Catena 218 (2022) 106563

Contents lists available at ScienceDirect

Catena

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

The importance of incorporating soil in the life cycle assessment procedure to improve the sustainability of agricultural management

Mauro De Feudis [a,*], Claudio Selmi [b], Gloria Falsone [a], Daniele Missere [b], Marcello Di Bonito [a,c], Livia Vittori Antisari [a]

[a] Department of Agricultural and Food Sciences, Alma Mater Studiorum - University of Bologna, Via Fanin, 40, 40127 Bologna, Italy
[b] RI.NOVA Soc. Coop., Via dell'Arrigoni, 120, 47522 Cesena, FC, Italy
[c] School of Animal, Rural and Environmental Sciences, Nottingham Trent University, Brackenhurst Campus, Southwell, Nottinghamshire NG250QF, UK

ARTICLE INFO

Keywords:
Cambisols
Fruit orchards
Soil organic C
CO_2eq
LCA

ABSTRACT

The formidable ability of soil to store carbon has attracted an increasing number of studies, but few of them included soil organic carbon (SOC) sequestration as part of a carbon balance assessment in the agroecosystem. This raises some interesting questions: 1) how orchards conversion increase soil capacity to mitigate the green–house gases (GHG) emissions by storing C? 2) can it be considered in life cycle assessment (LCA)? 3) can SOC pools and soil biochemical properties determination improve LCA interpretation? To answer these questions, this study selected a ten– and fifteen–years–old peach orchards, a twenty–years–old pear orchard, a thirty–years–old kiwi orchard in south-east part of Emilia–Romagna Region (Italy), and a cereals' field as reference. Soil samples were collected from 0 to 15 and 15–30 cm depths, and the SOC pool amounts (i.e., labile and recalcitrant) determined. LCA was used to estimate the GHG emissions (CO_2eq) from the orchards. Results showed that the conversion from cereals to orchard production increased OC stock (+82 % on average) suggesting that orchards cultivation systems have the capacity to enrich soil organic matter. Fertilization had the greatest impact on CO_2eq emission accounting for at least 40 % of total CO_2eq emissions. Kiwi cultivation had the highest impact on GHG emissions mainly due to the high water and nutrient demand (0.045 and 0.149 kg CO_2eq kg^{-1} fruit yr^{-1}, respectively). When taking into account the C–CO_2eq loss by fruit cultivation and C storage in soils, results would indicate that peach and pear orchard agroecosystems promote C sequestration. Conversely, kiwi cultivation showed large CO_2eq emissions only partly counterbalanced by SOC sequestration. This study highlights the importance of including soils in LCA: if made mandatory this would allow a wider, yet more detailed, picture of the impact of agricultural practices on C budget. This simple step could help optimise resource management and at the same time improve agroecosystem sustainability.

1. Introduction

In 2020, carbon dioxide (CO_2) concentration in the atmosphere reached values greater than 410 ppm due to the human activities (The World Meteorological Organization, 2020). Agriculture is recognised as a significant contributor to anthropogenic emissions of CO_2 (Smith et al., 2014; Lynch et al., 2021). Recent studies (Gkisakis et al., 2020; Goossens et al., 2017; Mousavi-Avval et al., 2017; Pryor et al., 2017) pointed out that the GHG emissions from agricultural crop production systems are mainly related to the fossil-fuel consuming and to the manufacturing and distribution of chemical fertilizers. Noteworthy is also the production of nitrous oxide (N_2O) gas due to soil nitrogen input (Lawrence et al., 2021). Consequently, a reduced utilization of both fuels and fertilizers could improve the sustainability of agricultural management. For example, Aguilera et al. (2015) compared the environmental impact of several conventional and organic cropping systems in Spain, highlighting greater GHG emissions in the formers compared to the latter, mainly due to the use of chemical fertilizers in the conventional system. Similarly, Pergola et al. (2017) found a greater impact on climate change of apricot orchards under integrated system compared to those under biodynamic one. In the context of the current climate change, soil plays a central role in the mitigation of GHGs emission from agriculture through soil carbon sequestration, defined by Chenu et al. (2019) as "*the process of transferring CO_2 from the atmosphere into the soil of a land unit,*

* Corresponding author.
E-mail address: mauro.defeudis2@unibo.it (M. De Feudis).

https://doi.org/10.1016/j.catena.2022.106563
Received 6 May 2022; Received in revised form 7 July 2022; Accepted 1 August 2022
Available online 6 August 2022

through plants, plant residues and other organic solids which are stored or retained in the unit as part of the soil organic matter". In this sense, worldwide there is a strong agreement to implement the carbon–farming initiatives with the main aim to increase the soil organic carbon (SOC) stock which is a way to mitigate the current climate change (Wiesmeier et al., 2020; Bradford et al., 2019; Chenu et al., 2019). However, to reach this goal, the chemical, physical and edaphic conditions of the soil must allow the humification process and the accumulation of organic C to be carried out rather than the mineralization process. Soil stores three times the amount of C present in the atmosphere (Ciais et al., 2013) and could potentially remove from the atmosphere between 0.79 and 1.54 Gt yr^{-1} of C (Fuss et al., 2018) if land uses and management practices increased C inputs and/or reduced C losses. In this sense, promoting soil organic C (SOC) sequestration is one of the most important strategies to reduce atmospheric CO_2 concentrations with a significant potential to mitigate climate change (Lal, 2018). Bulk SOC is composed of multiple functional pools differing in turnover, in fact it ranges from the most labile form (i.e., the dissolved organic C) to the most stable one as the physically protected and the chemically recalcitrance forms (De Feudis et al., 2019; Poeplau et al., 2018). The long residence time associated with most of the SOC pools (e.g., De Feudis et al., 2019) makes soil a major player in the global carbon budget (Martin et al., 2014). Moreover, soils characterized by high SOC concentrations are recognized to be desirable because SOC improves soil nutrient availability, cation exchange capacity, water retention capacity, soil aeration, soil aggregation and structure, soil microbial biomass and its activity, plant yield and quality (Bationo et al., 2007; Bronick and Lal, 2005; Chavarria et al., 2018; Martínez-Mena et al., 2021).

There is general agreement that management practices are important factors influencing SOC contents in agricultural lands (Montanaro et al., 2017; Novara et al., 2019; Pardon et al., 2017). For example, the cultivation of cover crops has been identified as an effective practice to increase of SOC content (Poeplau and Don, 2015). Similarly, practices addressing the incorporation of the plant residues into the soil could prevent SOC reduction (Keel et al., 2019). The no-tillage has been claimed to be a potential option to decrease SOC loss in agricultural soils (Nath and Lal, 2017), but at global scale its effect on SOC content seems to be limited (Mondal and Chakraborty, 2022). Moreover, it is well known the increase of SOC content when organic fertilizers are applied (Morugán-Coronado et al., 2020).

In this context, life cycle assessment (LCA) is a well established approach to help accounting for all the various stages of any activity, including agricultural practices where it was introduced since 1990 (Haas et al., 2000). LCA is one of the most used standardized methodologies for estimating the environmental impacts linked to the entire cycle of fruit production (Vinyes et al., 2015). Among the environmental impacts, the evaluation of GHG emissions prevail compared to the other environmental problems (Adewale et al., 2019; Bartzas et al., 2017; Rebolledo-Leiva et al., 2017). Most of the studies concerning LCA in agroecosystems take in account yield, plant growth and all the input factors related to the crop cultivation such as human labour, machinery, fertilizer application, fossil fuel consuming and irrigations (e.g., Foteinis and Chatzisymeon, 2016; Kaab et al., 2019). Conversely, despite its high potential to store carbon, soil is generally not included in LCA approach for the evaluation of C budget (Garrigues et al., 2012). Only in few cases SOC was taken into account for the LCA (Arzoumanidis et al., 2014; Brandão et al., 2013; Petersen et al., 2013). Hence, although SOC is essential if LCA is to be applied in case studies where carbon balances must be calculated, the limited number of LCA studies that took into consideration SOC would highlight how soil is generally the forgotten part of the agro-ecosystems. In addition, although the estimation of the bulk SOC stock could be sufficient for C balance in LCA approach, the knowledge of SOC pools and their dynamics are necessary for improving the interpretation of LCA outputs. Specifically, since the important role of LCA to improve the management of agricultural systems for preventing environmental hazards (e.g., the GHGs emissions) in the long–term, the agricultural managements and/or systems able to promote the storage of the most stable SOC forms should be promoted. Therefore, for a reliable C balance through the LCA procedure, it is important that soil C is stored in the most stable forms. Further, because of the key role of soil microbial community to transform and stabilize SOC (Angst et al., 2021; Domeignoz-Horta et al., 2021), the evaluation of their properties (e.g., amount and activity) could be of interest in LCA to understand whether (or not) soil stabilize C.

This study tries to address this gap in the literature and provide a justification for a more widely accepted introduction of soil in agro-ecosystems LCA. In particular, the study will focus on *i)* how orchards conversion increase soil capacity to mitigate the green–house gases (GHG) emissions by storing C; *ii)* how soil C stock can therefore be included in LCA approach; and *iii*) if SOC pools and soil biochemical properties determination can improve LCA interpretation. In order to address these aims, the following hypotheses were set: 1) orchards increase soil C stock compared to grain fields; 2) and soil C storage capacity can mitigate the GHG emissions related to the fruit orchard agricultural practices.

2. Materials and methods

2.1. Study sites description

The present study was conducted in the south-east part of Emilia Romagna Region, Italy. This area had a mean cumulative annual precipitation of 763 mm and a mean annual air temperature of 14.2 °C for the period 1986 – 2015. The study was conducted in 2017, and the specific study site selected included a ten– and fifteen–years–old peach orchards (Ph10 and Ph15, respectively) with a tree density of 1,300 plants ha^{-1}; a twenty–years–old pear orchard (Pr20) with a tree density of 820 plants ha^{-1}, and a thirty–years–old kiwi orchard (Ki30) with a tree density of 710 plants ha^{-1} (see Fig. 1). Some more details of study sites are reported in Vittori Antisari et al. (2021a, 2021b). The choice of the selected tree species was based on their wide distribution in Italy. The mean yields of the selected orchards were 48, 35, 30 and 28 × 10^3 kg (fresh weight) ha^{-1} for Ph10, Ph15, Pr20, Ki30, respectively. According to the farmers, such yields were reached within the fifth year after the orchard establishment. However, because of the missing data about yield during first years of orchard cultivation, in the present study we arbitrarily considered the aforementioned yields also for the first years of cultivation after orchard establishment. All the soils were classified as Cambisols with a texture from silty clay loam to loam, a slight alkaline reaction (pH = 7.7 on average) and bulk density ranging from 1.14 to 1.59 g cm^{-3}, with lower values in 0–15 cm compared to 15–30 cm soil layer. On average, cation exchange capacity of the soils studied was 24.9 cmol(+) kg^{-1}, the exchangeable Ca^{2+}, Mg^{2+} and K^{+} concentrations were 15.7, 1.9 and 0.60 cmol(+) kg^{-1}, respectively, and the base saturation was of 75.4 %. Details about clay content and equivalent soil mass of the study sites are reported in Table S1 of the Supplementary materials.

