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REVIEW

SEQUESTERING ORGANIC CARBON IN SOILS THROUGH LAND USE CHANGE AND AGRICULTURAL PRACTICES: A REVIEW

Lianhai WU (✉)

Net Zero and Resilient Farming, Rothamsted Research, North Wyke, Okehampton, Devon, EX20 2SB, UK.

KEYWORDS

agroecosystems, climate change, negative emissions technology, net zero

HIGHLIGHTS

- Either increasing C input to or reducing C release from soils can enhance soil C sequestration.
- Afforestation and reforestation have great potential in improving soil C sequestration.
- Long-term observations about the impacts of biochar on soil C sequestration are necessary.

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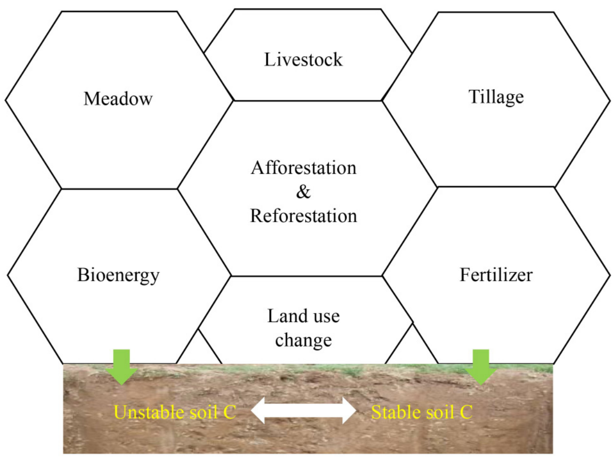
Correspondence: lianhai.wu@rothamsted.ac.uk

GRAPHICAL ABSTRACT



ABSTRACT

Climate change vigorously threats human livelihoods, places and biodiversity. To lock atmospheric CO_2 up through biological, chemical and physical processes is one of the pathways to mitigate climate change. Agricultural soils have a significant carbon sink capacity. Soil carbon sequestration (SCS) can be accelerated through appropriate changes in land use and agricultural practices. There have been various meta-analyses performed by combining data sets to interpret the influences of some methods on SCS rates or stocks. The objectives of this study were: (1) to update SCS capacity with different land-based techniques based on the latest publications, and (2) to discuss complexity to assess the impacts of the techniques on soil carbon accumulation. This review shows that afforestation and reforestation are slow processes but have great potential for improving SCS. Among agricultural practices, adding organic matter is an efficient way to sequester carbon in soils. Any practice that helps plant increase C fixation can increase soil carbon stock by increasing residues, dead root material and root exudates. Among the improved livestock grazing management practices, reseeding grasses seems to have the highest SCS rate.

1 INTRODUCTION

Extreme weather events and global warming are inevitable because of anthropogenic emissions of warming gases (greenhouse gases, GHG) in the future[1]. Over recent decades, the atmospheric carbon dioxide concentration and the global air temperature (Fig. 1) have been rising linearly, and the records of the extreme temperature and precipitation intensity kept being rewritten. Negative CO_2 emissions (i.e., CO_2 removals) to lock CO_2 up from the atmosphere through biological, chemical and physical processes are pathways to mitigate the global warming. Land management, soil management, bioenergy with carbon capture and storage, CO_2 capture from ambient air, enhanced weathering and ocean fertilization are purported to be main potential approaches for CO_2 removals[4–6]. Shepherd[7] categorized the CO_2 removal techniques into three groups according to places where they are applied to (land or ocean) and predominant interventions and assessed the techniques in terms of their effectiveness, timeliness, safety, affordability and reversibility. The first group is to sequester more C into their natural sinks for a long period. The second is to apply enhanced weathering techniques in land and oceans. The third is to apply advanced technologies to capture and store CO_2 directly elsewhere. The options also have been reviewed for their costs, potentials, side-effects, and innovation and scaling challenges for their implement[8–10]. Hepburn et al.[11] proposed 10 CO_2 removal pathways: (1) chemicals from CO_2, (2) fuels from CO_2, (3) products from microalgae, (4) concrete building materials, (5) CO_2-enhanced oil recovery, (6) bioenergy with carbon capture and storage, (7) enhanced weathering, (8) forestry techniques, (9) SCS techniques, and (10) biochar but stressed limited potentials for its removal. Agricultural soils have a significant C sink capacity[12,13]. It was estimated that the sink capacity is in the order of 20–30 Pg C (73–110 Pg CO_2 equivalent) over the next 50–100 years globally[12], which may offset 23%–35% of the net increase in atmospheric CO_2 between 2020 and 2100 with the target to limit warming to 1.5 °C in 2100 with a greater than 50% probability and a peak warming above 1.5 °C with less than 67% probability[14]. Soil organic C sequestration (SCS) can be accelerated through appropriate changes in land use and agricultural practices, such as converting crop land into land for non-crop fast growing plants[15], which is the focus of this review study. Cost assessment on net emission techniques suggested that appropriate land and soil management to boost SCS is cheap to deploy among the investigated techniques[8].

