Carbon Farming – A Working Paper Assessing the Potential for Soil C Sequestration

Keith Paustian[1,2], Eric Larson[3,4], Amy Swan[2], Ernie Marx[2], Jeffrey Kent[2] and Nickie Zenes[4]

[1] Dept. of Soil and Crop Sciences, Colorado State University, Fort Collins, CO 80523
[2] Natural Resource Ecology Laboratory, Colorado State University, Fort Collins, CO 80523
[3] Andlinger Center for Energy and the Environment, Princeton University, Princeton, NJ 08544
[4] Climate Central, Inc., 1 Palmer Square, Princeton, NJ 08540

Background and Summary

Most greenhouse gas (GHG) emission scenarios that keep the world from warming more than 2°C by the end of the century depend on removing significant amounts of CO_2 from the atmosphere and sequestrating it in the biosphere or geosphere – as much as 15 billion tonnes of CO_2 per year ($GtCO_2$/y) by later this century. A number of technological strategies for achieving these negative emissions have been proposed, but it is increasingly recognized in the scientific community that our knowledge of how to implement negative emissions systems *at scale*, and the associated collateral impacts, are still only just developing.

Research on negative emissions systems will need to be accelerated to ensure intended benefits of such technologies are achievable and at the scales and in the time frames relevant to meeting the goals of the Paris climate agreement. Understanding the potential for carbon negative technologies, quantitatively assessing the impacts on food, energy, and water of land-intensive strategies to limit global warming, and identifying and overcoming research and communications gaps are critical to developing policy solutions grounded in the best science.

To inform policies that can help bring negative emissions technologies to scale, this paper reviews the prospective need for negative emissions in the U.S. and globally, the state of science and technologies relevant to near-term deployment of negative emissions systems, and estimated potentials for agriculture-based processes for negative emissions. The latter are generally acknowledged to be the most promising options for the near term, both for the potential speed and scale at which they could be adopted and the collateral benefits that they could provide, such as improved water quality and soil health.

This review is part of a larger multi-disciplinary initiative involving researchers at Colorado State University, Princeton University and Climate Central to assess strategies and potential impacts of a greatly heightened role for agricultural and other land-based systems providing removal and sequestration of CO_2 via enhanced soil organic carbon sequestration, ecosystem restoration or afforestation on marginal and/or abandoned agricultural lands, and biomass energy with CO_2 capture and storage (BECCS). The technical, economic and societal feasibility of these biological negative emission (BNE) strategies and their implications for the future of food and water systems are far from clear. The larger initiative aims to address these uncertainties by seeking to answer several questions in depth:

- What are the potential consequences for land use, food production, water resources, the energy system, and net global greenhouse gas emissions of deploying BNE measures at scale in the U.S.? What are the key uncertainties?
- What strategies can help optimize the efficacy with which biomass, land, and water are used to achieve BNE for the U.S. and to decrease emissions from current and future agriculture, while feeding the world?
- Based on the best available evidence, what are *sustainable*, cost-constrained biogeophysical potentials (and associated uncertainties) in the U.S. for biological removal of CO_2 from the atmosphere via BECCS and other BNE measures?

- To what degree will development and deployment of these strategies meet the required negative emissions consistent with remaining at or below 2ºC globally?
- How best to communicate the science and policy options around BNE strategies to broad audiences?

Sequestering carbon in soils

Cropland soils are typically depleted in carbon relative to their native ecologies: most long-term cropped soils have lost 30-50% of the carbon stocks in top soil layers relative to their native condition. Grassland soils managed for grazing may or may not have suffered similar carbon losses, depending on how they have been managed. How croplands and grazing lands are managed affects both the rate at which they take up carbon from the atmosphere and the rate at which carbon returns to the atmosphere via decomposition. Mineral soils have an upper 'saturation' level of soil carbon above which no further increases are possible. Soils that already have high organic matter levels (e.g. > 5% carbon by mass) have a low propensity for further carbon gain. Soils that are carbon-depleted, however, have the ability to store substantial amounts of carbon. Storage can continue for several decades, depending on the initial condition of the soil. The initial rate of storage in such soils can be high, but will attenuate as the carbon stock builds toward a new equilibrium condition, determined by climate, soil and management conditions. Importantly, soil carbon gains are reversible, and much or all of the gained carbon can be lost if management practices resulting in carbon buildup are not maintained long term.

One can divide management interventions to increase soil carbon into two broad categories. The first category includes well-known, proven techniques that are conservation-oriented management practices, or "best management practices" (BMP) for building soil carbon. The impacts of these BMPs are relatively well-known from numerous field experiments and comparative field observations. Examples include increasing crop productivity coupled with residue retention, use of cover crops, conversion of marginal cropland to perennial grasses or legumes, modified tillage, adding manure or compost, and improved grazing-land management. A second broad category includes what we refer to as 'frontier technologies'. These are systems or practices 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 soil carbon increases over the longer term. Technologies that we consider in this category 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 carbon transfer to soil.

We reviewed published estimates made over the past two decades of global and U.S. soil carbon sequestration potential, mainly on cropland and managed grassland. These generally represent the biophysical potential for managed cropland and/or grassland systems to store additional carbon assuming widespread (near complete) adoption of BMPs. As such, these represent upper-bound estimates of the carbon sequestration potential without 'frontier technologies'. Most estimates, particularly at global scale, are based on highly aggregated data, but are useful for putting in perspective the potential of negative emissions from soil carbon sequestration relative to the need for negative emissions to achieve the goals of the Paris agreement. Our review of the literature suggests that 4 to 5 $GtCO_2/y$ is a good estimate of the global potential for soil carbon sequestration (with 0.2-0.3 $GtCO_2/y$ in the US alone), via a broad suite of well-understood BMPs on grasslands and croplands globally. In the longer-term, if 'frontier technologies' are successfully deployed, the global estimate might grow to 8 $GtCO_2/y$. For comparison, meeting the goals of the Paris agreement may require negative emissions globally of about 8 $GtCO_2/y$ by mid-century and 15 $GtCO_2/y$ by the end of the century. Current global CO_2 emissions from fossil fuel combustion are about 40 $GtCO_2/y$.

Negative emissions systems beyond agriculture-based options will ultimately be needed to meet the goals of the Paris agreement, but implementing best management practices in agriculture will provide significant steps forward at relatively low costs and with a variety of collateral benefits. The options to be implemented will be specific to local conditions. In this paper, we present spatially disaggregated (county-by-county) high-level estimates for the U.S. of existing BMP opportunities (i.e., not including frontier technologies) for negative emissions. Collectively, these represent about 0.2 $GtCO_2$/y of potential soil C stock increase in the country. The highest carbon sequestration potentials via management interventions on cropland are in the Midwest, northern Great Plains and Mississippi River Valley. The potential on irrigated croplands of the arid and semi-arid west are also significant. Potentials for U.S. grasslands (western rangelands and eastern pastures) are lower than those we estimated for croplands, but spatial patterns are more distinct due to differences in climate and in management practices: eastern pastures sequester more carbon per unit area, but the expanse of western rangelands leads to a higher total carbon sequestration on grasslands in the west than in the east.

In summary, 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. While much of the infrastructure for an effective MRV system for soil C sequestration is already in place, investment to buildout a full-scale, on-farm national soil monitoring network is a critical need. Likewise continued support for long-term field experiments and refinements in farm- and national-scale soil C inventory systems are needed. Finally, improved communication of the science issues and policy alternatives is needed to inform government, industry, NGOs, other stakeholders and the general public. These actions would enable the US to move rapidly to implement negative emission strategies in the agricultural sector and at the same time improve the health and resilience of the nation's soils.

1. Introduction: the need for negative emissions

The Paris Climate Agreement provides hope for limiting the impacts of global warming, with the world's nations agreeing to keep the global average temperature rise to *"well below 2°C above pre-industrial levels and to pursue efforts to limit the temperature increase to 1.5°C.*" Because long-term warming increases approximately linearly with total cumulative emissions of CO_2 to the atmosphere (Intergovernmental Panel on Climate Change [IPCC], 2014a), it is possible to estimate the total remaining amount of allowable CO_2 emissions that would commit the world to a warming of 2°C. This is estimated to be between 600 and 1200 billion tonnes of CO_2 ($GtCO_2$), starting from 2016 (Rogelj *et al.* 2016, Schleussner *et al.* 2016). At the current global rate of CO_2 emissions (about 40 $GtCO_2$/y), the lower end of this range would be reached by 2030, the upper end before 2050. Thus, even in the unlikely event that dramatic reductions in global CO_2 emissions begin soon, large-scale negative emissions, wherein CO_2 is captured from the atmosphere and held in storage in the biosphere, geosphere, or oceans will likely be required to bring cumulative emissions back under the cap after first "overshooting" the cap.

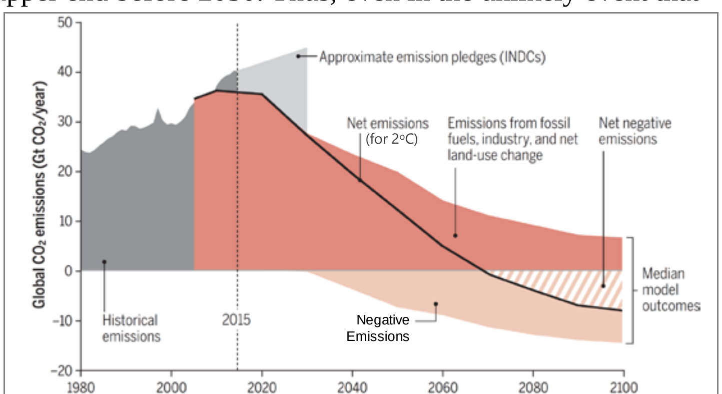

Figure 1. Median outcomes of 76 integrated assessment scenarios assessed by the IPCC that are consistent with not exceeding warming of 2°C (Anderson and Peters, 2016). Negative emissions appear as early as 2030 and reach 15 billion tCO_2/y ($GtCO_2$/y) by later this century.

The imperative of negative emissions is reflected in the many emissions scenarios assessed in the IPCC's *5th Assessment Report* (IPCC 2014b) that require negative emissions in future emissions scenarios to limit warming to less than 2°C (Fig. 1). In fact, negative emissions are also included in many integrated assessment model scenarios that fail to achieve the 2ºC target (Fuss et. al. 2014). In the long term, some level of sustained negative emissions will be required to maintain climate stability by offsetting the impacts of difficult-to-eliminate emissions of greenhouse gases (GHGs), such as methane from agriculture (Rockström *et al.* 2016).

Natural processes (ocean sinks and land sinks) today remove from the atmosphere the equivalent of more than half of CO_2 emitted by fossil fuel combustion and other anthropogenic activities. Negative emissions, as discussed here, would represent either an enhancement of these removal processes or a supplement to them.

