frontiers in Climate

REVIEW
published: 16 October 2019
doi: 10.3389/fclim.2019.00008

Soil C Sequestration as a Biological Negative Emission Strategy

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Soil carbon (C) sequestration is one of three main approaches to carbon dioxide removal and storage through management of terrestrial ecosystems. Soil C sequestration relies of the adoption of improved management practices that increase the amount of carbon stored as soil organic matter, primarily in cropland and grazing lands. These C sequestering practices act by increasing the rate of input of plant-derived residues to soils and/or by reducing the rates of turnover of organic C stocks already in the soil. In addition to carbon dioxide removal potential, increases in soil organic matter/soil C content are highly beneficial from the standpoint of soil health and soil fertility. Practices to increase soil C stocks include well-known, proven techniques, or "best management practices" (BMP) for building soil carbon. A second category includes what we refer to as frontier technologies for which significant technological and/or economic barriers exist today, but for which further R&D and/or economic incentives might offer the potential for greater sequestration over the longer term. We reviewed published estimates of global soil carbon sequestration potential, representing the biophysical potential for managed cropland and/or grassland systems to store additional carbon assuming widespread (near complete) adoption of BMPs. The majority of studies suggests that 4–5 $GtCO_2$/y as an upper limit for global biophysical potential with near complete adoption of BMPs. In the longer-term, if frontier technologies are successfully deployed, the global estimate might grow to 8 $GtCO_2$/y. There is a strong scientific basis for managing agricultural soils to act as a significant carbon (C) sink over the next several decades. A two-stage strategy, to first incentivize adoption of well-developed, conventional soil C sequestering practices, while investing in R&D on new frontier technologies that could come on-line in the next 2–3 decades, could maximize benefits. Implementation of such policies will require robust, scientifically-sound measurement, reporting, and verification (MRV) systems to track that policy goals are being met and that claimed increases in soil C stocks are real.

Keywords: biophysical potential, carbon sequestration, best management practice, carbon dioxide removal and storage, agricultural soils

OPEN ACCESS

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Specialty section:
This article was submitted to Negative Emission Technologies, a section of the journal Frontiers in Climate

Received: *30 June 2019*
Accepted: *19 September 2019*
Published: *16 October 2019*

Citation:
Paustian K, Larson E, Kent J, Marx E and Swan A (2019) Soil C Sequestration as a Biological Negative Emission Strategy. Front. Clim. 1:8. doi: 10.3389/fclim.2019.00008

INTRODUCTION

Together with other terrestrial ecosystem-based strategies for CO_2 removal [i.e., afforestation, bioenergy with carbon capture and storage (BECCS)], soil carbon sequestration relies on plant photosynthesis to carry out the initial step of carbon "removal" from the atmosphere. However, rather than increasing the storage of carbon contained in plant biomass, soil C sequestration relies on management practices that increase the amount of carbon stored as soil organic matter, primarily in cropland and grazing lands. Part of the attraction of soil C sequestration as a biological negative emission (BNE) strategy is that carbon stocks are most depleted on lands currently under agricultural management and thus this approach to CO_2 removal doesn't require land use conversions (e.g., to forests) and competition for land resources. In addition, increases in soil organic matter/soil C content are highly beneficial from the standpoint of soil health and soil fertility, which provides additional incentives for adopting soil carbon sequestering practices. In this paper, we review and summarize data and understanding, from field to global scale, of the capacity for soil carbon sequestration to play a role in helping draw down atmospheric CO_2 concentration (NASEM, 2019).

Globally, soils contain about 1,500 Gt of organic carbon (C)[1] to 1 m depth and 2,400 GtC to 2 m depth (Batjes, 1996). Thus, the total size of the soil carbon reservoir exceeds the total mass of carbon in vegetation and atmosphere combined. About 45% of global soils are under some form of agricultural use (i.e., cropland and grazing land). In most soils, organic matter[2] makes up a small fraction (∼1–10%) of the total soil mass which is dominated by mineral matter (i.e., sand, silt, and clay particles); these are so-called "mineral soils." The vast majority of agricultural lands are on mineral soils. In contrast, "organic soils" (e.g., peat soils), as the name implies, have very high organic matter content. Organic soils form where anaerobic conditions restrict decomposition, such that partially decayed plant material accumulates, making up the matrix of the soil.

Most agricultural soils (both mineral and organic) are depleted in C relative to the native ecosystems from which they were derived, due to reduced net primary production and export of harvested biomass—which reduce C inputs to soil; nutrient depletion, intensive soil disturbance, and soil erosion are other contributing factors to soil C depletion (Paustian et al., 1997). Most cropland mineral soils have lost 30–50% of the C stocks in top soil layers (0–30 cm) relative to their native condition (Davidson and Ackerman, 1993). In contrast, grassland soils managed for grazing may or may not have suffered similar C losses relative to their native state, depending on how they have been managed. Grasslands that have been overgrazed and poorly managed are likely significantly depleted in soil C, whereas well-managed grasslands may have C stocks equal to or exceeding their original native condition (Conant et al., 2016).

[1] In this paper, C refers to carbon and CO_{2eq} refers to CO_2-equivalents. One tone of C is equivalent to 3.67 tones of CO_{2eq}.
[2] About 50% of the mass of soil organic matter is carbon.

The organic carbon content of soils is governed by the balance between the rate of C added to the soil from plant residues (including roots) and organic amendments (e.g., manure, compost), and the rate of C lost from the soils, which is mainly as CO_2 from decomposition processes (i.e., heterotrophic soil respiration). Other organic C can be lost as CH_4 from anaerobic (e.g., flooded) systems as well as from leaching of dissolved organic C, but these are minor loss processes in most ecosystems. Soil erosion can greatly affect C stocks at a particular location, but at larger scales erosion may not represent a loss process *per se* but rather a redistribution of soil C. Effects of erosion on the global C balance is a subject of continued research but soil erosion may actually result in a small net C sink, because burial of C-rich sediment reduces its decomposition rate and, with erosional exposure, low C subsurface soil layers can have a higher capacity to store additional C (van Oost et al., 2007).

In native ecosystems the rate of detrital C inputs is a function of the type (e.g., annual vs. perennial, woody vs. herbaceous) and productivity of the vegetation, largely governed by climate but also nutrient availability and other growth determining factors. Decomposition rates are controlled by a variety of factors including soil temperature and moisture, drainage (impacting soil O_2 status) and pH. Soil physical characteristics such as texture and clay mineralogy also impact the longevity and persistence (i.e., mean residence time) of soil C, by affecting organic matter stabilization processes, i.e., the extent to which organic matter is "protected" from decomposition through mineral-organic matter associations (Schmidt et al., 2011).

In managed ecosystems such as cropland and grazing land both the rate of C input as well as the rate of soil C loss via decomposition are impacted by the soil and crop management practices applied. In general, soil C stocks can be increased by: (a) increasing the rate of C addition to the soil, which removes CO_2 from the atmosphere, and/or (b) reducing the relative rate of loss (as CO_2) via decomposition, which reduces emissions to the atmosphere that would otherwise occur.

However, three key points need to be made regarding the pattern of gains or losses of soil C. The first is that with increased C inputs and/or decreased decomposition rates, soil C stocks tend toward a new equilibrium state and thus after a few decades C gains attenuate, becoming increasingly small over time (Paustian, 2014). Secondly, because the soil C balance is governed by biotic processes, changes in management that lead to C gains are potentially reversible, i.e., if management reverts back to its previous condition, much or all of the gained C can be lost. Thus, practices that led to increased soil C need to be maintained long term. Third, mineral soils (i.e., non-peat soils) have an upper limit or "saturation level" of soil C (Six et al., 2002). While this maximum soil C concentration is much higher than is found in most managed soils, it does mean that soils that already have very high organic matter levels (e.g., >5% C by mass) have a low propensity for further C gains.

