Ecosystem Services 38 (2019) 100948

Contents lists available at ScienceDirect

Ecosystem Services

journal homepage: www.elsevier.com/locate/ecoser

Agroecological management improves ecosystem services in almond orchards within one year

Vincent De Leijster[a,*], Maria João Santos[a,b], Martin J. Wassen[a], Maria Eugenia Ramos-Font[c], Ana Belén Robles[c], Mario Díaz[a], Maartje Staal[a], Pita A. Verweij[a]

[a] Copernicus Institute of Sustainable Development, Utrecht University, Princetonlaan 8a, 3584 CB Utrecht, the Netherlands
[b] University Research Priority Program in Global Change and Biodiversity and Department of Geography, University of Zürich, Winterthurerstrasse 190, 8057 Zürich, Switzerland
[c] Estación Experimental del Zaidín (CSIC), C/Profesor Albareda, 1, 18008 Granada, Spain

ARTICLE INFO

Keywords:
Ecosystem services
Agroecology
Woody crops
South-eastern Spain
Almonds
ES bundles

ABSTRACT

There is an increasing body of studies that show that land use intensification and homogenisation in agriculture landscapes, aimed at increasing food provisioning, decline other ecosystem services. Agroecological management has been proposed as an alternative to conventional agricultural management because of its presumed capacity to rehabilitate degraded ecosystem services. In this study we tested whether the agroecological principles of minimum mechanical soil disturbance, maintaining understory cover and application of organic amendments can improve the provisioning of ecosystem services and whether bundles of ecosystem services emerged. We experimentally implemented no-tillage (NT), green manure (GM), compost (CM) and conventional tillage (CT) as a control in five almond orchards in south-eastern Spain and monitored nutrient cycling, carbon stock, habitat provisioning, food provisioning, pest control and pollination after one year. We found that CM and NT had a higher overall ecosystem service performance than CT, and that GM did not differ from CT. The treatments significantly improved ecosystem services such as nutrient cycling, carbon stock, habitat provisioning and food provisioning, but not pest control and pollination. CM treatment resulted in higher soil enzyme activity (glucosidase and phosphatase), soil nutrient content (total N and extractable K), leaf nutrient content (P and K concentrations), soil organic carbon (SOC) content and almond kernel weight compared to other treatments. GM treatment resulted in higher phosphatase activity, understory carbon content and more understory cover than CT. NT treatment resulted in higher glucosidase, phosphatase and urease activity, understory plant diversity and more understory cover than CT. We also found an emerging bundle between SOC and soil enzyme activity and between individual almond weight and soil nutrient levels and SOC. This study shows that ecosystem services can rehabilitate rather quickly, given the one-year time frame of the study. Further, each agroecological practice may enhance a specific set of ecosystem services.

1. Introduction

Intensification and land use homogenisation in the agricultural sector have been identified as important drivers for land degradation (Foley et al., 2005; Mirzabaev et al., 2016; Tscharntke et al., 2005). Land degradation is an anthropogenic process causing terrestrial ecosystem services and biodiversity to decline (IPBES, 2018). Worldwide, land degradation is estimated to affect over 29% of the land surface and 25% of cropland (Le et al., 2016). The 2018 assessment of the Intergovernmental Panel on Biodiversity and Ecosystem Services (IPBES) reports that, although food provisioning in Europe is increasing, at least seven of the sixteen ecosystem services it examined are declining (IPBES, 2018). This suggests that food production and economic growth might be provided at the cost of other services (Raudsepp-Hearne et al., 2010; Scherr and McNeely, 2008). In the long term, land degradation is even expected to negatively affect crop productivity itself (Nkonya and Mirzabaev, 2016). This trend can already be observed in European woody fruit-crop systems, such as olive, citrus, vine and almond, which declined in productivity by 12% between 1982 and 2010 as a result of land degradation processes (Cherlet et al., 2013). Thus, rehabilitating degraded ecosystem services within agricultural landscapes and simultaneously meeting global food demands is a major challenge (Foley et al., 2005; Gaba et al., 2015).

Successful rehabilitation of individual ecosystem services has been

* Corresponding author.
E-mail address: v.deleijster@uu.nl (V. De Leijster).

https://doi.org/10.1016/j.ecoser.2019.100948
Received 14 January 2019; Received in revised form 10 May 2019; Accepted 24 May 2019

demonstrated for pollination (Saunders et al., 2013), pest control (Eilers and Klein, 2009), carbon storage (Lal, 2006) and soil fertility (Ramos et al., 2011). The simultaneous rehabilitation of multiple ecosystem services in productive agricultural landscapes is more challenging and is an emerging field of research (Bennett et al., 2009). When multiple ecosystem services are positively related to each other across space and time (Raudsepp-Hearne et al., 2010), bundles of ecosystem services emerge. For example, Gamfeldt et al. (2013) found a bundle including bilberry production, game hunting potential, understory plant species richness and dead wood across multiple forests in Sweden. On the contrary, when ecosystem services are negatively correlated, trade-offs may occur (Rodríguez et al., 2006). For example, trade-offs between carbon sequestration and biodiversity conservation have been described when comparing monoculture plantings for carbon credits and ecological restoration projects in Australia (Bryan, 2013; Crossman et al., 2011), and between biodiversity conservation and food production in Indonesian oil palm plantations (Teuscher et al., 2015). 'No-effect' relationships are more difficult to detect or less reported than trade-off and bundle relationships as there are less methods available that can effectively measure them (Lee and Lautenbach, 2016). Lee and Lautenbach (2016) did a meta-analysis on a wide range of land-use systems and showed no relationship between provisioning and cultural services, and suggested that this is likely because the services are not responding to a common driver.

The formation of bundles, trade-offs and no-effect relationships among ecosystem services due to restoration practices in agricultural landscapes has been addressed by only a few studies. For example, Schulte et al. (2017) showed that adding strips of natural vegetation in corn–soybean fields in Iowa (United States) can simultaneously improve pollination, habitat provisioning and mitigation of soil erosion and runoff without compromising revenue. However, Raudsepp-Hearne et al. (2010) found strong trade-offs between multiple provisioning and regulating services in an agricultural landscape in Canada. It is therefore still unclear whether there are generalised patterns of bundles and trade-offs, as the studies to date did not quantify the interactions between the multiple ecosystem services through time and space, neither have they identified potential drivers. In order to understand the complex relationships between ecosystem services, more research on the rehabilitation of ecosystem services and their mutual interactions is needed (Bennett et al., 2009).

Agroecological management has been proposed as an alternative to conventional agricultural management because of the presumed better balance in ecosystem services it provides and the rehabilitating effect it may have (Bommarco et al., 2013; Caron et al., 2014), making it an interesting system in which to investigate ecosystem service interactions. Kassam et al. (2012) and Altieri (2002) propose the following four principles of agroecological soil management: minimum mechanical soil disturbance, permanent understory vegetation, application of organic amendments, and diversification of the plant species.

In Europe, the Mediterranean region is highly susceptible to land degradation, due to the local biophysical and climatic conditions and the prevailing land management (Parras-Alcántara et al., 2016). Woody fruit-crop systems currently cover 22% of the southern Spanish autonomous community of Andalusia (Junta de Andalucía, 2015) and are mostly managed conventionally, i.e. by frequent tillage which removes understory vegetation (Meerkerk et al., 2008). Conventional agricultural management maximises the ecosystem service 'food provisioning', which may disrupt the balance with regulating, supporting and other provisioning ecosystem services (Kremen and Miles, 2012). For example, compared to agroecological management, conventional management in Mediterranean woody-crop systems resulted in 20–40% less understory vegetation cover (Cucci et al., 2016; Fracchiolla et al., 2015), 32–51% lower soil organic carbon content (Almagro et al., 2016; Ramos et al., 2011), 27–86% less efficient breakdown of organic phosphorus compounds in the soil (Hernández et al., 2005; Ramos et al., 2011), increased erosion and runoff, and reduced pest control and pollination services (Durán Zuazo et al., 2008; Eilers and Klein, 2009; Klein et al., 2012). In contrast, understory vegetation was associated with increased soil microbial activity (Ramos et al., 2011), less erosion and runoff (Durán Zuazo et al., 2006), and increased pollinator abundance (Saunders et al., 2013). Further, the combination of organic soil amendments and permanent understory cover resulted in 28–50% higher peach and kiwi yields and improved soil nutrient (N, P, K) levels (Montanaro et al., 2010).

In this paper we assess whether agroecological management affects the rehabilitation potential and interactions of regulating, supporting and provisioning ecosystem services in almond orchards in southern Spain. We tested the hypothesis that degraded ecosystem services in woody-crop systems can be rehabilitated by minimum mechanical soil disturbance, understory vegetation and application of organic amendments, while maintaining or enhancing food provisioning levels. Thus, we conducted a field experiment where we manipulated soil and understory management practices (no-tillage, green manure and compost) and compared it with conventional tillage in five degraded almond orchards. We analysed the effects of these practices on six ecosystem services: nutrient cycling (potential organic matter decomposition, soil nutrient availability, crop nutrient uptake), habitat provisioning (understory plant diversity, canopy arthropod diversity), carbon stock (soil and understory organic carbon content), pest control (abundance of pests and their natural enemies), pollination (abundance of pollinators and fruit set), and food provisioning (almond nut production). Finally, we assessed whether bundles of ecosystem services or trade-offs could be identified according to their response to the treatments. Our findings contribute to understanding the specific ecosystem services that can be targeted in rehabilitation and restoration projects of degraded Mediterranean woody-crop systems.

2. Methods

2.1. Study area

The study was conducted in the high plains of the provinces of Granada and Almeria in eastern Andalusia, SE Spain. This region was chosen because here almond is the most abundant woody crop, covering an area of 8.7%. Almonds are the fastest expanding woody crop in Andalusia, having increased by 18% between 2014 and 2017 (Consejería de Agricultura Pesca y Desarollo Rural, 2016; Ministerio de Agricultura Pesca y Alimentación, 2018). The almond farms in these high plains are typically located at elevations varying between approximately 700–1300 m. The study region is subject to biophysical land degradation processes, such as accelerated erosion and soil organic matter depletion (García-Ruiz, 2010), and has been classified in the highest category of desertification "very close to desertification" (Moreira Modueño and Rodríguez Surián, 2008). Mean annual rainfall varies between 300–400 mm per year (Cruz Pardo et al., 2010), with mean temperatures of 2–10 °C in winter and 20–28 °C in summer (Navarro López et al., 2012). The region experiences long periods of drought, on average 330 dry days per year (Cruz Pardo et al., 2010). However, rainfall intensity during the rest of the year can reach high values. Rainfall intensities of 1 mm min^{-1} are regularly recorded (de Castro et al., 2004), making the area vulnerable to erosion. A reduction in annual rainfall of 15 and 25% since 1961 has been documented for Almería and the Sierra Nevada (Ruiz Sinoga et al., 2011; de Castro et al., 2004).

