Assessment of potential greenhouse gas mitigation from changes to crop root mass and architecture

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Executive Summary

Reducing (and eventually reversing) the increase in greenhouse gases (GHGs) in the atmosphere due to human activities, and thus reducing the extent and severity of anthropogenic climate change, is one of the great challenges facing humanity. While most of the man-caused increase in GHGs has been due to fossil fuel use, land use (including agriculture) currently accounts for about 25% of total GHG emissions and thus there is a need to include emission reductions from the land use sector as part of an effective climate change mitigation strategy. In addition, analyses included in the recent IPCC 5th Climate Change Assessment report suggests that it may not be possible to achieve large enough emissions reductions in the energy, transport and industrial sectors alone to stabilize GHG concentrations at a level commensurate with a less than 2°C global average temperature increase, without the help of a substantial CO_2 sink (i.e., atmospheric CO_2 removal) from the land use sector. One of the potential carbon sinks that could contribute to this goal is increasing C storage in soil organic matter on managed lands.

There are numerous land management practices that can be adopted to increase soil carbon storage in agricultural soils (e.g., changes in crop rotations, tillage, fertilizer management, organic amendments, etc.) which have been extensively reviewed and assessed in the scientific literature. 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, which generally contribute 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. An option that has not yet been widely explored is to modify, through targeted breeding and plant selection, crop plants to produce more roots, deeper in the soil profile where decomposition rates are slower compared to surface horizons, as an analogous strategy to increase soil C storage.

This report details a preliminary scoping analysis, to assess the potential agricultural area in the US – where appropriate soil, climate and land use conditions exist – to determine the land area on which 'improved root phenotype' crops could be deployed and to evaluate the potential long-term soil C storage, given a set of 'bounding scenarios' of increased crop root input and/or rooting depth for major crop species (e.g., row crops (corn, sorghum, soybeans), small grains (wheat, barley, oats), and hay and pasture perennial forages). The enhanced root phenotype scenarios assumed 25, 50 and 100% increase in total root C inputs, in combination with five levels of modifying crop root distributions (i.e., no change and four scenarios with increasing downward shift in root distributions). We also analyzed impacts of greater root production on the soil-crop nitrogen balance, from the standpoint of increased need for additional N inputs and consequences for increased N_2O flux, as well as potential impacts if more and deeper roots contributed to reduced N leaching. In the enhanced root phenotype scenarios, the implicit assumption was that increases in overall plant production could be achieved (e.g., through increased CO_2 assimilation, greater growth efficiency) without reducing the harvested yield – that is, we did not include potential leakage and land substitution effects from potential decreased crop yield in the analysis.

We found that around 87% of total US cropland (major annual crops plus hay/pasture land) had soils of sufficient depth and lacking major root-restricting soil layers to allow for crops with enhanced phenotypes. In general, the areas showing the largest potential soil C increases were in the northern tier of the Great Plains and Corn Belt agricultural area and in irrigated croplands in the western US, with smaller increases in C storage in the south-eastern US. Long-term soil C stock changes (i.e., change to a new equilibrium state) ranged from a 4% increase in stocks (0-2 m) with no increased root C inputs but a small downward shift in root distributions, to a 70% increase with no additional C inputs but with the deepest rooting scenario, to a 3.4 fold increase in soil C stocks with a doubling of current root C inputs and the deepest root distribution scenario (with annual crop roots having root distributions similar to that observed in some deep rooting perennial grasses). Changes to a new equilibrium soil C state would take place over a several hundred year period (due to the long turnover time of some of the more recalcitrant soil organic fractions). However, a significant portion of the change would occur over the first few decades after new crop plants were introduced and we estimated that about 30% of the total change to a new equilibrium would be achieved in the initial 30 year period. Based on this calculation, average annual (averaged over the initial 30 yr period) soil C accrual rates (assuming 100% adoption of improved phenotypes) ranged up to 280 Tg C yr^{-1} (1026 Tg CO_2eq) for the most optimistic scenario of a doubling of root C inputs and an extreme downward shift in root distributions. This is equivalent to an average rate of increase of almost 1.8 Mg C ha^{-1} yr^{-1}, similar to rates of soil C increase that have been observed with conversion of annual cropland to high productivity perennial grasses.

Including impacts on the soil N balance, reduced somewhat the total GHG mitigation potential of some of the improved root phenotype scenarios, due to increased demand for N inputs and hence increased GHG emissions from N_2O and from embodied GHG emissions associated with fertilizer manufacture and distribution. Many cropland soils currently have a surplus N balance and thus for modest changes in root inputs and depth distributions, there is sufficient surplus N to meet the increased N demand and thus little impact on overall net GHG benefits due to altered N budgets. However, with increasing soil C storage as a consequence of greater root C inputs and deeper root distributions, the role of N in the overall GHG balance increases. For the most optimistic scenario of doubled root C inputs and an extreme downward shift in root distributions, total net GHG benefits were reduced by up to 28% (from 1026 to 746 Tg CO_2eq) due to the increased N_2O emissions associated with increased N inputs. However, a more detailed analysis using dynamic process-based models that couple plant and soil C and N fluxes, including N_2O, other gaseous N losses and N leaching is needed to better evaluate net GHG consequences.

In addition to helping meet GHG mitigation goals, changes in crop root production and root distributions that increase soil organic matter stocks, can provide a wide range of other benefits to soil health and sustainability, including improved soil physical characteristics, nutrient storage and cation retention, improved water retention and water quality and enhanced soil biodiversity.

Introduction:

Soils constitute the largest terrestrial organic C pool, estimated at ca. 2400 Pg C to 2 m depth (Batjes 1996), which is three times the amount of CO_2 currently in the atmosphere (~830 Pg C) and 240 times current annual fossil fuel emissions (~10 Pg) (Cias et al. 2013). The primary C exchange between the atmosphere and terrestrial ecosystems is via uptake of CO_2 through photosynthesis and C fixation into plant biomass (as a CO_2 flux **from** atmosphere) and the release of CO_2 from previously plant-fixed C through plant and soil microbial respiration (as a CO_2 flux **to** the atmosphere). Changes in terrestrial C stocks are given by the balance between these two fluxes. Hence if the soil C balance can be manipulated through management by even a few percent, this would represent a significant greenhouse gas (GHG) mitigation potential.

Fundamentally, soil C stocks can be increased by increasing the rate of C additions through plant residues (and other organic amendments) and/or by reducing the specific rate of decomposition of organic matter already present in the soil (Paustian et al. 1997b), both of which can be achieved to varying degrees through a variety of practices on managed (e.g. agricultural) soils (Paustian et al. 2016, Smith et al. 2015). Most of the organic carbon in soils is derived from roots (as exudates and through root death and turnover) (Rasse et al. 2005) and thus a potentially significant means to increase soil C stocks would be to develop crops and trees which allocate a greater proportion of their fixed C into the soil and/or that have deeper root systems where the decomposing organic C compounds would have a longer mean residence time (Kell 2012). If implemented widely over a large fraction of the global area of managed soils, the mitigation potential could be substantial – however, no formal analyses of such scenarios have yet been published.

In this report we present results from a preliminary 'bounding analysis' of potential atmospheric CO_2 removals from cropland and pasture/hay land in the US, given a set of scenarios for the adoption of crop species developed to produce more and deeper root C inputs. We also evaluated potential consequences for impacts on nitrogen (N) use and N use efficiency and N_2O emissions as well as provided supplementary information on impacts of increasing soil organic matter (SOM) stocks on soil health. The analysis builds on an extensive set of geospatial databases on climate, soil and management practices that determine the regional variability in ecosystem and land use practices that affect soil C and N dynamics and thus CO_2 and N_2O source/sink values. These data are used with a process-based model (Century) to estimate soil C sink potential using an equilibrium-based approach, adjusted for realized changes over a 30 year time horizon, for a set of scenarios specifying relative increases in root residue C inputs to soils and/or changes in the depth distributions of roots. Impacts of the projected soil C changes on soil N interactions, including potential changes in soil N_2O emissions, are also assessed using a data-driven soil N mass balance approach and empirical models of N_2O flux as a function N fertilizer and manure inputs. Scenarios for projected increases in root production and residues and for deeper root distributions, were developed in consultation with ARPA-E program managers and reflect biologically-based potentials, that range from no change in current conditions to changes deemed at the 'upper limit' of current biological and technical capacity.

Methods:

Task 1: Determination of 'feasible' area and stratification of land area by constraining factors for changed plant root profiles.

A detailed spatial analysis of the 48-conterminous land area of the US was done to stratify the potential land base for the deep soil carbon analysis using soils, land use, cropping history and land ownership data. For ownership, all non-Federal lands were included in the initial stratification.

National Cooperative Soil Survey data in the 2015 SSURGO database were used to identify soil depth and soil texture (sand/silt/clay fractions). Each of 37.5 million SSURGO soil polygons was processed independently. Soil profiles were divided into five depth layers: 0-20cm, 20-50 cm, 50-100 cm, 100-150 cm and 150-200cm. The dominant SSURGO soil texture were assigned to each layer in the profile. Shallow soils, i.e., less than 50 cm deep, were excluded from the analysis.

The 2011 National Land Cover Dataset (NLCD) was used to identify the dominant land use associated with each soil polygon. Land classified as Cultivated Crops or Pasture/Hay was included in the analysis (all other land cover/land uses, e.g., forest lands, urban lands, rangeland) were excluded. Specific vegetative cover was identified using USDA Cropland Data Layer (CDL) data for 2010 – 2014. The CDL identifies vegetative cover on an annual basis at a ca. 50 meter resolution. The dominant CDL vegetation was assigned to each soil polygon for each of the five years in the analysis.

Representative (dominant) crops were chosen as components for the cropping sequences, determined from the CDL time series, for computing average crop residue inputs (both aboveground and belowground) for the soil C modeling. The crops were corn, soybean, winter wheat, grass hay, alfalfa and summer fallow (i.e., no crop). For soil organic carbon (SOC) calculations, the primary differences among these crops are root architecture and carbon inputs (biomass production and residues). Crops with similar growth patterns and biomass were grouped together. For example, sorghum was included with corn, and small grains such as barley and oats were grouped with winter wheat.

Soil carbon values were calculated on a soil, depth, crop, year and scenario specific basis (as described below). For deep soils with SSURGO reported depths of at least 150cm, we assumed a potential rooting depth of 200cm. For soils with depth 50-150cm, SOC stock values were truncated at the appropriate layer. For example, a soil with a depth as reported in SSURGO of 120 cm was assigned SOC stocks from the top four layers of the profile, omitting SOC stocks from the 150-200cm layer.

After applying scenario specific models to each soil layer, SOC and carbon input values were summed for the profile to calculate the equilibrium SOC values (described below). To represent crop rotations, average plant residue inputs and SOC stocks for the 5 year CDL crop rotation period were calculated to arrive at an equilibrium SOC stock value for each polygon. Area-weighted SOC sums and rates were then calculated at county and national levels.