An additional consideration that has been raised regarding constraints on aggressive targets for soil C sequestration, is the need for additional inputs of nitrogen (van Groenigen et al., 2017). In most mineral soils, soil organic matter has a relatively narrow C:N stoichiometry, typically ranging from 8 to 20, with a

C:N of 10–12 as a general "rule-of-thumb" for agricultural soils. Thus, to maintain this balance, if soil organic matter stocks were to increase by say 4 billion tons CO_{2eq}/y (1.1 GtC/y), then about 100 million tons per year of N would need to be incorporated into the added soil organic matter. van Groenigen et al. (2017) point out that this is equivalent to about 75% of the current global synthetic N fertilizer production. While this is a valid point, many of practices being promoted for increasing soil C include using more legumes (e.g., N-fixing cover crops, legume hay/pastures in rotation with annual crops) that could help meet demands for additional N inputs into soil organic matter. Moreover, many cropland soils in North America, Europe, China, India, and SE Asia currently lose a significant amount of added N (from fertilizer, manure, N-fixation) as gaseous losses and leached nitrate, and thus improved practices that could "mop up" some of this N and incorporate it into soil organic matter would yield multiple environmental benefits. Undoubtedly, improved management of N inputs, both to sustain crop productivity and soil organic matter increases and to minimize N_2O emissions (the most potent biogenic GHG on a per mass basis) and other losses of pollution-causing reactive nitrogen to the environment, will be an important part of strategies for negative emissions from soils.

MANAGEMENT PRACTICES TO INCREASE SOIL C STORAGE AND NET CO_2 REMOVALS

In evaluating management interventions to increase soil C stocks, the recent National Academies report divides soil carbon sequestering management practices into two broad categories (NASEM, 2019). The first category includes known, proven conservation management systems that can increase soil C on lands with existing crops and management techniques. These are practices that are typically not (yet) in dominant use, but are being practiced by more conservation-minded farmers and have the potential to become much more widely adopted. Such management techniques can be referred to as BMPs ("Best Management Practices") for increasing soil carbon storage. With proper incentives, such BMPs can be quickly adopted to provide near-term soil C stock increases. The second category of practices are referred to as "frontier technologies" which represent systems or practices for which significant technological and/or economic barriers exist. Thus, they represent technologies and practices that are still largely experimental, with little or no occurrence in production agricultural systems and thus are not yet mature enough to deploy at scale. However, with further R&D and sufficient economic incentives these frontier technologies may offer the potential for greater soil C increases over the longer term.

Conventional Conservation Practices (BMPs) to Sequester Soil C

Conservation practices that can contribute to an increase in carbon stocks in soils are well-known from numerous field experiments and comparative field observations. **Table 1** lists several classes of practices, classified according to their main mode of action in either increasing C inputs to soils and/or reducing C losses from soils.

TABLE 1 | Examples of agricultural management actions that can increase organic carbon storage and promote a net removal of CO_2 from the atmosphere and the main mode of action on the soil C balance (from Paustian, 2014).

Management practice	Increased C inputs	Reduced C losses
Improved crop rotations and increased crop residues	✓	
Cover crops	✓	
Conversion to perennial grasses and legumes	✓	✓
Manure and compost addition	✓	
No-tillage and other conservation tillage		✓
Rewetting organic (i.e., peat and muck) soils		✓
Improved grazing land management	✓	

Improved Crop Rotations and Cover Cropping

Farmers may adopt a number of cropping choices that increase inputs of C into soils: planting of high-residue crops, seasonal cover crops/green manure, continuous cropping (reduced fallow frequency), and planting of permanent or rotated perennial grasses (CAST, 2004). For example, a recent global review of cover crops reported a mean annual sequestration rate of 0.32 tC/ha/y, with several studies reporting rates higher than 1 tC/ha/y (Poeplau and Don, 2015). In many dry climates, farmers fallow croplands every other year to conserve soil moisture and stabilize grain yields. Intensifying and diversifying crop rotations in such systems can increases average annual C inputs, leading to higher soil C stocks than high fallow frequency systems (e.g., West and Post, 2002; Sherrod et al., 2005; O'Dea et al., 2015). In moister environments, adding 2–3 years of perennial hay/forage crops to row crop rotations increases C inputs from fine roots and boosts SOC stocks (e.g., Dick et al., 1998).

Manure and Compost Addition

Organic matter additions such as compost and manures can increase soil C contents, both by virtue of the added C in the amendment itself and through improving soil physical attributes and nutrient availability, such that plant productivity and residue C inputs increase as well (Paustian et al., 1997). One difficulty in assessing the overall impact of organic amendments on net CO_2 removals is that the amendments typically originate from an "off-site" location and thus don't directly reflect on-farm CO_2 uptake from the atmosphere as with other practices described in this section. Hence a full life cycle assessment (LCA) approach, in which the boundaries of the assessment extend outside the farm to include the source of the amendment, is needed for an accurate accounting of C accrual and net GHG reductions. An example is given by work in California on compost addition to rangeland, in which Silver and coworkers (Ryals and Silver, 2013; Ryals et al., 2015) found substantial increases in soil C storage following modest compost additions (a one-time ∼1.3 cm thick surface dressing), in part attributed to improved infiltration and water retention, increased grass productivity and hence greater grass root and residue inputs to soil. Without counting C in the compost addition, they estimated an increase in C storage of 0.5 tC/ha (1.8 tCO_{2eq}/ha) and 3.3 tC/ha (12.1 CO_{2eq}/ha) at two contrasting rangeland sites, respectively, 3 years after compost addition. Further, where the compost was sourced from organic waste in which the business-as-usual case involved land filling and thus potential large emissions of methane, DeLonge et al. (2013) estimated an average net GHG mitigation of 23 tCO_{2eq}/ha, over the 3 year study duration, considering the full LCA including landfill waste emissions vs. compost production, transport, application, and subsequent soil improvement impacts. Considering the large amount of organic waste generated by urban centers and impacts of landfilling on GHG emissions and the potential benefits of organic amendments to soil, use of compost is a potentially attractive option that merits additional R&D to assess the full range of environmental costs and benefits.

Tillage

Tillage is used by farmers to manage crop residues and prepare a seed bed for crops, and is the main source of soil disturbance in annual croplands. Advances in tillage technology and agronomic practice have allowed farmers in recent decades to reduce tillage frequency and intensity, sometimes ceasing tillage altogether with a practice known as "no-till" (NT). The main impetus for many farmers to reduce tillage is to mitigate soil erosion. Studies have reported highly significant reductions in soil erosion under NT, often as high as 90% (Langdale et al., 1979; Ghidey and Alberts, 1998; Williams and Wuest, 2011). Tillage also acts to speed the breakdown of stable soil aggregates that can "protect" organic matter from decomposition (Six et al., 2002). Under NT, aggregation and aggregate stability is significantly enhanced, which is believed to be the main mechanism promoting increased C storage under NT (Six and Paustian, 2014). Many field studies and reviews have shown increases in soil organic carbon (SOC) following adoption of reduced till and NT, with variations due to soil texture and climate (Denef et al., 2011). For example, Ogle et al. (2005) estimated increases under NT of approximately 0.25 tC/ha/y and 0.29 tC/ha/y on sandy and non-sandy soils, respectively. In a global analysis, Six et al. (2004) reported increases in dry climates of 0.1 tC/ha/y and 0.22 tC/ha/y in humid climates. Sainju (2016) recently assessed the net impact of NT to the atmosphere, and found NT systems to have 66% lower Global Warming Potential (GWP) and 71% lower greenhouse gas intensity (GHG emissions per unit of yield) than conventionally tilled systems. However, there are instances in which no-tillage does not increase soil C relative to conventional tillage (Angers and Eriksen-Hamel, 2008), primarily in soils with already high surface C concentrations and often cooler (and wetter) areas where crop productivity and C inputs may be lower under NT, e.g., because of delayed germination (Ogle et al., 2012).

In humid and subhumid croplands, particularly for soils with moderate to poor drainage and with high C concentrations in surface layers relative to subsurface horizon, a one-time deep inversion tillage may be highly effective at promoting a significant increase in soil C stocks, over a multi-decadal period. This practice entails the burial of C-rich surface horizons to a depth of 60–80 cm depth and the transfer of low-C subsoil material to the surface. Burial of C-rich surface soil can significantly slow its decomposition (and promote deeper root penetration) while "conventional" C sequestering practices—e.g., high residue crops, cover crops, and no-till—applied to the newly exposed subsoil material, could rapidly build new C stocks in surface soil layers. For example, Alcantara et al. (2016) sampled 10 sites in Germany that had been subjected to a single deep tillage operation between 1965 and 1978 (done to alleviate compaction of subsurface layers) and found that the deep-tilled sites contained on average 42 t/ha greater SOC stocks (to 1.5 m depth) than similar soils that were not deep-tilled. Crop yields were similar on the fields that received the deep tillage treatment and on untreated fields. The implied average rate of soil C increase following the deep tillage operation was 0.96 tC/ha/y (3.5 tCO_{2eq}/ha/y), over a 45 year period.

