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1 Introduction

To meet the climate goal of the Paris Agreement, signed in 2015 by 196 parties at the COP 21 meeting, the mean global air temperature increase must be kept well below 2°C and efforts to limit the increase to 1.5°C above pre-industrial levels should be encouraged. In addition to drastic reductions in the use of fossil carbon and emissions of non-CO_2 greenhouse gases (GHG), negative carbon emissions technologies are needed to achieve this goal. Promising technologies include bioenergy with carbon capture and storage, direct air carbon capture and storage, afforestation and reforestation, enhanced weathering, ocean fertilization, biochar, and soil carbon sequestration (Minx et al., 2018). It is unlikely that one single technology will achieve the carbon uptake required to meet the climate target in a sustainable way and not all negative emissions technologies are available at a reasonable cost (Fuss et al., 2018; Bednar et al., 2019). The role of soils in the global carbon cycle and

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the importance of reducing GHG emissions from agriculture are increasingly being acknowledged (IPCC, 2018). Increased carbon storage in plants and soils through afforestation, reforestation, and soil carbon sequestration (SCS) are generally considered to be more cost-effective than the other methods mentioned above, with high potential of SCS in particular (Fuss et al., 2018; Minx et al., 2018). SCS has other advantages because agricultural techniques enhancing soil organic carbon (SOC) are known and well tested and do not require land-use changes.

The technical potential for SCS in all soils is high, 1.2–3.1 Pg C $year^{-1}$ according to Lal (2013). Conversion of biomass into biochar, which may be used for soil amelioration, increases this potential further, by about 0.5 Pg C $year^{-1}$, and according to Woolf et al. (2010), it can be achieved sustainably. Global estimates of SCS in cropland soils due to improved agricultural management practices, excluding biochar, range between 0.069 and 1.85 Pg C $year^{-1}$ according to a recent review (Roe et al., 2019). Estimates at the higher end of this range refer solely to technical potential, while values up to about 0.5 Pg C $year^{-1}$ are regarded as economically feasible. On considering some limitations to implementation, Paustian et al. (2016) estimated that 0.08–0.4 Pg C $year^{-1}$ can be sequestered in upland agricultural soils, but that further technological developments in coming decades, such as plant breeding for enhanced root phenotypes of crops, could add an additional 0.3 Pg C $year^{-1}$. According to the Food and Agriculture Organization of the United Nations (FAO), the total area of harvested cropland in 2017 was 1.424 Gha, of which 0.165 Gha was paddy rice systems. Dividing the estimated annual SCS potential at the higher end of the range reported in several publications (0.3 Pg C $year^{-1}$) by the area of upland cropland soils (1.259 Gha) results in an average annual SCS potential of 0.24 Mg C ha^{-1} $year^{-1}$, which corresponds to 0.4% of average SOC stocks to 20 cm depth in European cropland (Lugato et al., 2021). This high-end estimate meets the target set by the '4 per mille' initiative to increase SOC by 0.4% per year (Minasny et al., 2017). On considering only low-cost and cost-effective measures to increase SOC in upland soil, that is, cover crops and more trees in agricultural landscapes, the estimated SCS potential decreases to less than half, 0.14 Pg C, corresponding to 0.11 Mg C ha^{-1} $year^{-1}$ (Bossio et al., 2020).

In this chapter, we synthesize the current state of knowledge on agricultural measures that could be implemented on cropland to increase SCS in upland mineral soils. Following a brief overview of the principles of SCS, we synthesize the results from reviews and meta-analyses quantifying the effect of different agricultural management practices at field scale and illustrate these with examples familiar to us. We also discuss uncertainties associated with the effect size of these measures and problems with upscaling data obtained in field studies to regional or global scale.

2 Basic principles of soil organic carbon storage and sequestration

The SOC balance is controlled by inputs of organic carbon and outputs resulting from decomposition (Fig. 1). Inputs derive from rhizodeposition during the growing season, post-harvest residues, and ex situ-derived inputs of other organic materials, such as manures and recycled organic wastes originating from agricultural production and/or food processing and consumption that may have been transformed through composting, fermentation, or pyrolysis. Carbon outputs from the soil are mainly controlled by pedoclimatic conditions and are therefore more difficult to manage than inputs, which are determined by farming practices (Kätterer et al., 2012). Losses in form of dissolved organic carbon (DOC) are often neglected but may be important to consider (Kindler et al., 2011). Soil erosion leads to loss of carbon at the plot scale, but the effect on the carbon balance at larger scales depends on where the eroded material is deposited (Lal, 2004). In some cases, erosion may even create a carbon sink at a larger scale (Doetterl et al., 2016). Net primary production (NPP), that is, the fraction of carbon fixed by photosynthesis that is converted to biomass, is the first step in capturing carbon from the atmosphere. The proportion of this fixed carbon that enters the soil depends on the crop grown, how it is managed, and how much carbon is retained in field, exported in products, or recycled through manure or through waste from society.

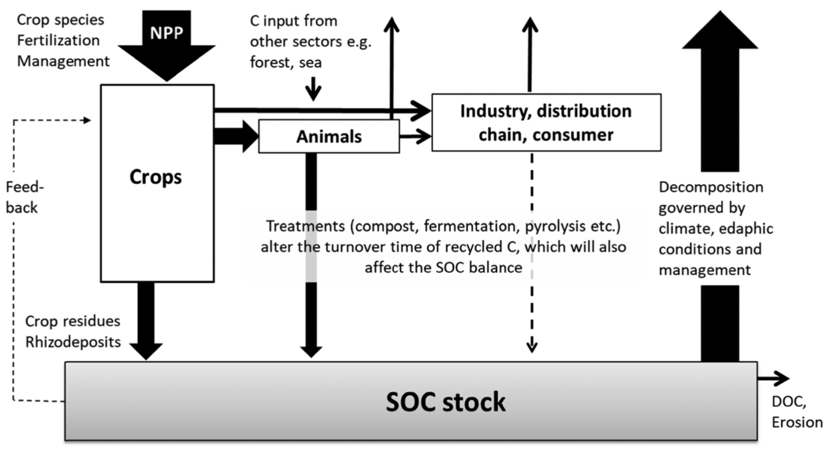

Figure 1 The soil organic carbon (SOC) balance of agricultural fields is driven by net primary production (NPP) and management decisions taken by farmers, that is, choice of crop, residue handling, input of organic amendments, etc. Losses of carbon, through decomposition, erosion, and dissolved organic carbon (DOC), are mainly under pedoclimatic control and less controllable by management (diagram modified after Bolinder et al., 2020).

The organic material that enters the soil through rhizodeposition, harvest residues, and recycled waste decomposes rapidly, but a fraction of it is stabilized through physical, chemical, and biological processes, depending on its characteristics and pedoclimatic conditions (Basile-Doelsch et al., 2020; see Chapter 3 of this book). These include microbial anabolism related to substrate quality and nutrient availability (Liang et al., 2017; Poeplau et al., 2019). The turnover time differs between different sources of organic inputs. Root-derived carbon is particularly important since it has a longer turnover time than aboveground plant material (Hénin and Dupuis, 1945; Bolinder et al., 1999; Wilhelm et al., 2004; Rasse et al., 2005; Kätterer et al., 2011). Plant breeding for cultivars with deep and extended root systems could be a way to increase SOC stocks (Eckersten et al., 2017). Organic material that has been digested in the gut of animals or through composting, fermentation, or pyrolysis prior to soil application generally has longer turnover times than the fresh plant material (e.g. Kätterer et al., 2011; Dechow et al., 2019; Levavasseur et al., 2020).

