Climate-smart soils

Article *in* Nature · April 2016

DOI: 10.1038/nature17174

6 authors, including:

Keith Paustian
Colorado State University
355 PUBLICATIONS **36,901** CITATIONS

Johannes Lehmann
Cornell University
427 PUBLICATIONS **56,000** CITATIONS

Dave S Reay
The University of Edinburgh
238 PUBLICATIONS **6,328** CITATIONS

G Philip Robertson
Michigan State University
327 PUBLICATIONS **26,480** CITATIONS

doi:10.1038/nature17174

Climate-smart soils

Keith Paustian[1,2], Johannes Lehmann[3], Stephen Ogle[2,4], David Reay[5], G. Philip Robertson[6] & Pete Smith[7]

Soils are integral to the function of all terrestrial ecosystems and to food and fibre production. An overlooked aspect of soils is their potential to mitigate greenhouse gas emissions. Although proven practices exist, the implementation of soil-based greenhouse gas mitigation activities are at an early stage and accurately quantifying emissions and reductions remains a substantial challenge. Emerging research and information technology developments provide the potential for a broader inclusion of soils in greenhouse gas policies. Here we highlight 'state of the art' soil greenhouse gas research, summarize mitigation practices and potentials, identify gaps in data and understanding and suggest ways to close such gaps through new research, technology and collaboration.

Evidence points to agriculture as the first instance of human-caused increases in greenhouse gases (GHGs), several thousand years ago[1]. Agriculture and associated land-use change remain a source for all three major biogenic GHGs: carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). Land use contributes ~25% of total global anthropogenic GHG emissions: 10%–14% directly from agricultural production, mainly via GHG emissions from soils and livestock management, and another 12%–17% from land cover change, including deforestation[2,3]. Although soils contribute a major share (37%; mainly as N_2O and CH_4) of agricultural emissions[3], improved soil management can substantially reduce these emissions and sequester some of the CO_2 removed from the atmosphere by plants, as carbon (C) in soil organic matter (in this Perspective, our discussion of soil C refers solely to organic C). In addition to decreasing GHG emissions and sequestering C, wise soil management that increases organic matter and tightens the soil nitrogen (N) cycle can yield powerful synergies, such as enhanced fertility and productivity, increased soil biodiversity, reduced erosion, runoff and water pollution, and can help buffer crop and pasture systems against the impacts of climate change[4].

The inclusion of soil-centric mitigation projects within GHG offset markets[5,6] and new initiatives to market 'low-carbon' products[7] indicate a growing role for agricultural GHG mitigation[8]. Moreover, interest in developing aggressive soil C sequestration strategies has been heightened by recent assessments, which project that substantial terrestrial C sinks will be needed to supplement large cuts in GHG emissions to achieve GHG stabilization levels of 450 parts per million CO_2 equivalent or below, consistent with the goal of a mean global temperature increase of less than 2 °C (ref. 9). Soil C sequestration is one of a few strategies that could be applied at large scales[9] and potentially at low cost; as an example, the French government has proposed[10] to increase soil C concentration in a large portion of agricultural soils globally, by 0.4% per year, in conjunction with the Conference of the Parties to the UN Framework Convention on Climate Change (UNFCCC) negotiations in December 2015. This would produce a C sink increase of 1.2 petagrams (Pg) of C per year (ref. 10).

An extensive body of field, laboratory and modelling research over many decades demonstrates that improved land use and management practices can reduce soil GHG emissions and increase soil C stocks. However, implementing effective soil-based GHG mitigation strategies on a large scale will require the capacity to measure and monitor GHG reductions with acceptable accuracy, quantifiable uncertainty and at relatively low cost. Targeted research to improve predictive models, expanded observational networks to support model validation and uncertainty bounds, 'Big Data' approaches to integrate land use, management and environmental drivers, and technologies with which to engage actively with land users at the grass-roots level, are all key elements in realizing the potential GHG mitigation from climate-smart agricultural soils.

Process controls and mitigation practices

Soil C sequestration via improved management

Soils constitute the largest terrestrial organic C pool (~1,500 Pg C to a depth of 1 m; 2,400 Pg C to 2 m depth[11]), which is three times the amount of CO_2 currently in the atmosphere (~830 Pg C) and 240 times the current annual fossil fuel emissions (~10 Pg)[9]. Thus, increasing net soil C storage by even a few per cent represents a substantial C sink potential.

Proximal controls on the soil C balance include the rate of C addition as plant residue, manure or other organic waste, minus the rate of C loss (via decomposition). Hence, C stocks can be increased by increasing organic matter inputs or by reducing decomposition rates (for example, by reducing soil disturbance), or both, leading to net removal of C from the atmosphere[12]. However, soil C accrual rates decrease over time as stocks approach a new equilibrium. Therefore net CO_2 removals are of limited duration, often attenuating after two to three decades[13].

Unmanaged forests and grasslands typically allocate a large fraction of their biomass production belowground and their soils are relatively undisturbed; accordingly, native ecosystems usually support much higher soil C stocks than their agricultural counterparts, and soil C loss (typically 0.5 to >2 megagrams (Mg) of C per hectare per year) following land conversion to cropland has been extensively documented[14,15]. Total losses once the soil approaches a new equilibrium are typically ~30%–50% of topsoil (for example, 0–30 cm depth) C stocks[15]. Hence, avoiding conversion and degradation of native ecosystems is a strong mitigation alternative. Conversely, restoration of marginal or degraded lands to perennial forest or grassland increases soil C storage (Fig. 1), although usually at a slower rate than the original conversion losses[16,17]. Restoring wetlands that have been drained for agricultural use reduces ongoing decomposition losses, which can be as high as 5–20 $Mg\,C\,ha^{-1}\,yr^{-1}$ (ref. 18), and can also restore C sequestration (Fig. 1), though methane emissions may increase[19,20]. Land-use conversions may, however, conflict with agricultural production and food security objectives, entailing the need for a broad-based accounting of net GHG implications (Table 1)[21].

[1]Department of Soil and Crop Sciences, Colorado State University, Fort Collins, Colorado, USA. [2]Natural Resource Ecology Laboratory, Colorado State University, Fort Collins, Colorado, USA. [3]Atkinson Center for a Sustainable Future, Department of Soil and Crop Sciences, Cornell University, Ithaca, New York, USA. [4]Department of Ecosystem Science and Sustainability, Colorado State University, Fort Collins, Colorado, USA. [5]School of Geosciences, University of Edinburgh, Edinburgh, UK. [6]W. K. Kellogg Biological Station and Department of Plant, Soil and Microbial Sciences, Michigan State University, East Lansing, Michigan, USA. [7]Institute of Biological and Environmental Sciences, University of Aberdeen, Aberdeen, UK.

