have evaluated the effect of olive grove abandonment on B. oleae and the arthropod groups that include its most important natural enemies, especially in traditional landscapes with different management regimes. In this context, we set out to determine (i) the possible effect of organic management and farmland abandonment on B. oleae and the structure and composition of the natural enemy community in traditional olive groves in a complex landscape; (ii) the differences in richness, abundance, and diversity of natural enemies between organic and abandoned olive groves; and (iii) the response of the dominant natural enemy families and B. oleae to olive grove abandonment. Considering the intermediate disturbance hypothesis and the intermediate landscape complexity hypothesis, we hypothesized that B. oleae and the natural enemy community will barely differ between traditional olive groves (with intermediate levels of perturbation) and organic olive groves (with slightly lower levels of perturbation). Nevertheless, the natural enemy community structure and composition would be expected to differ between managed and abandoned groves, with the latter harboring a richer and more abundant natural enemy community. We also hypothesized that managed groves will host more B. oleae individuals.

2. Materials and Methods
2.1. Study Area

The study was carried out in the municipality of Ahigal de los Aceiteros (Salamanca, Western Spain) (40°52’ N, 6°44’ W). The location of this region, at the Portuguese border, with small villages and limited infrastructure, has led to a continuous process of isolation, depopulation, and farmland abandonment. Conversely, a significant number of the managed plots are transitioning to organic agriculture, with some of them having been organic certified since 2014. Therefore, the agricultural landscape of this region is practically devoid of synthetic pesticides and fertilizers, and structurally, it is a mosaic formed by the combination of managed plots (in many cases with the absence of mechanization and low investment) and abandoned plots in various stages of plant succession, interspersed with fragments of natural vegetation. These remnants of natural vegetation typical of Mediterranean sclerophyllous scrub are formed mainly by Cytisus shrublands (Cytisus multiflorus (L’Hér.) Sweet, Cytisus scoparius (L.) Link), rock rose (Cistus ladanifer L.), French lavender (Lavandula pedunculata (Mill.) Cav.), and thymes (Thymus mastichina (L.) L., Thymus zygis subsp. zygis Loefl. ex L.). There are also areas of Mediterranean forest, mainly composed of holm oaks (Quercus ilex subsp. ballota (Desf.) Samp.) and oaks (Quercus pyrenaica Willd.), although they coexist with European nettle trees (Celtis australis L.) and junipers (Juniperus oxycedrus L.), which have great ecological value and persist mainly because of
the difficulty of cultivating on the steep slopes of the area, which has considerably limited agricultural practices.

The study area comprises 435 ha, its altitude ranging between 405 and 662 m a.s.l. It is located in the transition zone between landscape units 84 (gorges and valleys on the Portuguese border) and 49 (peneplains of Zamora and Salamanca, as well as the foothill of the Montes de León) [38], within the territory of the Arribes del Duero Natural Park (Figure 1). The climate is mild, generally warm, and temperate, with an average annual rainfall of 541 mm and an average annual temperature of 13.7 °C. The predominant soil types are cambisol and leptosol. The study area is bordered by two small streams and in its southwest region by the Águeda River, which runs through a deep canyon. The landscape is dominated by olive groves and, to a lesser extent, by vineyards and almond orchards under a traditional farming system without the application of pesticides and synthetic fertilizers. Olive cultivation is the base of the economic activity within the municipality, with the production and commercialization of extra virgin olive oil and extra virgin organic olive oil as derivative products from different varieties such as Zorzal de Arribes (endemic to Arribes del Duero), Manzanilla Cacerena, and Picual, among others.

**Figure 1.** Location of the study area in Salamanca (Western Spain) and the location of the nine olive groves sampled within the study area. Traditional olive groves: yellow squares, organic groves: red triangles, and abandoned groves: blue circles.
2.2. Sampling Design

We selected nine olive groves: three organic certified, three traditional, and three abandoned (mean area: 12,822.2 ± 4237.3 m², mean distance to the nearest grove: 564.4 ± 120.4 m). The organic olive groves obtained their certification in 2014; therefore, synthetic pesticides and fertilizers have not been used in these groves at least since 2011. In the traditional category, we included orchards under traditional management that did not have an organic certification but where synthetic pesticides and fertilizers have not been used for at least 10 years, according to the owners. We included them in our study to check for differences between organic-certified orchards and traditional orchards that follow an organic-like type of management but that have not been officially certified as organic and where an occasional and infrequent application of synthetic pesticides or fertilizers could still be performed. Ground cover vegetation was controlled in both systems by mowing once or twice a year, all groves are rainfed, and they have an average production of 12 kilos of olives per tree. Finally, all abandoned groves had been abandoned for at least 15 years.

Sampling was performed from May to October 2018 every seven to eight weeks, starting with the flowering period (early May) and finishing just before the olive harvest (late October), attending to the period of highest arthropod abundance [39] and following the *B. oleae* life cycle [40]. Each sampling took place over six consecutive days, randomly assigning the order in which the groves were sampled. Weather conditions were kept as uniform as possible between the sampling periods, avoiding rainy and windy days.

To capture edaphic fauna (spiders, carabids, staphylinids, and ants) six uncovered pitfall traps (9 cm diameter, 12.3 cm depth) were placed in each grove, three under the olive tree canopy and three between rows (54 in total). The traps were filled to a third with a mixture of 70% alcohol and antifreeze (10% ethylene glycol) in a 3:2 ratio (600 mL of alcohol and 400 mL of antifreeze per liter). The traps were also placed 20 m apart from each other and the groves’ edges to reduce trap-to-trap interference and edge effects, and remained in the field for 72 h. To collect hymenopteran parasitoids and vegetation spiders, we randomly selected four trees in each olive grove and vacuumed each tree and the surrounding vegetation in a 2 m × 2 m quadrant for three minutes using a gardener’s leaf-blower (Garland GAS 550 G) [41] modified as a suction machine. For the capture of *B. oleae* specimens, we placed 10 chromatic sticky traps (25 cm × 10 cm) (Koppert Biological Systems—Horiver) in each grove. The traps were hung from the lower branches of the olive trees (at a height of 1.5–2 m above the ground), arranged with a south-facing orientation and separated by a minimum of 10 m from each other, and remained active for 72 h (Figure 2).