Tasks 2 & 3: Estimation of potential soil C stock changes with increased root carbon inputs (T2) and deeper root distributions (T3)

Overview

We computed plant residue C inputs for each of the dominant/representative crops in the analysis using county-level average crop yields reported by the National Agricultural Statistics Service (NASS) and allometric equations to estimate above-ground and below ground crop residue values for the current crop phenotypes and cropping systems in the US (i.e., baseline condition). Computed residue C inputs were then used to drive the steady-state Century model analysis. The increased root C input scenarios changed only the root fraction of plant residue production – aboveground harvest residues (e.g., leaves, straw) were held constant at baseline levels

We used the Century model (Parton et al. 1987, 1994), with an analytical solution of steady-state conditions (Paustian et al. 1997a, Ogle et al. 2012) to estimate steady-state soil organic C stocks for the soil-climate-crop combinations determined in the spatial analysis (described above). Calculations were done first for the baseline condition (i.e., current crop residue input rates) and then for scenarios of 25, 50 and 100% increase (above the baseline) in the annual rate of belowground (i.e., root-derived C inputs). In the second phase of the analysis, we used empirical-based root C distributions for each of the modeled crop species to partition C inputs to five depth increments: 0-20cm, 20-50 cm, 50-100 cm, 100-150 cm and 150-200cm. Empirically-based decay rate adjustment factors were derived to reflect decreased specific decomposition rates at depth due to lower temperatures, reduced O_2 and less active microbial populations. For the baseline condition, we used depth distributions for current crop phenotypes and then for the 'deeper root' scenarios we perturbed the root distribution function to represent a range of plausible conditions, based on measured characteristics of deeper rooting plants.

Finally we did a series of long-term transient simulation runs for a distributed set of sites across the US to estimate the proportion on the total change between baseline and *equilibrium* soil C stocks for the scenarios that would be reached over different time intervals. We chose a 30 year projection time and calculated the percentage of the total soil C equilibrium shift achieved over 30 years and annualized SOC gains to integrate with N_2O emissions (Task 4) to derive total GHG mitigation in CO_2eq terms.

A. Derivation of crop residue inputs

Plant residue C inputs were estimated from county-average NASS yields (averaged for 1980-2009) and allometric equations (which relate plant C allocation to above-ground vs below-ground plant parts and, within above-ground biomass, the fractions allocated to vegetative biomass vs grain) for each of the crop types using methods developed for the Intergovernmental Panel on Climate Change national GHG (De Klein et al., 2006). More details on the C input estimation for individual crops are given in Ogle et al. (2012).

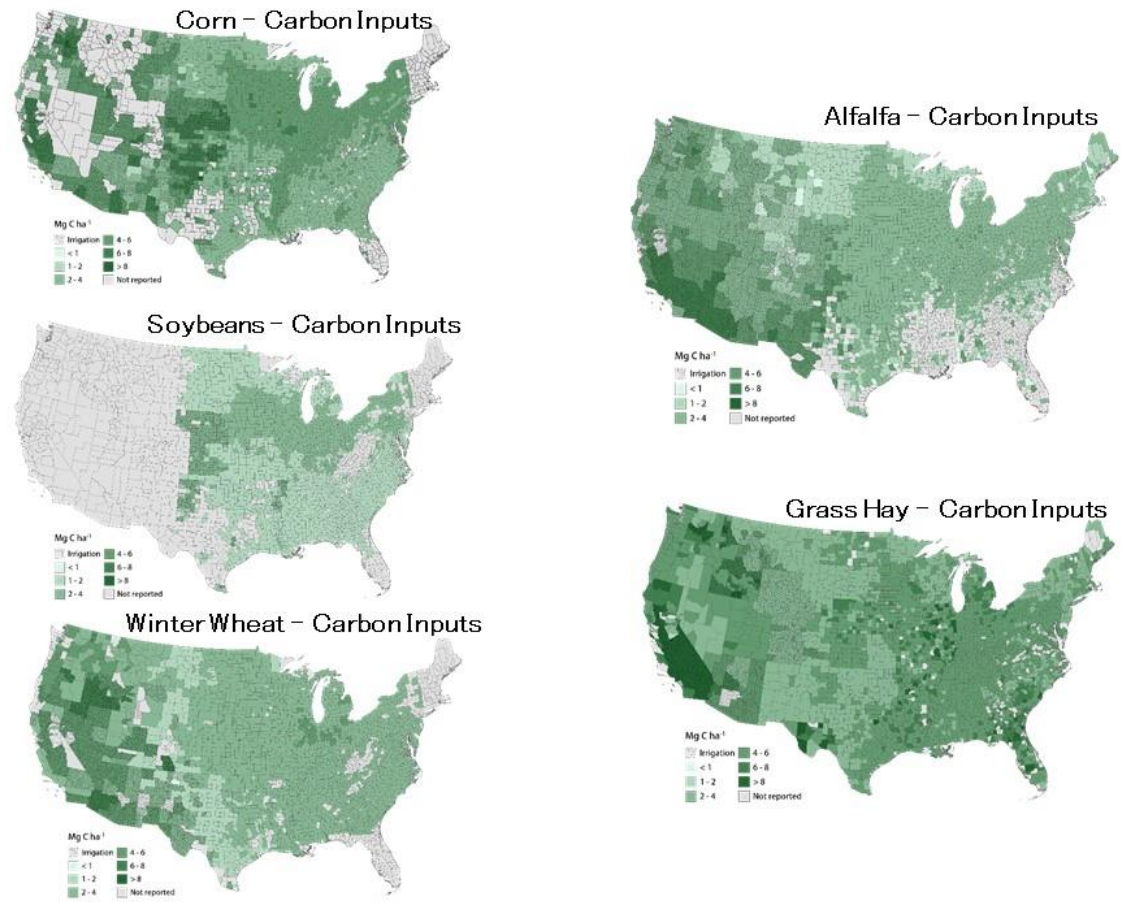


Fig. 1. Average county-scale baseline plant C inputs (above-ground + below ground) for each major crop type.

To compute average C inputs for each crop X soil type combination, inputs for each crop type in the 5-year CDL time series (i.e., representative crop rotation) for each overlay polygon were averaged as inputs to the model.

B. Root distribution and depth impacts on decomposition rates

Several factors give subsurface soil layers (i.e. deeper than 20 – 30cm) the potential to increase soil C storage with increases in C inputs. First, root inputs and soil organic matter are typically concentrated in the surface soil layer (i.e. from 0 – 20 or 30 cm), often showing a quasi-exponential decline with depth (Trumbore *et al* 1995, Torn *et al* 2002, Gill *et al* 1999). Soils in deeper depths have a lower SOC concentration relative to surficial layers and therefore often have a greater physical capacity to stabilize and retain increased C inputs, as compared to surface soils where C stabilization capacity may already be saturated (Six *et al* 2002, Stewart *et al* 2007). Second, there is evidence that organic matter in subsurface soil environments—with lower temperatures, reduced aeration, and less active and smaller microbial populations—have longer turnover times (Paul *et al* 1997, Trumbore *et al* 1995). Thus soil organic matter decays more slowly in deeper soils, with a potential for longer-term soil C stabilization. Combining these factors, management practices that increase root biomass in deeper soil layers may have the potential to yield long-term increases in soil C stocks.

The default parameterization for the organic matter decay rate constants in the Century model is set for the top 20-30 cm soil depth. To account for reduced organic matter decay rates deeper in the profile, we derived values for the relative decrease in decay constants as a function of depth from the deep soil version of the RothC model (Jenkinson and Coleman 2008), which along with Century is one of the most widely used ecosystem soil C models. The scaling factor N applied to each decay constant in a given layer was calculated as;

$$N = \frac{\frac{-1}{(1 + e^{(-s(F-f)})}}{\frac{-1}{(1 + e^{(-s(-f)})}} \qquad \text{Eq 1}$$

where "s is a constant, in cm^{-1}, F is the distance, in cm, from the middle of the top layer to the middle of the layer in question, and f is the distance in cm, from the surface to the middle of the top layer" (Jenkinson and Coleman 2008, pg 403). In the Jenkinson and Coleman (2008) analysis, s was a fitted parameter that ranged from -0.08 to -0.04. In our analysis, we used a value of -0.04 to scale decay constants in each soil layer from 0 - 100cm. Jenkinson and Coleman (2008) only evaluated decay adjustment factors down to 97 cm depth. We therefore used the value calculated with Eq 1 for the 50-100 cm layer for the depth increments below 100 cm. Information was lacking in the literature for decay dynamics below 100 cm. Thus we choose to use a conservative assumption of no additional reduction in potential decay rates in the 100 - 200cm soil layers (Table 1).

Table 1. Inputs and resulting calculated N scaling factors used to adjust decay constants in each soil layer.

Soil Layer	**Top (cm)**	**Bottom (cm)**	**F**	**f**	**s**	**N**
1	0	20	0	10	-0.04	1
2	20	50	35	10	-0.04	0.449
3	50	100	75	10	-0.04	0.115
4	100	150	na	na	na	0.115
5	150	200	na	na	na	0.115

To estimate root biomass distribution across depths, we used an exponential decay function. We calculated the percentage of root biomass inputs L in each layer i as:

$$L_i = \left(\left(1 - e^{(-p*B_i)}\right) - \left(1 - e^{(-p*T_i)}\right)\right) \qquad \text{Eq 2}$$

where B_i is the bottom, in cm, of layer i, T_i is the top, in cm, of layer i, and p is a constant that determines how uniform (i.e. when p is closer to 0) or skewed towards the surface layer (i.e. when p is closer to 1) root biomass is allocated across the soil profile. The exponential decay function is truncated at 200cm. The truncated remainder R was calculated as

$$R = 1 - \sum_{i=1}^{5} L_i \qquad \text{Eq 3}$$

The truncated remainder R was then allocated evenly across the soil profile. Total percentage root biomass G at layer i was calculated as:

$$G_i = L_i + (R * M_i)$$

Eq 4

where M_i is the percentage of the truncated remainder allocated to layer i. We chose to evenly distribute the truncated remainder across the soil profile, with 10% in layer 1, 15% in layer 2, and 25% each in layers 3 – 5.

To fit values of p for baseline conditions of crop-specific root biomass depth distributions, we assembled measured root biomass distribution data for five plant types: corn, soy, wheat, alfalfa, and grass/hay. Corn and soy data were drawn from a review of literature completed by NREL research associate S. Williams (for Table 11.2 in Intergovernmental Panel on Climate Change Guidelines for National Greenhouse Gas Inventories (2006)). Wheat data were drawn from the S. Williams review and from Slobodian *et al* (2002), while alfalfa data came from Abdul-Jabbar *et al* (1982), and grassland/pasture data came from Nippert *et al* (2012) and Gill *et al* (1999). We assembled data on deep-rooted tropical grasses from Saraiva *et al* (2014) and Paciullo *et al* (2010), as 'model' plant types with more uniform deep root distibutions through the soil profile. We then wrote a one dimensional optimization analysis in R (R Core Team 2014), using the optimize function (Brent 1973), to fit a baseline value of p for each plant type (Table 2).

Table 2. Baseline values of p for each plant type considered in this analysis.

Crop Type	*p*
corn	0.085
wheat	0.0695
soy	0.043
alfalfa	0.0332
grass/hay	0.0296
Model deep roots	0.0135

Fitted values of p were used to simulate baseline root growth across the soil profile (Figure 3).