In this chapter, we review measures that lead to increased carbon storage compared with a reference state. This does not necessarily mean that carbon is sequestered (Chenu et al., 2019). According to Olson et al. (2014), SCS is defined as 'the process of transferring CO_2 from the atmosphere into the soil of a land unit, through plants, plant residues and other organic solids which are stored or retained in the unit as part of the soil organic matter (humus). The retention time of sequestered carbon in the soil (terrestrial pool) can range from short-term (not immediately released back to the atmosphere) to long-term (millennia) storage'.

Whether increased carbon storage will lead to SCS, and thereby to negative emissions, or only to mitigation of carbon losses compared with the status quo depends on historical land use and management at a specific site (Fig. 2). The carbon inputs required to keep the soil at a steady-state are proportional to the initial SOC stock. Increased carbon storage after a change in management contributes to climate change mitigation either by real SCS or by decreasing carbon losses compared with the status quo. The advantage of focusing on changes in carbon storage rather than changes in carbon sequestration is that the former is independent of site history, at least in theory, which makes it much easier to compare the impact of agricultural measures between sites (Fig. 2).

Annual changes in SOC are small compared with the amount stored in the soil. Due to spatial variation within fields, it takes many years until changes in SOC are measurable. Long-term field experiments (LTEs) are therefore indispensable for quantifying the effect of agricultural management practices on soil carbon stocks. Primary results from LTEs have been compiled in several reviews and meta-analyses, in which changes in SOC are presented as either stock change rate (SCR, Mg C ha^{-1} $year^{-1}$) or response ratio (RR; %), that is, relative changes in SOC concentrations or stocks caused by a specific experimental treatment

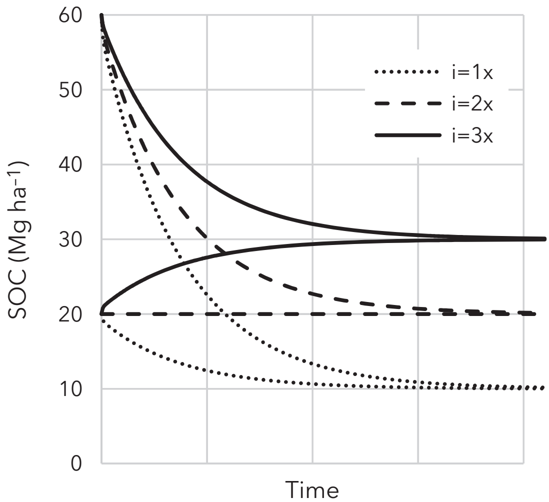

Figure 2 Soil organic carbon (SOC) dynamics in two hypothetical soils with different land use and management history, both receiving carbon inputs, *i*, at three different rates, 1×, 2×, and 3× (highest rate). In this example, we used a two-pool SOC model (ICBM; Andrén and Kätterer, 1997), but any other commonly used carbon model would produce similar curves. The soil with high initial SOC loses much more carbon over time than the soil starting with low initial SOC, but absolute differences in SOC between input rates are identical at any point in time, irrespective of initial conditions. Response ratio, i.e. the ratio of SOC stocks at different input rates, is dependent on both time and initial conditions. Only the low SOC soil at the highest input rate leads to net soil carbon sequestration, but carbon storage increases in both soils when input rates increase and both soils approach the same steady-state SOC stock when carbon inputs equal outputs.

compared with a control. Response ratio is a relative measure and is therefore scale-dependent, that is, a 10% change corresponds to twice as much in absolute terms in a soil with 2% SOC than in a soil with 1% SOC. However, since both SOC concentration and bulk density measurements are required to calculate SCR, RR based on concentration measurements is more frequently reported, although recent studies suggest that, when feasible, it is important to discuss SCR and RR in order to avoid misleading conclusions (e.g. Bolinder et al., 2020; Gross and Glaser, 2021). Therefore, we considered both these effect size indices in the present synthesis. Changes in SOC can also be expressed using absolute differences in SOC content (mg g^{-1}; see, e.g. Land et al., 2021). For scaling-up purposes, SCR is the most common and useful measure.

3 Crop residue retention, cover crops, recycling of organic materials and nitrogen fertilization

In this section we report the effect of four management practices (crop residue retention, cover crops, recycling of organic materials, and nitrogen fertilization) on SOC, based on the main outcomes of synthesis by Bolinder et al. (2020) of 20 reviews and meta-analyses, some covering several management practices. A majority of the publications were derived from the Systematic Map (Haddaway et al., 2015) and applied according to a systematic review protocol that includes a detailed and transparent description of the objectives and methods (Söderström et al., 2014). All publications used paired comparisons, involving a total of 2388 data entries on RR and 855 observations on SCR obtained from LTEs, some of which permitted assessment of their interactions with pedo-climatic conditions and cropping system characteristics. The mean duration of LTEs on which the publications were based generally ranged from 10 to 25 years, and they generally addressed changes in SOC in the topsoil layer (0–30 cm), with mean sampling depth typically between 20 cm and 30 cm. The reviews and meta-analyses rarely included subsoil samples (>30 cm), so the proportion in the databases was very small (for details, see Bolinder et al. 2020). In our discussion below on the four management practices, we also consider the main conclusions from other studies, in particular some recent meta-analyses not included in Bolinder et al. (2020).

3.1 Aboveground crop residue retention

Nine of the reviews covered by Bolinder et al. (2020) addressed issues relating to aboveground crop residue retention (Liu et al., 2014; Lemke et al., 2010; Powlson et al., 2011; Lehtinen et al., 2014; Smith et al., 1997; VandenBygaart et al., 2003; Wang et al., 2015; Lu, 2015; Luo et al., 2010a). Compared with the reference treatment (crop residue removal), calculated across reviews, retaining aboveground crop residues in the field resulted in a mean RR value of 10.3% (range of mean values in all reviews = 2.7–18.2%, number of paired observations (N = 995), and an SCR value of 0.12 Mg C ha^{-1} year^{-1} (0.053–0.59 Mg C ha^{-1} year^{-1}, N=279)). There was a large variation in mean values between the reviews. A few extreme treatments, such as the inclusion of bare fallow as a reference treatment or application of large amounts of straw in the residue retention treatments, can partly explain this variation. The reviews had different degrees of coverage (i.e. country-specific, continental, and global), further contributing to the variation. Most importantly, there were also variations in average soil depth considered and in the duration of the LTEs on which the different meta-databases were based.

The positive effects on SOC of leaving crop residues in the field are mainly a consequence of increasing inputs of carbon to the soil, although reduced losses of carbon from the topsoil via erosion is another major mechanism. According to two of the reviews, crop residue retention can also affect crop yields, thereby increasing NPP and leading to higher residue carbon inputs of the subsequent crop to the soil, which may in turn increase SOC and positively influence SCR (Lu, 2015; Wang et al., 2015). All reviews mainly considered small-grain cereal-based agroecosystems, although the relative importance of retaining aboveground residues increases with the amount of aboveground biomass produced. For example, grain maize produces around twice as many aboveground crop residues as small-grain cereals (Wilhelm et al., 2004). As shown for maize cropping systems in North American studies, SCR can then be as high as 0.3–0.8 Mg C ha^{-1} $year^{-1}$ (Anderson-Texeira et al., 2009). Indeed, one of the reviews found that the effect on RR increased (from 9% to 14%) with the proportion of maize in the rotation and that RR was higher for chopped (13%) compared with unchopped (9%) aboveground crop residues (Lu, 2015).

