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Management of a Mediterranean Forage/Cereal-Based Cropping System: An Ecosystem Service Multisectoral Analysis in the Perspective of Climate Change

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Abstract: Within Mediterranean cropping systems, legume forage crops that last up to 6 years or more (e.g., alfalfa) are replaced with cereal crops (e.g., wheat). The change from forage to cereal crops has negative climate and environmental impacts that must be addressed with mitigation actions. This study evaluated the synergies and tradeoffs between the ecosystem services provided by three management systems after forage legume. A field trial was set up from October 2017 to September 2019 on a 6-year-old alfalfa field subjected to the following management systems: (i) alfalfa termination followed by wheat for 2 years (WW, control); (ii) alfalfa termination followed by single amendment with 60 Mg ha^{-1} recalcitrant biochar and then by wheat for 2 years (WW_{B60}); and (iii) extension of alfalfa for 2 years (A_{EXT}). A range of regulating, supporting, and provisioning ecosystem services were assessed during the 2018 and 2019 cropping seasons. The results highlight that WW_{B60} can guarantee carbon sequestration without causing tradeoffs with other services, while A_{EXT} can enhance soil conservation while not increasing soil greenhouse gas emissions. Future policies should support the WW_{B60} system if the goal is to increase the supporting services. Conversely, the A_{EXT} system should be used if the goal is to increase the regulating and provisioning services.

Keywords: alfalfa; carbon sequestration; CH_4; CO_2; gasification biochar; greenhouse gases; N_2O; soil carbon stock; wheat

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1. Introduction

Globally, the agricultural sector provides about 25% of total greenhouse gas (GHG) emissions, and thus contributes substantially to climate change [1]. Within the different cropping systems, the implementation of different management practices can mitigate or adapt these effects of agricultural practices on climate change [2,3]. Widely used practices (e.g., soil tillage) can significantly influence the mineralisation rates of the soil organic matter [4], thus also contributing to soil GHG emissions [5–7]. In contrast, conservative practices (e.g., no tillage) are increasingly being adopted to reduce soil GHG emissions, which in some cases can also enhance crop yields [8]. The inclusion of perennial legume forage crops into crop rotation has been shown to enhance soil conservation by increasing soil organic matter [9] and limiting soil erosion [10]. Other widely used practices include nitrogen fertilisation, which can increase soil GHG emissions [2–8], and can also result in nitrate leaching [11,12].

Within the current climate change scenario, it is necessary to move towards sustainable agriculture, with the adoption of the best management practices to mitigate or adapt to climate change, while ensuring the current need for food security [2,13]. The positive

effects that can be derived from management practices adopted within cropping systems to enhance food production, and more in general, for the economic sustenance of farmers and the whole environment, have an influence on human well-being [14–16]. Such positive effects are defined as 'ecosystem services' [17].

It is well known that the different crop production systems can provide a wide array of ecosystem services to humans [14], and also that the provision of such services varies as a function of the management practices applied [16]. According to the framework proposed by the Millennium Ecosystem Assessment [17,18], ecosystem services can be divided into four categories: provisioning services (e.g., production of food, wood, fuel); supporting services (e.g., nutrient cycling, soil formation, habitat provision); regulating services (e.g., climate regulation, reduction in water runoff, soil erosion); and cultural services (e.g., spiritual, aesthetic, educational services).

The framework of analysis proposed by the Millennium Ecosystem Assessment includes a multisectoral and multiscale approach, and requires the inclusion of different types of knowledge by different stakeholders [17,18]. Despite the large numbers of studies that have used the Millennium Ecosystem Assessment framework, recent reviews show that most of these studies have largely ignored the potential tradeoffs between ecosystem services (e.g., increasing the flow of one service obtained from a system—such as provision of food, and those that might decrease the flow from others—such as carbon sequestration) [19].

Most Mediterranean cropping systems are characterised by cyclic alternation of perennial legume forage crops (mainly alfalfa) with winter cereals (e.g., durum wheat) and spring crops (e.g., corn, sunflower) [6,7,9]. Within this forage/cereal-based cropping system, alfalfa stands are normally terminated after 3–4 years, because of their loss of productivity [5,6]. After this period, a winter cereal is cultivated (mainly durum wheat), sometimes for two consecutive years. However, farmers sometimes choose not to terminate some alfalfa stands after 3–4 years, settling for the reduced quantitative and qualitative forage production normally obtained by a single mowing per year. This management system is not uncommon for fields in marginal areas and characterised by pronounced slope [5,6]. After mowing, these alfalfa fields are often grazed by transhumant sheep farms during the winter period [20,21]. The Mediterranean basin area is considered a 'hot spot' in terms of climate change, and recent climate projections show increased extremes within the next few decades, such as heat stress, drought, and heavy rain, which can all lead to crop yield losses [22,23]. Different climate change adaptation and mitigation systems for the Mediterranean cropping management have been proposed, which have included minimum tillage [24], use of cover crops [25], and postponed tillage [6]. A more recent and promising management system in terms of climate change adaptation/mitigation for Mediterranean cropping management might be the use of biochar [26,27]. Biochar is a carbonaceous material that is produced by heating organic material to high temperatures (350–1200 °C) and in a low-oxygen supply [26]. Biochar can be specifically produced to obtain a soil amendment to be used in agriculture, e.g., [28], or it can be obtained as a byproduct or coproduct resulting from thermal and electrical energy plants [29]. In this second case, companies need to find a suitable disposal solution for the biochar, and its utilisation as a soil amendment is among the possible options [26,29–32]. In this perspective, policies could support the use of biochar in agriculture by covering the farmers' costs (e.g., transportation, distribution), while increasing the soil carbon stock during the long term [33]. Recent studies have suggested that under Mediterranean conditions, the transition from perennial legume forage crops to cereals can result in rapid depletion of soil organic matter [34], with potential implications also on soil CO_2 emissions [5,7]. The addition of biochar between the transition from forage to cereal crops (e.g., from long-term alfalfa to durum wheat) would guarantee long-term carbon sequestration in the soil [26,27,33]. However, mitigation of soil GHG emissions by biochar remains context dependent (e.g., soil pH and texture; biochar feedstock, production temperature, pH, and application rate) [35,36].