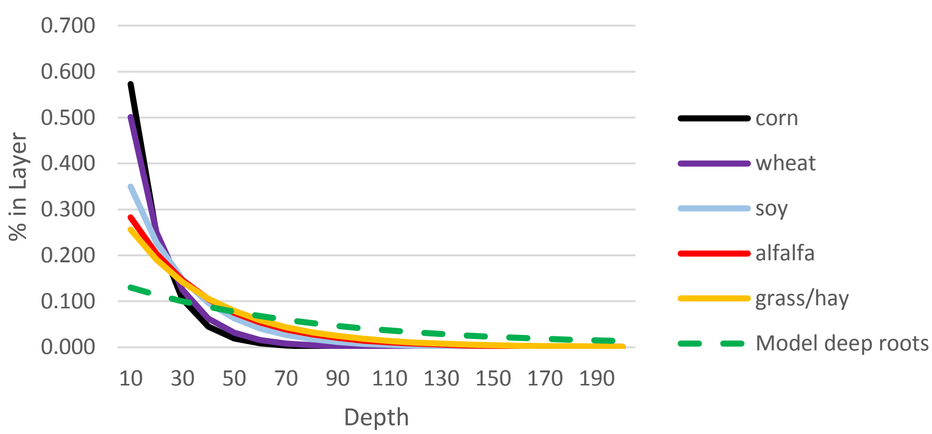


Fig.2. Baseline root distributions for all plant types included in this analysis, using parameter values from Table 2.

We then developed four hypothetical altered root growth scenarios:

1) ***'5% moderate adjustment'***: re-parameterized p values using adjusted measured root biomass distribution data, such that an additional 5% of biomass in each layer was moved to the next deeper layer

2) ***'20% moderate adjustment'***: re-parameterized p values using adjusted measured root biomass distribution data, such that an additional 20% of biomass in each layer was moved to the next deeper layer
3) ***'strong adjustment'***: all crops changed to grass/hay p value (from Table 2), grass/hay changed to Model deep roots p value (Table 2)
4) ***'extreme adjustment***: all crops and grass/hay changed to Model deep roots p value (Table 2)

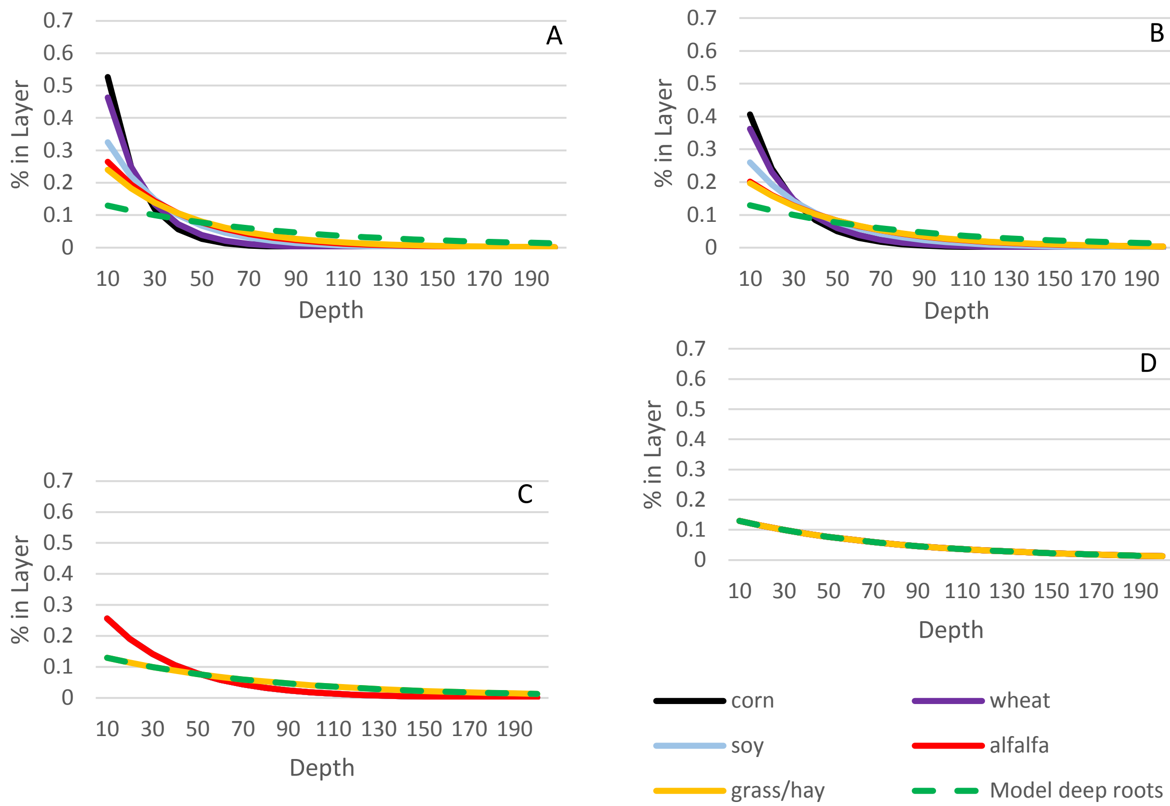


Fig. 3. Root distributions for scenario 1 (A), scenario 2 (B), scenario 3 (C) and scenario 4 (D), showing percentage allocation of root biomass in each layer to 200cm.

We determined adjusted values for p (see Table 3) and used them to simulate root distributions that reflect these scenarios (Fig. 3).

Table 3. Re-parameterized p values for hypothetical root growth scenarios.

Plant Type	Scenario 1-Moderate 5%	Scenario 2-Moderate 20%	Scenario 3-Strong	Scenario 4-Extreme
Corn	0.0747	0.0522	0.0296	0.0135
Wheat	0.0622	0.045	0.0296	0.0135
Soy	0.0393	0.0301	0.0296	0.0135
Alfalfa	0.0307	0.0244	0.0296	0.0135
Grass/Hay	0.0274	0.0218	0.0135	0.0135

D

C. Century model analysis of steady-state soil C stocks

To estimate the potential long-term impact of changes in root C inputs on SOC stocks we used a steady-state solution (Paustian et al. 1997a) to the residue and SOM pools represented in the Century model (Parton et al. 1987).

The steady-state solution for total soil organic carbon as represented in Century (Parton et al. 1987) and previously derived by Paustian et al. (1997a) is shown in Eq. 5,

$$X_{tot} = I\left\{\frac{\beta}{k_1} + \frac{(1-\beta)}{k_2}\left[\frac{1}{k_3} + \frac{f_4}{k_4} + \frac{f_5 + f_4 f_6}{k_5}\right]\alpha + \left[\frac{1}{k_4} + \frac{f_6}{k_5}\right] f_3 \lambda (1-\beta)\right\} \quad \text{Eq 5}$$

$$\alpha = \frac{f_1\beta + [f_2(1-\lambda) + f_3\lambda\,(f_7 + f_6 f_8)](1-\beta)}{(1 - f_4 f_7 - f_5 f_8 - f_4 f_6 f_8)} \quad \text{Eq 6}$$

where,

X_{tot} is total soil organic carbon, which is the sum of the metabolic (X_1), structural (X_2), active (X_3), slow (X_4) and passive (X_5) C pools;

I is the C (residue) input rate;

$k_{1,2,3,4,5}$ are the specific decay rates for the metabolic, structural, active, slow and passive pools, respectively;

f_1 and f_2 are the stabilization efficiency for metabolic and structural decay products, respectively, entering the active pool;

f_3 and f_4 are the stabilization efficiency for structural pool and active pool decay products, respectively, entering the slow pool;

f_5 and f_6 are the stabilization efficiency for active and slow pool decay products, respectively, entering the passive pool;

f_7 and f_8 are the stabilization efficiency for slow and passive pool decay products, respectively, entering the active pool;

β is the metabolic fraction of residue input (which is an empirical function of residue lignin (Λ) to nitrogen ratio);

Λ is lignin content of residue;

λ is the lignin to structural ratio of litter input [=Λ/(1- β)].

Most of the parameters in the model (e.g., maximum specific decay rates, stabilization efficiency partitioning parameters) are constants and the default values in the model were used. Actual decay constants for each pool are modified by temperature and moisture conditions and management conditions, and we used the same data and assumptions for county-scale values as described in Ogle et al. (2012). For estimating soil C stocks as a function of depth we included the depth adjustment factors described above (under ‘B. Root distributions and depth impacts’) as multipliers on the climate-adjusted specific decay rates (k_i) for each SOM pool. The main variables needed as model inputs were the residue C inputs, described above, and average values for crop residue lignin and N contents, estimated from literature values.

Following a significant perturbation (such as a sustained increase in C inputs), the trajectory of SOC stock change towards a new equilibrium condition follows a roughly hyperbolic shape. In Century (and most other SOC models), based on first-order kinetics, the approach to a new equilibrium state following a change in C inputs is towards an asymptote (strictly speaking, 'true equilibrium' is never reached). Since a significant portion of SOM is hundreds to thousands of years old, changes in this SOM fraction (represented by the 'Passive' pool in the Century model) occur very slowly. Thus to estimate the proportion of the total equilibrium C shift that would be attained over a finite time frame that was relevant for policy purposes, we conducted a series of dynamic Century model simulations runs, starting at equilibrium under current baseline conditions and then simulated a constant rate of organic matter addition (proportional to average annual simulated C inputs) for several thousand years until the model closely approached a new equilibrium condition. Sites chosen were well characterized long-term field experiments distributed across the major agricultural regions of the US. From these results we derived estimates of the percentage of the equilibrium C differential achieved as a function of time. For this analysis we chose a 30 year future projection period to calculate soil C change over that time horizon.

Task 4. Impacts on soil N and N_2O emissions of more and deeper roots

If crop plants are developed to produce more biomass (in this study, the assumption is that aboveground production remains constant, with the increase allocated to root biomass), additional N (and other plant nutrients) will be needed to support the increased plant N demands. Moreover, if increased root biomass (and deeper root distributions) lead to increases in soil organic matter stocks, N will be incorporated into SOM in roughly a fixed proportion relative to C. In fact, mineral soils under agricultural land use have a remarkably stable C/N ratio of around 10, which holds across a wide range of climate and soil types. Since, over the long-term, the only significant persistent stocks of C and N in cropland ecosystems are soil C and N (i.e., plant biomass is transient), accrual of ecosystem N stocks will be largely governed by the characteristic C/N ration of SOM. Thus, assuming a C/N ratio of 10 for soil organic matter, every 10 units of soil C sequestration will carry with it 1 unit of soil N sequestration. Depending on the N balance of the system, this ecosystem N demand could be met by surplus mineral N that currently exists in the system. Alternatively, it might require increased external N inputs which could increase N_2O emissions as well as embodied GHG emissions from fertilizer production and thereby offset some or all of the GHG reductions achieved through C sequestration. Finally, deeper more extensive root systems might also reduce N leaching losses and convert part of those losses into increased soil organic nitrogen (SON) storage.

To assess impacts of increased root production and deeper root distributions on the agroecosystem N budget and potential changes in N2O fluxes, we used a simple mass balance approach to estimate surplus N available under current conditions and a range of scenarios for potential reductions in N leaching losses due to deeper and more extensive root systems.

<u>A. N balance estimates</u>

Baseline nitrogen budgets for five crops (corn, wheat, soybeans, alfalfa, and grass hay) were generated using nitrogen input (Table 4) and output (Table 5) data sources along with a few other adjustments. Data was filtered so that only counties with > 5,000 ha of the crop were included in the analysis. Occasionally, NASS data on crop area has issues when the cropland area or number of farms reporting are small (e.g., < 5K ha) and thus counties under this threshold were removed for this analysis. To calculate average annual N balances for representative crop rotations, crops were area-weighted at the county level based on the NASS reported hectares within the county for each crop (using the same procedure described above to calculate average C inputs per unit area).

Estimates of average annual N inputs (Ninputs in Eq 7 below) were calculated, from a variety of data sources (Table 4), as the sum of additions through mineral fertilizer (Nfert), animal manure (Nomad), symbiotic N fixation (Nfix) in legume crops and atmospheric deposition (Ndep).

Table 4. Nitrogen input sources used for baseline nitrogen budgeting.