Reviews with global (Liu et al., 2014) or European coverage (Lehtinen et al., 2014) were unable to establish any interactions between climate and changes in RR or SCR with crop residue retention. However, some potential interactions were identified in country-scale reviews for China (Lu, 2015) and Australia (Luo et al., 2010a), with, for example, the latter showing greater increases in RR in areas with low precipitation. Among the reviews assessing interactions between soil texture and RR or SCR, two reported insignificant interactions for China (Lu, 2015; Wang et al., 2015). Interactions were significant in three of the reviews (Luo et al., 2010a; Liu et al., 2014; Lehtinen et al., 2014), but with contrasting results (i.e., either lower or higher RR with lower or higher clay content).

3.2 Cover crops

Five reviews on the effect of cover crops, using a reference treatment without cover crops, reported mean RR of 9% (7.8–13.1%, N = 129) and SCR of 0.33 Mg C ha^{-1} $year^{-1}$ (0.27–0.43 Mg C ha^{-1} $year^{-1}$, N = 176) (Poeplau and Don, 2015; McDaniel et al., 2014; Aguilera et al., 2013; Blanco-Canqui, 2013; Poeplau et al., 2015). Compared with the effect of aboveground crop residue retention, the variation in mean RR and SCR calculated across reviews was less variable for cover crops. Reviews on cover crops at a global scale showed similar values of RR (McDaniel et al., 2014) and SCR (Poeplau and Don, 2015) to the mean values calculated across all reviews.

However, studies and reviews on the effect of cover crops are complex, involving different species of main crops and cover crops, mixture of specifies, or reference treatments. Poeplau and Don (2015), for example, were comparing a winter cover crop with a reference cropland with winter fallow, and Poeplau et al. (2015) examined only the effect of perennial ryegrass (mostly undersown) as a cover crop. In the review by McDaniel et al. (2014), almost all paired comparisons (97%) involved leguminous cover crops. Aguilera et al. (2013) specifically examined scenarios where cover crops were substituting bare soils. Blanco-Canqui (2013) also included a specific discussion on the effect of multispecies (i.e., mixtures) of cover crops. The mechanism behind the positive effect of cover crops in agroecosystems is primarily increased carbon inputs into the soil since cover crops provide an additional source of aboveground and belowground crop residue carbon entering the soil. However, cover crops can also play an important role in reducing losses of carbon via soil erosion, particularly in permanent woody cropping systems, such as olive groves and vineyards, where RR can be as high as 27–55% (Palese et al., 2014; Gonzalez-Sanchez et al., 2012; Favretto et al., 1992). As seen for retention of aboveground crop residues, cover crops can stimulate NPP of the main crop, and thereby increase carbon inputs to the soil. Some studies specifically addressing the effect of cover crops on the yield of the main crop indicate grain yield increases in small-grain cereals of up to 5% with undersown legume crops, while non-legume cover crops may have no or even negative effects on yield (Valkama et al., 2015). Daryanto et al. (2018) reviewed the effects of cover crops on both RR and yield and reported a mean RR of 9% from 28 studies. This is similar to the mean values calculated across all reviews by Bolinder et al. (2020). Daryanto et al. (2018) reported a relative yield increase of the cash crop of up to 27% following leguminous cover crops (N = 1005) and 6% with non-leguminous cover crops (N = 1282). The 'no cover crop' reference treatment in that study was a fallow period and the analysis only involved annual cash crops following this cover crop period, which may explain the high yield increases. In our current synthesis of the reviews, we noted that only Poeplau et al. (2015) assessed the effect on yield of the main crop under Swedish conditions (i.e., cover crops undersown in the main crop consisting mostly of cereals) and found that the relationship was not significant. A global review by Poeplau and Don (2015) also found no influence on SCR when testing the effect of plant functional types such as non-legume versus legume cover crops. These reviews were the only two examining the interactions of soil texture and climate with RR and SCR, and both concluded that such interactions were not significant.

Higher SOC changes were recently found in a global meta-analysis (1195 paired comparisons, 60% from North America) by Jian et al. (2020), which showed that including cover crops in rotations resulted in an overall RR of 15.5% and SCR of 0.56 Mg C ha^{-1} year^{-1}. That meta-analysis also reported higher RR for fine-textured soils (39.5%) than for medium- and coarse-textured soils (10.3–11.4%), an interaction with climate, with RR being twice as high in temperate (18.7%) compared with tropical (7.2%) climates and no significant effect of grass cover crop species on SOC changes. That was the only review on cover crops to include a relatively large number of observations including both topsoil and subsoil >30 cm (20% of the database). The authors stressed that, although SOC changes are generally reported to be negligible in deeper layers, this may be a consequence of limited subsoil observations.

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3.3 Recycling of organic materials

The effect of applying organic materials (mainly manures) was a mean RR of 30% (23.5–43.4%, N = 418) and SCR of 0.41 Mg C ha^{-1} year^{-1} (0.20–1.3 Mg C ha^{-1} year^{-1}, N = 217), based on seven reviews reporting effect size indices using a mineral-fertilized treatment as the reference (Maillard and Angers, 2014; Ladha et al., 2011; Kopittke et al., 2017; Körschens et al., 2013; Smith et al., 1997; Aguilera et al., 2013; VandenBygaart et al., 2003). However, some reviews derived effect size indices using either a mineral-fertilized or an unfertilized treatment as the reference, with RR and SCR being much higher in the latter case, for example, RR of 46% instead of 33% (Körschens et al., 2013) and SCR of 0.52 Mg C ha^{-1} year^{-1} instead of 0.31 Mg C ha^{-1} year^{-1} (Maillard and Angers, 2014).

The effect of recycling organic materials to soil on SOC obviously depends on the quantity applied and also on the quality, which determines the relative proportions transformed into more stable SOC components. The organic materials applied in the studies compiled by the reviews presented in Bolinder et al. (2020) were broadly defined as manures. In addition to varying amounts applied, some of these manures may have been more or less decomposed, fermented, or composted. This lack of exact information contributed to the variation in the RR and SCR values. The review by Maillard and Angers (2014) was the only one to assess the influence of soil texture on RR and SCR after manure addition and found that it was not significant. The review by Ladha et al. (2011) concluded that RR is not climate-dependent, while Maillard and Angers (2014) reported a trend of higher SCR only for temperate compared with tropical climates.

In another review, Han et al. (2016) conducted a global meta-analysis with 652 paired observations comparing manure and chemical fertilizer treatments against an unfertilized reference treatment and found a mean RR of 36.2%. In a more recent worldwide meta-analysis by Gross and Glaser (2021) using 529 paired comparisons, the mean RR was 35.4%, with a variation in RR similar to that reported in the synthesis by Bolinder et al. (2020). Gross and Glaser (2021) was attributing this variation to factors such as management, site properties, and manure characteristics. Their study considered a wide variety of manure types and found that, for example, pig, cattle, and farmyard manures had the highest RR values (50%, 32%, and 41%, respectively). They also found that more clayey soils showed higher SCR than sandy soils, but that sandy soils had higher RR than clayey soils, partly because of lower initial SOC contents. Similar to Bolinder et al. (2020), Gross and Glaser (2021) stressed the importance of considering changes in both RR and SCR to avoid misleading conclusions. They also found a trend for lower SCR in tropical soils compared with soils in cool and humid climates although, due to small numbers of observations for tropical soils, the trend was not significant.