Conversion to Perennial Grasses and Legumes

Where croplands are converted to perennial vegetation (grasses, trees), we observe both an increase in C inputs and a reduction in soil disturbance (Denef et al., 2011). Lands retired from cropland cultivation are often referred to as "set-aside." In the U.S., the Conservation Reserve Program (CRP) pays farmers to retire marginal and highly erodible croplands, with peak cumulative enrollments of just over 35 million acres (USDA FSA, 2012). The EPA National Greenhouse Gas Inventory report credits CRP land as a key contributor to agricultural soil carbon sinks in the U.S. (USEPA, 2017). A synthesis by Conant et al. (2016) estimated C stock increases of 39% after conversion of annual cropland to permanent vegetation, with an average rate of almost 0.9 tC/ha/y. Initial rates of SOC accumulation can be high under set-aside, and long-term field studies have noted that accumulations can continue for several decades, approaching levels of native SOC stock (Baer et al., 2010; Munson et al., 2012).

Rewetting Organic Soils

The soils and practices discussed to this point relate to "mineral soils," soils in which the bulk of the soil mass is made up of mineral matter, i.e., sand, silt and clay, and where organic matter normally constitutes only a few percent of the total mass. In contrast, organic soils (referred to as "histosols" in formal soil classification systems), include peat and muck-derived soils for which the total mass consists mainly of organic matter. These soils are formed under waterlogged conditions (hence very low O_2 concentrations) which strongly inhibit decomposition processes, leading to the buildup of deep layers of partially decomposed plant material. In contrast to mineral soils, organic soils are not subject to saturation in the same way—that is, organic matter can continue to accumulate, with the soil "depth" increasing, as long as the conditions inhibiting decomposition remain. When organic soils are exploited for agriculture they are typically drained, limed, and fertilized. They can be very productive for annual cropping, but conversion to agriculture gives rise to extremely high rates of CO_2 emissions, as much as 40–80 tCO_2/ha/y (as well as substantial N_2O emissions; IPCC, 2006) as the soil mass is being oxidized, which can continue as long as organic layers remain exposed to aerobic (i.e., ambient O_2

concentrations) conditions. Consequently, where organic soils can be taken out of production and hydrological conditions restored (referred to as "rewetting"), the very high CO_2 and N_2O emissions can be abated and the soil C accumulation can resume (Wilson et al., 2016). When wetland conditions are restored, CH_4 emissions can increase but, overall, restoring cultivated organic soils provides very large per hectare net emission reductions. However, the area of cultivated organic soils is very small in comparison to that of mineral soils so that that overall mitigation potential is relatively modest (Paustian et al., 2016a).

Improved Grazing Land Management

In the US, non-forested grazing lands are typically differentiated into two main categories: pastures and rangelands. Generally speaking, rangelands refer to grasslands dominated by native species, often occurring in drier environments, and where conventional management interventions are largely restricted to manipulating grazing intensity and duration. In contrast, pastures are often made up of non-indigenous and/or non-native species, are often derived from other land covers and support more intensive and more diverse management options (e.g., fertilization, irrigation, plant species introduction, as well as grazing management).

With the exception of some managed pastures, grazing lands are generally never tilled. Therefore, increasing SOC stocks under perennial grasses relies mainly on enhancing C inputs from plant roots and residues. Ranchers may achieve this by managing plant biomass removal from grazing or increasing forage production through improved species, irrigation and fertilization, yielding increases in SOC stocks of as much as 10% (Conant et al., 2016). Other analyses of grazing land BMPs (including adjusting animal stocking rates and managing plant species) found SOC stock increases of 0.07–0.3 tC/ha/y on rangelands and 0.3–1.4 tC/ha/y on managed pastures (Morgan et al., 2010). Looking at individual practices, Conant et al. (2016) estimated average positive stock changes for improved grazing (0.28 tC/ha/y), sowing legumes (0.66 tC/ha/y) and fertilization (0.57 tC/ha/y).

For improving productivity and soil condition on grazing lands, there is heightened interest in intensive grazing practices employing high animal stocking rates for short durations, from a few hours to a few days, on an area of pasture, with frequent movement of animals and relatively long "rest periods" for the vegetation between grazing events. Various terms including rotation grazing, mob grazing, or adaptive multi-paddock (AMP) grazing are used to label such management systems although terminology is far from standardized. Some studies suggest very dramatic effects from AMP grazing systems in terms of improved productivity and soil physical properties and increased soil carbon stocks. Teague et al. (2011) reported rates of soil C accumulation of about 3 tC/ha/y in AMP systems compared to heavy, continuous grazed systems and Machmuller et al. (2015) reported even higher C accrual rates of up to 8 tC/ha/y on annually cropped soils converted to intensive rotational grazing systems. However, others have questioned whether AMP/rotational grazing systems are superior to well-managed continuous grazing systems (Briske et al., 2008) and there is an ongoing debate within the scientific community. A confounding issue is that adaptive grazing systems, by definition, are dynamic in response to varying weather and other environmental conditions that affect grassland productivity. Thus it is difficult to set up traditional replicated field experiments to compare different grazing systems at the landscape scale (Teague et al., 2013). In any case, additional research and better understanding of grazing impacts on SOC stocks is needed determine optimal management conditions for increasing soil C stocks and minimizing N_2O and CH_4 emissions from livestock in these systems.

"Frontier Technologies" to Sequester Soil C

Several "non-conventional" management practices offer considerable promise for producing negative emissions but require further research to develop the necessary technology and/or better constrain estimates of costs and life-cycle emissions under large-scale deployment. Technologies that we consider here include application of biochar to cropland soils, deployment of perennial grain crops, and adoption of annual crops that have been bred to produce deeper and larger root systems for enhanced C inputs.

Biochar Additions

Biochar is a carbon-rich solid produced from biomass using a thermochemical conversion process known as pyrolysis. A range of temperatures can be used in pyrolysis, with lower temperatures/longer residence times favoring solid biochar formation and higher temperatures/shorter residence times producing a greater proportion of gases and liquid bio-oil and less char (Tripathi et al., 2016). Tradeoffs therefore arise between energy production, which generally favors maximal production of volatiles and bio-oil, and soil applications which favors maximal production of biochar. Biochar also occurs in the soils of many fire-prone ecosystems (where it is typically referred to as pyrogenic carbon), including grasslands, savannas and woodlands, and can make up as much as 35% of the total organic C in these systems (Skjemstad et al., 2002; Glaser and Amelung, 2003; Bird et al., 2015). Hence biochar/pyrogenic carbon is a natural constituent of many soils and soil function is not generally impaired (and may be enhanced) with the addition of large quantities (e.g., 100 t/ha or more) of biochar. Thus, many soils have a potential large storage capacity for added biochar.

Biochar amendments can impact soil C storage and net CO_2 removals from the atmosphere in three different ways. For biochars produced as a coproduct of biofuel pyrolysis processes, when added to soils, most of the biochar mass (80–95%) is highly resistant to microbial decay, with a mean residence time of 100s of years or more (Santos et al., 2012; Wang et al., 2016). Hence, the biochar itself represents a carbon stock that once added to soil tends to persist for a long time. Secondly, biochar additions can also interact with the native organic matter already present in soils, and either stimulate or reduce the rate of decomposition of the native soil organic matter. These interactions could involve a number of factors including impacts on soil water holding capacity and soil moisture, changes in pH or nutrient availability and direct impacts of biochar additions on microbial community activity and composition. Both positive and negative effects on native SOM decomposition following biochar addition have been found (e.g., Song et al., 2016; Wang et al., 2016), but in most cases these effects on the long-term soil C balance are small (Wang et al., 2016). Finally, biochar additions can influence plant productivity and hence C inputs to soil in the form of plant residues. Impacts of biochar addition on plant productivity can vary widely depending on the characteristics of the biochar and soil/plant characteristics. Results from meta-analyses suggest that biochar additions generally have neutral or positive effects on plant growth, with small increases on average (typically <10%) in temperate cropping systems and larger increases (e.g., 10–25%) in tropical systems, particularly on acid, nutrient-poor soils (Liu et al., 2016).