2.2. Experimental design

We conducted an experiment with full factorial design with four treatments in five almond plantations experiencing land degradation. Selected almond plantations, hereafter referred to as farms, were applying conventional tillage management (> 2 times per year for understory removal), except for one farm (farm 3) that applied reduced

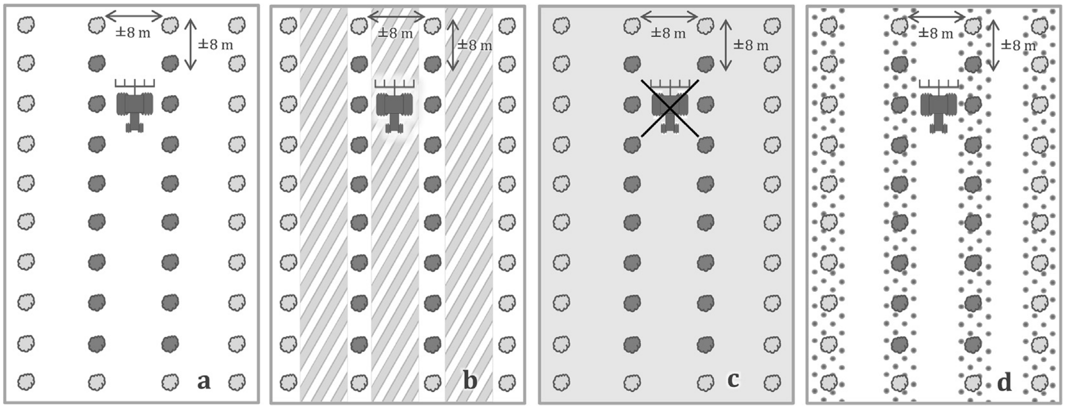

Fig. 1. Schematic diagram of the four treatments. a) conventional tillage (CT), b) green manure (GM), c) no-tillage (NT), and d) compost (CM). White indicates bare soil, stripes indicate green manure seed mixture, grey indicates permanent vegetation, and polka-dots indicate compost, tractor or tractor with cross indicate whether or not tillage is applied. Darker trees are sampled and lighter trees are not.

tillage management of 1–2 times per year. In all the farms tillage has been applied for decades, as -at least- minimum tillage is required to be eligible for subsidy. These farms were also selected, because the landowners consented to participate in the study. To apply the experimental treatments on each farm, we selected a site with similar altitude, slope, aspect, soil colour and almond tree size, age and variety, as farms were heterogenous in these characteristics.

The four experimental treatments were assigned randomly to the sites selected in each farm, and on farm 5 we repeated the treatments in a second parcel. Each treatment was therefore replicated six times, except for compost which was replicated five times, as it was only applied once on farm 5. A treatment within a farm is henceforth referred to as a 'plot'. Each plot corresponded to a rectangular area of at least four by eight trees, but to optimise the effect of the treatment and minimise the influence of adjacent management, only the inner two rows with almond trees were included in the research. The dimensions of the research plots were $14\,m \times 56\,m = 784\,m^2$ (7 m average distance between trees) and included at least 16 trees (Fig. 1).

The conventional tillage (CT) plots were tilled 2–3 times a year, using a chisel plough to remove the understory. The no-tillage (NT) plots were not harrowed, which allowed the wild plant species to grow. The green manure (GM) plots were sown manually in December (Table 1) with a legume–cereal mixture and then harrowed to incorporate the seeds in the soil. The mixture consisted of common vetch (*Vicia sativa*; 50 kg ha^{-1}), bitter vetch (*Vicia ervilia*; 50 kg ha^{-1}) and barley (*Hordeum vulgare*; 20 kg ha^{-1}). In early June the understory was harrowed into the soil. The plots with compost application (CM) were fertilised in December with compost (fermented sheep manure and straw; type bokashi) purchased locally (Table 1), applied manually near the almond trees at an approximate rate of 6 m^3 ha^{-1} and incorporated in the soil. This follows the advises from the local compost vendor and extension services, and is comparable to previous studies on woody crop systems in the region (Montanaro et al., 2010). In addition, the CM plots were harrowed 1–2 times to remove weeds. The same compost and seed mixture were used on all farms. During the study period, neither pesticides nor fertilisers were applied to the plots, except on Farm 5, where additional organic fertiliser (Fercrisa 5N-5P-5K, 300 kg ha^{-1}) was applied to all treatments, because of farmer's preference.

2.3. Field data collection

2.3.1. Soil properties

In each plot three soil samples of 1–2 kg were taken (Table 2), each consisting of ten sub-samples randomly taken from three different areas in the plot from the 0–20 cm soil layer because this was the plough layer (Almagro et al., 2016). The samples were sieved at field moisture through a 2 mm sieve. The sampling was done twice, in April 2017 and again in August 2017. The first samples were taken for enzymatic activity analysis, and were stored at 4 °C. The second set of samples were air dried and used for quantifying soil nutrients and organic matter. On farm 5 only soil samples were taken from the parcel with all four treatments.

Table 1
Characteristics of the almond farms used for the experiments and the details on treatment implementation per farm. CT refers to conventional tillage (> 2 times), RT to reduced tillage (1–2 times), GM to green manure, NT to no-tillage and CM to compost.

	Farm 1	Farm 2	Farm 3	Farm 4	Farm 5
Soil texture	Loam	Loam	Sandy clay loam	Sandy loam	Loam
Soil type	Calcic Regosols	Eutric Cambisols	Calcic Cambisols	Eutric Fluvisols	Calcic Cambisols
Elevation (m)	1250	900	1100	1100	1250
Slope (%)	7	24	4	2	8
Average rainfall in the past 5 years (mean ± st.dev.; mm)	262 ± 23	284 ± 24	296 ± 20	333 ± 23	327 ± 27
Location	Los Vélez, Almería	Alto Almanzora, Almería	Huéscar, Granada	Guadix, Granada	Los Vélez, Almería
Crop density (trees ha^{-1})	156	89	51	204	156
Crop age (year)	15	25	40	15	13
Almond variety	Guara	Marcona/Desmayo	Verdieri	Guara	Antoñeta/Guara
Historical management	CT	CT + occasional sheep grazing	RT + occasional sheep grazing	CT	CT + occasional sheep grazing + organic fertiliser
Implementation of treatments	All: Dec 2016	All: Jan 2017	GM & CM: Dec. 2016 NT: Dec 2015	GM & CM: Dec 2016 NT: Dec 2015	All: Dec 2016

Table 2
The selected ecosystem services for this study, corresponding indicators, units of measurement, sample size and month of data collection.

Ecosystem service indicator	Unit	No. of farms	samples/plot	Date of data collection
Nutrient cycling				
Enzymatic activity				
Dehydrogenase	µg INTF $g^{-1}h^{-1}$	5	3	April 2017
Glucosidase	µg PNG $g^{-1}h^{-1}$	5	3	April 2017
Phosphatase	mg PHP $g^{-1}h^{-1}$	5	3	April 2017
Urease	mg NH^{4+} $g^{-1}h^{-1}$	5	3	April 2017
Soil nutrient availability				
Total soil N content	%	5	3	Aug–Sept 2017
Soil extractable P content	ppm	5	3	Aug–Sept 2017
Soil extractable K content	ppm	5	3	Aug–Sept 2017
Leaf nutrient status				
Leaf N content	mg g^{-1}	6	3	Aug–Sept 2017
Leaf P content	ppm	6	3	Aug–Sept 2017
Leaf K content	ppm	6	3	Aug–Sept 2017
Carbon stock				
Soil organic carbon content	tC ha^{-1}	5	3	Aug–Sept 2017
Understory carbon content	tC ha^{-1}	6	6	April–May 2017
Habitat provisioning				
Arthropod abundance	# of individuals	6	4	April–May 2017
Arthropod species richness	# of orders	6	4	April–May 2017
Understory cover	%	6	6	April–May 2017
Understory plant species richness	# of species	6	6	April–May 2017
Food provisioning				
Almond yield	kg $tree^{-1}$	5	4	Aug–Sept 2017
Kernel weight	g $kernel^{-1}$	4	4	Aug–Sept 2017
Pest control				
Pest abundance	# of individuals	6	4	April–May 2017
Natural enemy abundance	# of individuals	6	4	April–May 2017
Enemy:Pest ratio	ratio	6	4	April–May 2017
Pollination				
Pollinator abundance	# of individuals	6	5	March 2017
Fruit set	Fruit:Flower ratio	6	4	March–May 2017

2.3.1.1. Enzymatic activity. Soil enzymatic activity, which relates to soil microbial activity and nutrient cycling processes (Sinsabaugh et al., 2008), has proven to be a powerful tool for assessing soil quality, as enzymes respond rapidly to changes in soil management (Burns et al., 2002). For this study we chose the enzymes β-glucosidase, phosphatase, urease and dehydrogenase, because they give a broad view of the potential decomposition processes in the soil, as they are indicators for the breakdown of cellulose (glucosidase), the P-cycle (phosphatase), N-cycle (urease) and C-cycle (dehydrogenase) (Das and Varma, 2011). Dehydrogenase activity was measured according to the methodology described by García et al. (1997). To assess phosphatase and β-glucosidase activity we used the methodology described by Ramos et al. (2011). Urease activity was determined using the method described in Kandeler and Gerber (1988).

2.3.1.2. Soil organic matter, carbon and chemical properties. Soil organic matter (OM) content was determined using the method described by Walkley and Black (1933). Soil organic carbon (SOC) content was calculated by multiplying OM content by 0.5, as proposed by Pribyl (2010). Total nitrogen content (N) was determined using the Kjeldahl method (Bremner, 1960). Extractable phosphorus content (P) was determined using the ascorbic acid Olsen method (Watanabe and Olsen, 1965). Extractable potassium (K) was determined with the ammonium acetate method (Pratt, 1965). The pH was determined in a 1:2.5 (w:v) aqueous solution.

2.3.2. Understory properties

We sampled the understory in the second and third week of May 2017 as in this period the herbaceous vegetation is fully developed (Ramos et al., 2010a,b). All plant species that grew underneath and in between almond trees were considered the understory.

2.3.2.1. Plant species richness and cover. The understory plant species composition was assessed using the point-intercept method, modified as proposed by Ruiz-Mirazo and Robles (2012) for natural grasslands in the same territory. We assessed six 10 m long transects that were laid out in each plot, going diagonally from one row of almond trees through the middle to the next row of almond trees. Each transect consisted of one hundred points at intervals of 10 cm. At each point of the transect, a pin 30 cm long and 2 mm of diameter was thrust into the ground. All plants that touched the pin were identified to the species level. When there was no plant touching the pin, we recorded bare soil or litter. From these data, we calculated species richness and percent understory cover.

2.3.2.2. Aboveground biomass and carbon content. Additionally, eight sub-plots of 25 × 25 cm were randomly distributed within each treatment plot. All the aboveground vegetation in these plots was cut manually and taken to the laboratory, where it was oven-dried at 60 °C and then weighed, to obtain dry biomass. The carbon content was calculated by multiplying dry biomass by the carbon fraction of 0.47 (IPCC, 2006).

