Climate-smart soils

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‘Climate-smart’ soils: a new management paradigm for global agriculture

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Preface

Soils are integral to the function of all terrestrial ecosystems and for sustaining food and fibre production. An overlooked aspect of soils is their potential to mitigate greenhouse gas (GHG) emissions. Although proven practices exist, implementation of soil-based GHG mitigation activities are early-stage and accurately quantifying emissions and reductions remains a significant challenge. Emerging research and information technology developments provide the

potential for broader inclusion of soils in GHG policies. We highlight 'state-of-the-art' soil GHG research, summarize mitigation practices and potentials, identify gaps in data and understanding and suggest ways to close gaps through new research, technology and collaboration.

Introduction

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]. While 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 paper, 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] and new initiatives to market 'low-carbon' products[6] indicate a growing role for agricultural GHG mitigation[7].

Moreover, interest in developing aggressive soil C sequestration strategies has been heightened by recent IPCC assessments, which project that substantial terrestrial C sinks will be needed to supplement large cuts in GHG emissions to achieve GHG stabilization levels of 450ppm CO_2 equivalent or below, consistent with the goal of $<2^{o}$ C mean global temperature increase[8]. Soil C sequestration is one of a few strategies that could be applied at large scales[8] and potentially at low cost; as an example, the French government is proposing a plan to increase soil C concentration in a large portion of agricultural soils globally, by 0.4% per year, producing a C sink increase of 1.2 Pg C yr^{-1}[9].

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 at scale will require 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 to actively engage with land users at the grass-roots, are key elements to 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 (ca. 1500 Pg C to 1 m depth; 2400 Pg C to 2 m depth[10]), 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)[8]. Thus, increasing net soil C storage by even a few percent represents a significant C sink potential.

Proximal controls on the soil C balance include the rate of C addition as plant residue, manure or other organic waste, less the rate of C loss (*via* decomposition); hence, C stocks can be increased by increasing organic matter inputs or by reducing decomposition rates (e.g., by reducing soil disturbance), or both, leading to net removal of C from the atmosphere[11]. However, soil C accrual rates decrease over time as stocks approach a new equilibrium. Thus net CO_2 removals are of limited duration, often attenuating after 2-3 decades[12].

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 significantly higher soil C stocks than their agricultural counterparts, and soil C loss (typically 0.5 to >2 Mg C ha^{-1} yr^{-1}) following land conversion to cropland has been extensively documented[13,14]. Total losses once the soil approaches a new equilibrium are typically ~30-50% of topsoil (e.g. 0-30 cm) C stocks[14]. Hence, avoided 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[15,16]. 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} [17], and can also restore C sequestration (Fig. 1), though methane emissions may increase[18,19]. 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[20].

[Fig 1 about here]

In general, soil C sequestration rates on land maintained in agricultural use are less than for land restoration/conversion, and vary on the order of 0.1 to 1 Mg C ha^{-1} yr^{-1}, as a function of land use history, soil/climate conditions, and the combination of management practices applied [2,14]. Practices that increase C inputs include *(i)* improved varieties or species with greater root mass to deposit C in deeper layers where turnover is slower[21], *(ii)* adopting crop rotations that provide greater C inputs[22], *(iii)* more residue retention[23], and *(iv)* cover crops during fallow periods to provide year-round C inputs (Fig. 1).[22,24] Cover crops can also reduce nutrient losses, including nitrate that is otherwise converted to N_2O in riparian areas and waterways[25] – an example of synergy between practices that sequester C and also tighten the N cycle to limit emissions of N_2O. Other practices to increase C inputs include irrigation in water-limited systems[18] and additional fertilizer input to increase productivity in low-yielding, nutrient deficient systems (Fig. 1)[26]. Although additional nutrient and water inputs to boost yields may increase non-CO_2 emissions[27], 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[20,22].

Some croplands can sequester C through less intensive tillage, particularly zero tillage[14], due to less disruption of soil aggregate structure[28]. Some authors have argued that benefits are small because increased C content in surface horizons are offset by C losses deeper in the profile[29], although others have noted that the larger variability in sub-surface horizons and lack of statistical power in existing studies makes such conclusions questionable[30].

