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Original
Carbon footprints and social carbon cost assessments in a perennial energy crop system: A comparison of fertilizer management practices in a Mediterranean area / Solinas, S.; Tiloca, M. T.; Deligios, P. A.; Cossu, M.; Ledda, L.. - In: AGRICULTURAL SYSTEMS. - ISSN 0308-521X. - 186:(2021). [10.1016/j.agsy.2020.102989]

Availability:
This version is available at: 11566/286591 since: 2024-12-06T13:17:25Z

Publisher:

Published
DOI:10.1016/j.agsy.2020.102989

(Article begins on next page)

Carbon footprints and social carbon cost assessments in a perennial energy crop system: a comparison of fertilizer management practices in a Mediterranean area

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Abstract

Agriculture is strongly linked to climate change and has a two-sided relationship with climate change. Although climate change contributes to reducing agricultural productivity, the primary sector is responsible for the production of greenhouse gas (GHG) emissions; on the other hand, the primary sector could mitigate emissions to foster soil carbon sequestration. Specifically, perennial energy crop systems could produce relevant environmental and socio-economic benefits. This study aimed to highlight the potential efficacy of various fertilizer management strategies in reducing GHG emissions and increasing the social value obtained from carbon storage. Using two methodological approaches, namely, the carbon footprint (CF) and social carbon cost (SCC) methods, five nitrogen fertilization patterns (low input, LI; high input, HI; LI + biochar, LI + Bi; LI + cover crop, LI + CC; and LI + Bi + CC) were compared in an experiment on cardoon cultivation for three consecutive growing seasons. GHG release exceeded GHG removal and ranged from 0.20 (HI) to 0.14 (LI + CC) t CO_2e per production unit. LI + CC reduced GHG emissions and optimized yield. The rates of carbon sequestration ranged from 72.7 (HI) to 26.2 (LI) t CO_2e t^{-1} of biomass. Furthermore, the combined use of biochar and a cover crop had no positive effects on C sequestration or GHG emission reduction, unlike these treatments individually. In fact, LI + Bi provided the highest value for C storage (61.1 t CO2e t -1 of biomass), and LI + CC had the best GHG balance (0.14 t CO_2e per production unit). The monetary evaluation of C storage showed that HI would produce the greatest benefits until 2050 (i.e., 9K US dollars per t CO_2e). Although a single best option was not identified among the fertilizer management practices, identifying the optimal trade-offs among productivity, GHG emissions reduction and SCC value is important in ensuring that an energy crop will provide food security as well as environmental and socio-economic sustainability. Furthermore, a potential optimal solution could allow improvements in

long-term crop system planning and land use and the development of effective strategies to combat climate change.

Keywords: cardoon, climate change, sustainability, life cycle assessment, carbon storage, nitrogen supply

1. Introduction

Agriculture and climate change are characterized by critical and controversial cause-effect linkages. These linkages may in turn affect the environmental, economic and social spheres and make it difficult to exclude farming from strategies to combat climate change. On the one hand, in 2016, agriculture produced 431 Mt CO_2 equivalents (CO_2e) of greenhouse gas (GHG) emissions in the European Union - 28 (EU-28) + Iceland (ISL). Specifically, methane (CH_4), nitrogen dioxide (N_2O) and carbon dioxide (CO_2) emitted by agriculture corresponded to 47.5%, 72.2%, and 0.3% of the total EU-28 + ISL emissions, respectively (EEA, 2018).

From a diagnostic perspective, life cycle assessment (LCA) may be an appropriate instrument to identify and quantify the GHG emissions and, more generally, the environmental impacts caused by a crop production system (Rebolledo-Leiva et al., 2017; Goglio et al., 2018). Specifically, within the LCA context, the carbon footprint (CF) represents the overall quantity of CO_2 and other GHG emissions related to a certain product produced throughout its life cycle (Baldo et al., 2014; Al-Mansour and Jejcic, 2017). On the other hand, agricultural management practices aimed at enhancing soil carbon stocks might play a key role in mitigating climate change (Söderström et al., 2014). Moreover, soil organic carbon (SOC) sequestration may be considered one of the most cost-effective options for counteracting the effects of climate change (Nayak et al, 2019). In this sense, the social carbon cost (SCC) might be a useful indicator of the potential efficacy of climate change mitigation measures. In principle, it estimates the monetized damage caused by an incremental increase in C emissions in a given year (Greenstone et al., 2013).

Agriculture could adopt a set of GHG mitigation strategies that, although they encompass different contexts (e.g., from the management of croplands and pastures to the restoration of degraded land and organic cultivated soils), are closely related to soil quality (i.e., SOC stocks) (Smith et al., 2008). The uncertainty about the efficacy of different management practices for improving soil carbon may depend on the soil type and climatic conditions (Ingram et al., 2014).

The Mediterranean Basin can be considered one of the most sensitive regions to climate change because of its specific location, namely, a transition zone between the arid climate of North Africa and the temperate and rainy climate of Central Europe (Planton et al., 2016). As highlighted by

Sanz-Cobeña et al. (2017), these varying conditions lead to the existence of two counteracting cropping systems (i.e., irrigated and rainfed) that require the selection and combination of different management practices (e.g., fertilization, soil tillage, use of cover crops, crop residues, and biochar) that might mitigate GHG emissions and, at the same time, enhance SOC content. Furthermore, Mediterranean agricultural areas are characterized by a low SOC level that makes these agroecosystems vulnerable to land degradation and desertification (Aguilera et al., 2013). These risks might be exacerbated by inappropriate land use change or land management (e.g., transformation from a forest or natural grassland to a pasture or cropland), and removing biomass or disturbing soil may lead to soils becoming deficient in carbon and other nutrients (Smith et al., 2016).

Bioenergy crops can contribute to the development of effective measures for climate change mitigation even though environmental and socio-economic sustainability, especially in terms of both land suitability and availability, is a key aspect of producing these crops correctly (Cronin et al., 2020). In 2050, the total land occupied by dedicated energy crops in the EU-28 may reach approximately 13,500 kha, namely, 3.6% of the total available land (1.3% in 2020), at the expense of areas for food and feed crops (90%) as well as forest and natural land (9% and 1%, respectively) (Perpiña Castillo et al., 2016). The use of marginal or abandoned land for bioenergy production is frequently suggested to reduce the controversy about land use change and land competition between food/feed and energy crops, even though this option might have implications for soil carbon and GHG production (Don et al., 2012; Albanito et al., 2016; Mehmood et al., 2017).

Perennial energy crops may be less harmful than annual crops in terms of GHG emissions, especially because of their lower nitrogen (N) requirements; thus, their long-term N management requirements might be less intense than those of annual crops (Drewer et al, 2012). The conversion of an annual cropping system to perennial bioenergy may enhance SOC storage due to the greater capacity of perennial crops to sequester carbon, which is likely due to the deposition and decomposition processes of perennial plant material on the soil surface; in addition, their massive root growth and belowground senescence processes may contribute to the SOC content (Panda, 2016). The increase in soil C under a perennial crop system is characterized by significant variability that is likely due, on the one hand, to complex interactions among climate, soil texture and soil biota and, on the other hand, to the choice of soil management practices, which should reduce the disturbance and destruction of aggregates (Tiemann and Grandy, 2014).

This study aimed to evaluate the potential performance of different N management practices in perennial energy crop cultivation (cardoon) in a Mediterranean area in terms of their ability to reduce GHG emissions and foster SOC storage in the long term. The analysis was implemented by combining two methodological approaches, CF and SCC, to highlight the potential relevance of fertilization patterns to addressing the effects of climate change from both environmental and socio-economic perspectives.

2. Materials and methods

2.1. Study site

The study was carried out in Sardinia (Italy), an island located in the Mediterranean Basin that has a subtropical dry-summer climate, also known as a Mediterranean climate (Belda et al., 2014). This climate was already described by Kottek et al. (2006) as being characterized by a hot-dry summer with an average temperature in the warmest month above 22°C and mild, wet winters. In Sardinia, most of the annual rainfall is concentrated in fall and winter at levels ranging between 500 mm along the southern coast and 1300 mm in the mountainous areas. The mean annual temperature is also affected by the distance from the coastline; the value ranges from 17°C on the southern coast to 12°C inland, and the maximum temperature exceeds 30°C in the summer (Salis et al., 2013).

This region may be considered a suitable territory for residual crop biomass energy exploitation (De Menna et al., 2018) or for energy crop system introduction (Ledda et al., 2013). In fact, the economic crisis for local agricultural and livestock activities on the island is exacerbating the abandonment of productive areas and is leading to the conversion of arable land into grasslands in areas served by irrigation infrastructure (Solinas et al., 2015). In this context, local biomass production or the development of energy crop systems might minimize the risk of land abandonment and provide farmers with new opportunities for additional income.

2.2. Cardoon

Cynara cardunculus L. is one of the most promising crops for use as feedstock for the energy sector (e.g., solid fuel and biodiesel) in addition to being useful for various industrial applications (e.g., cellulose, pulp and paper, phytochemical and pharmacological products) (Gominho et al., 2018). It is a perennial herbaceous species that includes three botanical taxa (i.e., globe artichoke (var. *scolymus* L. Fiori), cultivated cardoon (var. *altilis* DC.) and wild cardoon (var. *sylvestris* Lam. Fiori)) and is native to the Mediterranean Basin (Gatto et al., 2013). Although the three cardoon varieties' performances in terms of biomass and/or energy yield are different, cardoon is adaptable to poor pedo-climatic and input conditions (Ierna et al., 2012; Francaviglia et al., 2016; Neri et al., 2017). The capacity to grow under stressed conditions such as Mediterranean rainfed conditions depends on the drought-escape strategy: the aboveground plant parts dry up over the summer, whereas the underground plant parts survive by becoming quiescent; this strategy has been observed in other vivacious plants (Fernández et al., 2006).

Cardoon cultivation represents an opportunity for the Sardinian region, where the poor competitiveness of some food/feed crops (e.g., cereals) could lead to structural farming shifts towards bioenergy production that might be a valid way to avoid land abandonment. Furthermore, the positive results in terms of biomass, seed, and energy yield provided by field experiments implemented with this species in Sardinia using different crop management practices highlighted that cardoon might be an effective option at the farm level (Deligios et al., 2017).

In Sardinia, the environmental performance of cardoon is better than that of other energy crops, such as giant reed (*Arundo donax* L.), sorghum (*Sorghum vulgare* Pers.) and milk thistle (*Silybum marianum* L. Gaertn.) because of the lack or minimal use of some agricultural practices (e.g., irrigation, tillage); however, N fertilizers are relatively more important for cardoon cultivation than for the other crops (Solinas et al., 2019).

To our knowledge, no monetary estimation related to carbon storage from cardoon cultivation has been performed at the local scale.

2.3. Experimental site

A field trial was conducted on cardoon (*Cynara cardunculus* L. var. *altilis* DC.) cultivation for three consecutive crop years (from 2014-15 to 2016-17) at the “Mauro Deidda” experimental farm of the University of Sassari located in northwest Sardinia (Lat. 41°N, Long. 9°E, 81 m a.s.l.). Cardoon is considered one of the most promising perennial energy crops in the Mediterranean region since its adaptability to water and soil stress conditions prevents these stresses from undermining biomass production (Deligios et al., 2017). Throughout the trial, the average annual precipitation was 363 mm, and the mean maximum and minimum temperatures were 22°C and 12°C, respectively. At the experimental site, the soil is classified as a sandy clay loam, with 66% sand, 19% clay and 15% silt. At the beginning of the experiment, soil samples from a depth of 0-40 cm were collected and analyzed before applying the fertilization treatments. The soil samples had total C, total N and soil organic matter contents equal to 49 g kg $^{-1}$, 1.8 g kg $^{-1}$ and 31 g kg $^{-1}$, respectively.

2.4. Experimental design

Before starting the trial (2014-2015), cardoon was cultivated for seven consecutive years in the same location. To optimize SOC storage, longer field trials may be considered additionally valuable for detecting long-term SOC trends and the effects of crop continuity.

Cardoon removal was necessary since, after several years, the crop showed a physiological decline in production. Therefore, in 2014, the residual biomass from the previous multiyear cultivation period was incorporated into the soil before the new cardoon planting began. This activity, which most likely fostered an increase in SOC potentially available for the next crop, was the starting point for establishing the experimental design and the different N fertilization management treatments.

The trial was arranged in 7.5 m × 6 m plots in a randomized complete block design with four replicates. The different N fertilization options were selected in order to determine the possible N and C supply provided by each management treatment. Specifically, two conventional patterns, namely, local practices based on the use of synthetic fertilizers with high and low N inputs (HI and LI, respectively), were included to guarantee continuity with the previous cardoon cultivation, which used these N management strategies. Three alternative N fertilization practices, biochar (Bi) use, cover crop (CC) cultivation and their combination (CC + Bi), were established to evaluate their potential to reduce synthetic fertilizer use, increase SOC storage, optimize yields, and improve the overall environmental sustainability of perennial energy crop systems. Furthermore, since crop residues (cardoon and cover crops) and weeds were not incorporated throughout the experimental trial, all three alternative treatments were supplemented with the same synthetic N supply used in the LI treatment (i.e., LI + Bi, LI + CC and LI + Bi + CC) (Table 1). The use of biochar and cover crop together with the LI treatment was selected on the basis of the cardoon production level in order to improve its yield. In a previous experiment carried out in the same site of this study, the cardoon fertilized with a lower synthetic N rate, namely 50% less than the conventional one showed a worse crop growth, and thus a lower yield compared to the one achieved using a higher rate of N fertilizer (i.e., the conventional treatment) (Deligios et al., 2017).

Table 1

The use of biochar obtained from the thermochemical conversion of biomass (i.e., pyrolysis) may affect the physical and chemical properties of soil by enhancing its fertility and therefore fostering crop growth (Tan et al., 2017). Since cardoon biomass is grown for energy production, biochar application to soil might offset the amount of carbon removed by biomass harvesting. Specifically, biochar obtained from a slow pyrolysis process using rapeseed straw as the feedstock was applied (10 t ha $^{-1}$) only once at the beginning of the trial (November 2014) and was incorporated into the soil to a depth of 10 cm. In this study, biochar was considered as the amount of C obtained from feedstock pyrolysis (i.e., 71.34 wt %) on the basis of the report of Karaosmanoğlu et al. (2000).

