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

3
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11
12 **Abstract**

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

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

37

38 **Keywords:** carbon, climate change, sustainability, life cycle assessment, carbon storage, nitrogen
39 supply

40

41 **1. Introduction**

42 Agriculture and climate change are characterized by critical and controversial cause-effect
43 linkages. These linkages may in turn affect the environmental, economic and social spheres and
44 make it difficult to exclude farming from strategies to combat climate change. On the one hand, in
45 2016, agriculture produced 431 Mt CO₂ equivalents (CO₂e) of greenhouse gas (GHG) emissions in
46 the European Union - 28 (EU-28) + Iceland (ISL). Specifically, methane (CH₄), nitrogen dioxide
47 (N₂O) and carbon dioxide (CO₂) emitted by agriculture corresponded to 47.5%, 72.2%, and 0.3% of
48 the total EU-28 + ISL emissions, respectively (EEA, 2018).

49 From a diagnostic perspective, life cycle assessment (LCA) may be an appropriate instrument
50 to identify and quantify the GHG emissions and, more generally, the environmental impacts caused
51 by a crop production system (Rebolledo-Leiva et al., 2017; Goglio et al., 2018). Specifically, within
52 the LCA context, the carbon footprint (CF) represents the overall quantity of CO₂ and other GHG
53 emissions related to a certain product produced throughout its life cycle (Baldo et al., 2014; Al-
54 Mansour and Jecic, 2017). On the other hand, agricultural management practices aimed at
55 enhancing soil carbon stocks might play a key role in mitigating climate change (Söderström et al.,
56 2014). Moreover, soil organic carbon (SOC) sequestration may be considered one of the most cost-
57 effective options for counteracting the effects of climate change (Nayak et al, 2019). In this sense,
58 the social carbon cost (SCC) might be a useful indicator of the potential efficacy of climate change
59 mitigation measures. In principle, it estimates the monetized damage caused by an incremental
60 increase in C emissions in a given year (Greenstone et al., 2013).

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

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

69 Sanz-Cobeña et al. (2017), these varying conditions lead to the existence of two counteracting
70 cropping systems (i.e., irrigated and rainfed) that require the selection and combination of different
71 management practices (e.g., fertilization, soil tillage, use of cover crops, crop residues, and biochar)
72 that might mitigate GHG emissions and, at the same time, enhance SOC content. Furthermore,
73 Mediterranean agricultural areas are characterized by a low SOC level that makes these
74 agroecosystems vulnerable to land degradation and desertification (Aguilera et al., 2013). These
75 risks might be exacerbated by inappropriate land use change or land management (e.g.,
76 transformation from a forest or natural grassland to a pasture or cropland), and removing biomass or
77 disturbing soil may lead to soils becoming deficient in carbon and other nutrients (Smith et al.,
78 2016).

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

89 Perennial energy crops may be less harmful than annual crops in terms of GHG emissions,
90 especially because of their lower nitrogen (N) requirements; thus, their long-term N management
91 requirements might be less intense than those of annual crops (Drewer et al, 2012). The conversion
92 of an annual cropping system to perennial bioenergy may enhance SOC storage due to the greater
93 capacity of perennial crops to sequester carbon, which is likely due to the deposition and
94 decomposition processes of perennial plant material on the soil surface; in addition, their massive
95 root growth and belowground senescence processes may contribute to the SOC content (Panda,
96 2016). The increase in soil C under a perennial crop system is characterized by significant
97 variability that is likely due, on the one hand, to complex interactions among climate, soil texture
98 and soil biota and, on the other hand, to the choice of soil management practices, which should
99 reduce the disturbance and destruction of aggregates (Tiemann and Grandy, 2014).

100 This study aimed to evaluate the potential performance of different N management practices in
101 perennial energy crop cultivation (cardoon) in a Mediterranean area in terms of their ability to
102 reduce GHG emissions and foster SOC storage in the long term. The analysis was implemented by

103 combining two methodological approaches, CF and SCC, to highlight the potential relevance of
104 fertilization patterns to addressing the effects of climate change from both environmental and socio-
105 economic perspectives.

106

107 **2. Materials and methods**

108 *2.1. Study site*

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

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

124

125 *2.2. Cardoon*

126 *Cynara cardunculus* L. is one of the most promising crops for use as feedstock for the energy
127 sector (e.g., solid fuel and biodiesel) in addition to being useful for various industrial applications
128 (e.g., cellulose, pulp and paper, phytochemical and pharmacological products) (Gominho et al.,
129 2018). It is a perennial herbaceous species that includes three botanical taxa (i.e., globe artichoke
130 (var. *scolymus* L. Fiori), cultivated cardoon (var. *altilis* DC.) and wild cardoon (var. *sylvestris* Lam.
131 Fiori)) and is native to the Mediterranean Basin (Gatto et al., 2013). Although the three cardoon
132 varieties' performances in terms of biomass and/or energy yield are different, cardoon is adaptable
133 to poor pedo-climatic and input conditions (Ierna et al., 2012; Francaviglia et al., 2016; Neri et al.,
134 2017). The capacity to grow under stressed conditions such as Mediterranean rainfed conditions
135 depends on the drought-escape strategy: the aboveground plant parts dry up over the summer,

136 whereas the underground plant parts survive by becoming quiescent; this strategy has been
137 observed in other vivacious plants (Fernández et al., 2006).

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

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

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

151

152 2.3. *Experimental site*

153 A field trial was conducted on cardoon (*Cynara cardunculus* L. var. *altilis* DC.) cultivation for
154 three consecutive crop years (from 2014-15 to 2016-17) at the “Mauro Deidda” experimental farm
155 of the University of Sassari located in northwest Sardinia (Lat. 41°N, Long. 9°E, 81 m a.s.l.).
156 Cardoon is considered one of the most promising perennial energy crops in the Mediterranean
157 region since its adaptability to water and soil stress conditions prevents these stresses from
158 undermining biomass production (Deligios et al., 2017). Throughout the trial, the average annual
159 precipitation was 363 mm, and the mean maximum and minimum temperatures were 22°C and
160 12°C, respectively. At the experimental site, the soil is classified as a sandy clay loam, with 66%
161 sand, 19% clay and 15% silt. At the beginning of the experiment, soil samples from a depth of 0-40
162 cm were collected and analyzed before applying the fertilization treatments. The soil samples had
163 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,
164 respectively.

165

166 2.4. *Experimental design*

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

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

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

193

194 Table 1

195

196 The use of biochar obtained from the thermochemical conversion of biomass (i.e., pyrolysis)
197 may affect the physical and chemical properties of soil by enhancing its fertility and therefore
198 fostering crop growth (Tan et al., 2017). Since cardoon biomass is grown for energy production,
199 biochar application to soil might offset the amount of carbon removed by biomass harvesting.
200 Specifically, biochar obtained from a slow pyrolysis process using rapeseed straw as the feedstock
201 was applied (10 t ha⁻¹) only once at the beginning of the trial (November 2014) and was
202 incorporated into the soil to a depth of 10 cm. In this study, biochar was considered as the amount

203 of C obtained from feedstock pyrolysis (i.e., 71.34 wt %) on the basis of the report of
204 Karaosmanoğlu et al. (2000).