A change from annual to perennial crops typically increases belowground C inputs (and soil disturbance is reduced), leading to C sequestration[15]. In grasslands, soil C sequestration can be increased through optimal stocking/grazing density[31]. Improved management in fire-prone ecosystems *via* fire prevention or prescribed burning can also increase C sequestration[32].

Key knowledge gaps that affect our understanding of soil C sequestration processes and management options to implement them include questions about the differential temperature sensitivity of C turnover among SOM fractions[33], interactions among organic matter chemistry, mineral surface interactions and C saturation[34-36], and subsoil (> 30 cm) SOM accretion, turnover and stabilization[37]. 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[38]. And emerging evidence that stabilized SOM is of microbial rather than direct plant origin[34,39] 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 (i.e., 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-fold greater than un-composted organic matter [40], and biochar mineralizes 10-100 times slower than uncharred biomass [41]. 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[35] (e.g. moisture, temperature, texture).

However, because the organic matter originates from outside the ecosystem 'boundary', a broader life-cycle assessment approach is needed, that considers GHG impacts of: (i) offsite biomass removal, transport, and processing, (ii) alternative end uses of the biomass, (iii) interactions with other soil GHG-producing processes, and (iv) synergies between these soil amendments and the fixation and retention of *in situ* plant-derived C[42,43]. 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[42,43].

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'[44]) has often been observed immediately following biochar addition, but priming usually declines, sometimes becoming negative (i.e., inhibiting *in situ* soil C decomposition), over time [45,46]. Analogous time dependence of soil N_2O and CH_4 emissions has not received sufficient attention[40]. Increased plant growth in amended soils and the resultant feedbacks to soil C can make up a large proportion of the soil-based GHG balance[40,47] 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[2,18]; some 4.2 Tg of a global anthropogenic flux of 8.1 Tg N_2O-N yr^{-1}. Reducing this flux represents a significant 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).

N_2O is produced in soils by microbial activity – mainly nitrification and denitrification – which occur 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 [48], which is the basis for estimating emissions using default IPCC methods[17]. 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[27]. 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/downwind from the field[49].

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 up to 40% in some soils, although other soils show little reduction and more research is needed to understand variable site-level responses[50]. Likewise, tillage and water management can affect N_2O fluxes by altering the soil microenvironment[51,52].

Another means for 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. By one recent study[53], corn farmers in the U.S. Midwest 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[49].

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[54]. 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[55], 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}) [8] of global methane (CH_4) emissions occur through the microbial breakdown of organic compounds in soils under anaerobic conditions[56]. As such, wetlands (177-284 Tg yr^{-1}) and rice cultivation (33-40 Tg yr^{-1}) [8] 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 [57].

Key determinants of soil CH_4 fluxes include aeration, substrate availability, temperature and N inputs[58]; 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[59]. 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[18], although cycles of wetting and drying of soils may also enhance N_2O production[60] and soil C mineralisation[61], thereby reducing the net mitigation effect.

With global rice production projected to expand by ~40% between 2000-2023 [62], the potential for further GHG mitigation via soil management appears large, although the global distribution and diverse nature of rice production systems – including irrigated, rain-fed and deepwater – 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[63].

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[64], 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[65]. 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[17], 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.

[Fig 2. about here]

Global potential for soil GHG mitigation

How significant, 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-related practices by Smith et al. [18] 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 social incentive to pay for mitigation). They estimated total soil GHG mitigation potential ranging from 5.3 Pg CO_2eq yr^{-1} (absent economic constraints) to 1.5 Pg CO_2eq yr^{-1} at the lowest specified C price ($20 per Mg CO_2eq). Average rates for the majority of management interventions are modest, < 1 Mg CO_2eq ha^{-1} yr^{-1}. Thus, achieving globally significant 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 Smith et al.[18]) are less well known, Woolf et al. [66] estimated a global technical potential of 1-1.8 Pg CO_2eq 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[21,67]. 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 substantially alter root characteristics[68] 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

[67]. Greater root C inputs is well-recognized as a main reason for the higher soil C stocks maintained under perennial grasses compared to annual crops [15]. 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 ~1 Pg CO_2eq yr^{-1} or more if applied over a large portion on global cropland area (Fig 2).