All the collected specimens were sorted in the laboratory and identified at the family level. This higher taxa approach (e.g., family taxonomic resolution) was found to be a reliable approach for revealing species richness and compositional patterns [42].
19 estimated spider families, and 80% of the 51 estimated natural enemy families). Therefore, we estimated parasitoid families, 96% of the 25 estimated spider families, and 93% of the 41 estimated parasitoid families, 96% of the 23 estimated parasitoid families, 84% of the 25 estimated spider families, and 93% of the 41 estimated parasitoid families, 96% of the 23 estimated parasitoid families, 84% of the 25 estimated spider families, and 93% of the 41 estimated parasitoid families, 96% of the 23 estimated parasitoid families, 84% of the 25 estimated spider families, and 93% of the 41 estimated parasitoid families, 96% of the 23 estimated parasitoid families, 84% of the 25 estimated spider families, and 93% of the 41 estimated parasitoid families, 96% of the 23 estimated parasitoid families, 84% of the 25 estimated spider families, and 93% of the 41 estimated parasitoid families, 96% of the 23 estimated parasitoid families, 84% of the 25 estimated spider families, and 93% of the 41 estimated parasitoid families, 96% of the 23 estimated parasitoid families, 84% of the 25 estimated spider families, and 93% of the 41 estimated parasitoid families, 96% of the 23 estimated parasitoid families, 84% of the 25 estimated spider families, and 93% of the 41 estimated parasitoid families, 96% of the 23 estimated parasitoid families, 84% of the 25 estimated spider families, and 93% of the 41 estimated parasitoid families, 96% of the 23 estimated parasitoid families, 84% of the 25 estimated spider families, and 93% of the 41 estimated parasitoid families, 96% of the 23 estimated parasitoid families, 84% of the 25 estimated spider families, and 93% of the 41 estimated parasitoid families, 96% of the 23 estimated parasitoid families, 84% of the 25 estimated spider families, and 93% of the 41 estimated parasitoid families, 96% of the 23 estimated parasitoid families, 84% of the 25 estimated spider families, and 93% of the 41 estimated parasitoid families, 96% of the 23 estimated parasitoid families, 84% of the 25 estimated spider families, and 93% of the 41 estimated parasitoid families, 96% of the 23 estimated parasitoid families, 84% of the 25 estimated spider families, and 93% of the 41 estimated parasitoid families, 96% of the 23 estimated parasitoid families, 84% of the 25 estimated spider families, and 93% of the 41 estimated parasitoid families, 96% of the 23 estimated parasitoid families, 84% of the 25 estimated spider families, and 93% of the 41 estimated parasitoid families, 96% of the 23 estimated parasitoid families, 84% of the 25 estimated spider families, and 93% of the 41 estimated parasitoid families, 96% of the 23 estimated parasitoid families, 84% of the 25 estimated spider families, and 93% of the 41 estimated parasitoid families, 96% of the 23 estimated parasitoid families, 84% of the 25 estimated spider families, and 93% of the 41 estimated parasitoid families, 96% of the 23 estimated parasitoid families, 84% of the 25 estimated spider families, and 93% of the 41 estimated parasitoid families, 96% of the 23 estimated parasitoid families, 84(162,401),(854,994)
and the site as a random factor (when applicable) on the family richness, abundance, and Shannon’s index values for the natural enemies, spiders, and parasitoids and for the abundance of the most dominant families of the whole natural enemy community (>60 individuals, 17 out of 52 families). The residuals of each model were checked to ensure normality, independence, and homoscedasticity. Additionally, a factorial correspondence analysis (FCA) was carried out to represent the association between these families and the two types of olive groves (abandoned and organic).

To analyze the possible effect of the system (organic certified, traditional, and abandoned) on the populations of B. oleae, we applied a generalized linear mixed-effects model (GLMM), with the variables system, the sampling month, and their interaction as fixed factors, and the site as a random factor. Since we did not detect any B. oleae adults in the two summer sampling periods, we excluded them from the analysis, as the high number of zeros could be a source of error in the analysis. Then, we performed the analysis only with the data from the first and last sampling periods (May and October, respectively).

For the analyses, the statistical packages PRIMER v6 (PERMANOVA, MDS) (PRIMER-E Ltd., Plymouth, UK) [46], R 3.6.2 (linear least-squares models, linear mixed models, generalized linear models, generalized linear mixed models, and zero-inflated models) [47], and XLStat 2014 (factorial correspondence analysis) [48] were used.

Table 1. Results of the PERMANOVA for the variables system and sampling month (full model results and pairwise comparisons among the three systems).

| Response Variable | Explanatory Variables | d.f. | Pseudo-F | p-Value |
|-------------------|-----------------------|------|----------|---------|
| Natural enemies   | System                | 2    | 2.3117   | 0.0001  |
|                   | Sampling month        | 3    | 8.9107   | 0.0001  |
| Spiders           | System                | 2    | 3.0515   | 0.0001  |
|                   | Sampling month        | 3    | 6.2039   | 0.0001  |
| Parasitoids       | System                | 2    | 0.7927   | 0.7016  |
|                   | Sampling month        | 3    | 5.3291   | 0.0001  |

Pairwise Comparisons

| Response Variable | Explanatory Variables | Pseudo-t | p-Value |
|-------------------|-----------------------|----------|---------|
| Natural enemies   | Abandoned, organic    | 1.7466   | 0.0001  |
|                   | Abandoned, traditional| 1.7311   | 0.0002  |
|                   | Organic, traditional  | 0.9225   | 0.6299  |
|                   | Abandoned, organic    | 2.0979   | 0.0001  |
| Spiders           | Abandoned, traditional| 2.0336   | 0.0001  |
|                   | Organic, traditional  | 0.7412   | 0.8070  |
|                   | Abandoned, organic    | 0.6894   | 0.8338  |
| Parasitoids       | Abandoned, traditional| 1.0252   | 0.4277  |
|                   | Organic, traditional  | 0.9476   | 0.5434  |

3. Results

A total of 13,300 arthropods belonging to the focal groups were collected—1730 spiders (26 families), 2125 parasitoids (22 families), 27 carabids, 255 staphylinids, 8483 ants and, and 680 B. oleae individuals.

The PERMANOVA revealed a significant effect of the system on the natural enemy and spider communities (pseudo-F = 2.312, p = 0.0001 and pseudo-F = 3.052, p = 0.0001, respectively) but none for the parasitoid community (pseudo-F = 0.7927, p = 0.702); these differences occurred only between the managed and abandoned systems but not between the organic and traditional plots (Table 1). These results were also noticeable in the MDS, where, for the natural enemy and spider communities, the abandoned groves were clearly separated from the managed groves, which was not the case for the parasitoid community (Figure 3).
Insects 2022, 13, 48

![Figure 3. MDS of the natural enemy (A), spider (B), and parasitoid (C) communities sampled (Bray–Curtis index, square-root transformed abundances). Triangles: organic groves; circles: abandoned groves; squares: traditional groves.](image)

When considering the whole natural enemy community, we did not find differences in richness, abundance, or diversity between organic and abandoned groves (Table 2). Nevertheless, the models showed that spiders were significantly affected by the system, with the abandoned plots harboring richer and more abundant communities, although we also found a marginally significantly higher spider diversity in the organic groves toward the end of the sampling period. Parasitoids were partly affected by the system, showing no differences in richness or abundance, but their diversity was significantly higher in the abandoned plots. This difference decreased throughout the sampling period, and at the end of the season, organic plots harbored a higher parasitoid diversity (Table 2), showing a trend similar to that of the spiders.