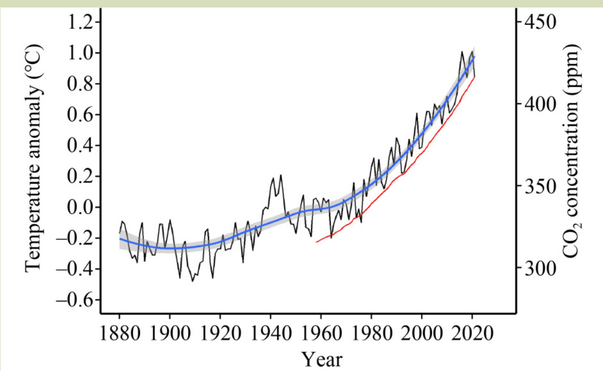


Fig. 1 Change in global surface temperature relative to 1951–1980 average temperatures (black line) and monthly atmospheric carbon dioxide concentrations observed at NOAA Mauna Loa Observatory in Hawaii (red line). The blue line is the smoothed conditional means with the 95% confidence band indicated by the gray area. Data sourced from Global Monitoring Laboratory website for CO_2 concentrations[2] and Global Climate Change website for temperature[3].

There have been various meta-analyses performed by combining data sets to interpret the influences of methods on SCS. For example, SCS rates by the tillage practice were globally reviewed in 2002[16] and 2022[17,18] and by afforestation in 2000[19], 2002[20], 2009[21], 2014[22], 2018[23], and 2021[24]. In this review, SCS rates with different land-based techniques are updated based on the latest publications and complexity to assess the impacts of the techniques on soil C accumulation is discussed. There are considerable co-benefits of the reviewed techniques for production and environment[13,25], which is beyond the scope for the review.

2 LAND-BASED METHODS TO ENHANCE SOIL CARBON SEQUESTRATION

Carbon cycling through plants, animals and soils in terrestrial ecosystems is shown in Fig. 2. For C recycling processes, plant

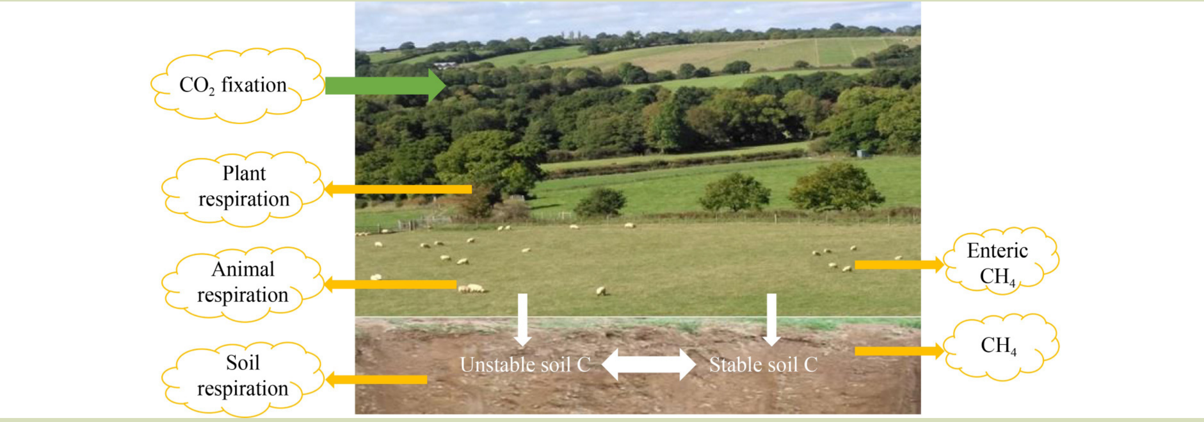


Fig. 2 Carbon cycling through plants, animals, and soils. Downward arrows indicate carbon input to soils through plant dead materials, animal excreta and external sources.

residues and dead roots can be incorporated into soil organic C (SOC) and a proportion of intake C by animals returns to soils as excreta. A fraction of plant fixed C is in standing biomass that can be used for other purposes or primary products. Similarly, a fraction of animal intake C is taken away as animal products (milk, meat and wool). For a land-based system with spatial boundaries, external input C (e.g., biochar and farmyard manure) and losses with water movement (e.g., bonded C in sediment and groundwater transport) should be considered. To sequester more C in soils, the target with different techniques is to increase the size of stable soil C stock via increasing C input to and reducing C release from soils.

2.1 Land use management

2.1.1 Afforestation and reforestation

Afforestation is the process that plants trees on land on which has no trees in recent history while reforestation is to replant trees where trees have been lost because of natural or anthropogenic disturbances. Afforestation has been described as one of the most natural and technologically simple methods to reduce atmospheric CO_2. It is realized by natural regeneration (planting native trees that will self-propagate from seed), agroforestry (incorporating trees into agricultural systems), or commercial plantations (for commercial products). A recent review suggested that afforestation significantly increased SOC by 44% at 0–100 cm soil depth, mainly occurred in the top 40 cm of soil[24]. A summary on the SCS rate by the methods from publications is shown in Table 1. New evidence is supportive of positive responses of SCS to afforestation.