Generally, perennial legume forage crops such as alfalfa are linked to higher levels of biodiversity and ecosystem services compared to more intensive crops [37]. Perennial crops do not require tillage or fertilisation, and for these reasons they can mitigate direct and indirect CO_2 and N_2O emissions [1,38]. Moreover, they increase the soil organic matter [9], fix atmospheric nitrogen in the soil [39], and provide for an almost constant soil cover, to prevent soil erosion [10]. However, as alfalfa stands age, they progressively reduce their yield production while also increasing the weed ratio and density [40,41], and therefore they are usually terminated in favour of other crops. The perennial legume termination by tillage decreases the soil C stock, while increasing soil CO_2 fluxes through mineralisation process of the soil organic matter [7,9]. However, the advantages of soil C stock enhancement could be lost in the short term after tillage [5,34].

While several studies have analysed both perennial forage and cereal crops under Mediterranean conditions, there is the need for more studies that focus on the effects of forage-to-cereal transition, in terms of ecosystem services, such as increased soil fertility or reduced GHG emissions. Moreover, studies that have investigated this transition from the perspective of ecosystem services and have also considering synergies and their tradeoffs are rare, if not available at all. Such multisectoral analysis of the effects produced by forage-to-cereal transition would be a useful tool for the identification of sustainable management systems and to provide recommendations for future policies in the light of ongoing climate change.

In this perspective, the aim of the present study was to evaluate the synergies and tradeoffs between ecosystem services that are generated by two alternative management systems compared to the commonly adopted termination of perennial forage legumes (e.g., alfalfa) followed by a cereal crop (e.g., wheat). In this study, this conventional system (alfalfa termination followed by 2 years of durum wheat; WW, control) was compared with alfalfa termination followed by single soil amendment with 60 Mg ha^{-1} recalcitrant biochar, and then cultivation of durum wheat for 2 years (WW_{B60}), and extension of alfalfa for 2 years (A_{EXT}). For these management systems, we formulated the hypothesis that WW_{B60} would provide carbon sequestration without generation of tradeoffs with other ecosystem services. Alternatively, A_{EXT} would ensure greater soil conservation due to greater soil cover, while not significantly increasing soil GHG emissions.

2. Materials and Methods

2.1. Study Area

This study was carried out in a hilly area of the Marche region, central Italy (43°33′ N, 13°25′ E; 100 m a.s.l.; 23% slope; SW exposure). The climate of the study area is classified as a variant of the temperate oceanic sub-Mediterranean climate, and characterised by a mean annual precipitation of ~800 mm and a mean annual temperature of ~14.5 °C. The meteorological data during the study period were recorded by a weather station located 0.3 km from the experimental field, and are reported in Figure 1. The soil of the study area is classified as Inceptisol according to the United States Department of Agriculture soil taxonomy system, with a pH of 8.13, a loam texture (36.1% sand; 25.6% clay), 1.5% soil organic matter, and a field capacity of 24.6%, and a wilting point of 17.8% [6].

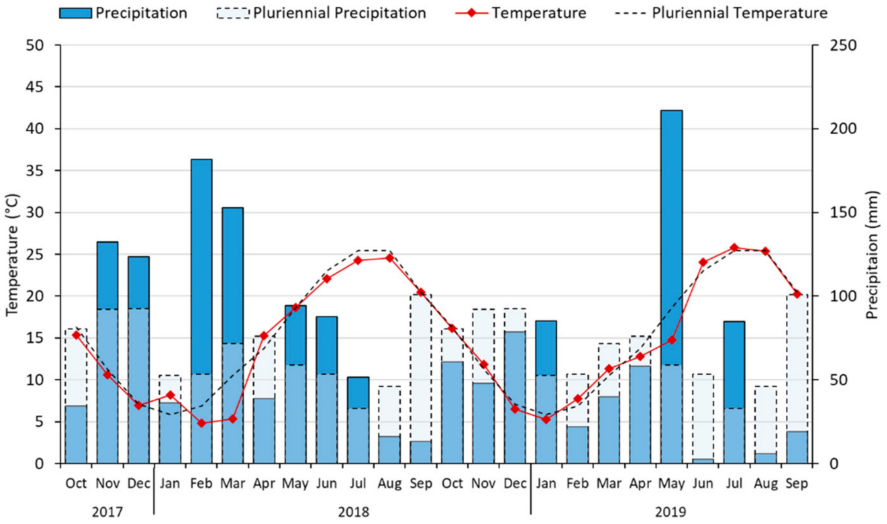

Figure 1. Monthly mean precipitation and air temperatures for the study area during the study period (October 2017–September 2019), and during the long term (pluriennial; 1998–2012).

2.2. Experimental Design and Management Practices

2.2.1. Experimental Design

The field trial lasted from October 2017 to September 2019. In the summer of 2017, an area that was homogeneous for soil, crop vegetation and topographic conditions was identified within a 6-year-old alfalfa ley and was fenced off to prevent any disturbance. This area was subjected to three different management systems that were allocated according to a complete randomised block design with three replicates, with individual plots of 2.5 m $\times$ 10.0 m. The management systems were: (i) alfalfa termination followed by sowing of durum wheat (*Triticum turgidum* L. ssp. *durum* (Desf.) Husn.) for two cropping seasons (WW, control); (ii) alfalfa termination followed by single amendment with 60 Mg ha^{-1} biochar and then by sowing of durum wheat for two cropping seasons (WW_{B60}); and (iii) extension of the 6-year-old alfalfa field for two additional cropping seasons (A_{EXT}).

2.2.2. Management Practices

For both WW and WW_{B60}, the termination of the 6-year-old alfalfa field was performed on 11 October 2017, using a spading machine to a depth of 0.2 m. This practice provided approximately 2.53 $\pm$ 0.14 Mg dry matter ha^{-1} of alfalfa aboveground biomass that was incorporated into the soil. In the three WW_{B60} plots, 60 Mg ha^{-1} biochar was applied on the soil surface in the form of powder on 16 October 2017. For the WW and WW_{B60} plots, tillage was subsequently performed with a rotary harrow to a depth of 0.15 m on the same day as the biochar application. This incorporated the biochar into the soil, which was applied only once during the study period. The biochar used originated from a mix of beech (*Fagus sylvatica* L.), pine (*Pinus pinea* L.), and fir (*Abies alba* Mill.) wood, and was obtained by gasification at ~850 °C in an industrial plant system. For both WW and WW_{B60}, a second tillage with a rotary harrow to a depth of 0.15 m was performed on 21 November 2017. For the WW and WW_{B60} plots, durum wheat (cv. 'Antalis') was sown on 23 November 2017 at a sowing rate of 400 seeds m^{-2} and at a depth of 0.03 m, which was harvested on 4 July 2018.