2. Overview of Negative Emissions Technologies

A variety of ways in which negative emissions might be achieved have been proposed:

1. Enhancing natural processes for CO_2 removal via land management, e.g., reforestation/afforestation, changing agricultural practices and crop phenotypes to increase soil uptake of carbon, or establishing perennial grasses on carbon-depleted soils unsuitable for arable agriculture.
2. Using biomass (residues or purpose-grown plant matter) for energy and capturing the resulting byproduct CO_2 for storage away from the atmosphere, for example in deep geologic formations. This approach is often called BECCS, for biomass energy with carbon capture and storage.
3. Directly extracting CO_2 from the atmosphere by passing ambient air over a solvent that selectively absorbs CO_2, and then storing the CO_2 away from the atmosphere, for example in deep geologic formations. This process is often called DAC, for direct air capture.
4. Accelerating the natural weathering of certain widely-occurring rock types by bringing them into contact with concentrated CO_2, leading to storage of carbon as either solid carbonate mineral (on land or in the ocean) or as dissolved bicarbonate ions (in the ocean).
5. Fertilizing the ocean with iron to promote photosynthetic uptake of CO_2 by phytoplankton that subsequently sinks to the deep ocean, carrying carbon with it.

The ocean fertilization option (#5) has been studied for more than two decades (Williamson *et al.* 2012), and the understanding that has developed of potential ecosystem disruptions and other negative impacts has resulted in near-universal rejection of this as an acceptable approach for negative emissions (Williamson 2016). Accelerated weathering strategies (#4 above) involving use of the ocean also carry environmental risks, and weathering strategies using either land or ocean would have large, economically-challenging requirements for mining and processing of mineral material (National Research Council, 2015).

Of the five options listed above, BECCS and DACS (#2 and #3) are projected to have the largest potential for annual CO_2 removal (> 10 Gt CO_2/y globally) (National Research Council 2015). Equipment for capturing CO_2 in BECCS systems is already commercially used in other applications, and deep underground storage of CO_2 (captured from fossil fuels) is ongoing at commercial scale in a number of projects around the world (Global CCS Institute, 2017). Significant quantities of biomass residues from agricultural and forestry operations are available today (IPCC 2011, Turkenburg *et al.* 2012) and could be used for BECCS systems without impacting agricultural or forest production (Tilman *et al.* 2009). However, a key limitation of BECCS today is a lack of sufficiently strong policies to incentivize their economics. A carbon emission price of \$100/tCO_2 (National Research Council 2015) or higher (Hailey *et al.* 2016) would likely be needed for BECCS systems to be commercially viable. For the projected levels of negative emissions needed by mid-to-late century to limit global warming to 2ºC or less, there is additionally a concern about competition for land and water to grow biomass for BECCS versus to provide feed and fiber for increasing populations (Smith *et al.* 2016; Field & Mach 2017). There is not the same level of concern about land requirements for DACS, because these systems will have relatively smaller footprints on the land. However, DACS technologies are still under development today, and carbon emission prices projected for commercial viability once the technologies are ready are in the range of \$400 to \$1000/tCO_2 (National Research Council 2015, Socolow *et al.* 2011). Projected costs are high in large part because CO_2 occurs in a relatively low concentration in air (currently about 400 ppm), and removing a low-concentration species from a mixture of gases is thermodynamically (and thus practically) difficult.

Currently, the most technologically established and lowest projected-cost negative-emissions option is land management that enhances natural biological sinks, i.e., #1 in the above list. A theoretical upper bound estimate of the cumulative negative emissions potential for this strategy is 660 ± 300 Gt CO_2-equivalent, which is the estimated historical loss of carbon (C) from native ecosystems (forests, grassland, wetlands) globally, that resulted from human-induced land-use change, largely to agricultural uses (IPCC 2014b). Without reestablishing native ecosystems, negative emissions can also be achieved via enhanced storage of carbon in managed agricultural soils through increased plant productivity, cultivation of deeper-rooted or more decomposition-resistant crops, decreased soil disturbance, and with organic amendments such as biochar addition (Paustian *et al.* 2016a). Also, perennial grasses and trees established on land abandoned from agricultural use due to soil degradation or lands that are poorly suited for agricultural production (Zumkehr & Campbell 2013, Campbell *et al.* 2008) can result in significant carbon build-up over time (Tilman *et al.* 2006). Because the stock of organic carbon in soils today is large – estimated to be 8,600 GtCO_{2eq} globally (Stockmann *et al.* 2013) – small percentage changes in soil carbon via changes in land use or in land management can have large impacts on atmospheric CO_2. However, the total potential for natural accrual of organic carbon in any soil is limited by a number of biogeochemical controls, including climate and soil minerology (Schmidt *et al.*, 2011). Paustian *et al.* (2016a) have estimated the overall carbon mitigation potential for a combined set of natural and anthropogenic soil management strategies to be as high as 8 billion tCO_{2eq}/y, corresponding to 240 GtCO_{2eq} over a 30-year period. Field and Mach (2017) have recently pointed out that the land management option comes with potential environmental co-benefits, such as improving habitat quality or increasing agricultural yields. They also noted the important downside risk that most carbon sequestered in ecosystems can be re-released relatively easily to the atmosphere if that ecosystem is disturbed. Thus policy interventions need to focus both on the adoption and the long-term maintenance of CO_2 sequestering practices.

3. Soil C sequestration technology/management options

Globally, soils contain about 1500 Gt of organic carbon (C)[1] to 1 meter (m) depth and 2400 GtC to 2m depth (Batjes 1996). About 45% of global soils are under some form of agricultural use (i.e., cropland and grazing land). Cropland soils are typically depleted in C relative to the native ecosystems from which they were derived, due to reduced net primary production, export (as products) of biomass, intensive soil disturbance, and soil erosion as contributing factors (Paustian *et al.* 1997). Most long-term cropped soils have lost 30-50% of the C stocks in top soil layers (0-30 cm) relative to their native condition (Davidson and Ackermann 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-

[1] In this paper, C refers to carbon and CO_{2eq} refers to CO_2-equivalents. One tonne of C is equivalent to 3.67 tonnes of CO_{2eq}.

managed grasslands may have C stocks equal to or exceeding their original native condition (Conant *et al.* 2016).

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)[2]. 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 a decreased decomposition rates, soil C stocks tend towards 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, soils have an upper limit or 'saturation level' of soil C (Six *et al.* 2002) above which no further increases in soil C are possible. 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 8-20, 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 tonnes CO_{2eq}/y (1.1 GtC/y), then about 100 million tonnes 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 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

[2] Other organic C can be lost as CH_4 from anaerobic (e.g. flooded) systems as well as leaching of dissolved organic C, but the latter is a minor loss process in most ecosystems. Soil erosion can greatly affect C stocks at a particular location, but 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 result in a small net C sink, because burial of C-rich sediment reduces decomposition rates and erosional exposure of low C subsurface soil can have a higher capacity to store additional C (van Oost *et al.* 2007)

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 help '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 and other losses of pollution-causing reactive nitrogen to the environment, will be an important part of strategies for negative emissions from soils.

In evaluating management interventions to increase soil C stocks, one can divide them into two broad categories. The first category includes well-known, proven techniques that can increase soil C by adopting more conservation-oriented practices, with conventional crops and existing management systems. In other words, adopting 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 includes what we refer to as 'frontier technologies' which represent systems or practices for which significant technological and/or economic barriers exist, such that they are not yet ready 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.

3.1 Conventional conservation practices 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 (from Paustian 2014).

Management Practice	**Increased C inputs**	**Reduced C losses**
Increased productivity and residue retention	✓	
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	✓	

Modified crop rotations

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., Sherrod *et al.* 2005, O'Dea *et al.* 2015, West and Post 2002). 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 the 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 net GHG reductions. An excellent 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 about 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, three 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 three 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.

No-tillage and other conservation 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 croplands. Advances in tillage implement 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% (Ghidey and Alberts 1998, Langdale *et al.* 1979, Mickelson *et al.* 2001, Williams & 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 at 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) predicted 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. There are instances in which no-tillage does <u>not</u> increase soil C relative to conventional tillage (Angers & 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. 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 tha^{-1} 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 to untreated fields. The implied average rates 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 new 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
With the exception of some managed pastures, grazing lands are generally never tilled. Therefore, increasing SOC stocks under perennial grasses relies 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 (Conant *et al.* 2016). Overall, improved grassland practices may increase SOC stocks by as much as 10% (Conant *et al.* 2016). Other analyses of grazingland 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 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 crop 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 effect 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 potential mechanisms on 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.

Additional information on field studies and syntheses of the impacts of various 'conventional' management practices on soil carbon storage are given in the Appendix.

3.2 '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, most commonly 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 & Amelung 2003, Bird *et al.* 2015). Hence biochar/pyogenic 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., Wang *et al.* 2016, Song *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 of 9-12% while an earlier global assessment (Cayluela *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 are 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 CO_{2eq} 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 (Abalos *et al.* 2016, Glover *et al.* 2010b, Pimentel *et al.* 2012, Kantar *et al.* 2016, Crews & 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.* (2014) found that intermediate wheatgrass increased the amount of labile soil C after four 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.

Table 2. Observed rates of SOC change under various managed perennial systems. Results are annualized rates of change from multi-year studies.

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	-5.4-13.0	(Qin *et al.* 2016)
Miscanthus	1.97	-4.7-8.2	
Poplar	0.56	-3.4-6.0	

¶ Mean value from a global meta-analysis of 93 studies.

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 < 1000 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 an 4 year study in Southwestern Michigan, Culman *et al.* (2014) reported average yields ranging from 119 kg/ha/y (in 2012) to 1493 kg/ha/y (in 2011), with a mean over the four years of 485 kg/ha/y. In a four-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. However, the potential for mixed grain and forage production and targeting the use of marginal lands that are poorly suited for annual grain production may be key to successful commercialization (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

As described in section 3.1, one of the most effective means for increasing soil C sequestration is through changing land cover, such as converting annual cropland to forest or perennial grasses, either of which generally contributes much more plant residue to soils. However, if widely applied, such land use conversions would have negative consequences for food and fiber production from the crops that are displaced. One future option, already described above, might be the deployment of perennial cereals, which would provide grain as well as significantly increase soil C storage, on at least a portion of current annual cropland. A related option that has not yet been widely explored would be to modify, through targeted breeding and plant selection, annual crop plants to produce more roots, deeper in the soil profile. Thus 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 towards 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 towards those found in perennial grasses. The impacts on soil C stocks of altered root systems were then modeled using the Century process-based ecosystem C model. The analysis also evaluated impacts of the projected soil C changes on soil N interactions, including potential changes in soil N_2O emissions. Summary results of the analysis on deployment of annual crops with enhanced root phenotypes for deployment in the US are included in Table 4.

4. Review of published estimates of soil C sequestration potential

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 twenty 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

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

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.