In the same period, a self-reseeding legume cover crop (*Trifolium subterraneum* L. var. *Antas*) was sown (30 kg ha $^{-1}$) in interrow spaces to a depth of 5 cm. A legume was chosen as the cover crop due to its capacity to provide an additional source of N and C through N fixation and residue production, respectively. In fact, cover crop residues were not removed or incorporated into the soil during the study period to facilitate litter development and potentially reduce synthetic fertilizer application. The biochar-cover crop combination was implemented to observe its effect on the SOC content compared to that of the management practices individually and to determine whether this combination showed synergic effects. The potential synergy was assessed considering the SOCS value of each alternative treatment deprived of the SOCS value due to the LI treatment. Practically, the effect separately caused by BI (and CC) was calculated eliding by the LI + BI (LI + CC) value the LI value. Successively, we calculated the effects of the combination of BI and CC eliding the LI value by the LI + BI + CC value. The comparison between the latter value to the sum of the formers allowed to assess the potential synergy (i.e., synergy exists when the combined BI + CC effect is less than the sum of individual BI and CC effects).

2.5. Functional unit, system boundaries and data collection

The multifunctionality of agricultural systems allows the identification of their functional units, namely, the land management, financial and productive functions (Nemecek et al., 2011). In general, the choice of which functional unit to study depends on the objective of the study, the types of environmental impacts evaluated, and the kinds of processes under consideration (Notarnicola et al., 2015). As reported by International Organization for Standardization (ISO) 14040 (2006), the main purpose of a functional unit is to provide a reference to which inputs and outputs are connected. Given these conditions, and considering that the goal of this analysis was to estimate the environmental effects and social cost of different fertilizer management practices in terms of both SOC variation and crop yield optimization, the productive function was considered the most appropriate functional unit for this study. Specifically, the productive function was expressed in tons of biomass ha $^{-1}$ produced by cardoon cultivation throughout the experimental trial.

In this study, a "from cradle to field gate" approach was adopted to emphasize the environmental implications of agricultural practices applied to energy crop systems. Specifically, the system boundary considered in this investigation included, for each fertilizer management treatment, the whole life cycle of cardoon cultivation from the acquisition of raw material inputs to the farm gate (i.e., crop harvesting) (Figure 1). Hence, the LCA neglected product transport operations and stopped at product harvesting; the evaluation did not focus on activities beyond the edge of the field. All farming practices carried out throughout cardoon cultivation were included in an inventory to support subsequent steps (i.e., impact assessment and interpretation). The quantification of inventory, namely, the material and resource flows to and from the environment within the system boundaries, should be methodologically sound, complete and unbiased (Sauer, 2012). Therefore, the inventory of agricultural practices throughout the three years of the trial was based on primary data collected at the experimental site specifically regarding the agricultural machinery, fuel consumption, and types and application rates of synthetic fertilizers, pesticides and organic amendments.

Figure 1

During the cardoon life cycle, direct field measurements (i.e., yield and SOC content), physicochemical analysis of some soil samples, and climatic data detection (e.g., temperature and precipitation) were carried out. These measurements allowed various models (see paragraph 2.5) for assessing the GHG emissions resulting from the different agricultural management practices to be applied.

Since the data were not exhaustive, they were integrated with secondary data (i.e., the upstream and downstream processes of crop cultivation) derived from international databases, primarily the Ecoinvent 3 database. In this study, this database was used in order to include processes regarding technical input production (e.g., fertilizers, pesticides, seeds) and the implementation of mechanical operations such as tillage, sowing, crop maintenance (e.g., fertilization, weeding), and harvesting in the evaluation phase. Specifically, the data for these processes included data regarding the consumption of natural resources, raw material, fuels, and electricity as well as heat production and chemical emissions to the environment.

The crop under consideration, cardoon, was used only for biomass production for energy purposes; therefore, no allocation of impacts was necessary in this evaluation.

2.6. Calculation methodology

Different tools were applied to improve the accuracy of the results of this study since the performance of the tools was mainly based on primary data related to soil physicochemical properties, climatic parameters, crop management, and yield. The use of several models enabled us to better understand the effects of the different fertilization patterns in terms of CO_2e produced or avoided. In this way, we obtained more detailed information on the GHG fluxes in terms of their potential environmental and monetary damages.

2.6.1. Fertilizer and amendment emissions

The main nitrogen emissions caused by each management treatment (i.e., ammonia (NH_3) and nitrous oxide (N_2O) in the air and nitrate in water (NO_3^-) were included in the analysis using the Estimation of Fertilizer Emissions Software (EFE-So) (2015). This software uses the model developed by Brentrup et al. (2000) and allows us to obtain more accurate emission values than other methods since it requires various site-specific data to contextualize the fertilizer application and the possible losses without distinguishing between direct and indirect emissions. This model considers the difference between the supplied N and the absorbed N and requires information about the fertilizer type, soil characteristics, climate context (e.g., air temperature during distribution, summer and winter precipitation) as well as the N content in the harvested crop and its coproducts (Schmidt Rivera et al., 2017).

According to Brentrup et al. (2000), N emissions are affected by different parameters. For instance, the average air temperature, infiltration rate, time between distribution and incorporation, precipitation, radiation, and wind speed are necessary to evaluate NH_3 volatilization from organic fertilizers. In the case of synthetic fertilizers, NH_3 loss mainly depends on the ammonium or urea content of the synthetic fertilizer, the climatic conditions, and the soil properties. The complexity of interactions between soil and climate factors and the variability of crop system management make it difficult to assess N_2O emissions. Nevertheless, the model uses the default value proposed by Houghton et al. (1997) as the emission factor for N_2O. Finally, NO_3^- loss was reported by Brentrup et al. (2000) as nitrate leaching. The rate of NO_3^- loss is strictly dependent on different parameters related to agricultural activity (nitrogen balance) and to soil and climate conditions (field capacity in the effective rooting zone and water drainage rate, respectively). The value for atmospheric N deposition included in the EFE-So model was estimated based on the report of Markaki et al. (2010) regarding annual nitrogen deposition fluxes at different sites in the Mediterranean region, including Sardinia.

To obtain more detailed results, the amount of CO_2 fixed in the industrial urea production process and potentially emitted through fertilizer distribution was considered in this analysis using Eq. (1) (De Klein et al., 2006):

$$CO_2\text{-C Emissions} = M \times EF \tag{1}$$

where CO_2-C emissions is the annual carbon loss from urea application (tons C yr $^{-1}$); M is the annual amount of urea distributed (tons urea yr $^{-1}$); and EF is the emission factor (tons of C (ton of urea) $^{-1}$).

For the LI + Bi treatment, the reduction in N_2O emissions caused by biochar application to soil was computed with Eq. (2) (Wolf et al., 2010):

$$EN = RN\ (2.5\ \text{kg}\ N_2O\ \text{ha}^{-1}\ \text{yr}^{-1})\ Ab \tag{2}$$

where EN is the annual amount of soil N_2O emissions avoided; RN is a reduction factor equal to 25%; and Ab is the area of land amended by biochar. This computation was performed for only the first crop year since soil N_2O fluxes generally show a decrease over time; however, these results are highly variable depending on the complexity of the interactions between the organic amendments and the soil as well as the different experimental setups, soil properties, and conditions (Agegnehu et al., 2016; Borchard et al., 2019).

The addition of carbon to the soil in the form of biochar may be responsible for the so-called priming effect (Zimmerman et al., 2011; Singh and Cowie, 2014), i.e., a short-term change (increasing/positive or decreasing/negative) in the mineralization rate of soil organic matter following the addition of exogenous organic substrates (Kuzyakova et al., 2000). Therefore, biochar application might affect CO_2 dynamics at different time scales. In the short term, its labile carbon fraction may trigger microbial activity that, in turn, increases mineralization (positive priming effect); in the long term, biochar may stimulate physical protection mechanisms (sorption and aggregation) for organic carbon on the amendment surface (negative priming effect) (Maestrini et al., 2015; Sagrillo et al., 2015). Given these considerations, this study included possible changes in soil CO_2 emissions due to biochar addition based on Maestrini et al. (2015), who quantified short-term soil carbon losses (3% of the C from the organic amendment) caused by the biochar priming effect. No specific value was provided for the long term because of the variability of the factors that may influence the priming effect (e.g., repeated biochar addition, seasonal variations in soil temperature and moisture).

Phosphorous losses were not reported for any fertilizer management treatment since they were considered negligible at the study site.

2.6.2. Details about the LI + CC treatment

This study considered the N and C provided by the legume biomass in the LI + CC treatment. Specifically, the N content of the above- and belowground biomass produced by cover crops was calculated based on two specific values (2% and 1.65%, respectively) determined during a field trial carried out in the same geographical area as this study.

The organic matter content provided by the total legume biomass was estimated according to Eq. (3):

$$SOM = DM - A \tag{3}$$

where SOM is the soil organic matter (Mg ha $^{-1}$); DM is the dry matter (Mg ha $^{-1}$); and A is the total ash (as a percentage of DM), which was approximately equal to 12% DM according to Chiofalo et al. (2010); Pace et al. (2011); and Bozhanska et al. (2016).

The SOC value (Mg ha $^{-1}$) was obtained with Eq. (4) (Prybil, 2010):

$$SOC = SOM/2 \tag{4}$$

where 2 is the most widely used conversion factor based on the assumption that soil organic matter contains 50% carbon.

For the LI + Bi + CC treatment, the N and C values were estimated with the same references used for the individual treatments, i.e., LI + Bi and LI + CC.

2.6.3. Pesticide emissions

The on-field emissions from pesticide application were calculated using the PestLCI 2.0 model to assess the pesticide fraction that crosses the technosphere-environment boundary and thus disperses in the environment (air, surface water and ground water). The technosphere can be considered a "field box" that is bounded by the arable field borders, the soil up to 1 m depth and the air column up to 100 m above the soil (Dijkman et al., 2012). The model, according to Birkved and Haushild (2006), considers two emission steps within the technosphere box that are responsible for the fate of pesticides: a primary and a secondary distribution.

The primary distribution refers to the pesticides that are deposited on the crops (e.g., crop leaves) and on the soil surface or are blown away by the wind immediately after pesticide application. The secondary distribution refers mainly to the fate of pesticides on the field; active pesticide ingredients may be deposited on crops, topsoil, or subsoil, where they may undergo different processes. The pesticide fraction that settles on plants might be subject to volatilization, uptake or degradation. On the topsoil, the main processes affecting pesticides are volatilization, biodegradation and surface water runoff due to rainfall; pesticides might also reach the subsoil and thus the ground water through leaching.

This model enables the calculation of emissions due to the primary and secondary distributions by constructing a scenario that includes site-specific information such as the type of pesticide, application method and month, crop, climatic conditions, and soil type. Currently, PestLCI 2.0 is applicable to European conditions; therefore, it includes various site-specific climate and soil data that are representative of European regions and approximately one hundred active ingredients (Moraleda Melero, 2018).

2.6.4. Carbon footprint

The carbon footprint is a methodological tool used to quantify the total amount of GHGs that a product or a service disperses into the environment during its lifetime (i.e., from raw material production to the final use of the product) expressed as CO_2e (Ramachandra and Mahapatra, 2015). In this study, the CF assessment carried out with an LCA approach enabled the quantification of GHG emissions due to the agricultural management practices used in cardoon cultivation throughout the cardoon life cycle.

SimaPro 8.0.4.30 software (Goedkoop et al., 2013a, b) was used to perform the CF analysis based on the impact categories associated with the GHG Protocol. This protocol was developed by the World Resources Institute (WRI) and the World Business Council for Sustainable Development (WBCSD) in 1998 in order to develop accounting and reporting standards for GHG emissions that are specifically designed for different private and public sector activities such as agricultural activities and to reduce the potential negative effects of climate change on natural resources (WRI and WBCSD, 2011a).

The GHG Protocol provides guidance to facilitate the management of agricultural GHG fluxes by considering mechanical (i.e., equipment or machinery operated on farms) and nonmechanical (e.g., soil amendment and management, crop residue burning, and land use change) emission sources as well as upstream sources (e.g., raw material extraction; fertilizer, pesticide and feed production) in order to foster eco-friendly production practices (Russell, 2011). The GHG Protocol uses the Intergovernmental Panel on Climate Change (IPCC) calculation approach to quantify the GHG fluxes of a given activity (WRI and WBCSD, 2011b). The GHG emissions related to the life cycle of a product may be expressed as CO_2e using a characterization factor, the global warming potential (GWP), developed by the IPCC within the climate change impact category (JRC, 2007). The GWP enables us to compare the potential climate impacts of various gases using the GWP value of CO_2 as a reference unit; the GWP of CO_2 is equal to 1 and can be considered at three different time horizons, namely, 20, 50 and 500 years (WRI and WBCSD, 2011a). In this study, the CO_2e, that is, the CF of a certain process, was calculated with Eq. (5) (Morawicki and Hager, 2014):

$$\text{GHG emissions in } CO_2e_{(i)} = \text{emission factor} \times \text{activity rate} \times GWP_{(i)} \quad (5)$$

where CO_2e is the CF from a certain gas (kg CO_2e); the emission factor (i) is the amount of GHG produced per unit of activity rate; the activity rate is the level of a specific practice (e.g., liter of diesel consumed during fertilizer distribution); and $GWP_{(i)}$ is the characterization factor expressed in kg CO_2e/kg GHG.

The GHG Protocol method uses 100 years as the time horizon to calculate GHG emission impacts related to a product system. This method uses the impact categories carbon emissions from fossil sources (CEFS), biogenic carbon emissions (BCE), carbon emissions from land transformation (CELT), and carbon uptake (CU) (PRé, 2018).

The CEFS category refers to emissions arising from fossil sources (e.g., carbon from fossil fuels), and BCE is related to biogenic sources (i.e., carbon from living organisms or materials derived from biological matter). CELT refers to emissions from the conversion of one land use category to another. The last category, CU, refers to the CO_2 stored in plants and trees as they grow (WRI and WBCSD, 2011b). Since the analysis in this study concerns a perennial crop, all estimated impact categories were expressed in annual CO_2e, that is, the CF values of each impact category for cardoon were calculated considering their lifetime average impacts. Finally, the values of the impact categories provided by SimaPro are expressed on a land basis in kg CO_2e ha^{-1}, but this study adopted a production functional unit (i.e., tons of biomass produced by cardoon). Therefore, the outputs were converted with Eq. (6) (Cheng et al., 2015):

$$CFY = CFA/Y \quad (6)$$

where CFY is the carbon footprint of a generic impact category per production unit (t CO_2e/t of biomass produced); CFA is the value of one impact category on a land basis (t CO_2e/ha); and Y is the yield of a given crop (t/ha).

