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

2

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17

18 **Preface**

19 Soils are integral to the function of all terrestrial ecosystems and for sustaining food and fibre

20 production. An overlooked aspect of soils is their potential to mitigate greenhouse gas (GHG)

21 emissions. Although proven practices exist, implementation of soil-based GHG mitigation

22 activities are early-stage and accurately quantifying emissions and reductions remains a

23 significant challenge. Emerging research and information technology developments provide the

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

28

29 **Introduction**

30 Evidence points to agriculture as the first instance of human-caused increases in greenhouse
31 gases (GHGs), several thousand years ago¹. Agriculture and associated land use change remain a
32 source for all three major biogenic GHGs -- carbon dioxide (CO₂), methane (CH₄), and nitrous
33 oxide (N₂O). Land use contributes ~25% of total global anthropogenic GHG emissions: 10-14%
34 directly from agricultural production, mainly via GHG emissions from soils and livestock
35 management, and another 12-17% from land cover change, including deforestation^{2,3}. While
36 soils contribute a major share (37%; mainly as N₂O and CH₄) of agricultural emissions³,
37 improved soil management can substantially reduce these emissions and sequester some of the
38 CO₂ removed from the atmosphere by plants, as carbon (C) in soil organic matter (in this paper,
39 our discussion of soil C refers solely to organic C). In addition to decreasing GHG emissions
40 and sequestering C, wise soil management that increases organic matter and tightens the soil
41 nitrogen (N) cycle can yield powerful synergies, such as enhanced fertility and productivity,
42 increased soil biodiversity, reduced erosion, runoff and water pollution, and can help buffer crop
43 and pasture systems against the impacts of climate change⁴.

44

45 The inclusion of soil-centric mitigation projects within GHG offset markets⁵ and new initiatives
46 to market ‘low-carbon’ products⁶ indicate a growing role for agricultural GHG mitigation⁷.

47 Moreover, interest in developing aggressive soil C sequestration strategies has been heightened
48 by recent IPCC assessments, which project that substantial terrestrial C sinks will be needed to
49 supplement large cuts in GHG emissions to achieve GHG stabilization levels of 450ppm CO₂
50 equivalent or below, consistent with the goal of <2° C mean global temperature increase⁸. Soil C
51 sequestration is one of a few strategies that could be applied at large scales⁸ and potentially at
52 low cost; as an example, the French government is proposing a plan to increase soil C
53 concentration in a large portion of agricultural soils globally, by 0.4% per year, producing a C
54 sink increase of 1.2 Pg C yr⁻¹[9].

55

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

65

66 **Process controls and mitigation practices**

67 Soil C sequestration via improved management

68 Soils constitute the largest terrestrial organic C pool (ca. 1500 Pg C to 1 m depth; 2400 Pg C to 2
69 m depth¹⁰), which is three times the amount of CO₂ currently in the atmosphere (~830 Pg C) and

70 240 times current annual fossil fuel emissions ($\sim 10 \text{ Pg}$)⁸. Thus, increasing net soil C storage by
71 even a few percent represents a significant C sink potential.

72

73 Proximal controls on the soil C balance include the rate of C addition as plant residue, manure or
74 other organic waste, less the rate of C loss (*via* decomposition); hence, C stocks can be increased
75 by increasing organic matter inputs or by reducing decomposition rates (e.g., by reducing soil
76 disturbance), or both, leading to net removal of C from the atmosphere¹¹. However, soil C
77 accrual rates decrease over time as stocks approach a new equilibrium. Thus net CO₂ removals
78 are of limited duration, often attenuating after 2-3 decades¹².

79

80 Unmanaged forests and grasslands typically allocate a large fraction of their biomass production
81 belowground and their soils are relatively undisturbed; accordingly, native ecosystems usually
82 support significantly higher soil C stocks than their agricultural counterparts, and soil C loss
83 (typically 0.5 to $>2 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$) following land conversion to cropland has been extensively
84 documented^{13,14}. Total losses once the soil approaches a new equilibrium are typically $\sim 30\text{-}50\%$
85 of topsoil (e.g. 0-30 cm) C stocks¹⁴. Hence, avoided conversion and degradation of native
86 ecosystems is a strong mitigation alternative. Conversely, restoration of marginal or degraded
87 lands to perennial forest or grassland increases soil C storage (Fig. 1), although usually at a
88 slower rate than the original conversion losses^{15,16}. Restoring wetlands that have been drained for
89 agricultural use reduces ongoing decomposition losses, which can be as high as $5\text{-}20 \text{ Mg C ha}^{-1}$
90 yr^{-1} [17], and can also restore C sequestration (Fig. 1), though methane emissions may
91 increase^{18,19}. Land use conversions may, however, conflict with agricultural production and food
92 security objectives, entailing the need for a broad-based accounting of net GHG implications²⁰.