Several reviews or meta-analyses on SCS with agroforestry have been published[40–46]. In the temperate region, the practice can increase SCS at 0–20 cm soil depth at a rate of 0.21 $Mg{\cdot}ha^{-1}{\cdot}yr^{-1}$ C[45]. Ramachandran Nair and Nair[47] showed a range of 0.4–2.5 $Mg{\cdot}ha^{-1}{\cdot}yr^{-1}$ C sequestered in soils under five major agroforestry systems in Africa without specified soil depth and conversion age. Unquestionably, the SCS potential by agroforestry is affected by climatic conditions, prior land use, agroforestry type and conversion duration. Data collected in China reviewed that the SCS sequestration rate can reach 1.0 $Mg{\cdot}ha^{-1}{\cdot}yr^{-1}$ C in the top 20 cm of soil in the areas with hot, humid summers and 0.83 $Mg{\cdot}ha^{-1}{\cdot}yr^{-1}$ C in the regions with warm, dry winters[44]. The same data set also indicated a role of agroforestry types in the SCS potential: 0.92, 0.70 and 0.23 $Mg{\cdot}ha^{-1}{\cdot}yr^{-1}$ for shelterbelts, agrisilviculture and silvopasture, respectively. A meta-analysis from 250 observations mostly located in the Americas, Africa and Asia revealed that the conversion of agriculture to agroforestry led to a 34% increase in SOC stock at 0–100 cm soil depth and 10% at 0–30 cm for the conversion from pasture or grassland to agroforestry[41]. Agroforestry systems stored more C in deeper soil layers near a tree than away from the tree compared with treeless systems[48]. However, there might not be always an increase in SOC with the system. A 9-year study with a humid or semiarid tropical climate suggested that SOC in the top 40 cm of soil decreased[49].

Climate, previous land use (representing soil C status before afforestation) and established forest type are the most important factors to drive soil C change in afforestation[20]. Globally, a cooler, drier climate stimulates SOC storage in the top 60 cm of soil whereas a warmer, wetter climate can reduce SOC storage[26]. A larger data set collected from northern

Table 1 Soil carbon sequestration rate by afforestation and reforestation (summaries from the literature and other sources as indicated)

Climate	Location	Conversion	Age after conversion (year)	Soil depth (cm)	Rate (Mg·ha^{-1}·yr^{-1} C)
Afforestation					
	Worldwide review	Agriculture	>8	–	0.34[19]
	Worldwide review	Grassland	>6	–	0.33[19]
	Worldwide review		>30	30	0.32[20]
	Worldwide review	Agriculture	<10 11–20 21–30 >30	60	–0.19[26] 0.53 1.10 0.57
	Worldwide review	Grassland	<10 11–20 21–30 >30	60	–0.34[26] –0.96 0.06 0.64
	China review	Agriculture	<10	20	–0.04[27]
	China review	Agriculture	11–20	20	1.35[27]
	China review	Semi-natural grassland	<10	20	nc[27]
	China review	Semi-natural grassland	11–20	20	0.86[27]
	Harpenden, UK	Agriculture to wild woodland	>110	69	0.38–0.54[28]
Humid continental	Michigan, USA	Agriculture to deciduous and conifer	53	100	0.26–0.35[29]
Temperate semiarid	Sierra de Carrascoy, Spain	Shrublands to conifers	20	5	0.17–0.28[30]
Mid-latitude steppe	Qinghai, China	Agriculture	9–31	60	1.13–1.38[31]
Temperate semiarid	Shaanxi, China	Agriculture to deciduous	39	100	0.63–1.86[32]
Temperate semiarid	Shaanxi, China	Agriculture to shrublands	38	100	0.63–1.78[32]
Humid continental	South-east Lithuania	Agriculture to woodland	20	28	0.05[33]
Humid continental	South-east Lithuania	Agriculture to conifers	20	28	–0.18[33]
Subpolar oceanic	Iceland	Natural grassland to deciduous	15–50	30	0.42[34]
Boreal Temperate	Sweden	Agriculture to mixed species	9 8	30	–3.02 to 2.3[35] –1.91 to 0.79
Subhumid Mediterranean	North-east Spain	Agriculture to orchards	60	30	0.42[36]
Humid continental	Czech	Agriculture to mixed species	14	20	–0.01 to 0.95[37]
Reforestation					
Temperate humid continental	Kentucky, USA	Mined soils to deciduous	15	50	1.7[38]
Arid hot climate	South-west China	Abandoned soils to mixed species	30	80	1.28[39]

Note: nc, not notable change.

China showed the SCS potential is adversely affected by greater soil C stocks before afforestation, especially in deeper soil[50]. Similarly, global analyses suggested that afforestation on grassland (usually with greater SOC concentration) appears a slower and smaller SOC increase than that on agricultural land[24,26]. Plantation type can drive the SCS potential. For example, a meta-analysis showed that pine plantations caused a 15% decrease in soil C[21]. A similar conclusion was drawn from the data in Europe which there was no significant impact of afforestation on SOC stocks in 0 to 20 or 30 cm of soil with conifer, deciduous or mixed species[20]. As shown in Table 1 and previous reviews[20,24,26], afforestation at a young age of plantations cannot be beneficial for SCS, which links to fine root distribution and exudates with age. Therefore, SOC stocks over a range of soil depths should be dynamically monitored in order to accurately assess the impact of afforestation on SCS. In addition, SCS rates are dependent on the soil depth at which they are determined. Published SCS rates are derived from different soil depths, which makes it difficult to compare studies, and might mislead practitioners and policymakers on afforestation and reforestation. To audit the reported SCS rates for given a practice, it is necessary to consider the impacts of climate, plantation type, establishment age and soil depth.