In the second year of the study (i.e., 2018–2019 cropping season), the WW and WW_{B60} plots were subjected to the same operations as for the previous season, except that the biochar application was not repeated for WW_{B60}, and that 3.27 $\pm$0.50 Mg ha^{-1} wheat straw dry matter was incorporated into the soil during the tillage for WW and WW_{B60}. Wheat was then sown again on 16 November 2018, using the same procedures and rates as for 2017, which was here harvested on 4 July 2019, for both WW and WW_{B60}.

For A_{EXT}, three mowings per year were performed with a bar mower set at a cutting height of 0.05 m from the ground. In both 2018 and 2019, the first and second mowings were performed at the beginning of May and July, and the third mowing at the beginning of September 2018 and at the end of August in 2019. Soon after the mowing, the cut herbage was removed from each plot.

No fertilisers, amendments, or pesticides were applied to any of the experimental plots throughout the study period, in line with the common practices adopted in marginal areas of such production systems after legume crops [6]. Figure 2 illustrates the practices applied for each management system during the study period.

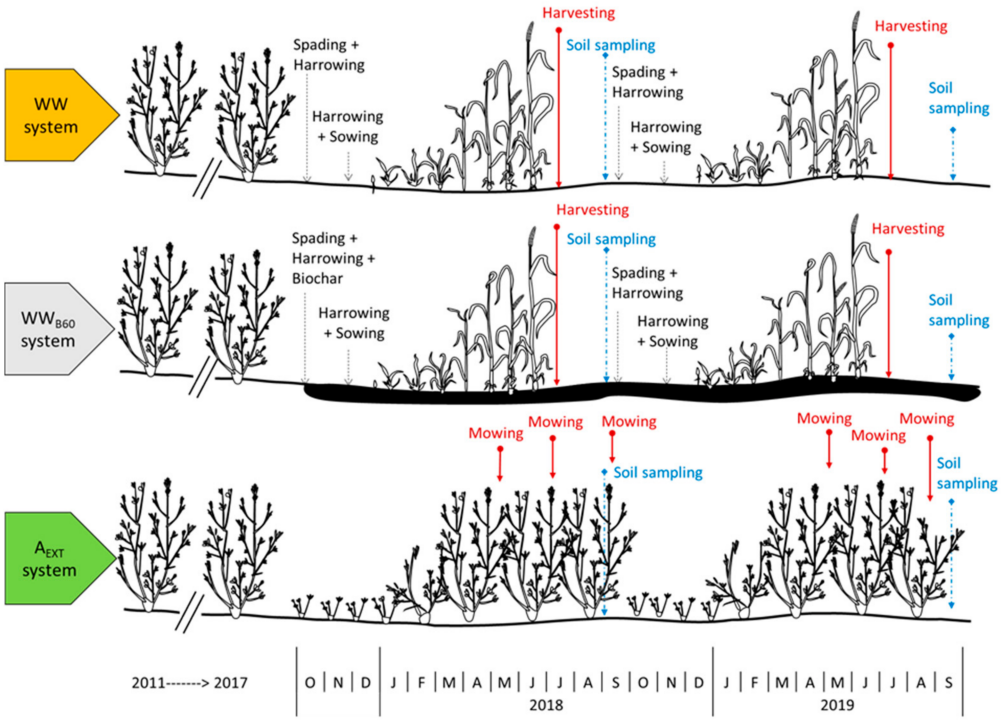

Figure 2. Management practices applied and samplings carried out for the different management systems during the study period (October 2017–September 2019).

2.3. Sampling

2.3.1. Botanical Survey

Botanical surveys were carried out after the wheat harvesting for the WW and WW_{B60} plots, and three times per year for the A_{EXT} plots, just before each mowing (Figure 2). For all of the plots, species composition and relative abundances were estimated according to Braun-Blanquet [42]. The taxonomical nomenclature followed Bartolucci et al. [43].

2.3.2. Biomass Sampling

For all of the management systems, the aboveground biomass was sampled in the central part of the plot, and immediately transported to the laboratory for further analysis. For the WW and WW_{B60} plots, the sample area was 2 m^2 (1.0 m × 2.0 m), while for the A_{EXT} plots, it was 1 m^2 (1.0 m × 1.0 m). For the WW and WW_{B60} plots, the biomass sampling was performed on the harvest dates, and thus once per year. For the A_{EXT} plots, the sampling dates corresponded to the dates of mowing, which were performed three times per year (Figure 2). For the A_{EXT} plots, the aboveground biomass was removed (i.e., alfalfa and other species), while for the WW and WW_{B60} plots, the wheat plants were cut at the base and removed (i.e., the weeds were left in the soil).

2.3.3. Soil Sampling

The soils were sampled on 5 September 2018 and 25 September 2019, for each plot (Figure 2). On each date, soil was collected from three points within each plot using a manual 0.05 m diameter auger, from a depth of 0–0.4 m. The soils collected from the same plot were mixed to obtain a single sample. The soil samples were taken immediately to the laboratory, sieved through a 2 mm mesh sieve, and stored in open plastic bags at ambient temperature for subsequent analysis.

2.3.4. Greenhouse Gas Sampling

Soil CO_2, N_2O, and CH_4 emissions were monitored during the study period using closed static polyvinylchloride chambers installed in the soil (depth, 0.1 m). Gas samples were collected between 9:00 a.m. and 12:00 noon (standard time), approximately every 15 days, except for the period between tillage and sowing, when they were increased to every 3–4 days. For all of these samples, CO_2 concentrations were measured using a Li-Cor 7000 system (Lincoln, NE, USA), while N_2O and CH_4 concentrations were measured by gas chromatography (GC8A; Shimadzu Corporation, Kyoto, Japan) with an electron capture detector. Further details on the calculation of CO_2, N_2O, and CH_4 efflux are provided in Trozzo et al. [6] and Toderi et al. [5].