4.1 Global potential

Despite somewhat different scope (land types included) and assumptions (practices considered), there is fairly close alignment among global estimates (Table 3), suggesting a technical soil C sequestration potential of 2 to 5 Gt CO_2 per year, for what were characterized in the section above as 'conventional' management practices. Estimates towards 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 to 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 of 2-3 decades before decreasing, as soil C levels approach a new equilibrium.

Table 3. Published estimates of **global** soil carbon sequestration potential. All values reflect technical or 'biophysical' potential estimates that are not constrained by carbon price or policy design.

Study/Citation	**Estimate** Gt CO_{2eq}/y	**Scope**
Paustian *et al.* 1998	1.5-3.3	Improved cropland management, setaside, restoration of degraded land
Lal & Bruce 1999	1.7-2.2	Improved cropland management, restoration of degraded land¶
IPCC 2000	3	Improved cropland & grassland management, setaside, agroforestry, restored peat soils
Lal 2004	1.5-4.4	Improved cropland & grassland management, setaside, agroforestry, restored degraded lands
Smith *et al.* 2008	5-5.4	Improved cropland & grassland management, setaside, agroforestry, restored degraded lands, restored peat soils**§**
Sommer & Bossio 2014	2.5-5.1	Improved cropland & grassland management, setaside, agroforestry, restored degraded lands
Paustian *et al.* 2016a	2-5	Improved cropland & grassland management, setaside, agroforestry, restored degraded lands, restored peat soils
Paustian *et al.* 2016a	4-8	Potential from practices above, plus unconventional technologies including high root C input crop phenotypes and biochar amendments

¶ An additional 1-1.5 GtCO_{2eq} emission reduction was projected from biofuel CO_2 offsets

§ This study also included an estimate of 'economic potential': about 2.5 GtCO_{2eq}/y was achievable for < $50 tonne CO_2

The estimate by Paustian *et al.* (2016a) that goes as high as 8 Gt CO_2 per year includes 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 technologies included in other global estimates. However, estimates of technical potentials for these unconventional practices 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 strongly 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. Budiman et al. 2017, Chambers et al. 2016, Lal 2016). In any case, as an aspiration 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 (Figure 1).

4.1 U.S. potential

Estimates of soil C sequestration for the US have employed roughly similar methods to those described above for global studies, although in some cases analyses have been done with more stratified and spatially disaggregated data to better account for the dependency of C accrual rates on climate and soil conditions.

Table 4. Published estimates of **US** soil carbon sequestration potential. Previously published studies reflect technical or 'biophysical' potential estimates that are not constrained by carbon price or policy design.[4]

Study/Citation	Estimate Million t CO_{2eq}/y	Scope
Lal *et al.* 1998	275-639	Land conversion and setasides, restoration of degraded land, improved management on cropland ¶
Sperow *et al.* 2003	305	Improved cropland management, setaside of marginal (highly erodible) cropland to grassland §

[4] The analysis done in this paper reflects somewhat more of a policy-constrained C sequestration potential in that modeled adoption rates are extrapolated from observed trends but they don't explicitly consider economic constraints.

Sperow 2016	240	Improved cropland management, setaside of marginal (highly erodible) cropland to grassland §
Chambers *et al.* 2016	250	Improved cropland and grassland management, setaside of marginal (highly erodible) cropland to grassland †
Paustian *et al.* 2016b	500-800	Deployment of enhanced root phenotypes for major annual crops (assumes 2X root C input and downward shift in root distribution equivalent to native prairie grasses) ‡
This study	216	Improved cropland and grassland management, setaside of marginal (highly erodible) cropland to grassland (see description below)

¶ Value corrected for error in the publication for C sequestration from residue management; also 'stacking' of multiple practices for the same land area increased their total estimate

§ Based on IPCC Tier 2 method

† Based on estimate for widespread adoption of USDA/NRCS conservation practices on all private lands

‡ Excluded non-irrigated semi-arid cropland with major water limitation on production

Estimates for US agricultural (cropland and grazing land) lands are on the order of 200-300 million t CO_{2eq} per year, with widespread adoption of soil C sequestering BMPs (Table 4). The higher upper limit for the Lal *et al.* (1998) study is likely an overestimate as rates multiple practices were in some cases combined for the same land area and the per ha estimates used may be most representative for humid and subhumid climates and hence would underestimate gains for semiarid cropland areas. The estimate of Paustian *et al.* (2016b) for widespread deployment of enhanced root phenotype crops on US cropland is nearly twice as high as that for all conventional C sequestering practices combined. However, these values are very speculative in that they presuppose the successful development of radically different crop varieties from those currently in use. A major assumption is that such crops would have much greater dry matter allocation belowground, while suffering minimal yield declines (i.e., an assumption in the analysis was that no additional annual cropland would be needed to maintain current commodity production levels).

Additional information on global and national estimates for soil carbon sequestration potential are given in the Appendix.

5. A new spatially-resolved assessment of soil C sequestration potential in the US

Although a few US-based assessments of potential soil C sequestration have been done in the past (Table 4), most have been based on highly aggregate data that minimally represent variability in climate, soil and land management practices. To get a more spatially resolved view of potential sequestration, we estimated the technical potential at the county level of U.S. croplands and grasslands to sequester carbon using a simple but robust approach based on the IPCC Tier 2 national GHG inventory methods (IPCC 2006) with U.S. specific reference carbon stocks and stock change factors (Eve *et al.* 2002). Baseline (current) SOC stocks were compared to two alternate future scenarios in which best management practices are adopted at a moderate level and at a high potential level.

Croplands

Within the IPCC method for croplands, reference SOC stocks are adjusted by land use, tillage and C inputs to the soil. Future scenarios were designed to represent realistic adoption rates of improved practices rather than complete conversion. Carbon sequestration scenarios for croplands were defined as follows:

- **Moderate Adoption Scenario**
  - All conventionally tilled lands are converted to reduced till (Table 5)
  - No-till adoption increases by 50% over current rates (Table 5)
  - Single practices increasing C inputs (such as cover crops, rotation intensification, perennials in rotation) are adopted on all croplands resulting in low input systems becoming medium input systems, and medium input systems becoming high input systems (Table 6)
  - 50% of marginal croplands are converted to native permanent vegetation or perennial bioenergy crops
- **Widespread Adoption Scenario**
  - All conventionally tilled lands are converted to reduced till (Table 5)
  - No-till adoption increases by 100% over current rates (Table 5)
  - Two or more practices increasing C inputs (such as cover crops, rotation intensification, perennials in rotation) are adopted on all croplands resulting in low and medium input systems becoming high input systems (Table 6)
  - 100% of marginal croplands are converted to native permanent vegetation or perennial bioenergy crops

Soil carbon reference stocks and stock change factors are unique to IPCC soil types and climate zones. As such, we classified SSURGO (USDA NRCS 2016) soils data into IPCC soil types according to the IPCC 2006 Guidelines (IPCC 2006). Soils were then overlaid with IPCC climates zones (IPCC 2006). To determine current C input levels of croplands as outlined in the IPCC guidelines (IPCC 2006), we evaluated USDA Cropland Data Layers (CDL) remotely sensed land cover data for 2009-2015 and classified each pixel into low, medium, and high C inputs based on the residue production of crops reported and frequency of fallowing (USDA-NASS 2009-2015). At this time, no national datasets report frequency of cover crop use. The Crop Residue Management Survey reported county-level rates of conventional, reduced, and no till, which were then area-weighted over the spatially distinct soil, climate and crop input combinations (CTIC 2004). We determined areas of marginal croplands by overlaying SSURGO pixels with land capability classes of 4 or greater with CDL croplands (USDA NRCS 2016).

Table 5. Average U.S. tillage percentages for baseline, moderate adoption, and widespread adoption scenarios.

Scenario	Conventional Till	Reduced Till	No Till
Baseline	38%	40%	22%
Moderate Adoption	0%	67%	33%
Widespread Adoption	0%	56%	44%

Table 6. Average U.S. carbon inputs grouped into IPCC input categories of low, medium and high. Carbon inputs are based on crops grown and fallow frequency.

	Carbon Inputs		
Scenario	**low**	**medium**	**high**
Baseline	8%	80%	11%
Moderate Adoption	0%	8%	92%
Widespread Adoption	0%	0%	100%

Grasslands

IPCC methods for grasslands focus on the condition of grasslands, ranking them as severely degraded, moderately degraded, nominal or improved. Baseline (current) and future scenarios assess the impact of changing grassland condition on western rangelands and eastern managed pastures:

- Baseline Scenario
    - Western rangelands: 80% are nominal, 10% are moderately degraded, 10% are severely degraded
    - Eastern pastures: All pastures are nominal condition
- Moderate Adoption Scenario
    - Western rangelands: 90% are nominal, 5% are moderately degraded, 5% are severely degraded
    - Eastern pastures: 50% are nominal and 50% are improved
- Widespread Adoption Scenario
    - Western rangelands: 95% are nominal, 5% are moderately degraded
    - Eastern pastures: 10% are nominal and 90% are improved

Spatial data for IPCC soil and climate zones (as described above) were used to determine soil carbon references stocks and stock change factors. We used 2011 National Land Cover Data (NLCD) (Homer *et al.* 2015) to determine areas of grassland, excluding any areas that overlapped with the CDL Cultivated Layer (USDA-NASS 2016). To separate extensive, western rangelands from managed, eastern pastures, we used a longitudinal separation (-100 degrees) that is consistent with the east-west precipitation gradient in the U.S. Herrick *et al.* (2010) estimated from an assessment of National Resource Inventory survey points, that approximately 20% of U.S. western rangelands were at least moderately degraded. No information was available for eastern pastures, so we assumed all pastures were in nominal condition.

Results

Potential C sequestration on U.S. croplands is predicted to be significant, with improved cropland management sequestering approximately 116 million t CO_{2eq}/y under a scenario of moderate adoption of C-sequestering practices (Table 7). Even wider adoption of these practices may sequester upwards of 127 million t CO_{2eq}/y. In this analysis, we removed marginal croplands (totaling about 17.5 Mha) from production, assuming a return to native permanent vegetation or planting of perennial bioenergy crops. Retiring half of all marginal croplands (8.8 million ha, in the moderate scenario) was predicted to remove approximately 12 million t CO_{2eq}/y from the atmosphere, while completely retiring marginal lands from cultivation (widespread adoption scenario) may remove 23 million t CO_{2eq}/y (Table 7). Overall, the potential of the current cropland land base may be as high as 150 million t CO_{2eq}/y, or 3,008 million t CO_{2eq} over 20 years.

The highest C sequestration potentials were predicted in the Midwest, northern Great Plains and Mississippi River Valley (Figure 2). We also predicted significant per ha potential in irrigated croplands of the arid and semi-arid west, although the total land area involved is relatively small.

Using the IPCC inventory methods, potentials for U.S. grasslands were lower than those predicted for croplands. We predicted a C sequestration potential of 36 million t CO_{2eq}/y in the moderate adoption scenario and 66 million t CO_{2eq}/y in the widespread adoption scenario (Table 7). Nationwide adoption of grassland best management practices may remove as much as 1,317 million t CO_{2eq} from the atmosphere after 20 years.