In comparison with the reviews summarized by Bolinder et al. (2020), the database analyzed by Gross and Glaser (2021) included a relatively large proportion of data for sampling depth >30 cm (*N* = 103). The results showed that recycling organic materials to soil affected both RR (24%) and SCR (4.6 Mg C ha^{-1}) in the subsoil.

Although some reviews presented data for liquid manures, it proved challenging to make realistic comparisons between solid and liquid manures because of a lack of data (Maillard and Angers, 2014), and the effects were often not significant (e.g. Aguilera et al., 2013). Manure management systems have changed drastically over time toward liquid forms, for example, in Sweden more than 90% of all dairy and pig manures are now handled in liquid form with a low dry matter content (about 5%). Other changes in the treatment and use of manures are also occurring, for example, in Sweden about 4% of manures were used as substrate in biogas plants in 2015 (Anonymous, 2017). The result is a byproduct commonly named biofertilizer, which is returned to agricultural land almost entirely in liquid form (i.e. dry matter content typically only 1–5%). There is a lack of research on the effect of this biofertilizer on changes in SOC.

Application of recycled organic materials such as sewage sludge and municipal solid waste can induce very strong effects on SOC, as shown by two reviews with RR ranging from 98% to 117% and SCR from 1.65 to 5.29 Mg C ha^{-1} $year^{-1}$, respectively (Smith et al., 1997; Aguilera et al., 2013). However, sewage sludge application is subject to statutory restrictions and, as shown by Kirchmann et al. (2017), this limits the SOC sequestration rate to around 0.08 Mg C ha^{-1} $year^{-1}$ for sewage sludge in Sweden.

3.4 Nitrogen fertilization

The effect of nitrogen fertilization was a mean RR of 6.2% (3.5–10.0%, *N* = 846) and SCS of 0.23 Mg C ha^{-1} $year^{-1}$ (0.20–0.48 Mg C ha^{-1} $year^{-1}$, *N* = 183). The six reviews assessing this used an unfertilized or control plot as the reference (Ladha et al., 2011; Alvarez, 2005; Lu et al., 2011; Körschens et al., 2013; Aguilera et al., 2013; VandenBygaart et al., 2003). With the exception of one study with a high SCR value (Aguilera et al., 2013), similar to that of cover crops, the nitrogen fertilization effect was fairly consistent between studies. Two global reviews on RR (Ladha et al., 2011) and SCR (Alvarez, 2005) showed the closest agreement with the mean calculated from all reviews. The review by Ladha et al. (2011) found that RR was highest for tropical conditions and lowest for temperate climates, while that by Alvarez (2005) found that SCR was higher under humid and temperate conditions than under dry conditions or in tropical climates. Only Alvarez (2005) assessed the effect of soil texture and found a greater effect of nitrogen fertilization on SCR for sandy soils than for fine-textured soils.

In contrast to the control used for other management practices, the reference treatment with no application of nitrogen fertilizer is not common agronomic practice. In situations where no mineral nitrogen is applied, farmers usually compensate for this using nitrogen from other sources, such as manures. However, nitrogen may sometimes not be applied in regions where there is a lack of alternative sources. Further, the nutrient supply affects SOC dynamics in a complex manner through its simultaneous effects on NPP and on the rate of heterotrophic respiration through nitrogen mining and mineralization of organic matter (Poeplau et al., 2016). Nonetheless, it is generally recognized for agroecosystems that the effect of nitrogen fertilization on SOC is positive because of increasing NPP (and yields), resulting in higher carbon inputs to the soil from aboveground and belowground post-harvest crop residues (Christopher and Lal, 2007; Kätterer et al., 2012). The review by Alvarez (2005) demonstrated this by establishing a relationship indicating that SOC storage increased by 2 kg C ha^{-1} for each additional 1 kg N ha^{-1} applied. On analyzing LTEs under Nordic conditions, Kätterer et al. (2012) obtained similar results, with SOC in the topsoil (0–20 cm) increasing by 1–2 kg C ha^{-1} $year^{-1}$ for each extra kg of nitrogen applied. According to VandenByggart et al. (2003), using an unfertilized treatment as the reference may overestimate the effect of nitrogen fertilization, since both SOC and yield responses are lower at higher nitrogen application rates. However, even when zero-nitrogen treatments are excluded, by considering only the range of values receiving nitrogen, such positive relationships between nitrogen fertilization and SOC are still present.

4 Crop rotations

Crop rotations that include perennial forage crops, often termed leys, have components of cropland and temporary grassland. Although leys occupy a significant proportion of arable land at high latitudes, for example, almost 50% in Sweden, their impact on SOC has not been as thoroughly documented as that of the four management practices discussed in the previous section. Since SOC stocks in grassland are generally, but not always, higher than those in cropland (Poeplau et al., 2011; Mukumbuta and Hatano, 2019), rotations dominated by leys can be expected to approach SOC stocks intermediate to those in cropland and grassland (Crème et al., 2020). Kätterer et al. (2013) compiled data for 15 sites in Canada, Estonia, Norway, Sweden, and the UK, taken from eight publications (Uhlen, 1991; Viiralt, 1998; Yang and Kay, 2001; VandenBygaart et al., 2003; Quenum et al., 2004; Reintam, 2007; Johnston et al., 2009; Bolinder et al., 2010) and four unpublished datasets. They calculated mean annual SOC stock changes in pairs of rotations for sites including leys with those consisting of only annual crops. The duration of the LTEs in those cases varied between 10 and 58 years and the soil depth to which SOC was measured varied between

20 cm and 70 cm. The studies differed in ley species composition and frequency of leys in the rotation. According to the analysis, on average 0.52 Mg C ha^{-1} $year^{-1}$ (range 0.3–1.1 Mg C ha^{-1} $year^{-1}$) more carbon was retained in soils in ley-arable systems (including pure grass and legume leys as well as grass/legume mixtures) than in exclusively annual cropping systems (Kätterer et al., 2013).

Recently, a comprehensive review on the effect of selected crop rotations on SOC was conducted following the strict protocol for systematic reviews (Land et al., 2017). The outcomes from this work, including a meta-analysis, were recently published in Swedish (Land et al., 2021), but with supplementary material including the database in English, which can be freely accessed. Among the different crop rotations investigated, the effect of perennial forage crops was strongest and highly significant. It was found that SOC concentrations were on average 3.49 mg g^{-1} higher in rotations with perennial forage crops than in rotations with different annual crops. The difference between the final SOC concentration in 39 individual treatment pairs divided by the duration of the LTE (on average 24.5 years) was 0.16 mg g^{-1} $year^{-1}$ soil (Land et al., 2017). Assuming a bulk density of 1.3 Mg m^{-3}, data on which were only provided in a few studies, this equates to an SCR of 0.51 Mg C ha^{-1} $year^{-1}$ to 25 cm depth.