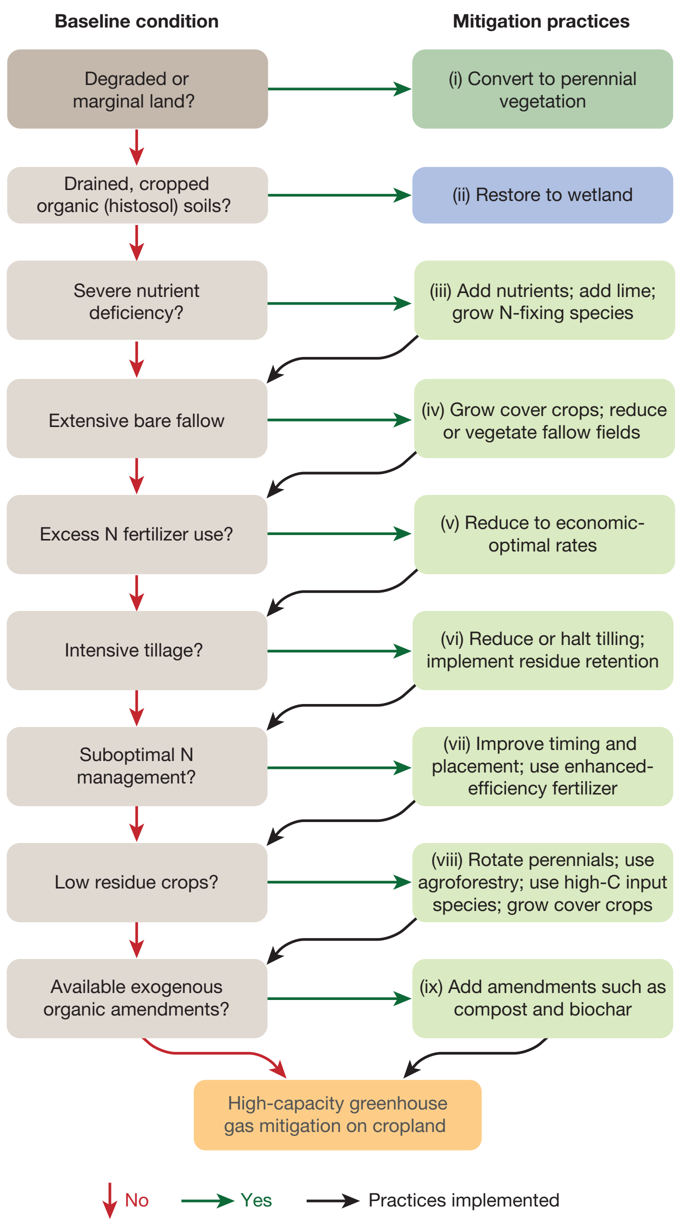

Figure 1 | Decision tree for cropland GHG mitigating practices. (Rice is not included.) For degraded, marginal lands, the most productive mitigation option is conversion to perennial vegetation either left unmanaged or sustainably harvested to offset fossil energy use (cellulosic biofuels). Histosol is soil with very high organic matter content, such as from peat bog. For more arable lands, multiple options could be implemented sequentially or in combination, depending on management objectives, cost and other constraints (see Table 1). The practices shown (see Table 1 and text for more discussion) are roughly ordered from lower-cost or higher-feasibility options towards more costly interventions (bottom of figure).

In general, soil C sequestration rates on land maintained in agricultural use are lower than for land restoration to grassland or forest, and vary on the order of 0.1–1 $\mathrm{Mg\,C\,ha^{-1}\,yr^{-1}}$, as a function of land-use history, soil or climate conditions, and the combination of management practices applied[2,15]. Practices that increase C inputs include (1) improved varieties or species with greater root mass to deposit C in deeper layers where turnover is slower[22], (2) adopting crop rotations that provide greater C inputs[23], (3) more residue retention[24], and (4) cover crops during fallow periods to provide year-round C inputs (Fig. 1)[23,25]. Cover crops can also reduce nutrient losses, including nitrate that is otherwise converted to N_2O in riparian areas and waterways[26]—an example of synergy between practices that sequester C and also tighten the N cycle to limit emissions of N_2O (Table 1). Other practices to increase C inputs include irrigation in water-limited systems[19] and additional fertilizer input to increase productivity in low-yielding, nutrient-deficient systems (Fig. 1)[27]. Although additional nutrient and water inputs to boost yields may increase non-CO_2 emissions[28], the emissions intensity of the system (GHG emissions per unit yield) may decline, providing a global benefit if the yield increase avoids land conversion for agriculture elsewhere[21,23].

Many croplands can sequester C through less intensive tillage, particularly zero tillage[15], owing to less disruption of soil aggregate structure[29]. Some authors have argued that benefits are small because increased C content in surface horizons are offset by C losses deeper in the profile[30], although others have noted that the larger variability in sub-surface horizons and lack of statistical power in existing studies makes such conclusions questionable[31].

A change from annual to perennial crops typically increases belowground C inputs (and soil disturbance is reduced), leading to C sequestration[16]. In grasslands, soil C sequestration can be increased through optimal stocking/grazing density[32]. Improved management in fire-prone ecosystems via fire prevention or prescribed burning can also increase C sequestration[33].

Key knowledge gaps that affect our understanding of soil C sequestration processes and management options to implement them include questions about the differential temperature sensitivities of C turnover among soil organic matter fractions[34], interactions among organic matter chemistry, mineral surface interactions and C saturation[35–37], and subsoil (>30 cm) soil organic matter accretion, turnover and stabilization[38]. Landscape processes, particularly the impact of erosion and lateral transport of C in sediments, contribute additional uncertainty on net sequestration occurring at a specific location[39]. And emerging evidence that stabilized soil organic matter is of microbial rather than direct plant origin[35,40] may offer a potential to manipulate the soil–plant microbiome to enhance C sequestration in the rhizosphere.

Soil C sequestration via exogenous C inputs

Addition of plant-derived C from external (that is, offsite) sources such as composts or biochar can increase soil C stocks, and may result in net CO_2 removals from the atmosphere (Fig. 1). Both compost and biochar are more slowly decomposed compared to fresh plant residues, with composts typically having mean residence times several times greater than un-composted organic matter[41], and biochar mineralizes 10–100 times more slowly than uncharred biomass[42]. Thus a large fraction of added C—particularly for biochar—can be retained in the soil over several decades or longer, although residence times vary depending on the amendment type, nutrient content and soil conditions[36] (such as moisture, temperature and texture).

However, because the organic matter originates from outside the ecosystem 'boundary', a broader life-cycle assessment approach is needed, that considers the GHG impacts of: (1) offsite biomass removal, transport, and processing; (2) alternative end uses of the biomass; (3) interactions with other soil GHG-producing processes; and (4) synergies between these soil amendments and the fixation and retention of *in situ* plant-derived C[43,44]. In many cases, net life-cycle emissions will largely depend on whether the biomass used as a soil amendment would have otherwise been burnt (either for fuel, thereby offsetting fossil fuel use, or as waste disposal), added to a landfill, or left in place as living biomass or detritus[43,44].

While slower mineralization of the amendment is an important determinant of net mitigation impact, effects on other soil emissions cannot be neglected. Mineralization of existing soil C in response to amendments (often referred to as 'priming'[45]) has often been observed immediately following biochar addition, but priming usually declines, sometimes becoming negative (that is, inhibiting *in situ* soil C decomposition), over time[46,47]. Analogous time dependence of soil N_2O and CH_4 emissions has not received sufficient attention[41]. Increased plant growth

Table 1 | Co-benefits, relative costs and constraints for the mitigation practices shown in Fig. 1

Mitigation practices	Practice co-benefits	Relative cost		Constraints and caveats
		Developed	Less developed	
(i)	↓Soil erosion ↑Biodiversity ↑Water quality	$$	$$	Alternate land/livelihood for subsistence farmers; opportunity cost of removing land base; potential for leakage (that is, land-use change impact)
(ii)	↑Biodiversity ↑Water quality	$$$	$$$	High opportunity cost of lost crop production; potential increase in CH_4 emissions; potential for leakage (that is, land-use change impact)
(iii)	↑Food security ↑Water quality	$	$$	Availability or access to fertilizer; potential increase in N_2O emissions
(iv)	↓Soil erosion ↑Water quality ↑Soil health ↑Food security	$	$$	Limited applicability in dry areas
(v)	↑Water quality	$	$	Risk of crop production loss
(vi)	↓Soil erosion ↑Water quality ↑Soil health	$	$$	Limited applicability in cold climates; potential increased equipment cost; increased herbicide use
(vii)	↑Water quality	$$	$$$	Availability or access to enhanced efficiency fertilizer
(viii)	↑Biodiversity ↑Water quality ↑Soil health	$$$	$$	Less applicability in dry areas and shallow soils; potential opportunity costs of lost crop production
(ix)	↑Soil health ↑Food security	$$$	$$	Dependent on life-cycle emissions of producing the amendment

Mitigation practices are numbered according to Fig. 1. Co-benefits include non-GHG ecosystem services from practice implementation. Relative costs are provided as examples based on a developed region such as North America and a less developed region such as sub-Saharan Africa; however, a specific option in one region may have a higher cost or be a less feasible option in another region. Potential constraints include factors that might limit or preclude practice adoption or increase other GHG emissions as a consequence of practice adoption. All options require a region-specific full-cost carbon accounting (GHG life-cycle analysis) that considers potential indirect land-use effects in order to define specific mitigation potentials.

in amended soils and the resultant feedbacks to soil C can make up a large proportion of the soil-based GHG balance[41,48] and these feedbacks may be especially important for more persistent amendments, because of the longer duration of any effects.