2.3.3. Almond tree properties

2.3.3.1. Almond nut biomass. We measured almond production in farms 1, 3 and 4, and in one of the two parcels of farm 5 (the other parcel in farm 5 was -by mistake- harvested by the farmer before the measurements had been done). In farm 2 the production was measured by the farmer herself following our protocol. The production was measured during the harvest season in August–September 2017. Trees were harvested in groups of at least four, and this was repeated 3–4 times per plot. The harvest was hulled and bagged per group, and then weighed to obtain the in-shell fruit production per tree. Additionally, one sub-sample of 20 fruits was taken from each bag and brought to the lab, where they were air dried. There, each sub-sample was weighed and shelled them and then weighed the kernels to calculate in-shell:kernel ratios and kernel average weight. Finally, we obtained two variables; first, the production indicator of almond yield per tree, which was expressed as kernel weight per tree, and second, the almond fruit quality indicator of kernel weight in grams per individual kernel. No sub-samples of the fruits were obtained from farm 2 and neither from one of the parcels in farm 5.

2.3.3.2. Almond leaf nutrients. In August and September 2017 three almond leaf samples were taken from four trees in each plot. The samples were oven-dried at 60 °C for two days. Leaf nitrogen content was measured by the Dumas method (Simonne et al., 1994). Leaf phosphorus and potassium contents were determined by the digestion method with 1:4 of H_2O_2 30%:HNO_3 65%, and then analysed with an inductively coupled plasma – optical emission spectrometer (ICP-OES).

2.3.3.3. Fruit set ratio. In February and March 2017 four trees were marked, each with four branches with minimum length of 0.5 m. On each of these branches all flowers were counted. In April and May 2017 we counted the fruits on these same branches. Fruit set was expressed as the number of fruits divided by the number of flowers (Klein et al., 2012).

2.3.4. Arthropod community composition

2.3.4.1. Canopy arthropods. In May 2017 canopy arthropods were collected using the beating technique (Benhadi-Marin et al., 2011). One sample consisted of the arthropods collected from four branches of the same tree. Each branch was hit six times and all the arthropods that fell off were caught in a round beating tray with a diameter of 1 m. This was done for four trees in each plot and repeated within two weeks. Collection took place between 12 and 3 pm. The beating tray was emptied with a pooter and the arthropods were stored in 70% ethanol. Canopy arthropods in the sample were identified based on morphotype to the level of order and when possible to a more detailed taxonomical level, especially when more detailed taxonomic identification was necessary to attribute an ecological role and ecosystem service. Then, we classified the arthropod taxonomic groups into 'pests' and their 'natural enemies' based on existing literature on almond pests (Supplementary material; Table 7). In the pest group we included, from the order Hemiptera, the suborder Sternorrhyncha, which comprises almond aphids, and the family Tingidae (lace bugs, for example *Monosteira unicostata* -'tigre de almendro'-) (Almatni and Khalil, 2008; Benhadi-Marin et al., 2011; Santos et al., 2012). Further, we included the order Thysanoptera and the larvae of Lepidoptera, which have both been identified as pests in almonds (Minaei, 2014; Santos et al., 2012). In the natural enemy group we included the family Coccinellidae (Coleoptera) and the order Araneae (Benhadi-Marin et al., 2011). Syrphid flies, lacewings and parasitic wasps are also considered to be important natural enemies (Almatni and Khalil, 2008), but the beating method is not sensitive to these arthropods so these could not be considered (Müther and Vogt, 2003). The data is expressed as pest abundance (see Section 2.3.5. for 'pest absence'), natural enemy abundance, and natural enemy-pest ratio (NE:P), which was calculated by dividing the number of individuals categorised as natural enemies by the number of individuals categorised as pests (Denys and Tscharntke, 2002). Moreover, we calculated 'pest absence' from the pest abundance to maintain a more-is-better scale. This approach is explained in Section 2.3.5 'Ecosystem service index'.

2.3.4.2. Aerial arthropods. In February and March 2017 aerial arthropods were collected using the pan trap method, which has been demonstrated to be the most efficient method for sampling aerial arthropods in European agricultural areas (Westphal et al., 2008). We followed the method as proposed by Westphal et al. (2008), using five clusters of pan traps per plot, each with three coloured plates (blue, yellow and white). Pan traps were set up between 9 and 10 am in all plots, and removed between 4 and 5 pm. The method was not repeated. The arthropods caught were stored per cluster in 70% ethanol. After field sampling, individuals were identified based on morphotype to the level of sub-family, family or more detailed taxonomic level if possible. Then we classified the following three taxonomic groups as pollinators based on existing studies on pollinators of almonds: the superfamily Apoidea and the order Diptera (Klein et al., 2012; Ortiz-Sánchez and Tinaut, 1993; Saunders et al., 2013) (see Supplementary material; Table 7).

2.3.5. Statistical analyses

2.3.5.1. Ecosystem service index. To compare the different ecosystem services, we rescaled each ecosystem service indicator to range between 0.1–1 using the transformation proposed by Kearney et al. (2017):

$$Y_i = 0.1 + \left(\frac{x_i - min_i}{max_i - min_i}\right) \times 0.9$$

where *i* is the ecosystem service indicator index, Y is the response index value of *i*, and max and min represent the maximum and minimum of *i*. For pest control we applied the reverse transformation of pest abundance to 'pest absence' to maintain a more-is-better scale. Therefore the ecosystem service indicator index is subtracted from 1.1, as proposed by Kearney et al. (2017). The values of all ecosystem service indicator indices were then averaged per ecosystem service and per treatment to obtain the so-called 'single ecosystem service index'. Then, we averaged the data of all single ecosystem service indices to get an 'overall ecosystem service index' per treatment. The rescaled data was used in a principal component analysis and used to test the effect of treatment on ecosystem service index with Generalised Linear Mixed Models (GLMM).

2.3.5.2. Generalised Linear Mixed Models (GLMM). To assess the effect of treatment on ecosystem service indicators (Table 2) we fitted GLMMs using maximum likelihood and a frequentist approach for hypothesis testing (Faraway, 2016). The indicators for ecosystem services were considered the response variables and treatment the fixed factor, while farm location was included as a random factor. We did not include an interaction between treatment and farm because of the small sample size. Additionally, we applied a GLMM on the ecosystem service index to test whether treatments differed significantly from each other. Therefore, we made a model with treatment as the fixed factor and both farm and the ecosystem service indicator as categorical random factors. We first tested if the assumptions of a Gaussian distribution in the response variables were met, by analysing homoscedasticity and normal distribution of the residuals. If the model failed to meet one of these assumptions we considered alternative distributions (Supplementary material; Table 5). A negative binomial distribution (arthropod, pollinator and pest abundance), Poisson distribution (arthropod richness) and a zero inflated Poisson (natural enemies) were considered for count data and a gamma distribution for continuous data (soil K content, understory carbon, almond yield and fruit set). Non-Gaussian distributed models were tested for overdispersion using the method described by Zuur et al. (2013), which was not the case for any of the models. We calculated the models' marginal R^2 to explain the variance of the fixed factor and the conditional R^2 for the variance of the fixed and random factors together, by using the *MuMIn* package (Nakagawa et al., 2017). Likelihood ratio test (*function* drop1) was used to test for significant effects of the fixed factor. Tukey's pairwise comparisons (*function* glht) were applied to models that showed significant effects. Effects were considered significant at $\alpha < 0.05$. The GLMM analysis was conducted using the *lme4* package and the functions *glmer* and *lmer* in R (Version 3.4.3).

2.3.5.3. Principal component analysis. We used a PCA to determine whether ecosystem services formed bundles or trade-offs and assess whether treatments would cluster based on their effects on ecosystem services. Bundles were defined as multiple ecosystem services that appear in the same quadrant defined by the first three PCA axes. To select which PCA axes were relevant, we used the contribution of each axis to variance explained and then picked the minimum number of axes that explains > 70% of the variance. For display purposes, we choose 2D graphs with all PCA axes combinations (in our case 72% of the variance was explained by the first 3 PCA axes, and we plotted three 2D plots: PCA1-PCA2, PCA1-PCA3, and PCA2-PCA3). A trade-off occurred when one ecosystem service was on the positive side of a PCA axis and another ecosystem service on the negative side of that axis. When ecosystem services were related to different axes it was assumed that they did not interact because of the orthogonality imposed in the PCA. The abundance of natural enemies of pests was excluded from this analysis because the response variable had too little variance for rescaling. We applied the *prcomp* function in R (Version 3.4.3) on the rescaled ecosystem service indicators.

3. Results

We found that on average the overall ecosystem service index was highest in the compost treatment (CM), followed by green manure (GM) and no-tillage (NT) and the lowest for conventional tillage (CT) treatments (Fig. 2 and Supplementary material Table 6). Compared to CT, the single ecosystem service indices for nutrient cycling, carbon stock and habitat provisioning were 11–36% higher in NT, 8–76% higher in GM and 31–73% higher in CM; the single ecosystem services indices for food provisioning, pest control and pollination were −23 to +10% (NT), −12 to +4% (GM) and −11 to +14% (CM) compared to CT. The overall ecosystem service indices of NT and CM were significantly higher than that of CT, but there was no significant difference between CT and GM (Supplementary material; Table 6; Fig. 2).

3.1. Ecosystem services

3.1.1. Nutrient cycling

CT plots had the lowest enzymatic activity for all four enzymes. We found significantly higher glucosidase activity for NT and CM, compared to CT (Table 3). The phosphatase activity was significantly higher for NT, GM and CM compared to CT. The urease activity was higher for NT than for CT and GM. There was no significant effect of treatment for dehydrogenase activity. CM had significantly higher soil N and K content than all the other treatments; however, we found no significant effect of treatment for soil P content. CM also resulted in the highest leaf nutrient contents. For CM, leaf P and K content were significantly higher than for CT, NT and GM. We found no differences in leaf N content across treatments.

3.1.2. Carbon stock

CM plots contained significantly higher SOC content, than NT and GM plots, but there was no difference with CT. The carbon stock in understory was significantly higher for GM and NT than for CT. Soil carbon content (0–20 cm depth) was 12–44 times higher than the carbon content in the understory.

3.1.3. Habitat provisioning

We recorded 7026 individual plants and identified 126 plant species in 138 transects. The most abundant species were *Vicia ervilia* (bitter vetch), *Lolium rigidium* (ryegrass), *Vicia sativa* (common vetch) and *Hordeum vulgare* (barley), of which all except *L. rigidium* had been sown in GM. *L. rigidium, Anacyclus clavatus* (whitebuttons) and *Bromus rubens* (red brome) were the abundant wild species. Both NT and GM treatments had significantly more understory cover than CT and CM; NT also had higher plant species richness than CT and CM (Table 3). The arthropod abundance was higher for CM than for GM, but no significant effects of treatment were found for both arthropod abundance and richness. In total, we recorded 987 individual arthropods, the most abundant arthropod orders being Hemiptera, Thysanoptera and Coleoptera.

3.1.4. Food provisioning

The NT treatment had the lowest almond yield on average, however, the difference was not significant (Table 3). Almond yield varied greatly between the farms, which explains high standard deviations. CM resulted in significantly higher kernel weights than NT and GM, but did not significantly differ to CT.

3.1.5. Pest control and pollination

In the beating method we caught 1983 individuals in total of which 86% were classified as pests (94% belonged to the suborder Sternorrhyncha) and 3% were considered natural enemies. In the pan trap method a total of 2421 individuals were caught, of which 77% were considered pollinators (64% belonged to the order Diptera and 13% to the sub-family Apoidea). Treatment did not have a significant effect on abundance of pests, natural enemies and pollinators.

