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

Carbon footprint and social carbon cost assessment in a perennial energy crop system: a comparison among fertilizer managements in a Mediterranean area

Authors

Stefania Solinas [a], Maria Teresa Tiloca [a], Paola A. Deligios [a], Marco Cossu [a]*, Luigi Ledda [a]

[a] Department of Agriculture, University of Sassari, Viale Italia 39, 07100 Sassari, Italy

E-mail address: ssolinas@uniss.it (S. Solinas); mtiloca@uniss.it (M.T. Tiloca); pdeli@uniss.it (P.A. Deligios); marcocossu@uniss.it (M. Cossu); lledda@uniss.it (L. Ledda).

* **Corresponding author**: Marco Cossu; e-mail: marcocossu@uniss.it; Full postal address: Department of Agriculture, University of Sassari, Viale Italia 39, 07100 Sassari, Italy.

Abstract

Agriculture as victim and perpetrator of climate change might contribute to reduce GreenHouse Gas (GHG) emissions and to foster carbon sequestration. Specifically, perennial energy crop systems might produce relevant benefits, both environmental and economic. This study is aimed to highlight the potential efficacy of various fertilizer managements in reducing GHG emissions and increasing economic value of carbon storage. Using two methodological approaches, namely Carbon Footprint (CF) and Social Carbon Cost (SCC), 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 a long term experiment on cardoon cultivation. The GHG release exceeded removal showing a range from 0.20 (HI) to 0.14 (LI + CC) t CO_2e per production unit and highlighting that LI + CC reduces GHG emissions and optimizes yield. As regards carbon sequestration, it was ranged from 72.7 (HI) to 26.2 (LI) t CO_2e t^{-1} of biomass. Furthermore, the combined use of biochar and cover crop showed no positive effects on C sequestration and reduction of GHG emissions unlike the performance of the single scenario. In fact, LI + Bi showed the highest value of C storage (61.1 t CO_2e 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 underlined that HI might produce the most convenient flow of benefits up to year 2050 (i.e. 9K US dollars for t CO_2e). Although a winning option among fertilizer managements was not found, identifying an optimal trade-off among productivity, GHG decrease and SCC value is a key factor so that an energy crop may guarantee food security, environmental and economic sustainability. Furthermore, this potential solution might allow to improve long-term crop system planning and land use in order to develop winning strategies to contrast climate change.

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

1. Introduction

Agriculture and climate change are characterized by tricky and controversial cause-effect linkage that might in turn affect environmental, economic and social spheres and make it difficult to exclude farming from strategy to combat climate change. In 2016, agriculture produced 431 Mt CO_2 equivalent (CO_2e) of GreenHouse Gas (GHG) emissions in European Union - 28 (EU-28) + Iceland (ISL). Specifically, CH_4, N_2O and CO_2 emitted by agriculture corresponded to 47.5%, 72.2%, and 0.3% of total EU-28 + ISL emissions, respectively (EEA, 2018).

On the other hand, there is evidence that seasonal changes in precipitation and temperature along with extreme weather events might jeopardize disposability of natural resources essential for agricultural production and yield level (FAO, 2017) even though different impacts are expected on the basis of geographical location (Altieri et al., 2015). For instance, the Mediterranean Basin might be considered as one of the most sensitive region 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 so varying conditions lead to the existence of two contrasting crop systems (i.e. irrigated and rainfed) requiring different selection and combination of managements that might foster the mitigation of a single GHG and, at the same time, enable the release of other GHGs in addition to influence Soil Organic Carbon (SOC) content.

Furthermore, the Mediterranean region is affected by a relevant land abandonment mainly due to a set of environmental and socio-economic drivers such as high risk of soil degradation, low inherent potential of crop productivity, agricultural policies, and globalization (Álvaro-Fuentes et al., 2018; Rodrigo-Comino et al., 2018). Agricultural land abandonment might lead to two opposite scenarios depending on environmental context and management: soil degradation or vegetation recovery that might have significant but opposite consequences for SOC content.

The perennial energy crop systems might be a winning alternative for Mediterranean agroecosystems whose soils are usually characterized by a low SOC content (Aguilera et al., 2013) and to avoid or, at least minimize the risk of land abandonment typically covered by food/feed crops (Cocco et al., 2014). Generally, the capacity of perennial crops to well suit to environmental stress conditions as well as low requirement of inputs and high biomass yield make these crops an ideal feedstock for bioenergy production in a changing climate context (Cosentino et al., 2018). However, perennial energy crops are not neutral to GHG emission production, even though they might be less harmful than the annual crops, especially because of their lower nitrogen (N) requirement and, thus the long run N management might be less intense (Drewer et al, 2012). Most of perennial crops might foster SOC storage due to their high capacity to sequester carbon compared to annual cropping system (Anderson-Teixeira et al., 2013). More specifically, deposition and decomposition of perennial plant material on the soil surface, the massive root growing and senescence process in the below ground might contribute to SOC content (Panda, 2016).

Although perennial energy crops are prone to foster an increase in SOC stock, this capacity is characterized by significant variability likely due, on the one hand, to complex interactions occurring between climate, soil texture and soil biota, and, on the other hand, to choice of soil management practices that should reduce its disturbance and destruction of aggregates (Tiemann and Grandy, 2014). For instance, fertilizer management might affect soil carbon content raising biomass production (Poeplau et al., 2018) but N fertilization generally causes GHG emissions by releasing ammonia, nitrate and N oxides that are potential harmful for climate and environment (Bashir et al, 2013). Furthermore, long-term N fertilizations demonstrated to have effects on soil carbon dynamics in terms of turnover and decomposition time (Neff et al., 2002).

Bioenergy crop cultivation entails environmental costs which among variation in SOC content resulting from land use change, crop type, and management practices may affect C dynamic as carbon dioxide (CO_2) emissions or sequestration (Ferchaud et al., 2015). Hence, in order to evaluate the potential use of a perennial energy cropping system as a winning strategy to combat climate change, it is necessary to assess its sustainability.

In the light of the above, the management of agricultural practices, in primis N fertilization along with crop planning and land use, should not be neglected in order to reduce harmful effects of farming on climate change, for instance in terms of soil C loss.

Life Cycle Assessment (LCA) might be an appropriate instrument in order to identify and quantify GHG emissions and more generally environmental impacts caused by a crop production system (Goglio et al., 2018). Furthermore, the application of LCA procedure to agricultural systems may not disregard two aspects: i) the most of environmental effects due to crop management adopted in a certain cropping system might occur throughout subsequent years; ii) thus, evaluations should be carried out for more years and they would need to be supported by findings arisen from long run field trials and by modeling that may provide information on C and N dynamics (Goglio et al., 2014).

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 occurring throughout its life cycle (Baldo et al., 2014). Therefore, the CF application to agriculture might enable to detect sources and sinks of GHGs caused by farming (Pandey and Agrawal, 2014).

This ability might support the identification of strategy to reduce GHG emissions and to strengthen GHG sinks (Adewale et al., 2016). Nevertheless, substantial differences in agricultural management between annual and perennial crops might play a crucial role in the reliability of CF calculation. For instance, the length of perennial crop life cycle should be considered since the crop performance and input requirements are related to the age of plants (Peter et al., 2017).

SOC is also a key factor for the most of ecosystem services including soil formation and retention, nutrient cycling and climate regulation (Francaviglia et al., 2017). An assessment of ecosystem services based on C storage might make it easier transparency in the debate on monetary and non-monetary trade-offs between various ecosystems services and associated beneficiaries when different development options are taken into account (Beaumont et al., 2014).

In this respect, 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 of C emissions in a given year (Greenstone et al., 2013). A complete appraisal of SCC should include effects on yields and the measures adopted by farmers towards these variations and changes to the value of ecosystem services (Smith and Braathen, 2015).

This study is aimed to evaluate the potential performances related to a perennial energy crop cultivation (cardoon) in a Mediterranean area in terms of ability to reduce GHG emissions and to foster SOC storage in the long run on the basis of different N managements. The analysis was implemented by combining two methodological approaches, CF and SCC, in order to highlight the potential efficacy of different fertilizer managements adopted in a long-term field experiment as response to climate change. The environmental performance of each fertilization scenario - expressed in CO_2e of GHGs released and removed - was also associated with a monetary estimation of ecosystem service related to SOC dynamic - to emphasize the relevance of agricultural practices in tackling climate change effects from both environmental and economic perspectives.

The dual methodological approach allows to provide more detailed information on individuating the fertilization pattern able to ensure higher productivity, higher carbon storage in the long run, and lower greenhouse gas emissions related to a perennial energy crop system. This assessment might be used as supporting for agricultural sector to which is currently required to guarantee food security, environmental and economic sustainability. This analysis may be considered a functional tool both in identifying a trade-off among productivity, GHG dynamics and the value of ecosystem service related to C sequestration and in developing innovative agricultural management suited for perennial crop and able to optimize crop system planning and land use in the long run.

2. Materials and methods

2.1. Study site

The study was carried out in Sardinia (Italy), an island located in the Mediterranean Basin that showed a subtropical dry-summer climate, also known as Mediterranean climate (Belda et al., 2014) and already described by Kottek et al. (2006) as a climate characterized by a hot-dry summer, namely with average temperature in the warmest month above 22°C and mild, wet winters. In Sardinia, most of annual rainfall is concentrated in fall and winter showing values ranges between 500 mm along the southern coast and 1300 mm in the mountainous areas. The mean annual temperature is also affected by distance from coastline; in fact, the value ranges from 17°C in 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 crop residual biomass energetic exploitation (De Menna et al., 2018) or for energy crop systems introduction (Ledda et al., 2013). In fact, the economic crisis that local agricultural and livestock activities are tackling in the island is exacerbating abandonment phenomenon in the productive areas or is leading to conversion of arable land into grasslands also in areas served by irrigation infrastructures (Solinas et al., 2015). In this context, local biomass production or the development of energy crop systems might contribute to minimize the risk of land abandonment and provide to farmers new opportunities for an additional income.

2.2. Experimental site

A field trial was conducted on a 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 the northwest of Sardinia (Lat. 41°N, Long. 9°E, 81 m a.s.l.). Cardoon may be considered one of the most promising energy perennial crops in the Mediterranean context since its adaptability to water and soil stress conditions do not enable to undermine the biomass production (Deligios et al., 2017). Throughout the trial, the average precipitation value was equal to 363 mm and the mean maximum and minimum temperature values were 22°C and 12°C, respectively. At the experimental site, soil is classified as a sandy-clay-loam soil showing 66% sand, 19% clay and 15% silt. At the beginning of experiment soil samples from a depth of 0-40 cm were collected and analyzed before applying fertilization treatments. The soil samples showed a total C content, total N content and soil organic matter equal to 49 g kg $^{-1}$, 1.8 g kg $^{-1}$ and 31 g kg $^{-1}$, respectively.