Input	Source, notes
Manure (Nomad)	Manure application rates that are used in the US EPA GHG inventory (US-EPA, 2010) were used in this report. The data used the 2000s manure fractions and, CN ratio, and application rates per crop that are generated at the county level by the Eastern Research Group[1].
Fertilizer (Nfert)	Fertilizer rates are based on the USDA–Economic Research Service Cropping Practices Surveys (USDA-ERS 2011), and are provided at a state level for each crop. Fertilizer application rates are provided for both irrigated and non-irrigated management scenarios for the crop. We used an area weighted average for fertilizer application based on percent of the crop irrigated in the county[2].
Nitrogen Deposition (Ndep)	National Atmospheric Deposition Program (NADP, http://nadp.isws.illinois.edu/) average nitrogen deposition for 2010-2012, area weighted to a county level average.
Yield	National Agriculture Statistic Survey (NASS) reported yield by county. Average yield (1980-2009) based on irrigated and non-irrigated acres within a county.
Fixation (Nfix)	Nitrogen fixation rates for soybeans and alfalfa were estimated using equations from peer-reviewed literature[2].
Irrigation	Fertilizer application rates were area-weighted based on the reported irrigation rate for the crop in order to apply the proper amount of fertilizer.

[1]Eastern Research Group, https://www.erg.com/ compiles manure and fertilizer application rates that are also used in the US GHG inventory.

[2]Soybean fixation is based on Salvagiotti et al. (2008) while alfalfa fixation is based on Anglade et al. (2015). Some grass hay area in the US GHG inventory includes grass-clover mixes that are used throughout parts of the US. Values of Nfix include N inputs for these grass-legume clover fixing areas where they occur.

B. Leaching and N_2O losses

Leaching of nitrate (NO_3^-) can occur where precipitation or irrigation water exceed evapotranspiration (ET) rates and hence a positive drainage flux is part of the soil water balance. We took a simple approach to approximate leaching, by assigning counties which fell into broad IPCC defined climate zones (which are partially defined by precipitation/potential ET ratio) – namely, zones 5 (warm temperate moist) and 7 (cool temperate moist) – as locations where drainage (and leaching) typically occurs, whereas areas in drier climates were assumed not to drain water below the root zone, unless they are under irrigation. IPCC climate zones where overlaid on US county delineations, and counties that were more than 50% within these zones were identified as counties were leaching was likely. In order to account for the potential leaching in drier regions for crops under irrigation, counties where irrigation was used for >50% of the crop acreage were also selected as counties were drainage and N leaching would be expected.

Table 5. Output source methodology used in nitrogen budget baseline.

Outputs	Source, notes
N_2O loss (N_2O)	IPCC estimated 1% of applied nitrogen (fertilizer and manure) as well as from fixed nitrogen is lost as direct N_2O. Indirect N_2O losses were not calculated.
Harvested N (HarvN)	Harvested N was calculated by multiplying the biomass yield of the crop by the nitrogen concentration of the harvested part of the crop. The nitrogen concentration was calculated by protein percent divided by 6.25 to get % nitrogen.
Leached N (Nleach)	IPCC climate zones were located in the US where leaching was likely, as well as irrigated counties that did not fall within these zones., A crop dependent fraction of nitrogen inputs and fixation N was estimated as leaching based on literature that is outlined below.

The amount of nitrogen leached per hectare is dependent on many factors including but not limited to: soil properties, precipitation and irrigation amount, nitrogen inputs, crop, and management practices, making it a difficult value to estimate. We were not able to attain any up-to-date spatially resolved estimates of leaching for the land base included in our analysis. Instead we used the IPCC Tier 1 approach for annual crops, supplemented with a separate literature-based estimate for hay crops. Thus for soils where there is normally a significant drainage flux, we assumed that on average 30% of nitrogen inputs are lost via leaching under annual crops (IPCC 1996). This is consistent with other work (e.g. Cardenas et al. 2013 suggested leaching rates of 28% for arable land and 9% for grasslands across the UK; Masarik et al. 2014 18-20% for corn in US prairie regions; Kucharik et al. 2003 suggested 31-44% for corn in Wisconsin, USA). In a review of the IPCC methodology, Nevison (2000) suggested 30% was a good estimate and pointed to six studies within the Midwestern US, planted with soybean and corn that suggested 20% of applied nitrogen leached. Based on the leaching value used as the IPCC standard (mean 30%, range of 10-80% of fertilizer/manure additions) and the other literature mentioned, we estimated leaching for corn, soybeans, wheat, and alfalfa at 30% of managed nitrogen inputs (manure and fertilizer). Due to the lower leaching rates for grasslands (Cardenas et al., 2013) we estimated leaching at grasslands at 10% of nitrogen inputs. Alfalfa and soybeans generate a lot of their N requirements through N_2 fixation (ca. 70% of total N requirements), which was not included in the 30% of N inputs lost as leaching. Due to generally larger and deeper roots for legume fixers and studies suggesting that N_2 fixation leads to lower nitrate leaching losses, we estimated leaching from fixed nitrogen at 10% of the amount of nitrogen fixed by perennial legumes (Crews and Peoples 2004).

Direct N_2O losses (kg N_2O-N ha^{-1} yr^{-1}) were calculated as 1% of annual N input from all sources, which is the default emission factor in the IPCC National Greenhouse Gas Inventory Guidelines (IPCC 2006). Indirect N_2O emissions (i.e., offsite losses from previously leached or volatilized and redeposited reactive N species) were not calculated. Embodied fertilizer GHG emissions were estimated using the mean of emission factors (3.93 kg CO_2e/kg N) for the two dominant N types in the US, anhydrous ammonia and urea (Johnson et al. 2013).

C. Nitrogen removals in harvest products

Each of the crops examined has a different protein concentration, resulting in different amounts of nitrogen harvested in grain or biomass. A percent protein to N conversion factor of 6.25 was used to calculate the nitrogen concentration (%N). If %N value was used more commonly than %protein, the %N value was used in place of the protein conversion equation. The crop specific characteristics that were used in order to inform the nitrogen harvest (HarvN) variable in the budget are outlined in Appendix A1.

D. Calculation of N surpluses

Based on the nitrogen inputs and losses outlined above, we calculated simplified nitrogen budgets, aggregated at county scale, using the following equation,

$$\text{envN} = \text{Ninputs} - N_2O - \text{HarvN} - \text{Nleach} \qquad \text{Eq 7}$$

where envN represents 'surplus N' not accounted for by harvest removal, N leaching and N_2O emissions. Thus it may include other gaseous losses (i.e., NO_X, N_2, NH_3) not accounted for, as well as N remaining in the field in unharvested crop residues and accumulated increases in soil N storage (as organic or inorganic N stocks). While some of this surplus N (envN) may be lost from the system as gaseous losses, particularly where surpluses are large and in moist environments (e.g., denitrification), in many cases these other gaseous losses are small (Liu et al., 2005) and much of the estimated surplus N could be available to support additional crop (root) assimilation and growth and incorporation into soil organic matter N.

E. Potential increased N demand and N_2O emissions

To calculate increased ecosystem N demand under the increased root scenarios, the ΔSOC for the 30 year projection period, was annualized (i.e., divided by 30), and then divided by the C/N ratio of 10. The difference between this additional N demand and current surplus N in the N balance (Nenv) is the additional N inputs (assumed to come from increased fertilizer and/or manure additions) required under the enhanced root scenarios. If the surplus (envN) exceeded the additional N requirement, no additional N inputs were needed.

We also included three specific N balance scenarios to assess the potential that deeper and more extensive root profiles could also reduce current N leaching rates and thus further conserve N already being added to the system, reducing the needed for increased external N inputs. Scenarios were 0 (i.e., baseline), 25, 50 or 75% reductions in current leaching rates for each of the enhanced root production scenarios. Thus, in calculating potential additional N inputs, both current surplus N and the reduced N leaching amounts were considered as available to meet part or all of additional N requirements.

Where additional N inputs were required to meet the higher N demand (accounting for N surplus and reduced leaching), increased N_2O emissions were calculated for the additional N inputs, assuming a 1% emission factor. To compute net GHG emission changes on a CO_2 equivalent basis, a GWP of 298 (adopted by USEPA for the US national GHG inventory) for N_2O was used.

Results:

Task 1: Determination of 'feasible' area and stratification of land area by constraining factors for changed plant root profiles.

The total land area meeting the criteria for the analysis was 156.6 million hectares, or 87.5 percent of cropland and pastureland in the conterminous United States. The remaining cropland contained ineligible soils or crops not included in the analysis (e.g., perennial crops such as orchards and vineyards as well as specialty and minor crops by land area).

Analysis of soil characteristics (Fig. 1) suggest that soil depth (either due to shallow bedrock or presence of severe root restricting layers) would be the main soil constraint on introduction of deeper root crop phenotypes. There is a very low incidence of highly acid subsoils (mainly occurring in highly weathered tropical soils) that could have chemical limitations (e.g. Al^{+++} toxicity). In some areas high water tables could be a constraint but in most cropland soils water tables during the growing season are likely to be below 2 m and/or are subject to control through existing artificial drainage systems. Lack of sufficient subsoil moisture could also be a constraint on rain-fed semiarid cropland, which we have not yet fully analyzed.

Considering only the soil depth constraint, only about 5% of the current cropland/pasture/hayland has potential rooting depth of < 50%, which we would consider 'not suitable' and about 8% have potential root depths of 50 to <100 cm, whereas 82% of the land area have soils with potential rooting depth of > 100 cm.

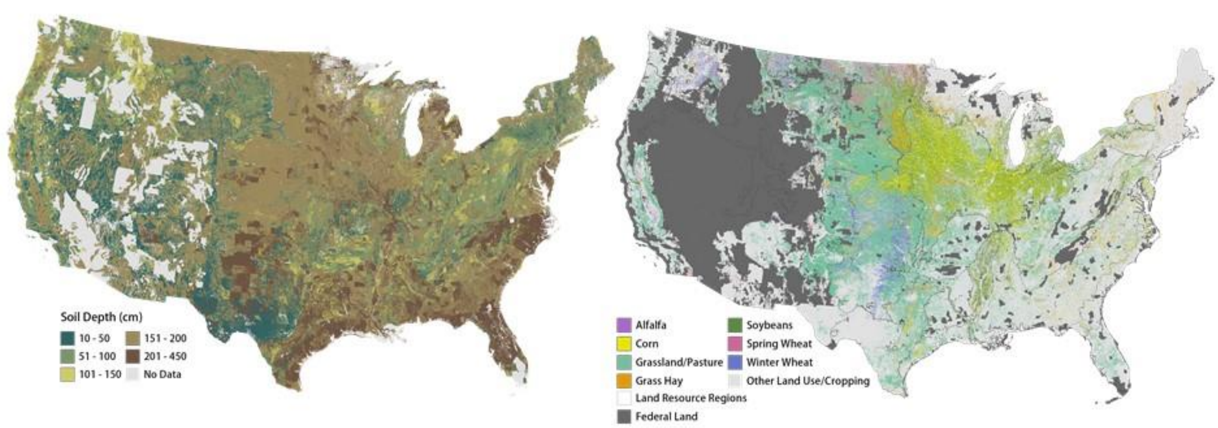
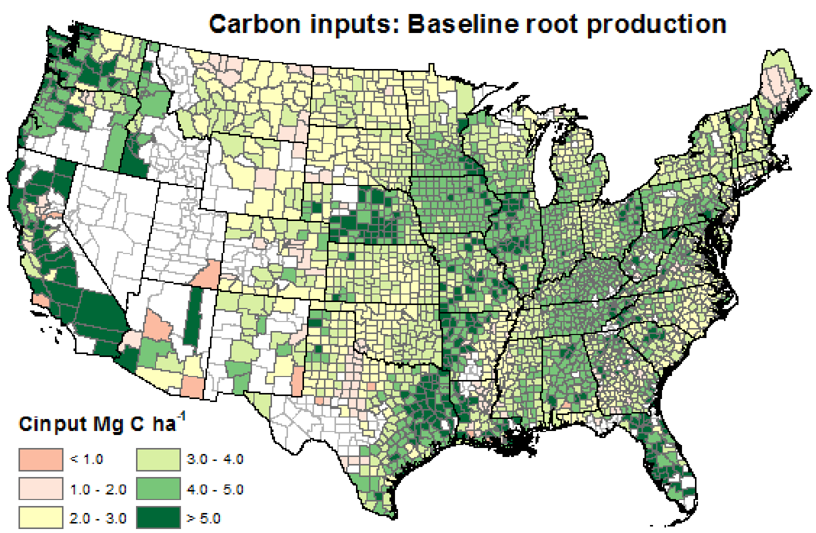


Fig. 4. Soil depth for potential rooting (derived from SSURGO) and distribution of major crop types (derived from CDL) used in the analysis.