**Table 2.** Results of the different models for the richness, abundance, and diversity (Shannon index (H)) of natural enemies, spiders, and parasitoids. Estimates, standard errors, test statistics, p-values, and significance levels (ns > 0.1, · < 0.1, * < 0.05, and *** < 0.001) for the intercept and the explanatory variable system, as well as the interaction between the variables system and the sampling month (when significant), are provided. The complete results, including the explanatory variable sampling month, are given in Table S1.

| Response Variable | Explanatory Variable | Value/Estimate | Std. Error | t-/Value| p-Value | * |
|-------------------|----------------------|----------------|------------|---------|---------|---|
| Natural enemy family richness a | System (organic) | 0.667 | 1.080 | -0.617 | <0.001 | *** |
| Natural enemy abundance a | System (organic) | 247.081 | 33.023 | 7.482 | <0.001 | *** |
| Natural enemy diversity (H) a | System (organic) | 4.5057 | 31.15 | 0.144 | 0.887 | ns |
| Spider family richness (square root) a | System (organic) | 2.838 | 0.094 | 30.134 | <0.001 | *** |
| Spider abundance b | System (organic) | 3.200 | 0.126 | 25.352 | <0.001 | *** |
| Spider diversity (H) a | System (organic) | 1.557 | 0.159 | 9.789 | <0.001 | *** |

Sampling month (June): system (organic) | Inter | 0.617 | 0.544 | ns |
Sampling month (August): system (organic) | 0.667 | 1.080 | ns |
Sampling month (October): system (organic) | 0.467 | 0.247 | 1.894 | 0.077 | ns |
Table 2. Cont.

| Response Variable                       | Explanatory Variable     | Value/Estimate | Std. Error | t-Value/z-Value | p-Value *   |
|-----------------------------------------|--------------------------|----------------|------------|----------------|-------------|
| Parasitoid family richness c            | Intercept                | 2.368          | 0.107      | 22.136         | <0.001 ***  |
|                                          | System (organic)         | −0.034         | 0.099      | −0.347         | 0.732 ns    |
| Parasitoid abundance d                  | Intercept                | 48.917         | 8.685      | 5.632          | <0.001 ***  |
|                                          | System (organic)         | −5.833         | 10.328     | −0.565         | 0.602 ns    |
| Parasitoid diversity a                  | Intercept                | 2.144          | 0.130      | 16.445         | <0.001 ***  |
|                                          | System (organic)         | −0.473         | 0.184      | −2.568         | 0.021 *     |
|                                          | Sampling month (June): system (organic) | 0.421 | 0.200 | 2.103 | 0.052 ns |
|                                          | Sampling month (August): system (organic) | 0.108 | 0.277 | 0.389 | 0.702 ns |
|                                          | Sampling month (October): system (organic) | 0.701 | 0.303 | 2.317 | 0.034 * |

* a GLS; b GLMM with a Poisson distribution; c quasi-GLM model; d LME.

The results of the linear models applied to the most abundant families showed different responses to the variable system across taxa. Nevertheless, we found a general tendency of natural enemy families to associate with abandoned groves (nine out of seventeen), with only four families being more abundant in the organic groves (Gnaphosidae, Linyphiidae, Philodromidae, and Staphylinidae) (Table 3). Four families were not associated with either type of system (Formicidae, Pteromalidae, Salticidae, and Scelionidae). These results were consistent with those of the correspondence analysis (Figure 4), except for four families. Mymaridae and Encyrtidae were more abundant in the abandoned groves according to the models, but they were not associated with these groves in the correspondence analysis; the contrary applied to the family Pteromalidae, which was associated with the abandoned groves in the correspondence analysis but not according to the generalized linear mixed model. The results of the model for the family Philodromidae revealed a significantly higher abundance in the organic groves that was not supported by the correspondence analysis (Figure 4). In the case of the main olive pest, B. oleae, the managed groves harbored significantly higher abundance than the abandoned groves in October (Table 3). In May, the B. oleae abundance was too low to detect differences between systems (105 versus 575 individuals in October). We did not capture any adults during the summer sampling.

Table 3. Results of the different linear models for the abundance of the most dominant families. Estimates, standard errors, test statistics, p-values, and significance levels (ns > 0.1, · < 0.1, * < 0.05, ** < 0.01, and *** < 0.001) for the intercept and the explanatory variable system, as well as the interaction between the variables system and the sampling month (when significant), are provided. No results are shown for Formicidae because the variable system did not remain in the optimal model. The complete results, including the explanatory variable sampling month, are given in Table S2.

| Response Variable                  | Explanatory Variable             | Value/Estimate | Std. Error | t-Value/z-Value | p-Value *   |
|------------------------------------|----------------------------------|----------------|------------|----------------|-------------|
| Araneidae abundance a              | Intercept                        | 1.680          | 0.244      | 6.875          | <0.001 ***  |
|                                    | System (organic)                 | −1.069         | 0.244      | −4.376         | <0.001 ***  |
| Gnaphosidae abundance a            | Intercept                        | 0.784          | 0.491      | 1.597          | 0.110 ns    |
|                                    | System (organic)                 | 0.584          | 0.650      | 0.899          | 0.369 ns    |
|                                    | Sampling month (June): system (organic) | −0.657 | 0.557 | −1.179 | 0.238 ns |
|                                    | Sampling month (August): system (organic) | 0.208 | 0.601 | 0.346 | 0.730 ns |
|                                    | Sampling month (October): system (organic) | 1.594 | 0.865 | 1.843 | 0.065 ns |
### Table 3. Cont.