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

219

220 2.5. Functional unit, system boundaries and data collection

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

232 In this study, a “from cradle to field gate” approach was adopted to emphasize the
233 environmental implications of agricultural practices applied to energy crop systems. Specifically,
234 the system boundary considered in this investigation included, for each fertilizer management
235 treatment, the whole life cycle of cardoon cultivation from the acquisition of raw material inputs to
236 the farm gate (i.e., crop harvesting) (Figure 1). Hence, the LCA neglected product transport

237 operations and stopped at product harvesting; the evaluation did not focus on activities beyond the
238 edge of the field. All farming practices carried out throughout cardoon cultivation were included in
239 an inventory to support subsequent steps (i.e., impact assessment and interpretation). The
240 quantification of inventory, namely, the material and resource flows to and from the environment
241 within the system boundaries, should be methodologically sound, complete and unbiased (Sauer,
242 2012). Therefore, the inventory of agricultural practices throughout the three years of the trial was
243 based on primary data collected at the experimental site specifically regarding the agricultural
244 machinery, fuel consumption, and types and application rates of synthetic fertilizers, pesticides and
245 organic amendments.

246

247 Figure 1

248

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

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

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

264

265 2.6. Calculation methodology

266 Different tools were applied to improve the accuracy of the results of this study since the
267 performance of the tools was mainly based on primary data related to soil physicochemical
268 properties, climatic parameters, crop management, and yield. The use of several models enabled us
269 to better understand the effects of the different fertilization patterns in terms of CO₂e produced or

270 avoided. In this way, we obtained more detailed information on the GHG fluxes in terms of their
271 potential environmental and monetary damages.

272

273 *2.6.1. Fertilizer and amendment emissions*

274 The main nitrogen emissions caused by each management treatment (i.e., ammonia (NH₃) and
275 nitrous oxide (N₂O) in the air and nitrate in water (NO₃⁻) were included in the analysis using the
276 Estimation of Fertilizer Emissions Software (EFE-So) (2015). This software uses the model
277 developed by Brentrup et al. (2000) and allows us to obtain more accurate emission values than
278 other methods since it requires various site-specific data to contextualize the fertilizer application
279 and the possible losses without distinguishing between direct and indirect emissions. This model
280 considers the difference between the supplied N and the absorbed N and requires information about
281 the fertilizer type, soil characteristics, climate context (e.g., air temperature during distribution,
282 summer and winter precipitation) as well as the N content in the harvested crop and its coproducts
283 (Schmidt Rivera et al., 2017).

284 According to Brentrup et al. (2000), N emissions are affected by different parameters. For
285 instance, the average air temperature, infiltration rate, time between distribution and incorporation,
286 precipitation, radiation, and wind speed are necessary to evaluate NH₃ volatilization from organic
287 fertilizers. In the case of synthetic fertilizers, NH₃ loss mainly depends on the ammonium or urea
288 content of the synthetic fertilizer, the climatic conditions, and the soil properties. The complexity of
289 interactions between soil and climate factors and the variability of crop system management make it
290 difficult to assess N₂O emissions. Nevertheless, the model uses the default value proposed by
291 Houghton et al. (1997) as the emission factor for N₂O. Finally, NO₃⁻ loss was reported by
292 Brentrup et al. (2000) as nitrate leaching. The rate of NO₃⁻ loss is strictly dependent on different
293 parameters related to agricultural activity (nitrogen balance) and to soil and climate conditions
294 (field capacity in the effective rooting zone and water drainage rate, respectively). The value for
295 atmospheric N deposition included in the EFE-So model was estimated based on the report of
296 Markaki et al. (2010) regarding annual nitrogen deposition fluxes at different sites in the
297 Mediterranean region, including Sardinia.

298 To obtain more detailed results, the amount of CO₂ fixed in the industrial urea production
299 process and potentially emitted through fertilizer distribution was considered in this analysis using
300 Eq. (1) (De Klein et al., 2006):

301

$$302 \quad \text{CO}_2\text{-C Emissions} = M \times \text{EF} \quad (1)$$

303

338 calculated based on two specific values (2% and 1.65%, respectively) determined during a field trial
339 carried out in the same geographical area as this study.

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

342

$$343 \quad \text{SOM} = \text{DM} - \text{A} \quad (3)$$

344

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

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

349

$$350 \quad \text{SOC} = \text{SOM}/2 \quad (4)$$

351

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

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

356

357 *2.6.3. Pesticide emissions*

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

365 The primary distribution refers to the pesticides that are deposited on the crops (e.g., crop
366 leaves) and on the soil surface or are blown away by the wind immediately after pesticide
367 application. The secondary distribution refers mainly to the fate of pesticides on the field; active
368 pesticide ingredients may be deposited on crops, topsoil, or subsoil, where they may undergo
369 different processes. The pesticide fraction that settles on plants might be subject to volatilization,
370 uptake or degradation. On the topsoil, the main processes affecting pesticides are volatilization,

371 biodegradation and surface water runoff due to rainfall; pesticides might also reach the subsoil and
372 thus the ground water through leaching.

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

379

380 *2.6.4. Carbon footprint*

381 The carbon footprint is a methodological tool used to quantify the total amount of GHGs that a
382 product or a service disperses into the environment during its lifetime (i.e., from raw material
383 production to the final use of the product) expressed as CO₂e (Ramachandra and Mahapatra, 2015).
384 In this study, the CF assessment carried out with an LCA approach enabled the quantification of
385 GHG emissions due to the agricultural management practices used in cardoon cultivation
386 throughout the cardoon life cycle.