2.3. Experimental design

Before starting of trial (2014-2015), cardoon was cultivated for seven consecutive years in the same site. This species represents an opportunity for the Sardinian region, where the poor competitiveness of some food/feed crops (e.g. cereals) might lead to structural farming changes towards bioenergy production that might be a valid alternative to avoid land abandonment phenomenon. In order to optimize SOC storage, the duration of field trial may be considered an added value useful to detect the SOC trend in the long run and the effects resulting from crop continuity.

The cardoon removal was necessary since, after several years, the crop showed a physiological decline of production. Therefore, in 2014 the residual biomass of previous multi-year cultivation was incorporated into the soil before proceeding with the new cardoon planting. This activity, that most likely fostered a raise of SOC potentially available for the next crop, was the starting point to set up the experimental design and the different N fertilization managements.

As regards the first aspect, 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, the second aspect, were selected with the aim of paying attention to the possible N and C supply provided by each considered management. Specifically, two conventional patterns, based on high and low N input (HI and LI, respectively), were considered in order to guarantee continuity to the previous cardoon cultivation that was characterized by the same N managements. Three alternative N fertilizations - based on biochar (Bi) use, cover crop (CC) cultivation and the combination between both previous managements (CC + Bi) - were set up in order to evaluate their potentiality in terms of mineral fertilizer saving, SOC storage increasing, yield level optimizing, and environmental sustainability of perennial energy crop system on the whole. Furthermore, since neither crop residues (cardoon and cover crop) nor weeds were incorporated throughout the experimental trial, all three alternative scenarios were supported by the same mineral N supply characterizing LI (i.e. LI + Bi, LI + CC and LI + Bi + CC) (Table 1).

Table 1

The use of biochar obtained from thermochemical conversion of biomass (i.e. pyrolysis) might potentially affect the soil physical and chemical properties enhancing its fertility and therefore fostering crop growth (Tan et al., 2017). Since the purpose of cardoon biomass is energy production, the biochar application to soil might enable to offset the carbon amount removed by biomass harvesting. Specifically, biochar - obtained from a slow pyrolysis process using rapeseed straw as feedstock - was applied (10 t ha $^{-1}$) only once at the beginning of trial (November 2014) and it was incorporated into the soil to a depth of 10 cm. In this study, biochar was included in the form of C content obtained from feedstock pyrolysis (i.e. 71.34 wt %) on the basis of what reported by to 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 inter-row spaces to a depth of 5 cm. The legume choice as cover crop was due to its capacity to provide an additional source of N and C, especially by N fixation activity and residue production, respectively. In fact, the latter were not removed or incorporated into the soil year after year in order to make easier the litter development and potentially to reduce mineral fertilizer addition. The latest scenario, namely the combination between biochar and cover crop, was focused on highlighting both the effect on SOC content compared to the single management and if this combination showed a synergic effect or not.

2.4. Functional unit, system boundaries and data collection

The multifunctionality of agricultural systems allows to identify the functional unit, namely land management, financial and productive functions (Nemecek et al., 2011). In general, the choice depends on the objective of study, the typology of environmental impact evaluated, and the kind of process under consideration (Notarnicola et al., 2015). As reported by (ISO) 14040 (2006), the main purpose of a functional unit is to provide a reference to which inputs and outputs are connected. In the light of the above, and considering that the goal of this analysis was to estimate the environmental and economic effects of different fertilizer managements regarding both SOC variation and crop yield optimization, the productive function was considered the most appropriate functional unit. Specifically, it was expressed in ton 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 system. Specifically, the system boundary considered in this investigation included, for each fertilizer management, the whole life cycle of cardoon cultivation from raw material acquisition of inputs to farm gate (i.e. crop harvesting) (Figure 1). Hence, the LCA analysis 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 in order to support subsequent steps (i.e. impact assessment and interpretation). The quantification of inventory, namely the material and resource flow to and from environment within system boundaries, should be methodologically sound, complete and unbiased (Sauer, 2012). Therefore, the inventory of agricultural practices throughout the three years of trial was based on primary data collected at the experimental site specifically regarding agricultural machineries, fuel consumption, type and application rate of mineral fertilizers, pesticides and organic amendment.

Figure 1

During the cardoon life cycle, direct field measurements (i.e. yield and SOC content), physicochemical analysis on some soil samples, and climatic data detection (e.g. temperature and precipitation) were carried out allowing to apply various models (see paragraph 2.5) useful to assess GHG emissions resulting from the different agricultural managements.

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

Given that the crop under consideration, namely cardoon, was completely devoted to biomass production for energy purpose, no allocation of impacts was necessary in this evaluation.

2.5. Calculation methodology

Different tools were applied in this study in order to improve the accuracy of results since their performance is mainly based on primary data related to soil physicochemical properties, climatic parameters, crop management, and yield. Basically, the use of more models enabled to better understand the effects due to the different fertilization patterns in terms of CO_2e produced or avoided and thus to provide more detailed information on the GHG fluxes in terms of potential environmental and monetized damage.

2.5.1. Fertilizer and amendment emissions

The main nitrogen field emissions caused by each management (i.e. ammonia (NH_3) and nitrous oxide (N_2O) in air, and nitrate in water (NO_3^-) were included in the analysis using Estimation of Fertilizers Emissions Software (EFE-So) (2015). It uses as reference the model developed by Brentrup et al. (2000), which allows to obtain more accurate emission values since it needs various site-specific data in order to contextualize the fertilizer application and the possible loss. This model considers the difference between the supplied N and the absorbed one and needs some information on fertilizer type, soil characteristics, climate context (e.g. air temperature during distribution, summer and winter precipitation), and the N content in crop harvested and co-products (Schmidt Rivera et al., 2017).

According to Brentrup et al. (2000), the N emissions are affected by different parameters. For instance, average air temperature, infiltration rate, time between distribution and incorporation, precipitation, radiation, and wind speed are necessary to evaluate NH_3 volatilization due to an organic fertilizer. In the case of mineral fertilizer, NH_3 loss mainly depends on the ammonium or urea content of mineral fertilizer, the climatic conditions, and soil properties. The complexity of interaction existing between soil and climate factors and the variability characterizing the crop system management makes it difficult to assess N_2O emission. Nevertheless, the model takes as emission factor for N_2O the default value proposed by IPCC (2006). Finally, NO_3^- loss was reported by Brentrup et al. (2000) as nitrate leaching whose rate 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 drainage water rate, respectively). As regards the atmospheric N deposition, the value includes in the EFE-So model was estimated on the basis of what reported by Markaki et al. (2010) regarding annual deposition fluxes of nitrogen in different sites of the Mediterranean region including Sardinia.

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

$$CO_2\text{-C Emission} = M \times EF \quad (1)$$

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

As regards the LI + Bi management, the reduction of N_2O emissions caused by biochar application to soil was computed on the basis of Eq. (2) (Wolf et al., 2010):

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

where EN is the annual avoided soil N_2O emissions; RN is a reduction factor equal to 25%; and Ab is land amended by biochar. This computation was carried out only regarding the first crop year since the soil N_2O fluxes generally show a decrease over time albeit these results are highly variable depending on the complexity of the interactions between organic amendment and soil along with different experimental set-up, soil properties, and conditions (Agegnehu et al., 2016; Borchard et al., 2019).

The carbon addition to soil in the form of biochar may be responsible for the so-called priming effect (Zimmerman et al., 2011; Singh and Cowie, 2014), namely a short term change (increasing/positive or decreasing/negative) in the mineralization rate of soil organic matter following to the addition of exogenous organic substrates (Kuzyakova et al., 2000). Therefore, biochar application might affect CO_2 dynamic since in the short term its labile carbon fraction may trigger the microbial activity that, in turn would increase mineralization process (positive priming effect) whereas in the long term biochar may stimulate the physical protection mechanisms (sorption and aggregation) of organic carbon on amendment surface (negative priming effect) (Maestrini et al., 2015; Sagrillo et al., 2015). In the light of the above, this study included the possible changes in soil CO_2 emissions due to biochar addition according to Maestrini et al., (2015) who provided a quantification on soil carbon loss in the short term (3% of C added by organic amendment) due to biochar priming effect whereas no specific value was indicated on the long term because of the variability that the potential factors affecting priming effect might show (e.g. repeated addition of biochar, seasonal variations of soil temperature and moisture).

Losses of phosphorous were not reported for any fertilizer management since they were considered negligible in the observed site.

2.5.2. Some details of the LI + CC scenario

This study considered N and C supplies due to legume biomass in the LI + CC management. In particular, N content was included regarding above- and below-ground biomass produced by cover crop on the basis of two specific values (2% and 1.65%, respectively) evaluated during a field trial carried out in the same geographical area considered in this study.

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

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

where SOM is Soil Organic Matter (Mg ha $^{-1}$); DM is Dry Matter (Mg ha $^{-1}$); and A is total ashes (percentage of DM) 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 on the basis of the Eq. (4) (Prybil, 2010):

$$SOC = SOM / 2 \quad (4)$$

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

As regards the LI + Bi + CC scenario, the analysis considered the N and C values were estimated on the basis of the same references used for the single management, namely LI + Bi and LI + CC.

2.5.3. Pesticide emissions

The on-field emissions due to the pesticide distribution were calculated using PestLCI 2.0 model in order to provide an assessment of pesticide fraction that crosses the technosphere - environment boundary and thus disperses in the environment (air, surface water and ground water). The technosphere may be considered as a "field box" which includes the arable field borders characterized by soil up to 1 m depth and the air column up to 100 m above the soil (Dijkman et al., 2012). The model rationale, as according to Birkved and Haushild (2006), considers two subsequent steps of emission within the technosphere box and that are responsible for the pesticide fate: a primary and a secondary distribution.

The primary process regards the part of pesticide that is deposited on crop (e.g. leaves) and on soil surface or drifted away by wind immediately after application. The secondary distribution mainly regards the pesticide fate on field since the fraction of active ingredient may be deposited on crops, on topsoil, and subsoil where it may be undergone to different processes. The pesticide fraction on plants might be subject to volatilization, uptake or degradation. On the topsoil, the processes affecting the pesticide fate are mainly volatilization, biodegradation and runoff in surface water by rainfall or it might reach the subsoil and thus ground water by leaching.

The model enables to calculate the emissions due to primary and secondary distribution through constructing a scenario that includes site-specific information such as type of pesticide, application method and month, crop, climatic conditions, and soil type. To date, PestLCI 2.0 is applicable for European conditions; therefore it includes various climate and soil site -specific data representative of European regions and approximately a hundred active ingredients (Moraleda Melero, 2018).

2.5.4. Carbon Footprint

This is a methodological tool useful to quantify the total amount of GHGs that a product or a service disperse into the environment during its lifetime (i.e. from raw material production to final use of product) expressed as CO_2e (Ramachandra and Mahapatra, 2015). In this study, the CF assessment - that was carried out according to the LCA approach - enabled to quantify GHG emissions due to agricultural managements applied to cardoon cultivation throughout its life cycle.