Although afforestation has merits for increasing SCS, it is of concern when used in inappropriately targeted areas (e.g., some countries in Africa[51] or grasslands and savannas[52]), negative impact on biodiversity when replacing semi-natural ecosystems[53–55], and reduction of the surface albedo[56,57] that could potentially raise the surface temperature and inadvertently reduce water availability, particularly in drier areas[58]. Others have raised concerns that afforestation can be difficult to manage[59] and take up large amounts of land, which can increase food prices[60], and impact food security and farmer incomes[61]. Also, there are some obstacles to the adoption of such practices by farmers with the reasons varying between countries. For example, financial support and knowledge gaps are major issues in the UK[62]. Reforestation is expensive and slow[63]. Therefore, it is necessary to identify priority areas where the early benefits from reforestation can be maximized, especially to meet commercial imperatives. Afforestation and reforestation can increase the chance of wildfires and insect outbreaks that release fixed C to the atmosphere[64] and adversely affect biodiversity. With these risks, Di Sacco et al.[65] proposed rules to maximize SCS and biodiversity, and improve livelihoods including engagement of all stakeholders, multiple goals and species selection.

2.1.2 Land use change

Changes in land use might favor soil C accumulation, especially conversion from cultivated to forest (i.e., afforestation) or permanent grassland. SCS rates under various land use changes are summarized in Table 2. It is rare to find literature on SCS with the land to be converted from bioenergy or agroforestry to other land use types. Prior land use type, climate, new land use type and conversion age are major factors to determine the SCS rate with a land use change. It is apparent that SOC stocks increase in a conversion from agricultural land to other uses, but not in the conversion of natural or semi-natural grassland to other uses. Long-term experiments at Rothamsted Research showed that soil C content in the 0–23 cm depth increased 64 $Mg \cdot ha^{-1}$ C over 120 years from 1881 to 1999 in the regenerating woodland[83] with only 23.4 $Mg \cdot ha^{-1}$ C over the same period in the adjacent arable plots with continuous winter wheat growth and 35 $Mg \cdot ha^{-1} \cdot yr^{-1}$ farmyard manure (equivalent to 3 $Mg \cdot ha^{-1} \cdot yr^{-1}$ C) application since 1883[84]. Published data from different climate regimes show that an average soil C accumulation rate at 0.39 $Mg \cdot ha^{-1} \cdot yr^{-1}$ C in converted forest and 0.33 $Mg \cdot ha^{-1} \cdot yr^{-1}$ C in converted grassland after agricultural use[19]. Plot experiments in the UK showed that the conversion of grassland to either silvopastoral or woodland systems, C stock in the top 20 cm of soil did not significantly differ between the land use types[85,86]. However, the consequence of the conversion from forest on SCS is inconclusive. The data collected from 51 sites and published before 2014 show a decline of SCS when a forest was changed from one type of coverage to another[66]. The conversion of forest to grassland can increase SCS[87] but decrease it in a conversion to an agroforestry system[88].

2.2 Practices on cropland

Agronomic practices that are able to increase organic matter input, partition more C to stable organic C pools, or reduce turnover rates in soil C pools can improve SCS[89]. In cropland, these can be direct-drilling or no-tillage, cover cropping, chemical and organic fertilizer applications, improved cropping and organic systems, erosion control, proper irrigation and water management, integrated pest management

Table 2 Impacts of land use change on soil carbon sequestration rate ($Mg \cdot ha^{-1} \cdot yr^{-1}$ C)

From/To	Arable	Grassland	Bioenergy	(Semi-)natural grassland	Agroforestry	Forest
From arable	x	−0.89 to 1.00[66–68]		−2.27[69]	4.55–6.75[70]	−1.74 to −0.60[66–68]
To grassland	0.16–0.92[66–68,71,72]	x		− 0.04 to 0.27[69,71]	0.21–16.9[70]	−0.10 to 0.68[66,67]
From bioenergy	1.02–1.09[73]	−0.67 to 0.33[73,74]	x	− 5.2 to −1.22[75,76]		− 4.04[73]
From (semi-)natural grassland	0.128[72]	0.031 [77]		x		− 0.17[78]
From agroforestry	−0.17 to 0.29[79,80]			−2.07[69]	x	0.9–5.6[81]
From forest	−0.28 to 1.40[66–68]	−1.28 to 0.43[68,79]		−0.09 to −0.061[82]		x

Note: x, no investigation on the same land use change; empty cells indicate no data available.

and precision agriculture[90,91]. In the literature, many publications on the impacts of tillage and fertilizer application can be found. In grasslands, fertilizer application, sowing legumes, improved grass species and irrigation can lead to an increase in soil C[71], which is reviewed below. Minasny et al.[92] compiled a list of management practices that are reported to enhance SCS. Meta-analysis on the impacts of recommended management practices focusing on organic inputs and tillage management on SCS in Mediterranean cropping systems showed that those practices with large quantities of C input have the highest influence compared with the standard practice[93]. Table 3 presents SOC sequestration rates under various agricultural practices as reported in the recent publications.