Hence, the overall mitigation potential of existing (and potential future) soil management practices could be as high as ~8 Pg CO_2eq 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 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. Thus engaging a significant number of these people is a massive undertaking in itself. Furthermore, agricultural soil GHG emissions are challenging to quantify due 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[52,69,70]. 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[69].

[Box 1 about here]

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 labeling' 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[6,71].

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 (i.e., 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 capture 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 [5,52]. Here, accurately quantifying C sequestration and/or emission reductions is more challenging due 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[72]. In contrast to projects involving major land cover changes, where remote sensing can provide much of the activity monitoring (e.g., retention of forested land over time), such options are poorly-suited for monitoring crop type, fertilizer, residue and water management, and organic matter amendments[73]; for such practices the best source 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 [71], being used by farmers participating in low C supply chain management, and the COMET-Farm tool, which allows farmers to compute full farm-scale GHG budgets, for support of government-sponsored conservation initiatives and participation in mitigation projects[74]. 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 existing infrastructure to collect and manage land use data is weak[75].

[Fig. 3 about here]

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[76] (e.g., weather, management and soils data). Empirical models generally rely on statistical analyses of measurement data to produce emission factors, along with an estimated uncertainty[14]. 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 programs[77].

Monitoring, reporting and verification (MRV) systems are a key element in a climate smart agricultural program. While MRV systems place different levels of importance on uncertainty depending on program type (see Box 1)[78], discounting payments based on the level of uncertainty is likely to be part of programs with financial incentives, such as cap-and-trade. Discounting encourages monitoring efforts to reduce uncertainty over time[17]. If discounting payments for C sequestration and emission reduction practices with larger uncertainty is adopted in climate smart agriculture programs, then more advanced methods with process-based models will likely emerge as the preferred method due to 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[76].

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[76]. Aggregation of many farms into larger projects will reduce uncertainties, which could be a viable approach for managing uncertainty and reducing 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, from essentially no requirements to annual evaluations[78]. 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 favored in terms of costs relative to alternatives, such as expert judgement[78], 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[79] and for model bias adjustment, using empirically-based methods[80].

Another approach to verification is to use atmospheric observations of trace gas concentrations and inverse modeling to estimate fluxes between the atmosphere and land surface[81,82]. This 'top-down' modeling, utilizing a network of tower-based observations of CO_2 concentrations, was used to verify 'bottom-up' inventory modeling based on observed management activities, in the largely agricultural region of the central United States[83,84]. 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[85]. Satellite-based measurements are providing a new source of atmospheric trace gas data that can be used to estimate land surface fluxes with inverse modeling frameworks[86,87]. While atmospheric observations and satellite imagery may become a standard for verifying regional inventories in the future, the methods need further testing in the near term before deploying operational systems.

Conclusions and way forward

Climate change and GHG mitigation require an 'all of the above' approach[88], 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. Impediments to more aggressively implementing agricultural soil GHG mitigation strategies to date are primarily the feasibility of cost-effectively quantifying and verifying soil mitigation activities[89]. Overcoming these barriers therefore translates into: i) increasing the acceptance of soil management within compliance and voluntary C markets, ii) reducing costs to governments for providing environmental-based subsidies, and iii) meeting demands of consumers for 'low carbon' products.

Reducing and managing uncertainties are key to both improved 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[90] encompassing a wide variation in management, as well as 'on-farm' soil C monitoring networks[79] needs to be strengthened, in coordination with basic research (e.g., on SOM 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 for underrepresented soil/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[91]. While 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[92]. Large geospatial databases of soil biophysical properties and climate variables are critical to accurately quantify soil processes across the landscape (Fig. 3). igh resolution soil maps exist in most developed countries (and increasingly in developing countries[93]), and if made publically available[94], would greatly improve capabilities for modeling GHG emission at scale.