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

In this regard, GHG Protocol provides guidance to facilitate the management of agricultural GHG fluxes considering mechanical (i.e. equipment or machinery operated in farm) and non-mechanical (e.g. soil amendment and management, burning of crop residues, and land use change) emission sources along with upstream ones (e.g. raw material extraction and fertilizer, pesticide and feed production) in order to foster eco-friendly production practices (Russell, 2011). GHG Protocol refers to the Intergovernmental Panel on Climate Change (IPCC) calculation approach to quantify GHG fluxes from an activity (WRI and WBCSD, 2011b). In fact, the different GHG emissions related to a life cycle product may be expressed as CO_2e using a characterization factor, the so-called Global Warming Potential (GWP), developed by IPCC regarding the impact category of

Climate Change (JRC, 2007). GWP enables to compare the potential climate impact of various gases using as reference unit the GWP value of CO_2 that is equal to 1 related to three different time horizons, namely 20, 50 and 500 years (WRI and WBCSD, 2011a). In this study, CO_2e, that is CF relating to a process was calculated on the basis of the following Eq. (5) (Morawicki and Hager, 2014):

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

where CO_2e is basically CF due to a certain gas (kg CO_2e); emission factor (i) is amount of GHG produced per unit of activity rate; 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 used as time reference 100 years to calculate GHG emission impacts related to a product system and a distinction regarding impact categories is made between: Carbon Emission from Fossil Sources (CEFS), Biogenic Carbon Emissions (BCE), Carbon Emission from Land Transformation (CELT), and Carbon Uptake (CU) (PRé, 2018).

The CEFS category concerns emissions arisen from fossil sources (e.g. carbon from fuels), BCE is related to biogenic sources (i.e. carbon from living organism or materials derived from biological matter), CELT regards emissions due to conversion from one land use category to another, and the last category, namely CU is CO_2 quantity storing in plants and trees as they grow (WRI and WBCSD, 2011b). Since the analysis 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, as the value of impact categories provided by SimaPro are expressed in kg CO_2e ha $^{-1}$, namely on land basis, whereas this study adopted the production functional unit (i.e. ton of biomass produced by cardoon) the outputs were converted according to 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 land basis (t CO_2e /ha); and Y is yield of a given crop (t/ha).

These results enable to calculate a CF balance between the GHG loss and saving (i.e. CEFS, BCE, CELT, and CU impact category, respectively) in order to identify the best and the worst environmental performance related to each fertilizer pattern adopted for cardoon cultivation throughout the experimental trial.

2.5.5. Uncertainty analysis of Carbon Footprint

A Monte Carlo analysis was implemented to assess the uncertainty of the CF findings. The analysis was also performed to test possible significant difference in terms of CF per product unit comparing the environmental impacts due to each fertilizer management applied to cardoon cultivation. SimaPro 8.0.4.30 software was employed to run the Monte Carlo simulation (Goedkoop et al., 2013a, b). It was used at a 95% confidence interval and 1000 reiterations were performed.

2.5.6. Soil carbon storage

In view of the complexity of C dynamics and GHG fluxes due to the different N fertilizers, an additional impact category, namely Soil Organic Carbon Storage (SOCS) was considered in order to provide a more detailed framework for GHG exchanges related to a perennial energy crop system. In fact, this information might be useful to make it easier the identification of environmental impacts in the long run and to support crop system planning and land use.

The accounting for soil C change due to agricultural systems and land use was a thorny issue within LCA context and, as a consequence in CF of products, mainly because of the lack of an unique procedure, although attempts to consider SOC dynamics may be implemented depending on availability of quality data and on C cycle model performances (Goglio et al., 2015).

In this study, carbon storage were estimated using the Rothamsted carbon model (RothC) Ver. 26.3 that was specifically developed for the turnover of SOC in non-waterlogged topsoil also including 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 type of information, namely i) climatic data (i.e. monthly values of air temperature (°C), rainfall (mm), and evapotranspiration (mm)); ii) soil data, including clay content (%), Inert Organic Carbon (IOM), initial SOC stock (t C ha $^{-1}$), depth of the considered soil layer (cm); 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 SOC for each agricultural management adopted for cardoon cultivation on the basis of site-specific characteristic of soil and climatic conditions and regarding a time reference equal to 100 years, namely the same time horizon used by SimaPro to assess the CEFS, BCE, CELT, and CU impact categories.

All inputs were included in RothC as average value of experimental trial lifetime. In the model, SOC is divided in four active pools and a small amount of IOM that is resistant to decomposition process. The four active compartments in which crop C input to soil is allocated are: 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 to partition the C input between DPM and RPM on the basis of its provenance, namely crops, grassland or forests. These two pools are undergone a decomposition resulting in CO_2, BIO and HUM depending on the soil clay content. The decomposition process of one active compartment occurs by first-order decay at 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 by Eq. (7) (Gónzalez-Molina et al., 2017):

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

where Y is material quantity of a pool that decomposes in a certain month (t C ha $^{-1}$); Y_0 is 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 regarding temperature, moisture, and soil cover, respectively; and t is 1/12 to express k as monthly decomposition rate. As regards IOM, it was calculated on the basis of 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 to equilibrium, namely the C input was adjusted in order to the modelled SOC value matches the measured starting one in the experimental device (Kaonga and Coleman, 2008). The SOC stock used in the RothC model was calculated according to Eq. (9) (Lozano-García et al., 2017):

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

where -SOC-S is Soil Organic Carbon Stock (mg ha $^{-1}$); SOC is Soil Organic Carbon (g kg $^{-1}$); BD is Bulk Density (mg m $^{-3}$); d is thickness (cm); and δ_2 mm is fractional percentage (%) of gravel greater than 2 mm size.

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

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

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

The obtained values of CO_2 were expressed in CO_2e on the basis of GWP relative to CO_2 for 100 years, namely 1 (Forster et al., 2007). These outputs are basically CF of the SOCS impact category for each cardoon management that, as the previous impact categories, were also referred to production functional unit in order to make it easier the comparison among the different adopted fertilization patterns in terms of potential C storage.

2.5.7. Social Carbon Cost

This term basically represents the economic cost due to an additional ton of CO_2 emissions or its equivalent that, in more detail, 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 aimed to better understand changes in agricultural production level, human health, and the value of ecosystem services arisen from climate change (IGW, 2016). On the contrary, it may be also considered a measure of avoided damage in case of emission reduction, that is a socio-economic benefit.

In this study, the SCC computation was based on an assessment of benefits and cost, that is increase and decrease 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). The SCC evaluation for different time horizons is implemented through three integrated assessment models running with several input assumptions and simulating the possible connection between GHG emissions to climate change related to a baseline scenario and to different options to assess future damages arisen from an additional ton of CO_2 released or avoided (Rose et al., 2014).

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

In the light of the above, in this study the monetized estimation of ecosystem service due to SOCS was intended as an attempt to underscore strengths and weaknesses of different fertilization managements adopted for the cardoon cultivation in the long run as a strategy for tackling climate change challenge. The SCC computation was performed multiplying the SOCS values of each fertilizer scenario in year 2050 resulting from the RothC model application times the SCC in 2050, namely \$ 79 (2016 dollars per metric ton CO_2e) with the 3% discount rate (Niemi, 2018). In order to perform this appraisal, the SOCS values were previously converted in ton CO_2e for 100-year time horizon as reported at the end of subparagraph 2.5.6.

3. Results

3.1. Carbon footprint of GHG fluxes from fertilizer managements

The description of CF outputs are focused on effects (t CO_2e t $^{-1}$ of cardoon biomass) resulting from peculiarity of each fertilizer management, i.e. use of different N doses between HI and LI, application of biochar, cultivation of legume cover crop and their combination. This choice was due to the fact that the mechanical operations and production inputs basically did not change among scenarios except in a few cases reported from time to time. For this reasons, their environmental impacts were not considered since the CF values did not demonstrate difference when expressed on land basis and especially to maintain the consistency with the objective of this study, that is to evaluate the potential reduction of GHG emissions and SOC storage resulting from different N fertilizer managements applied to cardoon.

The environmental performance of the five scenarios showed a significant variability both inter and intra impact categories (Figure 2). In fact, in the first case CF ranged from 0.00041 to 0.2 t CO_2e per production unit in CELT (LI) and CEFS (HI), respectively. The detected difference between HI and LI - CEFS exceeded CELT little more than 480 times - may be further emphasized considering the CEFS value of all fertilization patterns on the whole. In fact, the CF of the CEFS category was 432, 40, and 14 times greater than CELT, CU, and BCE, respectively. As regards CU, all values reported hereinafter should be considered reliable in absolute terms since this impact category is related to a GHG saving whereas the others to a GHG loss.

Figure 2

Moving on to the effect of each management within the single impact category, HI demonstrated the worst environmental performance in CEFS exceeding the second worst management (LI) of 21%. The observed gap between HI and LI was mainly due to the different impact of agricultural inputs, especially fertilizer one. In fact, the mechanical operations were basically not showed changes except in the LI + Bi, LI + CC, and LI + Bi + CC managements where two additional agricultural inputs were introduced, namely biochar and legume that were sowed or distributed, and subsequently buried. Furthermore, the greater amount of N fertilizer (i.e. urea, top dressing) used in HI was to be the main responsibility for the worst environmental performance of this management within CEFS showing a double contribution compared to the second most impacting scenario (LI). This was 20% and 10% more impacting than LI + Bi and LI + CC, respectively albeit the last two categories included two additional mechanical operations and two additional production inputs, namely biochar, its distribution and burying (LI + Bi), and legume seed, its sowing and burying (LI + CC).

These further processes showed an impact contribution not significant for the CEFS category, since they were equal to less than 1% and little more than 3% for LI + Bi and LI + CC, respectively. As regards LI + Bi, its better environmental performance than the LI scenario was mostly likely caused by the biochar effect in terms of N emissions reduction on fertilizers used in the short term, i.e. urea and diammonium phosphate throughout the first growing season. In fact, the environmental incidence demonstrated by these fertilizers used along with biochar was 22% less than the impacts due to the same fertilizers in LI management.

The better environmental performance of LI + CC than the LI one was not due to N and C contribution provided by legume cultivation (little more than 3% of the CEFS category) but rather from the higher level of average produced 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 production basis. The CF difference between Li + CC and Li + Bi (i.e. 0.01 t CO2e t $^{-1}$ of cardoon biomass) in favor of the latter was most likely due to the biochar effect on GHG emissions coming from fertilizers since the mechanical operations (i.e. distribution and burying of biochar, and sowing and burying of legume) showed the same environmental impact (0.0007 t CO2e t $^{-1}$ of cardoon biomass).

Finally, the LI + Bi + CC management demonstrate an antagonist effect due to the interaction between biochar and cover crop that was responsible for an impact 13% lower than the sum of their single effects. Nevertheless, LI + Bi + CC showed a CF contribution per production unit greater than LI + CC and LI + Bi (6% and 15%, respectively) because of higher biomass yield obtained from LI + CC and LI + Bi compared to the cardoon yield of the LI + Bi + CC scenario.