In the orchards, soil was kept covered by natural grasses which were periodically cut (4–5 times per year). Pruned wood materials were shredded and left on the soil surface. According to the farmers, the average amount of pruned materials for Ph10, Ph15, Pr20 and Ki30 were 3.0, 3.0, 2.5 and 3.5 Mg dry matter ha^{-1}. Some differences occurred for fertilization treatments (Table 1). In Ph10, no chemical fertilization was performed, but exhausted substrate for mushroom cultivation at a rate of 7 Mg ha^{-1} was spread on soil surface every year. In Ph15, Pr20 and Ki30, fertilization was carried out both by fertigation, through drip irrigation lines (one line per plant row), and foliar spray. The amounts of elements applied by fertilization is reported in Table 1.

To estimate C accumulation/loss of fruit orchard soils, a field for grain production (wheat) was used as reference (CK). The rationale to use a field for grains production as reference soil was based both on the widespread cultivation of such crops in the northern Italy and because

Fig. 1. Study site locations. CK: field for grains production; Ph10: 10–years–old peach orchard; Ph15: 15–years–old peach orchard; Pr20: 20–years–old pear orchard; Ki30: 30–years–old kiwi orchard.

Table 1
Amounts of C, N, P_2O_5 and K_2O applied by soil fertilization (Soil), fertigation (Fert) and by foliar spray (Leaf) application to a 10–years–old peach orchard (Ph10), a 15–years–old peach orchard (Ph15), 20–years–old pear orchard (Pr20) and a 30–years–old kiwi orchard (Ki30) through organic or synthetized fertilizers.

Nutrient	Ph10	Ph15	Pr20	Ki30
	Organic	Synthetized	Synthetized	Synthetized
C	Soil = 3990	Soil = 0	Soil = 0	Soil = 0
(kg ha^{-1})	Fert = 0	Fert = 0	Fert = 0	Fert = 0
	Leaf = 0	Leaf = 0	Leaf = 0	Leaf = 0
N	Soil = 140	Soil = 0	Soil = 0	Soil = 54.0
(kg ha^{-1})	Fert = 0	Fert = 117.8	Fert = 79.8	Fert = 69.5
	Leaf = 0	Leaf = 1.4	Leaf = 5.2	Leaf = 0
P_2O_5	Soil = 80	Soil = 0	Soil = 0	Soil = 0
(kg ha^{-1})	Fert = 0	Fert = 36.1	Fert = 38.5	Fert = 54.3
	Leaf = 0	Leaf = 3.3	Leaf = 1.2	Leaf = 1.7
K_2O	Soil = 153	Soil = 0	Soil = 0	Soil = 0
(kg ha^{-1})	Fert = 0	Fert = 47.0	Fert = 148.5	Soil = 1.2
	Leaf = 0	Leaf = 2.6	Leaf = 1.2	Fert = 115.9

the considered fruit orchards were formerly wheat fields for at least 5 years.

2.2. Soil sampling and analyses

Within each field, three 30 cm depth soil pits were dug, and soil samples were collected from 0 to 15 cm (hereafter, surface soil) and 15–30 cm depths (hereafter, subsurface soil). This study used the convention to investigate the 0–30 cm soil depth interval because such interval is worldwide used for the SOC stock evaluation (Makipaa et al., 2012; Guevara et al., 2020; Tangen and Bansal, 2020). The surface and subsurface soil samples were air–dried, passed through a 2-mm sieve and then an aliquot was finely ground for SOC and total nitrogen (TN) concentrations determination.

SOC and TN were determined by a CHN elemental analyser (EA 1110 Thermo Fisher, USA) after addition of hydrochloric acid to remove carbonates. The relative abundance of C and N stable isotopes were determined by continuous flow- isotope ratio mass spectrometry (CF-IRMS) using an isotopic mass spectrometer Delta V advantage (Thermo-Finnigam, DE). Measurements were expressed in standard δ ($\delta^{13}C$ and $\delta^{15}N$) notation (‰) relative to Vienna Pee Dee Belemnite and air, respectively.

Different SOM fractions, like particulate organic matter (POM), fulvic–like and humic–like substances, and non–extractable organic matter (NEOM), were chemically extracted (Agnelli et al., 2014). A volume of 100 mL of distilled water were added to 10 g of soil and shaken on a horizontal shaker for 16 h at 25 °C, centrifuged and the supernatant was separated from the precipitate. The supernatant was passed through a 53 μm sieve and the particles > 53 μm represented the POM. The precipitate remaining into the centrifugation tubes was re–suspended in 100 mL 0.1 M NaOH + 0.1 M Na_4P2O_7 solution and the samples were shaken for 24 h at 25 °C and then again centrifuged. The NaOH extract was passed through a 0.45 μm polycarbonate filter, while the remaining precipitate, containing NEOM was washed using deionized water to remove the excess of Na until the pH of the rinsed solution was ≤ 7. The 0.45 μm filtered NaOH extract was acidified to about pH 1.5 with 6 M HCl and allowed to settle overnight to separate fulvic–like and humic–like substances and centrifuged. To remove the excess of Na from the obtained fractions, the supernatant (fulvic–like substances) was dialyzed through 1000 Da cut–off membranes (Spectra/Por® Dialysis membrane) against distilled water, while the residual (humic–like substances) was washed with 0.002 M HCl. Both purified fractions were freeze–dried. The POM and NEOM fractions were dried at 40 °C. The organic C (OC) and N contents of POM, fulvic–like, humic–like substances and NEOM were determined by a CHN elemental analyser (EA 1110 Thermo Fisher, USA).

Soil microbial respiration was determined according to Falsone et al. (2015). Soil samples were adjusted to 60 % of water holding capacity and incubated for 28 days at 25 °C. The CO_2 emitted from incubated soils was measured through alkali (0.5 M NaOH solution) absorption of the produced CO_2 from each sample. Then, the titration of the rest of NaOH solution was carried out using 0.05 M HCl in presence of 0.75 M $BaCl_2$. The soil basal respiration (SBR) of each soil sample was computed

as the hourly flux of CO_2 per gram of soil, while the cumulative soil basal respiration (RCUM) was expressed as the total amount of CO_2 evolved during the 28 days of incubation.

Soil microbial biomass C (C_{mic}) was measured on soil samples at 60 % of WHC using chloroform fumigation extraction method with 0.5 M K_2SO_4 solution (Vance et al., 1987). Both fumigated and non–fumigated extracts were analysed using a TOC–V CPN total organic carbon analyser (Shimadzu, Japan). C_{mic} was calculated as EC × 2.64, where EC was the difference between organic C extracted from fumigated soils and organic C extracted from non–fumigated soils (Vance et al., 1987). The organic C inside the filtered solution obtained from non-fumigated soil samples were considered as water-extractable organic C (WEOC) (Chantigny et al., 2007).

2.3. Life cycle assessment (LCA) of peach, pear, and kiwi production

The LCA methodology used in the present study aimed to assess the annual impact on global warming potential of fruit production expressed as kg equivalent CO_2 kg fruit^{-1} yr^{-1} (ISO14040, 2006; ISO14044, 2006). The following assumptions were made for this LCA:

- The system boundary of this study is considered from the extraction of raw materials of inputs up to the farm gate when the fruits are harvested.
- Data for LCA were taken for the whole life cycle starting from the period of farm establishment till the time of performing this study. Specifically, the LCA was carried out taking in account orchard establishment, cultivation, harvesting and final disposal stages. The nursery stage was excluded, mainly due to the lack of reliable data regarding this phase. The orchard establishment stage included soil preparation, the construction of the fixed structures (irrigation system and supporting structures) and trees plantation. During this stage, the fuel consumption was 430 kg ha^{-1} for peach and pear orchards, and 1117 kg ha^{-1} for Ki30. The cultivation stage included production of fertilisers and their application to the field, pest and weed management substances manufacture and their application, irrigation, pruning, energy use for irrigation and fuel consumption, and machinery use. The mean yearly consumption of electricity, fuel and agrochemicals for the considered orchards are reported in Table 2. The electricity was used for irrigation purposes. In particular, the average water use was 2400, 3240, 2300 and 4130 m^3/ha for Ph10, Ph15, Pr20 and Ki30, respectively. The plants were watered through drip irrigation system. The disposal stage considered the disposing of wastes collected during orchard establishment and cultivation stages to thermal–power plants or to landfills. During the period going from orchards establishment until 2017, the waste production was on average 5.3, 12.5, 15.8 and 25.1 kg ha^{-1} year^{-1} for Ph10, Ph15, Pr20 and Ki30, respectively.
- The LCA took into account the production of the materials (e.g., concrete poles, iron wires and irrigation tubes) used for the construction of the fixed structures in the orchards.
- For fertilizers and agrochemicals production, LCA includes the transport of primary and secondary materials to the production plants, the synthesis of the chemical components and the waste treatment or disposal.
- The LCA included emissions to air of nitrous oxide (N_2O) coming from soil after fertilizations were calculated according to Stehfest and Bouwman (2006).
- For machinery, the performed LCA did include the manufacture, transport, maintenance, repair, and waste management of the machinery used for field operations.
- LCA did not include the transport of raw materials (pesticides, fertilisers, plantlets, poles, etc.) from the local storehouse to farms as well as the production of the packaging used for such raw materials.
- LCA did not include the human labour.

The data used for the life cycle inventory (e.g, fuel consumption, used fertilizers and irrigation) were retrieved from the farmers.

Table 2
Amounts of fuel, electricity and agrochemicals consumed in a 10–years–old peach orchard (Ph10), a 15–years–old peach orchard (Ph15), 20–years–old pear orchard (Pr20) and a 30–years–old kiwi orchard (Ki30).

Input	Unit	Ph10	Ph15	Pr20	Ki30
Fuel consumption	kg ha^{-1}	414	405	528	484
Electricity	kwh ha^{-1}	600	810	575	1944
Agrochemicals	kg ha^{-1}	223	21	51	29

2.4. Calculations and statistical analyses

For the investigated study sites, various calculations were performed, encompassing: soil C stock, expressed as Mg ha^{-1}; the yearly soil C stock gain or loss rate (Csoil) in 0–30 cm depth since the conversion of CK up today, expressed as Mg ha^{-1} yr^{-1}; C balance (C_{bal}), expressed as Mg ha^{-1} yr^{-1}, which is the yearly loss or gain of C of the fruit orchards (with exclusion of plant biomass); the metabolic quotient (qCO_2), expressed as mg C-CO_2 h^{-1} mg C^{-1}_{mic}, which is an indicator of stress in soils (Anderson and Domsch, 1993) and describes the efficiency of the microbial biomass in C use (Pinzari et al., 2017); the microbial quotient (qMIC), expressed as mg C_{mic} g SOC^{-1}, which represents the microbial ability to assimilate soil C (Sun et al., 2020); and the Dilly index which relates soil quality to microbial biomass and respiration (Dilly, 2005) as follows:

$$Cstock = SOC \times th \times BD \times (1 - \%gravel) \times 0.1 \quad (1)$$

where th is the considered soil thickness and %gravel is the gravel amount in the considered soil thickness;

$$C_{soil} = \frac{Cstock\ in\ orchard - Cstock\ in\ CK}{orchard\ age} \quad (2)$$

$$C_{bal} = \frac{Cstock\ in\ orchard - Cstock\ in\ CK}{orchard\ age} - orchard\ mean\ annual\ age \times CLCA \quad (3)$$

where Cbal is the carbon balance, CK is the reference field and CLCA is the C–CO_2eq.