Finally, realising 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 [95,96]. As such, there needs to be 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 utilizing innovative strategies[75] (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, establishing a new paradigm for climate-smart soil management.

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Figure text.

Fig. 1. A potential decision-tree ordering management practices towards creating GHG mitigating cropland (rice not included). For degraded, marginal lands (top of diagram) the most productive mitigation option is conversion to perennial vegetation either left unmanaged or sustainably harvested to offset fossil energy use (cellulosic biofuels). For more arable lands, multiple options could be implemented sequentially or in combination, depending on management objectives, cost and other constraints. Practices shown (see text for more discussion) are roughly arrayed from lower cost/higher feasibility options towards more costly interventions (bottom of figure). However, low cost options in one region may be a higher cost/less feasible option in another region. All options require a region-specific full-cost carbon accounting (GHG life cycle analysis) that includes potential indirect land use effects in order to define specific mitigation potentials. *Relative costs, provided as examples based on a developed region such as North America and a less developed region such as sub-Saharan Africa. †Denotes potential for major co-benefits as non-GHG ecosystem services. ‡Potential constraints that might limit or preclude practice adoption as well as potential increases in other GHGs as a consequence of practice adoption.

Fig. 2. Global potential for agricultural-based GHG mitigation, relating average per ha net GHG reduction rates and potential area (in Mha) of adoption (note log-scales). Unless otherwise noted, estimates are from Smith et al.[18] based on cropland and grassland area projections for 2030. Ranges in total Pg CO_2eq yr^{-1} represent varying adoption rates as a function of C pricing ($20, $50, and $100 per Mg CO2_eq), to a maximum technical potential, i.e., full implementation of practices on the available land base. Multiple practices are aggregated for

cropland (e.g. improved crop rotations and nutrient management, reduced tillage) and grazing land (e.g., grazing management, nutrient and fire management, species introduction) categories. Practices that increase net soil C stocks and/or reduce emissions of N_2O and CH_4 are combined in each practice category. The portion of projected mitigation from C stock increase (ca. 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[18]. Estimates for biochar application from Woolf et al.[66] represent a technical potential only, but 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 ~1 Pg CO_2eq yr^{-1} is shown, using as an analog, global average C accrual rates (0.23 Mg C ha^{-1} yr^{-1}) for cover crops[24], applied to 50% of the cropland land area used by Smith et al.[18].

Fig. 3. Expanding the role for agricultural soil GHG mitigation will require an integrated research support and implementation platform. Targeted basic research on soil processes (a few examples of priority areas shown here), expanding measurement/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 implementation of climate-smart soil management policies, via cap-and-trade systems, product supply chain initiatives for 'low-carbon' consumer products, national and international

GHG mitigation policies and also promote more sustainable and climate-resilient agricultural systems, globally.

[BOX 1]

Implementation strategies for soil GHG mitigation

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

1) Regulation/taxation: Direct regulatory measures to reduce soil GHGs at the entity scale are likely politically unfeasible and costly. Taxation of N fertilizer, already used in parts of the US and Europe to reduce nitrate pollution, could function as an indirect tax to reduce N_2O emissions.

2) Subsidies: Targeted government payments/subsidies for implementing GHG-reducing practices is emerging as a policy alternative. For example, US Dept. of Agriculture programs 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[97]. 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 [98].