Soil management to reduce N_2O emissions

Arable soils emit more N_2O to the atmosphere than any other anthropogenic source[2,19]; some 4.2 teragrams (Tg) of a global anthropogenic flux of 8.1 Tg N_2O-N yr^{-1}. Reducing this flux represents a substantial mitigation opportunity, particularly since N_2O is often the major source of radiative forcing in intensively managed cropland. Better N management to reduce emissions would also ameliorate other environmental problems such as nitrate pollution of ground and surface waters caused by excess reactive N in agroecosystems (Fig. 1, Table 1).

N_2O is produced in soils by microbial activity—mainly nitrification and denitrification—which occurs readily when stimulated by the abundant N that cycles rapidly in virtually all agroecosystems. During nitrification, ammonium added as fertilizer, fixed from the atmosphere by legumes, or mineralized from soil organic matter, crop residue, or other inputs is oxidized to nitrite and eventually to nitrate in a series of reactions that can also produce N_2O. Likewise, when denitrifiers use nitrate as an electron acceptor when soil oxygen is low, N_2O is an intermediate product that can readily escape to the atmosphere.

Arable soils managed to support high crop productivity have the capacity to produce large quantities of N_2O, and fluxes are directly related to N inputs. On average, about 1% of the N applied to cropland is directly emitted as N_2O (ref. 49), which is the basis for estimating emissions using default Intergovernmental Panel on Climate Change methods[18]. However, recent evidence suggests that this value is too high for crops that are under-fertilized and too low for crops that are fertilized liberally[28]. When crops compete with microbes for available N, N_2O fluxes are lower. In addition to direct in-field emissions, high N applications cause N losses from leaching and volatilization that contribute to 'indirect' N_2O emissions, downstream or downwind from the field[50].

Since N_2O has no significant terrestrial sink, abatement is best achieved by attenuating known sources of N_2O emissions, by altering the environmental factors that affect N_2O production (soil N, oxygen, and C) or by biochemically inhibiting conversion pathways using soil additives. For example, nitrification can be inhibited with commercial additives such as nitrapyrin and dicyandiamide, which slow ammonium oxidation, and field experiments suggest that inhibitors can reduce N_2O fluxes by up to 40% in some soils, although other soils show little reduction and more research is needed to understand variable site-level responses[51]. Likewise, tillage and water management can affect N_2O fluxes by altering the soil microenvironment[52,53].

Another means of reducing N_2O emissions from arable soils is more precise N management to minimize excess N not used by the crop, while maintaining sustainable high yields. Fertilized crops typically take up less than 50% of the N applied; the remainder is available for loss. According to one recent study[54], corn farmers in the midwestern states of the USA could reduce N_2O loss by 50% with more conservative fertilizer practices. Nitrogen conservation can be achieved by: (1) better matching application rates of N to crop needs using advanced statistical and quantitative modelling; (2) applying fertilizer at variable rates across a field based on natural patterns of soil fertility, or within the root zone rather than broadcast on the soil surface; and (3) applying fertilizer close to when the crop can use it, such as several weeks after planting, or adding it earlier but using slow-release coatings to delay its dissolution (Fig. 1, Table 1)[50].

High temporal and spatial variability make predictions of changes in N_2O fluxes in response to management surprisingly difficult. Particularly lacking are empirical data for multi-intervention strategies that may interact in unexpected ways. Aligned to this paucity are gaps in our understanding of how N cycling and net N_2O flux in managed soils will respond to future climate change[55]. The limited number of field manipulation studies to date indicate that changing temperature and precipitation patterns may have large and strongly coupled effects on net N_2O emissions[56], yet our understanding of the processes that underpin these effects and their robust representation in models is far from complete.

Soil management to reduce CH_4 emissions

More than one-third (>200 Tg yr^{-1})[9] of global methane (CH_4) emissions occur through the microbial breakdown of organic compounds in

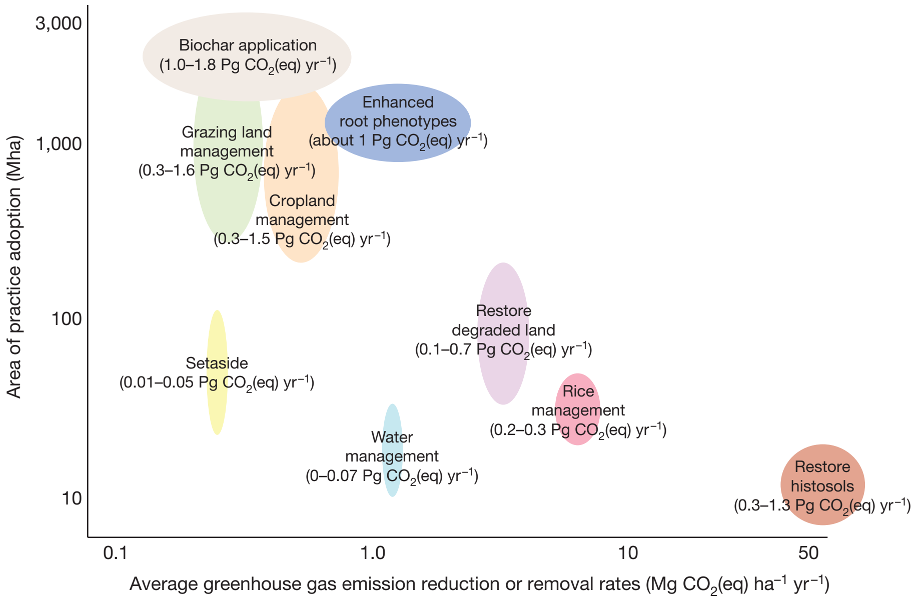

Figure 2 | Global potential for agricultural-based GHG mitigation practices. Management categories are arranged according to average per hectare net GHG reduction rates and potential area (in millions of hectares) of adoption (note log-scales). Unless otherwise noted, estimates are from ref. 19, based on cropland and grassland area projections for 2030. Ranges given in units of total Pg CO_2(eq) yr^{-1} represent varying adoption rates as a function of C pricing (US\$20, US\$50 and US\$100 per Mg CO_2(eq)), to a maximum technical potential—that is, the full implementation of practices on the available land base. Multiple practices are aggregated for cropland (for example, improved crop rotations and nutrient management, reduced tillage) and grazing land (for example, grazing management, nutrient and fire management, species introduction) categories. Practices that increase net soil C stocks or reduce emissions of N_2O and CH_4 are combined in each practice category. The portion of projected mitigation from soil C stock increase (about 90% of the total technical potential) would have a limited time span of 20–30 years, whereas non-CO_2 emission reduction could, in principle, continue indefinitely[19]. Estimates for biochar application[67] represent a technical potential only, but it is based on a full life-cycle analysis applicable over a 100-year time span. Although global estimates of the potential impact of enhanced root phenotypes for crops have not been published, a first-order estimate of about 1 Pg CO_2(eq) yr^{-1} is shown, using the global average C accrual rates (0.23 Mg C ha^{-1} yr^{-1}) for cover crops[25], applied to 50% of the cropland land area used by ref. 19. 'Setaside' land is arable land, usually for annual crops, that is taken out of production and converted to perennial vegetation (often grassland) and not actively managed for agricultural production, such as conservation reserves.

soils under anaerobic conditions[57]. As such, wetlands (177–284 Tg yr^{-1}) and rice cultivation (33–40 Tg yr^{-1})[9] represent the largest soil-mediated sources of CH_4 globally. In contrast, well aerated soils act as sinks for CH_4 (estimated at ~30 Tg yr^{-1}) from the atmosphere via CH_4 oxidation, the bulk of this net sink being in unmanaged upland and forest soils[58].

Key determinants of soil CH_4 fluxes include aeration, substrate availability, temperature and N inputs[59]; therefore, soil management can radically alter CH_4 fluxes. For example, in most soils, conversion to agriculture severely restricts CH_4 oxidation, related to the suppression of methanotrophs by accelerated N cycling[60]. In flooded rice, alterations in drainage regimes and organic residue incorporation could reduce emissions by ~25% or 7.6 Tg CH_4 yr^{-1} globally[19], although cycles of wetting and drying of soils may also enhance N_2O production[61] and soil C mineralization[62], thereby reducing the net mitigation effect.