The study by Land et al. (2021) also analyzed the effect of various rotations and monocultures on SOC changes. Selected major findings from these meta-analyses are highlighted here. Effects of diverse rotations in general compared with repeated monocultures (203 pairs) were not significant. However, diverse rotations including legumes led to significantly higher SOC content than repeated monocultures without legumes (113 pairs, mean effect 0.351 mg g^{-1}), which roughly equates to stock changes of about 0.05 Mg C ha^{-1} $year^{-1}$, based on the same assumptions for bulk density presented above. Subsets from this dataset showed that the effect of legumes in diverse rotations compared with repeated monoculture without mineral fertilization was highly positive (18 pairs, 1.425 mg g^{-1}, corresponding to about 0.2 Mg C ha^{-1} $year^{-1}$). Due to the higher production of legumes under nitrogen-limited conditions, resulting in higher carbon inputs to soil, this response was expected. However, this comparison only applied to practical situations where no mineral fertilizer was applied when growing non-legume crops. Compared with repeated monocultures receiving fertilizer, the inclusion of legumes in diverse rotations still had a slightly positive mean effect on SOC, but this effect was not significantly different from zero (102 pairs). Comparing diverse rotations with and without legumes, the inclusion of legumes had a significantly positive effect on SOC (22 pairs, 0.611 mg g^{-1}, corresponding to about 0.09 Mg C ha^{-1} $year^{-1}$). In general, the inclusion of perennial forage crops in rotations was far more effective in increasing SOC than the inclusion of legumes (Land et al., 2017).

The positive impact of leys in arable rotations on SOC described above was seen in studies with different proportions of leys and other crops in rotations. Thus, the pure impact of leys compared with annual crops was probably diluted, since rotations entirely dominated by leys would probably have a stronger effect. This is supported by results from three LTEs in northern Sweden, where leys are grown with different frequencies in arable rotations (5, 3, 2, and 1 years in 6-year rotations A, B, C, and D, respectively) (Bolinder et al., 2010). Using annual SOC stock estimates according to Bolinder et al. (2010), which were based on equivalent mass principles, the calculated mean annual SOC mass increase compared with rotation D was 0.55 Mg C ha^{-1} year^{-1} for rotation A and 0.30 Mg C ha^{-1} year^{-1} for rotations B and C across the three sites. However, these SOC changes did not represent a pure ley effect, since crops and amendments differed between the four rotations. Nevertheless, ley frequency was probably the main driver according to a modeling study for one of the sites (Bolinder et al., 2012) and the estimated changes were within the range reported in the reviews cited earlier.

5 Soil tillage

Reducing the intensity of soil disturbance through tillage, particularly no-till practices, is desirable in many respects, for example, for saving labor and fuel and for preserving soil moisture and fauna in soil. Early assessments of no-till in the USA (Paustian et al., 1997) led to optimistic extrapolations of potential carbon stock increases at regional and global scale (Kern and Johnson, 1993; Freibauer et al., 2004). During recent years, several meta-analyses have revealed both positive effects (West and Post, 2002; Angers and Eriksen-Hamel, 2008; Gonzalez-Sanchez et al., 2012; Virto et al., 2012; Aguilera et al., 2013) and null effects on SOC (Dimassi et al., 2014; Powlson et al., 2014). Global meta-analyses have shown increases in SOC concentrations in the upper topsoil layers and decreases in the layers below, compared with conventional tillage (Luo et al., 2010b; Ogle et al., 2019). Only a few studies have included subsoil layers (i.e., below 30 cm), and these found that the positive effect of no-till was highly variable (Govaerts et al., 2009), reduced (Angers and Eriksen-Hamel, 2008), or even absent for some regions (VandenByggart et al., 2003; Luo et al., 2010b). Due to the high variability of its effects on SOC sequestration, the value of reduced tillage practices for climate change mitigation is now being questioned (Powlson et al., 2014; VandenBygaart, 2016).

The most comprehensive recent systematic review on this subject, including 351 LTEs, reported rather moderate effects (4.61 Mg C ha^{-1} on average) of no-till over conventional tillage on SOC stocks in the topsoil (0–30 cm), while no significant tillage effect was detected in the full soil profile (Haddaway et al., 2017). In the majority of studies, effect size indices were calculated considering a fixed soil depth but, as bulk density is often affected by tillage, this may introduce considerable bias (Ellert et al., 2002). Meurer et al. (2018) selected a subset of 101 LTEs from the database provided by Haddaway et al. (2017), which reported both SOC and bulk densities for calculating SOC stocks with the equivalent soil mass approach (Ellert and Bettany, 1995). According to their analysis, the carbon gain in no-till over conventional tillage decreased with depth and even became non-significant in studies where SOC stocks could be calculated to an equivalent depth of 60 cm, that is, 1.15 Mg ha^{-1}, which divided by the mean duration of the LTEs gave 0.06 Mg ha^{-1} year^{-1} (Meurer et al., 2018). The effect size of no-till compared with conventional tillage to an equivalent soil depth of 30 cm (0.2 Mg C ha^{-1}) was lower in their study than that presented by Haddaway et al. (2017) (4.2 Mg C ha^{-1} year^{-1}), which may partly be due to slightly higher bulk densities in no-till treatments. Thus, the sequestration potential of no-till seems indeed to be over-estimated when neglecting deeper soil depths. However, only 11 LTEs permitted calculation of SOC stocks to 60 cm depth, so there is a need for additional assessments in ongoing long-term tillage studies including deeper depth layers to confirm these findings.

Moving to the other extreme of tillage operations, deep plowing to 1 m depth to improve soil structure and/or break up hard pans in podzols has been shown to increase SOC stocks significantly at some sites (Alcántara et al., 2016; Schneider and Don, 2019). However, this practice is irreversible and can affect soil ecosystem services in several ways, an issue that needs to be further assessed (Schiedung et al., 2019). Studies from New Zealand show that full inversion tillage of pastures to 30 cm depth resulted in increased SOC stocks and that the additional costs associated with this can be offset by the increase in productivity already in the first year after pasture renewal (Beare et al., 2020; Lawrence-Smith et al., 2021). The generality of these results also needs further consideration.

6 Subsoil

The amount of SOC stored in subsoil (>30 to 100 cm depth) is similar to that in the topsoil arable layer at 0–30 cm depth (Morari et al., 2019). However, LTEs examining common management practices for SCS often only consider the arable layer, where most SOC changes are assumed to occur because carbon cycling is more dynamic in topsoil than in deeper soil layers (Bolinder et al., 2020). In fact, the mean carbon age of SOC decreases sharply with depth (Balesdent et al., 2018), but this does not imply that subsoil SOC is insensitive to agricultural management practices. There is evidence that common management practices affect SOC stocks in the upper part of the subsoil or in deeper layers at decadal time scales (e.g., Börjesson et al., 2018; Dal Ferro et al., 2020; Kätterer et al., 2014; Kirchmann et al., 2013; Menichetti et al., 2015). However, this is not the case at all sites (Jarvis et al., 2017; Börjesson et al., 2018). For instance, Börjesson et al. (2018) presented SCRs calculated according to equivalent mass principles to

50 cm depth for two Swedish LTEs, on loam and sand, with an identical experimental set-up comparing a cereal monoculture with a ley-dominated rotation. During the 35 years of the trials, SCR was positive in the fully fertilized ley-dominated rotation at both sites (0.47 and 0.17 Mg ha^{-1} year^{-1} in the loam and clay, respectively) and negative in cereal monoculture (−0.38 and −0.15 M ha^{-1} year^{-1}, respectively). Around 27% of the crop rotation effect occurred below 20 cm depth in the loam, but no effect at all on subsoil C was detected in the clay soil. This illustrates that site-specific conditions such as soil texture play an important role in both carbon inputs and retention in soil profiles (Poeplau and Kätterer, 2017).