Tasks 2 & 3: Estimation of potential soil C stock changes with increased root carbon inputs (T2) and deeper root distributions (T3)

Baseline average residue carbon inputs for crop rotations indicated by USDA-CDL are shown in Figure 5. The crop specific carbon input estimates shown in Figure 1 were applied to the CDL reported cropping histories for years 2010-2014 to derive an average annual carbon input estimate. Some crop histories include fallow years, with carbon inputs close to zero, which reduces average annual carbon inputs in regions dominated by wheat-fallow systems. The map also shows the increased plant carbon production due to irrigation in parts of the arid West.

Root growth scenarios included production increases of 25, 50 and 100 percent. The relative spatial distribution under these scenarios remains the same (given the assumption of uniform proportional increases), with the absolute amounts of carbon inputs increasing most in the eastern half of the US and in irrigated areas with high productivity in the western US. Under the 50% increased C input scenario, much of the cropland in the US, except for rainfed areas in the more arid western regions, would have average residue C inputs exceeding 5 Mg C ha^{-1} yr^{-1} (Fig 5).



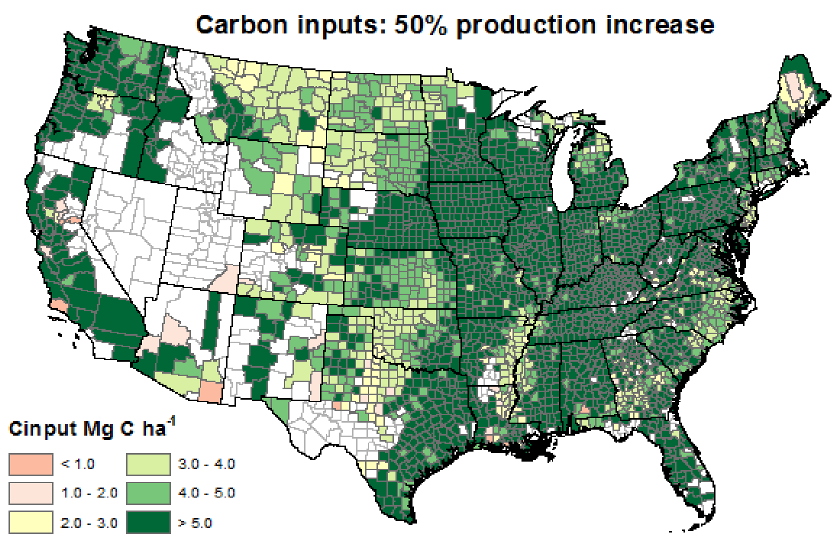


Fig. 5. Baseline average annual residue C inputs (top) based on CDL reported crop histories (2010 – 2015) and residue C under the 50% increased root production scenario (bottom).

The geographic distribution of potential equilibrium soil C stock changes for the baseline scenario and the scenario with a 50% increase in root production and a moderate downward shift in root distribution are shown below (Fig. 6). The highest per ha soil C change in equilibrium soil C stocks for cropland/pasture/hay land occurred in the areas with highest baseline root C inputs, including high productivity irrigated land in CA and the PNW, as well as through the northern tier of the Corn Belt and Northern Great Plains, where C inputs are relatively high, coupled with cooler temperatures (hence slower decomposition). Smaller differences in soil C stocks tend to be in the southeastern US which have higher rates of soil C turnover. Note, values shown are for per ha C stocks on agricultural land in each county and not per total land area nor adjusted for varying country size.

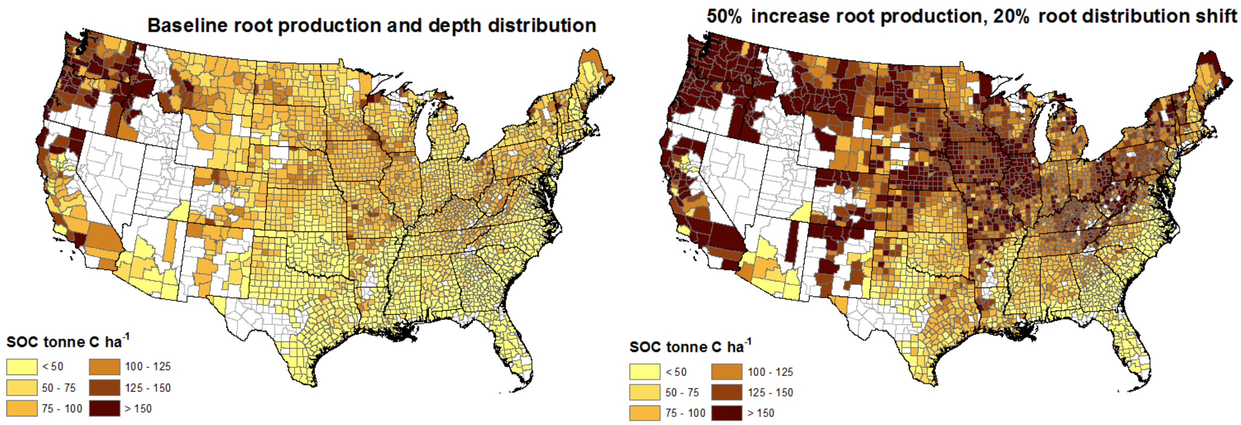


Fig. 6. Geographic distribution of steady-state soil organic C stocks (0-200 cm) on cropland and pasture/hay land under baseline (i.e. current) conditions and under a scenario for 50% increased root C inputs and moderately deeper root distributions.

The transient simulation runs to estimate the trajectory of ΔSOC were done for ten long-term experiment sites distributed across the US, in different climate zones and having different soil types. The percent of the total equilibrium change attained after 30 years ranged from 22 to 43% across the 10 sites, with a mean of 32%. The trajectory is dependent on factors affecting the mean residence time of the SOM pools in Century (Paustian et al. 1997a, which include climate, soil texture and residue quality (specifically the lignin/N ratio which varies by species). However, the fractional (relative) change per unit time is independent of the relative magnitude of the perturbation (Paustian et al. 1997a) and thus the percent change in SOC between equilibrium states for a given unit of time is the same for a 25, 50 or 100% increase in C inputs (example shown in Fig. 7). To best represent an average equilibrium approach per unit time, would require an extensive area-weighted sampling of a large combination of climate, soil and residue types, which was beyond the scope of this study. Thus we utilized the mean value of 32% in 30 years of the total equilibrium ΔSOC as a representative value and adopted a 30 year projection period for the scenario analysis.

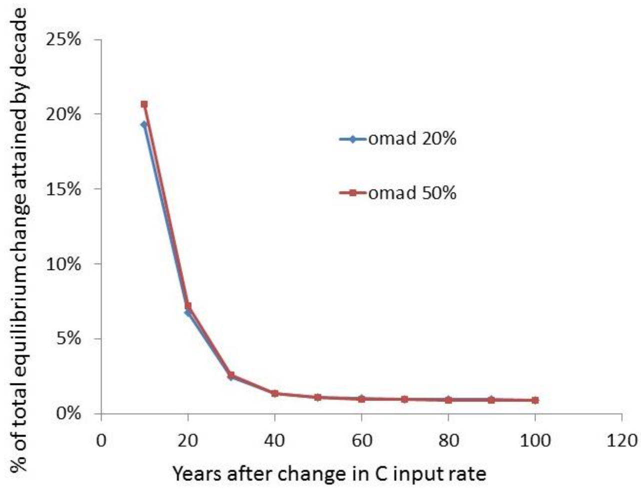


Fig. 7. Change in the relative rate of SOC accumulation (as a percentage of the difference between equilibrium soil C stocks) for the transition to a new equilibrium following a change in C input rates (+20% and +50% above baseline, respectively) for the Wooster, OH site. Simulations were run for 5000 yr – only the first 100 year period is shown here.

Table 6. Site characteristics for long-term sites analyzed for transient SOC dynamics following a fixed relative increase in residue C input rates. Percent of the total equilibrium SOC change attained after 30 years is given in the last column.

site	MAP (cm)	MAT (°C)	sand fraction	silt fraction	clay fraction	crop rotation	% of equilibrium SOC in 30 yr
Wooster,OH	88	9.5	0.19	0.57	0.24	continuous corn	28.6
Accokeek, MD	106	13.7	0.67	0.23	0.10	continuous corn	35.2
Lexington, KY	113	13.1	0.07	0.64	0.29	continuous corn	33.5
Mead, NE	68	11.6	0.05	0.60	0.35	corn-soybean	26.6
Lamberton, MN	67	7.6	0.36	0.33	0.31	corn-soybean	22.3
Watkinsville, GA	129	16.5	0.62	0.16	0.23	cotton-corn	30.1
Bushland, TX	50	14.7	0.17	0.53	0.30	fallow-winter wheat-sorghum	34.0
Akron, CO	40	9.3	0.29	0.46	0.25	fallow-w. wheat	40.1
Pendleton, OR	42	10.2	0.17	0.59	0.24	fallow-w. wheat	26.0
Mandan, ND	41	5.2	0.27	0.52	0.21	fallow-s. wheat	43.1
Mean							32%

Assuming 100% implementation of change scenarios on the cropland, pasture and hay land areas with suitable rating with respect to soil depth, average steady-state soil C stocks (0-200 cm) are shown for each of the root C perturbation scenarios (Table 7). Potential increases in equilibrium stocks range up to a 3.5 fold increase with a doubling of root C inputs and the most extreme downward shift in root distributions. More moderate scenarios with a 25-50% increase in C input rates and a more moderate shift in root distributions (5-20%) would increase steady-state stocks by approximately 50 to 100%.

Table 7. Average per ha SOC stocks (0-200 cm) for each of the altered root phenotype scenarios *at equilibrium*.

	Root depth distribution scenario				
	baseline	5% shift	20% shift	strong shift	extreme shift
Root C input scenario	Mg C ha^{-1}				
baseline	70.1	72.9	82.3	99.1	119.7
25% increase	87.7	91.1	102.8	123.8	149.6
50% increase	105.2	109.3	123.4	148.6	179.5
100% increase	139.6	145.0	163.7	197.1	238.1

Using the average estimate of 32% of the a independent DayCent simulations, we determined that 30 years from the start of the scenario SOC values had increased approximately 32 % of the total difference at equilibrium. Thus Table 8 shows the estimate of total annual SOC stock increase (relative to the baseline) over the 30 year projection period and Table 9 shows the same values but on an average per ha basis.