With global rice production projected to expand by ~40% between 2000 and 2023 (ref. 63), the potential for further GHG mitigation via soil management appears to be large, although the global distribution and diverse nature of rice production systems—including irrigated, rain-fed and deep-water—present challenges to developing effective mitigation strategies. For longer-term (>20 year) projections, climate change and land–atmosphere interactions become increasingly important, with changes in N inputs, temperature, precipitation and atmospheric CO_2 concentration all likely to affect net CH_4 fluxes from soils[64].

This uncertainty highlights important gaps in understanding key processes and their underlying controls. The restoration of soil CH_4 uptake following agricultural conversion, for example, appears related to methanotroph community diversity[65], about which we know too little. Likewise the abatement of CH_4 generation in rice rhizospheres is related to C compounds exuded by roots, such that CH_4 mitigation might be achieved through further rice breeding and genetics[66]. Limited availability of field-scale CH_4 flux data means a greater reliance on regionally averaged emission factors and extrapolation from mesocosm and laboratory incubations[18], and thus less site and condition specificity in modelling fluxes. Importantly, establishing the net climate forcing effects of any intervention is a prime target for future soil management research.

Global potential for soil GHG mitigation

How important, in total, is this large, varied set of land-use and management practices as a GHG mitigation strategy? One of the challenges in answering this question is to distinguish between what is technically feasible and what might be achieved given economic, social and policy constraints. A comprehensive global analysis of agricultural practices[19] combined climate-stratified modelling of emission reductions and soil C sequestration with economic and land-use change models to estimate mitigation potential as a function of varying 'C prices' (reflecting a social incentive to pay for mitigation). They estimated total soil GHG mitigation potential ranging from 5.3 Pg CO_2(eq) yr^{-1} (without economic constraints) to 1.5 Pg CO_2(eq) yr^{-1} at the lowest specified C price (US\$20 per Mg of CO_2(eq)). Average rates for the majority of management interventions are modest, <1 Mg CO_2(eq) ha^{-1} yr^{-1}. Thus, achieving large global GHG reductions requires a substantial proportion of the agricultural land base (Fig. 2). Although the economic and management constraints on biochar additions (not assessed by ref. 19) are less well known, ref. 67 estimated a global technical potential of 1–1.8 Pg CO_2(eq) yr^{-1} (Fig. 2).

A more unconventional intervention that has been proposed is the development of crops with larger, deeper root systems, hence increasing plant C inputs and soil C sinks[22,68]. Increasing root biomass and selecting for root architectures that store more C in soils has not previously been an objective for crop breeders, although most crops have sufficient genetic plasticity to alter root characteristics substantially[69] and selection aimed at improved root adaptation to soil acidity, hypoxia and nutrient limitations could yield greater root C inputs as well as increased crop yields[68]. Greater root C input is well recognized as a main reason for the higher soil C stocks maintained under perennial grasses compared to annual crops[16]. Although there are no published estimates of the global C sink potential for 'root enhancement' of annual crop species, as a first-order estimate, a sustained increase in root C inputs might add $\sim$1 Pg CO_2(eq) yr^{-1} or more if applied over a large portion of global cropland area (Fig. 2).

Thus, the overall mitigation potential of existing (and potential future) soil management practices could be as high as $\sim$8 Pg CO_2(eq) yr^{-1}. How much is achievable will depend heavily on the effectiveness of implementation strategies and socioeconomic and policy constraints. A key strength is that a variety of practices can often be implemented on the same land area, to leverage synergies, while avoiding offsetting effects for different gases (Fig. 1). But regardless of which combination of management interventions are pursued, effective policies, that incentivize land managers to adopt them, will be needed. A common thread across implementation strategies is the role for strong science-based metrics with which to measure and monitor performance.

Implementation of mitigation practices

Relative to many other GHG source categories, agricultural soil GHG mitigation presents particular challenges. Rates on an individual land parcel are often low, but vast areas of land are devoted to agriculture globally, and the implementers of mitigation practices—the people using the land—number in the billions. Therefore, engaging a substantial number of these people is a massive undertaking in itself. Furthermore, agricultural soil GHG emissions are challenging to quantify owing to their dispersed and variable nature and the multiplicity of controlling factors—operating across heterogeneous landscapes. Direct measurement of fluxes requires specialized personnel and equipment, normally limited to research environments, and hence not feasible for most mitigation projects. Model-based methods, in which emission rates are quantified as a function of location, environmental conditions and management, provide a more feasible approach[54,70,71]. Process-based models, which dynamically simulate mechanisms and controls on fluxes as a function of climatic and soil variables and management practices, and empirical models based on statistical analysis of field-measured flux rates, represent differing but complementary approaches. In general, model-based quantification systems enable monitoring to focus on practice performance and thus dramatically reduce transaction costs for implementing mitigation policies[70].

Several implementation strategies for soil GHG mitigation exist (see Box 1), all of which require robust quantification and monitoring technologies. Those requiring the most rigorous methods involve offset projects participating in cap-and-trade markets, in which land managers are directly compensated for achieving emission reductions. Other market-linked strategies, such as 'green labelling' systems for agricultural products, will also require rigorous yet easy-to-use GHG quantification tools, enabling agricultural producers to meet standards set by product distributors and accepted by consumers[7,72].

Within the voluntary C offset market space, there are a growing number of projects that include soil GHG mitigation components[5]. Several large projects focus on preventing land conversion (that is, from forest and grassland), thus avoiding large CO_2 emissions from soils and liquidated biomass C stocks. Relatively simple empirical models supplemented with field measurements are commonly used for avoided land conversion projects. For more complex land-use projects, empirical models are less suited to capturing interactions across multiple emission sources, and may over- or under-credit projects where a practice has an influence on multiple emission sources. There are relatively fewer projects targeting GHG mitigation on existing agricultural lands, involving a broader suite of soil management practices, and early pilot-phase N_2O and CH_4 reduction projects are only now being developed[5,54]. Here, accurately quantifying C sequestration and/or emission reductions is more challenging owing to lower rates of change relative to baseline conditions, thus requiring more sophisticated models and supporting research infrastructure (Fig. 3).

Another challenge for projects on existing agricultural lands is obtaining and processing the management activity data. For example, the Kenya Agriculture Carbon Project (KACP) involves a total of 60,000 individual small-holder farmers[73]. In contrast to projects involving major land-cover changes, where remote sensing can provide much of the activity monitoring (for example, retention of forested land over time), remote-sensing options are poorly suited for monitoring crop type, fertilizer, residue and water management, and organic matter amendments[74]; for such practices the best sources of information are the land managers themselves (Fig. 3).

Thus another option is to engage land managers as information providers. Examples of this approach are the Cool Farm Tool (http://www.coolfarmtool.org/)[72], being used by farmers participating in low C supply chain management, and the COMET-Farm tool

BOX 1

Implementing soil GHG reductions

Incentives for farmers to adopt alternative practices that mitigate GHGs can take a variety of forms, including:

(1) Regulation and taxation. Direct regulatory measures intended to reduce soil GHGs at the farm scale are probably politically unfeasible and costly. Taxation of N fertilizer, already used in parts of the USA and Europe to reduce nitrate pollution, could function as an indirect tax that would reduce N_2O emissions.

(2) Subsidies. Targeted government payments or subsidies for implementing GHG-reducing practices is emerging as a policy alternative. For example, US Department of Agriculture programmes are including GHG mitigation as a conservation goal and provisions in the EU Common Agricultural Policy link subsidy payments to 'cross-compliance' measures that include maintenance of soil organic matter stocks[99]. A more direct link to soil GHG emissions follows from a recent decision to include cropland and grassland in EU commitments under the Kyoto Protocol[100].

(3) Supply-chain initiatives. Major food distributors are targeting sustainability metrics, including low GHG footprints, as a consumer marketing strategy (see refs 101 and 102 for examples of initial efforts involving agricultural producers and product distributors and retailers), setting performance standards for contracted agricultural producers, including the requirement of field-scale monitoring of production practices and quantification of GHG emissions.