Within the C balance, the C of plant biomass was not considered because it was burned at the end of plants' life.

$$qCO2 = \frac{100 \times SBR}{C_{mic}} \quad (4)$$

$$qMIC = \frac{C_{mic}}{SOC} \quad (5)$$

$$Dilly\ index = \frac{qCO2 \times 1000}{SOC} \quad (6)$$

Two–way analysis of variance was performed to assess the effect of both orchard crop type and soil depth on the selected soil physical, chemical and biochemical parameters. Because of the absence of orchard crop type × soil depth interaction ($P > 0.05$), the effects of both main factors were evaluated through one–way analysis of variance. Prior analysis of variance, the normality and homoscedasticity of residuals were evaluated through graphical analysis and the data were transformed if necessary. To identify statistically significant differences among the means the Tukey's honest significant difference test was conducted as multi–comparison test ($P < 0.05$). The results presented are based on mean values and their standard error. The data were analysed using R software 4.0.3.

Concerning to LCA, SimaPro 8.5.0 software was used to analyze the

life cycle inventory data. SimaPro 8.5.0 is an LCA tool that can be used to monitor the performance of the sustainability of a product or service. This software can analyse a complex life cycle systematically and can evaluate the environmental impact of a product or service at each stage of the life cycle. Ecoinvent 3.4 was chosen as background data sources (Weidema et al., 2013).

3. Results

3.1. Soil physical, chemical and biochemical properties

The SOC concentration and stocks in the 0 – 30 cm depth ranged from 8.02 in CK to 15.36 g kg^{-1} in Ph15 and from 31.6 in CK to 64.4 Mg ha^{-1} in Ph15 (Fig. 2a, b). The TN concentrations varied from 0.96 in CK to 2.03 g kg^{-1} in Pr20 (Fig. 2c).

Comparing the surface layer of the selected orchard crop types, CK had the lowest value of SOC and TN concentration and C stock, while Pr20 had the highest ones. In subsurface soil layer, instead, only the peach orchards showed higher SOC and TN concentrations than CK (Fig. 2a, c), and no differences in C stock occurred among orchard crop types (Fig. 2b).

Between soil layers (0–15 and 15–30 cm), CK soils did not show differences in SOC and TN concentrations, and C stock. Some differences instead occurred in orchards: Ph10, Pr20 and Ki30 showed higher SOC and TN concentrations in surface than in subsurface layer (Fig. 2a, c); Ph15, Pr20 and Ki30 showed higher C stock in surface soil layer than in subsurface one (Fig. 2b).

The water-extractable organic C varied from 112 to 294 mg kg^{-1}, and no differences were found, neither between soil depth nor among orchard crop types (Fig. 2d).

The $\delta^{13}C$ and $\delta^{15}N$ values ranged from −25.20 to −27.29 and from 2.06 to 9.59 ‰ (Fig. 3a and b), respectively. Soils under Pr20 showed less negative value of both $\delta^{13}C$ and $\delta^{15}N$ of organic matter compared to CK (Fig. 3) and this was more pronounced for N where $\delta^{15}N$ in surface

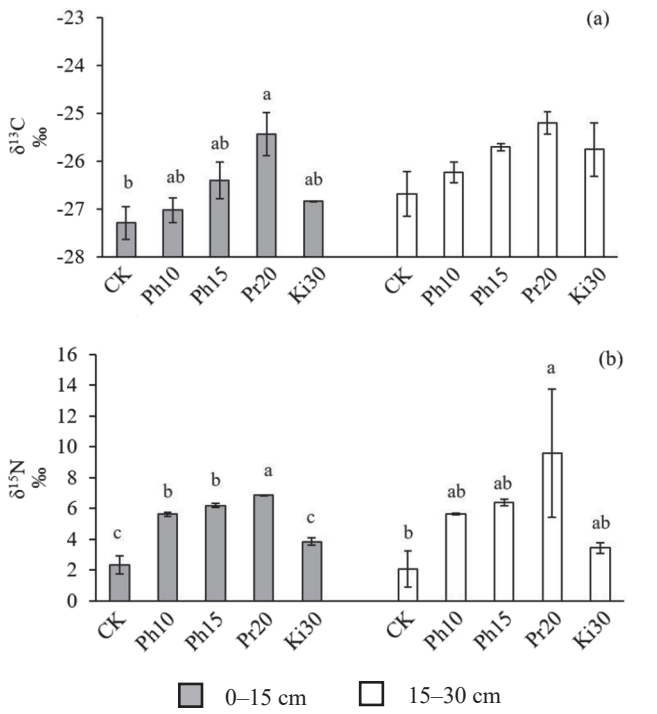

Fig. 3. $\delta^{13}C$ (a) and $\delta^{15}N$ (b) values in 0–15 (grey bars) and 15–30 cm (white bars) soil layers of a field for grains production (CK), a 10–years–old peach orchard (Ph10), a 15–years–old peach orchard (Ph15), a 20–years–old pear orchard (Pr20) and a 30–years–old kiwi orchard (Ki30). Error bars represent standard errors. Within the same soil layer, different lowercase letters indicate significant differences among the fields ($P < 0.05$).

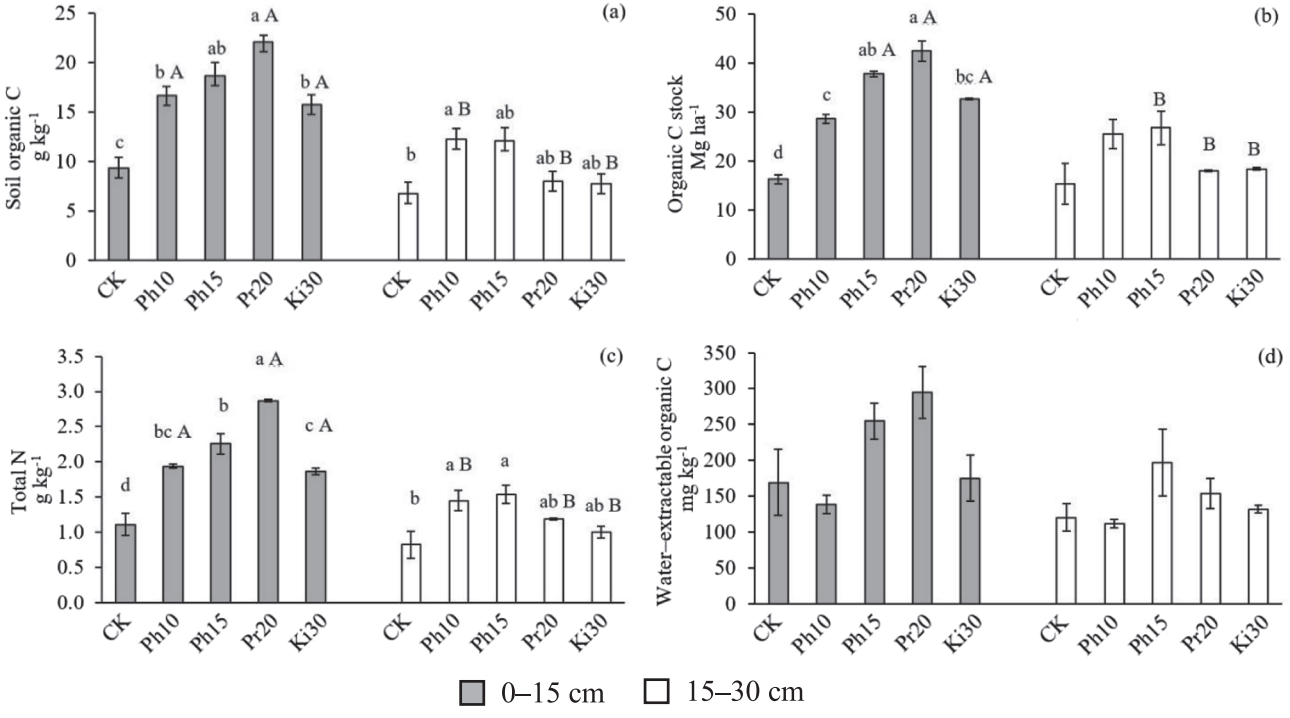

Fig. 2. Soil organic C content (a), organic C stock (b), total N content (c) and water–extractable organic C content (d) in 0–15 (grey bars) and 15–30 cm (white bars) soil depts of a field for grains production (CK), a 10–years–old peach orchard (Ph10), a 15–years–old peach orchard (Ph15), a 20–years–old pear orchard (Pr20) and a 30–years–old kiwi orchard (Ki30). Error bars represent standard errors. Within the same soil depth, different lowercase letters indicate significant differences among the fields ($P < 0.05$). Within the same field, different uppercase letters indicate significant differences between 0 and 15 and 15–30 cm soil depths ($P < 0.05$).

soil layer was the highest value (Fig. 3b).

The SOC pools obtained through chemical fractionation showed the major differences only for the more chemically stable ones (i.e., humic–like C and non–extractable organic C; Fig. 4). Specifically, no humic–like C was found in subsurface soil layers of CK and Pr20, moreover only in Ki30 the surface layer showed higher content of humic–like C compared to subsurface one (Fig. 4c). In the surface layer, the C content associated to NEOM (NEOC) assumed the lowest value in CK (7.35 g kg^{-1}) and it was lower in Ph10 compared to Ph15 and Pr20 (Fig. 4d). Furthermore, NEOC concentration decreased with soil depth in Pr20 and Ki30.

Both soil microbial respiration and C_{mic} content did not differ among the selected fields in surface soil, while some differences occurred for the subsurface soil (Fig. 5a, c). SBR showed higher values in Ph10 than in Pr20 (Fig. 5a) and C_{mic} content showed the lowest value in Pr20 and a higher value in Ph10 than in CK (Fig. 5c). Taking in consideration the soil depth, soil microbial respiration and C_{mic} generally were higher in surface compared to subsurface soil of Ph15, Pr20 and Ki30.

Like microbial respiration and C_{mic} content, no differences of qCO_2 and qMIC occurred among the selected fields in surface soil (Table 3). For the subsurface soil, instead, the Pr20 showed the highest qCO_2 and the lowest qMIC. Moreover, some differences occurred between the two soil depths in Pr20 and Ki30. Specifically, while qCO_2 increased with depth in Pr20 and decreased in Ki30, the opposite occurred for qMIC. The Dilly index showed similar values among the fields in the surface soil ranging from 170 to 570 (Table 3). In the subsurface soil, the Dilly index showed the highest value under Pr20 (2083) and the lowest ones under Ph10 and Ph15 (236 and 331, respectively). Generally, the Dilly index did not change with soil depth with the exception of Pr20 where the subsurface soil had a higher value compared to surface soil.

3.2. CO_2 loss estimation from orchards through life cycle assessment and carbon balance

When looking to the overall impact of the considered orchards on CO_2eq emission, kiwi production presented the greatest impact (Table 4). In all orchards, the main source of CO_2eq is attributed to fertilizers. Specifically, in the investigated orchards the contribution of fertilizers' manufacturing ranged from 21.97 to 33.91 % of the total CO_2eq emissions while the GHGs emission developed after the fertilizers' distribution ranged between 16.47 and 18.12 % of the total CO_2eq emissions. Comparing the considered orchards, Ki30 showed the highest CO_2eq emission from fertilizers use. The lowest CO_2eq emissions related to fertilizers production were observed in Pr20 (0.042 kg CO_2eq kg^{-1} fruit), while the lowest CO_2eq emissions related to fertilizers emissions were observed in Ph10 (0.029 kg CO_2eq kg^{-1} fruit). The agricultural practices during the cultivation period showed to be the second greatest source of GHG, with the exception of Ph10 where the use of agrochemicals accounted for the 22.4 % of total CO_2eq emissions followed by agricultural practices with 21.4 % (Table 4). Unlike fertilizers use, the agricultural practices showed the highest CO_2eq emission value in Pr20. It is interesting to observe the high relevance of orchard establishment on CO_2eq emission ranging from 5.8 % of Ph10 to 21.7 % for Ph15. Because of the scarcity of precipitations during the summer period, irrigation too showed a significant impact on CO_2eq emission, with the highest value in Ki30 (0.045 kg CO_2eq kg^{-1} fruit) and the lowest one in Ph10 (0.0081 kg CO_2eq kg^{-1} fruit).