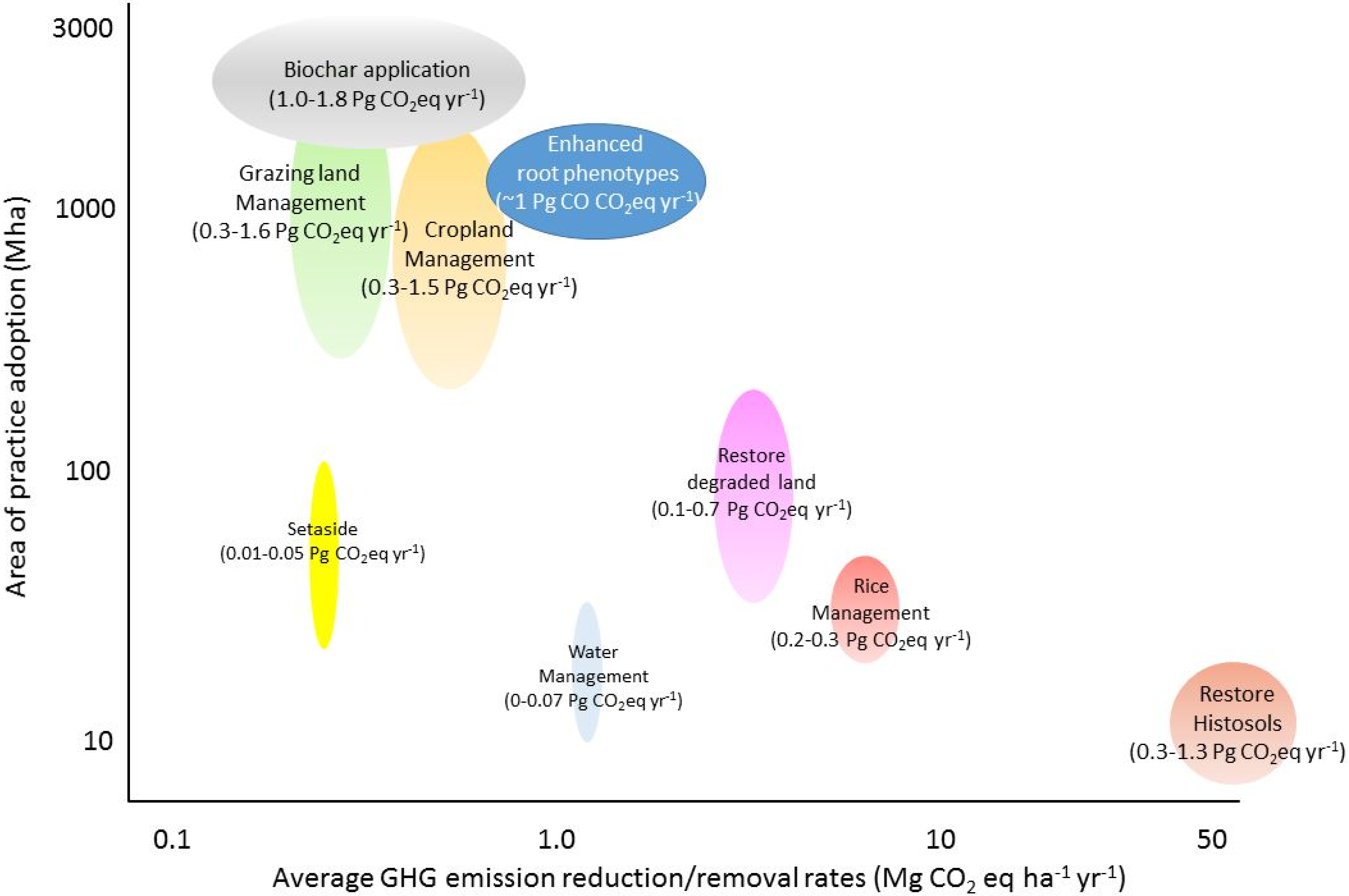
3) Supply chain initiatives: Major food distributors are targeting sustainability metrics, including low GHG footprints, as a consumer marketing strategy[99], setting performance standards for contracted agricultural producers, including requiring field-scale monitoring of production practices and quantification of GHG emissions.