The rationale for including deeper soil layers in any analysis of SOC stock changes is obvious when considering tillage practices. Including deeper soil layers then significantly influences the overall conclusions drawn on total SOC stock changes, since tillage affects the depth distribution of SOC (Angers and Eriksen-Hamel, 2008; Luo et al., 2010b; Meurer et al., 2018). Including subsoil is also important when evaluating and comparing annual and perennial crops with more or less well-developed root systems or other deep-rooting species (Carter and Gregorich, 2010; Collins et al., 2010; VandenBygaart et al., 2011; Guan et al., 2016). It has been shown that major land-use changes, such as cropland to grassland or pasture or cropland to forest, and vice versa, may in some cases also induce changes in subsoil carbon (Guo and Gifford, 2002; Poeplau and Don, 2013). A recent analysis on the conversion of former cropland on environmentally sensitive soils in the USA into grassland within the Conservation Reserve Program showed that gains in topsoil carbon may have been negated by net losses at depth (Yang et al., 2021). There is thus strong reason to include SOC changes in the subsoil in carbon accounting systems (Rumpel and Kögel-Knabner, 2011). However, the responses are site-specific and the mechanisms governing the effects on subsoil SOC have been insufficiently studied, which has been identified as a major knowledge gap (Lorenz and Lal, 2005; Chenu et al., 2019).

Since carbon changes are small relative to total SOC stocks, LTEs are indispensable when quantifying the net effect of different management practices on subsoil carbon. To our knowledge, reviews focusing on agricultural management impacts on subsoil SCS are currently lacking. Future efforts are urgently needed in data compilation and analysis of existing data, and further research is needed on the mechanisms controlling the movement of organic carbon into subsoils and its stabilization (Balesdent et al., 2018; Kögel-Knabner and Amelung, 2021; Simo et al., 2019; Torres-Sallan et al., 2017).

7 Case study: soil carbon and fertility

Rebuilding SOC in agricultural soils is frequently mentioned as a key component of sustainable intensification, due to its positive feedback on soil fertility and food security (e.g. Lal, 2004, 2013; Foley et al., 2011; Henryson et al., 2018). Although many studies show synergies between SOC and crop productivity (Lal, 2013; Oldfield et al., 2019), the responses seem not to be universal (Edmeades, 2003; Hijbeek et al., 2017) and may even be reversed in data analyses at regional or continental scale (Oelofse et al., 2015; Vonk et al., 2020). Confounding correlations, such as negative correlations between SOC and pH, further complicate this type of analysis (Kirchmann et al., 2020).

In the following, we present a case study of a Swedish LTE comparing two treatments, with and without the addition of cereal straw corresponding to 1.8 Mg C ha^{-1} year $^{-1}$ (Kätterer et al., 2011). The most frequent confounding factors, such as treatment differences in pH and crop nutrient supply, are most likely negligible in this comparison. Both treatments are fertilized annually with nitrogen, phosphorus, and potassium, and these elements are probably not limiting for crop growth. Differences in crop yield between the two treatments have increased over time since the start of the LTE in 1956. We regressed the significant linear slope of a regression model fitted to the time series of annual yield ratios in the straw-amended compared with control treatments on measured SOC concentration changes in the straw treatment compared with the control. The results clearly showed increasing yield with increasing SOC concentration, with a slope of 4.4% (Fig. 3).

This case study was performed on clay soil in Sweden, and extrapolation of the results is not straightforward. However, the results show that the synergy between SOC storage and yield potential can be considerable, at least in some regions. They also support findings in previous analyses of LTEs, for example, by Körschens et al. (2013), who reported a yield benefit of 6% in manured treatments compared with mineral fertilization alone. We interpreted the strong crop yield response to SOC increase in the Ultuna LTE as being mainly governed by soil structural changes related to SOC, through increasing the amount of plant-available water (Meurer et al., 2020) and creating a better environment for root growth, thereby increasing the uptake efficiency of water and nutrients. This complies with the interpretation of manure-induced yield increases in European LTEs (Zavatarro et al., 2017). The dynamic interactions between SOC and soil structure have recently been formalized in a modeling approach that represents the observed soil structural changes in this trial very well (Meurer et al., 2020). An important difference between the Ultuna experiment and most other LTEs, which may exaggerate the effect of SOC on soil structure to some extent, is that Ultuna is a small-plot trial (2 m × 2 m) that is managed entirely by hand, thus preventing soil compaction. In large plot experiments using full-scale machinery, the situation may differ, and under real-life field conditions, it is widely recognized by farmers that soil compaction results in yield reductions through reduced aeration and water infiltration, which increases the risk of waterlogging.

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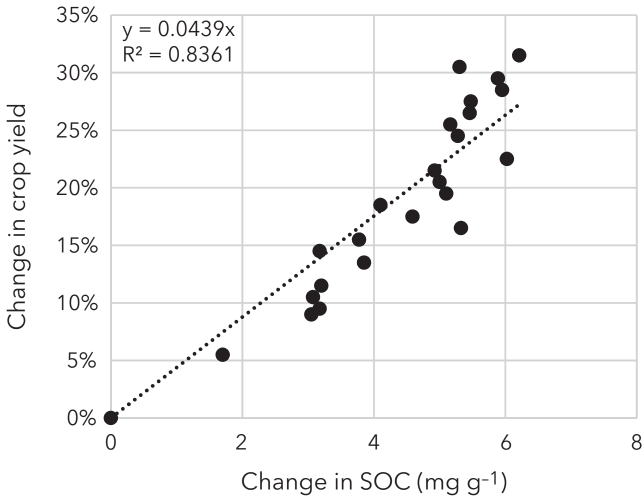

Figure 3 Crop yield response to increasing soil organic carbon (SOC) concentration in the topsoil during 63 years in the Ultuna long-term experiment (Kätterer et al., 2011). Yield changes correspond to linear regression (significant slope) of crop yield over time in a treatment amended with straw compared with treatment without straw. Changes in SOC between the two treatments were calculated from direct measurements recorded for several years and differences steadily increased over time. In 2019, 63 years after the start of the experiment, topsoil SOC concentration was 6.2 mg g^{-1} higher in the straw-amended treatment compared with the control. Both treatments received mineral fertilizer, so the yield response is probably mainly due to changes in soil physical properties.

8 Conclusion and future trends

Since annual changes in SOC are small compared with total SOC stocks, which vary greatly even at small scales, data from LTEs are essential when quantifying the impact of agricultural management on SOC stocks. The responses of different management practices on SCR synthesized in this review are summarized in Fig. 4. Studies reporting SCR were less common than those reporting RR because bulk density, which is laborious to measure, is not required to calculate RR based on SOC concentration measures. Among the practices considered, crop residue retention compared with removal was that most commonly investigated in LTEs, followed by manure addition, nitrogen fertilization, cover crops, and rotations. Changes in SOC stocks were highest when comparing perennial with annual crops and lowest, though still statistically significant when comparing diverse rotations including legumes with monocultures. LTEs investigating tillage effects were frequently described, but only a limited number of those reported bulk density values, which is a prerequisite for calculating SOC stock changes. Carbon stocks increased in no-till treatments compared with conventional