Aside from impacts on soil C storage, a number of studies suggest that biochar amendments may decrease soil N_2O emissions, which would further contribute to greenhouse gas mitigation. A recent meta-analysis by Verhoeven et al. (2017) reported average reductions of N_2O emissions of 9–12% while an earlier global assessment (Cayuela et al., 2014) suggested greater average reductions of almost 50%, compared to non-biochar amended soils. Differences in these meta-analyses are due to different selection criteria for the studies included and the weighting factors used. Regardless, there is an emerging consensus that, on average, biochar applications help to reduce N_2O emissions. The exact mechanisms involved are uncertain since many of the controls on nitrification and denitrification processes (by which N_2O emissions occur), for example pH, mineral N concentrations, soil moisture, and O_2 concentrations, can be impacted by the presence of biochar.

In summary, the main effect of biochar amendment on the GHG balance is associated with the long term storage of the biochar when added to soil. Because the production and transport of the biochar (and bioenergy coproducts) entail a number of different GHG emission sources, the actual mitigation attained (*vis a vis* the atmosphere) depends on the full biochar life cycle and emissions of the biomass feedstock production and harvesting, biochar production process, and field application. This net life cycle C offset value may vary considerably with system design and location, and better knowledge of biochar system LCAs is needed to support broad-scale deployment. One of the few global assessments of biochar amendments as a CO_2 mitigation strategy, by Woolf et al. (2010), suggested a climate change mitigation potential of 1.8 Gt C per year. Due to the complexity of biochar-bioenergy-agricultural systems, the viability of large-scale biochar production and soil application will be spatially variable and process dependent. One cost-benefit analysis found that (without a C price), the net present value of biochar application to soils was positive in a sub-Saharan African context but negative in a Northwestern European context, due to a combination of greater production costs and more modest yield benefits in the latter scenario (Dickinson et al., 2015).

Deployment of Perennial Grain Crops

There have been breeding efforts underway over the past three decades to develop cereal grains (and other annual crops) with a perennial growth habit. The perennial grasses selected for breeding stocks, such as intermediate wheatgrass, are notable in having deep and extensive root systems with a higher proportion of dry matter allocation belowground than conventional annual crops. Hence C inputs to soil are much greater than annual crops and thus will support greater SOC stocks. Perennial crops would also greatly reduce the need for tillage and its negative effects on SOC stocks and soil erosion. Larger and deeper root systems could also reduce nitrate leaching losses to waterways and possibly N_2O emissions to the atmosphere (Glover et al., 2010; Pimentel et al., 2012; Abalos et al., 2016; Crews and Rumsey, 2017).

Because of the relatively recent focus on developing agronomically-viable perennial grains, there are few long-term experiments that are of sufficient duration to document increases in SOC from adoption of perennial grain crops. Culman et al. (2013) found that intermediate wheatgrass increased the amount of labile soil C after 4 years compared to annual winter wheat in SW Michigan, but there was no significant increase in total SOC. However, results from other long-term studies and chronosequences involving perennial grass (e.g., hay, pasture) systems probably provide a reasonable proxy for what would be expected for the longer term response of soils under perennial grains. Some rates of SOC change observed following conversion of annual cropland to a variety of managed perennial grasslands systems are given in **Table 2**.

From **Table 2** it is reasonable to assume that perennial grains could sequester, on average, about 1 tC/ha/y (about 3.6 tCO_2/ha/y) over a number of years, on land converted from continuous annual crop production in the central US grain belt.

At present there are several barriers to adoption of perennial grains on significant areas of land currently allocated to conventional annual crops. Chief among these barriers are low yields and hence questionable economic viability if brought to scale. Yields for intermediate wheatgrass (presently the most commercially viable perennial grain) are typically <1,000 kg/ha, which is 5–10 times less than annual wheat yields at the same locations (Culman et al., 2016). Between-year variability is also high—in a 4 year study in Southwestern Michigan, Culman et al. (2016) reported average yields ranging from 119 kg/ha/y

TABLE 2 | Observed rates of SOC change under various managed perennial systems.

Cropping system	Mean ΔSOC (t C/ha/y)	Range (t C/ha/y)	Source
Restored prairie	0.77	0.62–0.91	Tilman et al., 2006
Hayed grassland	0.47	None given	Culman et al., 2010
Conversion of annual crops to pasture	0.87¶		Conant et al., 2016
Meta-analysis of perennial bioenergy crops			
Switchgrass	3.10	−5.4 to 13.0	Qin et al., 2016
Miscanthus	1.97	−4.7 to 8.2	
Poplar	0.56	−3.4 to 6.0	

¶*Mean value from a global meta-analysis of 93 studies.*
Results are annualized rates of change from multi-year studies.

(in 2012) to 1,493 kg/ha/y (in 2011), with a mean over the 4 years of 485 kg/ha/y. In a 4-year trial of more than 75 lines of perennial wheatgrass in Australia, several had first-year yields that approached a profitability threshold (without considering any value for potential carbon mitigation benefits), but yields for the following three seasons declined to negligible levels (Larkin et al., 2014). Other issues include problems with grain shattering, lodging, small seed size, and sparse knowledge on optimal agronomics. Such challenges are not unexpected given the few years of active breeding efforts so far, and thus further selection, breeding and field experimentation are likely to improve yields and agronomics[3]. However, there are likely persistent tradeoffs involving resource allocation by perennial plants between dry matter belowground to roots and aboveground to grain (Smaje, 2015; Vico et al., 2016) that will set limits on grain production capacity.

There are also clear tradeoffs in the case of replacing higher yielding annual crops with lower yielding perennials in terms of land use impacts at regional to global scales. If food/feed supply is decreased as a result of adopting lower yielding perennials, there would be pressure to replace that lost production through conversion of new land to agriculture elsewhere, leading to potentially large increases in GHG emissions due to land use conversion. This phenomenon, termed indirect land use change, has been extensively analyzed in the case of substituting energy crops for food crops (e.g., Searchinger et al., 2008; Creutzig et al., 2015) and tradeoffs between crop yields and land use choices are central to arguments underpinning sustainable land use intensification (e.g., Foley et al., 2011; Garnett et al., 2013). However, the potential for mixed grain and forage production and targeting the use of marginal lands that are poorly suited for annual grain production offer opportunities for successful initial commercialization of perennial grain crops (Bell et al., 2008; Culman et al., 2016). In summary, perennial grains show promise for broadening the array of ecosystem services provided by agriculture, including building SOC, but considerable work remains to produce cultivars with reliable regrowth and adequate grain yields, among other important agronomic traits (Cox et al., 2010; Crews et al., 2016).

Annual Crops Bred to Develop Deeper and Larger Root Systems

Another future option, somewhat similar to the deployment of perennial cereals, would be to modify, through targeted breeding and plant selection, existing annual crop plants to produce more roots, deeper in the soil profile. Thus, while the crops would still have an annual life cycle, both C inputs to soil would be increased and deeper root distributions, where decomposition rates are slower compared to surface horizons, would act to increase soil C storage. In a concept paper, Kell (2012) laid out a rationale for the potential to direct plant breeding efforts toward developing varieties for our major grain crops, e.g., corn, sorghum, wheat, and barley, that would have much greater allocation of C to roots and also deeper root distribution compared to current annual crop varieties.

In an analysis to support a new program launched by DOE's ARPA-E, Paustian et al. (2016b) performed a "bounding analysis" to estimate what level of soil C increase and total greenhouse gas mitigation (including N_2O emissions) might be possible based on specifying feasible increases in total root mass and changing root depth distributions toward those found in perennial grasses. They estimated that widespread adoption of annual crop phenotypes designed to have deeper and larger root systems could yield soil C stock increases of 0.5 Gt CO_2/ha/y on current US cropland.

ESTIMATES OF THE BIOPHYSICAL POTENTIAL FOR CO_2 REMOVAL AND SEQUESTRATION IN SOILS

As described in the preceding section, there are a wide variety of management practices that can be adopted on agricultural lands to remove CO_2 from the atmosphere and convert it into soil organic matter. The question then is "how much?"—how much carbon can actually be added to and maintained in soils and is it large enough to matter?

Over the past 20 years there have been several estimates of the soil C sequestration potential globally and for the US. In nearly all cases these represent the *biophysical potential* for managed cropland and/or grassland systems to store additional carbon assuming widespread (near complete) adoption of the sequestering practices. As such, these represent upper-bound estimates of the C sequestration potential. Economic or policy-related constraints are generally not considered as they require a detailed coupled ecosystem and economic modeling approach. In terms of methods, most estimates, particularly at global scale, are based on highly aggregated data on total area by land-use type, stratified into broadly defined climate types, and then applying estimates of representative per ha soil C sequestration rates for different management practices or suites of practices, based on measurements from long-term field experiments.