The GHG sampling was divided into two seasons of equal length (323 days each, 36 and 27 sampling dates for the first and the second season, respectively): the first started and ended with the spading performed for the WW and WW_{B60} plots, while the second started the day after the spading in 2019 and ended exactly 323 days afterwards (Figure 2). The soil temperatures were measured on each day of each GHG sampling, at a depth of 0.10 m with a soil thermometer (620-0909; VWR International, Milan, Italy).

2.4. Plant Biodiversity and Ecosystem Service Indicators

A variety of different ecosystem service indicators were determined from the sampling described above. These indicators were selected to assess the provision of a range of ecosystem services and associated plant biodiversity for each of the management systems.

Two indices were considered as proxy indicators of plant biodiversity: Raunkiaer's life forms and species richness. Global warming potential (GWP), temperature sensitivity to soil respiration (Q_{10}), soil GHG intensity (GHG-I), soil total organic carbon (TOC), and soil vegetation cover (SVC) were the indicators considered for the regulating ecosystem services. Soil pH, total nitrogen (N-tot), and carbon:nitrogen (C/N) ratio were the indicators considered for the supporting ecosystem services. Crop energy output (NRG) was the indicator considered for the provisioning ecosystem services.

2.4.1. Biodiversity Indicators

Raunkiaer's plant life forms followed Pignatti [44] and were used as functional groups, with the annual, biennial, and perennial species expressed as numbers and relative abundances (%) of species. Species richness was the sum of the new species for each sampling date for each management system (i.e., WW, WW_{B60}, A_{EXT}).

2.4.2. Regulating Services

For each management system, the relationship between soil CO_2 emission and soil temperature at 0.1 m depth was determined according to Equation (1):

$$y = a\, e^{bx} \quad (1)$$

where y is the measured soil CO_2 emission, b is the soil temperature at 0–10 cm depth, and x is the fitting parameter. Subsequently, the b parameter from Equation (1) was used to calculate Q_{10}, according to Equation (2) [45]:

$$Q_{10} = e^{10b} \quad (2)$$

For each management system and each season, the cumulative soil CO_2, N_2O, and CH_4 fluxes were calculated by linear interpolation between the two sampling dates, and numerical integration using the trapezoidal rule. The GWP for each management system was calculated according to the Intergovernmental Panel on Climate Change guidelines, assuming a 100-year time horizon [1], according to Equation (3):

$$GWP = CO_2 + 265 \times N_2O + 28 \times CH_4 \quad (3)$$

The GHG-I was calculated by dividing the GWP by the crop yield [46], according to Equation (5):

$$GHG\text{-}I = GWP/\text{grain yield} \quad (4)$$

The soil TOC at 0–0.4 m in depth was determined for each management system using the dry combustion Springer–Klee method.

2.4.3. Supporting Services

For each management system, the soil pH was measured in soil:water suspension (1:2.5, *w*/*v*) using the combined glass–calomel electrode potentiometric method, while the soil N-tot was determined by the Kjeldahl method, and the soil C/N ratio was obtained by dividing TOC by N-tot.

2.4.4. Provisioning Services

For WW and WW_{B60}, to determine the wheat grain production, the harvested plants were threshed using a combine harvester (Wintersteiger Delta). For A_{EXT}, the forage dry matter content was estimated by oven drying of the samples at 65 °C for 72 h. To estimate the production of gross energy per unit of surface, the different dry matter contents per unit surface were multiplied by their respective energy contents, as 18.1 and 18.2 MJ kg DM^{-1} for alfalfa (aboveground biomass) and wheat (grain), respectively (data from www.feedipedia.org).

2.4.5. Integrated Provision of Ecosystem Services and Data Analysis

To integrate the information provided by the various ecosystem service indicators into an overall assessment of ecosystem services provision, the indicator values for each management system were standardised and then used to yield a composite value for each management system and for each ecosystem service group, as follows: regulating services (averaging GWP, GHG-I, TOC, Q_{10}, SVC); supporting services (averaging soil pH, N-tot, C/N ratio); and provisioning services (averaging NRG). As GWP, GHG-I, and Q_{10} are related to soil GHG emissions, these were considered as 'ecosystem disservices' (i.e., the higher the value, the more detrimental to human well-being). For this reason, the GWP, GHG-I, and Q_{10} for each management system are 'sign changed' to highlight their actions to reduce the associated disservice.

The ecosystem service indicators were compared across the three management systems within the same season of the study using one-way analysis of variance (ANOVA), followed by post hoc analysis using Tukey tests. The data were Box–Cox transformed to meet the assumptions of normality and homoscedasticity [47].

3. Results

3.1. Plant Biodiversity

For the life forms, annual species (e.g., *Polygonum aviculare*, *Erigeron canadensis*) were more abundant for WW and WW_{B60} compared to A_{EXT}, where the perennials (including alfalfa) were instead more abundant than the annuals (e.g., *Avena sterilis* subsp. *ludoviciana*, *Anisantha sterilis*) (Table 1; Figure 3). The biennial species were always the least represented, with no significant variations between the management systems. These dynamics were generally confirmed for both the 2018 and 2019 seasons.

The species richness in 2018 was higher for A_{EXT} than WW and WW_{B60} (Table 1). In 2019 there were no significant differences between the management systems for species richness.

Table 1. Plant diversity as influenced by the management systems across the two seasons.

Season	Biodiversity Index	Detail	Management System		
			WW	**WW_{B60}**	**A_{EXT}**
2018	Life forms (n)	Perennial	1.3 b	2.0 ab	3.0 a
		Biennial	0.3	0.3	0.0
		Annual	6.3 a	5.3 b	7.7 a
	Life forms (%)	Perennial	3.1 b	2.7 b	68.6 a
		Biennial	0.1	0.1	0.0
		Annual	96.9 a	97.3 a	31.4 b
	Species Richness	-	8.0 b	7.7 b	10.7 a
2019	Life forms (n)	Perennial	3.3	1.7	3.7
		Biennial	2.0 a	1.7 ab	1.0 b
		Annual	9.0	8.3	9.3
	Life forms (%)	Perennial	2.2 b	0.3 b	64.7 a
		Biennial	3.6	3.7	1.5
		Annual	94.3 a	96.0 a	33.8 b
	Species richness	-	14.3	11.7	14.0

WW, alfalfa followed by wheat for 2 years, control; WW_{B60}, alfalfa followed by single biochar application followed by wheat for 2 years; A_{EXT}, alfalfa extended for 2 years. Data with different letters are significantly different within each index (row) ($p < 0.05$; ANOVA, post hoc Tukey tests).