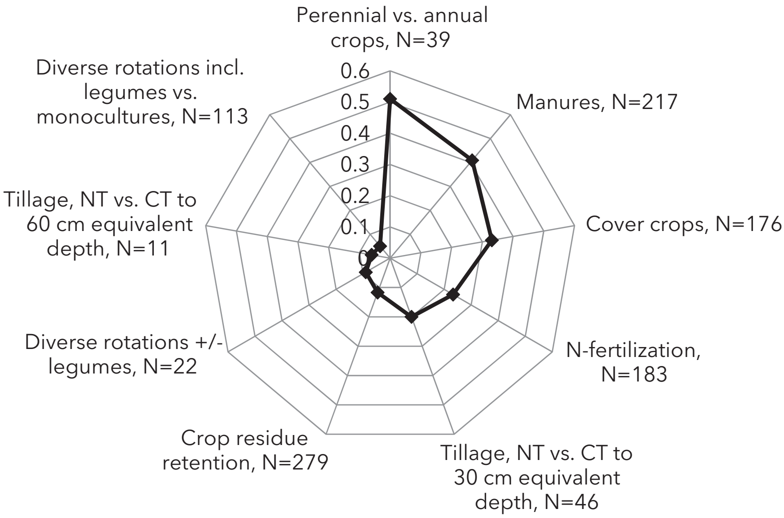

Figure 4 The overall average effect of different management practices on mean stock change rate (Mg C ha^{-1} $year^{-1}$), weighted by the number of paired comparisons (*N*), in the reviews presented in this chapter (Meurer et al., 2018; Bolinder et al., 2020; Land et al., 2021). NT, no-till; CT, conventional tillage.

tillage to 30 cm depth, but in studies reporting SOC to 60 cm depth, the SOC gain was much lower, and not significantly different from zero.

8.1 Carbon stock changes and long-term field experiments

The majority of the field-based estimates presented in Fig. 4 were found to be statistically significant in meta-analyses. Hence, they are valid for inclusion in large-scale analyses and planning. Most estimates of global SCS potential were directly derived from LTEs or indirectly from models based on LTEs. Nevertheless, extrapolation from LTEs to larger scales is not straightforward. For example, LTEs are more abundant in temperate than in tropical climate zones, and publications from certain regions, for example, Russia, are under-represented due to lack of cataloging in digital scientific databases or the use of native languages, making them difficult to find (Haddaway et al., 2015). There is a lack of LTEs in some regions, especially developing countries and the distribution of LTEs in reviews and meta-analyses is frequently also biased toward regions where certain management practices are common, as illustrated by the over-representation of tillage studies from North America (Haddaway et al., 2015, 2017).

Potential biases, especially in long time series, should also be considered, since management practices in LTEs, such as cultivars, fertilization, crop protection, or liming, do not always match common agricultural practices. Consequently, the yields are not necessarily optimized and the relative difference in yields, which is the main driver of annual crop residue carbon inputs to soil, may be lower in LTEs than on real farms. This could also cause some difficulties in the interpretation of effect size indices when combining data from shorter- and medium-term LTEs with data from much longer-term experiments. The RR and SCR values are relative assessments, most commonly using only the most recent measurements of SOC in LTEs, but it is possible that SOC content may have declined over time in both the management practice treatment and the control treatment, depending on the initial conditions at the start of the experiment (Bolinder et al., 2020). Using the time series of SOC measurements in LTEs could improve the estimates of RR and SCR (Haddaway et al., 2016).

Differences in tillage depth over time or in drainage conditions may also be prevalent and may create biases (Warren et al., 2020). Although the exact time when the changes take place is difficult to establish, some studies indicate that plowing depth may have increased by as much as 7.5–10 cm between around 1950–1960 and 1970–1990 (e.g. Van Meirvenne et al., 1996; Riley and Bakkegard, 2006). This can create a distortion in the time series of data used for SOC measurement through dilution of SOC by the inclusion of poor subsoil material into the topsoil. Hence, conclusions on absolute trends in SOC may be biased when comparing management practices from different databases, such as observations from LTEs with large plot sizes using modern full-scale machinery or soil monitoring programs. This dilution can also affect the calibration of SOC models and needs to be accounted for when modeling SOC stocks, as shown by, for example, van Wesemael et al. (2005). The movement of soil from one plot to another through tillage operations is another problem in small-plot experiments (Sibbesen, 1986). Furthermore, some LTEs include extreme treatments that are not scalable due to legal restrictions, as we pointed out with the example of application rates of sewage sludge in Sweden (Kirchmann et al., 2017). Another difficulty relating to the effects of recycled organic materials on SOC changes arises in LTEs with manures, since the amounts applied in these experiments do not necessarily reflect current agronomic practices and the SCR estimates are probably too optimistic in some cases. In fact, during recent decades there has been a lowering of application rates because of the introduction of nutrient-related regulations in many countries.

8.2 Other management practices

Several important agricultural management practices are under-represented in LTEs. Kirchmann et al. (2020) found that only 9 out of 735 LTEs that they reviewed investigated liming interventions according to the systematic map presented by Haddaway et al. (2015), despite soil acidity being a major yield-limiting factor in large agricultural areas. Lime application has been shown to decrease nitrous oxide emissions from soils (Scheer et al., 2008; Wang et al., 2018) and increase methane oxidation (Kunhikrishnan et al., 2016), but its effect on the carbon balance is understudied. It is well-known that soil acidity affects the composition and diversity of microbial communities, with associated feedbacks on carbon and nitrogen cycling in soil (Bahram et al., 2018; Malik et al., 2018), including stabilization mechanisms mediated, for example, by changes in bacteria/fungi ratio (Rousk et al., 2009). In their review, Paradelo et al. (2015) identified this topic as a knowledge gap and encouraged the soil science community to synthesize unpublished data from existing LTEs.

Field application of biochar is a measure that can increase SOC to the same order of magnitude as SCS (Paustian et al., 2016; Minx et al., 2018). The thermal decomposition of biomass under low oxygen conditions during pyrolysis makes biochar highly resistant to heterotrophic organisms in soil (Lehmann et al., 2006). As most of the biochar is not transformed to soil organic matter, the SOC increase after its application does not comply with the strict definition of SCS according to Olson et al. (2014). Nevertheless, due to its positive effects on soil fertility, biochar application generally leads to higher crop yields, especially in sub-tropical and tropical climates, as shown in several meta-analyses (Liu et al., 2013; Jeffery et al., 2017; Ye et al., 2020). Yield responses appear to persist for at least 20 growing seasons, as shown for LTEs in Kenya (Kätterer et al., 2019). The total climate change mitigation benefit of biochar technologies depends on the origin of the biomass and how efficiently the energetic gases evolving during pyrolysis are used, including substation effects (Tisserant and Cherubini, 2019; Sundberg et al., 2020). LTEs quantifying carbon stock changes due to biochar application is still scarce. Furthermore, biochar-induced nitrogen mining through priming of soil organic matter is a potential risk that should be evaluated before scaling-up this technology under nutrient-poor conditions (Wardle et al., 2008; Wang et al., 2016).

Irrigation is another topic that is insufficiently studied in relation to SOC. Despite its significance for food production and use on more than 0.3 billion ha, Trost et al. (2013) found only 14 LTEs reporting SOC changes due to irrigation practices. The results from these studies were highly variable, with up to 500% increases in SOC in irrigated desert soils, moderate increases in semi-arid regions, and no consistent effects in humid regions. Consideration of potential tradeoffs relating to water scarcity elsewhere, groundwater depletion, and nitrous oxide emissions are likely to be particularly relevant for irrigation practices.