3.2. PCA

The first PCA axis was interpreted as inversely related to phosphatase activity, urease activity and soil N and SOC contents (Table 4). The second PCA axis was positively related to plant species richness, understory cover and understory carbon content, and negatively to soil K content and almond production. The third PCA axis was negatively related to glucosidase activity, leaf P content, almond productivity and pollinator abundance. The three axes together explain 72 % of the variance. Fig. 3 shows a gradient in soil characteristics along PCA 1, corresponding to SOC content, phosphatase, urease and soil N content being higher on the negative side of PCA 1 than on the positive side. The plots of the farm 1, 2 and 4 are located on the positive side of PCA 1, while the plots of farm 3 and 5 are located on the negative side of PCA 1.

4. Discussion

We tested the hypothesis that degraded ecosystem services in woody-crop systems can be rehabilitated by agroecological management, while maintaining or enhancing food provisioning levels. We found some support for agroecological practices improving ecosystem service provisioning in the short term in woody-crop systems in southern Spain. The overall ecosystem service indices were higher in the agroecological treatments than in conventional management; +17% for green manure, +17% for no-tillage and +28% for compost. For individual ecosystem services, we observed that agroecological management increased nutrient cycling, carbon stock and habitat provisioning, but not food provisioning, pest control and pollination. These results are further evidence that agroecological land management practices can play a role in the rehabilitation of some ecosystem services (Tittonell, 2014; Bommarco et al., 2013; Caron et al., 2014; Bennett et al., 2009). The results also show that the rehabilitation process may occur rapidly, as we found significant improvements in less than one year. Previous studies have reported improvements over longer periods. For example, Ponisio et al. (2016) reported that ten years after hedgerow restoration in agroecological landscapes, habitat provisioning for pollinators had improved by 12% and Schulte et al. (2017) showed that after five years, natural vegetation strips had improved on-farm abundance of arthropods, pollinators and birds, and had reduced runoff by 37%, without reducing crop productivity in corn and soy fields. In our study, however, the responses of ecosystem services to the treatments differed strongly, as is summarised below.

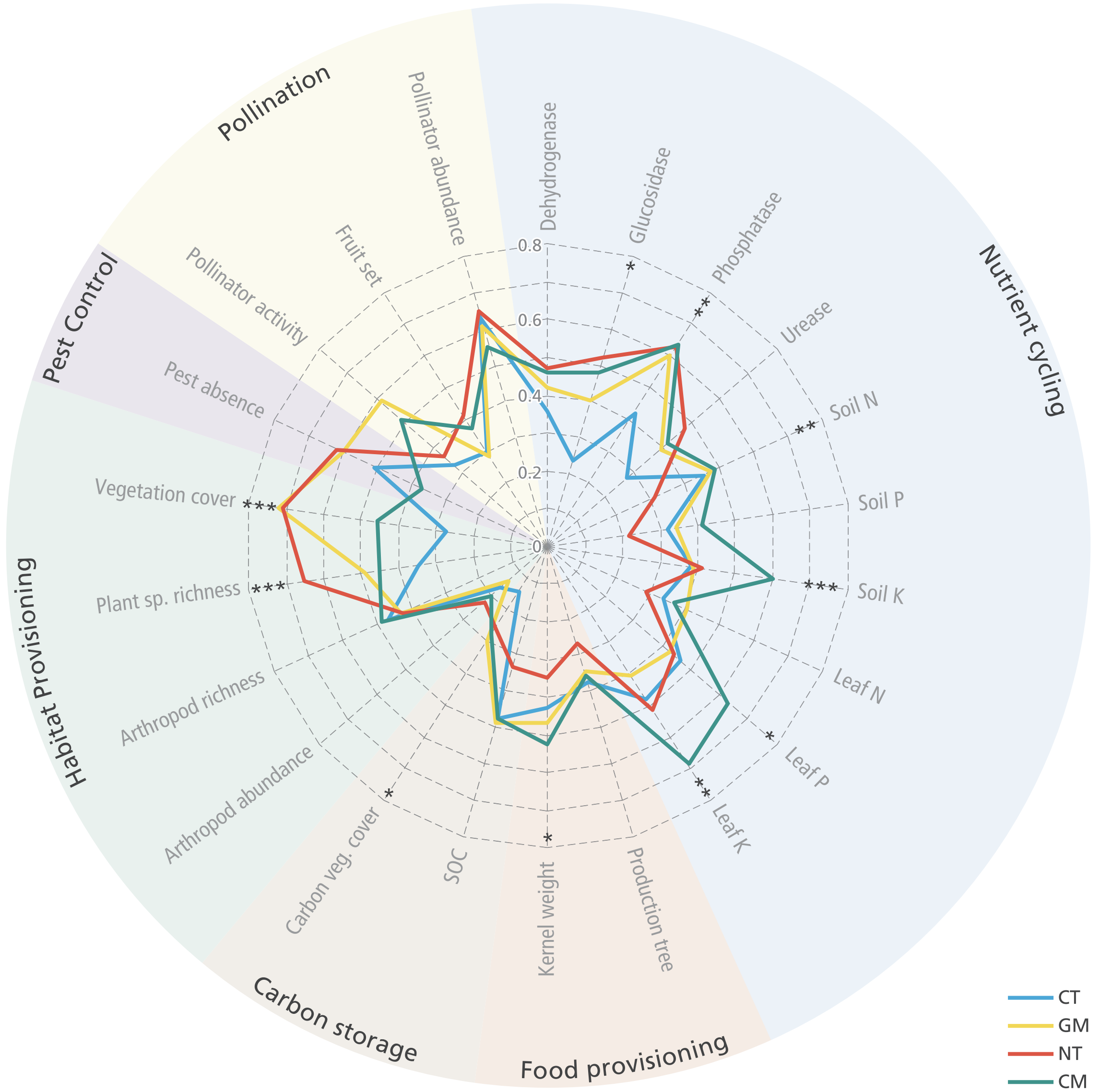

Fig. 2. Radar chart of individual ecosystem service indices values per treatment units. CT = conventional tillage, CM = compost, GM = green manure, NT = no-tillage (NT). Asterisks in figure show significance of differences between treatments, * < 0.05, ** < 0.001, *** < 0.0001.

4.1. Nutrient cycling

Nutrient cycling was the ecosystem service that improved the most with the agroecological treatments (11–36% higher) over the period of one year. Compost addition improved the nutrient cycling in soil organic decomposition processes (enzymatic activities of glucosidase and phosphatase), soil nutrient availability (total N and extractable K contents in soil) and crop nutrient contents (P and K contents in crop leaves). By contrast, conventional tillage had the lowest enzymatic activity and no-tillage the lowest nutrient content in the soil and crop. Our results are in line with those of another study in a non-agricultural system in Andalusia (Spain), where both plant-based and manure soil amendments effectively enhanced soil enzymatic activities (i.e. dehydrogenase, urease and β-glucosidase) within one year (Tejada et al., 2006). The activity of these three enzymes and of phosphatase is also known to increase as a result of compost application in other crops (Chang et al., 2007).

We found higher enzymatic activity in NT than CT, but lower soil total N and extractable P contents and leaf N and P contents. Another study in a Spanish almond system also found that by comparison with CT, NT resulted in higher enzymatic activity for dehydrogenase (+37%), β-glucosidase (+93%) and phosphatase (+87%). However, in contrast with our latter findings, lower soil and leaf nutrient levels in NT than in CT, the studies by Ramos et al. (2011) and Cucci et al. (2016) report higher soil N content for NT than for CT. The relationship between NT and soil N content seems to be variable, as Martínez-Mena et al. (2013) did not find differences between NT and CT in almond orchards, and Gómez et al. (2009) reported a reduction in soil N content with NT in olive orchards. Martínez-Mena et al. (2013) propose that NT increases soil compaction, which reduces both the N-mineralisation rate in the soil and crop root development, which in turn reduces nutrient uptake by the plant. This might explain the lower N content in the soil and almond leaves in NT in our study, but as we did not measure physical properties of the soil, we could not investigate this.

CM is known to have the capacity to improve the soil's physical and biological properties such as soil structure, water-holding capacity and macrofauna (Norris and Congreves, 2018; Ouédraogo et al., 2001), which, in theory, could offset the disadvantages of NT. A study in Mediterranean apricot orchards showed that applying CM and NT together can effectively restore soil nutrient contents but not SOC levels (Montanaro et al., 2010). More research on the combination of the considered agroecological treatments is needed to assess its effect on

Table 3
Summary of the GLMMs on the effect of treatments on ecosystem service indicators. Table includes the mean and standard deviation (SD) per treatment, P-value and, if significant, P-value in bold and Tukey post-hoc test grouping letters. CT = conventional tillage, NT = no-tillage, GM = green manure and CM = compost.

Ecosystem service indicator	Unit	GLMM *p*-value	CT	NT	GM	CM
Nutrient cycling						
Dehydrogenase	µg INTF $g^{-1}h^{-1}$	0.6	2.1 ± 1.4	2.7 ± 1.9	2.4 ± 1.9	2.7 ± 1.5
Glucosidase	µg PNG $g^{-1}h^{-1}$	**< 0.001**	140 ± 78^{b}	365 ± 215^{a}	272 ± 97^{ab}	333 ± 136^{a}
Phosphatase	mg PHP $g^{-1}h^{-1}$	**< 0.001**	102 ± 72.3^{b}	156 ± 58.6^{a}	150 ± 59.1^{a}	159 ± 66.9^{a}
Urease	mg NH^{4+} $g^{-1}h^{-1}$	**0.007**	33 ± 32^{b}	65 ± 50^{a}	51 ± 30^{b}	55 ± 35^{ab}
Total soil N content	%	**< 0.001**	0.09 ± 0.04^{b}	0.08 ± 0.03^{b}	0.09 ± 0.04^{b}	0.11 ± 0.04^{a}
Soil extractable P content	ppm	**0.006**	13 ± 7.9	8.3 ± 6.5	15 ± 8.9	21 ± 15
Soil extractable K content	ppm	**< 0.001**	140 ± 94^{b}	150 ± 120^{b}	140 ± 98^{b}	230 ± 140^{a}
Leaf N	mg g^{-1}	0.07	1.6 ± 0.26	1.58 ± 0.18	1.73 ± 022	1.75 ± 0.22
Leaf P	ppm	**< 0.001**	1.1×10^{3} ± 1.6×10^{3b}	1.1×10^{3} ± 1.4×10^{2b}	1.1×10^{3} ± 1.7×10^{3b}	1.2×10^{3} ± 2.0×10^{3a}
Leaf K	ppm	**< 0.001**	5.5×10^{3} ± 2.1×10^{3b}	5.9×10^{3} ± 1.3×10^{3b}	4.7×10^{3} ± 2.3×10^{3b}	7.7×10^{3} ± 2.0×10^{3a}
Carbon stock						
SOC	tC ha^{-1}	**< 0.001**	8.4 ± 5.1^{ab}	6.4 ± 3.7^{b}	8.2 ± 9.4^{b}	9.4 ± 4.0^{a}
Understory carbon stock	tC ha^{-1}	**< 0.001**	0.19 ± 0.34^{c}	0.44 ± 0.47^{ab}	0.54 ± 0.35^{a}	0.41 ± 0.58^{bc}
Habitat provisioning						
Arthropod abundance	# individuals $trap^{-1}$	**0.042**	18 ± 18^{ab}	14 ± 18^{ab}	12 ± 12^{b}	25 ± 28^{a}
Arthropod richness	# orders $trap^{-1}$	0.87	3.5 ± 2.2	3.2 ± 1.4	3.2 ± 1.5	3.6 ± 1.8
Understory cover	%	**< 0.001**	16.8 ± 23.2^{c}	56.2 ± 16.7^{a}	62.6 ± 12.0^{a}	36.5 ± 33.5^{b}
Understory plant richness	# species $transect^{-1}$	**< 0.001**	4.8 ± 5.3^{c}	11.6 ± 3.0^{a}	8.1 ± 3.4^{b}	7.4 ± 6.0^{b}
Food provisioning						
Almond yield	kg $tree^{-1}$	0.39	0.54 ± 0.47	0.37 ± 0.24	0.59 ± 0.59	0.48 ± 0.24
Kernel weight	g $kernel^{-1}$	**< 0.001**	1.08 ± 0.12^{ab}	1.02 ± 0.15^{b}	1.06 ± 0.35^{b}	1.24 ± 0.05^{a}
Pest control						
Pest abundance	# individuals $trap^{-1}$	0.39	12 ± 10	8.2 ± 5.8	8.5 ± 9.3	16 ± 13
Natural enemies abundance	# individuals $trap^{-1}$	0.77	0.87 ± 0.43	0.72 ± 0.39	0.58 ± 0.66	0.87 ± 0.61
NE:P ratio	Ratio	0.57	0.26 ± 0.38	0.16 ± 0.22	0.26 ± 0.38	0.08 ± 0.08
Pollination						
Pollinator abundance	# of individuals	0.13	6.0 ± 5.7	7.4 ± 7.3	5.6 ± 4.4	8.6 ± 8.7
Fruit set	Fruit:Flower ratio	0.062	0.12 ± 0.12	0.20 ± 0.21	0.13 ± 0.16	0.14 ± 0.20