As regards the CELT category, it showed the lowest CF contribution compared to the other three impact categories most likely due to the lack of a real land use change that de facto avoided a relevant production of GHG emissions. Nevertheless, impacts detected within the CELT category might be associated to CO_2 and N_2O emissions generated during agricultural land use and following a change in farm management in accordance with what reported by the GHG Protocol that emphasized the role of agricultural activity as sources and sink of CO_2 (WRI and WBCSD, 2011b).

Actually, the analysis showed similar CF values on land basis among scenarios identifying as key impact factor the same upstream processes, such as seed production that includes a land transformation. Although the CF difference per production unit was minimum (i.e. from 0.00035 to 0.00041 t CO_2e t $^{-1}$ of biomass for LI + CC and LI, respectively) it resulted from the different cardoon yield. The latter showed the lowest value for LI and thus the worst environment-friendly scenario in contrast to LI + CC that producing 18% more than LI, reduced GHG emission loss by 85% compared to the conventional management. Furthermore, the combination between biochar and legume cover crop showed as detected in the CEFS category, an antagonist effect even though the LI + Bi + CC environmental performance was worse than the LI + Bi and LI + CC one (8% and 10%, respectively). The LI + Bi and HI scenarios showed a very similar CF per production unit (about 0.0003 t CO_2e t $^{-1}$ biomass) and they were higher than LI + CC (2% and 3%, respectively) stressing that the potential effect of cover crop on increasing cardoon yield was most likely responsible for the lowest CF in the CELT category.

Moving on to the last two impact categories more specifically related to C dynamic, namely BCE and CU, both of them showed an intermediate order of magnitude compared to CEFS and CELT, but LI + Bi + CC was the worst and the best scenario in BCE and CU (0.03 and 0.01 t CO_2e t $^{-1}$ of biomass, respectively). This result might suggest that the use of organic material in addition to mineral fertilizers might act both as source and sink of C. The environmental performance of these alternative fertilization patterns might depend on how the additional inputs were included in the overall crop management. Specifically, the sum of CF resulted from LI + Bi + CC and LI + Bi represented 92% of the BCE category on the whole underlining the relevance of biochar as C source. In fact, the C contribution provided by biochar application exceeds the 90% in both scenarios. Although the harvesting of cover crop did not perform, the C supply resulting from legume was not relevant (7%) for BCE. The CF difference detected between LI + Bi + CC and LI + Bi (i.e. 0.002 t CO_2e t $^{-1}$ of biomass in favor of the first) was basically due to the simultaneous use of biochar and legume cover crop. Their combination showed a synergistic effect compared to the CF resulting from the single action of organic amendment and legume crop. This is because the CF of LI + Bi + CC exceeded of 9% the sum of CF arisen from single managements. In other words, in the LI + Bi + CC scenario biochar and legume crop might have acted so that strengthen the effect of one or both of them. LI + CC showed an environmental performance 17 times lower than the worst one further stressing the relevance of biochar in the BCE category. The two conventional managements, namely LI and HI showed the best contribution in terms of avoided CO_2 emission (6%) compared to the most impacting scenario because of the absence of the additional organic C source.

Among the four considered impact categories, CU is the one related to GHG emission removal, since it concerns the C storage in a crop throughout its life cycle. As mentioned above, the best environmental-friendly scenario within the CU category was the worst one in BCE. LI + Bi + CC showed a conflicting performance due to the combination between biochar and legume cover crop. The highest CF value showed by this scenario might be due to the synergistic effect that also in the CU category was caused by the interaction occurring between biochar and legume cover crop. Their simultaneous action - higher of 16% than the sum of single management effect - might have caused a greater C storage in the biomass than that occurred in the LI + Bi and LI + CC.

Furthermore, LI + Bi + CC showed a CF value greater than LI + CC and LI + Bi (13% and 170%, respectively) suggesting that the positive environmental performance in LI + Bi +CC might be due to the synergistic effect caused by biochar and legume able to enhance the C uptake from cardoon and legume cover crop. On the contrary, the lowest CF of LI + Bi underlines that the potential action of biochar on cardoon ability to store carbon might not have been adequate to guarantee a good performance.

Some agricultural inputs in addition to crop yield have had different impacts on the CU category on the basis of management. For instance, the cardoon seeds for sowing was showed a contribution of about 10% on average for the LI + Bi, LI + CC, and LI + Bi + CC categories. The mineral fertilizers used in LI + Bi had an incidence equal to 13% on CU whereas the C contribution of legume cover crop achieved 30% in LI + CC. The same inputs were equal to 5% and 29%, respectively in LI + Bi + CC. The environmental performance of LI in terms of CO_2 uptake was 8% higher than LI + Bi most likely since the yield of LI was greater than the LI + Bi one. The quantity of cardoon biomass might also have had a role in the CF value arisen from HI and LI managements. In fact, LI – that showed a lower average biomass production than HI - had the best environmental performance in CU category, with a contribution equal to little more than 7% compared to HI. Despite the use of a double N dose (HI vs LI), the N fertilizer incidence on CU was almost 2 times greater in HI scenario.

A more in-depth analysis of the single CF balances for each agricultural scenario (i.e. the comparison between GHG release and GHG removal) allowed to better understand the effects of fertilizer patterns on GHG fluxes (Fig. 3). All CF balances were in favor of GHG emission loss showing a range from 0.20 (HI) to 0.14 (LI + CC) t CO_2e per production unit. The other three balances were equal to 81%, 82%, and 90%, for LI + Bi, LI and LI + Bi + CC, respectively compared to the worst balance. Basically, the inclusion of a cover crop (i.e. legume) in a perennial energy system (cardoon) might be a winning option in terms of GHG emission reduction and yield optimization.

Figure 3

The second positive trade-off between GHG balance and crop production is provided by LI + Bi. Although this management showed the same GHG balance of LI (0.16 CO_2e t^{-1} of biomass), the cardoon yield achieved in case of biochar application is greater than the LI one (7.96 vs 6.91 t ha^{-1} on average). On the contrary, the balance of LI + Bi + CC showed the second worst value highlighting that the combination of biochar and cover crop did not foster a reduction of GHG emissions although the cardoon yield achieved with this management was between the biomass production provided by 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 pair-to-pair comparison among the fertilizer managements in terms of CF per production unit. The analysis showed (Table 2) that in CEFS three differences were not statistically significant for α = 0.05.

Table 2

Specifically, the analysis highlighted that HI, namely the most impacting scenario was significantly higher compared to the others. As regards the best eco-friendly scenario (i.e. LI + Bi) only the difference with LI was statistically significant. LI showed the worst result in CELT even though its performance was highly significantly different only compared to HI and LI + Bi + CC. Moving on to the BCE category, all comparisons demonstrated significant differences except for HI vs LI + CC. Finally, in CU the most impacting scenario, namely LI + Bi + CC were significant different compared to the second one (i.e. LI + CC) only for $\alpha = 0.10$ whereas it was highly significant relative to the other three managements.

3.3. Soil organic carbon stock from fertilizer managements

The analysis was completed considering the SOCS category aimed to detect change in SOC storage resulting from the implementation of the five fertilization patters. Although SOCS category was expressed in t CO_2e t $^{-1}$ cardoon biomass as the previous four categories, its environmental impact was arisen from direct measures detected in field throughout the experimental trial (Figure 4).

SOCS ranged from 72.7 (HI) and 26.2 (LI) t CO_2e per production unit highlighting that the two conventional managements were the best and the worst performance with a gap equal to little less 3 times in favor of the HI management. Its performance might be due to the greater N dose applied throughout the cardoon life cycle that in turn, most likely fostered a higher yield than BI. The three alternative scenarios 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) closer to the best management than the worst one underlining that the scenarios characterized by biochar, cover crop and their combination fostered SOCS. The simultaneous use of biochar and legume demonstrated an antagonistic effect in SOCS since the sum of biochar and cover crop effects was 2 times greater than the value of their interaction. LI + Bi showed an environmental performance better than LI + CC and LI + Bi + CC (13% and 15%, respectively) highlighting that the application of biochar might have caused a higher effect than the other two fertilizer managements in terms of soil carbon storage.

Figure 4

3.4. Social carbon cost from fertilizer managements

A monetary valuation was performed in order to estimate which fertilizer management might generate the greatest benefits flow related to the ecosystem service of SOCS. Results highlighted that HI might produce the most convenient flow in terms of benefits up to 2050 (Table 3). Specifically, these benefits might amount to about 9K US dollars for t CO_2e. On the contrary, the less benefits arisen from the other managements might entail the presence of a social cost (opportunity cost in terms of loss of benefits regarding the most favorable scenario). The LI management showed the highest SCC equal to about 5K US dollars for 1t CO_2e whereas the other three managements showed a SCC ranging between 1K (LI + Bi) and 2K (LI + Bi + CC) US dollars for 1t CO_2e.

Table 3

4. Discussion

4.1. Carbon footprint implications resulting from agricultural managements

The application of different assessment tools (e.g. simulation models for fertilizer and pesticide emissions and for carbon stock) based on site specific data collected throughout the experimental trial might be considered an attempt to mitigate the main weakness of LCA. As reported by Curran et al., (2013), this methodological approach is not free from limitations that might affect the accuracy of the results even though ISO developed a general framework to implement a LCA analysis. These limitations are mainly due to the lack of a well-defined procedure to encompass and estimate important site-specific factors, (e.g. soil quality and soil carbon sequestration) closely linked to both farm management and environmental performance of a crop system within the LCA context (Garrigues et al., 2012; Petersen et al., 2013). Although model use does not guarantee a high level of certainty - that, is in contrast attributed to direct observations - they are generally able to capture variability and soil and climatic interaction (Goglio et al., 2015). Therefore, in this study both models that field data were used in order to strengthen the reliability of the LCA analysis.

Our study emphasized that the double role played by farming, namely as victim and perpetrator of climate change, makes it difficult to identify a winning contribution able to guarantee an optimal food, energy, and environment security. Since it is virtually unthinkable to develop a set of measures valid worldwide, an assessment of farming practices would be necessary for single 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 results suggested that the optimization of agricultural practices, such as fertilization may have a positive effect on GHG fluxes in the long run. Furthermore, the management of a perennial energy crop is generally not devoid of environmental impacts whose value may often depend on fertilizer use (Wagner and Lewandowski, 2017; Fernando et al., 2018). This was consistent with our findings that basically identified the field emissions resulting from the fertilizer application as one of the main responsible for environmental performance of cardoon. A similar result were detected by Razza et al., (2017) although they considered a single value of GWP - without distinction between impact categories - due to a cardoon cultivation in Sardinia.

The findings stressed that the characterization of a perennial energy crop system in terms of agricultural management and land allocation should be considered a staging post to better support farmers' decisions aimed also to reduce GHG loss and to foster soil C storage in the long run. Specifically, the choice of farming management and land use might arise from a convenient trade-off between yield and environmental performance of energy crop such as to satisfy the present and future needs in terms of food and energy security, and environmental sustainability. This study might be a useful support in choosing the best option since the results enabled to highlight strengths and weaknesses of each fertilization pattern and their effects on GHG dynamics.