4) Cap and trade (C&T): In a C&T 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 and/or by purchasing surplus permits from capped entities that have exceeded their required reductions. Both compliance and voluntary markets can function as C&T systems[100]. Within many C&T systems, a limited amount of emission

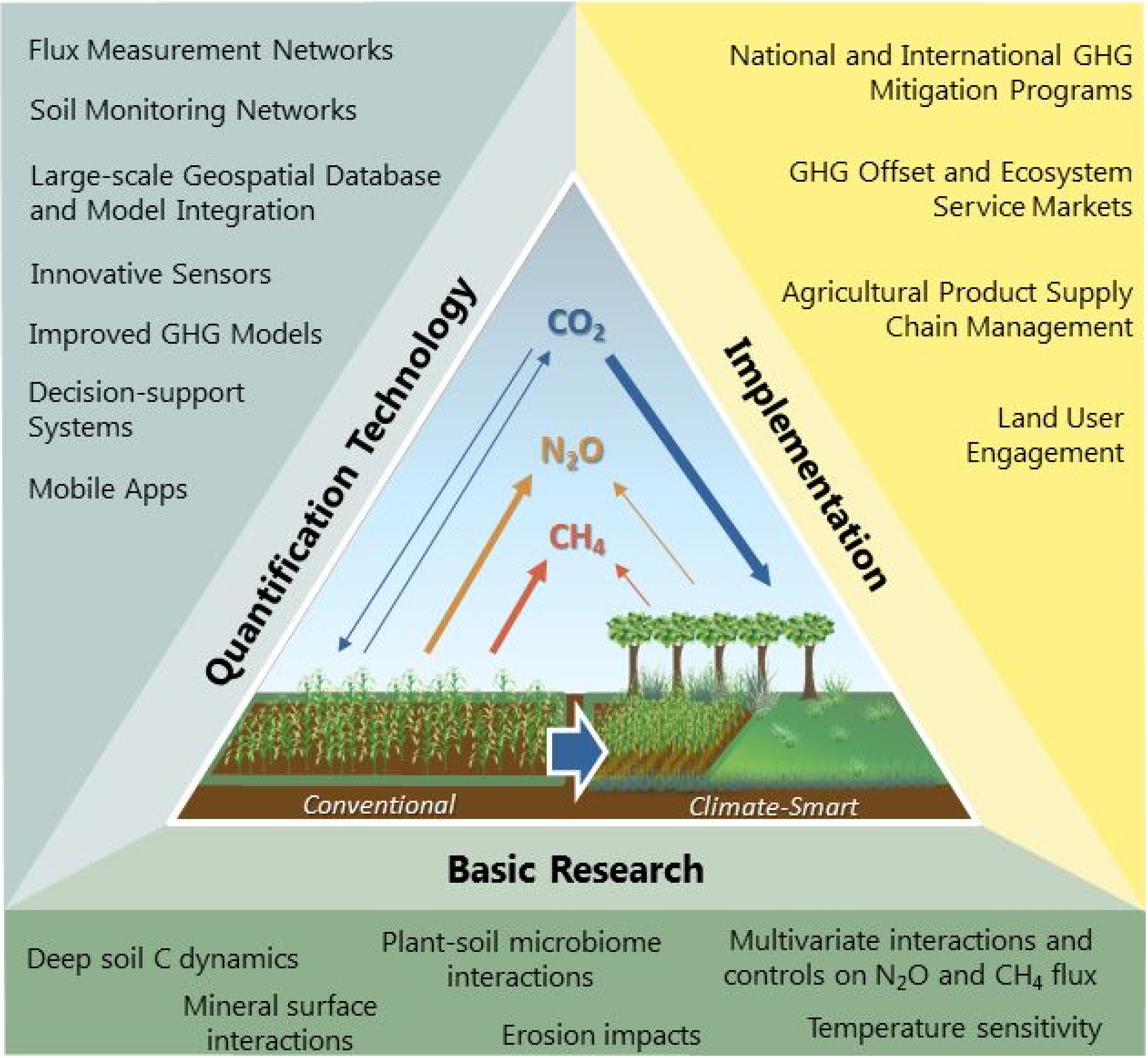
reductions (termed ‘offsets’) can be provided by non-capped entities. Inclusion of agricultural activities as offset providers has been growing, particularly within voluntary markets. To maintain the integrity of emission caps, key criteria for offset providers include demonstrating *additionality*, i.e., insuring that reductions result from project interventions and not simply business-as-usual trends, avoiding *leakage*, i.e., unintended emission increases elsewhere as a consequence of the project activities, and providing for *permanence* (e.g., that increased soil C storage, credited as a CO_2 removal, is maintained long-term).