The results of this conversion enabled the calculation of the CF balance between GHG emissions and sequestration (i.e., the CEFS, BCE, CELT, and CU impact categories, respectively) to identify the fertilizer treatments with the best and the worst environmental performance in cardoon cultivation throughout the experimental trial.

2.6.5. Carbon footprint uncertainty analysis

A Monte Carlo analysis was performed to assess the uncertainty of the CF findings. The analysis was also performed to test for possible significant differences in the environmental impacts of each fertilizer treatment in terms of their CF per product unit. SimaPro 8.0.4.30 software was used to run the Monte Carlo simulation (Goedkoop et al., 2013a, b) at a 95% confidence interval with 1000 reiterations.

2.6.6. Soil carbon storage

Due to the complexity of the C dynamics and GHG fluxes due to the different N fertilizers, an additional impact category, soil organic carbon storage (SOCS), was considered to provide a more detailed framework for GHG exchanges related to the perennial energy crop system. The results might be useful for facilitating the identification of environmental impacts in the long term and supporting crop system and land use planning.

Accounting for soil C changes due to agricultural systems and land use is difficult in the context of LCA and, consequently, in the context of product CFs. The difficulty arises mainly because of the lack of a specific procedure for soil C; however, attempts to consider SOC dynamics may be implemented depending on the availability of quality data and the performance of C cycle models (Goglio et al., 2015).

In this study, carbon storage was estimated using the Rothamsted carbon model (RothC) ver. 26.3. This model was specifically developed to estimate the turnover of SOC in nonwaterlogged topsoil and includes the effects of soil type, climate conditions and plant cover on the turnover process (Coleman and Jenkinson, 2014). Its performance is strongly dependent on site-specific data since it requires three different types of information: i) climatic data, i.e., monthly air temperature (°C), rainfall (mm), and evapotranspiration (mm) values; ii) soil data, including clay content (%), inert organic carbon (IOM), initial SOC stock (t C ha $^{-1}$), and depth of the considered soil layer (cm); and iii) land management data, such as soil cover and monthly quantity of plant residues (t C ha $^{-1}$) (Barančíková et al., 2010). RothC was used to estimate the SOC for each agricultural treatment adopted for cardoon cultivation based on site-specific soil and climatic conditions and with a time reference of 100 years, i.e., the same time horizon used by SimaPro to assess the CEFS, BCE, CELT, and CU impact categories.

All inputs were included in RothC as the average values for the experimental trial period. In the model, SOC is divided into four active pools and a small amount of IOM that is resistant to the decomposition process. Crop C inputs to soil are allocated into the categories decomposable and resistant plant material (i.e., DPM and RPM, respectively), microbial biomass (BIO), and humified organic matter (HUM) (Li et al., 2016). RothC allows the C input to be partitioned between DPM and RPM on the basis of its provenance, namely, crops, grassland or forests. These two pools undergo decomposition, resulting in CO_2, BIO or HUM depending on the soil clay content. The decomposition process for one active compartment occurs through first-order decay at a specific rate (year $^{-1}$) for DPM, RPM, BIO, and HUM (10, 0.3, 0.66, and 0.02, respectively) (Zimmermann et al., 2007).

The process is depicted in Eq. (7) (Gónzalez-Molina et al., 2017):

$$Y = Y_0 (1 - e^{-abckt}) \quad (7)$$

where Y is the material quantity of a pool that decomposes in a certain month (t C ha $^{-1}$); Y_0 is the initial C input (t C ha $^{-1}$); k is the decomposition rate specific to each compartment; a, b and c are factors that modify k related to temperature, moisture, and soil cover, respectively; and t is 1/12, to express k as the monthly decomposition rate. The IOM was calculated with Eq. (8) (Falloon et al., 1998):

$$IOM = 0.049 \times SOC \times 1.139 \quad (8)$$

where IOM and SOC are both expressed in t C ha $^{-1}$. Furthermore, RothC was performed at equilibrium, namely, the C input was adjusted such that the modeled SOC value matched the measured starting value in the experimental trial (Kaonga and Coleman, 2008). The SOC stock used in the RothC model was calculated according to Eq. (9) (Lozano-García et al., 2017):

$$SOC\text{-}S = SOC\ concentration \times BD \times d \times (1 - \delta_2\ mm) \times 10^{-1} \quad (9)$$

where -SOC-S is the soil organic carbon stock (mg ha $^{-1}$); SOC is the soil organic carbon (g kg $^{-1}$); BD is the bulk density (mg m $^{-3}$); d is the soil thickness (cm); and δ_2 mm is the fractional percentage (%) of gravel greater than 2 mm in size.

Finally, the SOC values provided by the RothC simulation for the time horizon of 100 years for each fertilization treatment used in cardoon cultivation throughout the experimental trial were converted to CO_2. This conversion was performed with Eq. (10) (Alani et al., 2017):

$$1\ \text{ton of soil C} = 3.67 \times \text{tons of } CO_2 \quad (10)$$

where the tons of CO_2 are the quantity of CO_2 emitted or stored depending on the ratio of the molecular weights of C (12) and CO_2 (44), namely, 44/12 = 3.67.

The values of CO_2 obtained were expressed in CO_2e based on the GWP of CO_2 for 100 years, i.e., 1 (Forster et al., 2007). These outputs are the CF of the SOCS impact category for each cardoon management treatment. As for the previous impact categories, these outputs were also converted to production functional units to facilitate comparisons of the different fertilization treatments in terms of their potential C storage.

2.6.7. Social Carbon Cost

The social carbon cost represents the cost of an additional ton of CO_2 emissions or its equivalent; in more detail, it describes the change in the discounted value of economic welfare resulting from an additional unit of CO_2e (Nordhaus, 2017). The monetized estimation of the potential damage caused by an increase in GHG emissions in a given year is performed in order to better understand the changes in agricultural production, human health, and the value of ecosystem services that arise due to climate change (IWG, 2016). In contrast, it may also be considered a measure of avoided damage in the case of emission reductions, which provide a socio-economic benefit.

In this study, the SCC was calculated based on an assessment of benefits and cost, that is, of the increases and decreases in human well-being due to GHG emissions, by linking the global carbon cycle and temperature variations to a global economic context (van den Bijgaart et al., 2016). SCC evaluations for different time horizons are performed with three integrated assessment models. These models run with several input assumptions and simulate the possible connections between GHG emissions and climate change compared to a baseline scenario as well as different options for assessing the future damages that may arise from an additional released or avoided ton of CO_2 emissions (Rose et al., 2014).

Each model runs 10K times, which provides thousands of results that are discounted and averaged to obtain an equivalent single number, called the present value. Specifically, the present value is computed for a number of years (x) in the future, and the previous values are reduced by a certain percentage (i.e., the discount rate) for each of the x years at three reference rates, namely, 2.5%, 3.0% and 5.0% (Niemi, 2018).

With the above methods, in this study, monetized estimations of the SOCS ecosystem service were performed as an attempt to underscore the long-term strengths and weaknesses of the different fertilization practices used in cardoon cultivation as strategies for addressing the challenges of climate change. The SCC was calculated by multiplying the SOCS values of each fertilizer treatment in 2050 obtained from the RothC model by the SCC in 2050, namely, 79 US dollars (2016 dollars per metric ton CO_2e), with the 3% discount rate (Niemi, 2018). To perform this calculation, the SOCS values were converted to tons CO_2e for a 100-year time horizon as described at the end of subparagraph 2.6.6.

3. Results

3.1. Carbon footprint of GHG fluxes from fertilizer management

The descriptions of the CF outputs are focused on the effects (t CO_2e t^{-1} of cardoon biomass) resulting from the specific characteristics of each fertilizer management treatment, i.e., the different N doses in HI and LI, biochar application, legume cover crop cultivation and their combination. These effects were the focus because the mechanical operations and production inputs did not change among treatments except in a few cases reported occasionally. The environmental impacts of these factors were not considered because the CF values did not differ among treatments when expressed on a land basis and because we wanted to remain consistent with the objective of this study, that is, to evaluate the potential reductions in GHG emissions and SOC storage resulting from different N fertilizer management strategies applied to cardoon.

The environmental performance of the five treatments showed significant variability in both inter- and intra-impact categories (Figure 2). In fact, in the former, CF ranged from 0.00041 to 0.2 t CO_2e per production unit in CELT (LI) and CEFS (HI), respectively. The difference detected between HI and LI - CEFS exceeded CELT slightly more than 480 times - is particularly interesting considering the CEFS value of all fertilization patterns taken together. In fact, the CF of the CEFS category was 432, 40, and 14 times greater than those of CELT, CU, and BCE, respectively. Regarding CU, all further values reported should be considered reliable in absolute terms since this impact category is related to GHG savings, whereas the other categories are related to GHG losses.

Figure 2

Considering the effect of each treatment in the single-impact category, HI demonstrated the highest environmental performance in CEFS exceeding the second worst management (LI) by 21%. The observed gap between HI and LI was mainly due to the different impacts of agricultural inputs, especially fertilizer inputs. In fact, the mechanical operations were the same except in the LI + Bi, LI + CC, and LI + Bi + CC treatments, in which two additional agricultural inputs were introduced, namely, biochar and legumes that were sown or distributed and subsequently buried. Furthermore, the higher amount of N fertilizer (i.e., urea as a topdressing) used in HI was mainly responsible for the poor environmental performance of this treatment in the CEFS category; HI had twice the impact of the second most impactful treatment (LI). HI was 20% and 10% more impactful than LI + Bi and LI + CC, respectively; however, the last two categories included two additional mechanical operations and two additional production inputs, namely, biochar and its distribution and burial (LI + Bi) and legume seeds and their sowing and burial (LI + CC).

These additional processes made contributions that were not significant in the CEFS category, since they were equal to less than 1% and slightly more than 3% for LI + Bi and LI + CC, respectively. LI + Bi showed better environmental performance than the LI treatment most likely due to the short-term effect of biochar on reducing N emissions from fertilizers, i.e., urea and diammonium phosphate, throughout the first growing season. In fact, the environmental impact of these fertilizers when used with biochar was 22% lower than the impact from the same fertilizers in the LI treatment.

LI + CC showed better environmental performance than LI due to the high average production of cardoon biomass (8.14 and 6.91 t DM ha $^{-1}$ for LI + CC and LI, respectively) that de facto reduced the CEFS value on a production basis rather than to the N and C provided by legume cultivation (slightly more than 3% of the CEFS category). The CF difference between Li + CC and Li + Bi (i.e., 0.01 t CO_2e t $^{-1}$ more cardoon biomass under Li + Bi) was most likely due to the effect of biochar on GHG emissions from fertilizers since the mechanical operations (i.e., biochar distribution and burial and legume sowing and burial) had the same environmental impact (0.0007 t CO_2e t $^{-1}$ of cardoon biomass).

Finally, the LI + Bi + CC treatment demonstrated an antagonistic effect between biochar and the cover crop that generated an environmental impact 13% lower than the sum of their individual effects. Nevertheless, the CF contribution per production unit of LI + Bi + CC was greater than those of LI + CC and LI + Bi (by 6% and 15%, respectively) because of the higher biomass yield from LI + CC and LI + Bi than from LI + Bi + CC.

The CELT category showed the lowest CF contribution of the four impact categories, most likely due to the lack of actual land use change, which de facto avoided the production of GHG emissions in this category. Nevertheless, impacts detected within the CELT category can be associated with CO_2 and N_2O emissions generated during agricultural land use and following a change in farm management practices according to the GHG Protocol, which emphasizes the roles of agricultural activity as sources of and a sink for CO_2 (WRI and WBCSD, 2011b).

The analysis showed similar CF values on a land basis among treatments that had the same upstream processes as key impact factors, such as seed production that includes a land transformation. The differences in CF per production unit were minimal (i.e., from 0.00035 to

0.00041 t CO_2e t^{-1} of biomass for LI + CC and LI, respectively) and resulted from the different cardoon yields. LI had the lowest cardoon yield and thus was the least environmentally friendly treatment. In contrast, LI + CC produced 18% more cardoon biomass than LI and reduced GHG emissions by 85% compared to those under conventional management. Furthermore, the combination of biochar and the legume cover crop showed, as detected in the CEFS category, an antagonistic effect even though the environmental performance of LI + Bi + CC was worse than those of LI + Bi and LI + CC (by 8% and 10%, respectively). The LI + Bi and HI treatments had a very similar CF per production unit (approximately 0.0003 t CO_2e t^{-1} biomass), and their CF values were higher than that of LI + CC (by 2% and 3%, respectively). This result highlights that the potential effect of the cover crop on increasing cardoon yield was most likely responsible for the low CF in the CELT category.

The last two impact categories, BCE and CU, which are more specifically related to C dynamics, showed intermediate values between those of CEFS and CELT. LI + Bi + CC was the worst and the best treatment for BCE and CU, respectively (0.03 and 0.01 t CO_2e t^{-1} of biomass). This result suggests that organic material used in addition to synthetic fertilizers might act as both a source and sink of C. The environmental performance of these alternative fertilization treatments might depend on how the additional inputs were included in the overall crop management. Specifically, the sum of the CFs resulting from LI + Bi + CC and LI + Bi represented 92% of the BCE category on the whole, underlining the relevance of biochar as a C source. In fact, the C contribution provided by biochar application exceeded 90% in both treatments. Although the cover crops were not harvested, the C supply from the legumes was not relevant (7%) to the BCE. The difference in CF between LI + Bi + CC and LI + Bi (i.e., 0.002 t CO_2e t^{-1} more biomass in LI + Bi + CC) was due to the simultaneous use of biochar and the legume cover crop. Their combination had a synergistic effect that increased the CF compared to those resulting from the biochar and legume crop individually. This is because the CF of LI + Bi + CC exceeded by 9% the sum of the CFs of the individual practices. In other words, in the LI + Bi + CC treatment, biochar and the legume crop might have acted to strengthen the effect of one or both of these practices. The environmental performance of LI + CC was 17 times lower than that of the worst treatment, further highlighting the relevance of biochar in the BCE category. The two conventional management treatments, namely, LI and HI, made the best contribution in terms of avoided CO_2 emissions (6%) compared to those from the treatment with the greatest impact because of the absence of the additional organic C source.