The conventional managements, namely HI and LI provided two completely different opportunities for trade-off most likely due to the different N doses (in HI it was twice LI). However, performances provided by scenarios considered in this study might be associated with the cardoon capacity to adapt to the Mediterranean climate and to uptake nutrient from deep soil layers by a well-developed root system that enables to increase soil organic matter and nutrient availability in the long term (Mauromicale et al., 2014). The adoption of a high mineral N dose for a perennial energy crop might be winning in terms of yield (HI production was about one ton more than LI) if the planning of energy crop system is aimed to use arable land that might be abandoned due to lack of an advantageous production purpose. On the other hand, the results related to LI might be a good trade-off in order to exploitation of lands unsuitable for food production where perennial biomass production - occasionally harvested for energy production purpose- might foster the restoration of vegetation and thus C storage in the long run. The introduction of a perennial energy crop in the farming planning - regardless of which management was applied - might prove to be more advantageous than annual ones. In fact, a perennial crop is generally characterized by lower input costs (e.g. tillage is carried out only once) and their long-lived roots might develop positive relationships with root symbionts fostering the nutrient availability and consequently reducing the fertilizer use (López-Bellido et al., 2014).

The potential trade-off regarding the conventional managements (i.e. HI and LI) might be achieved through adoption of innovative technologies. For instance, the application of precision agricultural practices might foster reduction of GHG emissions and enhancement of SOC storage since they may lower intensity of tillage practices, N supply and production input rates on the whole, and fuels consumption for implementing mechanical operations. Specifically, these innovative practices might optimize small amount of production inputs such as N fertilizers that if used in excessive quantity or on a wide agricultural area, might have relevant negative impacts in terms of environmental and economic sustainability (e.g. poor profit margin on land basis).

Furthermore, the exploitation of natural resources (e.g. water) or the distribution of N fertilizers potentially prone to leaching process might foster or exacerbate possible pollution phenomenon in particular 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 regarding variable rate application of nutrients, irrigation, pesticides and planting/seeding, and controlled traffic farming and machine guidance) based on high-tech equipment may optimize the use of inputs in space and time on the basis of site-specific characteristics causing a potential reduction of GHG emissions and an improvement of farm economic and production performance of the conventional managements.

These innovative practices were not contrary to the three alternative scenarios (i.e. LI + Bi, LI + CC and LI + Bi + CC) even though their effects must be interpreted with caution since their potential benefits in terms of GHG dynamics and SOCS might be affected by site-specific characteristics such as climate, soil type, and farming practices. The scientific studies regarding legume cover crop effects on GHG flux show a wide variability of results strongly connected to experimental context and that thus makes it difficult to associate our findings with a specific point of view. The LI + CC scenario confirmed the potential of legume cover crop to offset the cardoon N requirement reducing GHG release and guarantee the highest cardoon yield. This result was consistent with evidences from Daryanto et al. (2018) who underlined that the synchronization the timing of nutrient availability provided by cover crop and nutrient requirement from the main crop is strategic to ensure high productivity due to optimization of microbial activity. On the other hand, legume cultivation was able to foster a good SOC storage even though its contribution was not equally high compared to HI likely because of mineralization process of additional biomass produced by cover crop.

As regards LI + Bi management the positive effects in terms of C storage might be due to recalcitrant C content of biochar that interfering with C and N dynamics implemented by microbial community may foster the maintaining of a stable C pool into the soil. This condition might also have contributed to the achievement of a good yield level - just below HI and LI + CC - and to reduce the GHG loss. On the other hand, if the reliability of results showed by the previous managements lower on the basis of the reference context, this is even more correct for Li + Bi scenario.

The potential biochar effect on soil CO_2 emissions is still a tricky and poor understood issue because of the considerable uncertainty characterizing this aspect both in time (in the short and long term) and space (at the lab and field scale) (Fidel et al., 2018). In fact, CO_2 emissions showed different behaviors (increasing, decreasing and unchanged dynamics) as a result of organic amendment addition mainly due to the complicated interactions occurring between biochar feedstock and its physicochemical properties, application rate and mode (i.e. alone or combined with mineral and organic fertilizers), soil type, nutrient availability, microbial activity and crop management (e.g. incorporation of residual biomass, rate and time of mineral fertilizer application) (Kuppusamy et al., 2016; Shen et al., 2017). These complex interactions entail variable effects also on other GHG emissions from soil such as N_2O. In the light of the above, the performance showed by LI + Bi + CC is even more difficult to interpret since most likely affected by interaction between biochar and legume cover crop that is hard to well specify. In this sense, an attempt was performed analyzing results into each impact category in order to identify a synergistic effect.

Summarizing and considering all fertilization patterns, a winning option may not be identified since LI + CC is the management that maximized cardoon productivity and minimized GHG emission loss but C storage is maximized by HI in the long run.

The availability of site-specific data and specific information on crop system planning and land use are key factors to use a mixed methodological approaches useful to identify which fertilizer managements 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 results, the framework adopted in this study may be replicated in order to assess the potential of other perennial energy crop systems and innovative agricultural managements to achieve the best trade-off between production level and environmental sustainability.

4.2. Carbon economic effectiveness of agricultural managements

SCC reflects an economic measure related to negative externalities in 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 whose cost represents the benefit reduction in monetary terms switching from the HI management - i.e., the management that mostly contributes to C accumulation in the soil - to the other managements applied to cardoon cultivation. Basically, this cost is not sustained by farmer because, in absence of compensatory mechanisms of regulations, the responsibility is paid by collectivity in the long run (Havranek et al., 2015).

This represents a critical point because farmer is deprived of responsibility and he does not pay any direct cost for SOCS reduction in order to pursue his own economic objectives (basically the profit maximization). Furthermore, costs would not be equally distributed in collectivity since we would expect that the less developed countries bear more costs. In fact, the richer and more developed countries are more able to compensate costs related to negative externalities with greater benefits generated by higher agricultural productivity and profitability. This disparity implies that 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 to limit disparities is represented to introduction of normative mechanism based on property rights regarding C production able to foster internalization of these costs into the agricultural management chosen by farmer. In other terms, introduction of tax schemes or other mechanisms might transfer costs from society to farmers who produce externalities in order to create an incentive (disincentive) for increasing (decreasing) C storage. In this way, costs related to SOCS reduction become an “internal” costs for farmer as well as the other production costs and C storage would be an economic variable that concurs along with the other typical economic variables in defining farmers choices (aimed to increase productivity and thus to profit maximization).

In conclusion, more empirical evidence need to be found in order to extend this analysis to the managements of other perennial energy crop system and in geographical context different from Mediterranean one, to estimate the costs related to GHG emissions in the long run and to develop effective tools for “internalizing” SCC into farmer’s decision.

5. Conclusions

This study estimates the potential performances of the cardoon crop system in terms of GHG reduction and SOC storage in the long run by the combination between two methodological approaches (i.e. CF and SCC) on the basis of different fertilizer managements. The results stress the difficulty of denoting a unique fertilization patterns in terms of GHG production and SOC storage. In fact, the HI scenario showed the worst GHG balance and the best SOCS whereas LI + CC demonstrated a good performance in terms of GHG emission reduction and yield followed by LI + Bi. As regards LI + Bi + CC, the combined use of biochar and cover crop fostered neither the C sequestration nor the decrease of GHG emissions.

The monetary estimation related to ecosystem service provided by soil C storage highlighted the benefit reduction switching from the HI management to the others and the need to “internalize” into farmer’s choices SCC so that to handle environmental externality. This means that C storage should be considered on the same level of the other agricultural input costs in order to optimize practices also considering cardoon production and environmental performance.

More generally, a winning option able to guarantee an optimal level of food security, environmental and economic sustainability was not found. This study emphasizes the importance of finding a trade-off among productivity, GHG dynamics, and the economic value of ecosystem service (e.g. C sequestration) provided by agricultural management of a perennial energy crop. This potential solution would allow to optimize long-term crop system planning and land use in order to develop effective measures to tackle climate change.

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

Adewale, C., Higgins, S., Granatstein, D., Stöckle, C.O., Carlson, B.R., Zaher, U.E., Carpenter-Boggs, L., 2016. Identifying hotspots in the carbon footprint of a small scale organic vegetable farm. Agric. Sys. 149, 112–121. https://doi.org/10.1016/j.agsy.2016.09.004.

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.

Aguilera, E., Lassaletta, L., Gattinger, A., Gimeno, B.S., 2013. Managing soil carbon for climate change mitigation and adaptation in Mediterranean cropping systems: A meta-analysis. Agric. Ecosyst. Environ. 168, 25–36. http://dx.doi.org/10.1016/j.agee.2013.02.003.

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.

Altieri, M.A., Nicholls C.I., Henao A., Lana M.A., 2015. Agroecology and the design of climate change resilient farming systems. Agron. Sustain. Dev. 35, 869–890. https://doi.org/10.1007/s13593-015-0285-2.

Álvaro-Fuentes, J., Plaza-Bonilla, D., Arrúe, J. L., Bielsa, A., Cantero-Martínez, C., 2018. Soil Carbon Dynamics Under Different Land Uses in Dryland Mediterranean Conditions, in: Muñoz, M.Á., Zornoza, R. (Eds.), Soil Management and Climate Change: Effects on Organic Carbon, Nitrogen Dynamics, and Greenhouse Gas Emissions. Academic press, pp. 39–52.

Anderson-Teixeira, K.J., Masters, M.D., Black, C.K., Zeri, M., Hussain, M.Z., Bernacchi, C.J. DeLucia, E.H., 2013. Altered Belowground Carbon Cycling Following Land-Use Change to Perennial Bioenergy Crops. Ecosystems 16, 508–520. https://doi.org/10.1007/s10021-012-9628-x.

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.

Bashir, M.T., Ali, S., Ghauri, M., Adris, A., Harun, R., 2013. Impact of excessive nitrogen fertilizers on the environment and associated mitigation strategies. Asian J. Microbiol. Biotechnol. Environ. Sci. 15, 213–221. DOI: 10.13140/RG.2.1.2754.7606.

Beaumont, N.J., Jones, L., Garbutt, A., Hansom, J.D., Toberman, M., 2014. The value of carbon sequestration and storage in coastal habitats. Estuar. Coast. Shelf Sci. 137, 32–40. https://doi.org/10.1016/j.ecss.2013.11.022.

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.

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.

Cocco, D., Deligios, P.A. Ledda, L., Sulas, L., Virdis, A., Carboni, G., 2014. LCA Study of Oleaginous Bioenergy Chains in a Mediterranean Environment. Energies 7, 6258–6281. https://doi.org/10.3390/en7106258.

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).

Cosentino, S.L., Scordia, D., Testa, G., Monti, A., Alexopoulou, E., Christou, M., 2018. The Importance of Perennial Grasses as a Feedstock for Bioenergy and Bioproducts., in: Alexopoulou, E. (Eds.), Perennial Grasses for Bioenergy and Bioproducts. Academic press, pp. 1–33.

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.

Drewer, J., Finch, J.W., Lloyd, C.R., Baggs, E.M., Skiba, A., 2012. How do soil emissions of N2O, CH4 and CO2 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.