93

94 [Fig 1 about here]

95

96 In general, soil C sequestration rates on land maintained in agricultural use are less than for land
97 restoration/conversion, and vary on the order of 0.1 to 1 Mg C ha⁻¹ yr⁻¹, as a function of land use
98 history, soil/climate conditions, and the combination of management practices applied^{2,14}.

99 Practices that increase C inputs include (i) improved varieties or species with greater root mass
100 to deposit C in deeper layers where turnover is slower²¹, (ii) adopting crop rotations that provide
101 greater C inputs²², (iii) more residue retention²³, and (iv) cover crops during fallow periods to
102 provide year-round C inputs (Fig. 1).^{22,24} Cover crops can also reduce nutrient losses, including
103 nitrate that is otherwise converted to N₂O in riparian areas and waterways²⁵ – an example of
104 synergy between practices that sequester C and also tighten the N cycle to limit emissions of
105 N₂O. Other practices to increase C inputs include irrigation in water-limited systems¹⁸ and
106 additional fertilizer input to increase productivity in low-yielding, nutrient deficient systems
107 (Fig. 1)²⁶. Although additional nutrient and water inputs to boost yields may increase non-CO₂
108 emissions²⁷, the emissions intensity of the system (GHG emissions per unit yield) may decline,
109 providing a global benefit if the yield increase avoids land conversion for agriculture
110 elsewhere^{20,22}.

111

112 Some croplands can sequester C through less intensive tillage, particularly zero tillage¹⁴, due to
113 less disruption of soil aggregate structure²⁸. Some authors have argued that benefits are small
114 because increased C content in surface horizons are offset by C losses deeper in the profile²⁹,

115 although others have noted that the larger variability in sub-surface horizons and lack of
116 statistical power in existing studies makes such conclusions questionable³⁰.

117

118 A change from annual to perennial crops typically increases belowground C inputs (and soil
119 disturbance is reduced), leading to C sequestration¹⁵. In grasslands, soil C sequestration can be
120 increased through optimal stocking/grazing density³¹. Improved management in fire-prone
121 ecosystems *via* fire prevention or prescribed burning can also increase C sequestration³².

122

123 Key knowledge gaps that affect our understanding of soil C sequestration processes and
124 management options to implement them include questions about the differential temperature
125 sensitivity of C turnover among SOM fractions³³, interactions among organic matter chemistry,
126 mineral surface interactions and C saturation³⁴⁻³⁶, and subsoil (> 30 cm) SOM accretion, turnover
127 and stabilization³⁷. Landscape processes, particularly the impact of erosion and lateral transport
128 of C in sediments, contribute additional uncertainty on net sequestration occurring at a specific
129 location³⁸. And emerging evidence that stabilized SOM is of microbial rather than direct plant
130 origin^{34,39} may offer a potential to manipulate the soil-plant microbiome to enhance C
131 sequestration in the rhizosphere.

132

133 Soil C sequestration via exogenous C inputs

134 Addition of plant-derived C from external (i.e., offsite) sources such as composts or biochar can
135 increase soil C stocks, and may result in net CO₂ removals from the atmosphere (Fig. 1). Both
136 compost and biochar are more slowly decomposed compared to fresh plant residues, with
137 composts typically having mean residence times several-fold greater than un-composted organic

138 matter⁴⁰, and biochar mineralizes 10-100 times slower than uncharred biomass⁴¹. Thus a large
139 fraction of added C — particularly for biochar — can be retained in the soil over several decades
140 or longer, although residence times vary depending on the amendment type, nutrient content and
141 soil conditions³⁵ (e.g. moisture, temperature, texture).