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

394 The GHG Protocol provides guidance to facilitate the management of agricultural GHG fluxes
395 by considering mechanical (i.e., equipment or machinery operated on farms) and nonmechanical
396 (e.g., soil amendment and management, crop residue burning, and land use change) emission
397 sources as well as upstream sources (e.g., raw material extraction; fertilizer, pesticide and feed
398 production) in order to foster eco-friendly production practices (Russell, 2011). The GHG Protocol
399 uses the Intergovernmental Panel on Climate Change (IPCC) calculation approach to quantify the
400 GHG fluxes of a given activity (WRI and WBCSD, 2011b). The GHG emissions related to the life
401 cycle of a product may be expressed as CO₂e using a characterization factor, the global warming
402 potential (GWP), developed by the IPCC within the climate change impact category (JRC, 2007).
403 The GWP enables us to compare the potential climate impacts of various gases using the GWP
404 value of CO₂ as a reference unit; the GWP of CO₂ is equal to 1 and can be considered at three

405 different time horizons, namely, 20, 50 and 500 years (WRI and WBCSD, 2011a). In this study, the
406 CO_{2e}, that is, the CF of a certain process, was calculated with Eq. (5) (Morawicki and Hager, 2014):

407

$$408 \quad \text{GHG emissions in CO}_2\text{e}_{(i)} = \text{emission factor} \times \text{activity rate} \times \text{GWP}_{(i)} \quad (5)$$

409

410 where CO_{2e} is the CF from a certain gas (kg CO_{2e}); the emission factor (i) is the amount of
411 GHG produced per unit of activity rate; the activity rate is the level of a specific practice (e.g., liter
412 of diesel consumed during fertilizer distribution); and GWP_(i) is the characterization factor
413 expressed in kg CO_{2e}/kg GHG.

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

418 The CEFS category refers to emissions arising from fossil sources (e.g., carbon from fossil
419 fuels), and BCE is related to biogenic sources (i.e., carbon from living organisms or materials
420 derived from biological matter). CELT refers to emissions from the conversion of one land use
421 category to another. The last category, CU, refers to the CO₂ stored in plants and trees as they grow
422 (WRI and WBCSD, 2011b). Since the analysis in this study concerns a perennial crop, all estimated
423 impact categories were expressed in annual CO_{2e}, that is, the CF values of each impact category for
424 cardoon were calculated considering their lifetime average impacts. Finally, the values of the
425 impact categories provided by SimaPro are expressed on a land basis in kg CO_{2e} ha⁻¹, but this
426 study adopted a production functional unit (i.e., tons of biomass produced by cardoon). Therefore,
427 the outputs were converted with Eq. (6) (Cheng et al., 2015):

428

$$429 \quad \text{CFY} = \text{CFA}/\text{Y} \quad (6)$$

430

431 where CFY is the carbon footprint of a generic impact category per production unit (t CO_{2e}/t of
432 biomass produced); CFA is the value of one impact category on a land basis (t CO_{2e}/ha); and Y is
433 the yield of a given crop (t/ha).

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

438

439 *2.6.5. Carbon footprint uncertainty analysis*

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

445

446 *2.6.6. Soil carbon storage*

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

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

457 In this study, carbon storage was estimated using the Rothamsted carbon model (RothC) ver.
458 26.3. This model was specifically developed to estimate the turnover of SOC in nonwaterlogged
459 topsoil and includes the effects of soil type, climate conditions and plant cover on the turnover
460 process (Coleman and Jenkinson, 2014). Its performance is strongly dependent on site-specific data
461 since it requires three different types of information: i) climatic data, i.e., monthly air temperature
462 ($^{\circ}\text{C}$), rainfall (mm), and evapotranspiration (mm) values; ii) soil data, including clay content (%),
463 inert organic carbon (IOM), initial SOC stock (t C ha^{-1}), and depth of the considered soil layer
464 (cm); and iii) land management data, such as soil cover and monthly quantity of plant residues (t C
465 ha^{-1}) (Barančíková et al., 2010). RothC was used to estimate the SOC for each agricultural
466 treatment adopted for cardoon cultivation based on site-specific soil and climatic conditions and
467 with a time reference of 100 years, i.e., the same time horizon used by SimaPro to assess the CEFS,
468 BCE, CELT, and CU impact categories.

469 All inputs were included in RothC as the average values for the experimental trial period. In the
470 model, SOC is divided into four active pools and a small amount of IOM that is resistant to the
471 decomposition process. Crop C inputs to soil are allocated into the categories decomposable and
472 resistant plant material (i.e., DPM and RPM, respectively), microbial biomass (BIO), and humified

473 organic matter (HUM) (Li et al., 2016). RothC allows the C input to be partitioned between DPM
474 and RPM on the basis of its provenance, namely, crops, grassland or forests. These two pools
475 undergo decomposition, resulting in CO₂, BIO or HUM depending on the soil clay content. The
476 decomposition process for one active compartment occurs through first-order decay at a specific
477 rate (year⁻¹) for DPM, RPM, BIO, and HUM (10, 0.3, 0.66, and 0.02, respectively) (Zimmermann
478 et al., 2007).

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

480

$$481 \quad Y = Y_0 (1 - e^{-abckt}) \quad (7)$$

482

483 where Y is the material quantity of a pool that decomposes in a certain month (t C ha⁻¹); Y₀ is
484 the initial C input (t C ha⁻¹); k is the decomposition rate specific to each compartment; a, b and c
485 are factors that modify k related to temperature, moisture, and soil cover, respectively; and t is 1/12,
486 to express k as the monthly decomposition rate. The IOM was calculated with Eq. (8) (Falloon et
487 al., 1998):

488

$$489 \quad \text{IOM} = 0.049 \times \text{SOC} \times 1.139 \quad (8)$$

490

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

495

$$496 \quad \text{SOC-S} = \text{SOC concentration} \times \text{BD} \times d \times (1 - \delta_2 \text{ mm}) \times 10^{-1} \quad (9)$$

497

498 where -SOC-S is the soil organic carbon stock (mg ha⁻¹); SOC is the soil organic carbon (g kg⁻¹);
499 BD is the bulk density (mg m⁻³); d is the soil thickness (cm); and δ₂ mm is the fractional
500 percentage (%) of gravel greater than 2 mm in size.

501 Finally, the SOC values provided by the RothC simulation for the time horizon of 100 years for
502 each fertilization treatment used in cardoon cultivation throughout the experimental trial were
503 converted to CO₂. This conversion was performed with Eq. (10) (Alani et al., 2017):

504

$$505 \quad 1 \text{ ton of soil C} = 3.67 \times \text{tons of CO}_2 \quad (10)$$

506

507 where the tons of CO₂ are the quantity of CO₂ emitted or stored depending on the ratio of the
508 molecular weights of C (12) and CO₂ (44), namely, $44/12 = 3.67$.

509 The values of CO₂ obtained were expressed in CO₂e based on the GWP of CO₂ for 100 years,
510 i.e., 1 (Forster et al., 2007). These outputs are the CF of the SOCS impact category for each cardoon
511 management treatment. As for the previous impact categories, these outputs were also converted to
512 production functional units to facilitate comparisons of the different fertilization treatments in terms
513 of their potential C storage.

514

515 *2.6.7. Social Carbon Cost*

516 The social carbon cost represents the cost of an additional ton of CO₂ emissions or its
517 equivalent; in more detail, it describes the change in the discounted value of economic welfare
518 resulting from an additional unit of CO₂e (Nordhaus, 2017). The monetized estimation of the
519 potential damage caused by an increase in GHG emissions in a given year is performed in order to
520 better understand the changes in agricultural production, human health, and the value of ecosystem
521 services that arise due to climate change (IWG, 2016). In contrast, it may also be considered a
522 measure of avoided damage in the case of emission reductions, which provide a socio-economic
523 benefit.