Table 4
PCA scores for the indicators of ecosystem services.

Ecosystem services	Indicator	PC1	PC2	PC3
Nutrient cycling	Dehydrogenase activity	−0.17	0.18	−0.15
Nutrient cycling	Glucosidase activity	−0.24	0.11	−0.54
Nutrient cycling	Phosphatase activity	−0.43	0.00	0.10
Nutrient cycling	Urease activity	−0.42	−0.12	0.19
Nutrient cycling	Total soil N content	−0.38	−0.23	−0.08
Nutrient cycling	Soil extractable P content	−0.15	−0.16	0.17
Nutrient cycling	Soil extractable K content	−0.21	−0.28	0.10
Nutrient cycling	Leaf N content	−0.16	0.09	−0.05
Nutrient cycling	Leaf P content	0.02	−0.09	−0.38
Nutrient cycling	Leaf K content	0.10	−0.13	−0.01
Carbon stock	SOC	−0.38	−0.20	−0.08
Carbon stock	Understory carbon stock	−0.04	0.27	−0.07
Habitat provisioning	Arthropod abundance	0.09	0.04	0.16
Habitat provisioning	Arthropod order richness	0.00	0.01	0.15
Habitat provisioning	Plant species richness	−0.22	0.54	−0.04
Habitat provisioning	Understory cover	−0.17	0.43	−0.13
Food provisioning	Almond yield	0.12	−0.27	−0.49
Food provisioning	Kernel weight	−0.15	−0.21	−0.03
Pest control	Pest absence	−0.15	−0.01	0.30
Pollination	Fruit set	0.09	−0.21	−0.18
Pollination	Pollinator abundance	0.14	−0.01	0.29
	SD	0.55	0.43	0.32
	Proportion of variance explained	0.37	0.22	0.13

the nutrient cycling capacity.

4.2. Carbon stock

We found that the carbon stock in the agroecological treatments was 8–76% higher than in CT. Unsurprisingly, GM resulted in the highest carbon stock levels in the understory compared to CT. This is because GM plots received additional biomass from the seeded common vetch, bitter vetch and barley, whereas in CT the understory was removed. Our results are in line with the results of a 35-year-long study on almond orchards in Italy, which reported that green manure application increased carbon stock in the understory by 9% as compared to conventional tillage (Cucci et al., 2016), but we found a much larger increase. However, there seems to be a relation with water availability as reported by Ramos-Font et al. (2009), who found that in years with more dry periods the understory in NT stores more carbon than in GM, while this is the reverse in years with less water limitation. Cucci et al. (2016) also found 34% higher SOC for almonds with GM than for CT, which we did not observe, on the contrary we found lower SOC contents for GM and NT compared to CM and no differences with CT. Vicente-Vicente et al. (2016) showed, in a meta-analysis on soil carbon sequestration rates under alternative management practices, that adding organic amendments is the most effective strategy to increase SOC. No differences in soil carbon stock were found between CT and the agroecological treatment, which combined no-tillage with compost application, in a Spanish apricot study (Montanaro et al., 2010). The authors claim that Mediterranean soils with low organic matter content only show measurable changes in SOC levels after 7–10 years (Montanaro et al., 2012). As the build-up of SOC is slow, it is more likely that the higher levels of SOC in this study are the result of measuring the compost particles themselves, and these may still mineralise over time. Therefore, long-term monitoring of SOC after compost application is needed to demonstrate whether carbon stocks are also improved in the long term. Further, Luo et al. (2010) report that in a transition from CT towards NT management SOC increases in the top layer (0–10 cm), but decreases in deeper layers (10–40 cm). In the current study soil samples were taken from a depth between 0–20 cm, which includes both previous layers. Our results show that actual carbon storage is 12 to 44 times higher in this part of the soil than in understory vegetation, which implies that when targeting carbon stock

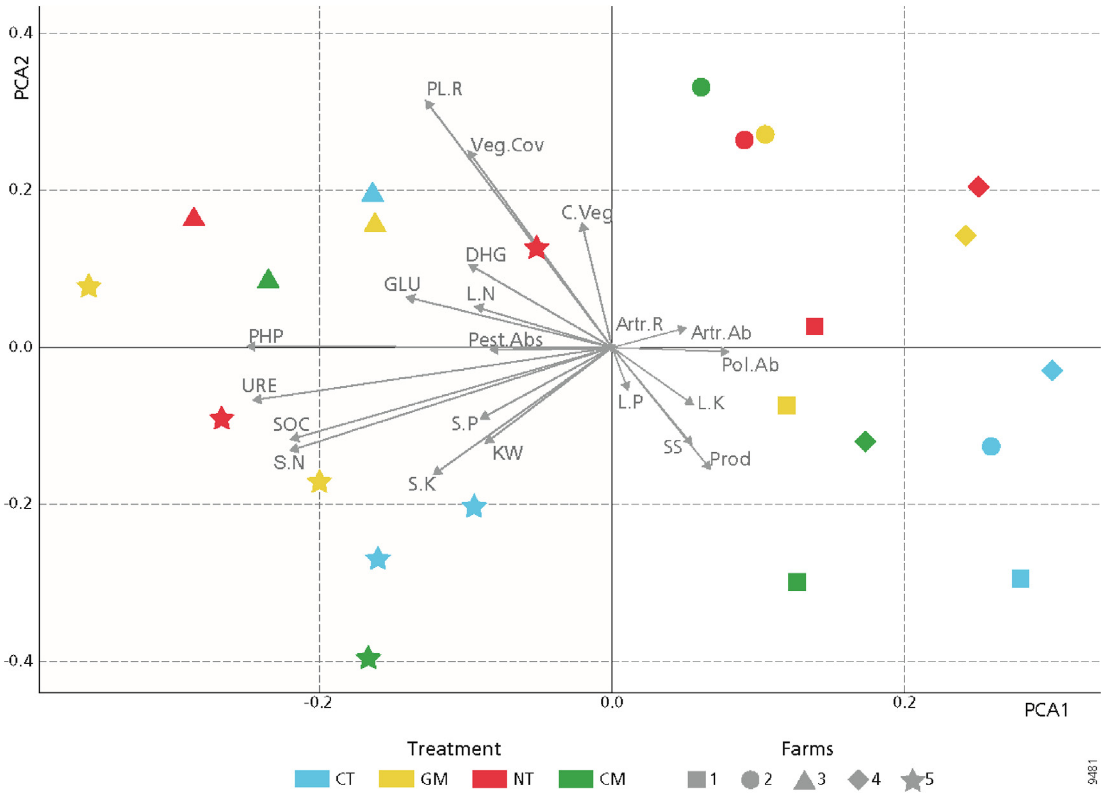

Fig. 3. Biplot visualising the scores of PCA1 (proportion of variance explained 0.37) and PCA2 (proportion of variance explained 0.22). The × axis corresponds to the scores of PCA1 scores, the y axis corresponds to the PCA2 scores. The colours correspond to the treatments, with: CM = compost, CT = conventional tillage, GM = green manure and NT = no-tillage. The symbols correspond to the farms, with: DHG = dehydrogenase activity, GLU = glucosidase activity, PHP = phosphatase activity, URE = urease activity, S.N = total soil N content, S.P = soil extractable P content, S.K = soil extractable K content, L.N = leaf N content, L.P = leaf P content, L.K = leaf K content, Prod = almond production, KW = kernel weight, SOC = soil organic carbon, Veg.C = understory carbon stock, Artr.Ab = arthropod abundance, Artr. R = arthropod order richness, PL.R = plant species richness, Veg.Cov = understory cover, Pest.Abs = pest absence, Pol.Ab = pollinator abundance, SS = fruit set ratio.

rehabilitation in woody-crop systems it is important to focus on the soil. This makes CM more effective in increasing carbon stock than the other agroecological management practices.

4.3. Habitat provisioning, pest control and pollination

We found 31–73% habitat provisioning improvements in the agroecological treatments by comparison with CM. This is strongly related to fraction of understory cover and plant species richness and not to arthropods abundance and richness. It is self-evident that the plant species richness and fractional cover are highest in the vegetation-related treatments (NT and GM) and is in line with previous research (Cucci et al., 2016). Understory vegetation in almond orchards has been shown to be positively related to pollinator abundance (Norfolk et al., 2016; Saunders et al., 2013), and parasitoid wasp abundance (Eilers and Klein (2009); however, we failed to find this relationship. One possible reason is that our study had a relatively small sample size and few sampling events over a short period of time (Westphal et al., 2008). A second possible reason is that pollination measurements were done in March, during the flowering season of the almonds, when the understory vegetation was not yet fully developed. Thirdly, the resolution of our taxonomic identification, which was only at the order level, might not be enough. Fourthly, for the analysis of natural enemies we may have missed important pest predators syrphid flies, parasitoid flies and wasps and lacewings as our method was not sensitive to these groups. Moreover, for the analysis of pollinators we missed butterflies (Lepidoptera) and pollinating wasps (Vespoidea), as the pan trap method was not sensitive to these groups. These groups, however, have a lower pollinator efficiency than bees (Apoidea) and flies (Diptera). A study in the southern Spain showed that butterflies, wasps and ants combined represent 17% of the flower visits, while bees and flies combined represent 76.6% (Ortiz-Sánchez and Tinaut, 1993). Finally, the population dynamics of farm arthropods largely depend on processes that occur at a scale larger than an agricultural plot (Silva et al., 2010), and our relatively small plots may have been ineffective in measuring these larger-scale dynamics.