In the selected orchards, soils showed a yearly increase of organic C stock (C_{soil}) in the 0 – 30 cm depth (Table 5). The highest soil organic C accumulation rate was observed in Ph10 (2294 kg C ha^{-1} $year^{-1}$), while the lowest one was found in Ki30 (646 kg C ha^{-1} $year^{-1}$).

The conversion of a field for grains production to peach and pear orchards had a positive effect on C immobilization (Table 5). Conversely, kiwi cultivation seemed to be an agroecosystem that

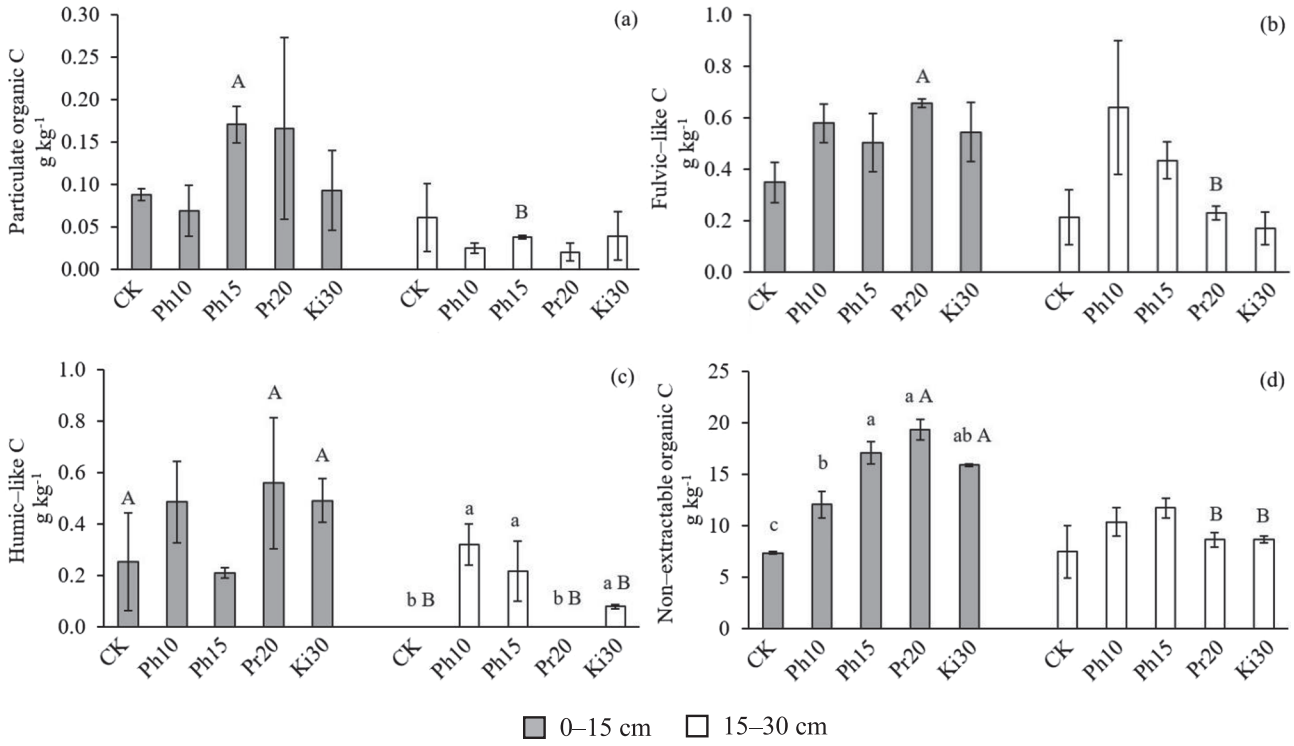

Fig. 4. Concentrations of particulate organic C (a), fulvic–like C (b), humic–like C (c) and non–extractable organic C (d) in 0–15 (grey bars) and 15–30 cm (white bars) soil depths of a field for grains production (CK), a 10–years–old peach orchard (Ph10), a 15–years–old peach orchard (Ph15), a 20–years–old pear orchard (Pr20) and a 30–years–old kiwi orchard (Ki30). Error bars represent standard errors. Within the same soil layer, different lowercase letters indicate significant differences among the fields ($P < 0.05$). Within the same field, different uppercase letters indicate significant differences between 0 and 15 and 15–30 cm soil depths ($P < 0.05$).

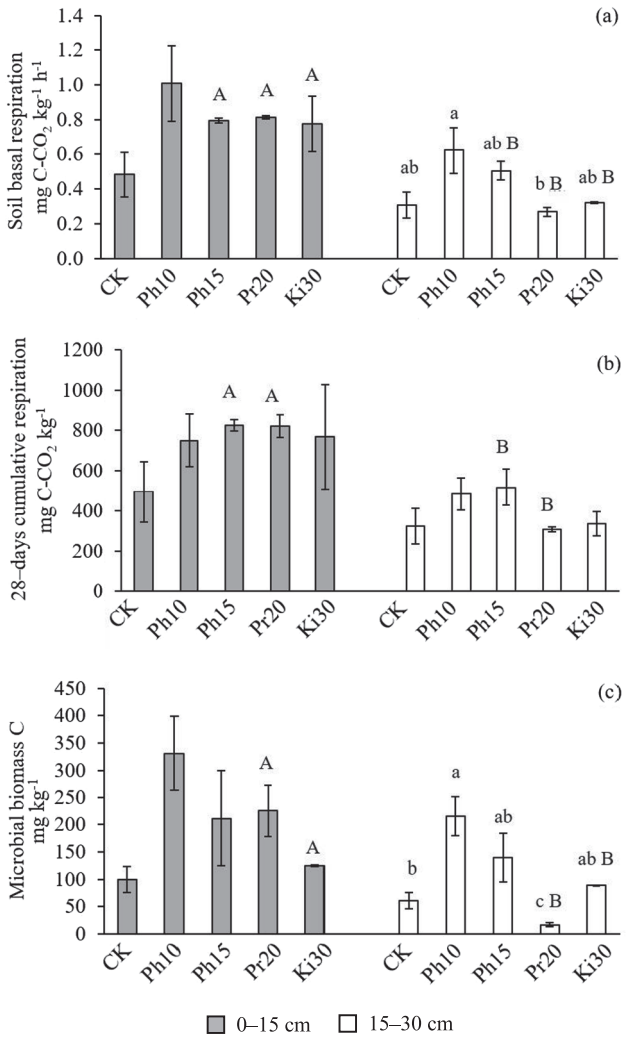

Fig. 5. Soil basal respiration (a), 28–days cumulative respiration (b) and microbial biomass C content (c) in 0–15 (grey bars) and 15–30 cm (white bars) soil depths of a field for grains production (CK), a 10–years–old peach orchard (Ph10), a 15–years–old peach orchard (Ph15), a 20–years–old pear orchard (Pr20) and a 30–years–old kiwi orchard (Ki30). Error bars represent standard errors. Within the same soil layer, different lowercase letters indicate significant differences among the fields ($P < 0.05$). Within the same field, different uppercase letters indicate significant differences between 0 and 15 and 15–30 cm soil depths ($P < 0.05$).

promotes C release to the atmosphere. Specifically, the highest C storage rates (C_{bal}) were observed in peach orchards (1515 and 1580 kg C ha^{-1} year^{-1} in Ph10 and Ph15, respectively), while Ki30 showed a C loss of 117 Mg ha^{-1} year^{-1}.

4. Discussion

4.1. Soil chemical properties

SOC content and C stock of the CK plot (8 g kg^{-1} and 31 Mg ha^{-1}, respectively) were similar to that found in Cambisols of croplands in the Emilia-Romagna region and in the plain of northern Italy (Vittori Antisari et al., 2021a; Brombin et al., 2020; Dal Ferro et al., 2020; Lugato et al., 2007) suggesting its representativeness as reference soil.

The increased SOC concentration and C stock in soils due to the land use change from wheat production to orchard would suggest the capacity of orchards cultivation systems to enrich soil of organic matter. Several studies (e.g., Massaccesi et al., 2018; Neilsen et al., 2014) found

Table 3
Metabolic quotient (qCO_2), microbial quotient (qMIC) and Dilly index in 0–15 and 15–30 cm depth intervals in a reference field (CK), a 10–years–old peach orchard (Ph10), a 15–years–old peach orchard (Ph15), 20–years–old pear orchard (Pr20) and a 30–years–old kiwi orchard (Ki30). Standard error is reported in brackets. Different uppercase letters indicate significant differences between 0 and 15 and 15–30 cm soil depth intervals, different lowercase letters indicate significant differences within the same soil depth interval ($P < 0.05$).

Soil indicator	Soil depth	CK	Ph10	Ph15	Pr20	Ki30
qCO_2 mg C-CO_2 h^{-1} mg Cmic^{-1}	0–15	5.20 (1.28)	3.42 (1.25)	4.59 (1.98)	3.77B (0.74)	6.24 A (1.39)
	15–30	5.07 ab (1.00)	2.89b (0.34)	4.14b (1.69)	16.73 a A (4.58)	3.59b B (0.02)
qMIC mg Cmic g SOC^{-1}	0–15	10.7 (2.2)	20.0 (4.5)	11.1 (3.9)	10.3 A (2.4)	7.9B (0.1)
	15–30	9.2 a (1.2)	18.0 a (3.4)	12.2 a (5.1)	2.1b B (0.44)	11.6 a A (0.6)
Dilly index qCO_2/SOC	0–15	570 (141)	203 (71)	256 (125)	170B (29)	397 (88)
	15–30	791 ab (242)	236c (10)	331c (99)	2083 aA (527)	466 bc (25)

SOC = soil organic carbon content.

an increase in organic carbon amount after orchards establishment. Specifically, a mean C stock of 57 Mg ha^{-1} in 0–30 cm depth was observed which was similar to the values reported by previous studies conducted in Europe (e.g., Álvaro-Fuentes et al., 2012; Bateni et al., 2021; Funes et al., 2019). The increased C stock could be mainly attributed to the presence of a permanent herbaceous plants established on whole surface of the fields which is worldwide recognized to increase soil C stock (de Torres et al., 2021; Xiang et al., 2022; Novara et al., 2019). In fact, the conversion of cropland to grassland promotes SOC storage (Auerswald and Fiener, 2019) due to the higher root turnover in grasslands compared to cropland and due to the harvest of the whole aboveground biomass in cropland (Poeplau and Don, 2013). Since root derived C through rhizodeposition processes and root turnover (De Feudis et al., 2016; Douglas et al., 2020) has been identified as the major source of SOC (Rasse et al., 2005), the presence of trees and perennial grasses may explain higher SOC accumulation in orchards compared to CK. Such differences were marked in surface soil mainly due the generally larger distribution of roots in the surface soil (Forey et al., 2017; Ruiz-Sánchez et al., 2005; Sokalska et al., 2009) and to the degradation of the chopped pruning residues left on soil surface (Massaccesi et al., 2018; Zhao et al., 2017). The greater influence of fruit orchards on surface soil compared to subsurface soil can be confirmed by the higher SOC content and C stock in the former in Ph15, Pr20 and Ki30. Because of the role of SOC on soil microbial activity (e.g., Martínez-García et al., 2018), the higher amount of organic matter in the surface soil might explain the generally higher soil microbial respiration and biomass in the superficial soil layer.