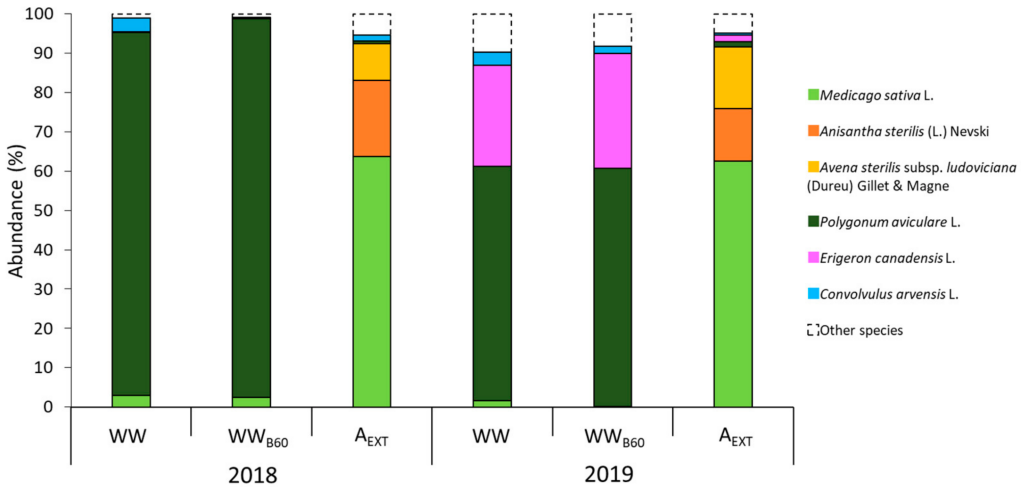

Figure 3. The six species that contributed at least 3% abundance for the three management systems across the two seasons. WW, alfalfa followed by wheat for 2 years, control; WW_{B60}, alfalfa followed by single biochar application followed by wheat for 2 years; A_{EXT}, alfalfa extended for 2 years.

3.2. Regulating Services

The GWP (i.e., soil GHG emissions, expressed as tons of equivalents CO_2 ha^{-1}) was higher for WW_{B60} compared to WW and A_{EXT} across both of the study seasons, although these differences did not reach significance (2018, 2019: $p = 0.14$, 0.11, respectively). However, in the 2019 season, GWP was decreased for all of the management systems compared to the 2018 season (Table 2). The sensitivity to soil respiration (Q_{10}) was significantly higher only for WW compared to A_{EXT} in 2018. The GHG-I was significantly higher for WW_{B60} compared to A_{EXT} in the 2018 season, and then in the 2019 season, significance was seen for both WW and WW_{B60} as higher than A_{EXT}. As expected, the TOC measured at 0–0.4 m in depth was significantly higher for WW_{B60} versus WW and for A_{EXT} across both of the

seasons. However, in 2019 WW showed a significant lower TOC also compared to A_{EXT} (Table 2). The SVC was always significantly higher for A_{EXT} compared to WW and WW_{B60}, which remained similar throughout the two seasons (Table 2).

Table 2. Indicators of ecosystem services as influenced by the management systems across the two seasons.

Season	Ecosystem Services		Management System		
	Group	Service	WW	WW_{B60}	A_{EXT}
2018	Regulating	GWP	19.68 ± 2.06	20.63 ± 1.73	17.33 ± 2.25
		Q_{10}	1.74 ± 0.07 a	1.67 ± 0.1 ab	1.59 ± 0.02 b
		GHG-I	9.02 ± 4.77 ab	10.05 ± 4.68 a	2.61 ± 0.53 b
		TOC	8.40 ± 0.87 b	16.95 ± 1.73 a	9.60 ± 2.61 b
		SVC	41.40 ± 1.94 b	37.96 ± 5.29 b	74.21 ± 1.21 a
	Supporting	pH	8.18 ± 0.05	8.23 ± 0.04	8.13 ± 0.04
		C/N ratio	8.68 ± 0.26 b	17.41 ± 2.09 a	9.82 ± 2.15 b
		N-tot	0.97 ± 0.07	0.98 ± 0.03	0.98 ± 0.11
	Provisioning	NRG	47.38 ± 22.45 ab	43.36 ± 19.53 b	122.08 ± 14.39 a
2019	Regulating	GWP	11.88 ± 0.48	14.98 ± 1.72	11.34 ± 1.52
		Q_{10}	1.89 ± 0.11	1.69 ± 0.16	1.93 ± 0.35
		GHG-I	4.01 ± 1.66 a	6.24 ± 1.22 a	1.02 ± 0.32 b
		TOC	9.10 ± 0.95 c	19.40 ± 2.21 a	11.25 ± 1.15 b
		SVC	50.27 ± 1.85 b	50.27 ± 1.85 b	71.50 ± 0.41 a
	Supporting	pH	8.28 ± 0.04 a	8.32 ± 0.02 a	8.19 ± 0.01 b
		C/N ratio	10.07 ± 0.48 b	19.62 ± 2.88 a	8.39 ± 0.74 b
		N-tot	0.91 ± 0.14	0.99 ± 0.04	1.34 ± 0.10
	Provisioning	NRG	59.00 ± 18.43 b	45.29 ± 12.58 b	209.90 ± 44.78 a

WW, alfalfa followed by wheat for 2 years, control; WW_{B60}, alfalfa followed by single biochar application followed by wheat for 2 years; A_{EXT}, alfalfa extended for 2 years; for ecosystem services abbreviations, see main text. Data with different letters are significantly different within each ecosystem service (row) ($p < 0.05$; ANOVA, post hoc Tukey tests).

3.3. Supporting Services

In 2019, soil pH was significantly lower for A_{EXT} compared to WW and WW_{B60}. For each of the study seasons, the soil C/N ratio was significantly higher for WW_{B60} compared to the other two management systems, which remained at similar levels. No significant differences were seen for soil N-tot across these management systems for both of the seasons (Table 2).