Among the four impact categories, CU is the most related to GHG emission removal since it concerns the C stored in a crop throughout its life cycle. As mentioned above, the most environmentally friendly treatment within the CU category was the worst treatment for BCE. LI + Bi + CC showed conflicting performance results due to the combination of biochar and legume cover crops. This treatment had the highest CF value, which might be due to the synergistic effect that was also observed in the CU category and was caused by the interaction between biochar and the legume cover crop. Their simultaneous action, which resulted in a CF value 16% higher than the sum of the CFs of the individual treatments, might have resulted in greater C storage in the biomass than that in the LI + Bi and LI + CC treatments.

Furthermore, LI + Bi + CC had a higher CF value than LI + CC and LI + Bi (by 13% and 170%, respectively), suggesting that the positive environmental performance in LI + Bi +CC might be due to the synergistic effect of biochar and the legume enhancing C uptake from cardoon and the legume cover crop. In contrast, the lowest CF occurring in LI + Bi underlines that the potential effect of biochar on the ability of cardoon to store carbon might not have been adequate to guarantee good performance.

In addition to crop yield, some agricultural inputs had various impacts on the CU category based on the management treatment. For instance, the cardoon seeds for sowing contributed approximately 10% on average to the LI + Bi, LI + CC, and LI + Bi + CC treatments. The synthetic fertilizers used in LI + Bi had an effect equal to 13% on CU, whereas the C from the legume cover crop contributed 30% to LI + CC. The same inputs made contributions of 5% and 29%, respectively, in LI + Bi + CC. The environmental performance of LI in terms of CO_2 uptake was 8% higher than that of LI + Bi, most likely since the yield of LI was greater than that of LI + Bi. The quantity of cardoon biomass might also have played a role in the CF values of the HI and LI treatments. In fact, LI, which had lower average biomass production than HI, had the best environmental performance in the CU category, with a contribution that was slightly more than 7% higher than that of HI. Due to the use of double the N dose (HI vs LI), the N fertilizer effect on the CU was almost 2 times greater in the HI treatment.

A more in-depth analysis of the individual CF balances for each agricultural treatment (i.e., the comparison of GHG release and GHG removal) allowed us to better understand the effects of fertilizer patterns on GHG fluxes (Figure 3). All CF balances showed GHG emission losses, ranging from 0.20 (HI) to 0.14 (LI + CC) t CO_2e per production unit. The balances for LI + Bi, LI and LI + Bi + CC were 81%, 82%, and 90%, respectively, of the highest balance. The inclusion of a cover crop (i.e., a legume) in a perennial energy system (cardoon) might be optimal for GHG emission reduction and yield optimization.

Figure 3

The second positive trade-off between the GHG balance and crop production was shown in LI + Bi. Although this treatment showed the same GHG balance as that of LI (0.16 CO_2e t $^{-1}$ of biomass), the cardoon yield achieved with biochar application was greater than the LI yield (7.96 vs 6.91 t ha $^{-1}$ on average). In contrast, the balance of LI + Bi + CC was the second highest, suggesting that the combination of biochar and the cover crop did not result in a reduction in GHG emissions, although the cardoon yield achieved with LI + Bi + CC was intermediate to the biomass production levels of LI + Bi and LI + CC.

3.2. Uncertainty analysis results

A Monte Carlo analysis was performed to evaluate the uncertainty of the LCA results by pairwise comparisons among the fertilizer management strategies in terms of their CF per production unit. The analysis showed (Table 2) that in CEFS, three differences were not statistically significant at $\alpha = 0.05$.

Table 2

Specifically, the analysis highlighted that the CEFS CF of HI, namely, the treatment with the highest impact, was significantly higher than those of the other treatments. Regarding the most eco-friendly treatment (i.e., LI + Bi), only its difference from LI was statistically significant. LI showed the worst result (i.e., the highest value) in CELT even though its performance was highly significantly different only from those of HI and LI + Bi + CC. In the BCE category, all the comparisons demonstrated significant differences except for HI vs LI + CC. Finally, in CU, the most impactful treatment, LI + Bi + CC, was significantly different from the second most impactful treatment (i.e., LI + CC) only at $\alpha = 0.10$, whereas it was highly significantly different from the other three treatments.

3.3. Soil organic carbon stocks under fertilizer management

The analysis was completed by considering the SOCS category in order to detect changes in SOC storage resulting from the implementation of the five fertilization patterns. Although the SOCS category was expressed in t CO_2e t $^{-1}$ cardoon biomass, as were the previous four categories, its environmental impact was calculated from direct measurements taken in the field throughout the experimental trial (Figure 4).

SOCS ranged from 72.7 (HI) to 26.2 (LI) t CO_2e per production unit, highlighting that the two conventional management strategies showed the best and the worst performance; the difference was equal to slightly less than 3 times in favor of HI management. The performance of HI might be due to the higher N dose applied throughout the cardoon life cycle which, in turn, most likely fostered a higher yield than that under LI. The three alternative treatments showed values (53.1, 53.9 and 61.1 t CO_2e t^{-1} of biomass for LI + Bi + CC, LI + CC and LI + Bi, respectively) that were closer to that of the best (i.e., the highest value) treatment than to that of the worst (i.e., the lowest value) treatment, highlighting that the treatments that included biochar, the cover crop or their combination fostered SOCS. The simultaneous use of biochar and the legume demonstrated an antagonistic effect on SOCS; the sum of the effects of biochar and the cover crop individually was 2 times higher than the value obtained from their combination. The environmental performance of LI + Bi was better than those of LI + CC and LI + Bi + CC (by 13% and 15%, respectively), highlighting that the application of biochar might have had a stronger effect than the other two fertilizer management strategies in terms of soil carbon storage.

Figure 4

3.4. Social carbon costs from fertilizer management

A monetary valuation was performed to estimate which fertilizer treatment might generate the greatest flow of benefits related to the SOCS ecosystem service. The results highlighted that HI might produce the greatest benefits until 2050 (Table 3). Specifically, these benefits could amount to approximately 9K US dollars per t CO_2e. In contrast, the lower benefits arising from the other treatments suggests the presence of a social cost (an opportunity cost in terms of lost benefits compared with those in the most favorable treatment). The LI treatment had the highest SCC, equal to approximately 5K US dollars per 1t CO_2e, whereas the other three treatments showed SCC values ranging from 1K (LI + Bi) to 2K (LI + Bi + CC) US dollars per 1t CO_2e.

Table 3

4. Discussion

4.1. Carbon footprint implications of agricultural management

The results highlight that the characterization of a perennial energy crop system in terms of agricultural management and land allocation should be used to better support farmers' decisions as well as to reduce GHG emissions and to increase soil C storage in the long term. Specifically, the choice of farming practices and land use might arise from a convenient trade-off between the yield and environmental performance of energy crops, for example, to satisfy present and future needs in terms of food and energy security as well as environmental sustainability. This study might provide useful support for selecting the best option since the results enabled us to highlight the strengths and weaknesses of each fertilization pattern and its effects on GHG dynamics (Figures 2-4).

The use of the three alternative treatments (i.e., LI + Bi, LI + CC and LI + Bi + CC), but their effects must be interpreted with caution since their potential benefits for GHG dynamics and SOCS might be affected by site-specific characteristics such as climate, soil type, and farming practices (Figures 3 and 4). Scientific studies regarding the effects of legume cover crops on GHG flux show highly variable results that are strongly connected to the experimental context. Therefore, it is difficult to associate our findings with a specific point of view. The LI + CC treatment confirmed the potential of legume cover crops to offset the cardoon N requirement, reducing GHG release and guaranteeing the highest cardoon yield (Figure 3). This result was consistent with evidence from Daryanto et al. (2018), who highlighted that the synchronization of nutrient availability from cover crops and nutrient requirements from the main crop is strategically necessary to ensure high productivity due to optimized microbial activity. On the other hand, legume cultivation was able to foster high SOC storage even though its contribution was not as high as that of HI, likely because of the mineralization of the additional biomass produced by the cover crop (Figure 4).

Regarding the LI + Bi treatment, its positive effects in terms of C storage might be due to the recalcitrant C in biochar. This C interferes with the C and N dynamics in the microbial community and may facilitate the maintenance of a stable C pool in the soil (Figure 4). These conditions might also have contributed to the high yield level - just below those of HI and LI + CC - and the reduction in GHG loss (Figures 2 and 3). On the other hand, the reliability of the results of previous studies is low due to the reference context, and this is particularly true for the Li + Bi treatment.

The potential effect of biochar on soil CO_2 emissions is still complicated and poorly understood because of the considerable uncertainties in both time (in the short or long term) and space (at the laboratory or field scale) (Fidel et al., 2018). In fact, CO_2 emissions showed different behaviors (increasing, decreasing or unchanged dynamics) as a result of organic amendment addition, mainly due to the complicated interactions between the biochar feedstock and its physicochemical properties; application rate and mode (i.e., alone or combined with synthetic or organic fertilizers); soil type, nutrient availability, and microbial activity; and crop management practices (e.g., incorporation of residual biomass, rate and time of synthetic fertilizer application) (Kuppusamy et al., 2016; Shen et al., 2017). These complex interactions also have variable effects on the emissions of other GHGs from soil, such as N_2O. In this context, the performance of LI + Bi + CC is even more difficult to interpret since it is most likely affected by the interaction between biochar and the legume cover crop, which is difficult to specify. Therefore, an attempt was made to analyze the results into each impact category to identify synergistic effects.

Conventional management, namely, HI and LI, provided two completely different opportunities for trade-offs, most likely due to the different N doses (in HI, it was twice LI). However, the performances of the treatments in this study might be associated with the ability of cardoon to adapt to the Mediterranean climate and to take up nutrients from deep soil layers with its well-developed root system, which increases soil organic matter and nutrient availability in the long term (Mauromicale et al., 2014). The use of a high synthetic N rate for a perennial energy crop might produce the highest yields (HI production was approximately one ton more than LI production) if the energy crop system is intended to use arable land that might be abandoned due to the lack of a useful production purpose. On the other hand, the results of LI might represent a good trade-off for the use of lands that are unsuitable for food production where perennial biomass production that is occasionally harvested for energy production purposes might foster the restoration of vegetation and thus C storage in the long term. The introduction of a perennial energy crop in farming planning might prove to be more advantageous than the introduction of an annual energy crop regardless of which management practices were applied. In fact, perennial crops are generally characterized by lower input costs (e.g., tillage is carried out only once), and their long-lived roots can develop positive relationships with root symbionts that foster nutrient availability and consequently reduce fertilizer use (López-Bellido et al., 2014).

The potential trade-offs in conventional practices (i.e., HI and LI) might be achieved through the adoption of innovative technologies. For instance, the application of precision agricultural practices can foster reductions in GHG emissions and increases in SOC storage since these practices may lower the intensity of tillage practices, the required N supply and other production inputs, and the consumption of fuel for mechanical operations. Specifically, these innovative practices can optimize a small amount of production inputs such as N fertilizers that, if used excessively or in a large agricultural area, can have relevant negative impacts in terms of environmental and economic sustainability (e.g., low profit margins on a land basis).

In our opinion, precision techniques may be considered a useful support for more efficient resource use (e.g., nutrient use) from a circular economy approach. In this paradigm, bioenergy production could offer a viable contribution for addressing challenges related to environmental concerns and resource scarcity (Pan et al., 2015). Although biomass plays an important role in the circular economy context as a feedstock alternative to nonrenewable energy sources, achieving high biomass crop yields involves energy and material costs due to, for instance, fertilizer use and production (Sherwood, 2020). The use of byproducts (e.g., biochar) would close the loop in agriculture, minimizing fertilizer nutrient dissipation in the environment and regenerating natural resources (Chojnacka et al., 2020). In this sense, biochar may be considered a promising option that is well suited to circular economy principles, even though its capacity to foster carbon sequestration, improve soil quality and support plant growth is strongly affected by its physicochemical characteristics and the production technology used (Bis et al., 2018; Olfield et al., 2018).

In summary, using synergies to close the natural resource cycle by developing integrated farming systems (e.g., the use of byproducts from one production process in another process) might increase the adoption of organic fertilizers and diversify production in addition to decreasing production costs and environmental impacts.

However, the exploitation of natural resources (e.g., water) and the application of N fertilizers that are prone to leaching may foster or exacerbate possible pollution phenomena, particularly in vulnerable agricultural areas devoted to profitable crop cultivation. As reported by Balafoutis et al. (2017), the application of precision agriculture practices (e.g., technologies that allow variable rate application of nutrients, irrigation, pesticides and planting/seeding; controlled traffic farming and machine guidance) with technological equipment may spatially and temporally optimize the use of inputs based on site-specific characteristics. These practices could cause a reduction in GHG emissions and an improvement in farm economic and production performance compared to those under conventional management.

In summarizing and considering all fertilization patterns, a clear best option did not emerge. LI + CC maximized cardoon productivity and minimized GHG emissions, but HI maximized C storage in the long term (Figures 3 and 4).

The availability of site-specific data and specific information on crop system planning and land use are key factors in using mixed methodological approaches to identify which fertilizer management strategies optimize the performance of cardoon in terms of productivity, GHG reduction and C sequestration.

Although more research needs to be done to improve the reliability of the results, the framework adopted in this study may be replicated to assess the potential of other perennial energy crop systems and innovative agricultural management practices to achieve the most favorable trade-off between production level and environmental sustainability.

4.2. LCA benefits in agricultural management

The application of different assessment tools (e.g., simulation models for fertilizer and pesticide emissions and for carbon stocks) based on site-specific data (e.g., pedo-climatic conditions and GHG production) collected throughout the experimental trial can be considered an attempt to mitigate the main weakness of LCA. As noted by Curran et al. (2013), this methodological approach is not free of limitations that might affect the accuracy of the results despite the general framework developed by ISO for implementing LCA. These limitations are mainly due to the lack of a well-defined procedure for encompassing and estimating important site-specific factors (e.g., soil quality, soil carbon sequestration, and gaseous N losses) that are closely linked to both farm management and the environmental performance of a crop system within the LCA context (Garrigues et al., 2012; Petersen et al., 2013). Although models, unlike direct observations, do not guarantee a high level of certainty, they are generally able to capture variability as well as soil and climatic interactions (Goglio et al., 2015). In this study, both models and field data were used to improve the reliability of the LCA.