| Response Variable                      | Explanatory Variable | Value/Estimate | Std. Error | t-Value/z-Value | p-Value       | * |
|----------------------------------------|----------------------|----------------|------------|----------------|---------------|---|
| Linyphiidae abundance                  | Intercept            | 0.654          | 0.474      | 1.379          | 0.168         | ns |
|                                        | System (organic)     | 0.758          | 0.599      | 1.266          | 0.205         | ns |
|                                        | Sampling month (June): system (organic) | 1.391          | 0.608      | 2.287          | 0.022         | * |
|                                        | Sampling month (August): system (organic) | −5.11E-05      | 0.625      | 0            | 0.999         | ns |
|                                        | Sampling month (October): system (organic) | −0.074         | 0.688      | −0.108         | 0.914         | ns |
| Oxyopidae abundance                   | Intercept            | 0.834          | 0.388      | 2.152          | 0.031         | * |
|                                        | System (organic)     | −1.946         | 1.068      | −1.823         | 0.068         | - |
|                                        | Sampling month (June): system (organic) | −0.251         | 1.500      | −0.168         | 0.867         | ns |
|                                        | Sampling month (August): system (organic) | −1.571         | 0.538      | −2.92          | 0.004         | ** |
|                                        | Sampling month (October): system (organic) | −2.683         | 0.624      | −4.301         | <0.001        | *** |
| Philodromidae abundance (square root) b | Intercept            | 0.334          | 0.378      | 0.884          | 0.388         | ns |
|                                        | System (organic)     | 0.576          | 0.114      | 5.058          | <0.001        | *** |
| Salticidae abundance b                 | Intercept            | 2.000          | 0.645      | 3.098          | 0.006         | ** |
|                                        | System (organic)     | −0.667         | 0.577      | −1.155         | 0.263         | ns |
| Theridiidae abundance a                | Intercept            | 0.245          | 0.466      | 0.525          | 0.600         | ns |
|                                        | System (organic)     | −1.287         | 0.310      | −4.15          | <0.001        | *** |
| Thomisidae abundance a                 | Intercept            | 0.835          | 0.390      | 2.141          | 0.032         | * |
|                                        | System (organic)     | −0.002         | 0.550      | −0.004         | 0.997         | ns |
|                                        | Sampling month (June): system (organic) | 0.167          | 0.673      | 0.248          | 0.804         | ns |
|                                        | Sampling month (August): system (organic) | −1.447         | 0.770      | −1.879         | 0.060         | - |
|                                        | Sampling month (October): system (organic) | −0.442         | 0.683      | −0.646         | 0.518         | ns |
| Braconidae abundance (square root) b   | Intercept            | 2.943          | 0.529      | 5.567          | <0.001        | *** |
|                                        | System (organic)     | −2.000         | 0.748      | −2.675         | 0.017         | * |
|                                        | Sampling month (June): system (organic) | 1.576          | 0.835      | 1.889          | 0.077         | - |
|                                        | Sampling month (August): system (organic) | 1.805          | 0.857      | 2.105          | 0.051         | - |
|                                        | Sampling month (October): system (organic) | 2.667          | 0.933      | 2.858          | 0.011         | * |
| Encyrtidae abundance a                 | Intercept            | 1.194          | 0.334      | 3.576          | <0.001        | *** |
|                                        | System (organic)     | −1.625         | 0.789      | −2.061         | 0.039         | * |
|                                        | Sampling month (June): system (organic) | 2.197          | 0.837      | 2.625          | 0.007         | ** |
|                                        | Sampling month (August): system (organic) | −0.056         | 0.876      | −0.064         | 0.949         | ns |
|                                        | Sampling month (October): system (organic) | 3.091          | 0.848      | 3.644          | <0.001        | *** |
| Eulophidae abundance a                 | Intercept            | 1.814          | 0.200      | 9.058          | <0.001        | *** |
|                                        | System (organic)     | −0.321         | 0.173      | −1.853         | 0.064         | - |
| Mymaridae abundance b                  | Intercept            | 1.843          | 0.197      | 9.337          | <0.001        | *** |
|                                        | System (organic)     | −0.530         | 0.175      | −3.034         | 0.007         | ** |
| Pteromalidae abundance a               | Intercept            | 0.668          | 0.530      | 1.259          | 0.208         | ns |
|                                        | System (organic)     | −0.372         | 0.664      | −0.561         | 0.575         | ns |
| Scelionidae abundance a                | Intercept            | 2.557          | 0.133      | 19.257         | <0.001        | *** |
|                                        | System (organic)     | 0.169          | 0.120      | 1.401          | 0.161         | ns |
| Trichogrammatidae abundance b          | Intercept            | 1.426          | 0.233      | 6.126          | <0.001        | *** |
|                                        | System (organic)     | −0.600         | 0.228      | −2.634         | 0.016         | * |
| Staphylinidae abundance a              | Intercept            | 1.936          | 0.353      | 5.479          | <0.001        | *** |
|                                        | System (organic)     | 1.405          | 0.469      | 2.998          | 0.003         | ** |
In accordance with our hypothesis, the structure of the natural enemy community differed between the abandoned and organic olive groves, except when considering parasitoid wasps alone. Parasitoid wasps were the most mobile group in the study, and they tend to be more affected by variables at a landscape scale [49–51]. They were most likely moving along patches adjacent to the groves in the matrix, using the multiple resources in abandoned groves and semi-natural habitats for foraging, refuge, and alternative hosts [52], and spilling over to managed groves, mostly searching for hosts [53]. In agreement with our hypothesis, we found no differences between organic and traditional groves. The landscape complexity of our study area would explain the lack of effect of the farming system (according to the intermediate disturbance hypothesis) [9], where local extensification measures, such as organic farming, are expected to have little effect on species richness [8]. Moreover, Bruggisser et al. [10] suggested that “the biodiversity benefits of organic farming

**Table 3. Cont.**

| Response Variable | Explanatory Variable | Value/Estimate | Std. Error | t-Value/z-Value | p-Value | * |
|-------------------|----------------------|----------------|------------|----------------|---------|---|
| Bactrocera oleae abundance | Intercept | 1.662 | 0.566 | 2.938 | 0.003 | ** |
|                    | System (traditional) | 0.346 | 0.788 | 0.439 | 0.660 | ns |
|                    | System (organic) | 0.859 | 0.775 | 1.108 | 0.268 | ns |
|                    | Sampling month (October): system (traditional) | 1.820 | 0.302 | 6.027 | <0.001 | *** |
|                    | Sampling month (October): system (organic) | 1.531 | 0.263 | 5.817 | <0.001 | *** |

* GLMM with a Poisson distribution; GLS.

**Figure 4.** Factorial correspondence analysis was performed on the abundance of the most relevant families of natural enemies on each type of system (organic and abandoned). Families in bold with black drawings were those significantly affected by the variable system, whereas underlined families with grey drawings were marginally affected by this variable, according to the results obtained from the linear models fitted on the abundances of the most dominant families. Red dot: principal coordinate value for each family in the first axis of the FCA; blue dot: principal coordinate value for each farming system in the first axis of the FCA. Significant association with the organic groves in the correspondence analysis; * significant association with the abandoned groves in the correspondence analysis.