Despite somewhat different scope (land types included) and assumptions (practices considered), there is fairly close alignment among global estimates (**Figure 1**), suggesting a technical soil C sequestration potential of 2–5 Gt CO_2 per year, for what were characterized in the section above as existing best conservation management practices. Estimates toward the lower end of this range consider either less land area (e.g., cropland only) and/or a more restricted set of practices. It is not surprising that these various estimates are in reasonably close alignment since the two main determining factors, land area by land use type and observed rates of soil C sequestration from long-term field trials, are fairly tightly constrained. Thus, there seems to be good support for an estimate of as much as 4–5 Gt CO_2 per year for widespread adoption of a broad suite of BMPs for soil C sequestration on global grassland and cropland. These rates of C storage could be sustained for a limited time period, on the order

[3] Glover et al. (2010) estimated that commercially viable perennial grains could be available by 2030.

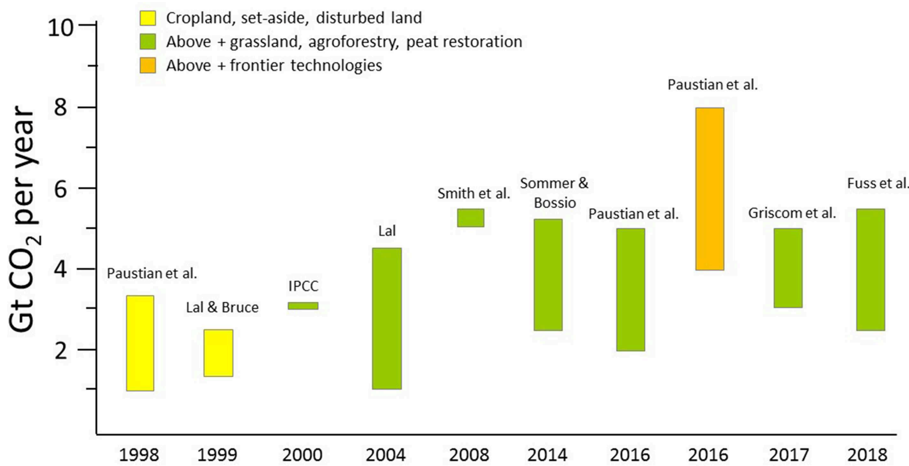

FIGURE 1 | Published estimates of global biophysical soil carbon sequestration potential, assuming (near) full adoption of C sequestering practices. Estimates vary in terms of land use types and mix of practices included; e.g., the pre-2000 estimates (Paustian et al., 1998; Lal and Bruce, 1999) focused on cropland and set-aside to grassland of marginal crop land, while later estimates (Lal, 2004; Smith et al., 2008; Sommer and Bossio, 2014; Paustian et al., 2016a; Griscom et al., 2017; Fuss et al., 2018) included a fuller range of options on all agricultural lands. The IPCC (2000) estimate provided a point estimate whereas other publications provide a range of per annum rates. The two estimates shown for Paustian et al. (2016a) denote with and without frontier technologies.

of 2–3 decades before decreasing, as soil C levels approach a new equilibrium.

The estimate by Paustian et al. (2016a) that goes as high as 8 Gt CO_2 per year, includes ∼3 Gt CO_2/y from what we've referred to as "frontier technologies," in this case biochar amendments and high root C input crop phenotypes, in addition to the conventional conservation technologies included in other global estimates. However, estimates of technical potentials for these frontier technologies are much more uncertain, either because empirical data on their performance in the field (e.g., in long-term field studies) is much scarcer, or in the case of novel crop types (e.g., perennial grains, enhanced root phenotype annual crops), the technologies themselves are still in an early developmental stage.

In conjunction with the negotiations for the Paris climate accords, the French government announced an initiative dubbed "4 per mille" which advocates for a massive effort to increase global soil C stocks as a core greenhouse gas mitigation strategy. As articulated by INRA, the French National Institute for Agricultural Research (INRA, 2017), if global soil C stocks in the top 40 cm (860 GtC) could be increased on average by 0.4% (i.e., 4 per mille) per year that is equivalent to about 3.4 GtC/y or 12.6 $GtCO_2$/y. That level of net CO_2 uptake would offset most of the current annual increase in atmospheric CO_2 (15.8 $GtCO_2$/y), assuming that the current ocean and terrestrial C sinks remained intact. There is considerable debate about whether this level of soil C sequestration is indeed possible, and whether all soils or mainly agricultural soils should be targeted (e.g., Chambers et al., 2016; Minasny et al., 2017). In any case, as an aspirational goal, the 4 per mille concept has certainly spurred debate and "raised the profile" of soils as a potentially key mitigation strategy.

As points of comparison, current global GHG emissions are about 40 $GtCO_{2e}$/year (with about 83% of that from fossil fuel combustion), and meeting the goals of the Paris agreement may require negative emissions of about 15 Gt CO_2/year by the end of the century (NASEM, 2019).

CONCLUDING REMARKS

There is strong scientific evidence for agricultural soils to act as a significant carbon (C) sink over the next several decades and thereby to contribute to meeting the objectives of the Paris Climate Accord. There are a wide variety of C sequestering practices that can be applied and the best solutions vary according to climate, soil, and farming practices. Many practices (e.g., improvements in crop rotations, use of cover crops, tillage changes, N fertilizer management) are already developed and their efficacy is relatively well-understood. Wide-scale adoption of such measures could take place quite rapidly. Other potential practices, requiring development of new crop varieties and broad-scale use of soil amendments such as biochar, require additional research and development to overcome technological hurdles and/or improve economic feasibility.

This suggests a "two-stage" strategy. Strong policy could be enacted immediately to begin an international effort to increase soil carbon sequestration, based on existing technologies. Key ingredients are efficient policies that incentivize farmers to adopt improved (C sequestering) practices, by compensating them for additional costs and/or added risk. Expanded education and outreach can also help to overcome knowledge or "know-how" barriers. Meanwhile, continued R&D, with increased investments could be devoted to further developing new crop varieties, both perennial grains (and "perennialization" of other crops such as oil seeds) and breeding for annual crops with larger and deeper root systems. This could lead to viability of these new crops for use by about 2030 and beyond, when the need for negative emission strategies will be growing.

Implementation of these policies will require a robust, scientifically-sound measurement, reporting, and verification (MRV) system to track that policy goals are being met and that claimed increases in soil C stocks are real. Much of the infrastructure for an effective MRV system for soil C sequestration could be assembled relatively quickly and with modest research and development investments (NASEM, 2019; Paustian et al., 2019). Existing ground-based data from long-term field experiments (e.g., Harden et al., 2018) together with national networks for on-farm soil monitoring (van Wesemael et al., 2011) can support the continued improvement and deployment of process-based predictive models. Expanded use of remote sensing can help to monitor management practices (e.g., Hively et al., 2018) and constrain local-scale estimates of CO_2 assimilation and C input to soils by crops (e.g., Guan et al., 2017). This extensive and broad-based melding of ground-based experiments and monitoring, dynamic predictive models, remote sensing and farmer-based knowledge of management practices can form the basis for quantification tools that can inform policy and program implementation, at field- (Paustian et al., 2018) to national-scales (Ogle et al., 2014).

In summary—by leveraging existing scientific knowledge and infrastructure, together with modest investment to further advance the knowledge base and develop new technologies, many countries could move to implement negative emission strategies in the agricultural sector and at the same time improve the health and resilience of their soils. This would stimulate and encourage global-scale initiatives (e.g., Schleussner et al., 2016; INRA, 2017), to help achieve the goal of limiting average global temperature increases to $<2^{\circ}$C.

AUTHOR CONTRIBUTIONS

KP and EL formulated the idea for the paper. KP led the development and writing of the paper. JK, EM, and AS helped compile and analyze data and contributed to the writing of the paper.