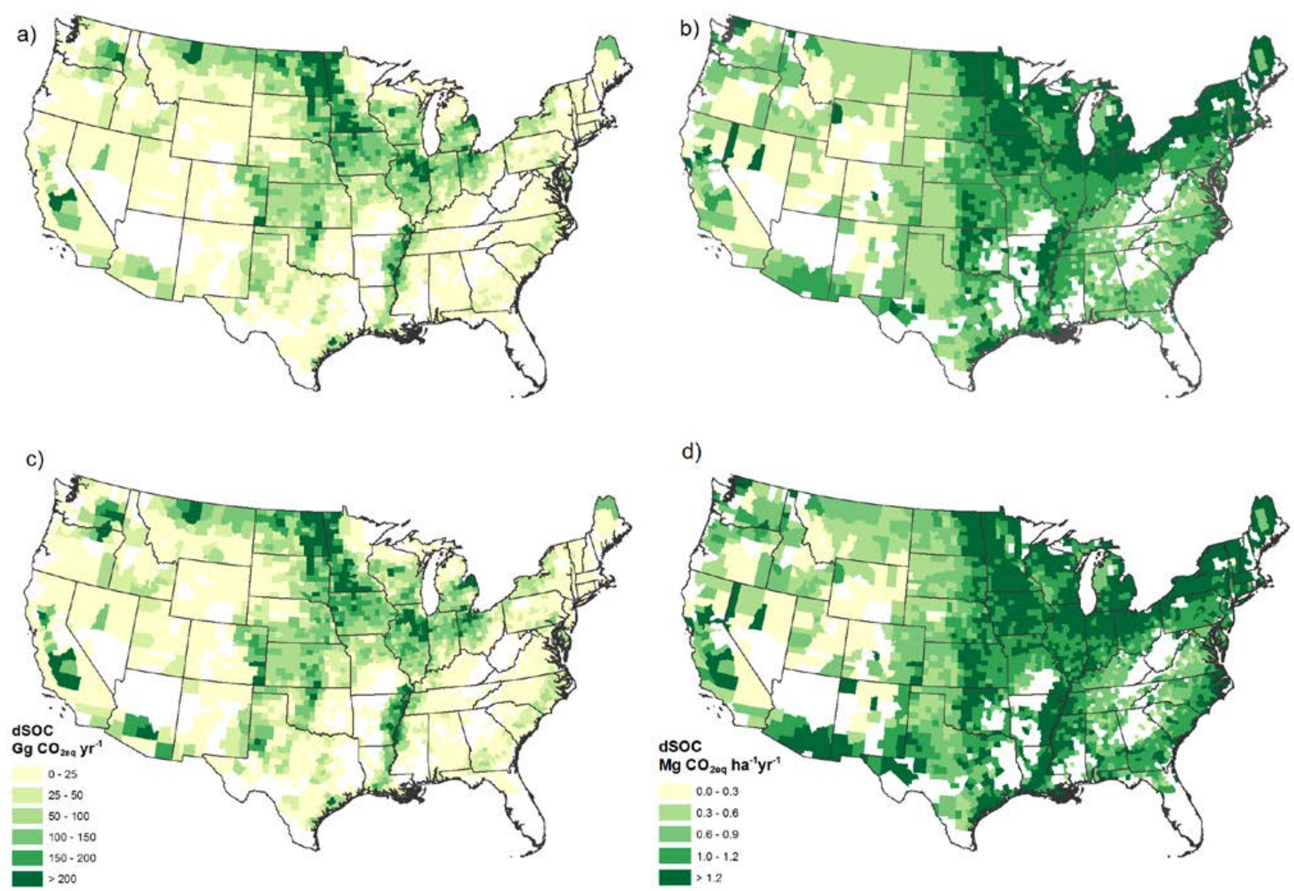

Figure 2. County-level predicted carbon sequestration for croplands: a) moderate adoption scenario, total county C sequestration in thousand t CO_{2eq}/y, b) moderate adoption scenario, average county C sequestration in t CO_{2eq}/ha/y, c) widespread adoption scenario, total county C sequestration in thousand t CO_{2eq}/y, d) widespread adoption scenario, average county C sequestration in t CO_{2eq}/ha/y.

Table 7. Total predicted soil carbon sequestration for moderate adoption and widespread adoption cropland and grassland scenarios.

Management interventions	**Medium adoption scenario** Million t CO_{2eq} y^{-1}	t CO_{2eq}/ha y^{-1}	**High adoption scenario** Million t CO_{2eq} y^{-1}	t CO_{2eq}/ha y^{-1}
Cropland Management	116	1.05	127	1.26
Conversion to set-aside§	12	1.35	23	1.35
Improved grazing land	36	0.36	66	0.65
Total	**164**	**-**	**216**	-

§ Conversion of annual cropland on marginal lands (based on Land Capability Class ≥ 4) to permanent grassland

Spatial patterns of carbon sequestration potentials in grasslands are more distinct, due to the differences climate and management of western vs. eastern pastures. While eastern pastures may sequester more C per hectare, the expansive areas of grasslands in the west may lead to higher total C sequestration potential (Figure 3).

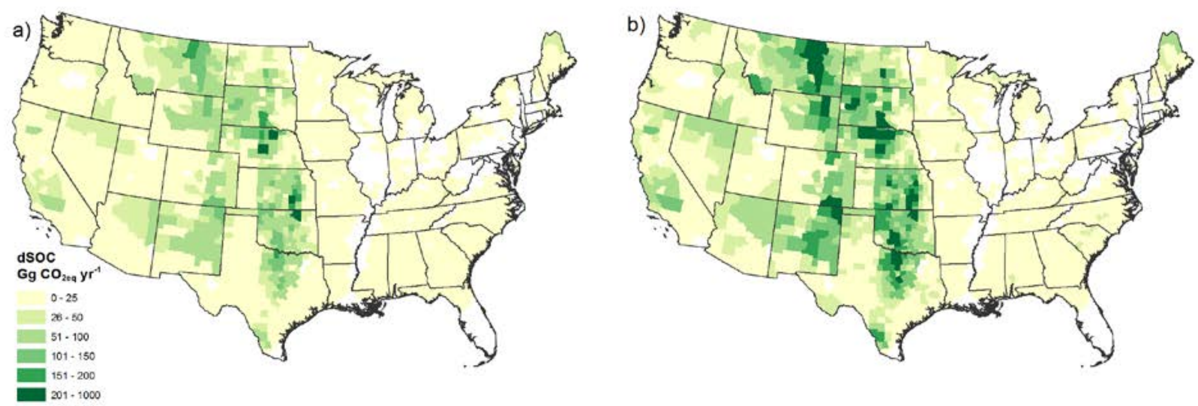

Figure 3. County-level predicted carbon sequestration for grasslands: a) moderate adoption scenario, total county C sequestration in thousand t CO_{2eq}/y and b) widespread adoption scenario, total county C sequestration in thousand t CO_{2eq}/y.

The estimates for soil C sequestration are similar to, but somewhat lower than other recent assessments for the soil C sequestration potential in the US. This is largely due to the fact the adoption rates in our scenarios were based more on extrapolations of historical trends and don't represent the absolute maximum potential assuming complete adoption of best practices for C sequestration. Thus our estimates are somewhat more conservative than published studies.

We used a similar methodology as Sperow (2014) but differing in data sources in that we used a 'wall-to-wall' analysis, at county resolution, based on intersecting spatial databases on land use and soil characteristics whereas Sperow (2014) used sample data from the National Resource Inventory (NRI) system. However, the main difference between our estimates is likely due to assumptions of complete adoption of carbon BMPs by Sperow (2014) whereas our estimates may be characterized as a more conservative in terms of adoption rates.

6. 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 a national 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 is already in place. The US has one of the most sophisticated inventory methods for agricultural soil C stock accounting at national-scale as part of the US national greenhouse gas inventory (Ogle *et al.* 2010, USEPA 2017). At the entity- (i.e., farm and ranch) scale, USDA has developed inventory methods for CO_2, CH_4 and N_2O for all major agricultural sinks and sources, including soil (and biomass) C stocks (Eve *et al.* 2014). The accounting methods are deployed in a free web-based data and computing platform, COMET-Farm™ (Paustian *et al.* 2012, 2017; http://cometfarm.nrel.colostate.edu) which has a user-friendly spatial graphical interface that is being used by farmers, ranchers, crop consultants, companies, NGOs, students and state and federal land managers. The methodologies are currently being used by USDA/NRCS to support conservation programs (Chambers *et al.* 2016), by companies to develop low C footprint supply-chains for agriculturally sourced projects, and by GHG registries (ACR, CAR, VCS,)[5] to support agriculturally-based C offset projects.

To further strengthen MRV capabilities, and in particular improve estimates of measurement uncertainty, a national on-farm soil C monitoring system could be established (van Wesemael *et al.* 2011, Ogle *et al.* 2014). A pilot network with support from USDA/NRCS and utilizing the National Resource Inventory (NRI) systems was initiated in 2007 (Spencer *et al.* 2011), ultimately involving about 500 NRI sample locations. However, following the US budget 'sequester' in 2013, further buildout of the network

[5] The most well-established registries involved in GHG offset project are: American Carbon Registry (ACR) - http://americancarbonregistry.org/; Climate Action Reserve (CAR) - http://www.climateactionreserve.org/; Verified Carbon Standard (VCS) - http://www.v-c-s.org/

towards the full design of 5,000-10,000 points was suspended. However, protocols and procedures are in place and hence the network build-out could be restarted immediately if funding were available.

Finally, continued investments are needed to maintain long-term experiments that include measurements of soil C change and N_2O and CH_4 fluxes as a function of different management practices, and to strengthen and expand coordinated site networks such as Gracenet (Jawson *et al.* 2005) and Ameriflux (Law 2005), helping to refine predictive models in these core MRV systems.

In summary – by leveraging existing scientific knowledge and infrastructure, together with modest investment to further advance the knowledge base and develop new technologies, the US could move rapidly to implement negative emission strategies in the agricultural sector and at the same time improve the health and resilience of the nation's soils. This would stimulate and encourage similar initiatives globally (e.g. INRA 2017), to help achieve the goal of limiting average global temperature to less than 2^oC.

References

Abalos, D., S.E. Brown, A.C. Vanderzaag, R.J. Gordon, K.E. Dunfield & C. Wagner-Riddle. (2016) Micrometeorological measurements over 3 years reveal differences in N2O emissions between annual and perennial crops. *Global Change Biology*, 22: 1244–1255.

Alcantara, V., A. Don, R. Well, and R. Nieder. (2016). Deep ploughing increases agricultural soil organic matter stocks. Global Change Biology 22:2939-2956.

Anderson K & G. Peters (2016). The trouble with negative emissions, *Science* 354:182–3.