FAO (Food and Agriculture Organization of the United Nations), 2017. Strategy on climate change. FAO, Rome.

Ferchaud, F., Vitte, G., Mary, B., 2016. Changes in soil carbon stocks under perennial and annual bioenergy crops. Glob. Change Biol. Bioenergy 8, 290–306. https://doi.org/10.1111/gcbb.12249.

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., Renzi, G., Ledda, L., Benedetti, A., 2017. Organic carbon pools and soil biological fertility are affected by land use intensity in Mediterranean ecosystems of Sardinia, Italy. Sci. Total Environ. 599–600, 789–796. https://doi.org/10.1016/j.scitotenv.2017.05.021.

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.

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., Grant, B.B., Smith, W.N., Desjardins, R.L., Worth, D.E., Zentner, R., Malhi, S.S., 2014. Impact of management strategies on the global warming potential at the cropping system level. Sci. Total Environ. 490, 921–933. https://doi.org/10.1016/j.scitotenv.2014.05.070.

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.

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.

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.

IPCC, 2006. IPCC Guidelines for National Greenhouse Gas Inventories, in: Egglestonne, H.S., Buendia, L., Miwa, K., Ngara, T., Tanabe, K. (Eds.), Prepared by the National Greenhouse Gas Inventories Programme. IGES, Japan.

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.

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.

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.

Neff, J.C., Townsend, A.R., Gleixner, G., Lehman, S.J., Turnbull, J., Bowman, W.D., 2002. Variable effects of nitrogen additions on the stability and turnover of soil carbon. Nature 419, 915–917. https://doi.org/10.1038/nature01136.

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.

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.

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.

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.

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.

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.

Poeplau, C., Zopf, D., Greiner, B., Geerts, R., Korvaar, H., Thumm, U., Don, A., Heidkamp, A., Flessa, H., 2018. Why does mineral fertilization increase soil carbon stocks in temperate grasslands? Agric. Ecosyst. Environ. 265, 144–155. https://doi.org/10.1016/j.agee.2018.06.003.

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.

Rodrigo-Comino, J., Martinez-Hernandez, C., Iserloh, T., Cerda, A., 2018. Contrasted Impact of Land Abandonment on Soil Erosion in Mediterranean Agriculture Fields. Pedosphere 28, 617–631. https://doi.org/10.1016/S1002-0160(17)60441-7.

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. Wprld Resources Institute. Washingtonne, 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.

Shen, Y., Zhu, L., Cheng, H., Yue, S., Li, S., 2017. Effects of biochar application on CO_2 Emissions from a cultivated soil under semiarid climate conditions in northwest China. Sustainability 9, 1–13. DOI: 10.3390/su9081482.

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., 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, S., Braathen, N., 2015. Monetary Carbon Values in Policy Appraisal: An Overview of Current Practice and Key Issues. OECD Environment Working Papers, No. 92, OECD Publishing, Paris. http://dx.doi.org/10.1787/5jrs8st3ngvh-en.

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.

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.

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).

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 scenario

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 mineral 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%)

Pair-to-pair 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 agricultural scenarios

	Discounted value (\$ tCO_2e^{-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

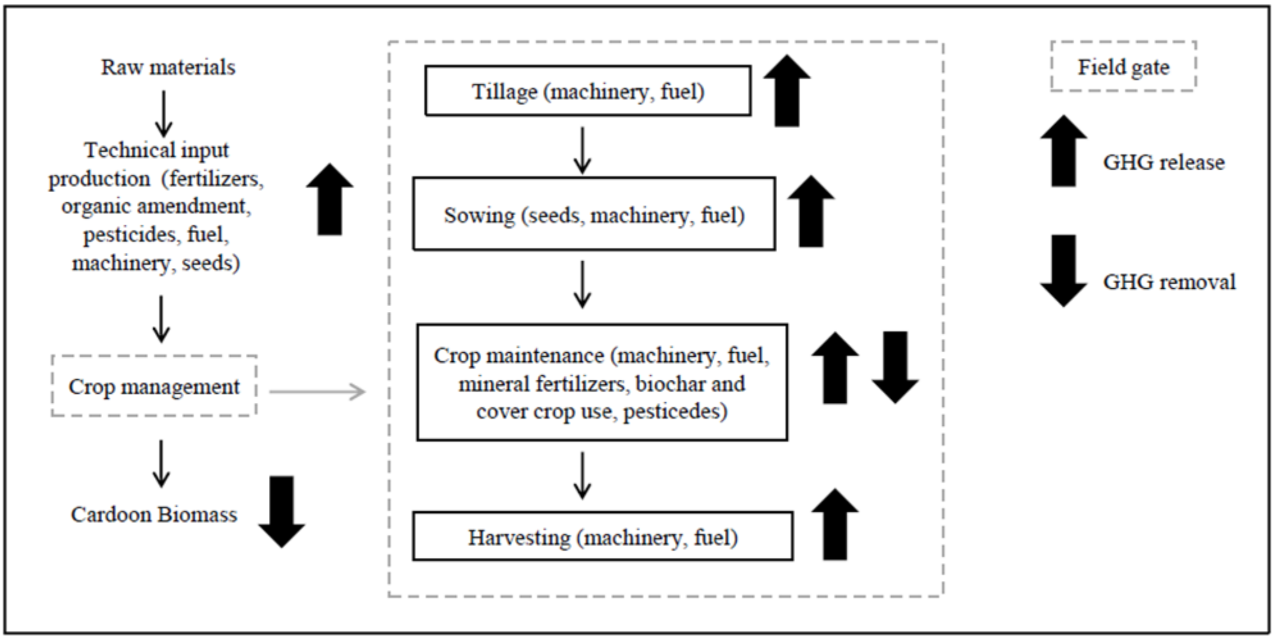

Fig. 1. The system boundary of the analysis

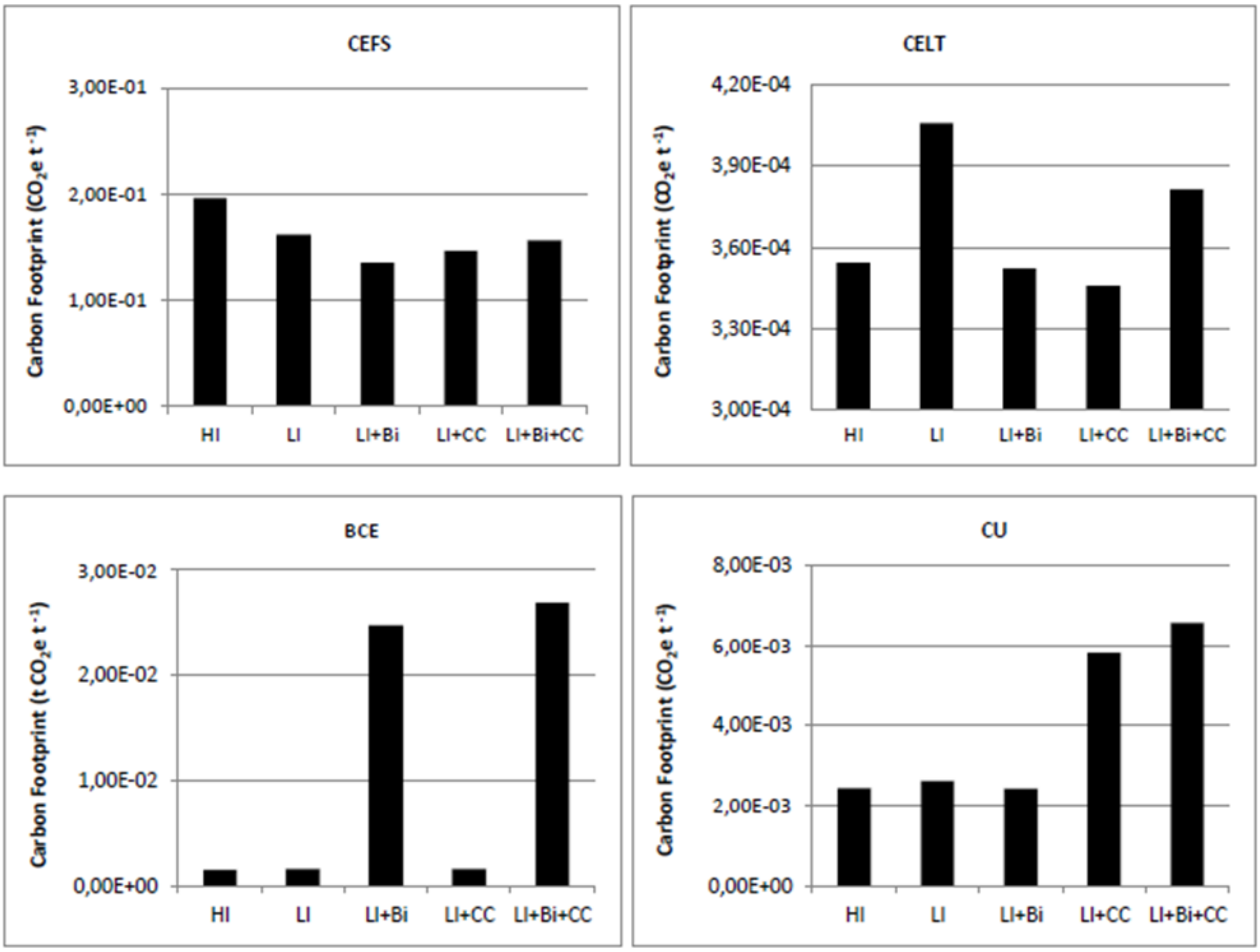

Fig. 2. Carbon Footprint (t CO_2e t $^{-1}$ cardoon biomass) of impact categories responsible for GHG fluxes (CEFS, Carbon Emission from Fossil Sources; BCE, Biogenic Carbon Emissions; CELT, Carbon Emission from Land Transformation; and CU, Carbon Uptake) due to five fertilization patterns (HI, High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

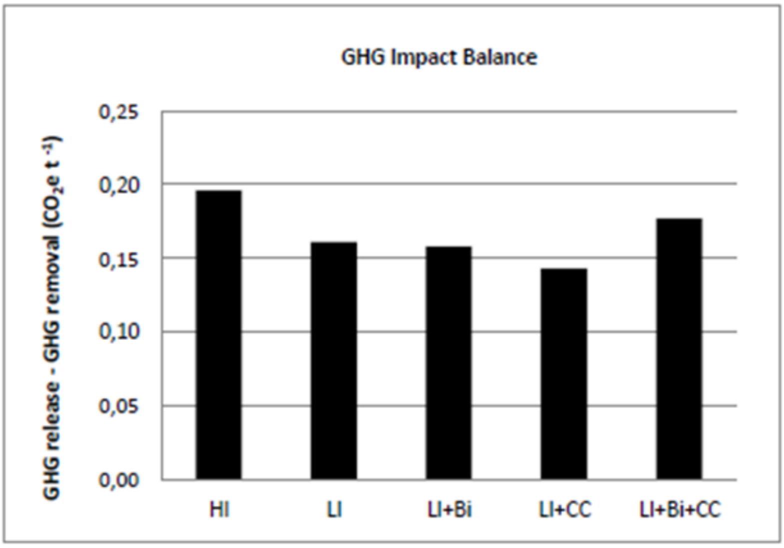

Fig. 3. Greenhouse gas (GHG) difference among impact categories for each management ((HI, High Input; LI, Low Input; LI + Bi, Low Input + Biochar; LI+CC, Low Input+ Cover Crop; LI + Bi + CC, Low Input + Biochar + Cover Crop) considering Carbon Emission from Fossil Sources (CEFS), Carbon Emission from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