4. Discussion

In accordance with our hypothesis, the structure of the natural enemy community differed between the abandoned and organic olive groves, except when considering parasitoid wasps alone. Parasitoid wasps were the most mobile group in the study, and they tend to be more affected by variables at a landscape scale [49–51]. They were most likely moving along patches adjacent to the groves in the matrix, using the multiple resources in abandoned groves and semi-natural habitats for foraging, refuge, and alternative hosts [52], and spilling over to managed groves, mostly searching for hosts [53]. In agreement with our hypothesis, we found no differences between organic and traditional groves. The landscape complexity of our study area would explain the lack of effect of the farming system (according to the intermediate landscape complexity hypothesis) [9], where local extensification measures, such as organic farming, are expected to have little effect on species richness [8]. Moreover, Bruggisser et al. [10] suggested that “the biodiversity benefits of organic farming...
in annual cropping systems may not hold for perennial crops, particularly if the use of pesticides is minimal,” based on the intermediate disturbance hypothesis [11]. Nevertheless, the similarity between organic and traditional groves also indicated that traditional agriculture, with a very occasional input of synthetic pesticides and fertilizers, may be able to maintain similar levels of natural enemy diversity as organic agriculture. Therefore, our results, in agreement with those of other studies, highlighted traditional agriculture as high-nature-value farming systems that supported high biodiversity in agricultural landscapes [54,55].

As we expected, the main olive pest, namely, *B. oleae*, was associated with the managed groves. In the abandoned groves, given the lack of management (especially pruning), olive trees produced less fruit; therefore, olive availability for *B. oleae* oviposition was scarce, and few adults developed in unmanaged habitats. In fact, olive production was so low that the *B. oleae* adults that we found in the abandoned groves may have dispersed from the managed groves. The abandoned groves would not act as reservoirs of *B. oleae*; they may even be sink habitats (i.e., resource-poor habitats where the death rate exceeds the birth rate and the populations are maintained, in the long term, by immigration [56]) because of the higher rate of natural enemies, functioning similarly to trap crops, which attract pests and induce a higher rate of pest mortality (e.g., through reduced larval survival or suppression by natural enemies) [57,58].

Surprisingly, we did not find differences in richness, abundance, or diversity for the whole natural enemy community between organic and abandoned olive groves, even though the structure of the communities tended to diverge. Both organic and abandoned groves would provide valuable and diverse microhabitats that harbor different natural enemy communities but be similarly rich and diverse given their high niche availability [15,59]. Abandoned groves, with an intermediate structure between the more open habitat of managed groves, and the closer one of the Mediterranean shrubland and forest, with structurally dense vegetation, maintain diverse communities that are different from managed groves and probably also from the natural habitats that surround them (see [22] for an example with spiders in European grasslands). Consequently, abandoned groves may increase landscape complexity, as well as microhabitat and resource availability, in agreement with other studies in different ecosystems (e.g., [59] for fruit orchards and [22] for grasslands; see [18] for a review).

Focusing on the most important groups of natural enemies considered in this study, namely, spiders and parasitoids, we found that the abandoned groves harbored richer and more abundant spider communities and higher parasitoid diversity. The structurally more complex vegetation of abandoned groves provided a wider range of microhabitats and especially more resources for feeding and refuge, which may have favored these groups. Several studies have shown that spiders are constrained by different habitat features at a local scale (e.g., [60,61]), favoring habitats with more complex vegetation architecture [62–65] and increased prey availability [66,67]. Therefore, the characteristics of the abandoned groves allowed them to sustain richer and more abundant spider communities. Concerning parasitoid wasps, the abandoned groves had a higher habitat complexity, which was found to enhance natural enemies in olive groves, but with little effects in parasitoid abundance [68]. Not only do these groves provide more resources (i.e., floral resources, overwintering sites, alternative hosts) that favor parasitoids [69] but also this may reduce negative interactions (such as intraguild predation, competition, or hyperparasitism) [70,71], which result in a more even community with higher diversity values.

Regarding community changes throughout the season, we detected a change of trend in the diversity observed in the two systems, with the organic groves harboring a higher diversity of both spiders and parasitoids than the abandoned groves toward the end of the sampling period. In the studied areas, farmers managed the natural vegetation cover in productive groves through conservation tillage (i.e., a form of non-intensive tillage that leaves at least 30% of the previous cover’s residue on the surface [72]) in July. It was shown that rapid perturbations, such as tillage, predominantly affect dominant species [73], reducing competition over hosts/prey and other resources. Therefore, the organic groves
harbored a slightly higher richness of spiders and parasitoids in October, which, when combined with the lower abundance found in this month, resulted in a significantly higher diversity. In fact, conservation tillage was shown to favor predators and parasitoids in simple and complex landscapes [74] and to increase predator diversity and evenness [75]. Our results are in accordance with those of previous studies that highlight that mild disturbances of traditional agroecosystems can favor biodiversity [19,21,54,55,59].

According to our expectations, the abandoned groves achieved a higher abundance of the most dominant families of natural enemies in general but with clear differences between taxa. Five out of seven parasitoid families were more abundant in the abandoned groves, with none associating with the organic groves. Parasitoids belong to a high trophic level, and they are especially sensitive to environmental change and agricultural disturbances [76,77]; thus, we would have expected to find differences in the community composition or shifts in richness or abundance between these two systems. A possible explanation is that an evaluation at the family level may not provide sufficient resolution to detect these differences [78]. However, when studying the dominant families separately, we do see a clear association of these families with the abandoned olive groves, again emphasizing their higher abundance of food and shelter resources that would favor parasitoids [69], and also their heightened chemical complexity, which may attract a greater diversity of parasitoids [24,79].

For the spiders, we found mixed results across families. The families Araneidae and Theridiidae (space and orb-web weavers, respectively [62,80]) and Oxyopidae and Thomisidae (active hunters and ambush hunters, respectively, mostly on the vegetation, with the latter especially on flowers [62,80]) are more abundant in the abandoned groves. As previously mentioned, these families probably favored the higher plant structural complexity of the abandoned groves, which provided more anchoring points for the webs of theridiids and araneids and foraging, as well as refuge resources for oxyopids and thomisids, along with a variety of locations for ambushing prey for the latter [62,63,67]. On the other hand, the families Gnaphosidae, Linyphiidae, and Philodromidae are more abundant in the organic groves. Gnaphosids are ground hunters, which favor open habitats, such as those of the organic groves [64,81,82]. Philodromids are active hunters either on the ground or the vegetation, and linyphiids include sheet-web weavers and also active hunters, both groups with ground and vegetation species. These two families may be more abundant in the organic groves because of less competition with species from other families that need more structurally complex habitats to flourish. Another possible explanation is that organic groves have filtered some ground-dwelling species of these families that occur mainly in open habitats [63,83] and that are dominating the spider community in these groves.

Staphylinids were more abundant in the organic groves, in contrast with the results of Baloj and Markó [84,85], who found more abundant staphylinid communities in abandoned apple orchards and vineyards than in productive fields (conventional and IPM and conventional and organic, respectively). This result may indicate that the community composition of staphylinids in our study area was dominated by generalist and open-habitat species that favored the agricultural management of the organic groves [86].