On the other hand, the effect of crop residues was not included in this analysis because of the lack of information, although it is known the influence of crop residues on soil N dynamics and N_2O emissions. Specifically, the agricultural use of crop residues can contribute to the maintenance of soil functions acting as source of organic matter and nutrients and thus able to improve crop production level (Lehtinen et al., 2014). Furthermore, the plant residue C/N ratio may influence the decomposition of residue and thus the soil N_2O fluxes (Pimentel et al., 2015). Although the use of crop residues with a high C/N ratio may encourage the N utilization by microbes leading to a reduction in N_2O emissions, the effects of crop residues with different C/N ratios on N_2O emissions might also depend on soil - climatic conditions, biochemical composition of plant residues, and agricultural management as a whole (Shan and Yan, 2013; Wu et al., 2016; Zhou et al., 2020).

Agricultural systems are closely related to various parameters (e.g., cropping intensity, input prices, climate and soil condition) whose high variability and addition to regional specificities make the data quality a key factor in application of LCA to agricultural products (Weidema and Meeusen, 2000). The fate of the emitted pollutants released by a product throughout its life cycle may be may affected by different locations where pollution occur. This spatial variability is traditionally disregarded in life cycle impact assessment (LCIA) although the impact highlights by LCIA may be considerably different from the actual one (Hauschild et al., 2006). On the other hand, the development of region-specific inventories and characterization factors might be relevant to improve the accuracy of LCA analysis (Yang et al., 2018; Patouillard et al., 2019). Regionalized LCIA still remains a challenge since on the one hand, regionalized LCIA characterization factors in combination with site-specific inventories might reduce the uncertainty of results. On the other hand, a proper development of the regionalized LCA might be limited by the lack of standardization in regionalized LCIA data formats, poor site-dependent inventory data availability, and a lack of widespread software support (Mutel et al., 2019).

In view of above, an additional limitation of the methodological approach adopted in this study concerns the sensitivity of the LCA tool in dealing with regional - based data.

Our study emphasized that the dual role played by farming, i.e., its vulnerability to climate change and its simultaneous contribution to the impacts of climate change, makes it difficult to identify the optimal management practices that would guarantee maximized food production, energy production, and environmental security. Since it is virtually unthinkable to develop a set of measures that are valid worldwide, an assessment of farming practices is necessary for each cropping system on the basis of site-specific characteristics (e.g., climatic and edaphic conditions, social context and historical land use and management) (Smith, 2012).

Our approach confirms this need, and the results suggest that the optimization of agricultural practices, such as fertilization, may have a positive effect on GHG fluxes in the long term. Furthermore, the management of a perennial energy crop is generally not devoid of environmental impacts, and the extent of these impacts often depends on fertilizer use (Wagner and Lewandowski, 2017; Fernando et al., 2018). This was consistent with our findings, which identified the field emissions resulting from fertilizer application as one of the main factors responsible for the environmental performance of cardoon cultivation. A similar result was detected by Razza et al. (2017) for cardoon cultivation in Sardinia, although they considered a single value for GWP without distinguishing among impact categories.

4.3. Socio-economic effectiveness of agricultural management

The SCC is an economic measure related to negative externalities from a climate change perspective (Anthoff and Tol, 2013). In this study, the ecosystem service corresponding to SOC storage provided by agricultural activity may be considered a positive externality. The cost of this service represents the monetary benefit reduction from changing from HI management, i.e., the practice that contributes the most to C accumulation in the soil, to the other management strategies for cardoon cultivation. This cost is not sustained by farmers because, in the absence of compensatory regulatory mechanisms, the cost is paid collectively in the long term (Havranek et al., 2015).

This is a critical point because farmers are deprived of responsibility and do not pay any direct costs from SOCS reduction in order to pursue their own economic objectives (typically profit maximization). Furthermore, the costs would not be equally distributed since we would expect that the less-developed countries would bear more of the costs. In fact, richer and more developed countries are more able to pay the costs related to negative externalities with the greater benefits generated by higher agricultural productivity and profitability. This disparity implies that the estimated SCC in our analysis would tend to increase in developing countries and, in parallel, to decrease in developed countries.

A general solution for avoiding social costs and limiting disparities would be the introduction of a normative mechanism regarding C production that is based on property rights and is able to internalize these costs into the agricultural practices selected by farmers. In other words, the introduction of tax schemes or other mechanisms might transfer the costs from society to the farmers who produce these externalities and create an incentive (disincentive) for increasing (decreasing) C storage. In this way, the costs related to SOCS reduction become an "internal" cost for farmers in addition to their other production costs, and C storage becomes an economic variable that is considered with the other typical economic variables in defining farmer choices (aimed at increasing productivity and thus maximizing profits).

In conclusion, more empirical evidence needs to be obtained to extend this analysis to the management of other perennial energy crop systems and to geographical contexts other than the Mediterranean region, to estimate the costs related to GHG emissions in the long term and to develop effective tools for "internalizing" the SCC into farmer decisions.

5. Conclusions

This study estimates the potential performance of a cardoon crop system in terms of long-term GHG reduction and SOC storage. Two methodological approaches were combined (i.e., CF and SCC) to assess different fertilizer treatments. The results stress the difficulty of identifying the optimal fertilization pattern in terms of GHG production and SOC storage. The HI treatment resulted in the worst GHG balance and the highest SOCS, whereas LI + CC demonstrated good performance in terms of GHG emission reduction and yield, followed by that of LI + Bi. In the LI + Bi + CC treatment, the combined use of biochar and a cover crop fostered neither C sequestration nor a decrease in GHG emissions.

The monetary estimation of the ecosystem service provided by soil C storage highlighted the benefit reduction involved in switching from HI management to the other practices and the need to "internalize" the SCC into farmer choices in order to address this environmental externality. This means that C storage should be considered on the same level as other agricultural input costs in order to optimize practices while also considering cardoon production and environmental performance.

More generally, a best option that could guarantee an optimal level of food security and environmental and socio-economic sustainability could not be identified. This study emphasizes the importance of finding trade-offs among productivity, GHG dynamics, and the monetary value of ecosystem services (e.g., C sequestration) provided by the agricultural management of perennial energy crops. This potential solution would allow the optimization of long-term crop system planning and land use to develop effective measures to address climate change.

The lack of a best option could lead to different choices by farmers and public decision makers. The former should move towards solutions that compromise between the need to maintain technical and economic productivity and the need to minimize GHG emissions. Social costs play a less important role in their choices, especially in the absence of compensation mechanisms that burden entrepreneurs. Conversely, this latter aspect is particularly important in the choices of public decision-makers who, in the absence of an optimal solution, should develop solutions aimed at containing social costs as much as possible from a long-term perspective.

At the same time, these results offer interesting insights for researchers for at least two reasons. First, research is needed to identify technical solutions capable of providing an appropriate level of productivity and minimizing the environmental impacts associated with cardoon fertilization. In this context, the dual methodological approach adopted in this research may be considered an attempt to obtain more detailed information for specifying a fertilization pattern that is able to ensure higher productivity, higher carbon storage in the long term, and lower greenhouse gas emissions for a perennial energy crop system.

Second, other empirical evidence relating to cardoon and other energy crops is needed to create a base of scientific information that will allow the main decision-makers - agricultural entrepreneurs and policy makers - to make the most rational choices.

Funding

This research did not receive any specific grant from funding agencies in the public, commercial, or not-for-profit sectors.

Acknowledgments

The authors thank Roberta Farina for her assistance in the application of RothC model, Leonardo Sulas for his valuable suggestions regarding legume cover crop, and Ester Spissu for some information provided on cardoon cultivation.

References

Agegnehu, G., Bass, A.M., Nelson, P.N., Bird, M.I., 2016. Benefits of biochar, compost and biochar–compost for soil quality, maize yield and greenhouse gas emissions in a tropical agricultural soil. Sci. Total Environ. 543, 295–306. https://doi.org/10.1016/j.scitotenv.2015.11.054.

Al-Mansour, F., Jejcic, V., 2017. A model calculation of the carbon footprint of agricultural products: The case of Slovenia. Energies 136, 7–15. http://dx.doi.org/10.1016/j.energy.2016.10.099.

Alani, R., Odunuga, S., Andrew-Essien, N., Appia, Y., Muyiolu, K., 2017. Assessment of the Effects of Temperature, Precipitation and Altitude on Greenhouse Gas Emission from Soils in Lagos Metropolis. J. Environ. Prot. 8, 98–107. http://dx.doi.org/10.4236/jep.2017.81008.

Albanito, F., Beringer, T., Corstanje, R., Poulter, B., Stephenson, A., Zawadzka, J., Smith, P., 2016. Carbon implications of converting cropland to bioenergy crops or forest for climate mitigation: a global assessment. GCB Bioenergy 8, 81–95. doi: 10.1111/gcbb.12242.

Anthoff, D., Tol, R.S. J., 2013. The uncertainty about the social cost of carbon: A decomposition analysis using fund. Climatic Change 117, 515–530. DOI 10.1007/s10584-013-0706-7.

Balafoutis, A., Beck, B., Fountas, S., Vangeyte, J., Wal, T.V., Soto, I., Gómez-Barbero, M., Barnes, A., Eory, V., 2017. Precision Agriculture Technologies Positively Contributing to GHG Emissions Mitigation, Farm Productivity and Economics. Sustainability 9, 1–28. https://doi.org/10.3390/su9081339.

Baldo, G.L., Marino, M., Montani, M., Ryding, S.-O., 2009. The carbon footprint measurement toolkit for the EU Ecolabel. Int. J. Life Cycle Ass. 14, 591–596. https://doi.org/10.1007/s11367-009-0115-3.

Belda, M., Holtanová, E., Halenka, T., Kalvová, J., 2014. Climate classification revisited: from Köppen to Trewartha. Clim. Res. 59, 1–13. https://doi.org/10.3354/cr01204.

Birkved, M., Michael Hauschild, Z., 2006. PestLCI—A model for estimating field emissions of pesticides in agricultural LCA. Ecol. Modell. 198, 433–451. https://doi.org/10.1016/j.ecolmodel.2006.05.035.

Bis, Z., Kobyłecki, R., Ścisłowska, M., Zarzycki, R., 2018. Biochar – Potential tool to combat climate change and drought. Ecohydrol. Hydrobiol. 18, 441–453. https://doi.org/10.1016/j.ecohyd.2018.11.005.

Borchard, N., Schirrmann, M., Cayuela, M.L., Kammann, C., Wrage-Mönnig, N., Estavillo, J.M., Fuertes-Mendizábal, T., Sigua, G., Spokas, K., Ippolito, J.A., Novak, J., 2019. Biochar, soil and land-use interactions that reduce nitrate leaching and N2O emissions: A meta-analysis. Sci. Total Environ. 651, 2354–2364. https://doi.org/10.1016/j.scitotenv.2018.10.060.

Bozhanska, T., Mihovski, T., Naydenova, G., Knotová, D., Pelikán, J., 2016. Comparative studies of annual legumes. Biotech. Anim. Husbandry 32, 311–320. DOI: 10.2298/BAH1603311B.

Brentrup, F., Küsters, J., Lammel, J., Kuhlmann, H., 2000. Methods to estimate on-field nitrogen emissions from crop production as an input to LCA studies in the agricultural sector. Int. J. Life Cycle Asses. 5, 349 –357. https://doi.org/10.1007/BF02978670.

Cheng, K., Yan, M., Pan, G., Luo, T., Yue, Q., 2015. Methodology for Carbon Footprint Calculation in Crop and Livestock Production, in: Kannan, S.S. (Eds.), The Carbon Footprint Handbook. CRC Press Boca Raton, pp. 61–84.

Chiofalo, B., Simonella, S., Di Grigoli, A., Liotta, L., Frenda, A.S., Lo Presti, V., Bonanno, A., Chiofalo, V., 2010. Chemical and acidic composition of longissimus dorsi muscle of Comisana lambs fed with Trifolium subterraneum and Lolium multiflorum. Small Rumin. Res. 88, 89–96. https://doi.org/10.1016/j.smallrumres.2009.12.015.

Chojnacka, K., Moustakas, K., Witek-Krowiak, A., 2020. Bio-based fertilizers: A practical approach towards circular economy. Bioresour. Technol. 295, 122223. https://doi.org/10.1016/j.biortech.2019.122223.

Coleman , K., Jenkinson, D.S., 2014. RothC - A model for the turnover of carbon in soil:Model Description and User’s Guide. Rothamsted Research Harpenden, UK. Available at: https://www.rothamsted.ac.uk/rothamsted-carbon-model-rothc. (accessed 25 February 2020).

Cronin, J., Zabel, F., Dessens, O., Anandarajah, G., 2020. Land suitability for energy crops under scenarios of climate change and land-use. GCB Bioenergy 12, 648–665. https://doi.org/10.1111/gcbb.12697.

Curran, M.A., 2013. Life Cycle Assessment: a review of the methodology and its application to sustainability. Curr. Opin. Chem. Eng. 2, 273–277. https://doi.org/10.1016/j.coche.2013.02.002.

Daryanto, S., Fua, B., Wang, L., Jacinthe, P.-A., Wenwu, Z., 2018. Quantitative synthesis on the ecosystem services of cover crops. Earth Sci. Rev. 185, 357-373. https://doi.org/10.1016/j.earscirev.2018.06.013.

De Klein, C., Novoa, R.S.A., Ogle, S., Smith, K.A., Rochette, P., Wirth, T.C., McConkey, B.G., Mosier, A., Rypdal, K., 2006. N2O emissions from managed soils, and CO2 emissions from lime and urea application, in: Egglestonne, H.S., Buendia, L., Miwa, K., Ngara, T., Tanabe, K. (Eds.), 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Published: IGES, Japan, pp. 11.1–11.54.

De Menna, F., Malagnino, R.A., Vittuari, M., Segrè, A., Molari, G., Deligios, P.A., Solinas, S., Ledda, L., 2018. Optimization of agricultural biogas supply chains using artichoke byproducts in existing plants. Agric. Sys. 165, 137–146. https://doi.org/10.1016/j.agsy.2018.06.008.

Deligios, P.A., Sulas, L., Spissu, E., Re, G.A., Farci, R., Ledda, L., 2017. Effect of input management on yield and energy balance of cardoon crop systems in Mediterranean environment. Eur. J. Agron. 82, 173–181. https://doi.org/10.1016/j.eja.2016.10.016.

Dijkman, T.J., Birkved, M., Hauschild, M.Z., 2012. PestLCI 2.0: A second generation model for estimating emissions of pesticides from arable land in LCA. Int. J. Life Cycle Assess. 17, 973–986. https://doi.org/10.1007/s11367-012-0439-2.