142
143 However, because the organic matter originates from outside the ecosystem ‘boundary’, a
144 broader life-cycle assessment approach is needed, that considers GHG impacts of: (i) offsite
145 biomass removal, transport, and processing, (ii) alternative end uses of the biomass, (iii)
146 interactions with other soil GHG-producing processes, and (iv) synergies between these soil
147 amendments and the fixation and retention of *in situ* plant-derived C^{42,43}. In many cases, net life-
148 cycle emissions will largely depend on whether the biomass used as a soil amendment would
149 have otherwise been burnt (either for fuel, thereby offsetting fossil fuel use, or as waste
150 disposal), added to a landfill, or left in place as living biomass or detritus^{42,43}.

151
152 While slower mineralization of the amendment is an important determinant of net mitigation
153 impact, effects on other soil emissions cannot be neglected. Mineralization of existing soil C in
154 response to amendments (often referred to as ‘priming’⁴⁴) has often been observed immediately
155 following biochar addition, but priming usually declines, sometimes becoming negative (i.e.,
156 inhibiting *in situ* soil C decomposition), over time^{45,46}. Analogous time dependence of soil N₂O
157 and CH₄ emissions has not received sufficient attention⁴⁰. Increased plant growth in amended
158 soils and the resultant feedbacks to soil C can make up a large proportion of the soil-based GHG
159 balance^{40,47} and these feedbacks may be especially important for more persistent amendments,
160 because of the longer duration of any effects.

161

162 Soil management to reduce N₂O emissions

163 Arable soils emit more N₂O to the atmosphere than any other anthropogenic source^{2,18}; some 4.2
164 Tg of a global anthropogenic flux of 8.1 Tg N₂O-N yr⁻¹. Reducing this flux represents a
165 significant mitigation opportunity, particularly since N₂O is often the major source of radiative
166 forcing in intensively managed cropland. Better N management to reduce emissions would also
167 ameliorate other environmental problems such as nitrate pollution of ground and surface waters
168 caused by excess reactive N in agroecosystems (Fig. 1).

169

170 N₂O is produced in soils by microbial activity – mainly nitrification and denitrification – which
171 occur readily when stimulated by the abundant N that cycles rapidly in virtually all
172 agroecosystems. During nitrification, ammonium added as fertilizer, fixed from the atmosphere
173 by legumes, or mineralized from soil organic matter, crop residue, or other inputs is oxidized to
174 nitrite and eventually to nitrate in a series of reactions that can also produce N₂O. Likewise,
175 when denitrifiers use nitrate as an electron acceptor when soil oxygen is low, N₂O is an
176 intermediate product that can readily escape to the atmosphere.

177

178 Arable soils managed to support high crop productivity have the capacity to produce large
179 quantities of N₂O, and fluxes are directly related to N inputs. On average, about 1% of the N
180 applied to cropland is directly emitted as N₂O⁴⁸, which is the basis for estimating emissions
181 using default IPCC methods¹⁷. However, recent evidence suggests that this value is too high for
182 crops that are under-fertilized and too low for crops that are fertilized liberally²⁷. When crops
183 compete with microbes for available N, N₂O fluxes are lower. In addition to direct in-field

184 emissions, high N applications cause N losses from leaching and volatilization that contribute to
185 ‘indirect’ N₂O emissions, downstream/downwind from the field⁴⁹.

186
187 Since N₂O has no significant terrestrial sink, abatement is best achieved by attenuating known
188 sources of N₂O emissions, by altering the environmental factors that affect N₂O production (soil
189 N, oxygen, and C) or by biochemically inhibiting conversion pathways using soil additives. For
190 example, nitrification can be inhibited with commercial additives such as nitrapyrin and
191 dicyandiamide, which slow ammonium oxidation, and field experiments suggest that inhibitors
192 can reduce N₂O fluxes up to 40% in some soils, although other soils show little reduction and
193 more research is needed to understand variable site-level responses⁵⁰. Likewise, tillage and
194 water management can affect N₂O fluxes by altering the soil microenvironment^{51,52}.

195
196 Another means for reducing N₂O emissions from arable soils is more precise N management to
197 minimize excess N not used by the crop, while maintaining sustainable high yields. Fertilized
198 crops typically take up less than 50% of the N applied; the remainder is available for loss. By
199 one recent study⁵³, corn farmers in the U.S. Midwest could reduce N₂O loss by 50% with more
200 conservative fertilizer practices. Nitrogen conservation can be achieved by: (1) better matching
201 application rates of N to crop needs using advanced statistical and quantitative modelling; (2)
202 applying fertilizer at variable rates across a field based on natural patterns of soil fertility, or
203 within the root zone rather than broadcast on the soil surface; and (3) applying fertilizer close to
204 when the crop can use it, such as several weeks after planting, or adding it earlier but using slow-
205 release coatings to delay its dissolution⁴⁹.