524 In this study, the SCC was calculated based on an assessment of benefits and cost, that is, of the
525 increases and decreases in human well-being due to GHG emissions, by linking the global carbon
526 cycle and temperature variations to a global economic context (van den Bijgaart et al., 2016). SCC
527 evaluations for different time horizons are performed with three integrated assessment models.
528 These models run with several input assumptions and simulate the possible connections between
529 GHG emissions and climate change compared to a baseline scenario as well as different options for
530 assessing the future damages that may arise from an additional released or avoided ton of CO₂
531 emissions (Rose et al., 2014).

532 Each model runs 10K times, which provides thousands of results that are discounted and
533 averaged to obtain an equivalent single number, called the present value. Specifically, the present
534 value is computed for a number of years (x) in the future, and the previous values are reduced by a
535 certain percentage (i.e., the discount rate) for each of the x years at three reference rates, namely,
536 2.5%, 3.0% and 5.0% (Niemi, 2018).

537 With the above methods, in this study, monetized estimations of the SOCS ecosystem service
538 were performed as an attempt to underscore the long-term strengths and weaknesses of the different
539 fertilization practices used in cardoon cultivation as strategies for addressing the challenges of
540 climate change. The SCC was calculated by multiplying the SOCS values of each fertilizer

541 treatment in 2050 obtained from the RothC model by the SCC in 2050, namely, 79 US dollars
542 (2016 dollars per metric ton CO₂e), with the 3% discount rate (Niemi, 2018). To perform this
543 calculation, the SOCS values were converted to tons CO₂e for a 100-year time horizon as described
544 at the end of subparagraph 2.6.6.

545

546 **3. Results**

547 *3.1. Carbon footprint of GHG fluxes from fertilizer management*

548 The descriptions of the CF outputs are focused on the effects (t CO₂e t⁻¹ of cardoon biomass)
549 resulting from the specific characteristics of each fertilizer management treatment, i.e., the different
550 N doses in HI and LI, biochar application, legume cover crop cultivation and their combination.
551 These effects were the focus because the mechanical operations and production inputs did not
552 change among treatments except in a few cases reported occasionally. The environmental impacts
553 of these factors were not considered because the CF values did not differ among treatments when
554 expressed on a land basis and because we wanted to remain consistent with the objective of this
555 study, that is, to evaluate the potential reductions in GHG emissions and SOC storage resulting
556 from different N fertilizer management strategies applied to cardoon.

557 The environmental performance of the five treatments showed significant variability in both
558 inter- and intra-impact categories (Figure 2). In fact, in the former, CF ranged from 0.00041 to 0.2 t
559 CO₂e per production unit in CELT (LI) and CEFS (HI), respectively. The difference detected
560 between HI and LI - CEFS exceeded CELT slightly more than 480 times - is particularly interesting
561 considering the CEFS value of all fertilization patterns taken together. In fact, the CF of the CEFS
562 category was 432, 40, and 14 times greater than those of CELT, CU, and BCE, respectively.
563 Regarding CU, all further values reported should be considered reliable in absolute terms since this
564 impact category is related to GHG savings, whereas the other categories are related to GHG losses.

565

566 Figure 2

567

568 Considering the effect of each treatment in the single-impact category, HI demonstrated the
569 highest environmental performance in CEFS exceeding the second worst management (LI) by 21%.
570 The observed gap between HI and LI was mainly due to the different impacts of agricultural inputs,
571 especially fertilizer inputs. In fact, the mechanical operations were the same except in the LI + Bi,
572 LI + CC, and LI + Bi + CC treatments, in which two additional agricultural inputs were introduced,
573 namely, biochar and legumes that were sown or distributed and subsequently buried. Furthermore,
574 the higher amount of N fertilizer (i.e., urea as a topdressing) used in HI was mainly responsible for

575 the poor environmental performance of this treatment in the CEFS category; HI had twice the
576 impact of the second most impactful treatment (LI). HI was 20% and 10% more impactful than LI +
577 Bi and LI + CC, respectively; however, the last two categories included two additional mechanical
578 operations and two additional production inputs, namely, biochar and its distribution and burial (LI
579 + Bi) and legume seeds and their sowing and burial (LI + CC).

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

587 LI + CC showed better environmental performance than LI due to the high average production
588 of cardoon biomass (8.14 and 6.91 t DM ha⁻¹ for LI + CC and LI, respectively) that de facto
589 reduced the CEFS value on a production basis rather than to the N and C provided by legume
590 cultivation (slightly more than 3% of the CEFS category). The CF difference between Li + CC and
591 Li + Bi (i.e., 0.01 t CO₂e t⁻¹ more cardoon biomass under Li + Bi) was most likely due to the effect
592 of biochar on GHG emissions from fertilizers since the mechanical operations (i.e., biochar
593 distribution and burial and legume sowing and burial) had the same environmental impact (0.0007 t
594 CO₂e t⁻¹ of cardoon biomass).

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

600 The CELT category showed the lowest CF contribution of the four impact categories, most
601 likely due to the lack of actual land use change, which de facto avoided the production of GHG
602 emissions in this category. Nevertheless, impacts detected within the CELT category can be
603 associated with CO₂ and N₂O emissions generated during agricultural land use and following a
604 change in farm management practices according to the GHG Protocol, which emphasizes the roles
605 of agricultural activity as sources of and a sink for CO₂ (WRI and WBCSD, 2011b).

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

609 0.00041 t CO₂e t⁻¹ of biomass for LI + CC and LI, respectively) and resulted from the different
610 cardoon yields. LI had the lowest cardoon yield and thus was the least environmentally friendly
611 treatment. In contrast, LI + CC produced 18% more cardoon biomass than LI and reduced GHG
612 emissions by 85% compared to those under conventional management. Furthermore, the
613 combination of biochar and the legume cover crop showed, as detected in the CEFS category, an
614 antagonistic effect even though the environmental performance of LI + Bi + CC was worse than
615 those of LI + Bi and LI + CC (by 8% and 10%, respectively). The LI + Bi and HI treatments had a
616 very similar CF per production unit (approximately 0.0003 t CO₂e t⁻¹ biomass), and their CF values
617 were higher than that of LI + CC (by 2% and 3%, respectively). This result highlights that the
618 potential effect of the cover crop on increasing cardoon yield was most likely responsible for the
619 low CF in the CELT category.