Don, A., Osborne, B., Hastings, A., Skiba, U., Carter, M.S., Drewer, J., Flessa, H., Freibauer, A., Hyvöne, N., Jones, M.B., Lanigan, G.J., Mander, Ü. Monti, A., Djomo, S.N., Valentine, J., Walter, K., Zegada-Lizarazu, W., Zenone, T., 2012. Land-use change to bioenergy production in Europe: implications for the greenhouse gas balance and soil carbon. GCB Bioenergy 4, 372–391. doi: 10.1111/j.1757-1707.2011.01116.x.

Drewer, J., Finch, J.W., Lloyd, C.R., Baggs, E.M., Skiba, A., 2012. How do soil emissions of N_2O, CH_4 and CO_2 from perennial bioenergy crops differ from arable annual crops? Glob. Change Biol. Bioenergy 4, 408–419. https://doi.org/10.1111/j.1757-1707.2011.01136.x.

EEA (European Environment Agency), 2018. Annual European Union greenhouse gas inventory 1990–2016 and inventory report 2018. European Commission, DG Climate Action European Environment Agency Brussels.

EFE-So, 2015. Estimation of Fertilisers Emissions-Software. Available at: http://www.sustainable-systems.org.uk/tools.php. (accessed 18 February 2020).

Falloon, P., Smith, P., Coleman, K., Marshall S., 1998. Estimating the size of the inert organic matter pool from total soil organic carbon content for use in the Rothamasted carbon model. Soil Biol. biochem. 30, 1207–1211. DOI: 10.1016/S0038-0717(97)00256-3.

Fernández, J., Curt, M.D., Aguado, P.L., 2006. Industrial applications of Cynara cardunculus L.
for energy and other uses. Ind. Crop. Prod. 24, 222–229. doi:10.1016/j.indcrop.2006.06.010.

Fernando, A. L., Costa, J., Barbosa, B., Monti, A., Rettenmaier, N., 2018. Environmental impact assessment of perennial crops cultivation on marginal soils in the Mediterranean Region. Biomass Bioenerg., 111, 174–186. https://doi.org/10.1016/j.biombioe.2017.04.005.

Fidel, R.B., Laird, D.A., Parkin, T.B., 2018. Effect of biochar on soil greenhouse gas emissions at the laboratory and field scales. Preprints 2018, 2018100315. doi: 10.20944/preprints201810.0315.v1.

Forster, P., Ramaswamy, V., Artaxo, P., Berntsen, T., Betts, R., Fahey, D.W., Haywood, J., Lean, J., Lowe, D.C., Myhre, G., Nganga, J., Prinn, R., Raga, G., Schulz, M., Van Dorland, R., 2007. Changes in Atmospheric Constituents and in Radiative Forcing, in: Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor M., Miller H.L. (Eds.), Cambridge University Press New York, pp. 129–234.

Francaviglia, R., Bruno, A., Falcucci, M., Farina, R., Renzi G., Russo, D.E., Sepe, L., Neri, U., 2016. Yields and quality of Cynara cardunculus L. wild and cultivated cardoon genotypes. A case study from a marginal land in Central Italy. Eur. J. Agron. 72, 10–19. http://dx.doi.org/10.1016/j.eja.2015.09.014.

Garrigues, E., Corsona, M.S., Angers, D.A., van der Werf, H.M.G., Walter, C., 2012. Soil quality in Life Cycle Assessment: towards development of an indicator. Ecol. Indic. 18, 434–442. https://doi.org/10.1016/j.ecolind.2011.12.014.

Gatto, A., De Paola, D., Bagnoli, F., Vendramin, G.G., Sonnante, G., 2013. Population structure of Cynara cardunculus complex and the origin of the conspecific crops artichoke and cardoon. Ann. Bot. 112, 855–865. doi:10.1093/aob/mct150.

Goedkoop, M., Oele, M., Leijting, J., Ponsioen, T., Meijer, E., 2013a. Introduction to LCA with SimaPro. PRé Consultants, The Netherlands.

Goedkoop, M., Oele, M., Vieira, M., Leijting, J., Ponsioen, T., Meijer, E., 2013b. SimaPro Tutorial. PRé Consultants, The Netherlands.

Goglio, P., Smith, W.N., Grant, B.B., Desjardins, R.L. McConkey, B.G., Campbell, C.A., Nemecek, T., 2015. Accounting for soil carbon changes in agricultural life cycle assessment (LCA): a review. J. Clean. Prod. 104, 23–39. https://doi.org/10.1016/j.jclepro.2015.05.040.

Goglio, P., Smith, W.N., Grant, B.B., Desjardins, R.L., Gao, X., Hanis, K., Tenuta, M., Campbell, C.A., McConkey, B.G., Nemecek, T., Burgess, P.J., Williams A.G., 2018. A comparison of methods to quantify greenhouse gas emissions of cropping systems in LCA. J. Clean. Prod. 172, 4010–4017. https://doi.org/10.1016/j.jclepro.2017.03.133.

Gominho, J., Curt, M.D., Lourenço, A., Fernández, J., Pereira, H., 2018. Cynara cardunculus L. as a biomass and multi-purpose crop: A review of 30 years of research. Biomass Bioenerg. 109, 257–275. https://doi.org/10.1016/j.biombioe.2018.01.001.

González-Molina, L., Etchevers-Barra, J.D., Paz-Pellat, F., Díaz-Solis, H., Fuentes-Ponce, M.H., Covaleda-Ocón, S., Pando-Moreno, M., 2011. Performance of the RothC-26.3 model in short-term experiments in Mexican sites and systems. J. Agric. Sci., 149, 415–425. DOI: https://doi.org/10.1017/S0021859611000232.

Greenstone, M., Kopits, E., Wolvertonne, A., 2013. Developing a Social Cost of Carbon for US Regulatory Analysis: A Methodology and Interpretation. Rev. Environ. Econ. Policy 7, 23–46. http://dx.doi.org/10.1093/reep/res015.

Hauschild, M.Z., Potting, J., Hertel, O., Schöpp, W., Bastrup-Birk, A., 2006. Spatial Differentiation in the Characterisation of Photochemical Ozone Formation. Int. J. LCA 11, 72–80. DOI: http://dx.doi.org/10.1065/lca2006.04.014.

Havranek, T., Irsova, Z., Janda, K., Zilberman, D., 2015. Selective reporting and the social cost of carbon. Energ. Econ. 51, 394–406. https://doi.org/10.1016/j.eneco.2015.08.009.

Houghton, J.T., Meira Filho, L.G., Lim, B., Treanton, K., Mamaty, I., Bonduki, Y., Griggs, D.J., Callender, B.A. (Eds.) 1997: Greenhouse Gas Inventory Reporting Instructions, Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories, Volumes 1-3. The intergovernmental Panel on Climate Change (IPCC), London, United Kingdom.

Ierna, A., Mauro, R.P., Mauromicale, G., 2012. Biomass, grain and energy yield in Cynara cardunculus L. as affected by fertilization, genotype and harvest time. Biomass Bioenerg. 36, 404–410. doi:10.1016/j.biombioe.2011.11.013.

Ingram. J., Mills, J., Frelih- Larsen, A., McKenna, D., Merante, P., Ringrose, S., Molnar, A., Sánchez, B., Ghaley, B.B., Karaczun, Z., 2014. Managing Soil Organic Carbon: A Farm Perspective. EuroChoices 13, 12–19. https://doi.org/10.1111/1746-692X.12057.

ISO 14040, 2006. Environmental Management – Life Cycle Assessment – Principles and Framework. International Standard Organization.

IWG, Interagency Working Group on Social Cost of Greenhouse Gases, United States Government, 2016. Technical Support Document: - Technical Update of the Social Cost of Carbon for Regulatory Impact Analysis - Under Executive Order 12866.

JRC, 2007. Carbon Footprint - what it is and how to measure it. European Commission.

Kaonga, M.L., Coleman, K., 2008. Modelling soil organic carbon turnover in improved fallows in eastern Zambia using the RothC-26.3 model. Forest. Ecol. Manag. 256, 1160–1166. https://doi.org/10.1016/j.foreco.2008.06.017.

Karaosmanoğlu F., Işiğigür-Ergüdenler A., Sever, A., 2000. Biochar from the straw-stalk of rapeseed plant. Energy Fuels 14, 336–339. DOI: 10.1021/ef9901138.

Kottek, M., Grieser, J., Beck, C., Rudolf, B., Rubel, F., 2006. World Map of the Köppen-Geiger climate classification updated. Meteorologische Zeitschrift, 15, 259–263. DOI: 10.1127/0941-2948/2006/0130.

Kuppusamy, S., Thavamani, P., Megharaj, M., Venkateswarlu, K., Naidu, R., 2016. Agronomic and remedial benefits and risks of applying biochar to soil: Current knowledge and future research directions. Environmental International 87, 1–12. https://doi.org/10.1016/j.envint.2015.10.018.

Kuzyakova, Y., Friedel, J.K., Stahr, K., 2000. Review of mechanisms and quantification of priming effects. Soil Biol. Biochem. 32, 1485–1498. http://dx.doi.org/10.1016/S0038-0717(00)00084-5.

Ledda, L., Deligios, P.A., Farci, R., Sulas, L., 2013. Biomass supply for energetic purpose from some Cardueae species grown in Mediterranean farming systems. Ind. Crop. Prod. 47, 218–226, http://dx.doi.org/10.1016/j.indcrop.2013.03.013.

Lehtinen, T., Schlatter, N., Baumgarten, A., Bechini, L., Krüger, J., Grignani, C., Zavattaro, L., Costamagna, C., Spiegel, H., 2014. Effect of crop residue incorporation on soil organic carbon and greenhouse gas emissions in European agricultural soils. Soil Use Manage. 30, 524–538. doi: 10.1111/sum.12151.

Li, S., Li, J., Li, C., Huang, S., Li, X., Li, S., Ma, Y., 2016. Testing the RothC and DNDC models against long-term dynamics of soil organic carbon stock observed at cropping field soils in North China. Soil Tillage Res. 163, 290–297. https://doi.org/10.1016/j.still.2016.07.001.

López-Bellido, L., Wery, J., López-Bellido, R.J., 2014. Energy crops: Prospects in the context of sustainable agricolture. Eur. J. Agron. 60, 1–12. https://doi.org/10.1016/j.eja.2014.07.001.

Lozano-García, B., Muñoz-Rojas, M., Parras-Alcántara, L., 2017. Climate and land use changes effects on soil organic carbon stocks in a Mediterranean semi-natural area. Sci. Total Environ. 579, 1249–1259. https://doi.org/10.1016/j.scitotenv.2016.11.111.

Maestrini, B., Nannipieri, P., Abiven, S., 2015. A meta- analysis on pyrogenic organic matter induced priming effect. Glob. Change Biol. Bioenergy 7, 577–590. https://doi.org/10.1111/gcbb.12194.

Markaki, Z., Loÿe-Pilot, M.D., Violaki, K., Benyahya, L., Mihalopoulos, N., 2010. Variability of atmospheric deposition of dissolved nitrogen and phosphorus in the Mediterranean and possible link to the anomalous seawater N/P ratio. Mar. Chem. 120, 187–194. https://doi.org/10.1016/j.marchem.2008.10.005.

Mauromicale, G., Sortino, O., Pesce, G.R., Agnello, M., Mauro, R.P., 2014. Suitability of cultivated and wild cardoon as a sustainable bioenergy crop for low input cultivation in low quality Mediterranean soils. Ind. Crops Prod. 57, 82–89. https://doi.org/10.1016/j.indcrop.2014.03.013.

Mehmood, M.A. Ibrahim, M., Rashid, U., Nawaz, M., , Shafaqat, Ali, Hussain, A., Gull, M., 2017. Biomass production for bioenergy using marginal lands. Sustain. Prod. Consump. 9, 3–21. https://doi.org/10.1016/j.spc.2016.08.003.

Moraleda Melero, C.M., 2018. PestLCI Pesticide Emission Fraction Estimation for LCA. Quantitative Sustainability Assessment, Department of Management Engineering, Technical University of Denmark. http://www.qsa.man.dtu.dk/research/research-projects/pestlci (accessed 10 February 2020).

Morawicki, R.O., Hager, T., 2014. Energy and greenhouse gases footprint of food processing, in: Van Alfen, N.K., (Eds.), Encyclopedia of Agriculture and Food Systems, Elsevier, pp.82-99.

Mutel, C., Liao, X., Patouillard, L., Bare, J., Fantke, P., Frischknecht, R., Hauschild, M., Jolliet, O., de Souza, D.M., Laurent, A., Pfister, S., Verones, F., 2019. Overview and recommendations for regionalized life cycle impact assessment. Int. J. Life Cycle Ass. 24, 856–865. https://doi.org/10.1007/s11367-018-1539-4.

Nayak, A.K., Rahman, M.M., Naidu, R., Dhal, B., Swaina, C.K., Nayak, A.D., Tripathi, R., Shahid, M., Islam, M.R., Pathak, H., 2019. Current and emerging methodologies for estimating carbon sequestration in agricultural soils: A review. Sci. Total Environ. 665, 890–912. https://doi.org/10.1016/j.scitotenv.2019.02.125.

Nemecek, T., Dubois, D., Huguenin-Elie, O., Gaillard, G., 2011. Life cycle assessment of Swiss farming systems: I. Integrated and organic farming. Agric. Syst. 104, 217–232. https://doi.org/10.1016/j.agsy.2010.10.002.

Neri, U., Pennelli, B., Simonetti, G., Francaviglia, R., 2017. Biomass partition and productive aptitude of wild and cultivated cardoon genotypes (Cynara cardunculus L.) in a marginal land of Central Italy. Ind. Crop Prod. 95, 191–201. http://dx.doi.org/10.1016/j.indcrop.2016.10.029.

Niemi, EG., 2018. The Social Cost of Carbon. Natural Resource Economics, Eugene, OR, United States, Elsevier.

Nordhaus, W.D., 2017. Revisiting the social cost of carbon. PNAS 114, 1518–1523. https://doi.org/10.1073/pnas.1609244114.

Notarnicola, B., Tassielli, G., Renzulli, P.A., Lo Giudice, A., 2015. Life Cycle Assessment in the agri-food sector: an overview of its key aspects, international initiatives, certification, labelling schemes and methodological issues, in: Notarnicola, B., Salomone, R., Petti, L., Renzulli, P.A., Roma, R., Cerutti, A.K. (Eds.), Life Cycle Assessment in the Agri-food Sector, Case Studies, Methodological Issues and Best Practices. Springer International Publishing: Switzerland, pp. 1–56.