(4) Cap and trade. In a cap-and-trade system, emitters are subject to an overall emissions level or 'cap', in which permitted emissions decrease over time. Emitters can stay below the capped levels by reducing their own emissions or by purchasing surplus permits from capped entities that have exceeded their required reductions. Both compliance and voluntary markets can function as cap-and-trade systems[103]. Within many cap-and-trade systems, a limited amount of emission reductions (termed 'offsets') can be provided by non-capped entities. The inclusion of agricultural activities as offset providers has been growing, particularly within voluntary markets. To maintain the integrity of emission caps, key criteria for offset providers include demonstrating additionality (that is, ensuring that reductions result from project interventions and not simply business-as-usual trends), avoiding leakage (that is, unintended emission increases elsewhere as a consequence of the project activities), and providing for permanence (meaning that increased soil C storage, credited as a CO_2 removal, is maintained long-term).

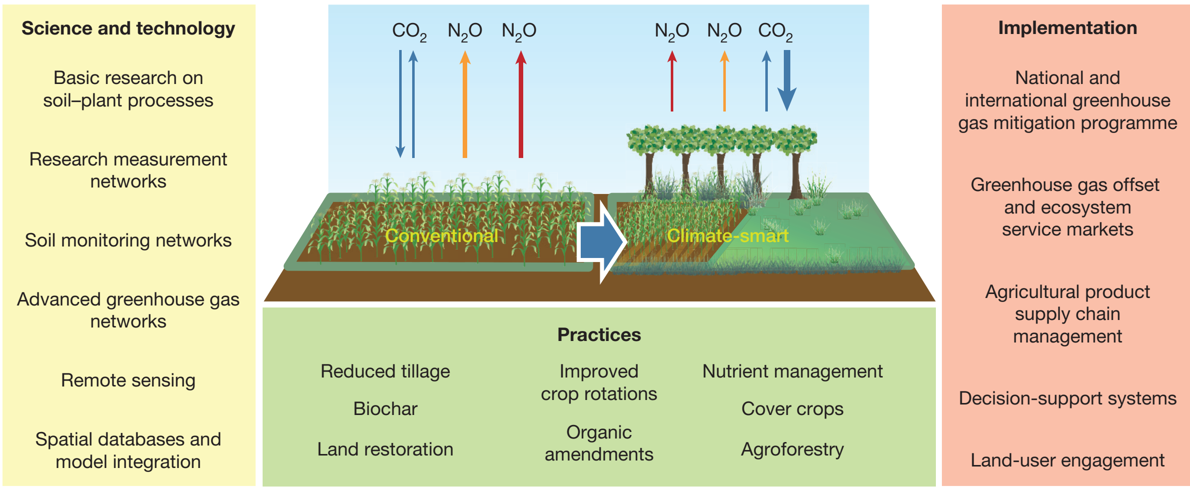

Figure 3 | Expanding the role of agricultural soil GHG mitigation will require an integrated research support and implementation platform. Targeted basic research on soil processes, expanding measurement and monitoring networks, and further developing global geospatial soils data can improve predictive models and reduce uncertainties. Ongoing advances in information technology and complex system and 'Big Data' integration offer the potential to engage a broad-range of stakeholders, including land managers, to 'crowd-source' local knowledge of agricultural management practices through web-based computer and mobile apps, and help drive advanced model-based GHG metrics. This will facilitate the implementation of climate-smart soil management policies, via cap-and-trade systems, product supply-chain initiatives for 'low-carbon' consumer products, and national and international GHG mitigation policies; it will also promote more sustainable and climate-resilient agricultural systems, globally.

(http://cometfarm.nrel.colostate.edu), which allows farmers to compute full farm-scale GHG budgets, for support of government-sponsored conservation initiatives and participation in mitigation projects[75]. Both tools provide web-based interfaces designed for non-specialists to enter land management information; Cool Farm utilizes empirical emission factor-type models, while COMET-Farm incorporates both empirical and process-based models. Such systems can be used to integrate local knowledge on management practices with detailed soil and climate maps, remote sensing and sophisticated models for emission calculations. Soon much of this functionality could be deployed in mobile applications (Fig. 3), which would be particularly advantageous in developing countries where the existing infrastructure to collect and manage land-use data is weak[76].

Quantifying uncertainties

Inventories of soil C stock changes and net GHG fluxes using process-based models will always have uncertainty due to lack of process understanding, inadequate parameterization, and limitations associated with model inputs[77] (such as weather, management and soils data). Empirical models generally rely on statistical analyses of measurement data to produce emission factors, along with an estimated uncertainty[15]. However, empirical models can be biased if measurements do not fully reflect the conditions for the agroecosystems in the project. Even with the limitations in process-based understanding, process-based models are likely to provide the most robust framework for estimating soil C stock and GHG flux changes in climate-smart agriculture programmes[78].

Monitoring, reporting and verification systems are a key element in a climate-smart agricultural programme. Such systems place different levels of importance on uncertainty depending on program type (see Box 1)[79], but discounting payments on the basis of the level of uncertainty is likely to be part of programmes with financial incentives, such as cap-and-trade. Discounting encourages monitoring efforts to reduce uncertainty over time[18]. If discounting payments for C sequestration and emission reduction practices with larger uncertainty is adopted in climate-smart agricultural programmes, then more advanced methods with process-based models will probably emerge as the preferred method because they will tend to have less uncertainty. For example, uncertainty was reduced by 24% when predicting national-scale C stock changes in the United States with process-based models compared to empirically derived factors[77].

Another consideration is that uncertainties in estimating C stock and GHG emissions with process-based models are considerably larger for reporting by single individuals, particularly if the amount of change on an individual farm is small[77]. Aggregation of many farms into larger projects will reduce uncertainties, which could be a viable approach for managing uncertainty and reducing the discounting of incentive payments.

Verification is an independent evaluation of estimated emissions intended to provide confidence that the reported results are correct, but in practice, the requirements for verification are highly variable across different GHG mitigation efforts, ranging from essentially no requirements to annual evaluations[79]. Verification typically focuses on the accuracy of the estimates, and possibly the most stringent approach is an independent set of measurements. Although independent data may be less favoured in terms of costs relative to alternatives, such as expert judgement[79], soil monitoring networks deployed at national or regional scales could produce independent data for evaluating model-based assessments of soil C stock changes and GHG emissions[80] and for model bias adjustment, using empirically based methods[81].

Another approach to verification is to use atmospheric observations of trace-gas concentrations and inverse modelling to estimate fluxes between the atmosphere and land surface[82,83]. This 'top-down' modelling, using a network of tower-based observations of CO_2 concentrations, was used to verify 'bottom-up' inventory modelling based on observed management activities in the largely agricultural region of the central USA[84,85]. Since atmospheric observations integrate all CO_2 fluxes in the region, the inventory included a full assessment of all sources and sinks. However, even with the fully integrated CO_2 flux, it is possible to statistically disaggregate individual sources as part of the analysis, such as contributions from soil C pools to the regional flux[86]. Satellite-based measurements are providing a new source of atmospheric trace-gas data that can be used to estimate land surface fluxes with inverse modelling frameworks[87,88]. Although atmospheric observations and satellite imagery may become a standard for verifying regional inventories in the future, these methods need further testing before operational systems can be reliably deployed.

Conclusions and recommendations

Climate change and GHG mitigation require an 'all of the above' approach[89], where all reduction measures that are feasible, cost-effective and environmentally sustainable should be pursued. For soils, a variety of management practices and technologies are known to reduce emissions and promote C sequestration, most of which also provide environmental co-benefits. The impediment to implementing agricultural soil GHG mitigation strategies more aggressively to date is primarily the feasibility of quantifying and verifying soil mitigation activities[90] in a cost-effective manner. Overcoming this barrier therefore translates into: (1) increasing the acceptance of soil management within compliance and voluntary C markets; (2) reducing costs to governments for providing environment-based subsidies; and (3) meeting the demands of consumers for 'low carbon' products.