620 The last two impact categories, BCE and CU, which are more specifically related to C
621 dynamics, showed intermediate values between those of CEFS and CELT. LI + Bi + CC was the
622 worst and the best treatment for BCE and CU, respectively (0.03 and 0.01 t CO₂e t⁻¹ of biomass).
623 This result suggests that organic material used in addition to synthetic fertilizers might act as both a
624 source and sink of C. The environmental performance of these alternative fertilization treatments
625 might depend on how the additional inputs were included in the overall crop management.
626 Specifically, the sum of the CFs resulting from LI + Bi + CC and LI + Bi represented 92% of the
627 BCE category on the whole, underlining the relevance of biochar as a C source. In fact, the C
628 contribution provided by biochar application exceeded 90% in both treatments. Although the cover
629 crops were not harvested, the C supply from the legumes was not relevant (7%) to the BCE. The
630 difference in CF between LI + Bi + CC and LI + Bi (i.e., 0.002 t CO₂e t⁻¹ more biomass in LI + Bi
631 + CC) was due to the simultaneous use of biochar and the legume cover crop. Their combination
632 had a synergistic effect that increased the CF compared to those resulting from the biochar and
633 legume crop individually. This is because the CF of LI + Bi + CC exceeded by 9% the sum of the
634 CFs of the individual practices. In other words, in the LI + Bi + CC treatment, biochar and the
635 legume crop might have acted to strengthen the effect of one or both of these practices. The
636 environmental performance of LI + CC was 17 times lower than that of the worst treatment, further
637 highlighting the relevance of biochar in the BCE category. The two conventional management
638 treatments, namely, LI and HI, made the best contribution in terms of avoided CO₂ emissions (6%)
639 compared to those from the treatment with the greatest impact because of the absence of the
640 additional organic C source.

641 Among the four impact categories, CU is the most related to GHG emission removal since it
642 concerns the C stored in a crop throughout its life cycle. As mentioned above, the most

643 environmentally friendly treatment within the CU category was the worst treatment for BCE. LI +
644 Bi + CC showed conflicting performance results due to the combination of biochar and legume
645 cover crops. This treatment had the highest CF value, which might be due to the synergistic effect
646 that was also observed in the CU category and was caused by the interaction between biochar and
647 the legume cover crop. Their simultaneous action, which resulted in a CF value 16% higher than the
648 sum of the CFs of the individual treatments, might have resulted in greater C storage in the biomass
649 than that in the LI + Bi and LI + CC treatments.

650 Furthermore, LI + Bi + CC had a higher CF value than LI + CC and LI + Bi (by 13% and
651 170%, respectively), suggesting that the positive environmental performance in LI + Bi + CC might
652 be due to the synergistic effect of biochar and the legume enhancing C uptake from cardoon and the
653 legume cover crop. In contrast, the lowest CF occurring in LI + Bi underlines that the potential
654 effect of biochar on the ability of cardoon to store carbon might not have been adequate to
655 guarantee good performance.

656 In addition to crop yield, some agricultural inputs had various impacts on the CU category
657 based on the management treatment. For instance, the cardoon seeds for sowing contributed
658 approximately 10% on average to the LI + Bi, LI + CC, and LI + Bi + CC treatments. The synthetic
659 fertilizers used in LI + Bi had an effect equal to 13% on CU, whereas the C from the legume cover
660 crop contributed 30% to LI + CC. The same inputs made contributions of 5% and 29%,
661 respectively, in LI + Bi + CC. The environmental performance of LI in terms of CO₂ uptake was
662 8% higher than that of LI + Bi, most likely since the yield of LI was greater than that of LI + Bi.
663 The quantity of cardoon biomass might also have played a role in the CF values of the HI and LI
664 treatments. In fact, LI, which had lower average biomass production than HI, had the best
665 environmental performance in the CU category, with a contribution that was slightly more than 7%
666 higher than that of HI. Due to the use of double the N dose (HI vs LI), the N fertilizer effect on the
667 CU was almost 2 times greater in the HI treatment.

668 A more in-depth analysis of the individual CF balances for each agricultural treatment (i.e., the
669 comparison of GHG release and GHG removal) allowed us to better understand the effects of
670 fertilizer patterns on GHG fluxes (Figure 3). All CF balances showed GHG emission losses,
671 ranging from 0.20 (HI) to 0.14 (LI + CC) t CO_{2e} per production unit. The balances for LI + Bi, LI
672 and LI + Bi + CC were 81%, 82%, and 90%, respectively, of the highest balance. The inclusion of a
673 cover crop (i.e., a legume) in a perennial energy system (cardoon) might be optimal for GHG
674 emission reduction and yield optimization.

675

676 Figure 3

677

678 The second positive trade-off between the GHG balance and crop production was shown in LI
679 + Bi. Although this treatment showed the same GHG balance as that of LI ($0.16 \text{ CO}_2\text{e t}^{-1}$ of
680 biomass), the cardoon yield achieved with biochar application was greater than the LI yield (7.96 vs
681 6.91 t ha^{-1} on average). In contrast, the balance of LI + Bi + CC was the second highest, suggesting
682 that the combination of biochar and the cover crop did not result in a reduction in GHG emissions,
683 although the cardoon yield achieved with LI + Bi + CC was intermediate to the biomass production
684 levels of LI + Bi and LI + CC.

685

686 3.2. Uncertainty analysis results

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

691

692 Table 2

693

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

703

704 3.3. Soil organic carbon stocks under fertilizer management

705 The analysis was completed by considering the SOCS category in order to detect changes in
706 SOC storage resulting from the implementation of the five fertilization patterns. Although the
707 SOCS category was expressed in $\text{t CO}_2\text{e t}^{-1}$ cardoon biomass, as were the previous four categories,
708 its environmental impact was calculated from direct measurements taken in the field throughout the
709 experimental trial (Figure 4).

710 SOCS ranged from 72.7 (HI) to 26.2 (LI) t CO₂e per production unit, highlighting that the two
711 conventional management strategies showed the best and the worst performance; the difference was
712 equal to slightly less than 3 times in favor of HI management. The performance of HI might be due
713 to the higher N dose applied throughout the cardoon life cycle which, in turn, most likely fostered a
714 higher yield than that under LI. The three alternative treatments showed values (53.1, 53.9 and 61.1
715 t CO₂e t⁻¹ of biomass for LI + Bi + CC, LI + CC and LI + Bi, respectively) that were closer to that
716 of the best (i.e., the highest value) treatment than to that of the worst (i.e., the lowest value)
717 treatment, highlighting that the treatments that included biochar, the cover crop or their combination
718 fostered SOCS. The simultaneous use of biochar and the legume demonstrated an antagonistic
719 effect on SOCS; the sum of the effects of biochar and the cover crop individually was 2 times
720 higher than the value obtained from their combination. The environmental performance of LI + Bi
721 was better than those of LI + CC and LI + Bi + CC (by 13% and 15%, respectively), highlighting
722 that the application of biochar might have had a stronger effect than the other two fertilizer
723 management strategies in terms of soil carbon storage.