REFERENCES

Abalos, D., Brown, S. E., Vanderzaag, A. C., Gordon, R. J., Dunfield, K. E., and Wagner-Riddle, C. (2016). Micrometeorological measurements over 3 years reveal differences in N_2O emissions between annual and perennial crops. *Glob. Change Biol.* 22, 1244–1255. doi: 10.1111/gcb.13137

Alcantara, V., Don, A., Well, R., and Nieder, R. (2016). Deep ploughing increases agricultural soil organic matter stocks. *Glob. Change Biol.* 22, 2939–2956. doi: 10.1111/gcb.13289

Angers, D., and Eriksen-Hamel, N. S. (2008). Full-inversion tillage and organic carbon distribution in soil profiles: a meta-analysis. *Soil Sci. Soc. Am. J.* 72, 1370–1374. doi: 10.2136/sssaj2007.0342

Baer, S. G., Meyer, C. K., Bach, E. M., Klopf, R. P., and Six, J. (2010). Contrasting ecosystem recovery on two soil textures: implications for carbon mitigation and grassland conservation. *Ecosphere* 1, 1–22. doi: 10.1890/ES10-00004.1

Batjes, N. H. (1996). Total carbon and nitrogen in the soils of the world. *Euro. J. Soil Sci.* 47, 151–163. doi: 10.1111/j.1365-2389.1996.tb01386.x

Bell, L. W., Byne, F., Ewing, M. A., and Wade, L. J. (2008). A preliminary whole-farm economic analysis of perennial wheat in an Australian dryland farming system. *Agric. Syst.* 96, 166–174. doi: 10.1016/j.agsy.2007.07.007

Bird, M. I., Wynn, J. G., Saiz, G., Wurster, C. M., and McBeath, A. (2015). The pyogenic carbon cycle. *Ann. Rev. Earth Planet. Sci.* 43, 273–298. doi: 10.1146/annurev-earth-060614-105038

Briske, D. D., Derner, J. D., Brown, J. R., Fuhlendorf, S. D., Teague, W. R., Havstad, K. M., et al. (2008). Rotational grazing on rangelands: reconciliation of perception and experimental evidence. *Rangeland Ecol. Manag.* 61, 3–17. doi: 10.2111/06-159R.1

CAST (2004). *Climate Change and Greenhouse Gas Mitigation: Challenges and Opportunities for Agriculture*. Task Force Report, No. 141. Ames, IA: Council for Agricultural Science and Technology.

Cayuela, M. L., van Zwieten, L., Singh, B. P., Jeffery, S., Roig, A., and Sanchez-Monedero, M. A. (2014). Biochar's role in mitigating soil nitrous oxide emissions: a review and meta-analysis. *Agric. Ecosys. Environ.* 191, 5–16. doi: 10.1016/j.agee.2013.10.009

Chambers, A., Lal, R., and Paustian, K. (2016). Soil carbon sequestration potential of US croplands and grasslands: implementing the 4 per thousand initiative. *J. Soil Water Conserv.* 71, 68A−74A. doi: 10.2489/jswc.71.3.68A

Conant, R. T., Cerri, C. E. P., Osborne, B. B., and Paustian, K. (2016). Grassland management impacts on soil carbon stocks: a new synthesis. *Ecol. Appl.* 27, 662–668. doi: 10.1002/eap.1473

Cox, T. S., Van Tassel, D. L., Cox, C. M., and Dehaan, L. R. (2010). Progress in breeding perennial grains. *Crop Pasture Sci.* 61, 513–521. doi: 10.1071/CP09201

Creutzig, F., Ravindranath, N. H., Berndes, G., Bolwig, S., Bright, R., Cherubini, F., et al. (2015). Bioenergy and climate change mitigation: an assessment. *Glob. Change Biol. Bioenergy* 7, 916–944. doi: 10.1111/gcbb.12205

Crews, T. E., Blesh, J., Culman, S. W., Hayes, R. C., SteenJensen, E., Mack, M. C., et al. (2016). Going where no grains have gone before: from early to mid-succession. *Agric. Ecosyst. Environ.* 223, 223–238. doi: 10.1016/j.agee.2016.03.012

Crews, T. E., and Rumsey, B. E. (2017). What agriculture can learn from native ecosystems in building soil organic matter: a review. *Sustainability* 9, 1–18. doi: 10.3390/su9040578

Culman, S.W., Pugliese, J., DeHaan, L., Crews, T., Sulc, R. M., Ryan, M., et al. (2016). "Can the perennial grain crop Kernza yield both forage and grain?" in *American Society of Agronomy meeting* (Phoenix, AZ). Available online at: https://scisoc.confex.com/scisoc/2016am/videogateway.cgi/id/27312?recordingid=27312

Culman, S. W., DuPont, S. T., Glover, J. D., Buckley, D. H., Fick, G. W., Ferris, H., et al. (2010). Long-term impacts of high-input annual cropping and unfertilized perennial grass production on soil properties and belowground food webs in Kansas, USA. *Agric. Ecosys. Environ.* 137, 13–24. doi: 10.1016/j.agee.2009.11.008

Culman, S. W., Snapp, S. S., Ollenburger, M., Basso, B., and DeHaan, L.R. (2013). Soil and water quality rapidly responds to the perennial grain kernza wheatgrass. *Agron. J.* 105, 735–744. doi: 10.2134/agronj2012.0273

Davidson, E. A., and Ackerman, I. L. (1993). Changes in soil carbon inventories following cultivation of previously untilled soils. *Biogeochemistry* 20, 161–164. doi: 10.1007/BF00000786

DeLonge, M. S., Ryals, R., and Silver, W. L. (2013). A lifecycle model to evaluate carbon sequestration potential and greenhouse gas dynamics of managed grasslands. *Ecosystems* 16, 962–979. doi: 10.1007/s10021-013-9660-5

Denef, K., Archibeque, S., and Paustian, K. (2011). *Greenhouse Gas Emissions From U.S. Agriculture and Forestry: A Review of Emission Sources, Controlling Factors, and Mitigation Potential*. Interim report from Colorado State University to USDA under Contract #GS23F8182H. Available online at: https://www.usda.gov/oce/climate_change/techguide/Denef_et_al_2011_Review_of_reviews_v1.0.pdf (accessed September 27, 2019).

Dick, W. A., Blevins, R. L., Frye, W. W., Peters, S. E., Christenson, D. R., Pierce, F. J., et al. (1998). Impacts of agricultural management practices on C sequestration in forest-derived soils of the eastern Corn Belt. *Soil Tillage Res.* 47, 235–244. doi: 10.1016/S0167-1987(98)00112-3

Dickinson, D., Balduccio, L., Buysse, J., Ronsse, F., van Huylenbroeck, G., and Prins, W. (2015). Cost-benefit analysis of using biochar to improve cereals agriculture. *GCB Bioenergy* 7, 850–864. doi: 10.1111/gcbb.12180

Foley, J. A., Ramankutty, N., Brauman, K. A., Cassidy, E. S., Gerber, J. S., Johnston, M., et al. (2011). Solutions for a cultivated planet. *Nature* 478, 337–342. doi: 10.1038/nature10452

Fuss, S., Lamb, W. F., Callaghan, M. W., Hilaire, J., Creutzig, F., Amann, T., et al. (2018). Negative emissions—part 2: costs, potentials and side effects. *Environ. Res. Lett.* 13:063002. doi: 10.1088/1748-9326/aabf9f

Garnett, T., Appleby, M. C., Balmford, A., Bateman, I. J., Benton, T. G., Bloomer, P., et al. (2013). Sustainable intensification in agriculture: premises and policies. *Science* 341, 33–34. doi: 10.1126/science.1234485

Ghidey, F., and Alberts, E. E. (1998). Runoff and soil losses as affected by corn and soybean tillage systems. *J. Soil Water Conserv.* 53, 64–70.