4.4. Food provisioning

During our study, the region experienced an exceptionally cold spring, which damaged almond blossom (personal observation) and affected yield. The trees on the farms in our study also suffered frost damage, causing low and variable production. The average minimum daily temperature in March from the weather stations nearby the farms was 27% lower in 2017 compared to the average from 2000 to 2016 (AEMET; supplementary material Fig. 4). Moreover, another study in southern Spain reported almond yields of 1–3 kg in kernel weight per tree, which is remarkably higher than what we found (Almagro et al., 2016; De Giorgio and Lamascese, 2005). The low spring temperatures may explain our results that almond yield and kernel weight did not differ significantly between CT and the other treatments. Previous studies on almond orchards reported a lower crop yield in NT than in CT (De Giorgio and Lamascese, 2005; Martínez-Mena et al., 2013). A 20-year-long study on tillage versus no-tillage management in corn fields in the United States found that no-tillage had an almost immediate positive effect on soil properties but its effect on crop yield lagged behind, with crop yields not outweighing those in tillage management until after 13 years (Ismail et al., 1994). To our knowledge, there are no comparable long-term studies in Mediterranean woody-crop systems and therefore it is unwise to extrapolate to our case. Moreover, we did not find any effect of GM or CM on almond yield. For GM our results agree with those found for almonds in Italy, where no significant difference in production was found between GM and CM (De Giorgio and Lamascese, 2005), but not with those in Spain, where production in GM was reported as 41% higher than in CT (Almagro et al., 2016). Nonetheless, we did find a significantly higher kernel nut weight for CM, which relates to a higher price for almonds. Although the combined effect of understory cover and soil amendment has not yet been investigated for almonds, it has been investigated for apricots, where combining understory cover and compost can increase crop yield by 28% within four years (Montanaro et al., 2010).

4.5. Ecosystem service bundles and trade-offs

The results from the PCA show a negative correlation between understory cover and crop yield for the first two axes (Fig. 4). Nevertheless, due to the climatic conditions during our study, crop yields were lower than average and therefore no conclusions on trade-offs can be drawn. De Giorgio and Lamascese (2005) and Martínez-Mena et al. (2013), did suggest a negative relationship between understory cover and crop yield, as they found lower almond yield in non-tilled plots with natural vegetation, compared to conventionally tilled plots in longer term experiments. There were no other negative correlations among individual ecosystem service indicators. Furthermore, in the PCA analysis we observed three pairs of positively correlated indicators that can be interpreted as early signals for emerging ecosystem service bundles. First, we observed that SOC and soil enzymatic activities are correlated (Fig. 4), which agrees with the findings of Ramos et al. (2011). Second, we found that crop yield related to the fruit set ratio on the first two axes, and that individual almond fruit weight related to soil enzymatic activities on the PCA1 axis and on both PCA1 and 2 to soil nutrient levels and SOC (Table 4). The importance of pollination for almond trees has been shown in Californian almond systems (Klein et al., 2012); another Californian almond study has demonstrated how nutrient availability is essential for fruit development (Muhammad et al., 2015). Nevertheless, the bundles containing yield data are uncertain, due to lower than average yield values.

One possible limitation to our ability to detect bundles and trade-offs at the scale of multiple ecosystem services during ecological rehabilitation in woody crop systems could be the length of the study. The rate of ecosystem services interaction, trade-off or bundling, is known to vary between systems and over space and time (Costanza et al., 2017). This suggests that even though certain trade-offs and bundles are not observable at a certain moment, they may yet develop over time. For example, Ponisio et al (2016) found that the ecosystem services pollination and habitat provisioning were higher after more than 10 years of planting hedgerows around intensive agricultural land than in the first 10 years. Applying this hypothesis on dryland tree-crop systems would suggest that trade-offs and/or bundles as a result of agroecological management implantation may arise over time. Therefore, research on larger spatial and temporal scales is required involving combinations of agroecological practises, such as organic amendments combined with understory vegetation. Each agroecological practice may enhance a specific set of ecosystem services, so by combining them ecosystem service bundles can be expected.

4.6. Limitations, uncertainties and knowledge gaps

We were able to detect rehabilitation of specific ecosystem service provisioning within a short time frame, but the limited time frame might also have prevented the detection of changes in ecosystem services that respond less rapidly (e.g. below ground carbon stock or arthropod mediated ecosystem services) or that do not respond at all. Therefore, long-term research is needed, to improve our understanding of trade-offs and bundle formation during the ecosystem service rehabilitation process. We found a larger variance in ecosystem service indicators across farms than between treatments, which suggests that the amplitude of ecosystem service response is also strongly influenced by farm characteristics such as location in the landscape, climate, soil type, and current and historical management (supplementary material Table 5). While the number of farms in our study was higher than those in previous studies (Almagro et al., 2016; Cucci et al., 2016; Ramos et al., 2010a,b), five farms might still be too few to reveal how farm characteristics influence the ecosystem service rehabilitation process. Research conducted on a larger spatial scale and involving a larger number of farms would be needed to provide better insight into the influence of site-specific factors on the rehabilitation process.

In this study we investigated potential trade-offs between food provisioning, the main objective of an almond farm, and other ecosystem services using almond yield and kernel weight as indicators. Nevertheless, net financial returns are more important than yield per se. In a financial analysis on Mediterranean olive orchards, it was shown that, although organic orchards have lower yields, they are more profitable than conventional orchards, largely because of higher subsidies and higher market price (Sgroi et al., 2015). Analogue to this, Jezeer et al. (2017) have shown that while shaded agroforestry management improves biodiversity and has higher cost-benefit ratios, the yields of coffee and cocoa plants are lower in shaded plantations. In this case agroforestry systems could reach high net economic benefits, because of lower management costs, additional sources of income and higher prices for the cash crop (Jezeer et al., 2018). These two examples suggest that to optimise farm performance, both for ecosystem services and financial performance, we need to better understand how agroecological practices affect economic metrics other than solely crop yields.

5. Conclusion

We experimentally implemented agroecological practices to investigate ecosystem service rehabilitation rates and interactions between ecosystem services in Mediterranean tree-crop systems. We found that green manure, no-tillage and compost resulted in higher provisioning of ecosystem services than conventional management. The results show that compost can be implemented in woody-crop systems without negatively affecting food provisioning on the short term. We also showed that the rehabilitation capacity of the individual agroecological practices of no-tillage, green manure and compost is ecosystem service specific. Therefore, based on our results we recommend compost to be applied to improve nutrient cycling and carbon stock, understory vegetation should be used to improve habitat provisioning, and green manure to improve habitat provisioning and carbon sequestration. However, no-tillage is expected to negatively affect almond yield on the short term. Applying multiple agroecological practices simultaneously may have the potential to optimise multiple ecosystem services, but this should be further investigated as we did not find strong ecosystem service bundles yet. The time frame of this study was relatively short (one year), suggesting that some ecosystem services can be rehabilitated rapidly. It is, however, possible that ecosystem services that were not affected by the treatments in this short study could still respond to agroecological practices after a longer period.

Acknowledgements

We would like to thank the farmers who participated in this study for their assistance and for allowing us to work on their farms. We are grateful to AlVelAl, which helped with the organisation of the fieldwork and the communication with the farmers. This study was supported by funding from the graduate program 'Nature Conservation, Management and Restoration' of The Netherlands Organisation for Scientific Research (NWO) and the Commonland Foundation, which financially supported field data collection. The English of a near-final version of the paper was edited and corrected professionally by Joy Burrough. The art-work of the paper was edited by Ton Markus. Finally, we thank the reviewers for their valuable feedback and contributions.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecoser.2019.100948.

References

Almagro, M., De Vente, J., Boix-Fayos, C., García-Franco, N., Melgares de Aguilar, J., González, D., Solé-Benet, A., Martínez-Mena, M., 2016. Sustainable land management practices as providers of several ecosystem services under rainfed Mediterranean agroecosystems. Mitig. Adapt. Strateg. Glob. Chang. 21, 1029–1043. https://doi.org/10.1007/s11027-013-9535-2.

Almatni, W., Khalil, N., 2008. A primary survey of aphid species on almond and peach, and natural enemies of Brachycaudus amygdalinus in As-Sweida, Southern Syria 109–115.

Altieri, M.A., 2002. Agroecology: the science of natural resource management for poor farmers in marginal environments. Agric. Ecosyst. Environ. 1971, 1–24.

Benhadi-Marin, J., Pereira, J.A., Barrientos, Bento, A., Santos, S.A.P., 2011. Diversity of predaceous arthropods in the almond tree canopy in Northeastern Portugal: a methodological approach. Entomol. Sci. 14, 347–358. https://doi.org/10.1111/j.1479-8298.2011.00444.x.

Bennett, E.M., Peterson, G.D., Gordon, L.J., 2009. Understanding relationships among multiple ecosystem services. Ecol. Lett. 12, 1394–1404. https://doi.org/10.1111/j.1461-0248.2009.01387.x.

Bommarco, R., Kleijn, D., Potts, S.G., 2013. Ecological intensification: harnessing ecosystem services for food security. Trends Ecol. Evol. 28, 230–238. https://doi.org/10.1016/j.tree.2012.10.012.

Bremner, J., 1960. Determination of nitrogen in soil by the Kjeldahl method. J. Agric. Sci. 55, 11–33. https://doi.org/10.1017/S0021859600021572.

Bryan, B.A., 2013. Incentives, land use, and ecosystem services: synthesizing complex linkages. Environ. Sci. Policy 27, 124–134. https://doi.org/10.1016/j.envsci.2012.12.010.

Burns, R.G., Nannipieri, P., Kandeler, E., Ruggiero, P., 2002. Enzymes in the Environment: Activity, Ecology, and Applications. Marcek Dekker.

Caron, P., Biénabe, E., Hainzelin, E., 2014. Making transition towards ecological intensification of agriculture a reality: the gaps in and the role of scientific knowledge. Curr. Opin. Environ. Sustain. 8, 44–52. https://doi.org/10.1016/j.cosust.2014.08.004.

Chang, E.-H., Chung, R.-S., Tsai, Y.-H., 2007. Effect of different application rates of organic fertilizer on soil enzyme activity and microbial population. Soil Sci. Plant Nutr. 53, 132–140. https://doi.org/10.1111/j.1747-0765.2007.00122.x.

Cherlet, M., Ivits, E., Sommer, S., Tóth, G., Jones, A., Montanarella, L., Belward, A., 2013. Land Productivity Dynamics in Europe Towards a Valuation of Land Degradation in the EU. JRC, Ispara, Italy https://doi.org/10.2788/70673.

Consejería de Agricultura Pesca y Desarollo Rural, 2016. Caracterización del sector de la almendra en Andalucía. Junta de Andalucía, Sevilla, Spain.

Costanza, R., de Groot, R., Braat, L., Kubiszewski, I., Fioramonti, L., Sutton, P., Farber, S., Grasso, M., 2017. Twenty years of ecosystem services: How far have we come and how far do we still need to go? Ecosyst. Serv. 28, 1–16. https://doi.org/10.1016/j.ecoser.2017.09.008.