Table 8. Annual increase in total SOC stocks (0-200cm) *during the first 30 years* for each of the altered root phenotype scenarios.

	Root depth distribution scenario				
	baseline	5% shift	20% shift	strong shift	extreme shift
Root C input scenario	Tg C yr^{-1}				
baseline	--	4.5	20.3	48.3	82.7
25% increase	29.3	35.0	54.6	89.6	132.6
50% increase	58.6	65.4	89.0	131.0	182.6
100% increase	116.0	125.0	156.3	212.1	280.5

Table 9. Annual rate of increase in SOC stocks (0-200cm) on a per hectare basis *during the first 30 years* for each of the altered root phenotype scenarios.

	Root depth distribution scenario				
	baseline	5% shift	20% shift	strong shift	extreme shift
Root C input scenario	kg C ha^{-1} yr^{-1}				
baseline	--	29.1	129.4	308.4	528.1
25% increase	187.0	223.4	348.7	572.5	847.2
50% increase	374.1	417.7	568.1	836.7	1166.3
100% increase	740.7	798.5	998.1	1354.4	1791.7

Maps of the annual increase in SOC stocks during the first 30 years are shown in Figure 8.

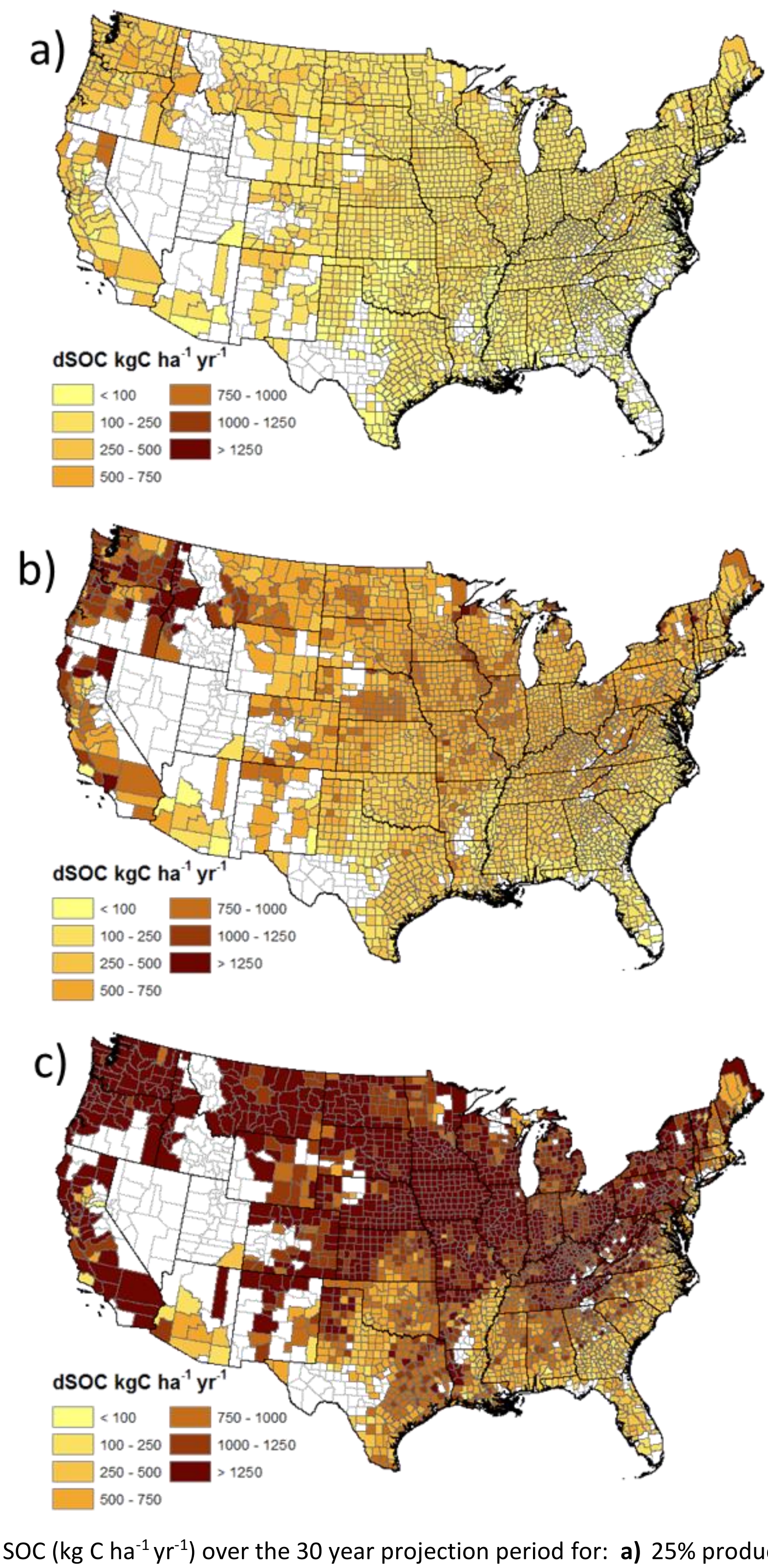


Fig. 8. Annual delta SOC (kg C ha^{-1} yr^{-1}) over the 30 year projection period for: **a)** 25% production increase, no shift, b) 50% production increase, 20% shift, **c)** 100% production increase, strong shift.

Task 4. Impacts on soil N and N_2O emissions of more and deeper roots

Per hectare components of the N budgets are shown in Fig 9. In the main crop growing areas for the five crops included in the analysis (corn, wheat, soy, alfalfa, grass hay), N inputs, harvest N removals and N leaching losses are all highest in the central tier states of the Corn Belt, stretching from central NE to OH. While there are high per ha N balance components shown in the southwestern US and parts of the Great Basin, these mainly reflect irrigated hay (grass and alfalfa) production. In these areas actual acreages under cropping are relatively small (note – the maps are colored by county, showing average per ha rates for the cropland in that county) and don't reflect the actual acreage of cropland nor area-weighted N fluxes present. Estimates of surplus N (i.e., not removed in harvest or through leaching) were highest in parts of New England, central Texas, and the mid-Atlantic coastal regional, mostly due to the high rates manure N (from dairy and feedlot operations). The majority of cropland in the eastern US, PNW and in irrigated parts of the intermountain west show N surpluses of > 25 kg ha^{-1} yr^{-1}.

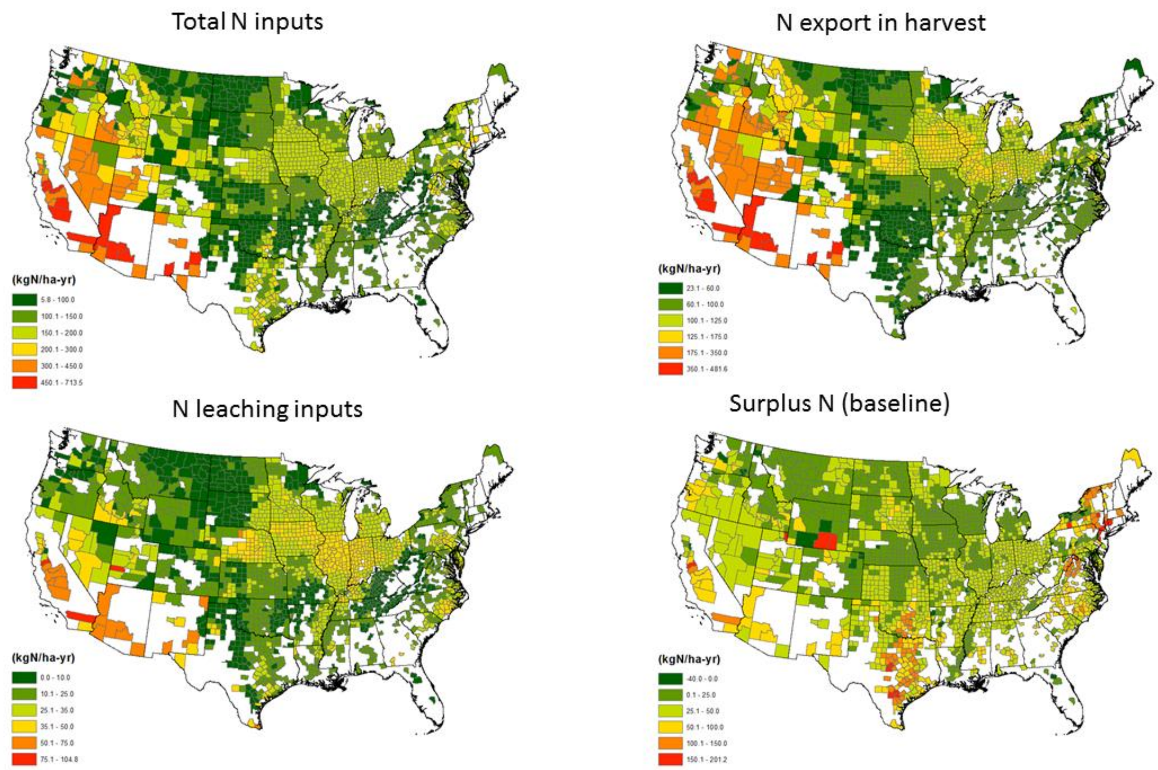


Fig. 9. Nitrogen balance components, showing average annual N flux rates for cropland included in the analysis. Values are in kg N ha^{-1} yr^{-1} for cropland in counties containing more than 10,000 acres in a county.

Depending on the scenario used (see Table 10), nationally-average N demands and N_2O emissions varied greatly but consistently for the scenario combinations. For the scenarios with modest changes in root production (<50%) and root architecture (20% downward shift), additional ecosystem N demands could be met by current 'surplus N' and negative values for additional N requirements suggests that current inputs would even be reduced (Fig. 10). For the scenarios, with a 50% increase or more in root biomass production, our analysis suggests that additional N inputs would be needed. Looking across changes in root architecture, as root distributions are shifted downward, resulting in increased SOM stabilization efficiency and greater SOC (and SON) accumulation, N requirements tend to increase monotonically. At the most extreme scenario, with a doubling of root C inputs and an extreme downward shift in root distributions, as much as 200 kg N ha^{-1} yr^{-1} of additional N inputs would be needed to support the projected ecosystem C & N accumulation, assuming a C/N ratio of 10 for SOM formation. Reductions in N leaching that could accompany changes in root architecture, decrease the needs for increased N addition somewhat (Fig. 10) relative to the baseline scenario (no leaching reductions).

Because direct N_2O emissions are proportional to N inputs, the pattern for additional N_2O emissions exactly track the changes in N requirements across the different scenarios (Fig. 10).

Table 10. Outline of C input and root distribution scenarios referred to in Fig. 10.