724

725 Figure 4

726

727 *3.4. Social carbon costs from fertilizer management*

728 A monetary valuation was performed to estimate which fertilizer treatment might generate the
729 greatest flow of benefits related to the SOCS ecosystem service. The results highlighted that HI
730 might produce the greatest benefits until 2050 (Table 3). Specifically, these benefits could amount
731 to approximately 9K US dollars per t CO₂e. In contrast, the lower benefits arising from the other
732 treatments suggests the presence of a social cost (an opportunity cost in terms of lost benefits
733 compared with those in the most favorable treatment). The LI treatment had the highest SCC, equal
734 to approximately 5K US dollars per 1t CO₂e, whereas the other three treatments showed SCC
735 values ranging from 1K (LI + Bi) to 2K (LI + Bi + CC) US dollars per 1t CO₂e.

736

737 Table 3

738

739 **4. Discussion**

740 *4.1. Carbon footprint implications of agricultural management*

741 The results highlight that the characterization of a perennial energy crop system in terms of
742 agricultural management and land allocation should be used to better support farmers' decisions as
743 well as to reduce GHG emissions and to increase soil C storage in the long term. Specifically, the

744 choice of farming practices and land use might arise from a convenient trade-off between the yield
745 and environmental performance of energy crops, for example, to satisfy present and future needs in
746 terms of food and energy security as well as environmental sustainability. This study might provide
747 useful support for selecting the best option since the results enabled us to highlight the strengths and
748 weaknesses of each fertilization pattern and its effects on GHG dynamics (Figures 2-4).

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

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

768 The potential effect of biochar on soil CO₂ emissions is still complicated and poorly understood
769 because of the considerable uncertainties in both time (in the short or long term) and space (at the
770 laboratory or field scale) (Fidel et al., 2018). In fact, CO₂ emissions showed different behaviors
771 (increasing, decreasing or unchanged dynamics) as a result of organic amendment addition, mainly
772 due to the complicated interactions between the biochar feedstock and its physicochemical
773 properties; application rate and mode (i.e., alone or combined with synthetic or organic fertilizers);
774 soil type, nutrient availability, and microbial activity; and crop management practices (e.g.,
775 incorporation of residual biomass, rate and time of synthetic fertilizer application) (Kuppusamy et
776 al., 2016; Shen et al., 2017). These complex interactions also have variable effects on the emissions
777 of other GHGs from soil, such as N₂O. In this context, the performance of LI + Bi + CC is even

778 more difficult to interpret since it is most likely affected by the interaction between biochar and the
779 legume cover crop, which is difficult to specify. Therefore, an attempt was made to analyze the
780 results into each impact category to identify synergistic effects.

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

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

806 In our opinion, precision techniques may be considered a useful support for more efficient
807 resource use (e.g., nutrient use) from a circular economy approach. In this paradigm, bioenergy
808 production could offer a viable contribution for addressing challenges related to environmental
809 concerns and resource scarcity (Pan et al., 2015). Although biomass plays an important role in the
810 circular economy context as a feedstock alternative to nonrenewable energy sources, achieving high
811 biomass crop yields involves energy and material costs due to, for instance, fertilizer use and

812 production (Sherwood, 2020). The use of byproducts (e.g., biochar) would close the loop in
813 agriculture, minimizing fertilizer nutrient dissipation in the environment and regenerating natural
814 resources (Chojnacka et al., 2020). In this sense, biochar may be considered a promising option that
815 is well suited to circular economy principles, even though its capacity to foster carbon
816 sequestration, improve soil quality and support plant growth is strongly affected by its
817 physicochemical characteristics and the production technology used (Bis et al., 2018; Olfeld et al.,
818 2018).

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

823 However, the exploitation of natural resources (e.g., water) and the application of N fertilizers
824 that are prone to leaching may foster or exacerbate possible pollution phenomena, particularly in
825 vulnerable agricultural areas devoted to profitable crop cultivation. As reported by Balafoutis et al.
826 (2017), the application of precision agriculture practices (e.g., technologies that allow variable rate
827 application of nutrients, irrigation, pesticides and planting/seeding; controlled traffic farming and
828 machine guidance) with technological equipment may spatially and temporally optimize the use of
829 inputs based on site-specific characteristics. These practices could cause a reduction in GHG
830 emissions and an improvement in farm economic and production performance compared to those
831 under conventional management.

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

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

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

843

844 *4.2. LCA benefits in agricultural management*

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

858 On the other hand, the effect of crop residues was not included in this analysis because of the
859 lack of information, although it is known the influence of crop residues on soil N dynamics and
860 N₂O emissions. Specifically, the agricultural use of crop residues can contribute to the maintenance
861 of soil functions acting as source of organic matter and nutrients and thus able to improve crop
862 production level (Lehtinen et al., 2014). Furthermore, the plant residue C/N ratio may influence the
863 decomposition of residue and thus the soil N₂O fluxes (Pimentel et al., 2015). Although the use of
864 crop residues with a high C/N ratio may encourage the N utilization by microbes leading to a
865 reduction in N₂O emissions, the effects of crop residues with different C/N ratios on N₂O emissions
866 might also depend on soil - climatic conditions, biochemical composition of plant residues, and
867 agricultural management as a whole (Shan and Yan, 2013; Wu et al., 2016; Zhou et al., 2020).

868 Agricultural systems are closely related to various parameters (e.g., cropping intensity, input
869 prices, climate and soil condition) whose high variability and addition to regional specificities make
870 the data quality a key factor in application of LCA to agricultural products (Weidema and Meeusen,
871 2000). The fate of the emitted pollutants released by a product throughout its life cycle may be may
872 affected by different locations where pollution occur. This spatial variability is traditionally
873 disregarded in life cycle impact assessment (LCIA) although the impact highlights by LCIA may be
874 considerably different from the actual one (Hauschild et al., 2006). On the other hand, the
875 development of region-specific inventories and characterization factors might be relevant to
876 improve the accuracy of LCA analysis (Yang et al., 2018; Patouillard et al., 2019). Regionalized
877 LCIA still remains a challenge since on the one hand, regionalized LCIA characterization factors in
878 combination with site-specific inventories might reduce the uncertainty of results. On the other

879 hand, a proper development of the regionalized LCA might be limited by the lack of standardization
880 in regionalized LCIA data formats, poor site-dependent inventory data availability, and a lack of
881 widespread software support (Mutel et al., 2019).

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

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

891 Our approach confirms this need, and the results suggest that the optimization of agricultural
892 practices, such as fertilization, may have a positive effect on GHG fluxes in the long term.
893 Furthermore, the management of a perennial energy crop is generally not devoid of environmental
894 impacts, and the extent of these impacts often depends on fertilizer use (Wagner and Lewandowski,
895 2017; Fernando et al., 2018). This was consistent with our findings, which identified the field
896 emissions resulting from fertilizer application as one of the main factors responsible for the
897 environmental performance of cardoon cultivation. A similar result was detected by Razza et al.
898 (2017) for cardoon cultivation in Sardinia, although they considered a single value for GWP
899 without distinguishing among impact categories.