Table 3 Soil carbon sequestration rate ($Mg \cdot ha^{-1} \cdot yr^{-1}$ C) by Agricultural practices (summaries from the literature review and other sources as indicated)

Practice	Location	System	Period (year)	Soil depth (cm)	Rate
NT vs ST	Global review	–	>6	Not specified	0.17[94]
Reduced tillage vs ST	Global review	–	>6	Not specified	–0.06[94]
NT vs ST	UK review	–	2–23	30	0.31[95]
NT vs ST	Central USA review	–	7–100	20	0.27[96]
NT vs ST	central USA review	–	5–30	30	0.45[96]
NT vs ST, 240 $kg \cdot ha^{-1} \cdot yr^{-1}$ urea-N & P	Quzhou, China	Wheat-maize	34	20	0.5[97]
NT vs ST, 2.25 $Mg \cdot ha^{-1} \cdot yr^{-1}$ straw mulch, 240 $kg \cdot ha^{-1} \cdot yr^{-1}$ urea-N & P	Quzhou, China	Wheat-maize	34	20	0.92[97]
NT vs ST, 4.5 $Mg \cdot ha^{-1} \cdot yr^{-1}$ straw mulch, 240 $kg \cdot ha^{-1} \cdot yr^{-1}$ urea-N & P	Quzhou, China	Wheat-maize	34	20	0.68[97]
NT vs ST, flatbed planting, 260 $kg \cdot ha^{-1} \cdot yr^{-1}$ urea-N, P & K	Tripura (W), India	Maize-maize-pea	2	30	0.53[98]
NT vs ST, 80 (40 for rotation) $kg \cdot ha^{-1} \cdot yr^{-1}$ N	Madrid, Spain	Winter wheat and wheat-vetch	32	20	0.36[99]
NT vs ST, straw mulch (NT) or incorporated (ST), 0, 60 or 120 $kg \cdot ha^{-1} \cdot yr^{-1}$ N, P & K	Catalonia, NE Spain	Barley	13	40	0.18[100]
NT vs ST, fertilizer rates and forms varied	Lopburi, Thailand	Maize-mung bean	5	15	–0.06[101]
NT vs ST, ridge and furrow planting, 260 $kg \cdot ha^{-1} \cdot yr^{-1}$ urea-N, P & K	Tripura (W), India	Maize-maize-pea	2	30	0.2[98]
Straw mulch vs removal, NT, 340 $kg \cdot ha^{-1} \cdot yr^{-1}$ urea-N, P & K	Tai'an, China	Wheat-peanut	3	30	0.38[102]
NT (straw mulch) vs ST (SR), 340 kg $\cdot ha^{-1} \cdot yr^{-1}$ urea-N, P & K	Tai'an, China	Wheat-peanut	3	30	–0.85[102]
RT vs ST, SR, 340 $kg \cdot ha^{-1} \cdot yr^{-1}$ urea-N, P & K	Tai'an, China	Wheat-peanut	3	30	–0.6[102]
Mineral fertilizer (200 $kg \cdot ha^{-1} \cdot yr^{-1}$ urea-N, P & K) vs no fertilizer	Gongzhuling, China	Maize	6	20	0.4[103]
Mineral fertilizer plus SR (3.2 $Mg \cdot ha^{-1} \cdot yr^{-1}$ C) vs mineral fertilizer (200 $kg \cdot ha^{-1} \cdot yr^{-1}$ urea-N, P & K)	Gongzhuling, China	Maize	6	20	0.4[103]
Mineral fertilizer plus compost (3.2 $Mg \cdot ha^{-1} \cdot yr^{-1}$ C) vs mineral fertilizer (200 $kg \cdot ha^{-1} \cdot yr^{-1}$ urea-N, P & K)	Gongzhuling, China	Maize	6	20	0.85[103]
Mineral fertilizer plus biochar (3.2 $Mg \cdot ha^{-1} \cdot yr^{-1}$ C) vs mineral fertilizer (200 $kg \cdot ha^{-1} \cdot yr^{-1}$ urea-N, P & K)	Gongzhuling, China	Maize	6	20	2.07[103]
Organic fertilizer (2.8 $Mg \cdot ha^{-1} \cdot yr^{-1}$ C & 47.2 $kg \cdot ha^{-1} \cdot yr^{-1}$ N) vs no fertilizer	Gujarat, India	Groundnut	16	100	0.63[104]
Organic (1.98 $Mg \cdot ha^{-1} \cdot yr^{-1}$ C & 15.6 $kg \cdot ha^{-1} \cdot yr^{-1}$ N) plus inorganic fertilizers (15.6 $kg \cdot ha^{-1} \cdot yr^{-1}$ N) vs no fertilizer	Gujarat, India	Groundnut	16	100	0.43[104]
Inorganic fertilizer (20:40:40 $kg \cdot ha^{-1} \cdot yr^{-1}$ N: P_2O_5:K_2O) vs no fertilizer	Gujarat, India	Groundnut	16	100	0.1[104]
Cattle slurry (240 $kg \cdot ha^{-1} \cdot yr^{-1}$ N) vs no input, P, K & S applied	Kiel, Germany	Continuous silage maize	8	30	0.1[105]
Cattle slurry (160 $kg \cdot ha^{-1} \cdot yr^{-1}$ N) vs no input, P, K & S applied	Kiel, Germany	Oats-wheat-pulses rotation	8	30	0.3[105]
Cattle slurry (160 $kg \cdot ha^{-1} \cdot yr^{-1}$ N) vs no input, P, K & S applied	Kiel, Germany	Maize/oats-wheat-ley rotation	8	30	0.4[105]

(*Continued*)