Reducing and managing uncertainties are key to both improving predictive models and decision-support tools and the design of effective policies that promote soil-based GHG mitigation. To advance these efforts, several research and development priorities are apparent (Fig. 3). First, support for research site networks of soil flux (N_2O, CH_4) and soil C measurements encompassing a wide variation in management, as well as 'on-farm' soil C monitoring networks[80] needs to be strengthened (see ref. 91 for an example of a large-scale research network measuring soil GHG flux in a geographically distributed field experiment, using uniform protocols, advanced instrumentation and data portals in Europe, and ref. 92 for an example in the USA). Such support should coordinate with basic research (for example, on soil organic matter stabilization processes, N_2O and CH_4 microbiology, plant–microbe interactions, plant breeding and root phenotyping) to advance process understanding, develop new mitigation practices and fill gaps in underrepresented soil- and climate-management systems. High-quality data generated from consistent measurement protocols is critical for evaluating and improving models. These efforts may benefit from development of new sensor technologies, enabling cheaper and quicker soil measurements[93]. Although multiple competing models are needed, both to spur innovation and because no single model will be best in all situations, model development will benefit from greater collaboration and cross-model testing among developers, moving towards a more open-source, community-development approach[94]. Large geospatial databases of soil biophysical properties and climate variables are critical to quantify soil processes accurately across the landscape (Fig. 3). High-resolution soil maps exist in most developed countries (and increasingly in developing countries[95]), and if made publicly available (there is free access to fine-spatial-scale (about 1:15,000 to 1:20,000) soil maps for the USA[96]), would greatly improve capabilities for modelling GHG emissions at scale.

Finally, realizing the potential for climate change mitigation through global soil management requires understanding cultural, political and socioeconomic contexts, and the ways in which widespread, sustained changes in practice can be successfully achieved within it[97,98]. As such, there needs to be a greater level of engagement with the land users themselves, who will be the ones implementing practices that abate GHG emissions and sequester C. Engagement means both education and outreach, highlighting the links between agriculture and GHGs and using innovative strategies[76] (Fig. 3) to involve stakeholders in gathering and using their local knowledge of how the land is being used now and how it might best be used in the future, thus establishing a new paradigm for climate-smart soil management.

Received 9 February 2015; accepted 27 January 2016.

1. Ruddiman, W. F. The anthropogenic greenhouse era began thousands of years ago. *Clim. Change* **61,** 261–293 (2003).
2. Smith, P. *et al.* Agriculture, Forestry and Other Land Use (AFOLU). In *Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* (eds Edenhofer, O. *et al.*) 813–922 (Cambridge Univ. Press, 2014).
3. Tubiello, F. N. *et al.* The contribution of agriculture, forestry and other land use activities to global warming, 1990–2012. *Glob. Change Biol.* **21,** 2655–2660 (2015).
4. Smith, P. Soils and climate change. *Curr. Opin. Environ. Sust.* **4,** 539–544 (2012).
5. *Verified Carbon Standard (VCS)* http://www.v-c-s.org/ (accessed May 2015).
6. *American Carbon Registry (ACR)* http://americancarbonregistry.org/ (accessed May 2015).
7. Lavallée, S. & Plouffe, S. The ecolabel and sustainable development. *Int. J. Life Cycle Assess.* **9,** 349–354 (2004).
8. Kahiluoto, H., Smith, P., Moran, D. & Olesen, J. E. Enabling food security by verifying agricultural carbon sequestration. *Nature Clim. Change* **4,** 309–311 (2014).
9. Ciais, P. *et al.* Carbon and other biogeochemical cycles. In *Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* (eds Stocker, T. F. *et al.*) 465–570 (Cambridge Univ. Press, 2013).
10. *Join the 4‰ Initiative Soils for Food Security and Climate* http://agriculture.gouv.fr/join-40-initiative-soils-food-security-and-climate-0 (accessed 23 Sept 2015).
11. Batjes, N. H. Total carbon and nitrogen in the soils of the world. *Eur. J. Soil Sci.* **47,** 151–163 (1996).
12. Paustian, K. *et al.* Agricultural soil as a C sink to offset CO_2 emissions. *Soil Use Manage.* **13,** 230–244 (1997).
13. West, T. O. & Six, J. Considering the influence of sequestration duration and carbon saturation on estimates of soil carbon capacity. *Clim. Change* **80,** 25–41 (2007).
14. Davidson, E. A. & Ackerman, I. L. Changes in soil carbon inventories following cultivation of previously untilled soils. *Biogeochemistry* **20,** 161–193 (1993).
15. Ogle, S. M., Breidt, F. J. & Paustian, K. Agricultural management impacts on soil organic carbon storage under moist and dry climatic conditions of temperate and tropical regions. *Biogeochemistry* **72,** 87–121 (2005).
16. Conant, R. T., Paustian, K. & Elliott, E. T. Grassland management and conversion into grassland: effects on soil carbon. *Ecol. Appl.* **11,** 343–355 (2001).
17. Guo, L. B. & Gifford, R. M. Soil carbon stocks and land use change: a meta analysis. *Glob. Change Biol.* **8,** 345–360 (2002).
18. IPCC. *Intergovernmental Panel on Climate Change Guidelines for National Greenhouse Gas Inventories* (eds Eggleston, S., Buendia, L., Miwa, K., Ngara, T. & Tanabe, K.) Vol. 4, *Agriculture, Forestry and Other Land Use (AFOLU)* http://www.ipcc-nggip.iges.or.jp/public/2006gl/vol4.html (Institute for Global Environmental Strategies, 2006).
19. Smith, P. *et al.* Greenhouse gas mitigation in agriculture. *Phil. Trans. R. Soc. B* **363,** 789–813 (2008).
A comprehensive analysis of agricultural GHG emissions and mitigation potentials including estimated C price impacts on mitigation activities.
20. Knox, S. H. *et al.* Agricultural peatland restoration: effects of land-use change on greenhouse gas (CO_2 and CH_4) fluxes in the Sacramento-San Joaquin Delta. *Glob. Change Biol.* **21,** 750–765 (2015).
21. Foley, J. *et al.* Solutions for a cultivated planet. *Nature* **478,** 337–342 (2011).
22. Kell, D. 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 (2012).
23. Burney, J. A. *et al.* Greenhouse gas mitigation by agricultural intensification. *Proc. Natl Acad. Sci. USA* **107,** 12052–12057 (2010).
24. Wilhelm, W. W., Johnson, J. M. F., Hatfield, J. L., Voorhees, W. B. & Linden, D. R. Crop and soil productivity response to corn residue removal: a literature review. *Agron. J.* **96,** 1–17 (2004).
25. Poeplau, C. & Don, A. Carbon sequestration in agricultural soils via cultivation of cover crops—a meta-analysis. *Agric. Ecosyst. Environ.* **200,** 33–41 (2015).
This paper combines an analysis of globally distributed field data with simulation modelling to quantify potential soil C increases with adoption of cover crops on previously fallow soils.
26. Tonitto, C., David, M. B. & Drinkwater, L. E. Replacing bare fallows with cover crops in fertilizer-intensive cropping systems: a meta-analysis of crop yield and N dynamics. *Agric. Ecosyst. Environ.* **112,** 58–72 (2006).
27. Lemke, R. L., VandenBygaart, A. J., Campbell, C. A., Lafond, G. P. & Grant, B. Crop residue removal and fertilizer N: effects on soil organic carbon in a long-term crop rotation experiment on a Udic Boroll. *Agric. Ecosyst. Environ.* **135,** 42–51 (2010).
28. Shcherbak, I., Millar, N. & Robertson, G. P. Global meta-analysis of the nonlinear response of soil nitrous oxide (N_2O) emissions to fertilizer nitrogen. *Proc. Natl Acad. Sci. USA* **111,** 9199–9204 (2014).
This paper shows that N_2O emissions are greater than previously thought for soils receiving high rates of N fertilizer.
29. Six, J., Elliott, E. T. & Paustian, K. Soil macroaggregate turnover and microaggregate formation: a mechanism for C sequestration under no-tillage agriculture. *Soil Biol. Biochem.* **32,** 2099–2103 (2000).
30. Powlson, D. S. *et al.* Limited potential of no-till agriculture for climate change mitigation. *Nature Clim. Change* **4,** 678–683 (2014).
31. Kravchenko, A. N. & Robertson, G. P. Whole-profile soil carbon stocks: the danger of assuming too much from analyses of too little. *Soil Sci. Soc. Am. J.* **75,** 235–240 (2011).
32. McSherry, M. E. & Ritchie, M. E. Effects of grazing on grassland soil carbon: a global review. *Glob. Change Biol.* **19,** 1347–1357 (2013).
33. Scholes, M. J. & Scholes, R. J. Dust unto dust. *Science* **342,** 565–566 (2013).
34. Conant, R. T. *et al.* Temperature and soil organic matter decomposition rates—synthesis of current knowledge and a way forward. *Glob. Change Biol.* **17,** 3392–3404 (2011).