Planting more trees or bushes in agricultural landscapes could also boost aboveground carbon storage and SCS (Menichetti et al., 2020). However, few LTEs on agroforestry cover more than a decade (Cardinael et al., 2018; Corbeels

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et al., 2018). Agroforestry systems are also highly diverse and region-specific, and their contribution as a carbon sink is often not accounted for in national inventories (Rosenstock et al., 2019). As seen for agroforestry, the introduction of perennial energy crops on cropland will often lead to SCR, as shown in a meta-analysis for woody species based mainly on European trials (Don et al., 2012). In addition, energy grasses have the potential to sequester carbon, as shown by Poeplau and Don (2012). Rough estimates of SCR provided in these two reviews lie around 0.4 Mg C ha^{-1} $year^{-1}$, which is similar to that leys in crop rotation trials. Much higher SCR values, that is, 1.56 Mg C ha^{-1} $year^{-1}$ and 0.68 Mg C ha^{-1} $year^{-1}$ on average for herbaceous and woody perennials, respectively, were reported in the meta-analysis by Agostini et al. (2015). However, if energy production is restricted to abandoned and marginal land in order to avoid competition for land with food productions, these estimates for SCR are probably not scalable.

Avoiding soil compaction through controlled traffic, i.e. constraining field traffic to the smallest possible area of permanent traffic lines, may also have the potential to increase crop productivity and SOC stocks, and mitigate nitrous oxide emissions (Antille et al., 2015; Hernandez-Ramirez et al., 2021). However, this technology is rather new and LTEs quantifying its effects on SOC are still lacking. Amelioration of subsoil conditions to improve soil fertility is another topic that should be investigated further. This may involve soil drainage, a measure for increasing both productivity and decomposition, and tillage operations aimed at loosening compact subsoil layers. Besides deep plowing operations, which have been shown to result in SOC sequestration in the few studies available, loosening of upper subsoil layers combined with injection of organic materials may be another way to increase soil fertility and SOC stocks (Getahun et al., 2018).

Organic farming practices are promoted in many countries, especially in the European Union, based in part on the argument that it leads to SCS (Freibauer et al., 2004). Higher average SOC stocks in organic than in conventional systems may be interpreted as an advantage of organic systems at farm scale (Smith, 2004; Goh, 2011; Gattinger et al., 2012). However, comparing conventional stockless farms with low external carbon inputs with organic mixed farms with inputs from manures, or even off-farm carbon sources transferred from conventional systems, requires system boundaries to be considered (Goulding et al., 2009; Kirchmann et al, 2016). Higher SOC stocks in organic farming can be due to high and often disproportionate application of organic fertilizers (Leifeld and Fuhrer, 2010; Leifeld et al., 2013) and transfer of nutrients from conventional to organic systems (Nowak et al., 2013). Moreover, organic farming often involves a combination of several of the management practices covered in this chapter, such as cover crops and leaving crop residues in the field, based on the belief that there is a synergetic effect. This may indeed be the case but needs further investigation, especially as these practices are also used in conventional farming systems. More importantly, all of the above-mentioned considerations make it

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very difficult to perform appropriate and equitable assessments on the effect of organic farming systems, both at farm scale and in LTEs, because these systems involve too many confounding factors acting simultaneously (Kirchmann et al., 2016). The lower yields in organic compared with conventional systems implies a SOC stock decrease in situ and a requirement for more cropland to produce an equivalent amount of food (Kirchmann et al., 2007). This may induce indirect land-use change, including deforestation, with accompanying losses of SOC at larger spatial scales (Kirchmann et al., 2016; Searchinger et al., 2018).

8.3 Uncertainty and scaling-up in space and time

At the 21st Conference of the Parties meeting in Paris, the French government launched the 4 per 1000 initiative (www.4p1000.org) to promote SCS, based on the rationale that an increase of 0.4% in global SOC stocks could offset all anthropogenic carbon dioxide emissions. Although the intention of the initiative was to set this as a normative target (Rumpel et al., 2020), it has created lively discussions in the scientific literature about the feasibility of SCS in practical implementation (Minasny et al., 2017; VandenBygaart, 2018; Poulton et al., 2018; Amelung et al., 2020). Among other issues, the sources of nitrogen and other nutrients required for building soil organic matter are a crucial climate-related issue as long as fertilizers are produced using fossil energy (van Groenigen et al., 2017). The carbon inputs required to reach the target would have to increase by 43%, or 0.66 Mg ha^{-1} year^{-1}, according to a recent study (Bruni et al., 2021). No single measure among those presented in Fig. 4 would be sufficient to reach this target.

Despite the uncertainties mentioned above and the incomplete understanding of how SCS is influenced by management and pedo-climatic factors (Chenu et al., 2019; Smith et al., 2020), the evidence from LTEs on several major management practices affecting SOC stocks is quite solid and can be used to design more climate-smart production systems and applied in bottom-up approaches to estimate biophysical SCS potential at larger scales. Several of these practices, like cover crops, more perennial crops, and surface mulch due to reduced tillage, will also directly lead to global cooling through associated increases in surface albedo (Lugato et al., 2020). However, some practices may lead to increased nitrous oxide emissions, which may lower the mitigation potential (Guenet et al., 2021). Moreover, when extrapolating the effects obtained from field measurements in one region to another with different soils, climate conditions, and agricultural practices, the estimates may become highly uncertain or even false. Proper boundary conditions must be set and tradeoffs with food production and other ecosystem services, leakage through indirect land-use change, non-CO_2 GHG, and the United Nations

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Sustainable Development Goals must be considered when upscaling SCS measures in space.

It must be emphasized that not all measures are scalable in space. For example, manures that increase SOC stocks at field or farm scale (Fig. 4) will not contribute to SCS at larger scales as long as the number of animals or manure handling/treatment methods in a given region does not change. The same applies to other limited organic resources, such as sewage sludge, composts, or other wastes from the food chain. To optimize the use of biomass, alternative pathways for any organic resource have to be evaluated. Although crop residue retention in the field will most likely improve soil health and lead to SOC gains, it may not always be the best option. Instead, part of the residues could be pyrolyzed and the evolving gases could be upgraded to fuels, replacing fuels of fossil origin, while the biochar produced could first be used for water purification or as a feed additive to cows to reduce methane emissions, and then applied to soil, where it will improve soil fertility and lead to negative carbon emissions. This is just one example of a win-win-win strategy that can result in more SOC with several other co-benefits. We view SCS as one component in the sustainable development of emerging bioeconomies. Transdisciplinary multi-actor approaches will be necessary to identify and develop region-specific pathways for more sustainable societies encompassing all dimensions of sustainable development.

9 Where to look for further information

Many of the reviews cited in this chapter provide excellent overviews of the research area.

Dr. Rattan Lal, Professor of Soil Science and Director of the Carbon Management and Sequestration Center at Ohio State University, has edited many textbooks that include articles from many leading scientists. He is easy to find on the internet. The following book is just an example: Lal, R. and Stewart, B.A. (2018), *Soil and Climate*. CRC Press, Boca Raton, doi:10.1201/b21225

Key research in this area can be found on the websites of the following organizations:

- United States Department of Agriculture (USDA; www.usda.gov).
- Joint European Research Program to develop knowledge, tools and an integrated research community to foster climate-smart sustainable agricultural soil (https://ejpsoil.eu/).
- The Global Research Alliance on Agricultural Greenhouse Gases https://globalresearchalliance.org/.
- All major soil science departments at universities and research institutes.

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