Oldfield, T.L., Sikirica, N., Mondini, C., López, G., Kuikman, P.J., Holden, N.M., 2018. Biochar, compost and biochar-compost blend as options to recover nutrients and sequester carbon. J. Environ. Manage. 218, 465–476. https://doi.org/10.1016/j.jenvman.2018.04.061.

Pace, V., Contò, G., Carfì, F., Chiariotti, A., Catillo, G., 2011. Short- and long-term effects of low estrogenic subterranean clover on ewe reproductive performance. Small Rumin. Res. 97, 94–100. https://doi.org/10.1016/j.smallrumres.2011.02.011.

Pan, S.-Y., Du, M.A., Huang, I.-T., Liu, I.-H., Chang , E.-E., Chiang, P.-C., 2015. Strategies on implementation of waste-to-energy (WTE) supply chain for circular economy system: a review. J. Clean. Prod. 108, 409–421. http://dx.doi.org/10.1016/j.jclepro.2015.06.124.

Panda, D., Mishra, S., Swain, K.C., Chakraborty, N.R., Mondal, S., 2016. Bio-Energy crops in mitigation of climate change. Int. J. Bio-res. Env. Agril. Sci 2, 242–250. ISSN 2454-3551.

Pandey D., Agrawal M., 2014. Carbon Footprint Estimation in the Agriculture Sector, in: Muthu S. (Eds.), Assessment of Carbon Footprint in Different Industrial Sectors, Volume 1. EcoProduction (Environmental Issues in Logistics and Manufacturing). Springer, Singapore, pp. 25–47.

Perpiña Castillo, C., Baranzelli, C., Maes, J., Zulian, G., Lopes Barbosa, A., Vandecasteele, I., Mari Rivero, I., Vallecillo Rodriguez, S., Batista, E., Silva, F., Jacobs, C., Lavalle, C., 2016. An assessment of dedicated energy crops in Europe under the EU Energy Reference Scenario 2013 Application of the LUISA modelling platform – Updated Configuration 2014. EUR 27644. doi:10.2788/64726.

Peter, C., Helming, K., Nendel, C., 2017. Do greenhouse gas emission calculations from energy crop cultivation reflect actual agricultural management practices? – A review of carbon footprint calculators. Renew. Sust. Energ. Rev. 67, 461–476. https://doi.org/10.1016/j.rser.2016.09.059.

Petersen, B.M., Knudsen, M.T., Hermansen, J.E., Halberg, N., 2013. An approach to include soil carbon changes in life cycle assessments. J. Clean. Prod. 52, 217–224. https://doi.org/10.1016/j.jclepro.2013.03.007.

Pimentel, L.G., Weiler, D.A., Pedroso, G.M., Bayer, C., 2015. Soil N_2O emissions following cover-crop residues application under two soil moisture conditions. J. Plant Nutr. Soil Sci. 178, 631–640. https://doi.org/10.1002/jpln.201400392.

Planton, S., Driouech, F., El Rhaz, K., Lionello, P., 2016. The climate of the Mediterranean regions in the future climate projections, in: Thiébault, S., Moatti J.P (Eds.), The Mediterranean region under climate change: a scientific update. IRD Éditions Institut De Recherche Pour Le Développement, Marseille, pp. 83–92.

Patouillard, L., Collet, P., Lesage, P., Tirado Seco, P., Bulle, C., Margni, M., 2019. Prioritizing regionalization efforts in life cycle assessment through global sensitivity analysis: a sector meta-analysis based on ecoinvent v3. Int. J. Life Cycle Ass. 24, 2238–2254. https://doi.org/10.1007/s11367-019-01635-5.

PRé, various authors, 2018. SimaPro Database Manual Methods Library. 2002-2013 PRé, Netherlands.

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.

Ramachandra, T.V., Mahapatra, D.M., 2015. The Science of Carbon Footprint assessment, in: Kannan, S.S. (Eds.), The Carbon Footprint Handbook. CRC Press Boca Raton, pp. 3–45.

Razza, F., Sollima, L., Falce, M., Costa, R.M.S., Toscano, V., Novelli, A., Ciancolini, A., Raccuia, S.A., 2016. Life cycle assessment of cardoon production system in different areas of Italy. Acta Hortic. 1147, 329–334. DOI: 10.17660/ActaHortic.2016.1147.46.

Rebolledo-Leiva, R., Angulo-Meza, L., Iriarte, A., González-Araya M.C., 2017. Joint carbon footprint assessment and data envelopment analysis for the reduction of greenhouse gas emissions in agriculture production. Sci. Total Environ. 593-594, 36–46. http://dx.doi.org/10.1016/j.scitotenv.2017.03.147.

Rose, S.K., Turner, D., Blanford, G., Bistline, J., de la Chesnaye, F., Wilson, T., 2014. Understanding the Social Cost of Carbon: A Technical Assessment. EPRI, Palo Alto, CA: 2014. Report #3002004657.

Russell, S., 2011. Corporate greenhouse gas inventories for agricultural sector: proposed accounting and reporting steps. WRI Working Paper. World Resources Institute. Washington, DC. pp. 29.

Sagrilo E., Jeffery, S., Hoffland, E., Kuyper, T.W., 2015. Emission of CO2 from biochar- amended soils and implications for soil organic carbon. Glob. Change Biol. Bioenergy 7, 1294–1304. https://doi.org/10.1111/gcbb.12234.

Salis, M., Ager, A.A., Arca, B., Finney, M.A., Bacciu, V., Duce, P., Spano, D., 2013. Assessing exposure of human and ecological values to wildfire in Sardinia, Italy. Int. J. Wildland Fire 22, 549–565. http://dx.doi.org/10.1071/WF11060.

Sanz-Cobeña, A., Lassaletta, L., Aguilera, E., del Prado, A., Garniere, J., Billen, G., Iglesias, A., Sánchez, B., Guardia, G., Abalos, D., Plaza-Bonilla, D., Puigdueta-Bartolomé, I., Moral, R., Galán, E., Arriaga, H., Merino, P., Infante-Amate, J., Meijide, A., Pardo, G., Álvaro-Fuentes, J., Gilsanz, C., Báez, D., Doltra, J., González-Ubierna, S., Cayuela, M.L., Menéndez, S., Díaz-Pinés, E., Le-Noë, J., Quemada, M., Estellés, F., Calvet, S., van Grinsven, H.J.M., Westhoek, H., Sanz, M.J., Gimeno, B.S., Vallejo, A., Smith, P., 2017. Strategies for greenhouse gas emissions mitigation in Mediterranean agriculture: A review. Agric. Ecosyst. Environ. 238, 5–24. https://doi.org/10.1016/j.agee.2016.09.038.

Sauer B., 2012. Life Cycle Inventory Modeling in Practice, in Curran M.A., (Eds.), Life Cycle Assessment Handbook: A Guide for Environmentally Sustainable Products. Co-published by John Wiley & Sons, Inc. Hoboken, New Jersey, and Scrivener Publishing LLC, Salem, Massachusetts, pp. 43–66.

Shan, J., Yan, X., 2013. Effects of crop residue returning on nitrous oxide emissions in agricultural soils. Atmos. Environ. 71, 170–175. http://dx.doi.org/10.1016/j.atmosenv.2013.02.009.

Shen, Y., Zhu, L., Cheng, H., Yue, S., Li, S., 2017. Effects of biochar application on CO2 Emissions from a cultivated soil under semiarid climate conditions in northwest China. Sustainability 9, 1–13. DOI: 10.3390/su9081482.

Sherwood, J., 2020. The significance of biomass in a circular economy. Bioresour. Technol. 300, 122755. https://doi.org/10.1016/j.biortech.2020.122755.

Singh, B.P., Cowie, A.L., 2014. Long-term influence of biochar on native organic carbon mineralisation in a low-carbon clayey soil. Scientific Reports 4, 1–9. https://doi.org/10.1038/srep03687.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F., Rice, C., Scholes, B., Sirotenko, O., Howden, M., McAllister, T., Pan, G., Romanenkov, V., Schneider, U., Towprayoon, S., Wattenbach, M., Smith, J., 2008. Greenhouse gas mitigation in agriculture. Phil. Trans. R. Soc. B 363, 789–813. doi:10.1098/rstb.2007.2184.

Smith, P., 2012. Agricultural greenhouse gas mitigation potential globally, in Europe and in the UK: what have we learnt in the last 20 years?. Glob. Change Biol. 18, 35–43. https://doi.org/10.1111/j.1365-2486.2011.02517.x.

Smith, P., House, J.I., Bustamante, M., Sobock, J., Harper, R., Pan, G., West, P.C., Clark, J.M., Adhya, T., Rumpel, C., Paustian, K., Kuikman, P., Cotrufo, M.F., Elliott, J.A., Mcdowell, R., Griffiths, R.I., Asakawa, S., Bondeau, A., Jain, A.K., Meersmans, J., Pugh, T.A.M., 2016. Global change pressures on soils from land use and management. Glob. Change Biol. 22, 1008–1028. doi: 10.1111/gcb.13068.

Söderström, B., Hedlund, K., Jackson, L.E., Kätterer, T., Lugato, E., Thomsen, I.K., Bracht Jørgensen, H., 2014. What are the effects of agricultural management on soil organic carbon (SOC) stocks?. Environ. Evid. 3, 2. https://doi.org/10.1186/2047-2382-3-2.

Solinas, S., Fazio, S., Seddaiu, G., Roggero, P.P., Deligios, P.A., Doro, L., Ledda, L., 2015. Environmental consequences of the conversion from traditional to energy cropping systems in a Mediterranean area. Eur. J. Agron. 70, 124–135. https://doi.org/10.1016/j.eja.2015.07.008.

Solinas, S., Deligios, P.A., Sulas, L., Carboni, G., Virdis, A., Ledda, L., 2019. A land-based approach for the environmental assessment of Mediterranean annual and perennial energy crops. Eur. J. Agron. 103, 63–72. https://doi.org/10.1016/j.eja.2018.11.007.

Tan, Z., Lin, C.S.K., Ji, X., Rainey, T.J., 2017. Returning biochar to fields: A review. Appl. Soil Ecol. 116, 1–11. https://doi.org/10.1016/j.apsoil.2017.03.017.

Tiemann, L.K., Grandy, S., 2014. Mechanisms of soil carbon accrual and storage in bioenergy cropping systems. Glob. Change Biol. Bioenergy 7, 161–174. https://doi.org/10.1111/gcbb.12126.

van den Bijgaart, I., Gerlagh, R., Liski, M., 2016. A simple formula for the social cost of carbon. J. Environ. Econ. Manag. 77, 75–94. https://doi.org/10.1016/j.jeem.2016.01.005.

Wagner, M., Lewandowski, I., 2017. Relevance of environmental impact categories for perennial biomass production. Glob. Change Biol. Bioenergy 9, 215–228. doi: 10.1111/gcbb.12372.

Weidema B.P., Meeusen, M.J.G., 2000. Agricultural data for Life Cycle Assessments. Agricultural Economics Research Institute (LEI), The Hague.

Woolf, D., Amonette, J.E., Street-Perrott, F.A., Lehmann, J., Joseph, S., 2010. Sustainable biochar to mitigate global climate change: Supplementary information. Nat. Commun. 1, 1–9. https://doi.org/10.1038/ncomms1053.

WRI and WBCSD, 2011a. Product Life Cycle Accounting and Reporting Standard. World Resources Institute and World Business Council for Sustainable Development. http://www.ghgprotocol.org/ (accessed 15 February 2020).

WRI and WBCSD, 2011b. GHG Protocol Agricultural Guidance, Interpreting the Corporate Accounting and Reporting Standard for the agricultural sector. World Resources Institute and World Business Council for Sustainable Development. http://www.ghgprotocol.org/ (accessed 15 February 2020).

Wu, Y., Lin, S., Liu, T., Wan, T., Hu, R., 2016. Effect of crop residue returns on N_2O emissions from red soil in China. Soil Use Manage. 32, 80–88. https://doi.org/10.1111/sum.12220.

Yang, Y., Tao, M., Sangwon, S., 2018. Geographic variability of agriculture requires sector-specific uncertainty characterization. Int. J. Life Cycle Assess. 23, 1581–1589. DOI 10.1007/s11367-017-1388-6.

Zhou, W., Jones, D.L., Hu, R., Clark, I.M., Chadwick, D.R., 2020. Crop residue carbon-to-nitrogen ratio regulates denitrifier N_2O production post flooding. Biol. Fertil. Soils 56, 825–838. https://doi.org/10.1007/s00374-020-01462-z.

Zimmermann, M., Leifeld, J., Schmidt, M.W.I., Smith, P., Fuhrer, J., 2007. Measured soil organic matter fractions can be related to pools in the RothC model. Eur. J. Soil Sci. 58, 658–667. https://doi.org/10.1111/j.1365-2389.2006.00855.x.

Zimmerman, A.R., Gao, B., Ahn, M.-Y., 2011. Positive and negative carbon mineralization priming effects among a variety of biochar-amended soils. Soil Biol. Biochem. 43, 1169–1179. https://doi.org/10.1016/j.soilbio.2011.02.005.

TABLES

Table 1

Nutrient supply for each treatment

Fertilizer/Soil amendment and cover crop	N input (kg ha^{-1} yr^{-1})	P input (kg ha^{-1} yr^{-1})	C input (kg ha^{-1} yr^{-1})	Fertilization type	Crop year
FERTILIZER INPUTS					
HI [a]					
Urea (46) [b]	79			Basal dressing	2014-2015
Diammonium phosphate (18-46) [b]	39	100		Basal dressing	2014-2015
Urea (46) [b]	100			Top dressing	2014-2015; 2015 2016; 2016-2017
Diammonium phosphate (18-46) [b]	25	65		Top dressing (sprounting stage)	2015 2016; 2016-2017
LI [a]					
Urea (46) [b]	79			Basal dressing	2014-2015
Diammonium phosphate (18-46) [b]	39	100		Basal dressing	2014-2015
Urea (46) [b]	50			Top dressing	2014-2015; 2015 2016; 2016-2017
Diammonium phosphate (18-46) [b]	25	65		Top dressing (sprounting stage)	2015 2016; 2016-2017
LI + Bi [a, c]					
Biochar			2,38 [d]	Basal dressing	2014-2015
LI + CC [a, c]					
Legume	12 [e]		274 [f]	Top dressing	2015 2016; 2016-2017

LI + Bi + CC [a, c]				
Biochar		2,38 [d]	Basal dressing	2014-2015
Legume	2.1 [e]	47.7 [f]	Top dressing	2015-2016; 2016-2017

[a] Fertilization patterns: HI, High Input; LI, Low Input; LI + Bi, Low Input + Biochar; LI+CC, Low Input+ Cover Crop; LI + Bi + CC, Low Input + Biochar + Cover Crop;

[b] Fertilizer title;

[c] LI + Bi, LI + CC and LI + Bi + CC scenarios were characterized by the same synthetic fertilizer inputs of LI;

[d] Value was obtained on the basis of what reported by Karaosmanoğlu et al. (2000);

[e] Value was estimated on the basis of an experimental trial on the same legume used in this study;

[f] Value was estimated on the basis of the information reported by Chiofalo et al. (2010); Prybil (2010); Pace et al. (2011); Bozhanska et al. (2016).