Practice	Location	System	Period (year)	Soil depth (cm)	Rate
Organic fertilizer (15 $Mg \cdot ha^{-1} \cdot yr^{-1}$) vs no fertilizer, P, K, rotation with 100% cereal, SR from barley and rye	Berlin, Germany	Barley-barley-rye-oats	24	20	0.1[106]
Organic fertilizer (15 $Mg \cdot ha^{-1} \cdot yr^{-1}$) vs no fertilizer, P, K, rotation with 75% cereal, SR from barley and rye	Berlin, Germany	Beets-barley-rye-rye	24	20	nc[106]
Organic fertilizer (15 $Mg \cdot ha^{-1} \cdot yr^{-1}$) vs no fertilizer, P, K, rotation with 50% cereal, RR from barley and rye	Berlin, Germany	Beets-barley-rye-silage maize	24	20	0.03[106]
Straw incorporated (2.25 $Mg \cdot ha^{-1} \cdot yr^{-1}$) vs no straw, ST, 240 $kg \cdot ha^{-1} \cdot yr^{-1}$ urea-N & P	Quzhou, China	Wheat-maize	34	20	1.76[97]
Straw incorporated (4.5 $Mg \cdot ha^{-1} \cdot yr^{-1}$) vs no straw, ST, 240 $kg \cdot ha^{-1} \cdot yr^{-1}$ urea-N & P	Quzhou, China	Wheat-maize	34	20	2.44[97]
Straw mulch vs no organic matter input, fertilizer rates and forms varied	Lopburi, Thailand	Maize-mung bean	5	15	0.39[101]
SR vs straw removal, irrigation and inorganic fertilisers (352.5:82.2:146.3 $kg \cdot ha^{-1} \cdot yr^{-1}$ N:P:K)	Shaanxi, China	Wheat-maize	25	20	0.11[107]
Irrigation vs rainfed, no fertilizer	Shaanxi, China	Wheat-maize	25	20	0.03[107]
Rotation vs continuous	Global review		25 (Average)	22 (Average)	0.15[16]
With vs without catch crop	Argentina	Soybean	8	20	0.09–0.39[108]

Note: NT, no-tillage; ST, standard tillage; RT, rotary tillage; SR, straw return. nc, no substantive change.

It has been suggested that the SCS potential of no-tillage is high compared to other tillage practices[18]. Several meta-analyses indicated that the no-tillage practice can increase SOC stock about 8%[94] or over 0.4 $Mg \cdot ha^{-1} \cdot yr^{-1}$ C[16,109] compared with that under the standard tillage practice. However, the role of the no-tillage practice on SCS is not conclusive. Data collected from north America showed positive, negative and no significant differences in SOC between no-tillage and standard tillage[110]. A meta-analysis on SCS related deep tillage from 43 publications until 2019 suggested that deep plowing increased SOC by 1.1% while subsoiling by 8.9% compared to standard tillage[111]. It was noted that uncertainty in comparison of SOC stocks in topsoil between no-tillage and standard tillage might exist if SOC stock is derived from SOC content (g C g^{-1} soil) and soil bulk density that is treated as a constant over the reporting period, i.e., assumed not to be influenced by tillage practices.

Fertilizer application cannot strengthen SCS directly, instead increase inputs from residue and belowground biomass. In the long term experiment at Rothamsted, SOC content in the top 23 cm of soil on the fertilized plot with 144 $kg \cdot ha^{-1} \cdot yr^{-1}$ N together with P (35 $kg \cdot ha^{-1} \cdot yr^{-1}$) and K (90 $kg \cdot ha^{-1} \cdot yr^{-1}$) application for over 170 years is 1.2%, greater than in the unfertilized plot (0.87%) and initial concentration (1.0%)[84]. The sequestration potential with fertilizer applications can decline with time even with straw incorporated into soils[112]. Quantifying the impact of fertilizer application on SCS depends on reporting soil depth. For example, in the long term rotation experiment of the Morrow Plots in the USA showed the mineral fertilizer applications for 51 growing seasons from 1955 to 2005 increase SOC in the top 30 cm of soil but have no change in the top 46 cm of soil compared with the unfertilized treatment[113].

Recent evidence has demonstrated that external SOC input can improve SCS in various cropping systems in different climate zones (Table 3). However, the extent of increase in SOC depends on its initial concentration. For example, in the Thyrow long-term experiment, farmyard manure (FYM) was applied at 15 $Mg \cdot ha^{-1} \cdot yr^{-1}$ from 1937 until 1971, which led to a greater concentration of SOC in the topsoil. With new treatments, SOC did not show substantive difference between the plots where FYM application was discontinued and those FYM was applied at the same rate for another 24 years[106]. In a separate experiment with a short history of organic matter input, SOC with organic fertilizer application significantly increased compared without organic fertilizer input[104]. Further, the method of organic fertilizer or straw application affects the SCS rate. In general, organic fertilizer or straw is incorporated into soil is better than surface application[97,101].

Again, published SCS rates have been derived from different soil depths, which makes it difficult to compare them between studies. Apart from organic material amendment, the majority of the practices improve SCS through increased retention of surface and belowground plant residues. Roots of different plant species differ in their penetration to depth. If SCS rates are considered in the topsoil only, they might be underestimated with the contribution of the dead roots in the subsoils ignored. Another difficulty for comparing data from different studies, even with the same nominal tillage practice, is that the practice might have different implications. For example, a reported minimal tillage practice might not define how often the field was plowed, which is likely to determine the final SCS rates. SCS through agricultural practices is affected by environmental factors. A review on SCS under conservation tillage on light-textured soils in Australia revealed that significantly greater SOC concentrations compared with multi-pass tillage were only found in the wetter areas (> 500 mm annual rainfall) and restricted to the top 2.5–10.0 cm[114].