206

207 High temporal and spatial variability make predictions of changes in N₂O fluxes in response to
208 management surprisingly difficult. Particularly lacking are empirical data for multi-intervention
209 strategies that may interact in unexpected ways. Aligned to this paucity are gaps in our
210 understanding of how N cycling and net N₂O flux in managed soils will respond to future climate
211 change⁵⁴. The limited number of field manipulation studies to date indicate that changing
212 temperature and precipitation patterns may have large and strongly-coupled effects on net N₂O
213 emissions⁵⁵, yet our understanding of the processes that underpin these effects and their robust
214 representation in models is far from complete.

215

216 Soil management to reduce CH₄ emissions

217 More than one-third (>200 Tg yr⁻¹)⁸ of global methane (CH₄) emissions occur through the
218 microbial breakdown of organic compounds in soils under anaerobic conditions⁵⁶. As such,
219 wetlands (177-284 Tg yr⁻¹) and rice cultivation (33-40 Tg yr⁻¹)⁸ represent the largest soil-
220 mediated sources of CH₄ globally. In contrast, well-aerated soils act as sinks for CH₄ (estimated
221 at ~ 30 Tg yr⁻¹) from the atmosphere *via* CH₄ oxidation, the bulk of this net sink being in
222 unmanaged upland and forest soils⁵⁷.

223

224 Key determinants of soil CH₄ fluxes include aeration, substrate availability, temperature and N
225 inputs⁵⁸; therefore, soil management can radically alter CH₄ fluxes. For example, in most soils,
226 conversion to agriculture severely restricts CH₄ oxidation, related to the suppression of
227 methanotrophs by accelerated N cycling⁵⁹. In flooded rice, alterations in drainage regimes and
228 organic residue incorporation could reduce emissions by ~ 25% or 7.6 Tg CH₄ yr⁻¹ globally¹⁸,

229 although cycles of wetting and drying of soils may also enhance N₂O production⁶⁰ and soil C
230 mineralisation⁶¹, thereby reducing the net mitigation effect.

231

232 With global rice production projected to expand by ~40% between 2000-2023 [62], the potential
233 for further GHG mitigation via soil management appears large, although the global distribution
234 and diverse nature of rice production systems – including irrigated, rain-fed and deepwater –
235 present challenges to developing effective mitigation strategies. For longer-term (>20 year)
236 projections, climate change and land-atmosphere interactions become increasingly important,
237 with changes in N inputs, temperature, precipitation and atmospheric CO₂ concentration all
238 likely to affect net CH₄ fluxes from soils⁶³.

239

240 This uncertainty highlights important gaps in understanding key processes and their underlying
241 controls. The restoration of soil CH₄ uptake following agricultural conversion, for example,
242 appears related to methanotroph community diversity⁶⁴, about which we know too little.

243 Likewise the abatement of CH₄ generation in rice rhizospheres is related to C compounds exuded
244 by roots, such that CH₄ mitigation might be achieved through further rice breeding and
245 genetics⁶⁵. Limited availability of field-scale CH₄ flux data means a greater reliance on
246 regionally-averaged emission factors and extrapolation from mesocosm and laboratory
247 incubations¹⁷, and thus less site and condition specificity in modelling fluxes. Importantly,
248 establishing the net climate forcing effects of any intervention is a prime target for future soil
249 management research.

250

251 [Fig 2. about here]

252

253 **Global potential for soil GHG mitigation**

254 How significant, in total, is this large, varied set of land use and management practices as a GHG
255 mitigation strategy? One of the challenges in answering this question is to distinguish between
256 what is technically feasible and what might be achieved given economic, social and policy
257 constraints. A comprehensive global analysis of agricultural-related practices by Smith et al.¹⁸
258 combined climate-stratified modelling of emission reductions and soil C sequestration with
259 economic and land use change models to estimate mitigation potential as a function of varying
260 'C prices' (reflecting social incentive to pay for mitigation). They estimated total soil GHG
261 mitigation potential ranging from 5.3 Pg CO₂eq yr⁻¹ (absent economic constraints) to 1.5 Pg
262 CO₂eq yr⁻¹ at the lowest specified C price (\$20 per Mg CO₂eq). Average rates for the majority
263 of management interventions are modest, < 1 Mg CO₂eq ha⁻¹ yr⁻¹. Thus, achieving globally
264 significant GHG reductions requires a substantial proportion of the agricultural land-base (Fig.
265 2). Although the economic and management constraints on biochar additions (not assessed by
266 Smith et al.¹⁸) are less well known, Woolf et al.⁶⁶ estimated a global technical potential of 1-1.8
267 Pg CO₂eq yr⁻¹ (Fig. 2).