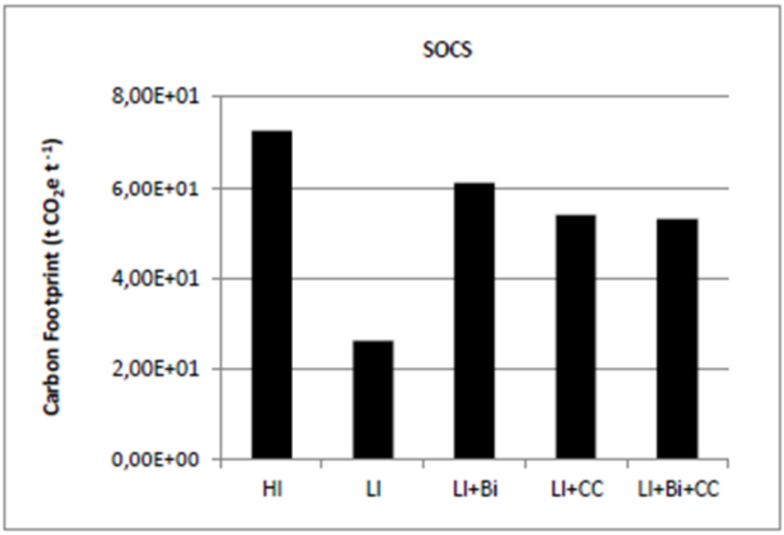

Fig. 4. Carbon Footprint (t CO_2e t $^{-1}$ cardoon biomass) of soil organic carbon storage (SOCS) category due to five fertilization patterns (HI, High Input; LI, Low Input; LI + Bi, Low Input + Biochar; LI+CC, Low Input+ Cover Crop; LI + Bi + CC, Low Input + Biochar + Cover Crop).

Declaration of interests

☒ The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

☐The authors declare the following financial interests/personal relationships which may be considered as potential competing interests:

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

Authors

Stefania Solinas [a], Maria Teresa Tiloca [a], Paola A. Deligios [a], Marco Cossu [a]*, Luigi Ledda [a]

[a] Department of Agriculture, University of Sassari, Viale Italia 39, 07100 Sassari, Italy

E-mail address: ssolinas@uniss.it (S. Solinas); mtiloca@uniss.it (M.T. Tiloca); pdeli@uniss.it (P.A. Deligios); marcocossu@uniss.it (M. Cossu); lledda@uniss.it (L. Ledda).

* Corresponding author: Marco Cossu; e-mail: marcocossu@uniss.it; Full postal address: Department of Agriculture, University of Sassari, Viale Italia 39, 07100 Sassari, Italy.

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):

$$CO_2\text{-C Emissions} = M \times EF \quad (1)$$

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):

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

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):

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

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):

$$CFY = CFA/Y \tag{6}$$

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):

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

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):

$$IOM = 0.049 \times SOC \times 1.139 \tag{8}$$

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):

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

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):

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

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_2e^{-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

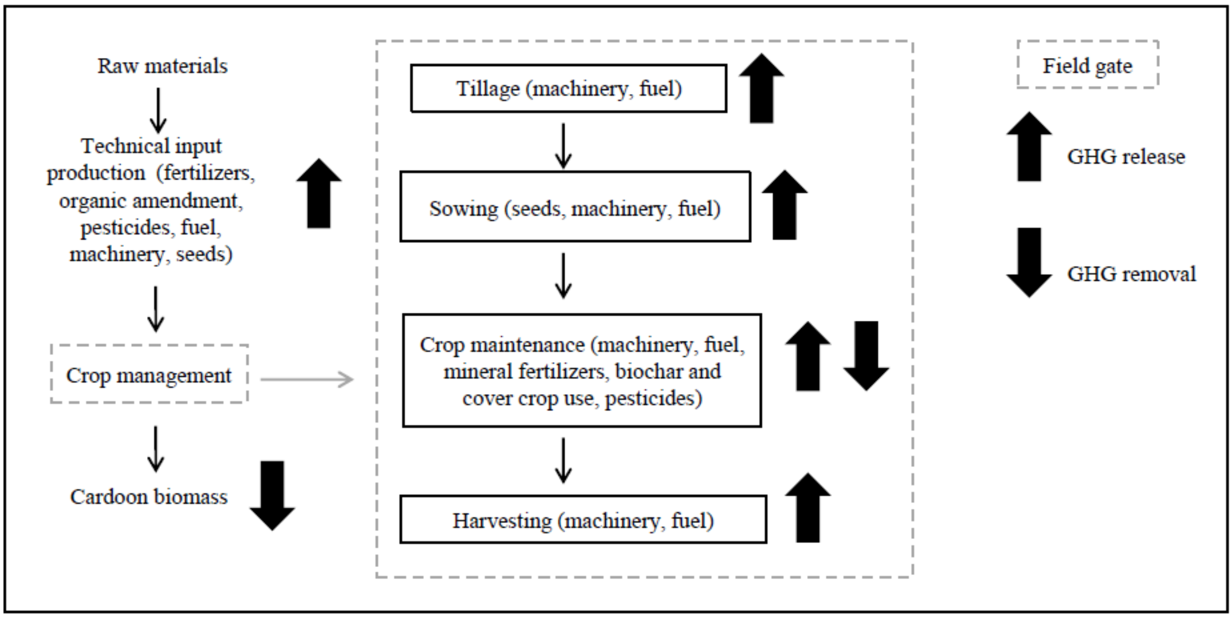

Fig. 1. The system boundary of the analysis

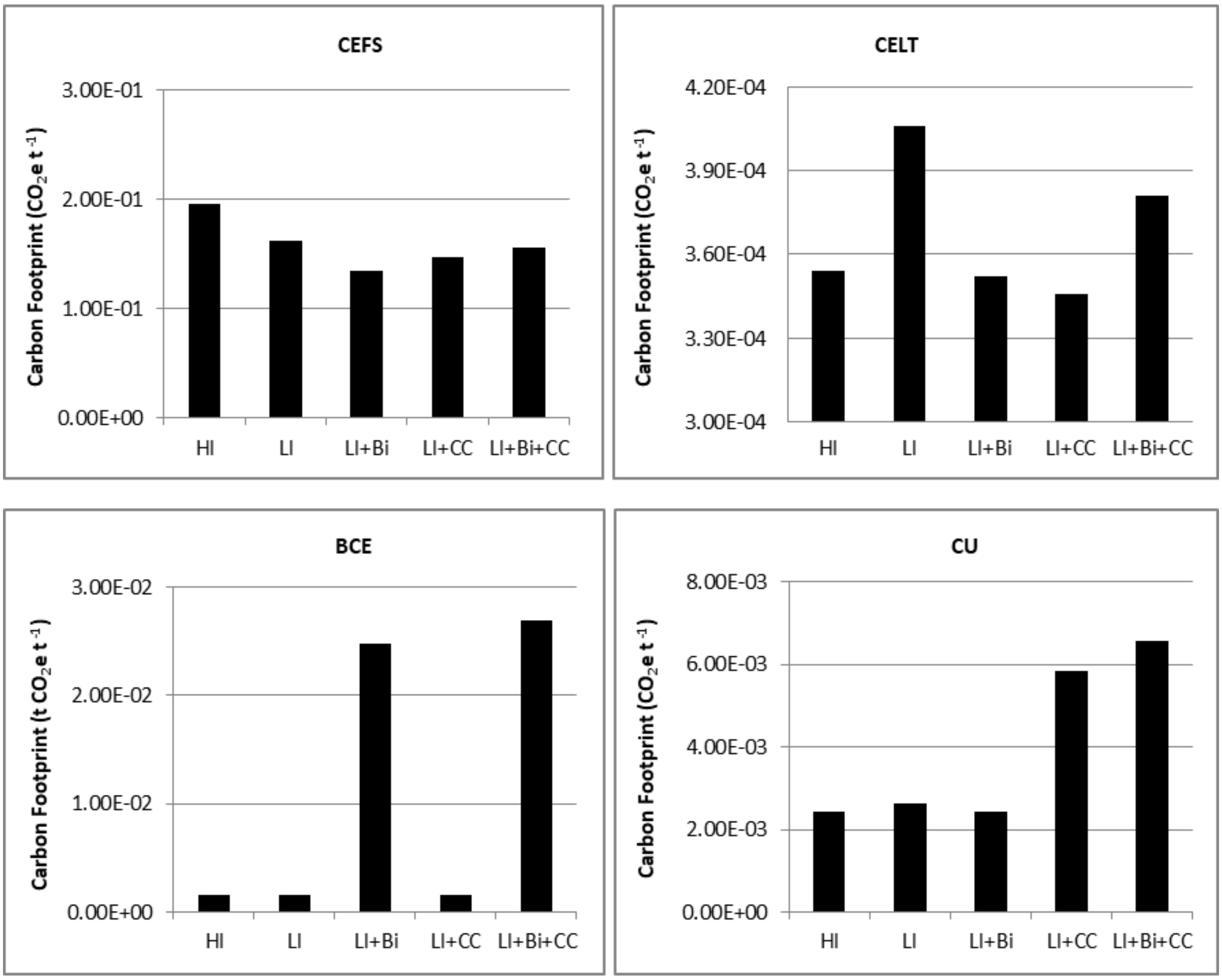

Fig. 2. Carbon Footprint (t CO_2e t $^{-1}$ cardoon biomass) of impact categories responsible for GHG fluxes (CEFS, Carbon Emission from Fossil Sources; BCE, Biogenic Carbon Emissions; CELT, Carbon Emission from Land Transformation; and CU, Carbon Uptake) due to five fertilization patterns (HI, High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