Crossman, N.D., Bryan, B.A., Summers, D.M., 2011. Carbon payments and low-cost conservation. Conserv. Biol. 25, 835–845. https://doi.org/10.1111/j.1523-1739.2011.01649.x.

Cruz Pardo, J., Yanes Puga, M., Sánchez Rojas, C.P., Simón Mata, M., 2010. Altiplano estepario. Ambientes semiáridos del sureste andaluz. Consejería de Medio Ambiente. Junta de Andalucía, Sevilla, Spain.

Cucci, G., Lacolla, G., Crecchio, C., Pascazio, S., De Giorgio, D., 2016. Impact of long term soil management practices on the fertility and weed flora of an almond orchard. Turk. J. Agric. For. 40, 194–202. https://doi.org/10.3906/tar-1502-87.

Das, S.K., Varma, A., 2011. Role of enzymes in maintaining soil health. In: Soil Enzymology, Soil Biology 22. Springer Berlin Heidelberg, pp. 25–42 https://doi.org/10.1007/978-3-642-14225-3.

De Giorgio, D., Lamascese, N., 2005. Long-term comparison among different soil tillage systems and weed control methods on almond tree growing in southern Italy. Options Méditerranéennes 63, 257–264.

Denys, C., Tscharntke, T., 2002. Plant-insect communities and predator-prey ratios in field margin strips, adjacent crop fields, and fallows. Oecologia 130, 315–324. https://doi.org/10.1007/s004420100796.

de Castro, M., Martín-Vide, M., Alonso, S., 2004. The climate of Spain: Past, present and scenarios for the 21st century. Internet 1–62.

Durán Zuazo, V.H., Francia Martínez, J.R., Rodríguez Pleguezuelo, C.R., Martínez Raya, A., Carcéles Rodríguez, B., 2006. Soil-erosion and runoff prevention by plant covers in a mountainous area (SE Spain): implications for sustainable agriculture. Environmentalist 26, 309–319. https://doi.org/10.1007/s10669-006-0160-4.

Durán Zuazo, V.H., Rodríguez Pleguezuelo, C.R., Martínez-Raya, A., Francia Martínez, J.R., Arroyo Panadero, L., Cárceles Rodriguez, B., 2008. Environmental and agronomic benefits of aromatic and medicinal plant strips for rainfed almond orchards in semiarid slopes (SE, Spain). Open Agric. J. 2, 15–21. https://doi.org/10.2174/1874331500802010015.

Eilers, E.J., Klein, A.M., 2009. Landscape context and management effects on an important insect pest and its natural enemies in almond. Biol. Control 51, 388–394. https://doi.org/10.1016/j.biocontrol.2009.07.010.

Faraway, J.J., 2016. Extending the Linear Model with R; Generalized Linear, Mixed Effects and Nonparametric Regression Models, second ed. CRC Press, Boca Raton, United States America.

Foley, J.A., Defries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin, F.S., Coe, M.T., Daily, G.C., Gibbs, H.K., Helkowski, J.H., Holloway, T., Howard, E.A., Kucharik, C.J., Monfreda, C., Patz, J.A., Prentice, I.C., Ramankutty, N., Snyder, P.K., 2005. Global consequences of land use. Science (80-) 309 (5734), 570–574. https://doi.org/10.1126/science:1111772.

Fracchiolla, M., Terzi, M., Frabboni, L., Caramia, D., Lasorella, C., De Giorgio, D., Montemurro, P., Cazzato, E., 2015. Influence of different soil management practices on ground-flora vegetation in an almond orchard. Renew. Agric. Food Syst. 31, 300–308. https://doi.org/10.1017/S1742170515000241.

Gaba, S., Lescourret, F., Boudsocq, S., Enjalbert, J., 2015. Multiple cropping systems as drivers for providing multiple ecosystem services: from concepts to design. Agron. Sustain. Dev. 35, 607–623. https://doi.org/10.1007/s13593-014-0272-z.

Gamfeldt, L., Snäll, T., Bagchi, R., Jonsson, M., Gustafsson, L., Kjellander, P., Ruiz-Jaen, M.C., Fröberg, M., Stendahl, J., Philipson, C.D., Mikusinski, G., Andersson, E., Westerlund, B., Andrén, H., Moberg, F., Moen, J., Bengtsson, J., 2013. Higher levels of multiple ecosystem services are found in forests with more tree species. Nat. Commun. 4, 1–8. https://doi.org/10.1038/ncomms2328.

García-Ruiz, J.M., 2010. The effects of land uses on soil erosion in Spain: a review. Catena 81, 1–11. https://doi.org/10.1016/j.catena.2010.01.001.

Gómez, J., Sobrinho, T.A., Giraldez, J.V., Fereres, E., 2009. Soil management effects on runoff, erosion and soil properties in an olive grove of Southern Spain. Soil Tillage Res. 102, 5–13. https://doi.org/10.1016/j.still.2008.05.005.

Hernández, A.J., Lacasta, C., Pastor, J., 2005. Effects of different management practices on soil conservation and soil water in a rainfed olive orchard. Agric. Water Manag. 77, 232–248. https://doi.org/10.1016/j.agwat.2004.09.030.

IPBES, 2018. Summary for policymakers of the regional assessment of biodiversity and ecosystem services for Europe and Central Asia of the Intergovernmental Science-Policity Platform on Biodiversity and Ecosystem Services, M. Fischer, M. Rounsevell, A. Torre-Marin Rando, A. Mader, A. Church, M. Elbakidze, V. Elias, T. Hahn. P.A. Harrison, J. Hauck, B. Martín-López, I. Ring, C. Sandström, I. Sousa Pinto, P. Visconti, N.E. Zimmermann, M. Christie (Eds.). Bonn, Germany.

IPCC, 2006. IPCC guidelines for national greenhouse gas inventories – Volume 4 Agriculture, forestry and other land use. In: Eggleston, H.S., Buendia, L., Miwa, K., Ngara, T., Tanabe, K. (Eds.), Prepared by the National Greenhouse Gas Inventories Programme. IGES, Japan.

Ismail, I., Blevins, R.L., Frye, W.W., 1994. Long-term no-tillage effects on soil properties and continuous corn yields. Soil Sci. Soc. Am. J. 58, 193. https://doi.org/10.2136/sssaj1994.03615995005800010028x.

Jezeer, R.E., Santos, M.J., Boot, R.G.A., Junginger, M., Verweij, P.A., 2018. Effects of shade and input management on economic performance of small-scale Peruvian coffee systems. Agric. Syst. 162, 179–190. https://doi.org/10.1016/j.agsy.2018.01.014.

Jezeer, R.E., Verweij, P.A., Santos, M.J., Boot, R.G.A., 2017. Shaded coffee and cocoa – double dividend for biodiversity and small-scale farmers. Ecol. Econ. 140, 136–145. https://doi.org/10.1016/j.ecolecon.2017.04.019.

Junta de Andalucía, 2015. Anuario de estadísticas agrarias y pesqueras año 2015. Sevilla, Spain.

Kandeler, E., Gerber, H., 1988. Short-term assay of soil urease activity using colorimetric determination of ammonium. Biol. Fertil. Soils 6, 68–72. https://doi.org/10.1007/BF00257924.

Kassam, A., Friedrich, T., Derpsch, R., Lahmar, R., Mrabet, R., Basch, G., González-Sánchez, E.J., Serraj, R., 2012. Conservation agriculture in the dry Mediterranean climate. F. Crop. Res. 132, 7–17. https://doi.org/10.1016/j.fcr.2012.02.023.

Kearney, S.P., Fonte, S.J., García, E., Siles, P., Chan, K.M.A., Smukler, S.M., 2017. Evaluating ecosystem service trade-offs and synergies from slash-and-mulch agroforestry systems in El Salvador. Ecol. Ind. 1–15. https://doi.org/10.1016/j.ecolind.2017.08.032.

Klein, A.M., Brittain, C., Hendrix, S.D., Thorp, R., Williams, N., Kremen, C., 2012. Wild pollination services to California almond rely on semi-natural habitat. J. Appl. Ecol. 49, 723–732. https://doi.org/10.1111/j.1365-2664.2012.02144.x.

Kremen, C., Miles, A., 2012. Ecosystem Services in Biologically Diversified versus Conventional Farming Systems: Benefits, Externalities, and Trade-Offs. Ecol. Soc. 17.

Lal, R., 2006. Enhancing crop yield in the developing countries through restoration of soil organic carbon pool in agricultural lands. L. Degrad. Dev. 209, 197–209.

Le, Q.B., Nkonya, E., Mirzabaev, A., 2016. Biomass productivity-based mapping of global land degradation hotspots. In: Economics of Land Degradation and Improvement – A Global Assessment for Sustainable Development. Springer International Publishing, Cham, pp. 55–84. https://doi.org/10.1007/978-3-319-19168-3_4.

Lee, H., Lautenbach, S., 2016. A quantitative review of relationships between ecosystem services. Ecol. Ind. 66, 340–351. https://doi.org/10.1016/j.ecolind.2016.02.004.

Luo, Z., Wang, E., Sun, O.J., 2010. Can no-tillage stimulate carbon sequestration in agricultural soils? a meta-analysis of paired experiments. Agric. Ecosyst. Environ. 139, 224–231. https://doi.org/10.1016/j.agee.2010.08.006.

Martínez-Mena, M., Garcia-Franco, N., Almagro, M., Ruiz-Navarro, A., Albaladejo, J., de Aguilar, J.M., Gonzalez, D., Querejeta, J.I., 2013. Decreased foliar nitrogen and crop yield in organic rainfed almond trees during transition from reduced tillage to no-tillage in a dryland farming system. Eur. J. Agron. 49, 149–157. https://doi.org/10.1016/j.eja.2013.04.006.