3.4. Provisioning Services

In the 2018 season, the NRG was significantly lower for WW_{B60} compared to A_{EXT}, with the same trend seen for WW versus A_{EXT} ($p = 0.06$). In 2019, these differences were consolidated, where the NRG was significantly lower for both WW and WW_{B60} compared to A_{EXT} ($p < 0.01$) (Table 2).

3.5. Ecosystem Service Global Value

Figure 4 shows the overall assessment of the variations among the different management systems for the provision of the three groups of ecosystem services analysed. The WW control was the worst management system in terms of the supply of regulating and supporting services. Supporting services were greater for WW_{B60}, while A_{EXT} resulted in the greatest supply of regulating and provisioning services (Figure 4).

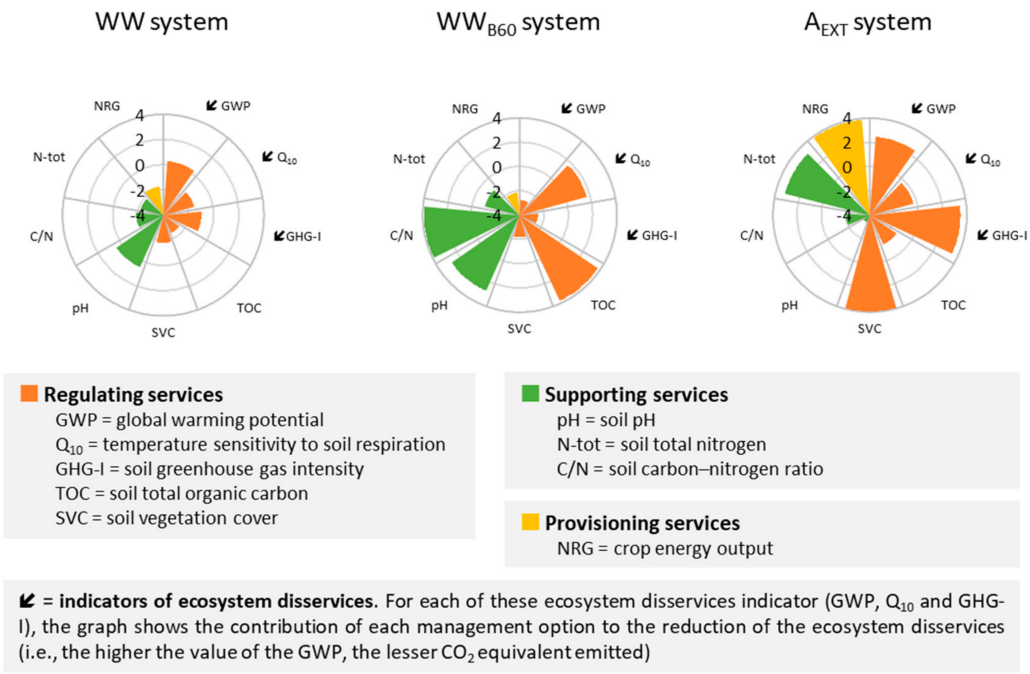

Figure 4. Relative contributions of each management system to provisioning of the ecosystem services. The data are based on the averages of the standardised values for the various indicators considered. WW, alfalfa followed by wheat for 2 years, control; WW_{B60}, alfalfa followed by single biochar application followed by wheat for 2 years; A_{EXT}, alfalfa extended for 2 years.

4. Discussion

Many cropping systems in the Mediterranean are characterised by cyclic alternations of forage legumes and cereal crops. While forage crops require low production inputs and are generally considered conservative, cereal crops are more resource demanding, as they require tillage also after sowing (e.g., corn), fertilisation, weeding treatments, and irrigation [7,9,48]. In the specific case of legume-to-cereal transition (e.g., alfalfa to durum wheat), detrimental effects on the environment are likely to occur. These detrimental effects, which can be defined as ecosystem disservices [49], can also include higher direct and indirect GHG emissions [6,7,9,13,14,50].

The present study examined two alternative management systems compared to termination of 6-year-old alfalfa stands followed by 2 years of conventional durum wheat cultivation (WW, control). In the first management system, after alfalfa termination, there was addition of a high dose of stable biochar (60 Mg ha^{-1}) before the durum wheat (WW_{B60}), with the main aim being to increase the soil carbon stock [26,27]. In the second management system, the duration of the alfalfa was extended for an additional 2 years (A_{EXT}), with the intention being to reduce direct and indirect soil GHG emissions [1,38] while providing continuous soil cover [10]. The results obtained confirmed a doubling of the soil total organic carbon (TOC) through the 0–0.4 m depth for WW_{B60} compared to WW, with no change in soil GHG emissions. However, from an ecosystem services perspective, the increases in the specific ecosystem services are likely to generate tradeoffs with other services, as also for A_{EXT}, and a multisectoral analysis should be performed for these alternative management systems [17].

In the following sections, the alternative management systems of WW_{B60} and A_{EXT} are discussed within each ecosystem service group (i.e., regulating, supporting, provisioning services) in the light of the Millennium Ecosystem Assessment framework, to highlight synergies and tradeoffs across these ecosystem services [18]. Finally, recommendations for future agri-environmental policies for these alternative management systems are proposed.

4.1. Regulating Services

In the present study, the soil CO_2 emissions represented about ~98% of the annual cumulative soil GWP for all of these management systems. Other studies have reported contributions of >90% for CO_2 in the calculation of soil GWP for Vertisols and Inceptisols [5,51]. Recent studies carried out under Mediterranean conditions have suggested that most of the peaks of N_2O emissions mainly occur after nitrogen fertilisation [52]. Under conditions such as those for the WW (i.e., control), soil GHG emissions would be expected to be limited, especially for negligible contributions of N_2O to GWP. This is because after long-term alfalfa, nitrogen fertilisation can be avoided, as it is assumed that the soil incorporation of the aboveground biomass provides enough nitrogen for the following crop [6]. As no significant differences emerged in terms of GWP between WW and WW_{B60}, it is possible that the high dose of biochar (i.e., 60 Mg ha^{-1}) added for WW_{B60} did not affect the soil organic matter mineralisation, which suggests that this biochar did not promote any priming effects (Table 2). This can be attributed to the biochar feedstock and its production temperature. Indeed, according to Leng et al. [53], biochar produced from lignocellulose biomass under high heating temperatures tends to be highly stable. Gasification biochar is generally characterised by high levels of condensed aromatic rings, which results in a more stable material that is less prone to microbial mineralisation [54].