900

901 *4.3. Socio-economic effectiveness of agricultural management*

902 The SCC is an economic measure related to negative externalities from a climate change
903 perspective (Anthoff and Tol, 2013). In this study, the ecosystem service corresponding to SOC
904 storage provided by agricultural activity may be considered a positive externality. The cost of this
905 service represents the monetary benefit reduction from changing from HI management, i.e., the
906 practice that contributes the most to C accumulation in the soil, to the other management strategies
907 for cardoon cultivation. This cost is not sustained by farmers because, in the absence of
908 compensatory regulatory mechanisms, the cost is paid collectively in the long term (Havranek et al.,
909 2015).

910 This is a critical point because farmers are deprived of responsibility and do not pay any direct
911 costs from SOCS reduction in order to pursue their own economic objectives (typically profit
912 maximization). Furthermore, the costs would not be equally distributed since we would expect that

913 the less-developed countries would bear more of the costs. In fact, richer and more developed
914 countries are more able to pay the costs related to negative externalities with the greater benefits
915 generated by higher agricultural productivity and profitability. This disparity implies that the
916 estimated SCC in our analysis would tend to increase in developing countries and, in parallel, to
917 decrease in developed countries.

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

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

931

932 **5. Conclusions**

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

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

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

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

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

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

970

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979

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1350

1351 TABLES

1352 Table 1

1353 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 (sprouting 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 (sprouting 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

1354 ^a Fertilization patterns: HI, High Input; LI, Low Input; LI + Bi, Low Input + Biochar; LI+CC, Low Input+ Cover Crop;

1355 LI + Bi + CC, Low Input + Biochar + Cover Crop;

1356 ^b Fertilizer title;

1357 ^c LI + Bi, LI + CC and LI + Bi + CC scenarios were characterized by the same synthetic fertilizer inputs of LI;

1358 ^d Value was obtained on the basis of what reported by Karaosmanoğlu et al. (2000);

1359 ^e Value was estimated on the basis of an experimental trial on the same legume used in this study;

1360 ^f Value was estimated on the basis of the information reported by Chiofalo et al. (2010); Prybil (2010); Pace et al.
1361 (2011); Bozhanska et al. (2016).

1362

1363 **Table 2**

1364 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 -

1365 ^a Impact categories: CEFS, Carbon Emission from Fossil Sources; BCE, Biogenic Carbon Emissions; CELT, Carbon
 1366 Emission from Land Transformation; and CU, Carbon Uptake;
 1367 ^b Fertilization patterns: HI, High Input; LI, Low Input; LI + Bi, Low Input + Biochar; LI+CC, Low Input+ Cover Crop;
 1368 LI + Bi + CC, Low Input + Biochar + Cover Crop.