Four families were not associated with either system. In the case of scelionids and ptinomaiids, some species may favor the organic groves, masking the possible differences between organic and abandoned groves (e.g., parasitoids of ground beetles, which are more abundant in the organic groves, or parasitoids of olive pests). We also did not find any differences in ant abundance. Since we did not sort them into species or functional groups, we cannot assess the structure of the ant community. Based on the few studies that compare ant diversity in abandoned and managed agricultural fields ([87] in orchards and [88] in cereal fields), we can hypothesize that abandoned groves may harbor a different, more diverse ant community because of their higher niche and different resource availability, with organic groves harboring fewer species, but they are exploiting the resources successfully, resulting in similar abundance levels. Finally, the family Salticidae includes some species
adapted to hunting on the ground, while others hunt on the vegetation [80]; thus, open-habitat species may be more abundant in organic groves, and the more complex vegetation of abandoned groves may sustain more individuals of species that hunt on the vegetation, although they tend to prefer widely spaced over dense structures [89].

Although arthropod interannual population variability may result in biased conclusions [90], short-term studies provide valuable results that are useful for making local comparisons [91], and most studies conducted with arthropods in agroecosystems have one-year sampling periods. In any case, our results should be interpreted with caution due to their dependence on spatial and temporal context.

5. Conclusions

Our study provided the first evidence of the effects of olive grove abandonment on *B. oleae* and different groups of natural enemies in traditional agroecosystems and contributed to increasing the knowledge of these effects on woody crops, which are clearly underrepresented in the literature, even when considering the limitations derived by the fact that our sampling period was restricted to one year. The studied traditional olive groves may be able to sustain a different natural enemy community (but equally rich and diverse) than that of the more structurally complex abandoned groves, emphasizing the role that traditional olive groves have in conserving biodiversity in agricultural landscapes. Abandoned groves could constitute landscape elements of high importance that provide a wide variety of microhabitats and resources that are exploited by natural enemies but not by the olive fruit fly and, therefore, may not act as reservoirs for this pest in our study area. The presence of both systems, traditional and abandoned groves, at a landscape scale provides complimentary habitats that promote diverse communities of some beneficial groups, such as spiders. However, some management measures should be carried out in the abandoned groves to halt plant succession, preventing them from becoming Mediterranean scrubland or forest, similar to the one surrounding them, which would result in the loss of landscape-scale heterogeneity and biodiversity. A greater priority is to stop the abandonment of traditional olive groves that are still in use. The high biodiversity of these groves depends on their traditional management, and further abandonment would translate not only into the loss of the aesthetic and cultural values associated with these agroecosystems but also the possibly irretrievable loss of the biodiversity that they harbor. For this reason, the measures aimed at reducing the depopulation of rural areas, where these traditional systems occur, are especially important to stop the abandonment process. Organic agriculture—with the associated increase in the price of olive oil, which results in higher profits for farmers—and agricultural tourism are measures that were shown to be very effective in halting this process in our study area. More studies are needed to disentangle the effects of olive grove abandonment on the associated arthropod communities, especially in different regions and with longer sampling periods.

**Supplementary Materials:** The following are available online at https://www.mdpi.com/article/10.3390/insects13010048/s1, Table S1: Complete results of the different models for the richness, abundance, and diversity (Shannon index (H)) of natural enemies, spiders, and parasitoids; Table S2: Complete results of the different models for the abundance of the most dominant families and *B. oleae*; Supplementary Data Matrix: Our complete data set.

**Author Contributions:** Conceptualization: V.d.P., E.T., N.R.-R., J.T., J.D.A. and L.B.-P.; methodology: V.d.P., E.T., N.R.-R., J.T., J.D.A. and L.B.-P.; formal analysis: V.d.P., E.T., N.R.-R., J.D.A. and L.B.-P.; data curation: V.d.P., E.T., N.R.-R., J.D.A. and L.B.-P.; writing—original draft preparation: V.d.P.; writing—review and editing: V.d.P., E.T., N.R.-R., J.T., J.D.A. and L.B.-P. All authors have read and agreed to the published version of the manuscript.

**Funding:** This research was funded by Programa I USAL 2017/00129/001; 2019-00, and Víctor de Paz was funded by the Ministerio de Educación y Formación Profesional, grant number FPU17/03556.

**Institutional Review Board Statement:** Not applicable.
Data Availability Statement: The data presented in this study are available as Supplementary Materials (Supplementary Data Matrix).

Acknowledgments: The authors wish to thank the farmers for allowing access to their groves, specifically Loli Sánchez, for her help in finding and accessing the groves, and the Servicio Territorial de Medio Ambiente de Castilla y León for providing the necessary permits to carry out the samplings. We also wish to thank Sara Vicente Alonso for her help with all the fieldwork. We appreciate the comments and suggestions of three anonymous reviewers that have greatly increased the quality of the manuscript.

Conflicts of Interest: The authors declare no conflict of interest. The funders had no role in the design of the study; in the collection, analyses, or interpretation of data; in the writing of the manuscript; or in the decision to publish the results.