The high stability of the biochar used is also partially supported by the Q_{10} values, which were not lower for WW_{B60} compared to WW (p = 0.14, and 0.5 in 2017 and 2018, respectively). In general, the Q_{10} for all of these management systems was within the range estimated for croplands in a recent global meta-analysis [55]. Conversely, the GHG-I for both WW and WW_{B60} were considerably higher than reported for similar crops in semiarid environments [46,56], which can be attributed to low grain yields compared to the averages of the study area when a moderate amount of fertiliser is applied [34].

Other studies that have investigated similar forage/cereal-based systems under the same climatic conditions have suggested that soil CO_2 emissions from alfalfa fields are higher than those of spring wheat, and that this is due to higher soil organic matter accumulation during the forage-based part [7]. Indeed, in the second year of the present study, the TOC was significantly higher for A_{EXT} compared to WW, although this did not result in higher GWP (i.e., soil GHG emissions, expressed as tons of equivalents CO_2 ha^{-1}) (Table 2). While this higher TOC for A_{EXT} might be attributable to the relatively high aboveground biomass production obtained for the first mowing in 2019 (i.e., 7.52 $\pm$1.2 Mg ha^{-1}), and thus presumably alfalfa root presence, the relatively low GHG emissions observed in 2019 for all of these management systems might be linked to the rainfall patterns of May and June 2019 (Figure 1). The soils of the study area are characterised by an ustic moisture regime [57], and while in May 2019 the exceptional amount of cumulate rainfall (~211 mm) might have created an anoxic soil environment, in the following month, the scarcity of precipitation (~2.54 mm) might have caused a scarcity of available water in the soil [7]. This contrasting rainfall pattern might have limited the contribution of both heterotrophic and autotrophic soil respiration, which was relatively low in 2019 for all of these management systems (Table 2).

Extreme rainfall events such as those observed in March 2019 are likely to happen more frequently [22], and are likely to cause extensive damage in terms of erosion in similar systems if the soil is left uncovered [58]. For all of the study period, A_{EXT} guaranteed a constant soil cover, which was only interrupted for a few days after the mowing (Figure 2); this will thus result in the most appropriate management to limit soil erosion in the case of extreme events [10,16,25]. Indeed, the higher soil cover for A_{EXT} consisted of both perennial and biennial species (68.6%, 66.2%, in 2018, 2019, respectively), such as the cespitose *Lolium perenne*, *P. pratensis* and *P. trivialis*, which as hemicryptophytic plants are better adapted to reduce soil erosion.

4.2. Supporting Services

The soils of the study area developed from thinly layered marine sediments, and are characterised by alkaline or subalkaline pH [57]. The C/N ratio of such soils is generally low, due to the high mineralisation rates that are typical of Mediterranean climates [12].

Despite the high dose of biochar (60 Mg ha^{-1}) provided for WW_{B60}, the soil pH did not undergone any significant change during the study period (Table 2). This might be explained by the subalkalinity of the soil, as it has been demonstrated that biochars have liming effects only in acid soils, while in alkaline soils, biochar addition does not generally result in soil pH increases [59]. On the other hand, for WW_{B60}, the C/N ratio increased significantly, and as expected, remained the highest for these different management systems also during the second season of the study (Table 2). This further confirms the stability of the biochar used, which was obtained by gasification, and which is usually characterised by a high degree of aromaticity and resistance to chemical oxidation [54]. Thus, WW_{B60} would be expected to positively affect both the soil C/N ratio and carbon sequestration during the long term [27]. In agreement with Hansen et al. [30], the addition of this type of biochar did not increase soil nitrogen mineralisation compared to WW, at least not during the 2-year study period (Table 2). However, medium-term to long-term studies on biochar ageing are needed, as other studies have suggested that aged biochar can result in detrimental effects, such as enhancement of N_2O emissions [60].

The lowest soil pH was observed for A_{EXT} during the second year of the study (Table 2), which might be ascribable to an increase in soil oxygenation for both WW and WW_{B60} due to the soil tillage. Indeed, lowering of soil pH in no-tillage systems has been reported elsewhere [61,62]. Conversely, A_{EXT} did not expose the soil organic matter to mineralisation, as was seen for both WW and WW_{B60}, where the alfalfa was instead terminated by tillage [5,6,34]. Previous studies have suggested that the timing of alfalfa termination is crucial for mineralisation of soil organic matter [63]. In particular, the rates of soil organic matter mineralisation under Mediterranean conditions are expected to be lower in autumn compared to summer, where the soil temperatures are much higher [7]. In the present study, the postponed alfalfa termination (i.e., performed in early autumn rather than in summer) probably slowed the mineralisation of the soil organic matter and reduced the time window between alfalfa termination and the nitrogen uptake by the next crop [6].

It is possible that in long-term alfalfa stands (such as for A_{EXT}, which had lasted for 6 + 2 years), smaller amounts of atmospheric nitrogen are fixed in the soil compared with standard duration alfalfa fields (3–4 years duration). Indeed, other studies on long-term alfalfa stands that were carried out under semiarid conditions showed that with prolonged planting years, alfalfa can consume more of the available soil nutrients than that of its fixed nitrogen [41]. Usually, the mineralised nitrogen after alfalfa termination is used by the next crop (i.e., by wheat for both WW and WW_{B60} in the present study), and for this reason in the total balance of the 2 years of the study period, the soil nitrogen was not significantly different among these three management systems (Table 2).

4.3. Provisioning Services

For both WW and WW_{B60}, lower yields were expected compared to those that can be obtained from fertilised cropping systems under similar climate conditions. Lower yields for A_{EXT} were expected as well, due to the progressive reduction in alfalfa and its contribution to the herbage production after the first mowing. Moreover, these results are supported by the presence of species belonging to *Artemisietea vulgaris* and to the *Molinio–Arrhenathereteа* classes [64], which after the spring cut do not revegetate, as both are perennials (e.g., *P. pratensis*, *P. trivialis*, *L. perenne*) and annuals (*Avena sterilis* subsp. *ludoviciana*, *Anisantha sterilis*). Other studies have shown that durum wheat under similar Mediterranean conditions can reach 5 Mg ha^{-1} of grain yield when fertilised, while alfalfa can reach 13.5 Mg ha^{-1} [34].