Glaser, B., and Amelung, W. (2003). Pyrogenic carbon in native grassland soils along a climosequence in North America. *Glob. Biogeochem. Cycles* 17, 1–8. doi: 10.1029/2002GB002019

Glover, J. D., Reganold, J. P., Bell, L. W., Borevitz, J., Brummer, C., Buckler, E., et al. (2010). Increased food and ecosystem security via perennial grains. *Science* 328, 1638–1639. doi: 10.1126/science.1188761

Griscom, B. W., Adams, J., Ellis, P. W., Houghton, R. A., Lomax, G., Miteva, D. A., et al. (2017). Natural climate solutions. *Proc. Natl. Acad. Sci. U.S.A.* 114, 11645–11650. doi: 10.1073/pnas.1710465114

Guan, K., Wu, J., Kimball, J., Anderson, M., Frolking, S., Li, B., et al. (2017). The shared and unique values of optical, fluorescence, thermal and microwave satellite data for estimating large-scale crop yields. *Remote Sens. Environ*. 199, 333–349. doi: 10.1016/j.rse.2017.06.043

Harden, J. W., Hugelius, G., Ahlström, A., Blankinship, J. C., Bond-Lamberty, B., Lawrence, C. R. et al. (2018). Networking our science to characterize the state, vulnerabilities, and management opportunities of soil organic matter. *Glob Change Biol.* 24, e705–e718. doi: 10.1111/gcb.13896

Hively, W., Lamb, B., Daughtry, C., Shermeyer, J., McCarty, G., and Quemada, M. (2018). mapping crop residue and tillage intensity using worldview-3 satellite shortwave infrared residue indices. *Remote Sens.* 10:1657. doi: 10.3390/rs10101657

INRA (2017). *4 per 1000. Carbon Sequestration in Soils*. Available online at: http://www.inra.fr/en/Public/Global-warming/All-magazines/Four-parts-per-1000-carbon-storage-in-the-soil#

IPCC (2000). *Land Use, Land-Use Change and Forestry*. Special Report. Cambridge University Press. p. 377.

IPCC (2006). "2006 IPCC Guidelines for National Greenhouse Gas Inventories," in *Agriculture, Forestry and Other Land Use*, Vol. 4. eds H. S. Eggleston, L. Buendia, K. Miwa, T. Ngara, and K. Tanabe (Published IGES, Japan). (Prepared by the National Greenhouse Gas Inventories Programme).

Kell, D. (2012). Large-scale sequestration of atmospheric carbon via plant roots in natural and agricultural ecosystems: why and how. *Phil. Trans. R. Soc. B* 367, 1589–1597 doi: 10.1098/rstb.2011.0244

Lal, R. (2004). Soil carbon sequestration impacts on global climate change and food security. *Science* 304, 1623–1627. doi: 10.1126/science.1097396

Lal, R., and Bruce, J. P. (1999). The potential of world cropland soils to sequester C and mitigate the greenhouse effect. *Environ. Sci. Policy* 2, 177–185. doi: 10.1016/S1462-9011(99)00012-X

Langdale, G., Barnett, A., Leonard, R., and Fleming, W. (1979). Reduction of soil erosion by the no-till system in the Southern Piedmont. *Transact. ASAE* 22, 82–86. doi: 10.13031/2013.34970

Larkin, P. J., Newell, M. T., Hayes, R. C., Aktar, J., Norton, M. R., Moroni, S. J., et al. (2014). Progress in developing perennial wheats for grain and grazing. *Crop Pasture Sci.* 65, 1147–1164. doi: 10.1071/CP13330

Liu, S., Zhang, Y., Zong, Y., Hu, Z., Wu, S., Zhou, J., et al. (2016). Response of soil carbon dioxide fluxes, soil organic carbon and microbial biomass carbon to biochar amendment: A meta-analysis. *GCB Bioenergy* 8, 392–406. doi: 10.1111/gcbb.12265

Machmuller, M. B., Kramer, M. G., Cyle, T. K., Hill, N., Hancock, D., et al. (2015). Emerging land use practices rapidly increase soil organic matter. *Nat. Commun.* 6:6995. doi: 10.1038/ncomms7995

Minasny, B., Malone, B. P., McBratney, A. B., Angers, D. A., Arrouays, D., Chambers, A., et al. (2017). Soil carbon 4 per mille. *Geoderma* 292, 59–86. doi: 10.1016/j.geoderma.2017.01.002

Morgan, J. A., Follett, R. F., Allen, L. H. Jr., Del Grosso, S., Derner, J. D., Dijkstra, F., et al. (2010). Carbon sequestration in agricultural lands of the United States. *J. Soil Water Conserv.* 65, 6–13A. doi: 10.2489/jswc.65.1.6A

Munson, S. M., Lauenroth, W. K., and Burke, I. C. (2012). Soil carbon and nitrogen recovery on semiarid Conservation Reserve Program lands. *J. Arid Environ.* 79, 25–31. doi: 10.1016/j.jaridenv.2011.11.027

NASEM (2019). *Negative Emissions Technologies and Reliable Sequestration: A Research Agenda. National Academies of Sciences, Engineering, and Medicine*; Washington, DC: The National Academies Press.

O'Dea, J. K., Jones, C. A., Zabinski, C. A., Miller, P. R., and Keren, I. N. (2015). Legume, cropping intensity, and N-fertilization effects on soil attributes and processes from an eight-year-old semiarid wheat system. *Nutr. Cycl. Agroecosyst.* 102, 179–194. doi: 10.1007/s10705-015-9687-4

Ogle, S. M., Breidt, F. J., and Paustian, K. (2005). Agricultural management impacts on soil organic carbon storage under moist and dry climatic conditions of temperate and tropical regions. *Biogeochemistry* 72, 87–121. doi: 10.1007/s10533-004-0360-2

Ogle, S. M., Olander, L., Wollenberg, L., Rosenstock, T., Tubiello, F., Paustian, K., et al. (2014). Reducing greenhouse gas emissions and adapting agricultural management for climate change in developing countries: providing the basis for action. *Glob. Change Biol.* 20, 1–6. doi: 10.1111/gcb.12361

Ogle, S. M., Swan, A., and Paustian, K. (2012). No-till management impacts on crop productivity, carbon inputs and soil carbon sequestration. *Agri. Ecosys. Environ.* 149, 37–49. doi: 10.1016/j.agee.2011.12.010

Paustian, K. (2014). Carbon sequestration in soil and vegetation and greenhouse gas emissions reduction. *Glob. Environ. Change* 1, 399–406. doi: 10.1007/978-94-007-5784-4_10

Paustian, K., Andren, O., Janzen, H. H., Lal, R., Smith, P., Tian, G., et al. (1997). Agricultural soils as a sink to mitigate CO_2 emissions. *Soil Use Manag.* 13, 230–244. doi: 10.1111/j.1475-2743.1997.tb00594.x

Paustian, K., Campbell, N., Dorich, C., Marx, E., and Swan, A. (2016b). *Assessment of Potential Greenhouse Gas Mitigation From Changes to Crop Root Mass and Architecture*. Final report to ARPA-E, 34. Available online at: https://arpa-e.energy.gov/sites/default/files/documents/files/Revised_Final_Report_to_ARPA_Bounding_Analysis.pdf (accessed August 22, 2019).

Paustian, K., Cole, C. V., Sauerbeck, D., and Sampson, N. (1998). CO_2 mitigation by agriculture: an overview. *Clim. Change* 40, 135–162. doi: 10.1023/A:1005347017157

Paustian, K., Collier, S., Baldock, J., Burgess, R., Creque, J., DeLonge, M., et al. (2019). Quantifying carbon for agricultural soil management: from the current status toward a global soil information system. *Carbon Manag* 21. doi: 10.1080/17583004.2019.1633231

Paustian, K., Easter, M., Brown, K., Chambers, A., Eve, M., Huber, A., et al. (2018). "Field- and farm-scale assessment of soil greenhouse gas mitigation using COMET-Farm^{TM}," in *Precision Conservation: Geospatial Techniques for Agricultural and Natural Resources Conservation*, eds J. A. Delgado, G. F. Sassenrath, and T. Mueller Agronomy Monograph 59 (Madison, WI: ASA/CSSA/SSSA), p. 341–359.