The generally homogeneous $\delta^{13}C$ values would indicate that orchard cultivation did not affect the organic matter decomposition (Blagodatskaya et al., 2011; Solomon et al., 2002). The unchanged SOC decomposition rate could be confirmed by the negligible differences between CK and the considered orchards of those biochemical indicators (i.e., SBR, RCUM, C_{mic}, qCO_2 and qMIC) related to C cycle. The similar SOC degradation combined with the high organic material input due to the shredded pruning residues might have promoted an accumulation of NEOC in the surface soil of the orchards. The plant residues could release water–insoluble compounds (e.g., lignin and waxes) and labile substances readily available to microorganisms whose cell residues could bind to soil minerals increasing the NEOM fraction (Hayes et al., 2017; Wang et al., 2021).

Like SOC content and C stock, the cultivation of fruit orchards increased the TN content in surface soil. This can be attributed to the addition of N by amendment (i.e., in Ph10) and chemical fertilizers. The

Table 4
Amounts and percentage distribution of carbon dioxide equivalent emitted from the establishment, cultivation and disposal stages of a 10–years–old peach orchard (Ph10), a 15–years–old peach orchard (Ph15), 20–years–old pear orchard (Pr20) and a 30–years–old kiwi orchard (Ki30).

Site	Unit	Establishment stage	Cultivation stage					Disposal stage	Total
			Agricultural practices	Irrigation	Fertilizer production	Fertilizer emissions	Agrochemicals	Wastes	
Ph10	kg CO_2eq kg^{-1} fruit yr^{-1}	0.010	0.037	0.0081	0.049	0.029	0.039	0.00081	0.17
	%	5.84	21.42	4.67	28.40	16.78	22.43	0.47	
Ph15	kg CO_2eq kg^{-1} fruit yr^{-1}	0.041	0.043	0.015	0.053	0.034	0.0015	0.0012	0.19
	%	21.72	23.02	7.80	27.95	18.12	0.78	0.62	
Pr20	kg CO_2eq kg^{-1} fruit yr^{-1}	0.026	0.066	0.012	0.042	0.034	0.0094	0.0021	0.19
	%	13.26	34.52	6.39	21.97	17.88	4.90	1.07	
Ki30	kg CO_2eq kg^{-1} fruit yr^{-1}	0.033	0.067	0.045	0.100	0.049	0.0024	0.0026	0.30
	%	10.85	22.23	14.87	33.91	16.47	0.80	0.88	

The establishment stage included soil preparation, the construction of the fixed structures (irrigation system and supporting structures) and trees plantation. Agricultural practices included fuel consumption, machinery use, pruning, pest and weed control, fertilizers distribution. Fertilizer production equates to the kg CO_2eq emission related to the industrial production phase of fertilizers. Fertilizer emissions equates to the kg CO_2eq of green–house gas emissions form soil (e.g., N_2O) once the fertilizers were distributed. Agrochemicals equates to the kg CO_2eq emission related to the industrial production phase of them. Wastes equates to the disposing of wastes collected during orchard establishment and cultivation stages to thermal–power plants or to landfills.

Table 5
Yearly C loss from fruit production practices (C–LCA), yearly C stock change (C soil) and C soil - C–LCA (Cbal) of a 10–years–old peach orchard (Ph10), a 15–years–old peach orchard (Ph15), 20–years–old pear orchard (Pr20) and a 30–years–old kiwi orchard (Ki30).

	Unit	Ph10	Ph15	Pr20	Ki30
C–LCA	kg C–CO_2eq ha^{-1} year^{-1}	734	611	518	763
C soil	kg C ha^{-1} year^{-1}	2249	2191	1440	646
Cbal	kg C ha^{-1} year^{-1}	1515	1580	922	−117

higher $\delta^{15}N$ values in orchards compared to the wheat field might be attributed both to the contribution of N–enriched fertilizers to $\delta^{15}N$ values and to the preferential microbial utilization of ^{14}N compounds (Boström et al., 2007; Lobe et al., 2005). The latter maybe limited under Ki30.

It was interesting to note that for the subsurface soil, among the selected orchards, Pr20 showed the lowest humic–like C content which would cause a limited SOC stabilization (Martins Gomes et al., 2018). The limited SOC stabilization might be due to the less suitable conditions for the soil microbial community which did not allow the transformation of the soil organic matter (Liebich et al., 2007). In fact, the subsurface soil of Pr20 also showed the lowest C_{mic}, qMic and the highest qCO_2 indicating a lower C use efficiency by the microbial community (Anderson, 2003; Anderson and Domsch, 1989; Okolo et al., 2020) compared to other fields and, therefore, the occurrence of poor conditions (Vittori Antisari et al., 2021a, 2021b). Such unfavourable conditions in subsurface soil for Pr20 was confirmed by the very high Dilly index value, which would suggest the worsening of the energy use efficiency by the microbial community, in turn not promoting organic C accumulation (Dilly, 2005).

4.2. Life cycle assessment

In agreement with previous studies (e.g., Romero-Gámez et al., 2017; Vinyes et al., 2017), this study found that fertilization was the procedure that had the greatest impact on CO_2eq emission from the orchards, accounting for at least 40 % of total CO_2eq emission. In this context, it was interesting to observe that, although in Ph10 no chemical fertilizers were applied, the use of organic amendment had a great impact on CO_2eq emissions. In fact, organic amendment production is both an energy-intensive process and a source of methane and nitrous oxide while its application causes N_2O emission (Bacenetti et al., 2016; Galgani et al., 2014). However, because of the greatest use of N and P fertilizers, the highest CO_2eq emission related to fertilizers was observed in Ki30. Indeed, N and P fertilizers are considered highly impacting on climate change, fossil fuel depletion, acidification, eutrophication, and resources depletion (Hasler et al., 2015). This result, together with the highest CO_2eq emission related to the irrigation, would indicate the higher demands of nutrients and water of kiwi plants compared to peach and pear trees (Allen et al., 1998; Carranca et al., 2018; Peticila et al., 2015).

The consume of fuel related to agricultural practices as tillage, weed control and pruning showed to be the second most important CO_2eq source. In this sense, Milà I Canals et al. (2006) suggested the use of biofuel in order to limit the impact of the agricultural practices on CO_2 emission.

Several studies (e.g., Martin-Gorriz et al., 2020; Vinyes et al., 2017) reported the high impact of agrochemicals on CO_2eq released into the atmosphere. However, in this study the contribution of agrochemicals on CO_2eq emission in Ph15, Pr20 and Ki30 resulted low due to the sustainable approach used on the studied farms. In this context, it was important to highlight the greater contribution of agrochemicals on CO_2eq emissions for Ph10. In this case, the amounts of agrochemicals used was 10 times higher than those used in the other orchards, and they were mainly sulphur based. This higher amounts of agrochemicals can be attributed to the types of agrochemicals generally used in organic farming. These findings are in agreement with the work of Longo et al. (2017) which observed a larger use of pesticides to produce organic apples compared to those produced with conventional approaches.

Overall, this study clearly showed how kiwifruit cultivation had the highest impact on GHG emissions mainly due to the high water and nutrient demand, suggesting that such tree species is less suitable than peach and pear for the considered study area.

4.3. Carbon balance

When taking in account the C–CO_2eq loss by fruit cultivation and C gained and stored into the soil, results from this study would indicate that peach and pear orchard ecosystems promote C sequestration. The capability of the studied orchards to sequester C was mainly attributed to the soil on which they grow. In fact, the investigated soil was able to store each year a large amount of organic C. Notably, such C was stored in the most stable form preventing C to go back to the atmosphere as CO_2 in the short– or mid–term. It is important to note that in the present study we did not consider C fixed in plant biomass because it is not a long-living component. In fact, orchards for fruit production generally have a lifetime of few decades. Also, at the end of the cultivation period the plant biomass is removed and burnt on the field or in thermal power plants or processed for pellet production which are common practices for fruit orchards (Brand and Jacinto, 2020; Giuntoli et al., 2016). Conversely, the organic carbon stored as fulvic-like C, humic-like C and

NEOC could have a mean residence time which spans from centuries to thousands of years (Certini et al., 2004; Piccolo, 2002).

The generally similar soil microbial efficiency to use C and, therefore, to transform C in stable forms, together with similar $\delta^{13}C$ values and soil characteristics (e.g., clay content) between the orchards and the reference field would indicate that C sequestration was mainly related to the management practices carried–out in each orchard.

Taking in consideration each orchard type, it is important to mention the negative C balance (-117 kg C ha^{-1} $year^{-1}$) of Ki30. The negative value can be mainly attributed to the high inputs (fertilizers and irrigation) requested by the kiwi plants which caused large CO_2eq emission just partly counterbalanced by soil carbon storage processes. Indeed, when taking in consideration the soil environment, generally no differences in SOC content and its chemical forms were found among the selected orchards. Unlike Ki30, Pr20 showed similar values of CO_2eq emissions of peach orchards (Table 5) but a lower mean annual C storage increase (Table 5). The weak mean annual C storage increase in Pr20 could be attributed to the more stressful conditions for the microbial biomass in subsurface soils. Overall, the C balance performed in this study by taking in consideration the topsoil highlighted the importance of SOC sequestration into the LCA of agricultural systems. However, because of its pivotal role on C storage (Guillaume et al., 2022; Antony et al., 2022) and its greater influence on the agricultural managements compared to topsoil (Samson et al., 2021; Osanai et al., 2020), future LCA studies should take into consideration the subsoil and its key role in the overall C cycle.

5. Conclusions

The results from the present study suggest that the conversion of a field from grains production to the fruit orchards cultivation promoted soil carbon gain. The majority of the gained C was found in the most chemically recalcitrant form suggesting that in the selected fruit orchards the C stabilization processes were promoted. The organic C increase in orchards could be mainly attributed to the permanent grasses covering such fields. However, such increase could be also promoted both by the direct release from plant residues of chemically recalcitrant compounds and by the release of readily available C for microorganisms whose necromass could bind to soil mineral particles. However, the C gain rate is not unlimited as it depends on soil properties (e.g., clay content) as well as on orchard management. For example, in Ki30, soil stored C, but it was not able to counterbalance the GHG emissions coming from the cultivation of kiwi though it had similar clay content and similar biochemical properties of the reference field. A key tool in this sense may therefore be LCA as it allows us to take into consideration soil resources and their contribution. The systematic inclusion of soil in LCA would allow to enhance agroecosystems sustainability and give soil resources their rightful place in the quest to tackle sustainable development goals and combat climate change. Therefore, we propose to insert the soil C storage rate as CO_2 soil uptake from atmosphere lowing the environmental impacts of orchards management. Finally, although the present study only considered topsoil (0–30 cm depth), in future LCA procedures that also considered deep soil would provide an important additions to give a more realistic view of the role of soil on the mitigation of the GHG emissions coming from the cultivation practices.

Declaration of Competing Interest

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.

Data availability

Data will be made available on request.