Angers, D. & N.S. Eriksen-Hamel. (2008). Full-inversion tillage and organic carbon distribution in soil profiles: A meta-analysis. *Soil Sci. Soc. Am. J.* 72:1370-1374.

Baer, S.G., C.K. Meyer, E.M. Bach, R.P. Klopf, and J. Six. 2010. Contrasting ecosystem recovery on two soil textures: implications for carbon mitigation and grassland conservation. Ecosphere **1**.

Batjes, N.H. (1996). Total carbon and nitrogen in the soils of the world. *European Journal of Soil Science* 47:151-163.

Bell LW, Byne F, Ewing MA, Wade LJ. (2008) A preliminary whole-farm economic analysis of perennial wheat in an Australian dryland farming system. *Agricultural Systems* 96: 166–174.

Bird MI, Wynn JG, Saiz G, Wurster CM, McBeath A. (2015) The Pyogenic Carbon Cycle. *Annual Review of Earth and Planetary Sciences*, 43: 273–298.

Booker, K., L. Huntsinger, J. W. Bartolome, N. F. Sayre, and W. Stewart. (2013). What can ecological science tell us about opportunities for carbon sequestration on arid rangelands in the United States. *Global Environmental Change*. 23, 240-251.

Briske, D. D., J. D. Derner, J. R. Brown, S. D. Fuhlendorf, W. R. Teague, K. M. Havstad, R. L. Gillen, A. J. Ash, and W. D. Willms. 2008. Rotational grazing on rangelands: Reconciliation of perception and experimental evidence. *Rangeland Ecology & Management* 61:3-17.

Caldeira, K., M. Granger Morgan, D. Baldocchi, P. G. Brewer, C. T. A. Chen, G. J. Nabuurs, N. Nakicenovic, and G. P. Robertson. (2004). A portfolio of carbon management options. In: C.B. Field and M.R. Raupach, Editors, The Global Carbon Cycle, Island Press, Washington, DC, USA, pp. 103–109.

Campbell JE, Lobell DB, Genova RC, Field CB. (2008) The global potential of bioenergy on abandoned agriculture lands. *Environmental Science & Technology*, 42: 5791–4.

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

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

Chambers, A. R. Lal & K. Paustian. (2016) Soil carbon sequestration potential of US croplands and grasslands: Implementing the 4 per thousand initiative. *J. Soil Water Conserv*. 71:68A-74A.

Chan, K. Y., A. Cowie, G. Kelly, B. Singh, and P. Slavich. (2009). Scoping paper: soil organic carbon sequestration potential for agriculture in NSW. NSW DPI Science and Research Technical Paper. New South Wales, Australia: NSW Department of Primary Industries.

Conant, R. T., K. Paustian, and E. T. Elliot. (2001). Grassland management and conversion into grassland: effects on soil carbon. *Ecological Applications* 11:343-355.

Conant, R.T., C.E.P. Cerri, B.B. Osborne, & K. Paustian. (2016) Grassland management impacts on soil carbon stocks: a new synthesis. *Ecological Applications,* 27: 662-668.

Cole, C. V., J. Duxbury, J. Freney, O. Heinemeyer, K. Minami, K. Paustian, N. Rosenburg, N. Sampson, D. Sauerbeck, and Q. Zhao. (1997). Global estimates of potential mitigation of greenhouse gas emissions by agriculture. *Nutrient Cycling in Agroecosystems*. 49:221-228.

Cox TS, Van Tassel DL, Cox CM, Dehaan LR. (2010) Progress in breeding perennial grains. *Crop and*

Pasture Science, 61: 513–521.
Crews TE & Rumsey BE. (2017) What agriculture can learn from native ecosystems in building soil organic matter: A review. *Sustainability (Switzerland)*, 9: 1–18.
Crews TE, Blesh J, Culman SW et al. (2016) Going where no grains have gone before: From early to mid-succession. *Agriculture, Ecosystems and Environment*, 223: 223–238.
CTIC. (2004) The National Crop Residue Management Survey. Conservation Tillage Information Center, West Lafayette, IN.
Culman SW, DuPont ST, Glover JD, Buckley DH, Fick GW, Ferris H, Crews TE. (2010) Long-term impacts of high-input annual cropping and unfertilized perennial grass production on soil properties and belowground food webs in Kansas, USA. *Agri, Ecosys Enviro*n 137: 13–24.
Culman, S.W., S. Snapp, C. Sprunger, K. Cassida, A. Peralta, L. DeHaan. (2014). Ecosystem services provided by a novel perennial grain, Kernza. American Society of Agronomy meeting, Long Beach, CA, Nov 2-6, 2014. Viewable at: https://dl.sciencesocieties.org/publications/meetings/download/pdf/2014am/86995
Culman, S.W., J. Pugliese, L. DeHaan, T. Crews, R. M. Sulc, M. Ryan, J. Jungers, J.E. Maul, M.E. Schipanski and C.C. Sheaffer. (2016). Can the perennial grain crop Kernza yield both forage and grain? American Society of Agronomy meeting, Phoenix, AZ, Nov 9, 2016. Viewable at: https://scisoc.confex.com/scisoc/2016am/videogateway.cgi/id/27312?recordingid=27312
Davidson, E.A. & Ackerman, I.L. (1993). Changes in soil carbon inventories following cultivation of previously untilled soils. *Biogeochemistry* 20: 161-164.
DeLonge, M. S., R. Ryals, and W. L. Silver. 2013. A Lifecycle Model to Evaluate Carbon Sequestration Potential and Greenhouse Gas Dynamics of Managed Grasslands. Ecosystems **16**:962-979.
Denef, K., S. Archibeque, & K. Paustian. (2011) Greenhouse gas emissions from U.S. agriculture and forestry: A review of emission sources, controlling factors, and mitigation potential. Interim report to USDA under Contract #GS23F8182H.
Dick, W.A., R.L. Blevins, W.W. Frye, S.E. Peters, D.R. Christenson, F.J. Pierce & M.L. Vitosh. (1998). Impacts of agricultural management practices on C sequestration in forest-derived soils of the eastern Corn Belt. *Soil Tillage Research* 47:235-244.
Dickinson D, Balducció L, Buysse J, Ronsse F, van Huylenbroeck G, Prins W. (2015) Cost-benefit analysis of using biochar to improve cereals agriculture. *GCB Bioenergy*, **7**: 850–864.
Dumanski, J., R. L. Desjardins, C. Tarnocai, C. Monreal, E. G. Gregorich, V. Kirkwood, and C. A. Campbell. (1998). Possibilities for future carbon sequestration in Canadian agriculture in relation to land use changes. *Climate Change* 40:81-103.
Eve, M.D., M. Sperow, K. Paustian and R.F. Follett. (2002). National-scale estimation of changes in soil carbon stocks on agricultural lands. *Environmental Pollution* 116: 431-438.
Eve, M., D. Pape, M. Flugge, R. Steele, D. Man, M. Riley-Gilbert, & S. Biggar, (Eds). (2014) Quantifying Greenhouse Gas Fluxes in Agriculture and Forestry: Methods for Entity-Scale Inventory. Technical Bulletin Number 1939. Office of the Chief Economist, U.S. Department of Agriculture, Washington, DC.
Field, C.B. & K.J. Mach, (2017) Rightsizing carbon dioxide removal. *Science*, 356:706-707.
Follett, R. F. and G. E. Schuman. (2005). Grazing land contributions to carbon sequestration. Grazingland: a global resource. Wageningen, The Netherlands: Wageningen Academic Publishers. p.266-277.
Follett, R. F. and D. A. Reed. (2010). Soil carbon sequestration in grazing lands: societal benefits and policy implications. *Rangeland Ecology and Management* 63:4-15.
Franzlubbers, A. J. (2010). Achieving soil organic carbon sequestration with conservation agricultural systems in the Southeastern United States. *Soil Sci. Soc. Amer. J.* 74:347-357.
Fuss S, Canadell JG, Peters GP, Tavoni M, Andrew RM, Ciais P, et al. (2014) Betting on negative emissions. *Nature Clim Change*, 4:850–3.

Ghidey, F. & E.E. Alberts. (1998) Runoff and soil losses as affected by corn and soybean tillage systems. Journal of Soil and Water Conservation 53: 64-70.

Glaser B & Amelung W (2003) Pyogenic carbon in native grassland soils along a climosequence in North America. *Global Biogeochemical Cycles*, 17: 1–8.

Global CCS Institute, "Large-scale CCS facilities," global database, https://www.globalccsinstitute.com/projects/large-scale-ccs-projects (accessed 24 May 2017).

Glover JD, Reganold JP, Bell LW et al. (2010bs) Increased food and ecosystem security via perennial grains. *Science*, 328: 1638–1639.

Gupta, R. K. and D. L. N. Rao. (1994). Potential of wastelands for sequestering carbon by reforestation. *Current Science* 66:378-380.

Hailey AK, Meerman JC, Larson ED, Loo Y-L. (2016) Low-carbon 'drop-in replacement' transportation fuels from non-food biomass and natural gas. *Applied Energy*, 183: 1722-1730.

Hansen, J., P. Kharecha, M. Sato, V. Masson-Delmotte, F. Ackerman, D.J. Beerling, P.J. Hearty, O. Hoegh-Guldberg, S.-L. Hsu, C. Parmesan, J. Rockstrom, E.J. Rohling, J. Sachs, P. Smith, K. Steffen, L. Van Susteren, K. von Schuckmann, and J.C. Zachos. (2013). Assessing "Dangerous Climate Change": Required Reduction of Carbon Emissions to Protect Young People, Future Generations and Nature (JA Añel, Ed.). PLoS ONE 8, e81648.

He, Y., S. E. Trumbore, M. S. Torn, J. W. Harden, L. J. S. Vaughn, S. D. Allison, and J. T. Randerson. (2016). Radiocarbon constraints imply reduced carbon uptake by soils during the 21st century. *Science* 353:1419-1424.

Herrick, J.E., V.C. Lessard, K.E. Spaeth, P.L. Shaver, R.S. Dayton, D.A. Pyke, L. Jolley, & J.J. Goebel. (2010) National ecosystem assessments supported by scientific and local knowledge. *Frontiers in Ecology and the Environment*, 8: 403-408.

Homer, C.G., Dewitz, J.A., Yang, L., Jin, S., Danielson, P., Xian, G., Coulston, J., Herold, N.D., Wickham, J.D., & Megown, K. (2015) Completion of the 2011 National Land Cover Database for the conterminous United States-Representing a decade of land cover change information. *Photogrammetric Engineering and Remote Sensing*, 81(5): 345-354.

INRA (2017). 4 per 1000. Carbon sequestration in soils. 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 Univ. Press. 377 p.

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

IPCC. (2011) Special Report on Renewable Energy Sources and Climate Change Mitigation, O Edenhofer, R Pichs-Madruga, Y Sokona, K Seyboth, P Matschoss, S Kadner, T Zwickel, P Eickemeier, G Hansen, S Schlömer, C von Stechow (eds). Cambridge University Press.