References
1. Tilman, D.; Cassman, K.G.; Matson, P.A.; Naylor, R.; Polasky, S. Agricultural sustainability and intensive production practices. Nature 2002, 418, 671–677. [CrossRef]
2. Bongaarts, J. IPBES Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. Popul. Dev. Rev. 2019, 45, 680–681.
3. Raven, P.H.; Wagner, D.L. Agricultural intensification and climate change are rapidly decreasing insect biodiversity. Proc. Natl. Acad. Sci. USA 2021, 118, 1–6. [CrossRef]
4. Lichtenberg, E.M.; Kennedy, C.M.; Kremen, C.; Batáry, P.; Berendse, F.; Bommard, R.; Bosque-Pérez, N.A.; Carvalheiro, L.G.; Snyder, W.E.; Williams, N.M.; et al. A global synthesis of the effects of diversified farming systems on arthropod diversity within fields and across agricultural landscapes. Glob. Chang. Biol. 2017, 23, 4946–4957. [CrossRef] [PubMed]
5. Dainese, M.; Martin, E.A.; Albrecht, M.; Bartomeus, I.; Bommard, R.; Carvalheiro, L.G.; Chaplin-Kramer, R.; Gagic, V.; Garibaldi, L.A.; et al. A global synthesis reveals biodiversity-mediated benefits for crop production. Sci. Adv. 2019, 5, eaax0121. [CrossRef]
6. Tuck, S.L.; Winqvist, C.; Mota, F.; Ahnström, J.; Turnbull, L.A.; Bengtsson, J. Land-use intensity and the effects of organic farming on biodiversity: A hierarchical meta-analysis. J. Appl. Ecol. 2014, 51, 746–755. [CrossRef]
7. Bengtsson, J.; Ahnström, J.; Weibull, A.C. The effects of organic agriculture on biodiversity and abundance: A meta-analysis. J. Appl. Ecol. 2005, 42, 261–269. [CrossRef]
8. Concepción, E.D.; Díaz, M.; Kleinj, D.; Bäldi, A.; Batáry, P.; Clough, Y.; Gabriel, D.; Herzog, F.; Holzschuh, A.; Knop, E.; et al. Interactive effects of landscape context constrain the effectiveness of local agri-environmental management. J. Appl. Ecol. 2012, 49, 695–705. [CrossRef]
9. Tscharntke, T.; Tylianakis, J.M.; Rand, T.A.; Didham, R.K.; Fahrig, L.; Batáry, P.; Bengtsson, J.; Clough, Y.; Crist, T.O.; Dormann, C.F.; et al. Landscape moderation of biodiversity patterns and processes—Eight hypotheses. Biol. Rev. 2012, 87, 661–685. [CrossRef] [PubMed]
10. Bruggisser, O.T.; Schmidt-Entling, M.H.; Bacher, S. Effects of vineyard management on biodiversity at three trophic levels. Biol. Conserv. 2010, 143, 1521–1528. [CrossRef]
11. Connell, J.H. Diversity in Tropical Rain Forests and Coral Reefs. Science 1978, 199, 1302–1310. [CrossRef]
12. Cramer, V.; Hobbs, R.; Standish, R. What’s new about old fields? Land abandonment and ecosystem assembly. Trends Ecol. Evol. 2008, 23, 104–112. [CrossRef] [PubMed]
13. Benayas, J.M.R.; Martins, A.; Nicolau, J.M.; Schulz, J.J. Abandonment of agricultural land: An overview of drivers and consequences. CAB Rev. Perspect. Agric. Vet. Sci. Nutr. Nat. Resour. 2007, 2, 1–14. [CrossRef]
14. Baldock, D.; Beaufoy, G.; Bennett, G.; Clark, J. Nature Conservation and New Directions in the Common Agricultural Policy: The Potential Role of EC Policies in Maintaining Farming and Management Systems of High Nature Value in the Community; Institute for European Environmental Policy (IEEP): London, UK, 1993; ISBN 90-74095-03-8.
15. Fischer, J.; Hartel, T.; Kuemmerle, T. Conservation policy in traditional farming landscapes. Conserv. Lett. 2012, 5, 167–175. [CrossRef]
16. Keenleyside, C.; Beaufoy, G.; Tucker, G.; Jones, G. High Nature Value Farming throughout EU-27 and Its Financial Support under the CAP; Institute for European Environmental Policy (IEEP): London, UK, 2014; ISBN 9789279379581.
17. Queiroz, C.; Bellin, R.; Folke, C.; Lindborg, R. Farmland abandonment: Threat or opportunity for biodiversity conservation? A global review. Front. Ecol. Environ. 2014, 12, 288–296. [CrossRef]
18. Plieninger, T.; Hui, C.; Gaertner, M.; Huntsinger, L. The Impact of Land Abandonment on Species Richness and Abundance in the Mediterranean Basin: A Meta-Analysis. PLoS ONE 2014, 9, e93355. [CrossRef]
19. Cremene, C.; Groza, G.; Rakosy, L.; Schileo, A.A.; Baur, A.; Erhardt, A.; Baur, B. Alterations of Steppe-Like Grasslands in Eastern Europe: A Threat to Regional Biodiversity Hotspots. Conserv. Biol. 2005, 19, 1606–1618. [CrossRef]
20. Öckinger, E.; Eriksson, A.K.; Smith, H.G. Effects of grassland abandonment, restoration and management on butterflies and vascular plants. Biol. Conserv. 2006, 133, 291–300. [CrossRef]

Acknowledgments: The authors wish to thank the farmers for allowing access to their groves, specifically Loli Sánchez, for her help in finding and accessing the groves, and the Servicio Territorial de Medio Ambiente de Castilla y León for providing the necessary permits to carry out the samplings. We also wish to thank Sara Vicente Alonso for her help with all the fieldwork. We appreciate the comments and suggestions of three anonymous reviewers that have greatly increased the quality of the manuscript.

Conflicts of Interest: The authors declare no conflict of interest. The funders had no role in the design of the study; in the collection, analyses, or interpretation of data; in the writing of the manuscript; or in the decision to publish the results.
51. Puech, C.; Baudry, J.; Joannon, A.; Poggi, S.; Aviron, S. Organic vs. conventional farming dichotomy: Does it make sense for natural enemies? *Agric. Ecosyst. Environ.* 2014, 194, 48–57. [CrossRef]

52. Alignier, A.; Raymond, L.; Deconchat, M.; Menozzi, P.; Montel, C.; Sarthou, J.P.; Vialatte, A.; Ouin, A. The effect of semi-natural habitats on aphids and their natural enemies across spatial and temporal scales. *Biol. Control* 2014, 77, 76–82. [CrossRef]

53. Tschamntke, T.; Rand, T.A.; Bianchi, F.J.J.A. The landscape context of trophic interactions: Insect spillover across the crop-noncrop interface. *Ann. Zool. Fennici* 2005, 42, 421–432.

54. Cooper, T.; Arblaster, K.; Baldock, D.; Farmer, M.; Beaufoy, G.; Jones, G.; Poux, X.; McCracken, D.; Bignal, E.; Elbersen, B.; et al. *Final Report on the Study of HNV Indicators for Evaluation*, Institute for European Environmental Policy: London, UK, 2007.

55. Horak, J.; Pelanovn, A.; Podavkova, A.; Safarova, L.; Bogusch, F.; Romportl, D.; Zusadel, P. Biodiversity responses to land use in traditional fruit orchards of a rural agricultural landscape. *Agric. Ecosyst. Environ.* 2013, 178, 71–77. [CrossRef]

56. Pulliam, H.R. Sources, Sinks, and Population Regulation. *Am. Nat.* 1988, 132, 652–661. [CrossRef]

57. Badenes-Perez, F.R.; Shelton, A.M.; Nault, B.A. Evaluating Trap Crops for Diamondback Moth, Plutella xylostella (Lepidoptera: Plutellidae). *J. Econ. Entomol.* 2004, 97, 1365–1372. [CrossRef]

58. Fanadzo, M.; Dalicuba, M.; Dube, E. Application of Conservation Agriculture Principles for the Management of Field Crops Pests. In *Sustainable Agriculture Reviews* 28; Gaba, S., Smith, B., Lichtfouse, E., Eds.; Springer International Publishing: Cham, Switzerland, 2018; pp. 125–152, ISBN 978-94-007-5448-5.