Acknowledgement

This research has been co-funded by the Rural Development Program (RDP; Programma di Sviluppo Rurale-PSR) 2014–2020 of the Emilia-Romagna region, grant numbers: 5015501 (Fruttifi-CO project).

Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.catena.2022.106563.

References

Adewale, C., Reganold, J.P., Higgins, S., Evans, R.D., Carpenter-Boggs, L., 2019. Agricultural carbon footprint is farm specific: case study of two organic farms. J. Clean. Prod. 229, 795–805. https://doi.org/10.1016/j.jclepro.2019.04.253.

Agnelli, A., Bol, R., Trumbore, S.E., Dixon, L., Cocco, S., Corti, G., 2014. Carbon and nitrogen in soil and vine roots in harrowed and grass-covered vineyards. Agric. Ecosyst. Environ. 193, 70–82. https://doi.org/10.1016/j.agee.2014.04.023.

Aguilera, E., Guzmán, G., Alonso, A., 2015. Greenhouse gas emissions from conventional and organic cropping systems in Spain. II. Fruit tree orchards. Agron. Sustain. Dev. 35 (2), 725–737. https://doi.org/10.1007/s13593-014-0265-y.

Allen, R.G., Pereira, L.S., Raes, D., Smith, M., 1998. Crop evapotranspiration. Guidelines for computing crop water requirements. FAO Irrigation and Drainage Paper No. 56. FAO, Rome, Italy.

Álvaro-Fuentes, J., Easter, M., Paustian, K., 2012. Climate change effects on organic carbon storage in agricultural soils of Northeastern SpainAgric. Ecosyst. Environ. 155, 87–94. https://doi.org/10.1016/j.agee.2012.04.001.

Anderson, T.-H., 2003. Microbial eco-physiological indicators to asses soil quality. Agric. Ecosyst. Environ. 98 (1-3), 285–293. https://doi.org/10.1016/S0167-8809(03)00088-4.

Anderson, T.-H., Domsch, K.H., 1989. Ratios of microbial biomass carbon to total organic carbon in arable soils. Soil Biol. Biochem. 21 (4), 471–479. https://doi.org/10.1016/0038-0717(89)90117-X.

Anderson, T., Domsch, K., 1993. The metabolic quotient for CO2 (qCO2) as a specific activity parameter to assess the effects of environmental conditions, such as ph, on the microbial biomass of forest soils. Soil Biol. Biochem. 25 (3), 393–395. https://doi.org/10.1016/0038-0717(93)90140-7.

Angst, G., Mueller, K.E., Nierop, K.G.J., Simpson, M.J., 2021. Plant-or microbial-derived? A review onthe molecular compositionof stabilizedsoil organic matter. Soil Biol. Biochem. 156, 108189. https://doi.org/10.1016/j.soilbio.2021.108189.

Antony, D., Collins, C.D., Clark, J.M., Sizmur, T., 2022. Soil organic matter storage in temperate lowland arable, grassland and woodland topsoil and subsoil. Soil Use Manage. 00, 1–15. https://doi.org/10.1111/sum.12801.

Arzoumanidis, I., Fullana-I-Palmer, P., Raggi, A., Gazulla, C., Raugei, M., Benveniste, G., Anglada, M., 2014. Unresolved issues in the accounting of biogenic carbon exchanges in the wine sector. J. Clean. Prod. 82, 16–22. https://doi.org/10.1016/j.jclepro.2014.06.073.

Auerswald, K., Fiener, P., 2019. Soil organic carbon storage following conversion from cropland to grassland on sites differing in soil drainage and erosion history. Sci. Total Environ. 661, 481–491. https://doi.org/10.1016/j.scitotenv.2019.01.200.

Bacenetti, J., Fusi, A., Negri, M., Bocchi, S., Fiala, M., 2016. Organic production systems: sustainability assesement of rice in Italy. Agric. Ecosyst. Environ. 225, 33–44. https://doi.org/10.1016/j.agee.2016.03.046.

Bartzas, G., Vamvuka, D., Komnitsas, K., 2017. Comparative life cycle assessment of pistachio, almond and apple production. Inf. Process. Agric. 4 (3), 188–198. https://doi.org/10.1016/j.inpa.2017.04.001.

Bateni, C., Ventura, M., Tonon, G., Pisanelli, A., 2021. Soil carbon stock in olive groves agroforestry systems under different management and soil characteristics. Agrofor. Syst. 95 (5), 951–961. https://doi.org/10.1007/s10457-019-00367-7.

Bationo, A., Kihara, J., Vanlauwe, B., Waswa, B., Kimetu, J., 2007. Soil organic carbon dynamics, functions and management in West African agro-ecosystems. Agric. Syst. 94 (1), 13–25. https://doi.org/10.1016/j.agsy.2005.08.011.

Blagodatskaya, E., Yuyukina, T., Blagodatsky, S., Kuzyakov, Y., 2011. Turnover of soil organic matter and of microbial biomass under C3–C4 vegetation change: consideration of 13C fractionation and preferential substrate utilization. Soil Biol. Biochem. 43 (1), 159–166. https://doi.org/10.1016/j.soilbio.2010.09.028.

Boström, B., Comstedt, D., Ekblad, A., 2007. Isotope fractionation and 13C enrichment in soil profiles during the decomposition of soil organic matter. Oecologia 153 (1), 89–98. https://doi.org/10.1007/s00442-007-0700-8.

Bradford, M.A., Carey, C.J., Atwood, L., Bossio, D., Fenichel, E.P., Gennet, S., Fargione, J., Fisher, J.R.B., Fuller, E., Kane, D.A., Lehmann, J., Oldfield, E.E., Ordway, E.M., Rudek, J., Sanderman, J., Wood, S.A., 2019. Soil carbon science for policy and practice. Nat. Sustain. 2 (12), 1070–1072. https://doi.org/10.1038/s41893-019-0431-y.

Brand, M.A., Jacinto, R.C., 2020. Apple pruning residues: Potential for burning in boiler systems and pellet production. Renew. Energy 152, 458–466. https://doi.org/10.1016/j.renene.2020.01.037.

Brandão, M., Levasseur, A., Kirschbaum, M.U.F., Weidema, B.P., Cowie, A.L., Jørgensen, S.V., Hauschild, M.Z., Pennington, D.W., Chomkhamsri, K., 2013. Key issues and options in accounting for carbon sequestration and temporary storage in