IPCC. (2014a) Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to 5th Assessment Report.

IPCC. (2014b) Climate Change 2014: Mitigation of Climate Change, Working Group III contribution to the Fifth Assessment Report.

Jaikumar NS, Snapp SS, Murphy K, Jones SS. (2012) Agronomic assessment of perennial wheat and perennial rye as cereal crops. *Agronomy Journal*, 104: 1716–1726.

Jain, A. K., X. Yang, T. O. West, and W. M. Post. (2005). Assessing the effectiveness of soil carbon sequestration in North America and its impact on net terrestrial uptake of CO2. Presented at Seventh International Carbon Dioxide Conference. https://www.esrl.noaa.gov/gmd/icdc7/proceedings/abstracts/jainMC393.pdf

Jawson, M. D., S. R. Shafer, A. Franzluebbers, T. B. Parkin, and R. F. Follett. (2005). GRACEnet: Greenhouse gas reduction through agricultural carbon enhancement network. Soil & Tillage Research **83**:167-172.

Kane, D. (2015). Carbon sequestration potential on agricultural lands: a review of current science and available practices. Report. http://sustainableagriculture.net/wpcontent/uploads/2015/12/Soil_C_review_Kane_Dec_4-final-v4.pdf.

Kane DA, Rog P, & Snapp SS. (2016) A systematic review of perennial staple crops literature using topic modeling and bibliometric analysis. *PLoS ONE*, 11: 1–18.

Kantar MB, Tyl CE, Dorn KM et al. (2016) Perennial Grain and Oilseed Crops. *Annual Review of Plant Biology*, 67: 703–729.

Kell, D. B. (2011). Breeding crop plants with deep roots: their role in sustainable carbon, nutrient and water sequestration. *Annals of Botany* 108:407-418.

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

Lal, R. & J.P. Bruce. (1999) The potential of world cropland soils to sequester C and mitigate the greenhouse effect. *Environ. Science & Policy* 2:177-185.

Lal, R. (2004). Soil carbon sequestration impacts on global climate change and food security. Science, 1623-1627.

Lal, R. (2016). Beyond COP21: Potential and challenges of the "4 per Thousand" initiative. *J. Soil Water Conserv*. 71:20A-25A

Lal, R., J.M. Kimble, R.F. Follett and C.V. Cole. 1998. The potential of U.S. cropland to sequester carbon and mitigate the greenhouse effect. Sleeping Bear Press, Ann Arbor, MI, 128 p.

Lal, R., R. F. Follett, and J. M. Kimble. (2003). Achieving soil carbon sequestration in the United States: a challenge to the policy makers. *Soil Science* 168:827-845.

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

Larkin PJ, Newell MT, Hayes RC, Aktar J, Norton MR, Moroni SJ, Wade LJ. (2014) Progress in developing perennial wheats for grain and grazing. *Crop and Pasture Science*, 65: 1147–1164.

Law, B. 2005. Carbon dynamics in response to climate and disturbance: Recent progress from multi-scale measurements and modeling in AmeriFlux. In: Omasa, K, I. Nouchi and LJ DeKok (eds) Plant Responses to Air Pollution and Global Change, pp 205-213, 6th International Symposium on Plant Responses to Air Pollution and Global Changes, Tsukuba, Japan.

Liu S, Zhang Y, Zong Y 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.

Mackey, B., I. C. Prentice, W. Steffen, J. I. House, D. Lindenmayer, H. Keith, and S. Berry. (2013). Untangling the confusion around land carbon science and climate change mitigation policy. Nature *Climate Change*. 3:552-557.

Machmuller, M. B., M. G. Kramer, T. K. Cyle, N. Hill, D. Hancock, and A. Thompson. 2015. Emerging land use practices rapidly increase soil organic matter. *Nature Communications* 6.

McCalmont, J. P., A. Hastings, N. P. McNamara, G. M. Richter, P. Robson, I. S. Donnison, and J. Clifton-Brown. (2015). Enviromental costs and benefits of growing Micanthus for bioenergy in the UK. *GCB Bioenergy*. doi:10.1111/gcbb.12294.

Meyer, L., S. Dabney, & C. Murphree. (1999) Crop production systems to control erosion and reduce runoff from upland silty soils. *Transactions of the ASAE*, 42: 1645-1652.

Morgan, J.A., R.F. Follett, L.H. Allen Jr., S. Del Grosso, J.D. Derner, F. Dijkstra, A. Franzluebbers, R. Fry, K. Paustian, & M.M. Schoeneberger. (2010) Carbon sequestration in agricultural lands of the United States. *Journal of Soil and Water Conservation*, 65: 6-13A.

Munson, S. M., W. K. Lauenroth, and I. C. Burke. 2012. Soil carbon and nitrogen recovery on semiarid Conservation Reserve Program lands. *Journal of Arid Environments* **79**:25-31.

National Research Council. (2015) Climate Intervention: Carbon Dioxide Removal and Reliable Sequestration, National Academies Press, Washington DC.

O'Dea, J.K., C.A. Jones, C.A. Zabinski, P.R. Miller, & I.N. Keren. (2015) Legume, cropping intensity, and N-fertilization effects on soil attributes and processes from an eight-year-old semiarid wheat system. *Nutrient Cycling in Agroecosystems*, 102: 179-194.

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

Ogle, S.M., F.J. Breidt, M. Easter, S. Williams, K. Killian and K. Paustian. 2010. Scale and uncertainty in modeled soil organic carbon stock changes for US croplands using a process-based model. Global Change Biology 16:810-822.

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

Ogle, S. M., L. Olander, L. Wollenberg, T. Rosenstock, F. Tubiello, K. Paustian, L. Buendia, A. Nihart and P. Smith. 2014. Reducing greenhouse gas emissions and adapting agricultural management for climate change in developing countries: providing the basis for action. *Global Change Biology* 20(1): 1-6.

Paustian, K., O. Andren, H.H. Janzen, R. Lal, P. Smith, G. Tian, H. Tiessen, M. Van Noordwijk, P.L. Woomer. (1997) Agricultural soils as a sink to mitigate CO2 emissions. *Soil Use and Management*, 13: 230-244.

Paustian, K., C.V. Cole, D. Sauerbeck & N. Sampson. (1998) CO_2 mitigation by agriculture: An overview. *Climatic Change* 40:135-162.

Paustian, K. (2014). Carbon sequestration in soil and vegetation and greenhouse gas emissions reduction. In: Freedman, B. (ed) Global Environmental Change, pp. 399-406, Springer Reference, Springer Dordrecht Heidelberg New York London.

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

Paustian, K, N. Campbell, C Dorich, E Marx & A. Swan. (2016b) Assessment of potential greenhouse gas mitigation from changes to crop root mass and architecture. Final report to ARPA-E, 34 p.

Paustian, K., Schuler, J., Killian, K., Chambers, A., DelGrosso, S., Easter, M., Alvaro-Fuentes, J., Gurung, R., Johnson, G., Merwin, M., Ogle, S., Olson, C., Swan, A., Williams, S. and R. Vining. (2012). COMET 2.0 – Decision support system for agricultural greenhouse gas accounting. In: Liebig, M., Franzluebbers, A., and Follett, R., (eds.), Managing Agricultural Greenhouse Gases: Coordinated Agricultural Research through GraceNet to Address Our Changing Climate. Pp. 251-270. Academic Press, San Diego, CA.

Paustian, K., M. Easter, K. Brown, A. Chambers, M. Eve, A. Huber, E. Marx, M. Stermer, B. Sutton, A. Swan, C. Toureene, S. Verlayudhan and S. Williams. (2017). Field- and farm-scale assessment of soil greenhouse gas mitigation using COMET-Farm™. In: J. Delgado (ed). Precision Conservation: Geospatial Techniques for Agricultural and Natural Resources Conservation. SSSA, Madison WI. (in press).

Pimentel D, Cerasale D, Stanley RC et al. (2012) Annual vs. perennial grain production. *Agriculture, Ecosystems and Environment*, 161: 1–9.

Poeplau, C. & A. Don. (2015) Carbon sequestration in agricultural soils via cultivation of cover crops – a meta-analysis. *Agriculture, Ecosystems and Environment*, 200: 33-41.

Qin Z, Dunn JB, Kwon H, Mueller S, Wander MM. (2016) Soil carbon sequestration and land use change associated with biofuel production: Empirical evidence. *GCB Bioenergy*, 8: 66–80.

Rockström J, Schellnhuber HJ, Hoskins B, Ramanathan V, Schlosser P, Brasseur GP, et al. (2016) The world's biggest gamble. *Earth's Future*, 4:465–70.

Rogelj J, Schaeffer M, Friedlingstein P, Gillett NP, van Vuuren DP, Riahi K, et al. (2016) Differences between carbon budget estimates unraveled. *Nature Clim Change* 6:245–52.

Ryals, R., M. D. Hartman, W. J. Parton, M. S. DeLonge, and W. L. Silver. 2015. Long-term climate change mitigation potential with organic matter management on grasslands. Ecological Applications

25:531-545.
Ryals, R., and W. L. Silver. 2013. Effects of organic matter amendments on net primary productivity and greenhouse gas emissions in annual grasslands. Ecological Applications **23**:46-59.
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* February 22, 2016. DOI:10.1371/journal.pone.0148527
Santos, F., M. S. Torn, and J. A. Bird. 2012. Biological degradation of pyogenic organic matter in temperate forest soils. Soil Biology & Biochemistry **51**:115-124.
Schleussner, C. F., J. Rogelj, M. Schaeffer, T. Lissner, R. Licker, E. M. Fischer, R. Knutti, A. Levermann, K. Frieler, and W. Hare. (2016). Science and policy characteristics of the Paris Agreement temperature goal. *Nature Climate Change* **6**:827-835.
Schmidt MWI, Torn MS, Abiven S, Dittmar T, Guggenberger G, Janssens IA, et al. (2011) Persistence of soil organic matter as an ecosystem property. *Nature*, 478:49–56.
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 Science Society of America Journal*, 69:1600-1608.
Six, J., S.M. Ogle, F.J. Breidt, R.T. Conant, A.R. Mosier, & K. Paustian. (2004) The potential to mitigate global warming with no-tillage management is only realized when practised in the long term. *Global Change Biology*, 10: 155-160.
Six, J., R.T. Conant, E.A. Paul, & K. Paustian. (2002) Stabilization mechanisms of soil organic matter: implications for C-saturation of soil. *Plant and Soil*, 241: 155-176.
Six, J. & K. Paustian. 2014. Aggregate-associated soil organic matter as an ecosystem property and a measurement tool. *Soil Biology & Biochemistry* 68:A4-A9.
Skjemstad JO, Reicosky DC, Wilts AR, McGowan JA (2002) Charcoal carbon in U.S. agricultural soils. *Soil Science Society of America Journal*, 66: 1249.
Smaje C. (2015) The Strong Perennial Vision: A Critical Review. *Agroecology and Sustainable Food Systems*, 39: 500–515.
Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F., and Rice, C. (2008). Greenhouse gas mitigation in agriculture. *Philosophical Transactions of the Royal Society B*: Biological Sciences 363:789-813.
Smith P, Davis SJ, Creutzig F, Fuss S, Minx J, Gabrielle B, et al., (2016) Biophysical and economic limits to negative CO2 emissions, *Nature Clim Change*, 6:42–50.
Smith, P., D. S. Powlson, J. U. Smith, P. Falloon, and K. Coleman. (2000). Meeting Europe's climate change commitments: quantitative estimates of the potential for carbon mitigation by agriculture. *Global Change Biology* 6:525-539.
Socolow R, Desmond M, Aines R, Blackstock J, Bolland O, Kaarsberg T, et al. (2011) Direct air capture of CO2 with chemicals: a technology assessment for the APS Panel on Public Affairs, American Physical Society.
Sommer, R. & D. Bossio. (2014). Dynamics and climate change mitigation potential of soil organic carbon sequestration. J. *Environ. Manag.* 144:83-87.
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. *Ecosystem Health and Sustainability*, 2: n/a-n/a.
Song, X., H. Tian, X. Xu, D. Hui, G. Chen, G. Sommers, L. Marzan, and M. Liu. (2013) Projecting terrestrial carbon sequestration of the southeastern United States in the 21st century. *Ecosphere*. 4: 88.
Spencer, S., S. M. Ogle, F. J. Breidt, J. J. Goebel and K. Paustian. (2011). Designing a national soil carbon monitoring network to support climate change policy: a case example for US agricultural lands, *Greenhouse Gas Measurement and Management*, 1:3-4, 167-178.
Sperow, M., M.D. Eve & K. Paustian. (2003) Potential soil C sequestration on U.S. agricultural soils. *Climatic Change* 57:319-339.