59. Kajtoch, L. The importance of traditional orchards for breeding birds: The preliminary study on Central European example. *Acta Oecologica* 2017, 78, 53–60. [CrossRef]

60. Barton, P.S.; Evans, M.J.; Foster, C.N.; Cunningham, S.A.; Manning, A.D. Environmental and spatial drivers of spider diversity at contrasting microhabitats. *Austral Ecol.* 2017, 42, 700–710. [CrossRef]

61. Theron, K.J.; Gaigher, R.; Pryke, J.S.; Samways, M.J. High quality remnant patches in a complex agricultural landscape sustain high spider diversity. *Biol. Conserv.* 2002, 104, 108480. [CrossRef]

62. Uetz, G.W.; Halaj, J.; Cady, A.B. Guild Structure of Spiders in Major Crops. *In Entomol. Exp. Appl.* 2020, 2, 10–19. [CrossRef]

63. Marko, V.; Keresztes, B.; Fountain, M.T.; Cross, J.V. Prey availability, pesticides and the abundance of orchard spider communities. *Agric. Ecosyst. Environ.* 2019, 285, 244–253. [CrossRef]

64. Rosas-Ramos, N.; Baños-Picó, L.; Tobajas, E.; de Paz, V.; Tormos, J.; Asis, J.D. Value of ecological infrastructure diversity in the maintenance of spider assemblages: A case study of Mediterranean vineyard agroecosystems. *Agric. Ecosyst. Environ.* 2018, 265, 115–124. [CrossRef]

65. Spears, L.R.; MacMahon, J.A. An experimental study of spiders in a shrub-steppe ecosystem: The effects of prey availability and shrub architecture. *J. Arachnol.* 2012, 40, 218–227. [CrossRef]

66. Álvarez, H.A.; Morente, M.; Oi, F.S.; Rodríguez, E.; Campos, M.; Ruano, F. Semi-natural habitat complexity affects abundance and movement of natural enemies in organic olive orchards. *Agric. Ecosyst. Environ.* 2019, 285, 106618. [CrossRef]

67. Rusch, A.; Valantin-Morison, M.; Sarthou, J.-P.; Roger-Estrade, J. Biological Control of Insect Pests in Agroecosystems. In *Biology and Ecology of Natural Enemies in Agricultural Systems*; Elsevier Ltd.: Amsterdam, The Netherlands, 2010; Volume 109, pp. 219–259.

68. Meyhofer, R.; Hindayana, D. Effects of intraguild predation on aphid parasitoid survival. *Entomol. Exp. Appl.* 2000, 97, 115–122. [CrossRef]

69. Finke, D.L.; Denno, R.F. Intraguild predation diminished in complex-structured vegetation: Implications for prey suppression. *Ecology* 2002, 83, 643–652. [CrossRef]

70. Stinner, B.R.; House, G.J. Arthropods and Other Invertebrates in Conservation-Tillage Agriculture. *Annu. Rev. Entomol.* 1990, 35, 299–318. [CrossRef]

71. Arnoldi, J.-F.; Bideault, A.; Loreau, M.; Haegeman, B. How ecosystems recover from pulse perturbations: A theory of short- to long-term responses. *J. Theor. Biol.* 2018, 436, 79–92. [CrossRef]

72. Tamburini, G.; De Simone, S.; Sigura, M.; Boscutti, F.; Marini, L. Conservation tillage mitigates the negative effect of landscape simplification on biological control. *J. Appl. Ecol.* 2016, 53, 233–241. [CrossRef]

73. Rowen, E.K.; Regan, K.H.; Barbercheck, M.E.; Tooker, J.F. Is tillage beneficial or detrimental for insect and slug management? A meta-analysis. *Agric. Ecosyst. Environ.* 2020, 294, 106849. [CrossRef]

74. Barton, P.S.; Evans, M.J.; Foster, C.N.; Cunningham, S.A.; Manning, A.D. Environmental and spatial drivers of spider diversity at contrasting microhabitats. *J. Theor. Biol.* 2018, 436, 79–92. [CrossRef]

75. Bennett, A.B.; Gratton, C. Local and landscape scale variables impact parasitoid assemblages across an urbanization gradient. *Landsc. Urban Plan.* 2012, 104, 26–33. [PubMed]

76. Aartsma, Y.; Bianchi, F.J.J.A.; Werf, W.; Poelman, E.H.; Dicke, M. Herbivore-induced plant volatiles and tritrophic interactions across spatial scales. *New Phytol.* 2017, 216, 1054–1063. [CrossRef]
80. Cardoso, P.; Pekár, S.; Jocqué, R.; Coddington, J.A. Global Patterns of Guild Composition and Functional Diversity of Spiders. *PLoS ONE* 2011, 6, e21710. [CrossRef]

81. Rosas-Ramos, N.; Baños-Picón, L.; Tormos, J.; Asís, J.D. Farming system shapes traits and composition of spider assemblages in Mediterranean cherry orchards. *PeerJ* 2020, 8, e8856. [CrossRef] [PubMed]

82. Cardenas, M.; Castro, J.; Campos, M. Short-term response of soil spiders to cover-crop removal in an organic olive orchard in a mediterranean setting. *J. Insect Sci.* 2012, 12, 1–18. [CrossRef]

83. Bellmann, H. *Nueva Guía de Campo de los Arácnidos de Europa*; Omega: Barcelona, Spain, 2011; ISBN 9788428215589.

84. Balog, A.; Markó, V. Community structure of rove beetles (Coleoptera: Staphylinidae) in apple orchards under different pest management system programs in Hungary. *Acta Phytopathol. Entomol. Hung.* 2007, 42, 377–385. [CrossRef]

85. Balog, A.; Markó, V. Species composition and community structure of the rove beetles (Coleoptera: Staphylinidae) in an experimental vineyard under different vineyard management systems. *Acta Phytopathol. Entomol. Hung.* 2007, 42, 367–376. [CrossRef]

86. Honék, A.; Kocian, M.; Martinková, Z. Rove beetles (Coleoptera: Staphylinidae) in an apple orchard. *Plant Prot. Sci.* 2012, 48, 116–122. [CrossRef]

87. Altieri, M.A.; Schmidt, L.L. Abundance patterns and foraging activity of ant communities in abandoned, organic and commercial apple orchards in Northern California. *Agric. Ecosyst. Environ.* 1984, 11, 341–352. [CrossRef]

88. Gómez, C.; Casellas, D.; Oliveras, J.; Bas, J.M. Structure of ground-foraging ant assemblages in relation to land-use change in the northwestern Mediterranean region. *Biodivers. Conserv.* 2003, 12, 2135–2146. [CrossRef]

89. Robinson, J.V. The Effect of Architectural Variation in Habitat on a Spider Community: An Experimental Field Study. *Ecology* 1981, 62, 73–80. [CrossRef]

90. McCain, C.; Szewczyk, T.; Bracy Knight, K. Population variability complicates the accurate detection of climate change responses. *Glob. Chang. Biol.* 2016, 22, 2081–2093. [CrossRef]

91. Rivers-Moore, N.A.; Samways, M.J. Game and cattle trampling, and impacts of human dwellings on arthropods at a game park boundary. *Biodivers. Conserv.* 1996, 5, 1545–1556. [CrossRef]