life cycle assessment and carbon footprinting. Int. J. Life Cycle Assess. 18 (1), 230–240. https://doi.org/10.1007/s11367-012-0451-6.
Brombin, V., Mistri, E., De Feudis, M., Forti, C., Salani, G.M., Natali, C., Falsone, G., Vittori Antisari, L., Bianchini, G., 2020. Soil carbon investigation in three pedoclimatic and agronomic settings of northern Italy. Sustain. 12, 1–19. https://doi.org/10.3390/su12410539.
Bronick, C.J., Lal, R., 2005. Soil structure and management: a review. Geoderma 124 (1-2), 3–22. https://doi.org/10.1016/j.geoderma.2004.03.005.
Carranca, C., Brunetto, G., Tagliavini, M., 2018. Nitrogen nutrition of fruit trees to reconcile productivity and environmental concerns. Plants 7 (1), 4. https://doi.org/10.3390/plants7010004.
Certini, G., Agnelli, A., Corti, G., Capperucci, A., 2004. Composition and mean residence time of molecular weight fractions of organic matter extracted from two soils under different forest species. Biogeochemistry 71 (3), 299–316. https://doi.org/10.1023/B:BIOG.0000049345.11312.03.
Chantigny, M., Angers, D., Kaiser, K., Kalbitz, K., 2007. Extraction and Characterization of Dissolved Organic Matter. In: Carter, M.R., Gregorich, E.G. (Eds.), Soil Sampling and Methods of Analysis, Second Edition. CRC Press Taylor & Francis, Boca Raton. https://doi.org/10.1201/9781420005271.ch48.
Chavarria, D.N., Pérez-Brandan, C., Serri, D.L., Meriles, J.M., Restovich, S.B., Andriulo, A.E., Jacquelin, L., Vargas-Gil, S., 2018. Response of soil microbial communities to agroecological versus conventional systems of extensive agriculture. Agric. Ecosyst. Environ. 264, 1–8.
Chenu, C., Angers, D.A., Barré, P., Derrien, D., Arrouays, D., Balesdent, J., 2019. Increasing organic stocks in agricultural soils: knowledge gaps and potential innovations. Soil Tillage Res. 188, 41–52. https://doi.org/10.1016/j.still.2018.04.011.
Ciais, P., Sabine, C., Bala, G., Bopp, L., Brovkin, V., Canadell, J., Chhabra, A., DeFries, R., Galloway, J., Heimann, M., Jones, C., Le Quéré, C., Myneni, R.B., Piao, S., Thornton, P., 2013. Carbon and Other Biogeochemical Cycles. In: Plattner, M., Tignor, S.K., Allen, J., Boschung, A., Nauels, Y., Xia, V., Midgley, P.M. (Eds.), ClImate Change 2013: the PhysIcal ScIence BasIs. ContrIbutIon of WorkIng Group I to the FIfth Assessment Report of the Intergovernmental Panel on ClImate Change. UK and NY. Cambridge University Press, Cambridge.
Dal Ferro, N., Piccoli, I., Berti, A., Polese, R., Morari, F., 2020. Organic carbon storage potential in deep agricultural soil layers: Evidence from long-term experiments in northeast Italy. Agric. Ecosyst. Environ. 300, 106967. https://doi.org/10.1016/j.agee.2020.106967.
De Feudis, M., Cardelli, V., Massaccesi, L., Bol, R., Willbold, S., Cocco, S., Corti, G., Agnelli, A., 2016. Effect of beech (*Fagus sylvatica* L.) rhizosphere on phosphorous availability in soils at different altitudes (central Italy). Geoderma 276, 53–63. https://doi.org/10.1016/j.geoderma.2016.04.028.
De Feudis, M., Cardelli, V., Massaccesi, L., Trumbore, S.E., Vittori Antisari, L., Cocco, S., Corti, G., Agnelli, A., 2019. Small altitudinal change and rhizosphere affect the SOM light fractions but not the heavy fraction in European beech forest soil. Catena 181, 104091. https://doi.org/10.1016/j.catena.2019.104091.
Dilly, O., 2005. Microbial Energetics in Soils. In: Varma, A., Buscot, F. (Eds.), Soil BiologyMicroorganisms in Soils: Roles in Genesis and Functions. Springer-Verlag, Berlin/Heidelberg, pp. 123–138. https://doi.org/10.1007/3-540-26609-7_6.
Domeignoz-Horta, L.A., Shinfuku, M., Junier, P., Poirier, S., Verrecchia, E., Sebag, D., De Angelis, K.M., 2021. Direct evidence for the role of microbial community composition in the formation of soil organic matter composition and persistence. ISME Commun. 1, 64. https://doi.org/10.1038/s43705-021-00071-7.
Douglas, G., Mackay, A., Vibart, R., Dodd, M., McIvor, I., McKenzie, C., 2020. Soil carbon stocks under grazed pasture and pasture-tree systems. Sci. Total Environ. 715, 136910. https://doi.org/10.1016/j.scitotenv.2020.136910.
Falsone, G., Marinari, S., Vittori Antisari, L., Vianello, G., 2015. Soil processes related to organic matter modifications following Douglas-fir mature reforestation. Biol. Fertil. Soils 51 (3), 277–287. https://doi.org/10.1007/s00374-014-0971-2.
Forey, O., Temani, F., Wery, J., Jourdan, C., Metay, A., 2017. Effect of combined deficit irrigation and grass competition at plantation on peach tree root distribution. Eur. J. Agron. 91, 16–24. https://doi.org/10.1016/j.eja.2017.08.008.
Foteinis, S., Chatzisymeon, E., 2016. Life cycle assessment of organic versus conventional agriculture. A case study of lettuce cultivation in Greece. J. Clean. Prod. 112, 2462–2471. https://doi.org/10.1016/j.jclepro.2015.09.075.
Funes, I., Savé, R., Rovira, P., Molowny-Horas, R., Alcañiz, J.M., Ascaso, E., Herms, I., Herrero, C., Boixadera, J., Vayreda, J., 2019. Agricultural soil organic carbon stocks in the north-eastern Iberian Peninsula: drivers and spatial variability. Sci. Total Environ. 668, 283–294. https://doi.org/10.1016/j.scitotenv.2019.02.317.
Fuss, S., Lamb, W.F., Callaghan, M.W., Hilaire, J., Creutzig, F., Amann, T., Beringer, T., de Oliveira Garcia, W., Hartmann, J., Khanna, T., Luderer, G., Nemet, G.F., Rogelj, J., Smith, P., Vicente, J.L.V., Wilcox, J., del Mar Zamora Dominguez, M., Minx, J.C., 2018. Negative emissions—Part 2: costs, potentials and side effects. Environ. Res. Lett. 13 (6), 063002. https://doi.org/10.1088/1748-9326/aabf9f.
Galgani, P., van der Voet, E., Korevaar, G., 2014. Composting, anaerobic digestion and biochar production in Ghana. Environmental-economic assessment in the context of voluntary carbon markets. Waste Manag. 34 (12), 2454–2465. https://doi.org/10.1016/j.wasman.2014.07.027.
Garrigues, E., Corson, 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.
Giuntoli, J., Agostini, A., Caserini, S., Lugato, E., Baxter, D., Marelli, L., 2016. Climate change impacts of power generation from residual biomass. Biomass and Bioenergy 89, 146–158. https://doi.org/10.1016/j.biombioe.2016.02.024.
Gkisakis, V.D., Volakakis, N., Kosmas, E., Kabourakis, E.M., 2020. Developing a decision support tool for evaluating the environmental performance of olive production in terms of energy use and greenhouse gas emissions. Sustain. Prod. Consum. 24, 156–168. https://doi.org/10.1016/j.spc.2020.07.003.
Goossens, Y., Annaert, B., De Tavernier, J., Mathijs, E., Keulemans, W., Geeraerd, A., 2017. Life cycle assessment (LCA) for apple orchard production systems including low and high productive years in conventional, integrated and organic farms. Agric. Syst. 153, 81–93.
Guevara, M., Arroyo, C., Brunsell, N., Cruz, C.O., Domke, G., Equihua, J., Etchevers, J., Hayes, D., Hengl, T., Ibelles, A., Johnson, K., de Jong, B., Libohova, Z., Llamas, R., Nave, L., Ornelas, J.L., Paz, F., Ressl, R., Schwartz, A., Victoria, A., Wills, S., Vargas, R., 2020. Soil organic carbon across Mexico and the conterminous United States (1991–2010). Global Biogeochemical Cycles 34 (3). https://doi.org/10.1029/2019GB006219.
Guillaume, T., Makowski, D., Libohova, Z., Elfouki, S., Fontana, M., Leifeld, J., Bragazza, L., Sinaj, S., 2022. Carbon storage in agricultural topsoils and subsoils is promoted by including temporary grasslands into the crop rotation. Geoderma 422, 115937. https://doi.org/10.1016/j.geoderma.2022.115937.
Haas, G., Wetterich, F., Geier, U., 2000. Life cycle assessment framework in agriculture on the farm level. Int. J. Life Cycle Assess. 5, 345–348. https://doi.org/10.1007/BF02978669.
Hasler, K., Bröring, S., Omta, S.W.F., Olfs, H.W., 2015. Life cycle assessment (LCA) of different fertilizer product types. Eur. J. Agron. 69, 41–51. https://doi.org/10.1016/j.eja.2015.06.001.
Hayes, M.H.B., Mylotte, R., Swift, R.S., 2017. Humin: its composition and importance in soil organic matter. Adv. Agron. 143, 47–138. https://doi.org/10.1016/bs.agron.2017.01.001.
ISO International Organization for Standardization, 14040, 2006. Environmental Management - Life Cycle Assessment e Principles and Framework. ISO, Geneva.
ISO International Organization for Standardization, 14044, 2006. Environmental Management - Life Cycle Assessment e Requirements and Guidelines. ISO, Geneva.
Kaab, A., Sharifi, M., Mobli, H., Nabavi-Pelesaraei, A., Chau, K.W., 2019. Combined life cycle assessment and artificial intelligence for prediction of output energy and environmental impacts of sugarcane production. Sci. Total Environ. 664, 1005–1019. https://doi.org/10.1016/j.scitotenv.2019.02.004.
Keel, S.G., Anken, T., Büchi, L., Chervet, A., Fliessbach, A., Flisch, R., Huguenin-Elie, O., Mäder, P., Mayer, J., Sinaj, S., Sturny, W., Wüst-Galley, C., Zihlmann, U., Leifeld, J., 2019. Loss of soil organic carbon in Swiss long-term agricultural experiments over a wide range of management practices. Agric. Ecosyst. Environ. 286, 106654. https://doi.org/10.1016/j.agee.2019.106654.
Lal, R., 2018. Digging deeper: a holistic perspective of factors affecting soil organic carbon sequestration in agroecosystems. Glob. Change Biol. 24 (8), 3285–3301. https://doi.org/10.1111/gcb.14054.
Lawrence, N.C., Tenesaca, C.G., VanLoocke, A., Hall, S.J., 2021. Nitrous oxide emissions from agricultural soils challenge climate sustainability in the US Corn Belt. Proc. Natl. Acad. Sci. U. S. A 118 (46) e2112108118. https://doi.org/10.1073/pnas.211210811.
Liebich, J., Schloter, M., Schäffer, A., Vereecken, H., Burauel, P., 2007. Degradation and humification of maize straw in soil microcosms inoculated with simple and complex microbial communities. Eur. J. Soil Sci. 58 (1), 141–151. https://doi.org/10.1111/j.1365-2389.2006.00816.x.
Lobe, I., Bol, R., Ludwig, B., Du Preez, C.C., Amelung, W., 2005. Savanna-derived organic matter remaining in arable soils of the South African Highveld long-term mixed cropping: Evidence from 13C and 15N natural abundance. Soil Biol. Biochem. 37 (10), 1898–1909. https://doi.org/10.1016/j.soilbio.2005.02.030.
Longo, S., Mistretta, M., Guarino, F., Cellura, M., 2017. Life Cycle Assessment of organic and conventional apple supply chains in the North of Italy. J. Clean. Prod. 140, 654–663. https://doi.org/10.1016/j.jclepro.2016.02.049.
Lugato, E., Paustian, K., Giardini, L., 2007. Modelling soil organic carbon dynamics in two long-term experiments of north-eastern Italy. Agric. Ecosyst. Environ. 120 (2-4), 423–432. https://doi.org/10.1016/j.agee.2006.11.006.
Lynch, J., Cain, M., Frame, D., Pierrehumbert, R., 2021. Agriculture's Contribution to Climate Change and Role in Mitigation Is Distinct From Predominantly Fossil CO2-Emitting Sectors. Front. Sustain. Food Syst. 4, 518039 https://doi.org/10.3389/fsufs.2020.518039.
Makipaa, R., Liski, J., Guendehou, S., Malimbwi, R., Kaaya, A., 2012. Soil carbon monitoring using surveys and modelling: General description and application in the United Republic of Tanzania. Food and Agriculture Organization of the United Nations, Rome.
Martin, M.P., Orton, T.G., Lacarce, E., Meersmans, J., Saby, N.P.A., Paroissien, J.B., Jolivet, C., Boulonne, L., Arrouays, D., 2014. Evaluation of modelling approaches for predicting the spatial distribution of soil organic carbon stocks at the national scale. Geoderma 223–225, 97–107. https://doi.org/10.1016/j.geoderma.2014.01.005.
Martínez-García, L.B., Korthals, G., Brussaard, L., Jørgensen, H.B., De Deyn, G.B., 2018. Organic management and cover crop species steer soil microbial community structure and functionality along with soil organic matter properties. Agric. Ecosyst. Environ. 263, 7–17. https://doi.org/10.1016/j.agee.2018.04.018.
Martínez-Mena, M., Perez, M., Almagro, M., Garcia-Franco, N., Díaz-Pereira, E., 2021. Long-term effects of sustainable management practices on soil properties and crop yields in rainfed Mediterranean almond agroecosystems. Eur. J. Agron. 123, 126207. https://doi.org/10.1016/j.eja.2020.126207.
Martin-Gorriz, B., Gallego-Elvira, B., Martínez-Alvarez, V., Maestre-Valero, J.F., 2020. Life cycle assessment of fruit and vegetable production in the Region of Murcia (south-east Spain) and evaluation of impact mitigation practices. J. Clean. Prod. 265, 121656. https://doi.org/10.1016/j.jclepro.2020.121656.
Martins Gomes, E.T., Berbara, R.L.L., Pereira, M.G., Urquiaga, S.S., Tavares, O.C.H., Assunção, S.A., Zonta, E., do Amaral Sobrinho, N.M.B., García, A.C., 2018. Effects of