Scenario	Δ residue C input	Δ root distribution
P0_R0	No change	No change
P0_R5	No change	5% downward shift
P0_R20	No change	20% downward shift
P0_Rstr	No change	'strong' downward shift
P0_Rext	No change	'extreme' downward shift
P25_R0	+ 25%	No change
P25_R5	+ 25%	5% downward shift
P25_R20	+ 25%	20% downward shift
P25_Rstr	+ 25%	'strong' downward shift
P25_Rext	+ 25%	'extreme' downward shift
P50_R0	+ 50%	No change
P50_R5	+ 50%	5% downward shift
P50_R20	+ 50%	20% downward shift
P50_Rstr	+ 50%	'strong' downward shift
P50_Rext	+ 50%	'extreme' downward shift
P99_R0	+ 99%	No change
P99_R5	+ 99%	5% downward shift
P99_R20	+ 99%	20% downward shift
P99_Rstr	+ 99%	'strong' downward shift
P99_Rext	+ 99%	'extreme' downward shift

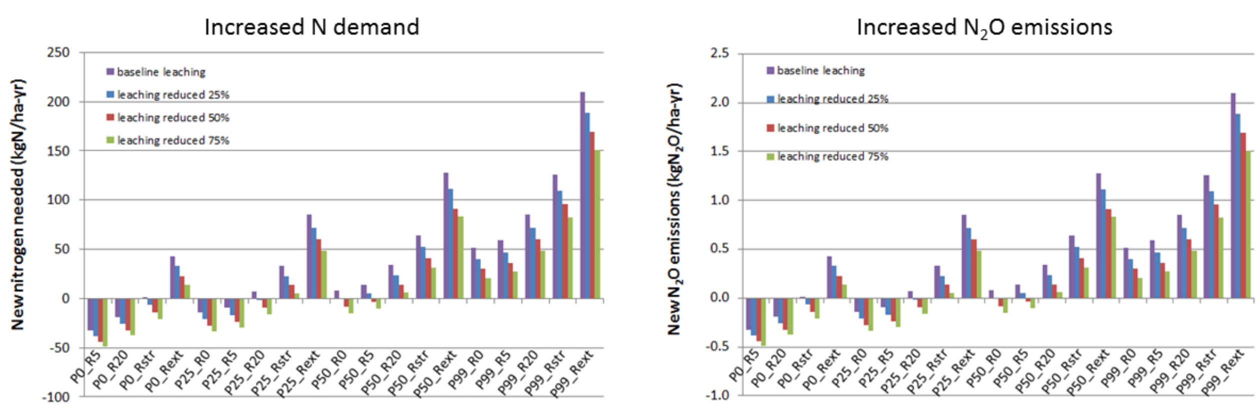


Fig. 10. Change in annual nitrogen demands (left) and N_2O emissions (left) over the 30 year (right) equilibrium scenarios.

Geographically, much of the increased N_2O emissions (under scenarios where surplus N is insufficient to meet increased N requirements with more and deeper roots) is predicted to occur in central Corn Belt states, which is where the lowest N surpluses relative to crop needs are mainly located. Baseline emissions (i.e., without increased root production) are show for comparison. If leaching losses are reduced (compare 25 vs 50% reductions shown in Fig. 11), emission rates decreased in the Corn Belt region as well as in intensive cropped regions in central TX, the Mississippi Delta and the mid-Atlantic region.

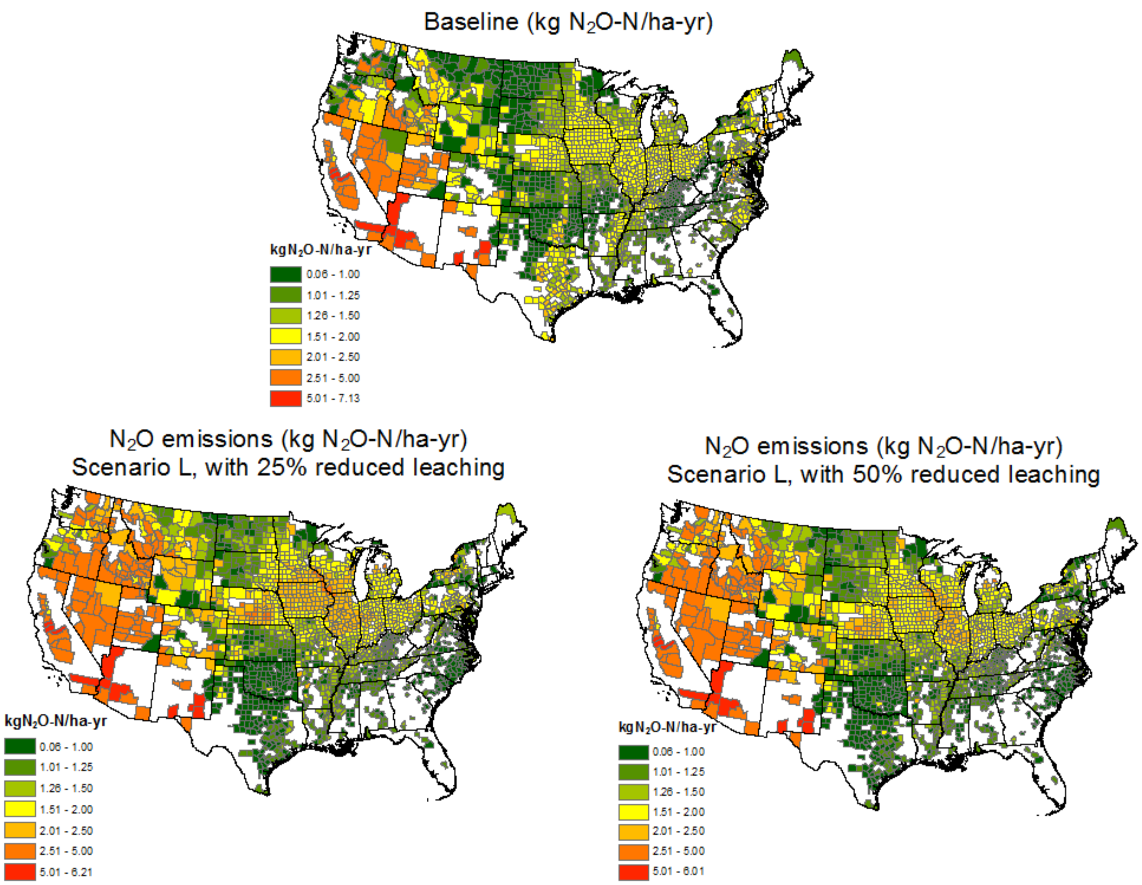


Figure 11. N_2O emissions under baseline (current) conditions vs N_2O emissions projected for the scenario with 50% increase in root C inputs and a 20% downward shift in root distribution, including with a 25% decrease (bottom left) or a 50% decrease (bottom right) in N leaching.

For the entire land base analyzed, average N_2O-N fluxes (kg N_2O-N ha^{-1} yr^{-1}) as a function of the root perturbation scenarios and potential leaching reduction scenarios (values only shown for 50% reduction scenario), over the 30 year projection period, are shown in Table 11.

Table 11. Average N_2O flux rates for the modeled scenarios. Values where surplus N was sufficient to supply additional N requirements are not included and it would be conservative to assume no change in N_2O emissions relative to the baseline.

Δ root C input	**Reduced leaching**	Δ root distribution				
		no change	5% shift	20% shift	Strong shift	Extreme shift
		kg N_2O-N ha^{-1} yr^{-1}				
0%	0% (baseline)	1.46	-	-	1.48	1.90
0%	-50%	na	-	-	-	1.69
+25%	0%	-	-	1.54	1.79	2.32
+25%	-50%	-	-	-	1.60	2.06
+50%	0%	1.55	1.60	1.80	2.11	2.74
+50%	-50%	-	-	1.61	1.88	2.37
+ 99%	0%	1.98	2.05	2.32	2.73	3.56
+ 99%	-50%	1.76	1.83	2.06	2.42	3.16

Summary outputs on overall GHG mitigation potential (Integration of T1-T4):

For the 30 year projection period, the estimated impacts on soil C and N_2O emissions, assuming 100% implementation of the change scenarios on the cropland, pasture and hay land areas with suitable rating with respect to soil depth (156.6 Mha), potential mitigation rates are substantial. For soil C sequestration, 25-50% increase in root C inputs coupled with moderate changes in root distributions, could increase soil C stores from 35 to >100 Tg C yr^{-1} (Table 8). Higher root C inputs and more extreme changes in root distributions could increase those values.

Combining soil C gains with projected increases in N_2O (where present) as well as increased embodied fertilizer emissions (associated with fertilizer production and distribution) due to additional plant N requirements and putting all values into CO_2 equivalents gives a net estimate for total GHG mitigation potential (Table 12). Total GHG mitigation is the net of CO_2 removals (sink) to soil minus increased N_2O emissions.

Table 12. Average per ha soil C sinks and N_2O source emissions by scenario. A GWP value (100 yr time horizon) of 298 was used to put N_2O emissions on a CO_2 equivalent basis.

	ΔSOC	Δ soil CO_2eq	N_2O (as CO_2 eq)		Embodied fertilizer (as kg CO_2 eq)		total net GHG reduction	
	kg ha^{-1} yr^{-1} for 30 yr projection period							
0% root C increase			base leaching	-50% leaching	base leaching	-50% leaching	base leaching	-50% leaching
base root distr.	--	--	--	--	--	--	--	--
5% shift	29	107	0	0	0	0	107	107
20% shift	129	474	0	0	0	0	474	474
strong shift	308	1131	6	0	5	0	1120	1131
extreme shift	528	1936	202	107	170	90	1564	1739
25% root C increase								
base root distr.	187	686	0	0	0	0	686	686
5% shift	223	819	0	0	0	0	819	819
20% shift	349	1279	34	0	28	0	1217	1279
strong shift	573	2099	154	63	129	53	1816	1983
extreme shift	847	3106	399	281	336	236	2371	2589
50% root C increase								
base root distr.	374	1372	39	0	33	0	1300	1372
5% shift	418	1532	64	0	54	0	1414	1532
20% shift	568	2083	158	67	133	56	1792	1960
strong shift	837	3068	302	193	253	162	2513	2713
extreme shift	1166	4276	596	425	501	357	3179	3494
99% root C increase								
base root distr.	741	2716	243	141	204	118	2269	2457
5% shift	799	2928	277	171	232	143	2419	2614
20% shift	998	3660	40	280	337	235	3283	3145
strong shift	1354	4966	591	448	497	376	3878	4142
extreme shift	1792	6570	983	794	826	667	4761	5109

Totals for the entire land area included in the analyses are given in Table 13.

Table 13. Aggregate net GHG reductions for the altered crop root scenarios (Tg CO_2eq yr^{-1} for 30 yr projection period).

	total GHG reduction	
	Tg CO_2eq yr^{-1} for 30 yr projection period	
0% root C increase	base leaching	-50% leaching
base root distr.	--	--
5% shift	17	17
20% shift	74	74
strong shift	175	177
extreme shift	245	272
25% root C increase		
base root distr.	107	107
5% shift	128	128
20% shift	190	200
strong shift	284	311
extreme shift	371	406
50% root C increase		
base root distr.	204	215
5% shift	221	240
20% shift	281	307
strong shift	393	425
extreme shift	498	547
99% root C increase		
base root distr.	355	385
5% shift	379	409
20% shift	514	492
strong shift	607	649
extreme shift	746	800

Overall the positive changes in soil C stocks (representing a removal of CO_2 from the atmosphere) are the predominant component in GHG balance projected from the analysis. Increased N_2O emissions and embodied fertilizer emissions do reduce somewhat the net GHG benefit, negating up to 28% of the soil C GHG benefit in the scenarios which have the highest demand for additional N. However, for the scenarios with less than 50% increase in C inputs and with less than the most extreme downward shift in root distributions, the impacts of increased N_2O and embodied emissions on net GHG benefits were negligible. Whave not included estimates of indirect N_2O (although these are typically considerably less than direct emissions) thus may be somewhat understated. However, even if total N_2O were 2X our estimates, they would not be sufficient to offset the climate benefits from soil C sequestration over the 30 year projection period.