Novel and innovative solutions can help sequester C. A laboratory-scale pilot study showed that products using cellulose-based waste materials and industrially sourced CO_2 can be significantly lower than usual C and resource footprints[115]. The authors claimed that the fertilizers can permanently sequestrate organic C that has been incorporated into them in soil in equal quantities to the fertilizer applied[116]. Mixed farming (an integrated crop-livestock system) could be an option to increase soil C stocks. Brewer and Gaudin[117] summarized how the system can be managed. However, studies comparing SOC sequestration in mixed farming systems with cropping or grazing systems alone are quite limited.

Although appropriate agricultural practices have potential for SCS, the capacity to remove atmospheric CO_2 should not be exaggerated. First, the capacity of soils to store additional C is finite so that mitigation benefits will be reached over a limited time period. The long-term spring barley field experiment at Rothamsted Research, UK, showed that the SCS rate for the treatment with annual FYM application at 35 $Mg \cdot ha^{-1}$ (equivalent to approx. 3.2 $Mg \cdot ha^{-1} \cdot yr^{-1}$ C) was 0.69 $Mg \cdot ha^{-1} \cdot yr^{-1}$ C in the 23 cm depth during the first 20 years but only 0.06 $Mg \cdot ha^{-1} \cdot yr^{-1}$ C during 140–160 years[61]. A modeling study also suggested the rate is very small in soils with greater SOC[118]. Secondly, an SCS potential of an agricultural practice will not be universal applicable. For example, the SCS potential of no-tillage is site specific and limited within surface soil[119].

2.3 Biochar

Biochar application to soils has been considered as a pathway to sequester C[120,121]. As majority of C in biochar is recalcitrant with the mean residence time estimated to be about 556 years[122], it is highly stable in soils. However, biochar decomposition both in field and laboratory experiments through different mechanisms have been reported[123,124]. Apart from added C with biochar for SCS, it has been reported that biochar amendment can increase SOC stock. Since 2012, at least six reviews or meta-analyses on SCS by biochar application have been published[125–130]. One meta-analysis showed a 4.9–43 $g \cdot kg^{-1}$ increase in SOC over a period between 5 months and 2 years[127]. Another, based on 56 publications, suggested that biochar can enhance SOC stock by 28% in the top 10 cm of soil under field conditions[94]. A recent global meta-analysis of 412 individual field treatments over experimental durations between 1 and 10 years at biochar application rates between 1 and 100 $Mg \cdot ha^{-1}$ showed a mean increase in SOC by 13 $Mg \cdot ha^{-1}$[131]. Zhang et al.[132] reported an SOC increase of 0.34–0.90 $g \cdot kg^{-1} \cdot yr^{-1}$ C in the top 15 cm of soil over 5 years when 20, 30 or 40 $Mg \cdot ha^{-1} \cdot yr^{-1}$ biochar was applied in an oilseed rape-sweet potato production system.

Although biochar applied to soils can enhance SOC stocks, it might lead to low nitrogen availability to plants via increasing N immobilization due to the high C/N ratio of biochar[133]. However, some reports claimed that biochar can increase plant N availability. For example, observations from four field experiments in boreal agricultural soils where biochar was applied 8 years before showed an increase in plant N availability through increased soil N mineralization in the short term[134]. In addition, feedstock type is an important factor for N availability. In their review on biochar effects on soil available inorganic N, Nguyen et al.[135] concluded that woody biochar did not decrease soil inorganic N as much as other types of biochar. Long-term impacts of biochar on SCS might be different from those derived from short-run experiments as indirect effects over time have not been determined. Observations have only been made over the last two decades, so compared to the potential residence time of biochar in soil, this period is too short to be meaningful. Consequently, it will be necessary to conduct long-term experiments to monitor the impacts of biochar on SCS and other indicators in agricultural systems.

2.4 Practices on grassland

Livestock systems can be divided into pastoral, mixed and feedlot systems. The first group is based on grazing ruminants, the mixed system integrates crop and livestock production in which livestock are fed with crop products, grasses and/or fodder. The third group is defined as a solely livestock system where less than 10% of dry matter fed to the animals is farm produced[136]. Pastoral and mixed systems are the focus in this review.

The contribution of grass-fed ruminants to SCS is reported to be small[137]. A review of the effects of grazing on soil C showed that different studies have found both strong positive and negative grazing effects on SOC[138]. However, proper grazing management practices, including appropriate stocking rates, introducing beneficial forage species, and allowing sufficient rest time for plant recovery between grazing, and adopting silvopasture in livestock production systems[139–143], can help increase C sequestration in grazing lands/pastures between 0.14 and 0.41 $Mg \cdot m^{-2} \cdot yr^{-1}$ C[71]. Madigan et al.[17] listed management practices in temperate grassland which affect SCS, and their co-benefits or disadvantages. A global meta-analysis of grazing impacts on soil health indicators showed that both rotational grazing and no grazing had greater SOC than continuous grazing[143]. A five-year on-farm study in Michigan, USA, showed that rotational grazing can sequester C at 3.59 $Mg \cdot m^{-2} \cdot yr^{-1}$ C for three soil types (sandy, sandy loam and clay loam)[144]. A 15-year field experiment on grassland showed that the long-term application of organic fertilizer can significantly increase soil C in the top 15 cm soil in both Arenosol and Andosols but inorganic fertilizer amendment did not have this effect[145]. SCS rates under different practices summarized from the literature and other recent sources are shown in Table 4.