farmed managements in sandy soils on composition and stabilization of soil humic substances. L. Degrad. Dev. 29 (1), 68–79. https://doi.org/10.1002/ldr.2839.
Massaccesi, L., De Feudis, M., Agnelli, A.E., Nasini, L., Regni, L., D'Ascoli, R., Castaldi, S., Proietti, P., Agnelli, A., 2018. Organic carbon pools and storage in the soil of olive groves of different age. Eur. J. Soil Sci. 69 (5), 843–855. https://doi.org/10.1111/ejss.12677.
Milà i Canals, L., Burnip, G.M., Cowell, S.J., 2006. Evaluation of the environmental impacts of apple production using Life Cycle Assessment (LCA): Case study in New Zealand. Agric. Ecosyst. Environ. 114 (2-4), 226–238. https://doi.org/10.1016/j.agee.2005.10.023.
Mondal, S., Chakraborty, D., 2022. Global meta-analysis suggests that no-tillage favourably changes soil structure and porosity. Geoderma 405, 115443. https://doi.org/10.1016/j.geoderma.2021.115443.
Montanaro, G., Xiloyannis, C., Nuzzo, V., Dichio, B., 2017. Orchard management, soil organic carbon and ecosystem services in Mediterranean fruit tree crops. Sci. Hortic. (Amsterdam) 217, 92–101. https://doi.org/10.1016/j.scienta.2017.01.012.
Morugán-Coronado, A., Linares, C., Gómez-López, M.D., Faz, Á., Zornoza, R., 2020. The impact of intercropping, tillage and fertilizer type on soil and crop yield in fruit orchards under Mediterranean conditions: a meta-analysis of field studies. Agric. Syst. 178, 102736. https://doi.org/10.1016/j.agsy.2019.102736.
Mousavi-Avval, S.H., Rafiee, S., Sharifi, M., Hosseinpour, S., Notarnicola, B., Tassielli, G., Renzulli, P.A., Khanali, M., 2017. Use of LCA indicators to assess Iranian rapeseed production systems with different residue management practices. Ecol. Indic. 80, 31–39. https://doi.org/10.1016/j.ecolind.2017.04.045.
Nath, A.J., Lal, R., 2017. Effects of tillage practices and land use management on soil aggregates and soil organic carbon in the North Appalachian Region, USA. Pedosphere 27 (1), 172–176. https://doi.org/10.1016/S1002-0160(17)60301-1.
Neilsen, G., Forge, T., Angers, D., Neilsen, D., Hogue, E., 2014. Suitable orchard floor management strategies in organic apple orchards that augment soil organic matter and maintain tree performance. Plant Soil 378 (1-2), 325–335. https://doi.org/10.1007/s11104-014-2034-8.
Novara, A., Minacapilli, M., Santoro, A., Rodrigo-Comino, J., Carrubba, A., Sarno, M., Venezia, G., Gristina, L., 2019. Real cover crops contribution to soil organic carbon sequestration in sloping vineyard. Sci. Total Environ. 652, 300–306. https://doi.org/10.1016/j.scitotenv.2018.10.247.
Okolo, C.C., Dippold, M.A., Gebresamuel, G., Zenebe, A., Haile, M., Bore, E., 2020. Assessing the sustainability of land use management of northern Ethiopian drylands by various indicators for soil health. Ecol. Indic. 112, 106092. https://doi.org/10.1016/j.ecolind.2020.106092.
Osanai, Y., Knox, O., Nachimuthu, G., Wilson, B., 2020. Contrasting agricultural management effects on soil organic carbon dynamics between topsoil and subsoil. Soil Res. 59, 24–33. https://doi.org/10.1071/SR19379.
Pardon, P., Reubens, B., Reheul, D., Mertens, J., De Frenne, P., Coussement, T., Janssens, P., Verheyen, K., 2017. Trees increase soil organic carbon and nutrient availability in temperate agroforestry systems. Agric. Ecosyst. Environ. 247, 98–111. https://doi.org/10.1016/j.agee.2017.06.018.
Pergola, M., Persiani, A., Pastore, V., Palese, A.M., Arous, A., Celano, G., 2017. A comprehensive Life Cycle Assessment (LCA) of three apricot orchard systems located in Metapontino area (Southern Italy). J. Clean. Prod. 142, 4059–4071. https://doi.org/10.1016/j.jclepro.2016.10.030.
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.
Peticila, A., Scaeteanu, G.V., Madjar, R., Stanica, F., Asanica, A., 2015. Fertilization effect on mineral nutrition of *Actinidia deliciosa* (kiwi) cultivated on different substrates. Agric. Agric. Sci. 6, 132–138. https://doi.org/10.1016/j.aaspro.2015.08.049.
Piccolo, A., 2002. The supramolecular structure of humic substances: a novel understanding of humus chemistry and implications in soil science. Adv. Agron. 75, 57–134. https://doi.org/10.1016/s0065-2113(02)75003-7.
Pinzari, F., Maggi, O., Lunghini, D., Di Lonardo, D.P., Persiani, A.M., 2017. A simple method for measuring fungal metabolic quotient and comparing carbon use efficiency of different isolates: application to Mediterranean leaf litter fungi. Plant Biosyst. 151 (2), 371–376. https://doi.org/10.1080/11263504.2017.1284166.
Poeplau, C., Don, A., 2013. Sensitivity of soil organic carbon stocks and fractions to different land-use changes across Europe. Geoderma 192, 189–201. https://doi.org/10.1016/j.geoderma.2012.08.003.
Poeplau, C., Don, A., 2015. Carbon sequestration in agricultural soils via cultivation of cover crops - A meta-analysis. Agric. Ecosyst. Environ. 200, 33–41. https://doi.org/10.1016/j.agee.2014.10.024.
Poeplau, C., Don, A., Six, J., Kaiser, M., Benbi, D., Chenu, C., Cotrufo, M.F., Derrien, D., Gioacchini, P., Grand, S., Gregorich, E., Griepentrog, M., Gunina, A., Haddix, M., Kuzyakov, Y., Kühnel, A., Macdonald, L.M., Soong, J., Trigalet, S., Vermeire, M.-L., Rovira, P., van Wesemael, B., Wiesmeier, M., Yeasmin, S., Yevdokimov, I., Nieder, R., 2018. Isolating organic carbon fractions with varying turnover rates in temperate agricultural soils – A comprehensive method comparison. Soil Biol. Biochem. 125, 10–26. https://doi.org/10.1016/j.soilbio.2018.06.025.
Pryor, S.W., Smithers, J., Lyne, P., van Antwerpen, R., 2017. Impact of agricultural practices on energy use and greenhouse gas emissions for South African sugarcane production. J. Clean. Prod. 141, 137–145. https://doi.org/10.1016/j.jclepro.2016.09.069.
Rasse, D.P., Rumpel, C., Dignac, M.-F., 2005. Is soil carbon mostly root carbon? Mechanisms for a specific stabilisation. Plant Soil 269 (1-2), 341–356. https://doi.org/10.1007/s11104-004-0907-y.
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. https://doi.org/10.1016/j.scitotenv.2017.03.147.
Repullo-Ruibérriz de Torres, M.A., Carbonell-Bojollo, R.M., Moreno-García, M., Ordóñez-Fernández, R., Rodríguez-Lizana, A., 2021. Soil organic matter and nutrient improvement through cover crops in a Mediterranean olive orchard. Soil Tillage Res. 210, 104977. https://doi.org/10.1016/j.still.2021.104977.
Romero-Gámez, M., Castro-Rodríguez, J., Suárez-Rey, E.M., 2017. Optimization of olive growing practices in Spain from a life cycle assessment perspective. J. Clean. Prod. 149, 25–37. https://doi.org/10.1016/j.jclepro.2017.02.071.
Ruiz-Sánchez, M.C., Plana, V., Ortuño, M.F., Tapia, L.M., Abrisqueta, J.M., 2005. Spatial root distribution of apricot trees in different soil tillage practices. Plant Soil 272 (1-2), 211–221. https://doi.org/10.1007/s11104-004-4781-4.
Samson, M.-E., Chantigny, M.H., Vanasse, A., Menasseri-Aubry, S., Royer, I., Angers, D. A., 2021. Response of subsurface C and N stocks dominates the whole-soil profile response to agricultural management practices in a cool, humid climate. Agric. Ecosyst. Environ. 320, 107590. https://doi.org/10.1016/j.agee.2021.107590.
Smith, P., Bustamante, M., Ahammad, H., Clark, H., Dong, H., Elsiddig, E.A., Haberl, H., Harper, R., House, J., Jafari, M., Masera, O., Mbow, C., Ravindranath, N.H., Rice, C. W., Robledo Abad, C., Romanovskaya, A., Sperling, F., Tubiello, F., 2014. Agriculture, forestry and other land use (AFOLU). Cambridge University Press, Cambridge.
Sokalska, D.I., Haman, D.Z., Szewczuk, A., Sobota, J., Dereń, D., 2009. Spatial root distribution of mature apple trees under drip irrigation system. Agric. Water Manag. 96 (6), 917–924. https://doi.org/10.1016/j.agwat.2008.12.003.
Solomon, D., Fritzsche, F., Lehmann, J., Tekalign, M., Zech, W., 2002. Soil organic matter dynamics in the subhumid agroecosystems of the ethiopian highlands. Soil Sci. Soc. Am. J. 66 (3), 969–978. https://doi.org/10.2136/sssaj2002.9690.
Stehfest, E., Bouwman, L., 2006. N_2O and NO emission from agricultural fields and soils under natural vegetation: summarizing available measurement data and modeling of global annual emissions. Nutr. Cycl. Agroecosyst. 74 (3), 207–228. https://doi.org/10.1007/s10705-006-9000-7.
Sun, T., Wang, Y., Hui, D., Jing, X., Feng, W., 2020. Soil properties rather than climate and ecosystem type control the vertical variations of soil organic carbon, microbial carbon, and microbial quotient. Soil Biol. Biochem. 148, 107905. https://doi.org/10.1016/j.soilbio.2020.107905.
Tangen, B.A., Bansal, S., 2020. Soil organic carbon stocks and sequestration rates of inland, freshwater wetlands: sources of variability and uncertainty. Sci. Total Environ. 749, 141444. https://doi.org/10.1016/j.scitotenv.2020.141444.
The World Meteorological Organization, 2020. Carbon dioxide levels continue at record levels, despite COVID-19 lockdown. World Meteorol. Organ.
Vance, E.D., Brookes, P.C., Jenkinson, D.S., 1987. An extraction method for measuring soil microbial biomass C. Soil Biol. Biochem. 19, 703–707. https://doi.org/10.1016/0038-0717(87)90052-6.
Vinyes, E., Gasol, C.M., Asin, L., Alegre, S., Muñoz, P., 2015. Life Cycle Assessment of multiyear peach production. J. Clean. Prod. 104, 68–79. https://doi.org/10.1016/j.jclepro.2015.05.041.
Vinyes, E., Asin, L., Alegre, S., Muñoz, P., Boschmonart, J., Gasol, C.M., 2017. Life Cycle Assessment of apple and peach production, distribution and consumption in Mediterranean fruit sector. J. Clean. Prod. 149, 313–320. https://doi.org/10.1016/j.jclepro.2017.02.102.
Vittori Antisari, L., Ferronato, C., De Feudis, M., Natali, C., Bianchini, G., Falsone, G., 2021a. Soil biochemical indicators and biological fertility in agricultural soils: A case study from northern Italy. Minerals 11, 1–15. https://doi.org/10.3390/min11020219.
Vittori Antisari, L., Trenti, W., De Feudis, M., Bianchini, G., Falsone, G., 2021b. Soil quality and organic matter pools in a temperate climate (Northern Italy) under different land uses. Agronomy 11, 1815. https://doi.org/10.3390/agronomy11091815.
Wang, B., Liang, C., Yao, H., Yang, E., An, S., 2021. The accumulation of microbial necromass carbon from litter to mineral soil and its contribution to soil organic carbon sequestration. Catena 207, 105622. https://doi.org/10.1016/j.catena.2021.105622.
Weidema, B.P., Bauer, C., Hischier, R., Mutel, C., Nemecek, T., Reinhard, J., Vadenbo, C., Wernet, G., 2013. Overview and Methodology: Data quality guideline for the ecoinvent database Version 3, Swiss Centre for Life Cycle Inventories.
Wiesmeier, M., Mayer, S., Burmeister, J., Hübner, R., Kögel-Knabner, I., 2020. Feasibility of the 4 per 1000 initiative in Bavaria: A reality check of agricultural soil management and carbon sequestration scenarios. Geoderma 369, 114333. https://doi.org/10.1016/j.geoderma.2020.114333.
Xiang, Y., Li, Y., Liu, Y., Zhang, S., Yue, X., Yao, B., Xue, J., Lv, W., Zhang, L., Xu, X., Li, Y., Li, S., 2022. Factors shaping soil organic carbon stocks in grass covered orchards across China: a meta-analysis. Sci. Total Environ. 807, 150632. https://doi.org/10.1016/j.scitotenv.2021.150632.
Zhao, D., Cheng, C., Lyu, D., Jiang, M., Du, G., 2017. Effects of residue coverage on the characteristics of soil carbon pools in orchards. Arch. Agron. Soil Sci. 63 (6), 771–783. https://doi.org/10.1080/03650340.2016.1241390.