Table 2

Results from Monte Carlo analysis (confidence interval = 95%)

Pairwise comparison of MC scores					
CEFS [a]					
	HI [b]	LI [b]	LI + Bi [b]	LI + CC [b]	LI + Bi + CC [b]
HI [b]	-	100.0%	100.0%	100.0%	100.0%
LI [b]		-	89.6%	100.0%	84.2%
LI + Bi [b]			-	99.9%	100.0%
LI + CC [b]				-	89.4%
LI + Bi + CC [b]					-
CELT [a]					
	HI [b]	LI [b]	LI + Bi [b]	LI + CC [b]	LI + Bi + CC [b]
HI [b]	-	99.8%	100.0%	94.7%	58.2%
LI [b]		-	51.5%	100.0%	57.4%
LI + Bi [b]			-	55.0%	99.9%
LI + CC [b]				-	52.3%
LI + Bi + CC [b]					-
BCE [a]					
	HI [b]	LI [b]	LI + Bi [b]	LI + CC [b]	LI + Bi + CC [b]
HI [b]	-	99.8%	100.0%	70.4%	100.0%
LI [b]		-	100.0%	100.0%	100.0%
LI + Bi [b]			-	100.0%	100.0%
LI + CC [b]				-	100.0%
LI + Bi + CC [b]					-
CU [a]					
	HI [b]	LI [b]	LI + Bi [b]	LI + CC [b]	LI + Bi + CC [b]
HI [b]	-	99.5%	56.5%	100.0%	99.9%
LI [b]		-	93.0%	100.0%	100.0%
LI + Bi [b]			-	100.0%	100.0%

LI + CC [b]	-	93.7%
LI + Bi + CC [b]		-

[a] Impact categories: CEFS, Carbon Emission from Fossil Sources; BCE, Biogenic Carbon Emissions; CELT, Carbon Emission from Land Transformation; and CU, Carbon Uptake;

[b] Fertilization patterns: HI, High Input; LI, Low Input; LI + Bi, Low Input + Biochar; LI+CC, Low Input+ Cover Crop; LI + Bi + CC, Low Input + Biochar + Cover Crop.

Table 3

Social carbon cost estimation for the five treatments

	Discounted value (\$ $tCO_{2}e^{-1}$); 2017-2050				
	HI [a]	LI [a]	LI + Bi [a]	LI + CC [a]	LI + Bi + CC [a]
Social Carbon Cost	8,815.20	3,876.49	7,781.98	7,201.69	6,797.86
Benefit flow	-	4,938.72	1,033.23	1,613.51	2,017.34

[a] Fertilization patterns: HI, High Input; LI, Low Input; LI + Bi, Low Input + Biochar; LI+CC, Low Input+ Cover Crop; LI + Bi + CC, Low Input + Biochar + Cover Crop.

FIGURES

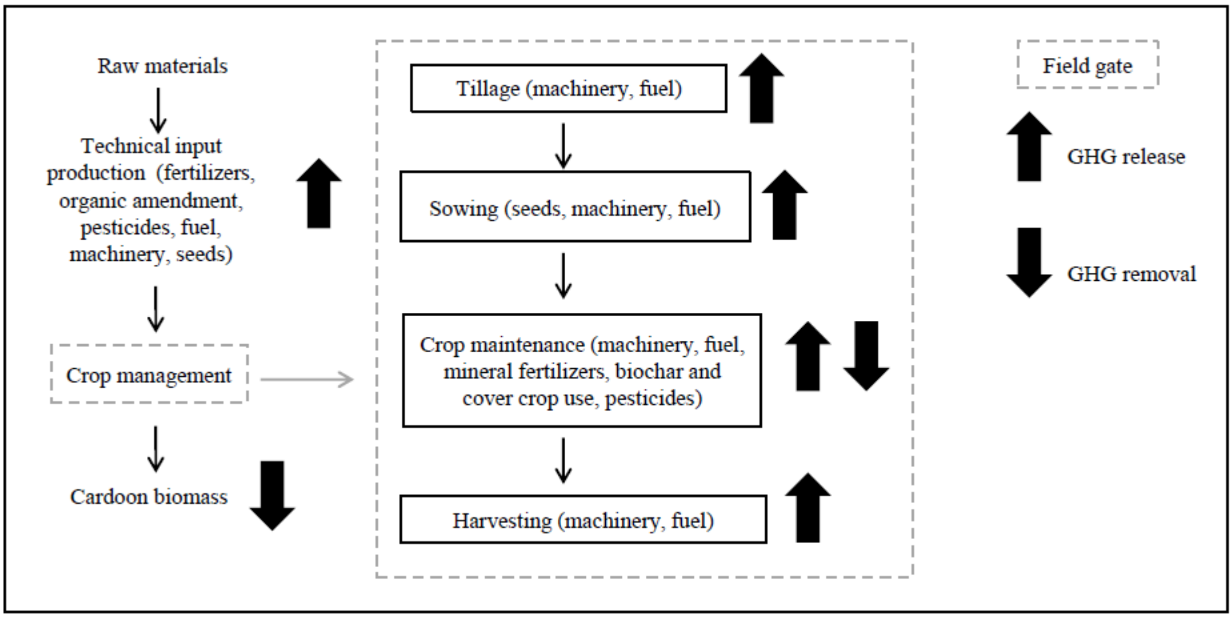

Fig. 1. The system boundary of the analysis

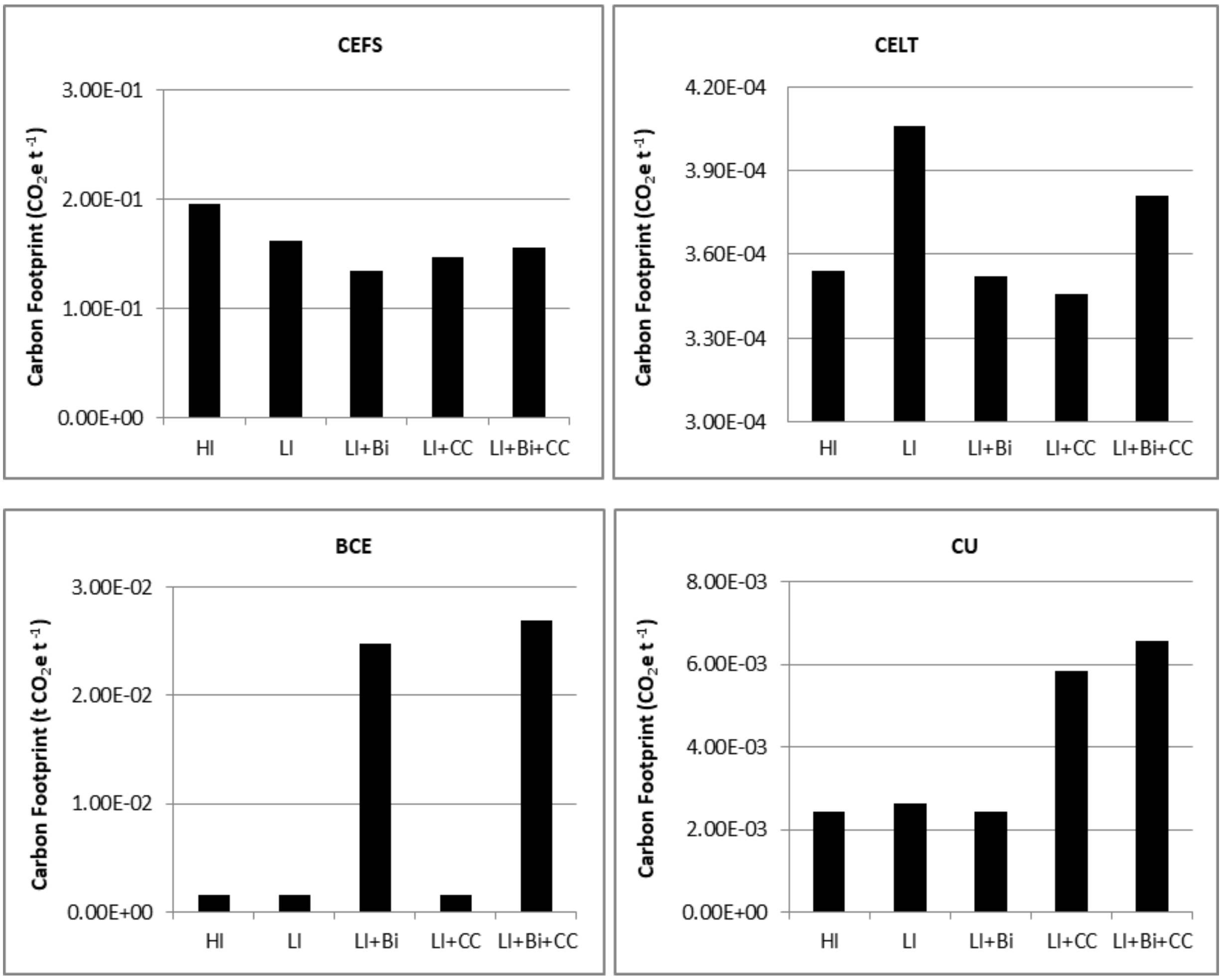

Fig. 2. Carbon Footprint (t CO_2e t $^{-1}$ cardoon biomass) of impact categories responsible for GHG fluxes (CEFS, Carbon Emission from Fossil Sources; BCE, Biogenic Carbon Emissions; CELT, Carbon Emission from Land Transformation; and CU, Carbon Uptake) due to five fertilization patterns (HI, High Input; LI, Low Input; LI + Bi, Low Input + Biochar; LI+CC, Low Input+ Cover Crop; LI + Bi + CC, Low Input + Biochar + Cover Crop).

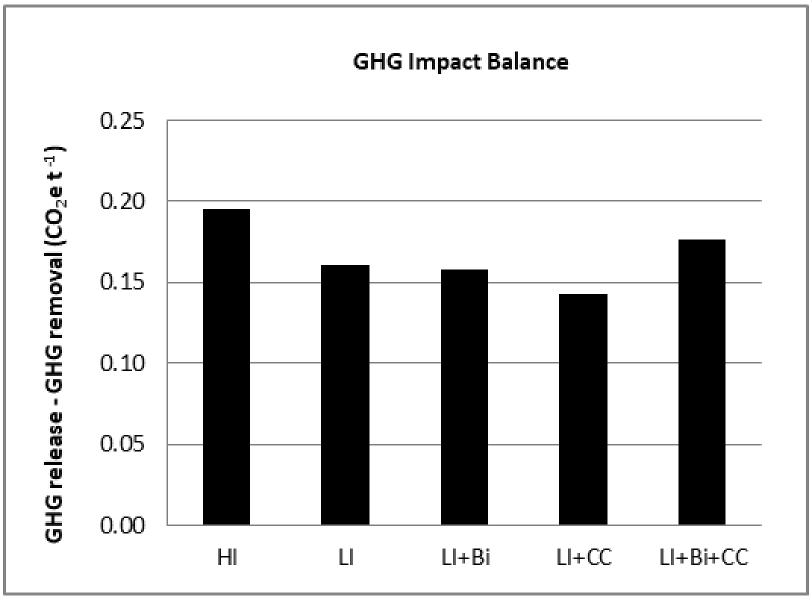

Fig. 3. Greenhouse gas (GHG) difference among impact categories for each treatment ((HI, High Input; LI, Low Input; LI + Bi, Low Input + Biochar; LI+CC, Low Input+ Cover Crop; LI + Bi + CC, Low Input + Biochar + Cover Crop) considering Carbon Emission from Fossil Sources (CEFS), Carbon Emission from Land Transformation (CELT), and Biogenic Carbon Emissions (BCE) categories as GHG release and Carbon Uptake (CU) category as GHG removal.

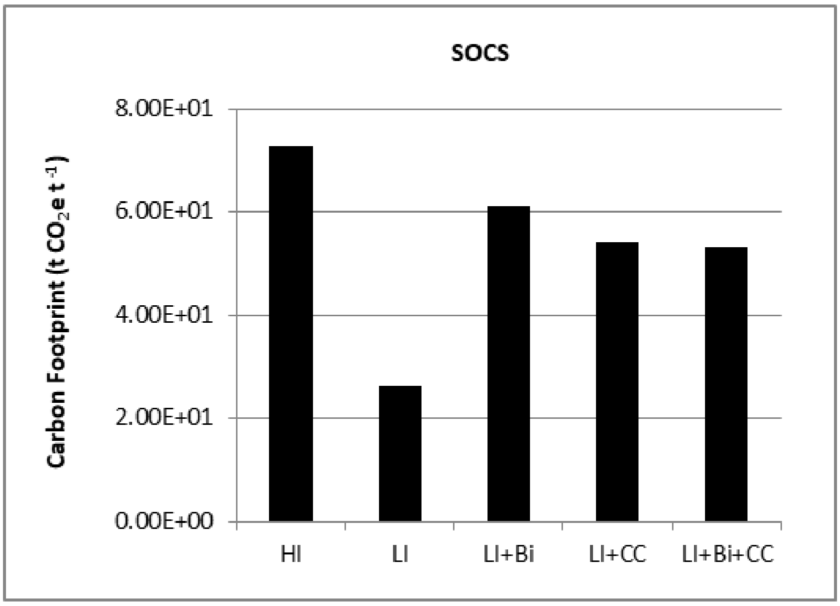

Fig. 4. Carbon Footprint (t CO_2e t $^{-1}$ cardoon biomass) of soil organic carbon storage (SOCS) category due to five fertilization patterns (HI, High Input; LI, Low Input; LI + Bi, Low Input + Biochar; LI+CC, Low Input+ Cover Crop; LI + Bi + CC, Low Input + Biochar + Cover Crop).