Sperow, M. (2016). Estimating carbon sequestration potential on US agricultural topsoils. *Soil & Tillage Research* **155**:390-400.

Stockmann U, Adams MA, Crawford JW, Field DJ, Henakaarchchi N, Jenkins M, et al. (2013) The knowns, known unknowns and unknowns of sequestration of soil organic carbon, *Agriculture, Ecosystems & Environment*, 164:80–99.

Teague, R., F. Provenza, U. Kreuter, T. Steffens, and M. Barnes. 2013. Multi-paddock grazing on rangelands: Why the perceptual dichotomy between research results and rancher experience? *Journal of Environmental Management* 128:699-717.

Teague, W. R., S. L. Dowhower, S. A. Baker, N. Haile, P. B. DeLaune, and D. M. Conover. 2011. Grazing management impacts on vegetation, soil biota and soil chemical, physical and hydrological properties in tall grass prairie. *Agriculture Ecosystems & Environment* 141:310-322.

Tilman D, Hill J, & Lehman C. (2006) Carbon-Negative Biofuels from Low-Input High-Diversity Grassland Biomass, *Science*, 314:1598–600.

Tilman D, Socolow R, Foley JA, Hill J, Larson E, Lynd L, Pacala S, Reilly J, Searchinger T, Sommerville C, Williams R. (2009) Beneficial Biofuels–The Food, Energy, and Environment Trilemma. *Science*, 325:270–1.

Todd-Brown, K. E. O., J. T. Randerson, F. Hopkins, V. Anora, T. Hajima, C. Jones, E. Shevliakova, J. Tjiputra, E. Volodin, T. Wu, Q. Zhang, and S. D. Allison. (2014). Changes in soil organic carbon storage predicted by Earth system models during the 21st century. *Biogeosciences* 11:2341-2356.

Tripathi M, Sahu JN, & Ganesan P (2016) Effect of process parameters on production of biochar from biomass waste through pyolysis: A review. *Renewable and Sustainable Energy Reviews*, 55: 467–481.

Turkenburg, et al., (2012) Renewable Energy Systems. Chapter 11 in The Global Energy Assessment, Cambridge University Press.

USDA FSA. (2012) Conservation Reserve Program Annual Summary and Enrollment Statistics. U.S. Dept. of Agriculture, Farm Service Agency. https://www.fsa.usda.gov/Assets/USDA-FSA-Public/usdafiles/Conservation/PDF/summary12.pdf

USDA-NASS. (2016) Cultivated Layer. Published cultivated/non-cultivated data layer [Online]. Available at https://nassgeodata.gmu.edu/CropScape/ (accessed May 2017). USDA, National Agricultural Statistics Service, Washington, DC.

USDA-NASS. (2009-2015) Cropland Data Layer. Published crop-specific data layer [Online]. Available at https://nassgeodata.gmu.edu/CropScape/ (accessed May 2016). USDA, National Agricultural Statistics Service, Washington, DC.

USDA NRCS. (2016) Soil Survey Geographic (SSURGO) Database. Soil Survey Staff, Natural Resources Conservation Service, United States Department of Agriculture. Available online at < http://www.nrcs.usda.gov/wps/portal/nrcs/detail/soils/survey/geo/?cid=nrcs142p2_053628>.

USEPA. (2017). Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2015. EPA Report 430-P-17-001. 564 p. Online at: https://www.epa.gov/sites/production/files/2017-02/documents/2017_complete_report.pdf (accessed June 2017)

van Groenigen, J. W., C. van Kessel, B. A. Hungate, O. Oenema, D. S. Powlson, and K. J. van Groenigen. (2017). Sequestering Soil Organic Carbon: A Nitrogen Dilemma. *Environmental Science &* Technology 51:4738-4739.

van Oost, K., Quine, T. A., Govers, G., De Gryze, S., Six, J., Harden, J. W., Ritchie, J. C., McCarty, G. W., Heckrath, G., Kosmas, C., Giraldez, J. V., da Silva, J. R. M. & Merckx, R. (2007). The impact of agricultural soil erosion on the global carbon cycle. *Science*, 318: 626-629.

van Wesemael, B., K. Paustian, O. Andrén, C.E.P. Cerri, M. Dodd, J. Etchevers, E. Goidts, P. Grace T. Kätterer, B. McConkey, S. Ogle, G. Pan and C. Siebner. (2011). How can soil monitoring networks be used to improve predictions of organic carbon pool dynamics and CO_2 fluxes in agricultural soils? *Plant and* Soil 338:247-259

Verhoeven E, Pereira E, Decock C, Suddick E, Angst T, Six J. (2017) Toward a Better Assessment of Biochar–Nitrous Oxide Mitigation Potential at the Field Scale. *Journal of Environment Quality*, 46: 237.

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

Wang, J. Y., Z. Q. Xiong, and Y. Kuzyakov. 2016. Biochar stability in soil: meta-analysis of decomposition and priming effects. Global Change Biology Bioenergy **8**:512-523.

West, T.O. & W.M. Post. (2002) Soil Organic Carbon Sequestration Rates by Tillage and Crop Rotation: A Global Data Analysis. *Soil Science Society of America Journal,* 66: 1930-1946.

Williams, J.D. & S.B. Wuest. (2011) Tillage and no-tillage conservation effectiveness in the intermediate precipitation zone of the inland Pacific Northwest, United States. *Journal of Soil and Water Conservation,* 66: 242-249.

Williamson, P. (2016) Scrutinize CO_2 Removal Methods. *Nature*, 530: 153-155.

Williamson P, Wallace DWR, Law CS, Boyd PW, Collose Y, Crootf P, Denmang K, Riebesellh U, Takedai S, Vivianj C. (2012) Ocean fertilization for geoengineering: A review of effectiveness, environmental impacts and emerging governance. *Process Safety and Environmental Protection*, 90(6): 475–488.

Wilson, D., D. Blain, J. Couwenberg, C. D. Evans, D. Murdiyarso, S. E. Page, F. Renou-Wilson, J. O. Rieley, A. Sirin, M. Strack, and E. S. Tuittila. 2016. Greenhouse gas emission factors associated with rewetting of organic soils. *Mires and Peat* 17:1-28.

Woolf D., J. Amonette, F. Street-Perrott, J. Lehmann and S. Joseph. (2010) Biochar as carbon negative in carbon credit under changing climate. *Nature Communications*, 1–9.

Zumkehr A & Campbell JE. (2013) Historical U.S. Cropland Areas and the Potential for Bioenergy Production on Abandoned Croplands. *Environmental Science & Technology*, 47:3840–7.

Appendix A

A1. Published estimates of global potential for greenhouse gas removal from the atmosphere via changes in land management. All values reflect technical or 'biophysical' potential estimates that are not constrained by carbon price or policy design.

Source	Estimate, Gt CO_2-eq yr^{-1}	Duration, years	Measures	Summary/Comments
Caldeira et al. 2004	3.3	100	All possible land use changes	Includes agricultural soils, reforestation/agroforestry (improved management, afforestation, cessation of deforestation), improvement of cropland management.
IPCC 2000	1.4-2.9	50 – 100	Improved croplands/grasslands mgmt., agroforestry, setaside	Capacity of global agricultural soils reaching a new carbon equilibrium after approximately 50 to 100 years.
Kane 2015	1.95-2.44	15 – 30	Use of fertilizer, no till and conservation tillage, cover crops and crop rotation, rotational grazing, perennial cropping systems	Kane also cities Hansen et al. (2013) with an optimistic global carbon soil sequestration rate of ~10% of current annual emissions. This matches the upper limit of Smith et al. (2008) technical estimates without factoring in economic limitation at different carbon prices.
Lal 2004	1.47-4.4	20 – 50	Improved mgmt. of cropland soils, restoration of degraded and desertified soils, irrigated soils, fire mgmt., improved species, graze land mgmt.	Methods differ depending on the biome and type of soil. Cropland soils 1350 Mha total with 250 Mha in South America, which have high potential. Restoration of degraded and desertified soils 1.1 billion ha. Irrigated soils - 275 Mha and includes both soil organic and inorganic carbon. Rangelands and grasslands - 3.7 billion ha in semi-arid and sub-humid regions. Timeframe of carbon sequestration can be much longer than 20-50 years, but unrealistic under on-farm conditions.
Lal and Bruce 1999	1.65-2.22	20 – 50	Improved mgmt. of world's croplands only	World cropland management including erosion control, restoration of degraded soil, reclamation of salt-affected soils, use of conversation tillage practices and crop residue management, and improvement of cropping system. Carbon soil sinks are finite and would reach a new equilibrium after 20-50 years. An additional 1.1-1.5 Gt CO_2 eq yr^{-1} for carbon offset through biofuel production.