[End BOX 1]

Cropland characteristics		Mitigation practices	Practice Co-benefits*	Relative expense† Developed	Relative expense† Less Developed	Constraints/caveats‡
Degraded or marginal land?	yes →	***Convert to perennial set-aside or cellulosic biofuel***	↓soil erosion ↑biodiversity ↑water quality	$$	$$	Alternate land/livelihood for subsistence farmers Opportunity cost of removing land base Potential for leakage (i.e., land use change impact)
no ↓ **Drained, cropped organic (histosol) soils?**	yes →	***Restore to wetland***	↑biodiversity ↑water quality	$$$	$$$	High opportunity costs of lost crop production Potential ↑CH_4 emissions Potential for leakage (i.e., land use change impact)
no ↓ **Severe nutrient deficiency?**	yes →	*Nutrient additions, liming, N-fixing species*	↑food security ↑water quality	$	$$	Availability/access to fertilizer Potential ↑N_2O emissions
no ↓ **Extensive bare fallow?**	yes →	*Cover crops, reduced fallow vegetated fallow*	↓soil erosion ↑water quality ↑soil health ↑food security	$	$$	Limited applicability in dry areas
no ↓ **Excess N fertilizer use?**	yes →	*Reduce to economic-optimal rates*	↑water quality	$	$	Risk of crop production loss
no ↓ **Intensive tillage?**	yes →	*Reduced till, no-tillage, residue retention*	↓soil erosion ↑soil health	$	$$	Limited applicability in cold climates Potential increased equipment cost Increased herbicide use
no ↓ **Suboptimal N management?**	yes →	*Improve timing, placement; enhanced efficiency fertilizer*	↑water quality	$$	$$$	Availability/access to enhanced efficiency fertilizer
no ↓ **Low residue crops?**	yes →	*Perennials in rotation, agroforestry, high C input species, cover crops*	↑biodiversity ↑soil health ↑food security	$$$	$$	Less applicability in dry areas, shallow soils Potential opportunity costs of lost crop production
no ↓ **Available exogenous organic amendments?**	yes →	*Add amendments (e.g. compost, biochar)*	↑soil health ↑food security	$$$	$$	Dependent on lifecycle emissions of producing the amendment
↓ ***High capacity GHG mitigation on cropland***						



3000
1000
100
10
Area of practice adoption (Mha)
Biochar application
(1.0-1.8 Pg CO_2eq yr^{-1})
Grazing land
Management
(0.3-1.6 Pg CO_2eq yr^{-1})
Cropland
Management
(0.3-1.5 Pg CO_2eq yr^{-1})
Enhanced
root phenotypes
(~1 Pg CO CO_2eq yr^{-1})
Setaside
(0.01-0.05 Pg CO_2eq yr^{-1})
Water
Management
(0-0.07 Pg CO_2eq yr^{-1})
Restore
degraded land
(0.1-0.7 Pg CO_2eq yr^{-1})
Rice
Management
(0.2-0.3 Pg CO_2eq yr^{-1})
Restore
Histosols
(0.3-1.3 Pg CO_2eq yr^{-1})
0.1
1.0
10
50
Average GHG emission reduction/removal rates (Mg CO_2 eq ha^{-1} yr^{-1})



Flux Measurement Networks
Soil Monitoring Networks
Large-scale Geospatial Database and Model Integration
Innovative Sensors
Improved GHG Models
Decision-support Systems
Mobile Apps
Quantification Technology
National and International GHG Mitigation Programs
GHG Offset and Ecosystem Service Markets
Agricultural Product Supply Chain Management
Land User Engagement
Implementation
CO_2
N_2O
CH_4
Conventional
Climate-Smart
Basic Research
Deep soil C dynamics
Plant-soil microbiome interactions
Multivariate interactions and controls on N_2O and CH_4 flux
Mineral surface interactions
Erosion impacts
Temperature sensitivity