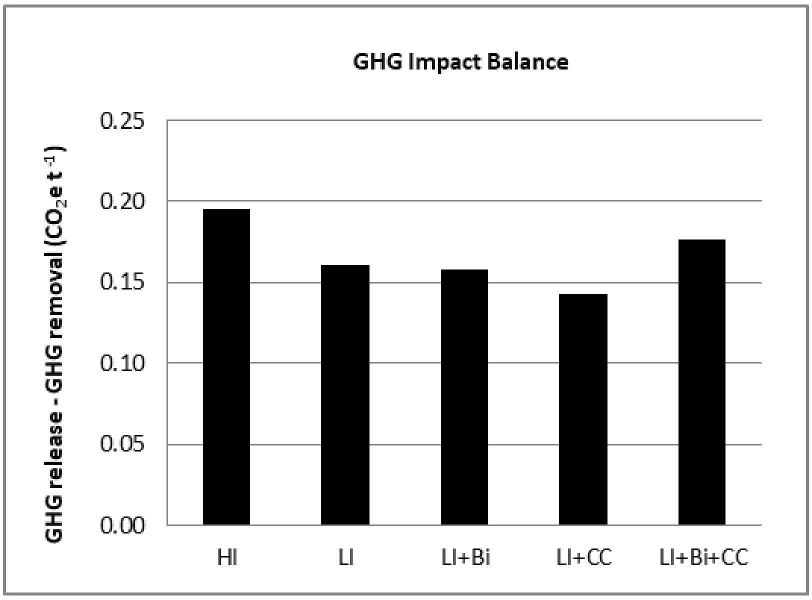

Fig. 3. Greenhouse gas (GHG) difference among impact categories for each treatment ((HI, High Input; LI, Low Input; LI + Bi, Low Input + Biochar; LI+CC, Low Input+ Cover Crop; LI + Bi + CC, Low Input + Biochar + Cover Crop) considering Carbon Emission from Fossil Sources (CEFS), Carbon Emission from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

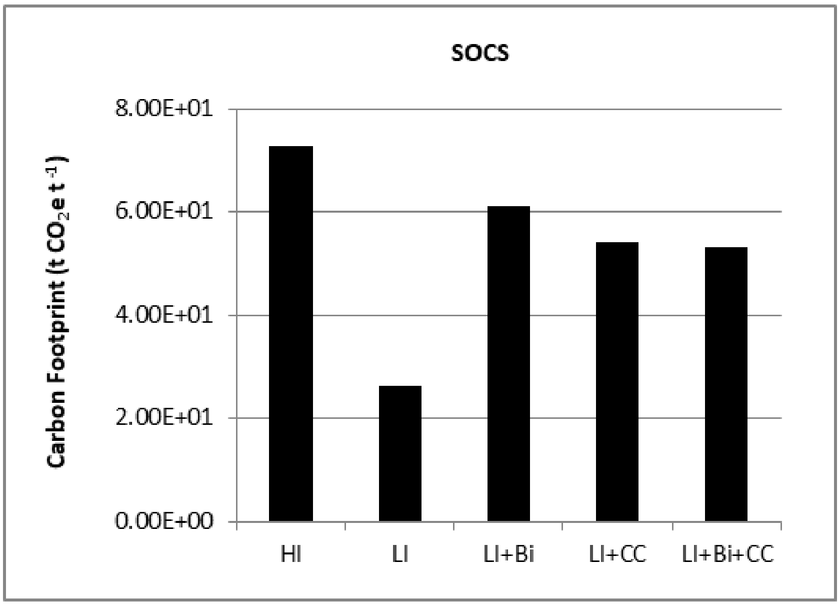

Fig. 4. Carbon Footprint (t CO_2e t $^{-1}$ cardoon biomass) of soil organic carbon storage (SOCS) category due to five fertilization patterns (HI, High Input; LI, Low Input; LI + Bi, Low Input + Biochar; LI+CC, Low Input+ Cover Crop; LI + Bi + CC, Low Input + Biochar + Cover Crop).

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

Authors

Stefania Solinas [a], Maria Teresa Tiloca [a], Paola A. Deligios [a], Marco Cossu [a]*, Luigi Ledda [a]

[a] Department of Agriculture, University of Sassari, Viale Italia 39, 07100 Sassari, Italy

E-mail address: ssolinas@uniss.it (S. Solinas); mtiloca@uniss.it (M.T. Tiloca); pdeli@uniss.it (P.A. Deligios); marcocossu@uniss.it (M. Cossu); lledda@uniss.it (L. Ledda).

* Corresponding author: Marco Cossu; e-mail: marcocossu@uniss.it; Full postal address: Department of Agriculture, University of Sassari, Viale Italia 39, 07100 Sassari, Italy.

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.

Commented [SS1]: Reviewer 1, answer 1.

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).

Commented [SS2]: Reviewer 1, answer 2.

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).

Commented [SS3]: Reviewer 1, answer 3.

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):

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

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):

$$IOM = 0.049 \times SOC \times 1.139 \tag{8}$$

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):

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

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):

$$1\ ton\ of\ soil\ C = 3.67 \times tons\ of\ CO_2 \tag{10}$$

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.

Commented [SS4]: Reviewer 1, answer 4.

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).

Commented [SS5]: Reviewer 2, answer 1.

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.

Commented [SS6]: Reviewer 2, answer 2.

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_2e^{-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

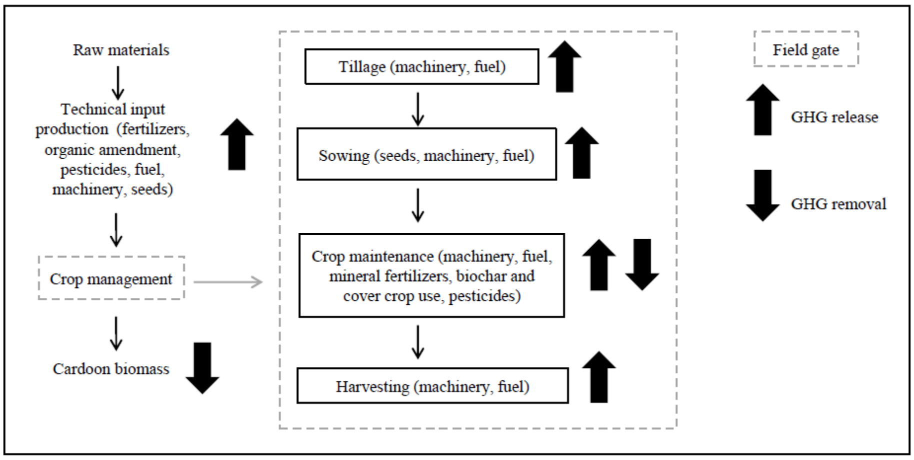

Fig. 1. The system boundary of the analysis

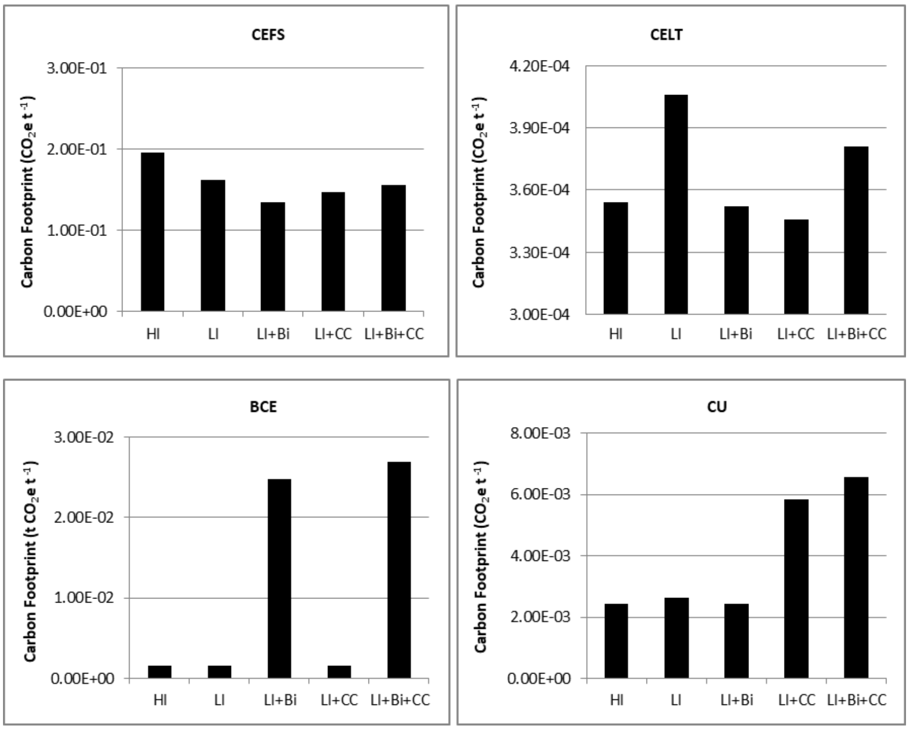

Fig. 2. Carbon Footprint (t CO_2e t $^{-1}$ cardoon biomass) of impact categories responsible for GHG fluxes (CEFS, Carbon Emission from Fossil Sources; BCE, Biogenic Carbon Emissions; CELT, Carbon Emission from Land Transformation; and CU, Carbon Uptake) due to five fertilization patterns (HI, High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

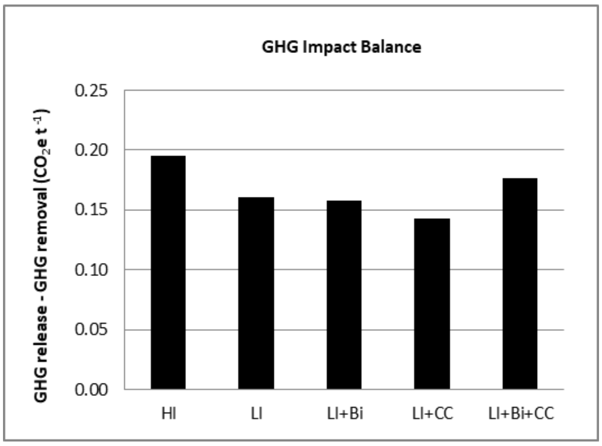

Fig. 3. Greenhouse gas (GHG) difference among impact categories for each treatment ((HI, High Input; LI, Low Input; LI + Bi, Low Input + Biochar; LI+CC, Low Input+ Cover Crop; LI + Bi + CC, Low Input + Biochar + Cover Crop) considering Carbon Emission from Fossil Sources (CEFS), Carbon Emission from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

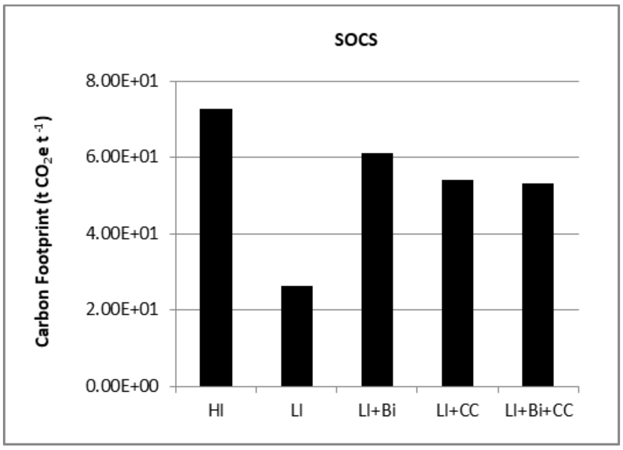

Fig. 4. Carbon Footprint (t CO_2e t $^{-1}$ cardoon biomass) of soil organic carbon storage (SOCS) category due to five fertilization patterns (HI, High Input; LI, Low Input; LI + Bi, Low Input + Biochar; LI+CC, Low Input+ Cover Crop; LI + Bi + CC, Low Input + Biochar + Cover Crop).