Paustian, K., Lehmann, J., Ogle, S., Reay, D., Robertson, G. P., Smith, P. (2016a). Climate-smart soils. *Nature* 532, 49–57. doi: 10.1038/nature17174

Pimentel, D., Cerasale, D., Stanley, R. C., Perlman, R., Newman, E. M., Brent, L. C., et al. (2012). Annual vs. perennial grain production. *Agric. Ecosyst. Environ.* 161, 1–9. doi: 10.1016/j.agee.2012.05.025

Poeplau, C., and Don, A. (2015). Carbon sequestration in agricultural soils via cultivation of cover crops–a meta-analysis. *Agric. Ecosyst. Environ.* 200, 33–41. doi: 10.1016/j.agee.2014.10.024

Qin, Z., Dunn, J. B., Kwon, H., Mueller, S., and Wander, M. M. (2016). Soil carbon sequestration and land use change associated with biofuel production: empirical evidence. *GCB Bioenergy* 8, 66–80. doi: 10.1111/gcbb.12237

Ryals, R., Hartman, M. D., Parton, W. J., DeLonge, M. S., and Silver, W. L. (2015). Long-term climate change mitigation potential with organic matter management on grasslands. *Ecol. Appl.* 25, 531–545. doi: 10.1890/13-2126.1

Ryals, R., and Silver, W. L. (2013). Effects of organic matter amendments on net primary productivity and greenhouse gas emissions in annual grasslands. *Ecol. Appl.* 23, 46–59. doi: 10.1890/12-0620.1

Sainju, U. M. (2016). A global meta-analysis on the impact of management practices on net global warming potential and greenhouse gas intensity from cropland soils. *PLoS ONE* 11:e0148527. doi: 10.1371/journal.pone.0148527

Santos, F., Torn, M. S., and Bird, J. A. (2012). Biological degradation of pyrogenic organic matter in temperate forest soils. *Soil Biol. Biochem.* 51, 115–124. doi: 10.1016/j.soilbio.2012.04.005

Schleussner, C. F., Rogelj, J., Schaeffer, M., Lissner, T., Licker, R., Fischer, E. M., et al. (2016). Science and policy characteristics of the Paris Agreement temperature goal. *Nat. Climate Change* 6, 827–835. doi: 10.1038/nclimate3096

Schmidt, M. W. I., Torn, M. S., Abiven, S., Dittmar, T., Guggenberger, G., and Janssens, I. A., et al. (2011). Persistence of soil organic matter as an ecosystem property. *Nature* 478, 49–56. doi: 10.1038/nature10386

Searchinger, T., Heimlich, R., Houghton, R. A., Dong, F. X., Elobeid, A., Fabiosa, J., et al. (2008). Use of US croplands for biofuels increases greenhouse gases through emissions from land-use change. *Science* 319, 1238–1240. doi: 10.1126/science.1151861

Sherrod, L. A., Peterson, G. A., Westfall, D. G., Ahuja, L. R. (2005). Soil organic carbon pools after 12 years in no-till dryland agroecosystems. *Soil Sci. Soc. Am. J.* 69, 1600–1608. doi: 10.2136/sssaj2003.0266

Six, J., Conant, R. T., Paul, E. A., and Paustian, K. (2002). Stabilization mechanisms of soil organic matter: implications for C-saturation of soil. *Plant Soil* 241, 155–176. doi: 10.1023/A:1016125726789

Six, J., Ogle, S. M., Breidt, F. J., Conant, R. T., Mosier, A. R., and Paustian, K. (2004). The potential to mitigate global warming with no-tillage management is only realized when practised in the long term. *Glob. Change Biol.* 10, 155–160. doi: 10.1111/j.1529-8817.2003.00730.x

Six, J., and Paustian, K. (2014). Aggregate-associated soil organic matter as an ecosystem property and a measurement tool. *Soil Biol. Biochem.* 68, A4–A9. doi: 10.1016/j.soilbio.2013.06.014

Skjemstad, J. O., Reicosky, D. C., Wilts, A. R., and McGowan, J. A. (2002). Charcoal carbon in U.S. agricultural soils. *Soil Sci. Soc. Am. J.* 66:1249. doi: 10.2136/sssaj2002.1249

Smaje, C. (2015). The strong perennial vision: a critical review. *Agroecol. Sustain. Food Syst.* 39, 500–515. doi: 10.1080/21683565.2015.1007200

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., et al. (2008). Greenhouse gas mitigation in agriculture. *Philos. Transact. R. Soc. B* 363, 789–813. doi: 10.1098/rstb.2007.2184

Sommer, R., and Bossio, D. (2014). Dynamics and climate change mitigation potential of soil organic carbon sequestration. J. *Environ. Manag.* 144, 83–87. doi: 10.1016/j.jenvman.2014.05.017

Song, X., Pan, G., Zhang, C., Zhang, L., Wang, H. (2016). Effects of biochar application on fluxes of three biogenic greenhouse gases: a meta-analysis. *Ecosyst. Health Sustain.* 2:e01202. doi: 10.1002/ehs2.1202

Teague, R., Provenza, F., Kreuter, U., Steffens, T., and Barnes, M. (2013). Multi-paddock grazing on rangelands: why the perceptual dichotomy between research results and rancher experience? *J. Environ. Manag.* 128, 699–717. doi: 10.1016/j.jenvman.2013.05.064

Teague, W. R., Dowhower, S. L., Baker, S. A., Haile, N., DeLaune, P. B., and Conover, D. M. (2011). Grazing management impacts on vegetation, soil biota and soil chemical, physical and hydrological properties in tall grass prairie. *Agric. Ecosyst. Environ.* 141, 310–322. doi: 10.1016/j.agee.2011.03.009

Tilman, D., Hill, J., and Lehman, C. (2006). Carbon-negative biofuels from low-input high-diversity grassland biomass. *Science* 314, 1598–600. doi: 10.1126/science.1133306

Tripathi, M., Sahu, J. N., and Ganesan, P. (2016). Effect of process parameters on production of biochar from biomass waste through pyolysis: a review. *Renew. Sust. Energy Rev.* 55, 467–481. doi: 10.1016/j.rser.2015.10.122

USDA FSA (2012). Conservation *Reserve Program Annual Summary and Enrollment Statistics*. U.S. Deparment of Agriculture; Farm Service Agency. Available online at: https://www.fsa.usda.gov/Assets/USDA-FSA-Public/usdafiles/Conservation/PDF/summary12.pdf (accessed September 27, 2019).

USEPA (2017). *Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2015*. EPA Report 430-P-17-001. Available online at: https://www.epa.gov/sites/production/files/2017-02/documents/2017_complete_report.pdf (accessed September 27, 2019).

van Groenigen, J. W., van Kessel, C., Hungate, B. A., Oenema, O., Powlson, D. S., and van Groenigen, K. J. (2017). Sequestering soil organic carbon: a nitrogen dilemma. *Environ. Sci. Technol.* 51, 4738–4739. doi: 10.1021/acs.est.7b01427

van Oost, K., Quine, T. A., Govers, G., De Gryze, S., Six, J., Harden, J. W., et al. (2007). The impact of agricultural soil erosion on the global carbon cycle. *Science* 318, 626–629. doi: 10.1126/science.1145724

van Wesemael, B., K., Paustian, O., Andrén, C. E. P., Cerri, M., Dodd, J., et al. (2011). How can soil monitoring networks be used to improve predictions of organic carbon pool dynamics and CO_2 fluxes in agricultural soils? *Plant Soil* 338, 247–259 doi: 10.1007/s11104-010-0567-z

Verhoeven, E., Pereira, E., Decock, C., Suddick, E., Angst, T., and Six, J. (2017). Toward a better assessment of biochar–nitrous oxide mitigation potential at the field scale. *J. Environ. Qual.* 46:237. doi: 10.2134/jeq2016.10.0396

Vico, G., Manzoni, S., Nkurunziza, L., Murphy, K., and Weih, M. (2016). Trade-offs between seed output and life span–a quantitative comparison of traits between annual and perennial congeneric species. *New Phytol.* 209, 104–114. doi: 10.1111/nph.13574

Wang, J. Y., Xiong, Z. Q., and Kuzyakov, Y. (2016). Biochar stability in soil: meta-analysis of decomposition and priming effects. *Glob. Change Biol. Bioenergy* 8, 512–523. doi: 10.1111/gcbb.12266

West, T. O., and Post, W. M. (2002). Soil organic carbon sequestration rates by tillage and crop rotation: a global data analysis. *Soil Sci. Soc. Am. J.* 66, 1930–1946. doi: 10.2136/sssaj2002.1930

Williams, J. D., and Wuest, S. B. (2011). Tillage and no-tillage conservation effectiveness in the intermediate precipitation zone of the inland Pacific Northwest, United States. *J. Soil Water Conserv.* 66, 242–249. doi: 10.2489/jswc.66.4.242

Wilson, D., Blain, D., Couwenberg, J., Evans, C. D., Murdiyarso, D., Page, S. E., et al. (2016). Greenhouse gas emission factors associated with rewetting of organic soils. *Mires and Peat* 17, 1–28. doi: 10.19189/MaP.2016.OMB.222

Woolf, D., Amonette, J. E., Street-Perrott, F.A., Lehmann, J., and Joseph, S. (2010). Sustainable biochar to mitigate global climate change. *Nat. Commun.* 1:56. doi: 10.1038/ncomms1053

Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.