35. Cotrufo, M. F., Wallenstein, M., Boot, C. M., Denef, K. & Paul, E. A. The Microbial Efficiency-Matrix Stabilization (MEMS) framework integrates plant litter decomposition with soil organic matter stabilization: do labile plant inputs form stable soil organic matter? *Glob. Change Biol.* **19,** 988–995 (2013).
36. Schmidt, M. W. I. *et al.* Persistence of soil organic matter as an ecosystem property. *Nature* **478,** 49–56 (2011).
37. Stewart, C. E., Plant, A. F., Paustian, K., Conant, R. & Six, J. Soil carbon saturation: linking concept and measurable carbon pools. *Soil Sci. Soc. Am. J.* **72,** 379–392 (2008).
38. Rumpel, C. & Koegel-Knabner, I. Deep soil organic matter—a key but poorly understood component of terrestrial C cycle. *Plant Soil* **338,** 143–158 (2011).
39. Nadeu, E., Gobin, A., Fiener, P., van Wesemael, B. & van Oost, K. Modelling the impact of agricultural management on soil carbon stocks at the regional scale: the role of lateral fluxes. *Glob. Change Biol.* **21,** 3181–3192 (2015).
40. Grandy, A. S. & Neff, J. C. Molecular C dynamics downstream: the biochemical decomposition sequence and its impact on soil organic matter structure and function. *Sci. Total Environ.* **404,** 297–307 (2008).
41. Ryals, R., Hartman, M. D., Parton, W. J., DeLonge, M. & Silver, W. L. Long-term climate change mitigation potential with organic matter management on grasslands. *Ecol. Appl.* **25,** 531–545 (2015).
Field experiments and modelling were used to show reductions in net GHG emissions from grazed annual grasslands using composted green waste applications.
42. Lehmann, J. *et al.* In *Biochar for Environmental Management: Science, Technology and Implementation* (eds Lehmann, J. & Joseph, S.) 235–282 (Taylor and Francis, 2015).
43. Roberts, K., Gloy, B., Joseph, S., Scott, N. & Lehmann, J. Life cycle assessment of biochar systems: estimating the energetic, economic and climate change potential. *Environ. Sci. Technol.* **44,** 827–833 (2010).
44. DeLonge, M. S., Ryals, R. & Silver, W. L. A lifecycle model to evaluate carbon sequestration potential and greenhouse gas dynamics of managed grasslands. *Ecosystems* **16,** 962–979 (2013).
45. Kuzyakov, Y. Priming effects: interactions between living and dead organic matter. *Soil Biol. Biochem.* **42,** 1363–1371 (2010).
46. Zimmerman, A., Gao, B. & Ahn, M. Y. Positive and negative mineralization priming effects among a variety of biochar-amended soils. *Soil Biol. Biochem.* **43,** 1169–1179 (2011).
47. Whitman, T., Zhu, Z. & Lehmann, J. Carbon mineralizability determines interactive effects on mineralization of pyrogenic organic matter and soil organic carbon. *Environ. Sci. Technol.* **48,** 13727–13734 (2014).
48. Whitman, T., Nicholson, C. F., Torres, D. & Lehmann, J. Climate change impact of biochar cook stoves in Western Kenyan farm households: system dynamics model analysis. *Environ. Sci. Technol.* **45,** 3687–3694 (2011).
49. Bouwman, A. F., Boumans, L. J. M. & Batjes, N. H. Emissions of N_2O and NO from fertilized fields: summary of available measurement data. *Glob. Biogeochem. Cycles* **16,** 1058 (2002).
50. Robertson, G. P. & Vitousek, P. M. Nitrogen in agriculture: balancing the cost of an essential resource. *Annu. Rev. Environ. Resour.* **34,** 97–125 (2009).
51. Akiyama, H., Yan, X. Y. & Yagi, K. Evaluation of effectiveness of enhanced-efficiency fertilizers as mitigation options for N_2O and NO emissions from agricultural soils: meta-analysis. *Glob. Change Biol.* **16,** 1837–1846 (2010).
52. Aguilera, E., Lassaletta, L., Sanz Cobeña, A., Garnier, J. & Vallejo Garcia, A. The potential of organic fertilizers and water management to reduce N_2O emissions in Mediterranean climate cropping systems. *Agric. Ecosyst. Environ.* **164,** 32–52 (2013).
53. van Kessel, C. *et al.* Climate, duration, and N placement determine N_2O emissions in reduced tillage systems: a meta-analysis. *Glob. Change Biol.* **19,** 33–44 (2013).
54. Millar, N., Robertson, G. P., Grace, P. R., Gehl, R. J. & Hoben, J. P. Nitrogen fertilizer management for nitrous oxide (N_2O) mitigation in intensive corn (maize) production: an emissions reduction protocol for US Midwest agriculture. *Mitig. Adapt. Strat. Glob. Change* **15,** 185–204 (2010).
55. Robertson, G. P. *et al.* Nitrogen–climate interactions in US agriculture. *Biogeochemistry* **114,** 41–70 (2013).
56. Reay, D. S. *et al.* Global agriculture and nitrous oxide emissions. *Nature Clim. Change* **2,** 410–416 (2012).
57. Conrad, R. The global methane cycle: recent advances in understanding the microbial processes involved. *Environ. Microbiol. Rep.* **1,** 285–292 (2009).
58. Le Mer, J. & Roger, P. Production, oxidation, emission and consumption of methane by soils. *Eur. J. Soil Biol.* **37,** 25–50 (2001).
59. Segers, R. Methane production and methane consumption: a review of processes underlying wetland methane fluxes. *Biogeochemistry* **41,** 23–51 (1998).
60. Suwanwaree, P. & Robertson, G. P. Methane oxidation in forest, successional, and no-till agricultural ecosystems: effects of nitrogen and soil disturbance. *Soil Sci. Soc. Am. J.* **69,** 1722–1729 (2005).
61. Linquist, B. A. *et al.* Reducing greenhouse gas emissions, water use, and grain arsenic levels in rice systems. *Glob. Change Biol.* **21,** 407–417 (2015).
62. Liu, Y. *et al.* Carbon dioxide flux from rice paddy soils in central China: effects of intermittent flooding and draining cycles. *PLoS ONE* **8,** e56562 (2013).
63. Organisation for Economic Co-operation and Development (OECD)–Food and Agriculture Organization (FAO) of the United Nations. *Agricultural Outlook* 2015–2024. http://stats.oecd.org (worldwide rice production for 2000 to 2024 period, accessed March 2016).
64. van Groenigen, K. J., van Kessel, C. & Hungate, B. A. Increased greenhouse-gas intensity of rice production under future atmospheric conditions. *Nature Clim. Change* **3,** 288–291 (2012).
65. Levine, U. *et al.* Agriculture's impact on microbial diversity and associated fluxes of carbon dioxide and methane. *ISME J.* **5,** 1683–1691 (2011).
66. Su, J. *et al.* Expression of barley SUSIBA2 transcription factor yields high-starch low-methane rice. *Nature* **523,** 602–606 (2015).
67. Woolf, D., Amonette, J. E., Street-Perrott, F. A., Lehmann, J. & Joseph, S. Sustainable biochar to mitigate global climate change. *Nature Commun.* **1,** 56 (2010).
This paper calculates the global technical potential of greenhouse gas emission reductions by biochar systems as a function of its effects on soil improvement.
68. Lynch, J. P. & Wojciechowski, T. Opportunities and challenges in the subsoil: pathways to deeper rooted crops. *J. Exp. Bot.* **66,** 2199–2210 (2015).
This paper looks at the potential for breeding plants with more root production and tolerance to 'problem soils', to increase yields, nutrient and water capture, and C sequestration.
69. Smith, S. & De Smet, I. Root system architecture: insights from *Arabidopsis* and cereal crops introduction. *Phil. Trans. R. Soc. B* **367,** 1441–1452 (2012).
70. Conant, R. T., Ogle, S. M., Paul, E. A. & Paustian, K. Measuring and monitoring soil organic carbon stocks in agricultural lands for climate mitigation. *Front. Ecol. Environ* **9,** 169–173 (2011).
71. Paustian, K., Ogle, S. M. & Conant, R. T. In *Handbook of Climate Change and Agroecosystems: Impact, Adaptation and Mitigation* (eds Hillel, D. & Rosenzweig, C.) 307–341 (Imperial College Press, 2011.)
72. Hillier, J. G. A farm-focused calculator for emissions from crop and livestock production. *Environ. Modell. Softw.* **26,** 1070–1078 (2011).
73. Swallow, B. M. & Goddard, T. W. Value chains for bio-carbon sequestration services: lessons from contrasting cases in Canada, Kenya and Mozambique. *Land Use Policy* **31,** 81–89 (2013).
74. National Academy of Sciences. *Verifying Greenhouse Gas Emissions: Methods to Support International Climate Agreements* http://www.nap.edu/download.php?record_id=12883 (eds Pacala, S. *et al.*) 1–110 (National Academies Press, 2010).
75. Paustian, K. *et al.* in *Managing Agricultural Greenhouse Gases: Coordinated Agricultural Research through GraceNet to Address Our Changing Climate* (eds Liebig, M., Franzluebbers, A. & Follett, R.) 251–270 (Academic Press, 2012).
76. Paustian, K. Bridging the data gap: engaging developing country farmers in greenhouse gas accounting. *Environ. Res. Lett.* **8,** 021001 (2013).
This paper proposes the development of mobile apps to 'crowd-source' local-scale knowledge of land use and management data to help improve GHG inventories and project-scale accounting in developing countries.
77. Ogle, S. M. *et al.* Scale and uncertainty in modeled soil organic carbon stock changes for US croplands using a process-based model. *Glob. Change Biol.* **16,** 810–822 (2010).
78. Smith, P. *et al.* Towards an integrated global framework to assess the impacts of land use and management change on soil carbon: current capability and future vision. *Glob. Change Biol.* **18,** 2089–2101 (2012).
79. Bellassen, V. *et al.* Monitoring, reporting and verifying emissions in the climate economy. *Nature Clim. Change* **5,** 319–328 (2015).
80. van Wesemael, B. *et al.* How can soil monitoring networks be used to improve predictions of organic carbon pool dynamics and CO_2 fluxes in agricultural soils? *Plant Soil* **338,** 247–259 (2011).
81. Ogle, S. M., Breidt, F. J., Easter, M., Williams, S. & Paustian, K. An empirically based approach for estimating uncertainty associated with modeling carbon sequestration in soils. *Ecol. Modell.* **205,** 453–463 (2007).
82. Peters, W. *et al.* An atmospheric perspective on North American carbon dioxide exchange: CarbonTracker. *Proc. Natl Acad. Sci. USA* **104,** 18925–18930 (2007).
83. Schuh, A. E. *et al.* Evaluating atmospheric CO_2 inversions at multiple scales over highly-inventoried agricultural landscape. *Glob. Change Biol.* **19,** 1424–1439 (2013).
84. Miles, N. L. *et al.* Large amplitude spatial and temporal gradients in atmospheric boundary layer CO_2 mole fractions detected with a tower-based network in the U.S. upper Midwest. *J. Geophys. Res.* **117,** G01019 (2012).
85. Lauvaux, T. *et al.* Constraining the CO_2 budget of the corn belt: exploring uncertainties from the assumptions in a mesoscale inverse system. *Atmos. Chem. Phys.* **12,** 337–354 (2012).
86. Cooley, D., Breidt, F. J., Ogle, S. M., Schuh, A. E. & Lauvaux, T. A constrained least-squares approach to combine bottom-up and top-down CO_2 flux estimates. *Environ. Ecol. Stat.* **20,** 129–146 (2013).
One of the first studies using independent ground- and atmosphere-based methods to validate GHG emissions estimates for a large agriculturally intensive region.