Meerkerk, A., van Wesemael, B., Cammeraat, E., 2008. Water availability in almond orchards on marl soils in southeast Spain: the role of evaporation and runoff. J. Arid Environ. 72, 2168–2178. https://doi.org/10.1016/j.jaridenv.2008.06.017.
Minaei, K., 2014. A new species of Eremiothrips from Iran (Thysanoptera: Thripidae). ACTA Entomol. Musei Natl. Pragae 54, 29–34.
Ministerio de Agricultura Pesca y Alimentación, 2018. Superficies y producciones anuales de cultivos [WWW Document]. URL https://www.mapama.gob.es/es/estadistica/temas/estadisticas-agrarias/agricultura/superficies-producciones-anuales-cultivos/.
Mirzabaev, A., Nkonya, E., Goedecke, J., Johnson, T., Anderson, W., 2016. Global drivers of land degradation and improvement. In: Economics of Land Degradation and Improvement – A Global Assessment for Sustainable Development, pp. 167–195. https://doi.org/10.1007/978-3-319-19168-3.
Montanaro, G., Celano, G., Dichio, B., Xiloyannis, C., 2010. Effects of soil-protecting agricultural practices on soil organic carbon and productivity in fruit tree orchards. L. Degrad. Dev. 21, 132–138. https://doi.org/10.1002/ldr.917.
Montanaro, G., Dichio, B., Briccoli, C., Xiloyannis, C., 2012. Soil management affects carbon dynamics and yield in a Mediterranean peach orchard. Agric. Ecosyst. Environ. 161, 46–54. https://doi.org/10.1016/j.agee.2012.07.020.
Moreira Modueño, J.M., Rodríguez Surián, M., 2008. Escenarios climáticos y desertificación en Andalucía. Sevilla.
Muhammad, S., Sanden, B.L., Lampinen, B.D., Saa, S., Siddiqui, M.I., Smart, D.R., Olivos, A., Shackel, K.A., DeJong, T., Brown, P.H., 2015. Seasonal changes in nutrient content and concentrations in a mature deciduous tree species: Studies in almond (Prunus dulcis (Mill.) D.A. Webb). Eur. J. Agron. 65, 52–68. https://doi.org/10.1016/j.eja.2015.01.004.
Müther, J., Vogt, H., 2003. Sampling methods in orchard trials: a comparison between beating and inventory sampling. Pestic. Benef. Org. 26, 67–72.
Nakagawa, S., Schielzeth, H., Johnson, P.C.D., 2017. The coefficient of determination R 2 and intra-class correlation coefficient from generalized linear mixed-effects models revisited and expanded. J. R. Soc. Interface 14, 1–35. https://doi.org/10.1098/rsif.2017.0213.
Navarro López, E.M., Navarro Sanchez, A.C., Roth, D., Schimansky, S., 2012. Vélez Blanco; Medio Natural, Historia y Patrimonio Cultural, Pueblos de Almeria. Vélez-Blanco, Alméria. https://doi.org/10.1017/CBO9781107415324.004.
Nkonya, E., Mirzabaev, A., 2016. Economics of Land Degradation and Improvement – A Global Assessment for Sustainable Development. Springer https://doi.org/10.1007/978-3-319-19168-3.
Norfolk, O., Eichhorn, M.P., Gilbert, F., 2016. Flowering ground vegetation benefits wild pollinators and fruit set of almond within arid smallholder orchards. Insect Conserv. Divers. 9, 236–243. https://doi.org/10.1111/icad.12162.
Norris, C.E., Congreves, K.A., 2018. Alternative management practices improve soil health indices in intensive vegetable cropping systems: a review. Front. Environ. Sci. 6, 50. https://doi.org/10.3389/fenvs.2018.00050.
Ortiz-Sánchez, F.J., Tinaut, A., 1993. Composición y variación latitudinal de la fauna de polinizadores potenciales del almendro, Prunus dulcis (Mill.), en la provincia de Granada (sur de España). Bol. San. Veg. Plagas 19, 491–502.
Ouédraogo, E., Mando, A., Zombré, N.P., 2001. Use of compost to improve soil properties and crop productivity under low input agricultural system in West Africa. Agric. Ecosyst. Environ. 84, 259–266. https://doi.org/10.1016/S0167-8809(00)00246-2.
Parras-Alcántara, L., Lozano-garcía, B., Keesstra, S., Cerdà, A., Brevik, E.C., 2016. Long-term effects of soil management on ecosystem services and soil loss estimation in olive grove top soils. Sci. Total Environ. 571, 498–506. https://doi.org/10.1016/j.scitotenv.2016.07.016.
Ponisio, L.C., M'Gonigle, L.K., Kremen, C., 2016. On-farm habitat restoration counters biotic homogenization in intensively managed agriculture. Glob. Chang. Biol. 22, 704–715. https://doi.org/10.1111/gcb.13117.
Pratt, P.F., 1965. Methods of soil analysis. Am. Soc. Agron. 1027–1028.
Pribyl, D.W., 2010. A critical review of the conventional SOC to SOM conversion factor. Geoderma 156, 75–83. https://doi.org/10.1016/j.geoderma.2010.02.003.
Ramos-Font, M.E., Robles, A.B., González-Rebollar, J.L., 2009. Sistemas agrosilvopastorales en el sudeste Ibérico: una alternative sostenible a los monocultivos leñosos de secano.
Ramos, María E., Benítez, E., García, P.A., Robles, A.B., 2010a. Cover crops under different managements vs. frequent tillage in almond orchards in semiarid conditions: effects on soil quality. Appl. Soil Ecol. 44, 6–14. https://doi.org/10.1016/j.apsoil.2009.08.005.
Ramos, M.E., Robles, A.B., González-Rebollar, J.L., 2010b. Ley-farming and seed dispersal by sheep: two methods for improving fallow pastures in semiarid Mediterranean environments? Agric. Ecosyst. Environ. 137, 124–132. https://doi.org/10.1016/j.agee.2010.01.012.
Ramos, M.E., Robles, A.B., Sánchez-Navarro, A., González-Rebollar, J.L., 2011. Soil responses to different management practices in rainfed orchards in semiarid environments. Soil Tillage Res. 112, 85–91. https://doi.org/10.1016/j.still.2010.11.007.
Raudsepp-Hearne, C., Peterson, G.D., Bennett, E.M., 2010. Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. PNAS 107, 5242–5247. https://doi.org/10.1073/pnas.0907284107.
Rodríguez, J.P., Beard Jr., T.D., Bennett, E.M., Cumming, G.S., Cork, S.J., Agard, J., Dobson, A.P., Peterson, G.D., 2006. Trade-offs across space, time, and ecosystem services. Ecol. Soc. 11. https://doi.org/10.5751/ES-01667-110128.
Ruiz-Mirazo, J., Robles, A.B., 2012. Impact of targeted sheep grazing on herbage and holm oak saplings in a silvopastoral wildfire prevention system in south-eastern Spain. Agrofor. Syst. 86, 477–491. https://doi.org/10.1007/s10457-012-9510-z.
Ruiz Sinoga, J.D., Garcia Marin, R., Martinez Murillo, J.F., Gabarron Galeote, M.A., 2011. Precipitation dynamics in southern Spain: Trends and cycles. Int. J. Climatol. 31, 2281–2289. https://doi.org/10.1002/joc.2235.
Santos, A.P., Raimundo, A., Bento, A., 2012. Species abundance patterns of coccinellid communities associated with olive, chestnut and almond crops in north-eastern Portugal. Agric. For. Entomol. 14, 376–382. https://doi.org/10.1111/j.1461-9563.2012.00578.x.
Saunders, M.E., Luck, G.W., Mayfield, M.M., 2013. Almond orchards with living ground cover host more wild insect pollinators. J. Insect Conserv. 17, 1011–1025. https://doi.org/10.1007/s10841-013-9584-6.
Scherr, S.J., McNeely, J.A., 2008. Biodiversity conservation and agricultural sustainability: towards a new paradigm of "ecoagriculture" landscapes. Philos. Trans. R. Soc. Lond. B Biol. Sci. 363, 477–494. https://doi.org/10.1098/rstb.2007.2165.
Schulte, L.A., Niemi, J., Helmers, M.J., Liebman, M., Arbuckle, J.G., James, D.E., Randall, K., Neal, M.E.O., Tomer, M.D., Tyndall, J.C., Drobney, P., Neal, J., Van Ryswyk, G., Schulte, L.A., Niemi, J., Helmers, M.J., Liebman, M., Arbuckle, J.G., James, D.E., 2017. Prairie strips improve biodiversity and the delivery of multiple ecosystem services from corn – soybean croplands. PNAS 114, 11247–11252. https://doi.org/10.1073/pnas.1719680114.
Sgroi, F., Foderà, M., Di Trapani, A.M., Tudisca, S., Testa, R., 2015. Cost-benefit analysis: a comparison between conventional and organic olive growing in the Mediterranean Area. Ecol. Eng. 82, 542–546. https://doi.org/10.1016/j.ecoleng.2015.05.043.
Silva, E.B., Franco, J.C., Vasconcelos, T., Branco, M., 2010. Effect of ground cover vegetation on the abundance and diversity of beneficial arthropods in citrus orchards. Bull. Entomol. Res. 100, 489–499. https://doi.org/10.1017/S0007485309990526.
Simonne, E.H., Mills, H.A., Jones, J.B., Smittle, D.A., Hussey, C.G., 1994. A comparison of analytical methods for nitrogen analysis in plant tissues. Commun. Soil Sci. Plant Anal. 25, 943–954. https://doi.org/10.1080/00103629409369090.
Sinsabaugh, R.L., Lauber, C.L., Weintraub, M.N., Ahmed, B., Allison, S.D., Crenshaw, C., Contosta, A.R., Cusack, D., Frey, S., Gallo, M.E., Gartner, T.B., Hobbie, S.E., Holland, K., Keeler, B.L., Powers, J.S., Stursova, M., Takacs-Vesbach, C., Waldrop, M.P., Wallenstein, M.D., Zak, D.R., Zeglin, L.H., 2008. Stoichiometry of soil enzyme activity at global scale. Ecol. Lett. 11, 1252–1264. https://doi.org/10.1111/j.1461-0248.2008.01245.x.
Tejada, M., Hernandez, M.T., Garcia, C., 2006. Application of two organic amendments on soil restoration: effects on the soil biological properties. J. Environ. Qual. 35, 1010–1017. https://doi.org/10.2134/jeq2005.0460.
Teuscher, M., Vorlaufer, M., Wollni, M., Brose, U., Mulyani, Y., Clough, Y., 2015. Trade-offs between bird diversity and abundance, yields and revenue in smallholder oil palm plantations in Sumatra, Indonesia. Biol. Conserv. 186, 306–318. https://doi.org/10.1016/j.biocon.2015.03.022.
Tittonell, P., 2014. Ecological intensification of agriculture — sustainable by nature. Curr. Opin. Environ. Sustain. 8, 53–61. https://doi.org/10.1016/j.cosust.2014.08.006.
Tscharntke, T., Klein, A.M., Kruess, A., Steffan-Dewenter, I., Thies, C., 2005. Landscape perspectives on agricultural intensification and biodiversity ecosystem service management. Ecol. Lett. 8, 857–874. https://doi.org/10.1111/j.1461-0248.2005.00782.x.
Vicente-Vicente, J.L., García-ruiz, R., Francaviglia, R., Aguilera, E., Smith, P., 2016. Soil carbon sequestration rates under Mediterranean woody crops using recommended management practices: a meta-analysis. Agric. Ecosyst. Environ. 235, 204–214. https://doi.org/10.1016/j.agee.2016.10.024.
Walkley, A., Black, I.A., 1933. An examination of the degtjareff method for determining soil organic matter and proposed modification of the chromic acid titration method. Soil Sci. 37, 29–38.
Watanabe, F.S., Olsen, S.R., 1965. Test of an ascorbic acid method for determining phosphorus in water and NaHCO3 extracts from soil. Soil Sci. Soc. Am. J. 29, 677–678.
Westphal, C., Bommarco, R., Carré, G., Lamborn, E., Petanidou, T., Potts, S.G., Roberts, S.P.M., Szentgyörgyi, H., Tscheulin, T., Vaissière, B.E., Woyciechowski, M., Biesmeijer, J.C., Kunin, W.E., Settele, J., Steffan-dewenter, I., Monographs, E., Biesmeijer, C., Kunin, E., 2008. Measuring bee diversity in different european habitats and biogeographical regions published by: Ecological Society of America Your use of the JSTOR archive indicates your acceptance of JSTOR's Terms and Conditions of Use. Ecol. Monogr. 78, 653–671 available at: http://www.js.
Zuur, A., Hilbe, J., Ieno, E., 2013. A Beginner's Guide to GLM and GLMM with R. Highland Statistics Ltd, UK.