The unfavourable weather conditions observed in the first year of the study probably compromised the yield and thus the NRG for both WW and WW_{B60}. Indeed, during the

second year of the study, slightly higher NRG was obtained for WW compared to those for the 2018 season, which were, however, low, although explainable because of the low available soil nitrogen (Table 2). The addition of 60 Mg ha^{-1} of wood gasification biochar (WW_{B60}) neither enhanced nor decreased the NRG compared to WW in either of the study years. Other studies have reported that crop yield stimulation usually derives from soil liming effects, and therefore, as expected, the application of this biochar to a subalkaline soil did not have any significant effects on the increase in crop yield [65].

Among these management systems, A_{EXT} resulted in the highest aboveground biomass, and thus the highest energy output, especially during the second year of the study, when the climate conditions probably stimulated the growth (Table 2, Figure 1). However, it should be noted that the energy balance was likely to have been overestimated for the alfalfa stand studied, taking into account the lower abundance of alfalfa (especially at the first utilisation) compared to a standard alfalfa crop.

4.4. Recommendations for Future Policies

Compared to the WW control, both of these alternative management systems (WW_{B60}, A_{EXT}) were associated with the enhancement of ecosystem service, although some differences emerged, depending on the systems. More specifically, WW_{B60} resulted in the greatest contributions to supporting services, while A_{EXT} generated greater regulating and provisioning services (Figure 4). Based on these results, different recommendations can be suggested to policy makers interested in enhancing ecosystem services provided by forage/cereal-based cropping systems under Mediterranean conditions and under climate change conditions.

While it is true that the biochar applied for WW_{B60} (i.e., wood as feedstock, obtained at high gasification temperatures) significantly increased the soil total organic carbon, future policies should encourage farmers to choose the most appropriate biochar also based on the context [27,32,36]. The biochar that was applied for WW_{B60} was probably an appropriate biochar to enhance carbon sequestration, because it did not result in higher soil GHG emissions (Table 2) and did not decrease the NRG (Table 2). However, future studies are needed to explore the long-term ageing effects of this type of biochar on such ecosystem services and disservices [60].

Policies could support farmers in the use of biochar in cropping systems minimising the costs associated with its application [31]. This support would be very interesting for the development of biochar value chains [27,28] also coming from energy cogeneration plants, where there is the need for suitable disposal options for biochar [29,30,54]. If the policy objective is to go beyond the increase in soil carbon stock and to also mitigate soil GHG emissions while enhancing crop yields, then the use of biochar in combination with a nitrogen fertiliser might be considered [66]. Given the recent climate projections for the Mediterranean areas in terms of precipitation patterns, and consequently their effects on crop yields [22,23], policies should be directed towards increasing grain yields to enhance food security [67]. Depending on the main goal for the use of biochar (e.g., carbon sequestration, soil GHG mitigation, crop yield enhancement), the following must always be considered: type of soil; biochar production feedstock, process, and temperatures (e.g., pyrolysis, gasification); and biochar application rate, ageing, and size (e.g., powder, pellets). This is especially the case for possible tradeoffs with other ecosystem services [26,49]. Moreover, biochar might be produced not only in large-scale systems (e.g., gasification, pyrolysis implants), but also in small-scale systems (e.g., pruning residues as feedstock, kilns as reactors). In this perspective, policies must also consider the overall assessment of the sustainability of biochar as a carbon credit scheme in agriculture through Life Cycle Assessments [27].

In the case of long-term alfalfa stands, Life Cycle Assessments could also allow a better evaluation of its environmental impacts, even if this approach would not take into account other benefits linked to this management systems. Among the local agri-environmental policies that support conservative agricultural management systems, there are no specific

measures for long-term alfalfa stands (Rural Development Programme of Marche Region 2014–2020 [68]). However, it should be noted that the presence of long-term alfalfa in these areas also allows winter grazing by sheep transhumant farms, which are able to provide a wide array of ecosystem services [7,19–21]. Among these services, the improvement of soil fertility due to excrement deposition in night-resting areas and the enhancement of spring hay quality derived from grazing the alfalfa stands during winter could be mentioned. Other services that go well beyond forage production include small ruminant traditional products (e.g., lamb meat, cheese, wool) [21]. Within these provisioning ecosystem services, all of the benefits must be considered that also arise from transhumant farm movements of flocks to the mountain areas during the summer grazing seasons. These benefits also include conservative management of biodiversity-rich permanent grasslands under protection regimes or under severe vegetation dynamics [69–72] and all of the related services, including the value associated with the landscape as shaped by the animals themselves, or as a part of the landscape aesthetic values, for example. Such cultural ecosystem services have not been extensively studied, but are perceived as essential by stakeholders, and therefore future studies are needed to also address these aspects [19].

5. Conclusions

This study analysed the synergies and tradeoffs between ecosystem services generated by two alternative management systems compared to termination of a 6-year-old alfalfa field followed by 2 years of durum wheat. The first alternative management system (i.e., 60 t ha^{-1} recalcitrant biochar added before sowing the wheat; WW_{B60}) showed a doubling of the soil total organic carbon that was maintained during the 2 years of this study. This addition did not result in higher soil GHG emissions nor in negative tradeoffs with other ecosystem services, at least not in the short term. The second alternative management system (i.e., extending the alfalfa duration by 2 years; A_{EXT}) resulted in significantly higher soil cover while not significantly increasing the emissions of soil GHGs.

The results from this study suggest that agri-environmental policies should support the WW_{B60} system if the main objective is to mitigate climate change, because it enables carbon sequestration while maximising the delivery of supporting services. If the main objectives are to enhance regulating services, then extension of the normal duration of alfalfa (i.e., maintained beyond 6 years) would be preferable, as this limits soil erosion while supplying high regulating and provisioning services.

Future studies should not only evaluate the evolution of tradeoffs between ecosystem services in both the medium term and long term, but also include analysis of cultural ecosystem services.

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