87. Kadygrov, N. *et al.* Role of simulated GOSAT total column CO_2 observations in surface CO_2 flux uncertainty reduction. *J. Geophys. Res.* **114,** D21208 (2009).
88. Basu, S. *et al.* Global CO_2 fluxes estimated from GOSAT retrievals of total column CO_2. *Atmos. Chem. Phys.* **13,** 8695–8717 (2013).
89. Pacala, S. & Socolow, R. Stabilization wedges: solving the climate problem for the next 50 years with current technologies. *Science* **305,** 968–972 (2004).
90. Alexander, P., Paustian, K., Smith, P. & Moran, D. The economics of soil C sequestration and agricultural emissions abatement. *Soil* **1,** 331–339 (2015).
91. *NitroEurope* http://www.nitroeurope.eu (accessed May 2015).
92. *GRACEnet* http://www.ars.usda.gov/research/programs/programs.htm?np_code=212&docid=21223 (USDA, accessed May 2015).
93. Dhawale, N. M. *et al.* Proximal soil sensing of soil texture and organic matter with a prototype portable mid-infrared spectrometer. *Eur. J. Soil Sci.* **66,** 661–669 (2015).
94. Yeluripati, J. *et al.* Global Research Alliance Modelling Platform (GRAMP): an open web platform for modelling greenhouse gas emissions from agro-ecosystems. *Comp. Elec. Agric.* **111,** 112–120 (2015).
95. Sanchez, P. Digital soil map of the world. *Science* **325,** 680–681 (2009).
96. *The Web Soil Survey* http://websoilsurvey.sc.egov.usda.gov/App/WebSoilSurvey.aspx (accessed May 2015).
97. Awan, M. I., van Oort, P. A. J., Ahmad, R., Bastiaans, L. & Meinke, H. Farmers' views on the future prospects of aerobic rice culture in Pakistan. *Land Use Policy* **42,** 517–526 (2015).
98. Smith, P. *et al.* Policy and technological constraints to implementation of greenhouse gas mitigation options in agriculture. *Agric. Ecosyst. Environ.* **118,** 6–28 (2007).
99. Louwagie, G. S.H., Sammeth, F. & Ratinger, T. The potential of European Union policies to address soil degradation in agriculture. *Land Degrad. Dev.* **22,** 5–17 (2011).
100. Decision No 529/2013/EU of the European Parliament and of the Council of 21 May 2013 on accounting rules on greenhouse gas emissions and removals resulting from activities relating to land use, land-use change and forestry and on information concerning actions relating to those activities. http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=OJ:JOL_2013_165_R_0080_01 (EUR-Lex, Access to European Union Law, accessed May 2015).
101. *Field to Market Alliance* https://www.fieldtomarket.org/ (accessed May 2015).
102. *Unilever Sustainable Agriculture Code* https://www.unilever.com/Images/sac-2015_tcm244-427050_en.pdf (accessed Feb 2016).
103. Horowitz, J. K. & Just, R. E. Economics of additionality for environmental services from agriculture. *J. Environ. Econ. Manage.* **66,** 105–122 (2013).

Acknowledgements We thank A. Swan for figure design. We thank the following organisations for support: USDA/NIFA (grant number 2011-67003-30205 to K.P. and S.O.); USDA/NRCS (grant number CESU-68-7482-15-507 to K.P.); the NSF (grant number DEB 1027253 to G.P.R.); the US DOE (grant number DE-FCO2-07ER64494 to G.P.R.); NERC (grant number NE/M016900/1 to P.S.); and the Belmont Forum/FACCE-JPI (grant number NE/M021327/1 to P.S.).

Author Contributions K.P. led the development of the manuscript and the integration of content. All authors contributed equally to drafting sections of the manuscript and making revisions.

Author Information Reprints and permissions information is available at www.nature.com/reprints. The authors declare no competing financial interests. Readers are welcome to comment on the online version of the paper. Correspondence and requests for materials should be addressed to K.P. (keith.paustian@colostate.edu).