Task 5. Ancillary benefits of increased SOM

Soil organic matter (SOM) is the organic (plant and animal) matter component of soil carbon which operates as the biological foundation for soil, supplying the energy to support soil microorganisms and faunal and influencing the physical (e.g., aggregation, bulk density, infiltration, porosity), and chemical (e.g., pH, cation exchange, nutrient supply) characteristics of soil that determine soil health and function (Allen et al. 2011). Through these interactions SOM serves as a reservoir for both water and nutrients, and provides the foundation for microbial actions that lead to changes in soil structure, aggregate size, water holding capacity, increased nutrient exchange, etc that are all part of what makes up a healthy soil. Soil organic matter contributes directly and locally to the ecosystem by impacting biomass production, biodiversity, and nutrient supply (Antle and Stoorvogel 2008) as well as indirectly, with non-local benefits as an intermediary service to water quality and climate change at regional to global scales (Victoria et al. 2012, Pascual et al. 2015).

The importance of SOM is unquestioned, but without an economic value for SOM our policies and practices have led to a shortsighted view of ecological value based on present day production. This shortsighted view of SOM has contributed to soil degradation soils throughout the world with negative consequences on production, water quality, and climate change (Lal 2009, Pascual et al. 2015). While degradation of a soil's SOM can be partially masked by fertilization or residue management, at least in the short term, the incurred costs of depleting SOM are not sustainable (Loveland and Webb 2003, Lal 2009). This reduced SOM represents a loss of value to farmers through reduced production (Lal 2009) as well as public good from lowered water quality and increased atmospheric carbon (Victoria et al. 2012, Campbell and Paustian 2015). While replenishing SOM has been called a necessary step in combating climate change (Lal 2011, Powlson et al. 2011) the slow-nature turnover and accumulation of SOM require many decades to centuries to restore SOM to natural levels (Pascual et al. 2015). The goals of this section are therefore to review some of the benefits of SOM, outline indicators for a healthy soil and their interactions with SOM, and finally to examine the current economic view of SOM in order to provide a starting point for future discussions.

An emerging topic around soils is how to quantify and determine soil health, through the measurement of soil quality indicators (SQI) in hopes of determining management pathways to improve the health of the soil and increase production (Allen et al. 2011). Amongst studies looking at SQI, soil water holding characteristics is one category that is universally important (Zornosa et al. 2007, Jokela et al. 2011; Armenise et al. 2013, Nosrati 2013, Moncada et al. 2014). Another study suggested that aeration characteristics in the upper 10 cm of the soil profile, and soil aggregation in the 10-20cm depth range had the largest impact on soil health and crop production (Shukla et al. 2006). While aeration characteristics lead to better water infiltration, larger aggregates with their increased surface area and pore space are needed to store this water and bind nutrients for plant use. This increased water storage and nutrient availability are both key for a healthier soil with increased productivity (Jokela et al. 2011). Inherently this makes sense as water-filled pore space is directly related to oxygen levels, and thus the amount and type of biological activity occurring. Soil water characteristics are both affected by SOM contents, via soil structural attributes such as aggregation, bulk density and porosity, and likewise soil water status affects SOM and GHG emissions, through soil moisture and aeration effects on plant and microbial activity, which influences decomposition rates, losses of nutrients via leaching, or altered oxygen levels which acts as a driver for quantity and type of GHG emissions (Campbell and Paustian 2015). One of the stronger SQI is soil porosity which can often be used as a quantitative measure to determine the amount of water stored in the root zone (Reynolds et al., 2002). While SOM is a regulating service for water flows, plant production also has feedbacks on SOM characteristics. For example, root development has a strong impact on soil porosity, as well as pore space and distribution (Allen et al., 2011). Therefore the amount and depth of root biomass could potentially play a large role in managing water and nutrient flow as well as carbon sequestration.

While the benefits of SOM are well known, they are not linear and there is also believed to be a critical limit for SOM, with an estimated average of ~3.4% SOM (of total soil mass) for temperate systems (Loveland and Webb, 2003), below which there can be substantial negative implications to both soil quality and crop productivity. Sites with SOM below the critical level are unlikely to reach maximum yield potentials due to increased leaching and erosion losses, as well as reduced mineralization rates (Loveland and Webb, 2003). This critical limit varies by region, crop, management, as well as climate trends making it an important topic of further research as we adjust to a changing climate (Lal, 2009).

As a necessary substance for agricultural production, SOM inherently has a value and given its ability to be depleted it can further be considered a scarce resource that is not freely available (Pascual et al. 2015). The natural capital provided from the direct and indirect benefits of SOM are poorly understood and vastly underestimated. While some obvious and rudimentary ecological valuations for carbon (carbon price, carbon tax or carbon credit schemes) and nitrogen or other nutrients (cost of fertilizer) could be applied to SOM these do not consider all the benefits that SOM provides. However, under these rudimentary analysis the value might be estimated as such; one ton of SOC holds roughly 100 kg of nitrogen and with the current cost of urea (46% N) at ~$270/ton ($590/ton N), one ton of lost SOC is equivalent to losing $59 in terms of its nitrogen value. As outlined above, one of the main roles of SOM and something not currently considered in its economic value is its role as a supporting and intermediary service for water flow. SOM controls on water flows lead to impacts on nutrient and agrochemical leaching, erosion, runoff, crop productivity, and GHG emissions (Victoria et al. 2012, Pascual et al. 2015). These direct and indirect benefits of SOM represent a substantial value that is currently overlooked by these rudimentary carbon and nitrogen valuations, resulting in an underestimation of the true economic value of SOM.

Due to both the private and public benefits of SOM, determining an appropriate economic value, as well as to whom that economic value should be allocated when SOM changes, remains a difficult task. The answers to these questions are important ones in terms of adjusting policy under a changing climate as well as being foundational to soil carbon credit markets. For example, soil organic matter is believed to have such a big impact on biomass production and ecosystem health that one study estimates that an average increase in SOC of 1 Mg C/ha on croplands could lead to increases in crop productivity of 6-12 million Mg/yr in sub Saharan Africa and 24-40 million Mg/yr in developing countries overall (Lal, 2009). Those benefits do not even address the additional economic benefits of increased production, potentially reduced leaching or erosion, jobs created, increased food security and more. While developing countries contain some of the most degraded agricultural land and would benefit the most from increased SOC, there are many barriers to restoring these lands (Lal, 2009). What benefits a local farmer in the tropics does not necessarily match global desires, for example for tropical rainforests, and thus is not incentivized or reflected in current policies (Izac 1997). Further, management needed to increase SOC generally requires capital up front that many farmers are unable to afford, especially when the benefits of increased SOC may not be seen immediately. Poor economic situations, short planning horizons and high discount rates make it unlikely that many farmers in developing can adjust their management, using conventional practices, in order to increase SOM without economic incentives (Izac 1997).

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Appendix A1 – Crop characteristics used in N balance calculations

Corn

Corn grain was assumed to be 9% protein (1.44%N) throughout the entire US (US Grain council, 2012) while corn silage used a harvest rate of 95% and 1.1 %N for harvested biomass (Brown, et al., 2010).

Wheat

Wheat protein percent is based on the US Wheat Associates 2015 wheat report (Wheat Associates, 2015). Based on wheat market class maps included in the report, a state was assigned a major wheat market class, with the respective protein percent for each wheat type (protein percent based on 5 year average) from the same report. There were some exceptions for what protein percent value was used that are outlined below

Table A1. Wheat protein percent at the state level, states missing data do not have any counties above the 10,000 acre threshold.

State	Major Wheat Market Class	%protein	#wheat counties	bushel/ weight
AL	soft red winter	10.0	1	58.7
AR	soft red winter	10.0	23	58.7
AZ	desert durum	13.4	3	62.8
CA[1]	avg(desert durum/hard red winter)	13.1	10	61.8
CO[2]	hard red winter	12.1	16	60.7
CT				
DE	soft red winter	10.0	2	58.7
FL				
GA	soft red winter	10.0	1	58.7
ID	soft white	10.0	18	60.6
IL	soft red winter	10.0	25	58.7
IN	soft red winter	10.0	6	58.7
IA				
KS[3]	hard red winter	12.3	88	60.7
KY	soft red winter	10.0	12	58.7
LA	soft red winter	10.0	6	58.7
ME				
MD	soft red winter	10.0	6	58.7
MA				
MI	soft red winter	10.0	21	58.7
MN	hard red spring	14.1	21	61.4
MS	soft red winter	10.0	6	58.7
MO	soft red winter	10.0	26	58.7
MT	hard red winter	12.7	36	60.7
NE[2]	hard red winter[1]	12.0	32	60.7
NV				
NH				

NJ				
NM	hard red winter	12.7	4	60.7
NY	soft red winter	10.0	4	58.7
NC	soft red winter	10.0	12	58.7
ND[4]	avg(hard red spring/northern durum)	13.8	53	60.7
OH	soft red winter	10.0	32	58.7
OK	hard red winter	12.7	36	60.7
OR	soft white	10.0	7	60.6
PA	soft red winter	10.0	4	58.7
RI				
SC	soft red winter	10.0	6	58.7
SD[4]	avg(hard red winter/hard red spring)	13.4	43	61.1
TN	soft red winter	10.0	7	58.7
TX	hard red winter	12.7	64	60.7
UT	hard red winter	12.7	2	60.7
VT				
VA	soft red winter	10.0	3	58.7
WA	soft white	10.0	14	60.6
WV				
WI	soft red winter	10.0	5	58.7
WY	hard red winter	12.7	3	60.7

[1]California wheat is taken as an even average between desert durum &hard red winter wheat.
[2]Colorado and Nebraska protein % is based on PlainsGrains crop reports (2009-2015), with average data from 2008-2014.
[3]Kansas protein % is based on USDA, Kansas wheat history report (2015), with the data an average protein concentration from 2000-2010.
[4]North and South Dakota are an even average of their respective wheat types reported.

Soybean

The American Soybean Association, soybean 2014 commodity report was used for soybean percent protein (Soybean Association, 2014). The US average from 1986-2014 of 35.2% protein was used to determine soybean nitrogen harvested. Salvagiotti et al., (2008) was used to estimate nitrogen fixation with the equation 0.66X – 19 (kgN/ha), where X is the total biomass nitrogen of the crop. Total biomass data was taken from the NASS yield model, and used a CN ratio of 13.5 (KBS LTER) to estimate total biomass nitrogen.

Alfalfa

An alfalfa protein percent of 16% was used for the entire US (Higginbotham et al., 2008). This protein percent was selected to reflect the various stages of when alfalfa is harvested and the impacts on protein percent. A nitrogen fixation equation was taken from Anglade et al., (2015), a review paper for estimating N_2 fixation in legumes of Europe. The equation used to estimate N_2 fixation for alfalfa was 0.81Ny – 13.9 where Ny is the nitrogen yield in the harvest. Because this does not account for fixed nitrogen needs of the unharvested and belowground biomass, an adjustment factor (x 1.67) was used to calculate residual nitrogen fixed, as ~40% of the fixed nitrogen is accounted for in unharvested biomass

(Anglade et al., 2015). A further adjustment was made that only affected a few counties, where manure nitrogen inputs were subtracted from biomass nitrogen needs. If the resulting value was less than the original fixed nitrogen amount, this value was now used for fixed nitrogen. This adjustment was used based on the thinking that higher nitrogen application rates (at least in the extreme) lead to lower nitrogen fixation by nitrogen fixing crops.

Grass

A grass hay protein percent of 8.5% was used for the entire US.