Paustian and Cole 1998	1.76 - 3.15	50	Cropland mgmt., restore degraded lands, setaside	Converting croplands back to forests/grasslands, improved management of agricultural lands, and restoration of degraded lands.
Paustian et al. 2016a	1.5 - 5.3	50	Conventional methods	Conventional methods: grazing land management, cropland management, set-aside, water management, restore degraded land, rice management.
	4 - 8	50	Conventional plus unconventional (advanced) technologies/methods	New technologies include genetic engineering of plants to enhance their roots as well as their tolerance to harsh soil conditions such as acidity, oxygen deficiency, and nutrient deficiencies, biochar application, and restore histosols.
Smith et al. 2008	4.9-5.34	50 – 100	Improved croplands mgmt., setaside, agroforestry	Methods for improvement include cropland management, grazing land management, restore cultivated organic soils, restore degraded lands, bioenergy (soil components), water management, and set-aside and agroforestry. An additional 0.6 Gt CO_2 eq yr^{-1} possible from offset of emissions through use of biofuels.
Sommer and Bossio 2014	2.5-5.1	87	Improved croplands and grasslands mgmt., setaside, agroforestry, restoration of degraded lands	Analysis includes estimates for soil carbon content for two different projections for future CO_2 emissions and for both arable land/permanent crops and permanent meadows/pastures from 2014 to 2100.

A2. Published estimates of potential for greenhouse gas removals from the atmosphere via changes in land management in the U.S.

Source	Estimates*	Duration, years	Measures	Region of US	Summary
Chambers et al. 2016	250 Mg CO_2 eq yr^{-1}	50	Improved cropland and grassland mgmt., setaside of marginal cropland to grassland	Lower 48	Estimates are based on widespread adoption USDA/NRCS conservation practices on all private lands.
Franzlubbers, 2010	1.5-1.8 Mg CO_2 eq ha^{-1} yr^{-1} 2.86-3.48 Mg CO_2 eq ha^{-1} yr^{-1}	Varied	Adopt conservation tillage Establish perennial pasture	Southeast	Reviews literature from Southeastern US to estimate soil carbon sequestration potential of degraded lands. Includes ample experimental data from 20 published studies.
Lal et al. 1998	275-639 Mg CO_2 eq yr^{-1}	50	Land conversion and setasides, restoration of degrade lands, improved cropland mgmt.	Not specified	Value corrected for error in the publication for C sequestration from residue management; also 'stacking' of multiple practices for the same land area increased total estimate
Lal et al. 2003	17.6-52.8 Tg CO_2 eq yr^{-1}	30	Adopt recommended mgmt practices on ag soils, grazing lands, restoration of degraded soils	Lower 48 and Alaska	Paper includes a breakdown of carbon sequestration in the continental US and Alaska by land type.
CAST. 2004	0.37-2.57 Mg CO_2 eq ha^{-1} yr^{-1}	10 – 30	Adoption of no till practices	Not specified	
Paustian et al. 2016b	500-800 Mg CO_2 eq yr^{-1}		Deployment of enhanced root phenotypes for major annual crops.	Lower 48	Estimate assumes 2 times root C input and a downward shift in root distribution equivalent to native prairie grasses. Excluded non-irrigated semi-arid cropland with major water limitation on production.
Song 2013	Soil C storage increasing from 31.2 kg CO_2 eq m^{-2} in	100	Changes in atmospheric CO2,	Southeast	This study is modeling changes in carbon storage and fluxes in the Southeast US under three climate scenarios using four different climate models. This does not include

Source	Estimates*	Duration, years	Measures	Region of US	Summary
	2000s to 35.6 kg CO_2 eq m^{-2} in 2090s.		nitrogen deposition, ozone		land use changes or active mitigation efforts, but instead takes into account environmental and climate changes by 2100. Study defined the SEUS as containing the following 13 states: FL, GA, NC, SC, VA, AL, AR, KY, LA, MS, OK, TN, and TX.
Sperow et al. 2003	305 Mg CO_2 eq yr^{-1}	15 and then dropping off over longer periods	Adoption of no till practices, improved agricultural soil mgmt.	Lower 48	Estimates are based on IPCC Tier 2 method and take into account changes in land management across the US. Includes elimination of summer fallow practices and using a winter cover crop. The rate is projected for 15 years, noting that this would decline over long periods of time.

*Mg = 1 Mt (metric tonne)
Tg = 1 MMt (million metric tonnes)

A3. Additional published estimates of greenhouse gas removal potential via changes in land management.

Source	Estimate	Duration, years	Measure	Region	Summary
Chan et al. 2009	18 Tg CO_2 eq yr^{-1}	Not specified	Improved mgmt. of croplands, rangelands, and grasslands	New South Wales	Estimate for New South Wales SOC sequestration in pasture land, cropping land and rangelands, which was equivalent to 11% of NSW's total GHG emissions in 2005.
Cole et al. 1997	7.33 Pg CO_2 eq for restoration of wetlands in India	50	Improved mgmt. for agricultural soils, setaside, restoration of degraded land	India	Cites Gupta and Rao (1994) who performed experiments in India on the restoration of salt and alkali impacted soils.
	1.61-3.23 Pg CO_2 eq yr^{-1}			World	Adapted from IPCC (1996) and are net storage rates for agriculture-based mitigation impacts.
Conant et al. 2001	0.40-11.1 Mg CO_2 eq ha^{-1} yr^{-1} with an average of 1.98 Mg CO_2 eq ha^{-1} yr^{-1}	40	Improved grassland mgmt. practices	World	This paper focused on improvement of grasslands to increase soil carbon sequestration. It looked at 115 studies with more than 300 data points. The first 40 years after treatment saw the highest rates of sequestration as well as in the top 10cm of soil. Results vary across biome and climate.
Dumanski et al. 1998	0.17 Pg CO_2 eq yr^{-1}	50	• Summer fallow replacement	Canada	Land use changes in Canada including reducing summer fallow area, improving erosion control, adoption of no-till practices, and changes in types of crops. The analysis incorporated both empirical data from long-term field experiments as well as the CENTURY model (version 4.0, 1993.)
Follett and Reed 2010	0.26-0.44 Mg CO_2 eq ha^{-1} yr^{-1}	40	Improved rangeland mgmt.	Colorado	Meta-analysis with rangeland management effects on soil carbon sequestration with empirical rates in different biomes with different techniques
	1.1 Mg CO_2 eq ha^{-1} yr^{-1}			Wyoming and North Dakota	
	5.87 Mg CO_2 eq ha^{-1} yr^{-1}			Kansas	

Source	Estimate	Duration, years	Measure	Region	Summary
	1.65-2.64 Mg CO_2 eq ha^{-1} yr^{-1} 1.21-5.72 Mg CO_2 eq ha^{-1} yr^{-1} 0.81 Mg CO_2 eq ha^{-1} yr^{-1} 7.0-10.1 Mg CO_2 eq ha^{-1} yr^{-1}			Saskatchewan South Dakota Kansas Argentina	
Follet and Schuman, 2005	0.73 Pg CO_2 eq m^{-2} yr^{-1}	40	Grazing land management only	World	Reviews climatic and land use factors that affect soil carbon sequestration and gives estimates for the potential of the world's global grazing land to sequester carbon. Also includes information about the retention of carbon in the soil and how policy will play a role in increasing SOC.
Jain et al. 2005.	2.14 Mg CO_2 eq ha^{-1} yr^{-1} with a total of 0.16 Pg CO_2 eq yr^{-1}	20	Adoption of no till practices	North America	Analysis was centered on adoption of no till from conventional plow tillage and the simulated uptake of CO2 in North America soils from 1981-2000. Model takes into account the 5 different climate regions across North America.
Kell 2011	367 Mg CO_2 eq ha^{-1} at a soil depth of 1-2 meters cumulative over 50 years	50	Breeding crop plants with deeper roots	N/A	This study looks at breeding plants with deeper root system to store carbon at the depth of 1-2 meters. Most of the other studies mentioned above only take into account the top meter of soil.
Kell 2012	733 Mg CO_2 eq ha^{-1} in the top metre of soil cumulative over 50 years	50	Breeding crops with improved below ground C sequestration traits	N/A	High potential for carbon storage not just in the topsoil layer, but the subsoils as well. There is evidence to support that soils have been depleted of carbon and at a bulk relative density of 1 percent C in the soil, 200 Mg ha-1 can be stored in just the top meter of soil alone. By extending root systems of agricultural crop, which are

Source	Estimate	Duration, years	Measure	Region	Summary
					generally not deeper than 1m, we can have the ability to store even more carbon in subsoil layers.
Mackey et al. 2013	6.6 Pg CO_2 eq yr^{-1}	100	Hypothetical restoration of land carbon buffer		Estimates based on the restoration of all cleared lands to their contemporary pre-agricultural state and carbon stock. Currently unachievable in our society.
McCalmont et al., 2015	1.54-14.0 Mg CO_2 eq ha^{-1} yr^{-1}	15 – 20	Growing *Miscanthus*, improvement of former arable land, grassland mgmt.	UK	Estimates are for the UK *Miscanthus* planted on former arable land. Crop can further reduce emissions when used as a perennial biomass crop up to 10% of the UK's current energy use.
Smith et al. 2000	0.12 Pg CO_2 eq yr^{-1}	50 – 100	Improved mgmt. of surplus arable land, improved land, improved mgmt. of agricultural soils	Europe	Soil carbon mitigation potential for land management practices that include no-tillage, straw incorporation, and organic amendments.
Smith 2012	5.5-6.0 Pg CO_2 eq yr^{-1} 0.2 Pg CO_2 eq yr^{-1}	15 – 20	Improved croplands/grasslands mgmt., agroforestry, setaside	World Europe cropland	5.5 – 6 represents global potential from agriculture by 2030 considering all greenhouse gases. Economic potential is much lower. 0.2 represents technical potential for Europe, but economic potential is much lower.
West and Post 2002	158-260 g CO_2 eq m^{-2} yr^{-1} 29-117 g CO_2 eq m^{-2} yr^{-1}	15 – 20 40 – 60	Changes from conventional tillage to no till Enhancement of rotation complexity	World	This analysis was conducted using a global database of 67 long-term agricultural experiments that compared the soil C sequestration rates of conventional tillage and no tillage practices in paired treatment groups. Also included estimates of SOC sequestration due to crop rotation.

A4: Estimates of soil carbon sequestration based on Earth System Models that take into account the changes in atmospheric CO2, temperature, and other green house gases. The estimates from ESMs do not take into account land management changes.

Source	Estimate	Duration	Measure	Summary
He et al. 2016	N/A	100	Incorporating radiocarbon restraint in ESMs	Analysis of 5 ESMs with radiocarbon restraint and found that ESMs estimates without were 40 +- 27% higher for soil carbon sequestration potential. This study indicates that we do not have a full understanding of the sequestration of carbon in soils on a long timescale.
Todd-Brown et al. 2014	-2.64-9.28 Pg CO_2 eq $yr^{-1}$with mean of 2.38 Pg CO_2 eq yr^{-1}	100	Changes in input of carbon to soil, changes in decomposition rate	Analysis from 11 Earth system models for the difference in 10-year means, 2090-2099 minus 1997-2006 soil carbon. Study does not specific what techniques specifically, i.e. land use changes contribute to the changes in soil carbon sequestration.