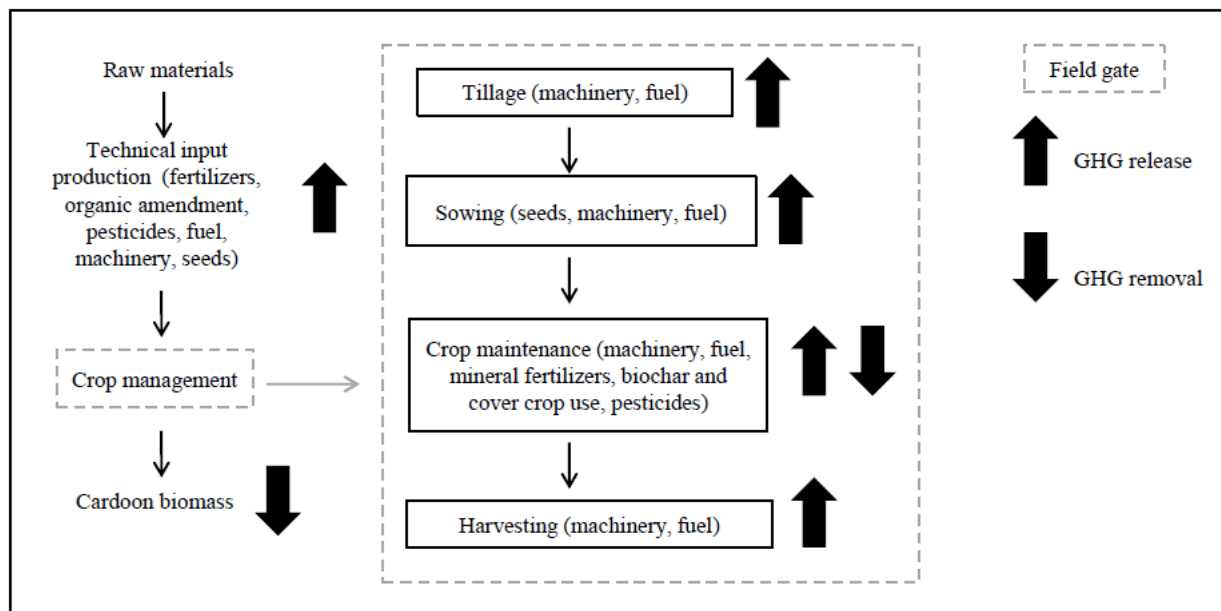
1371 **Table 3**

1372 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

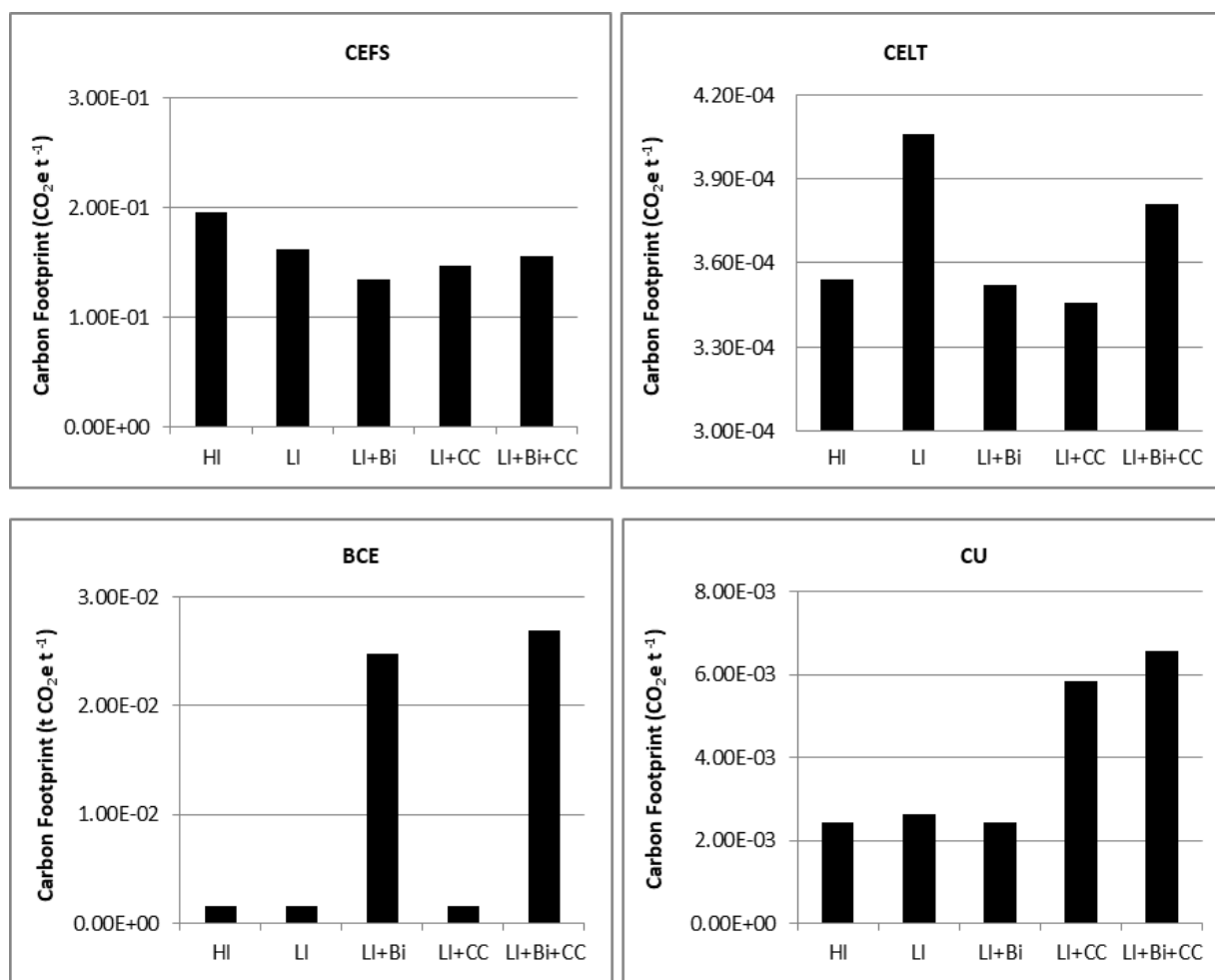
1373 ^a Fertilization patterns: HI, High Input; LI, Low Input; LI + Bi, Low Input + Biochar; LI+CC, Low Input+ Cover Crop;
 1374 LI + Bi + CC, Low Input + Biochar + Cover Crop.

1376 **FIGURES**



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 1379 **Fig. 1.** The system boundary of the analysis

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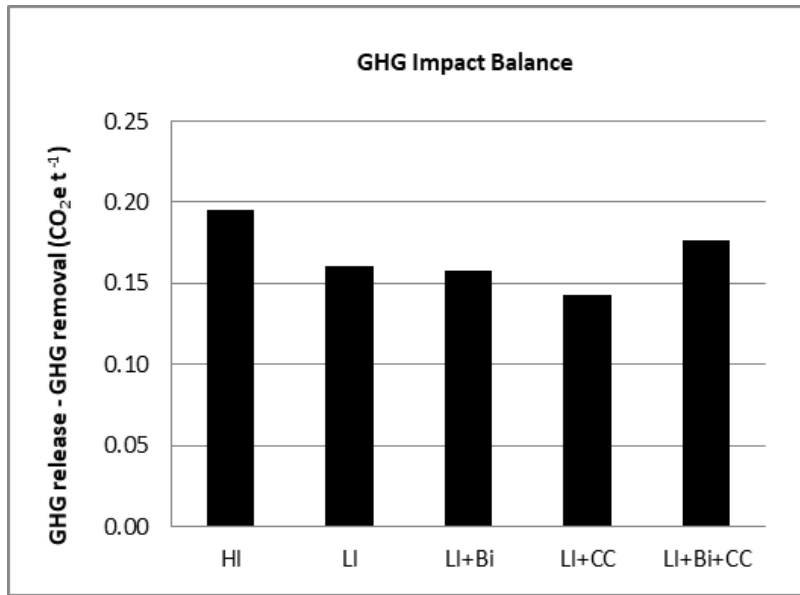
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Fig. 2. Carbon Footprint (t CO₂e t⁻¹ cardoon biomass) of impact categories responsible for GHG fluxes (CEFS, Carbon Emission from Fossil Sources; BCE, Biogenic Carbon Emissions; CELT, Carbon Emission from Land Transformation; and CU, Carbon Uptake) 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).



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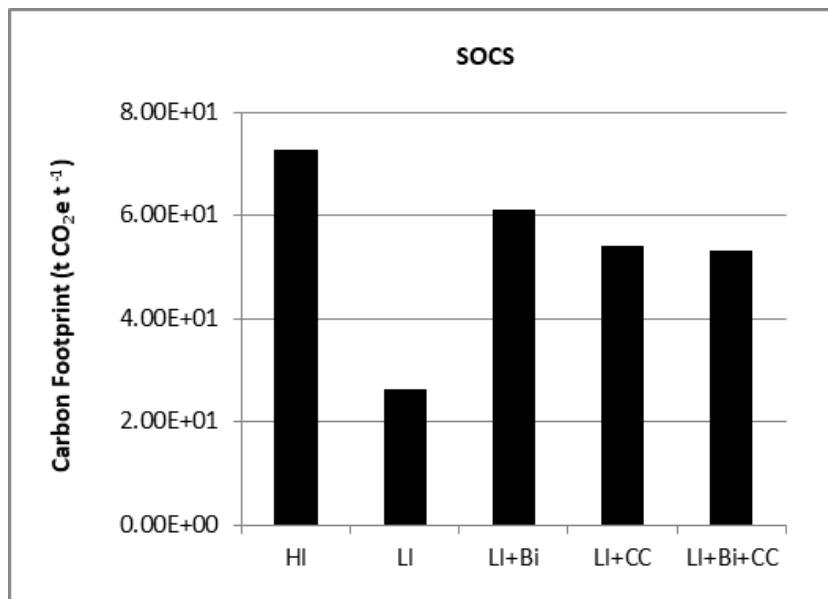
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Fig. 3. Greenhouse gas (GHG) difference among impact categories for each treatment ((HI, High Input; LI, Low Input; LI + Bi, Low Input + Biochar; LI+CC, Low Input+ Cover Crop; LI + Bi + CC, Low Input + Biochar + Cover Crop) considering Carbon Emission 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.



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Fig. 4. Carbon Footprint (t CO₂e t⁻¹ carbon 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).



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Supplementary Material

PBVFSGSD_B125-5525-63D0-E779-7872 (1).pdf