Among the improved management practices, reseeding grasses has the highest SCS rate[139]. However, the sequestration rate depends on the period for which a practice is undertaken. For example, in their meta-analysis on the impacts of livestock exclusion on C sequestration in a Chinese grassland, Deng et al.[148] reported that the rate was not significantly different from zero over a period of three years but was greater than zero in all the examined studies. It was noted that intensive production practices with high inputs and rates of removal can deplete SOC stocks[154].

Table 4 Soil carbon sequestration rates (as $Mg \cdot ha^{-1} \cdot yr^{-1}$ C or percentage change) for various livestock practices (summaries from the literature and other sources as indicated)

Climate	Type	Practices	Length (year)	Depth (cm)	Rate
Worldwide review	Grassland	Mineral fertilizer	–	–	0.54[71]
Worldwide review	Grassland	Organic fertilizer	–	–	0.84[71]
Worldwide review	Grassland	Introducing legumes	–	–	0.66[71]
Worldwide review	Grassland	Improved grass species	–	–	3.04[142]
Semi-arid tropical savannah	Rangeland	Managed (grazing in dry season) vs unmanaged	–	30	12.1%–22.2%[146]
Semi-arid tropical savannah	Rangeland	Managed (grazing in wet season) vs unmanaged	–	30	nc[146]
Arid and semiarid	Rangeland	Grazing exclusion vs grazing	6	20	26.9%[147]
Arid and semiarid	Rangeland	Grazing exclusion vs grazing	>1	30	0.23[148]
Cold desert	Rangeland	Grazing exclusion vs grazing	4	20	49%[149]
Cold steppe	Grassland	Grazing exclusion vs light grazing	12	30	−15.6%[150]
Cold steppe	Grassland	Grazing exclusion vs heavy grazing	12	30	14.1%[150]
Cold semi-arid	Grassland	Grazing exclusion vs light grazing	55	30	49.4%[150]
Cold semi-arid	Grassland	Grazing exclusion vs heavy grazing	55	30	46.9%[150]
Temperate	Grassland	Fertilized P vs non application	>20	60	25.5%[151]
Temperate	Grassland	Multiple sward (5 species)	9	30	1.6[152]
Temperate	Grassland	Multiple sward (2 species)	9	30	0.44[152]
Semiarid	Rangeland	Grazing exclusion	>75	60	0.128[153]
Semiarid	Rangeland	Light grazing (0.78 sheep Eq ha^{-1})	>75	60	0.097[153]
Semiarid	Rangeland	Heavy grazing (1.18 sheep Eq ha^{-1})	>75	60	0.093[153]

Note: nc, no substantive change.

3 SOIL CARBON SEQUESTRATION AND OTHER ECOSYSTEM SERVICES

Most of the land-based techniques for SCS need land that support multiple ecosystem services. Combined with food demands with a global population of 9 billion by the 2050s, different demands result in competition for land. Smith et al.[155] reviewed how competition for land is influenced by drivers and pressures and concluded that policies should address agriculture, food production and the primary drivers of competition for land concurrently. It was proposed to breed new plants that absorb and sequester C more efficiently in soils to lessen the land constraint[156]. Meanwhile, regenerative agriculture and carbon farming that use a suite of practices to achieve ecosystem services should be encouraged. Sequestering more C in soils through agricultural practices is not only for mitigating climate change but also strengthening resilience of agricultural systems to cope with weather variability. Long-term field experiments on fertilization in China showed a significant improvement in yield stability in wheat-maize cropping systems with a combination of manure and mineral fertilizer application in southern China[157]. A farm survey across China indicated high-quality soils led to both greater crop yield and yield stability[158]. Data from a 27-year experiment in Germany drew the same conclusion[159]. An integrated methodology for evaluating management practices to increase SCS and other services in agroecosystems should be examined[89]. Agricultural systems are complex because they are generally managed at a field scale and all have interactions with nutrient cycling including C dynamics, environmental conditions and plants, with uncertainties in the responses of the system to disturbances. In this respect, models provide an effective method for investigating agricultural systems allowing examination of the systems under different scenarios and predict SCS. Fuss et al.[6] also advocated to use biophysical-techno-engineering-process-based, economic and Earth system models to investigate sustainable potential of the sequestration methods.

4 CONCLUSIONS

Enhancing SCS can be achieved by either increasing C input to or reducing C release from soils with different land-based techniques. Afforestation and reforestation are slow processes but have great potential in improving SCS. However, these practices can only be applied in appropriate areas. When intensive land use (agricultural land and managed grassland) is changed to a type of extensive land uses (e.g., wildland and semi-natural/natural grassland), most published data indicated a consequent positive response in SCS. Among agricultural practices, adding organic matter is an efficient way to sequester C in soils. Any practice that helps plant increase C fixation can increase soil C stock by increasing residues, dead root material and root exudates. Novel and innovative solutions for SCS need to be explored. Regenerative agriculture and carbon farming should be encouraged. Biochar can slow the decomposition process and increase SOC. However, it would be necessary to set up long-term experiments to monitor the impacts of biochar on SCS and other indicators in agricultural systems. Among the improved livestock grazing management practices, reseeding grasses seems to have the highest SCS rate. Finally, modeling is an effective option for evaluating the impacts of agronomic and pastoral practices on SCS and other services in agroecosystems.

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