268

269 A more unconventional intervention that has been proposed is the development of crops with
270 larger, deeper root systems, hence increasing plant C inputs and soil C sinks^{21,67}. Increasing root
271 biomass and selecting for root architectures that store more C in soils has not previously been an
272 objective for crop breeders, although most crops have sufficient genetic plasticity to substantially
273 alter root characteristics⁶⁸ and selection aimed at improved root adaptation to soil acidity,
274 hypoxia and nutrient limitations could yield greater root C inputs as well as increased crop yields

275 ⁶⁷. Greater root C inputs is well-recognized as a main reason for the higher soil C stocks
276 maintained under perennial grasses compared to annual crops ¹⁵. Although there are no
277 published estimates of the global C sink potential for ‘root enhancement’ of annual crop species,
278 as a first-order estimate, a sustained increase in root C inputs might add ~1 Pg CO₂eq yr⁻¹ or
279 more if applied over a large portion on global cropland area (Fig 2).

280
281 Hence, the overall mitigation potential of existing (and potential future) soil management
282 practices could be as high as ~8 Pg CO₂eq yr⁻¹. How much is achievable will depend heavily on
283 the effectiveness of implementation strategies and socioeconomic and policy constraints. A key
284 strength is that a variety of practices can often be implemented on the same land area, to leverage
285 synergies, while avoiding offsetting effects for different gases (Fig. 1). But regardless of which
286 combination of management interventions are pursued, effective policies, that incentivize land
287 managers to adopt them, will be needed. A common thread across implementation strategies is
288 the role for strong science-based metrics to measure and monitor performance.

289

290 **Implementation of mitigation practices**

291 Relative to many other GHG source categories, agricultural soil GHG mitigation presents
292 particular challenges. Rates on an individual land parcel are often low, but vast areas of land are
293 devoted to agriculture globally, and the implementers of mitigation practices – the people using
294 the land – number in the billions. Thus engaging a significant number of these people is a
295 massive undertaking in itself. Furthermore, agricultural soil GHG emissions are challenging to
296 quantify due to their dispersed and variable nature and the multiplicity of controlling factors –
297 operating across heterogeneous landscapes. Direct measurement of fluxes requires specialized

308 personnel and equipment, normally limited to research environments, and hence not feasible for
309 most mitigation projects. Model-based methods, in which emission rates are quantified as a
310 function of location, environmental conditions and management, provide a more feasible
311 approach^{52,69,70}. Process-based models, which dynamically simulate mechanisms and controls on
312 fluxes as a function of climatic and soil variables and management practices, and empirical
313 models based on statistical analysis of field-measured flux rates, represent differing but
314 complementary approaches. In general, model-based quantification systems enable monitoring to
315 focus on practice performance and thus dramatically reduce transaction costs for implementing
316 mitigation policies⁶⁹.

307

308 [Box 1 about here]

309

310 Several implementation strategies for soil GHG mitigation exist (see Box 1), all of which require
311 robust quantification and monitoring technologies. Those requiring the most rigorous methods
312 involve offset projects participating in cap-and-trade markets, in which land managers are
313 directly compensated for achieving emission reductions. Other market-linked strategies, such as
314 ‘green labeling’ systems for agricultural products, will also require rigorous yet easy to use GHG
315 quantification tools, enabling agricultural producers to meet standards set by product distributors
316 and accepted by consumers^{6,71}.

317

318 Within the voluntary C offset market space, there are a growing number of projects that include
319 soil GHG mitigation components⁵. Several large projects focus on preventing land conversion
320 (i.e., from forest and grassland), thus avoiding large CO₂ emissions from soils and liquidated

321 biomass C stocks. Relatively simple empirical models supplemented with field measurements
322 are commonly used for avoided land conversion projects. For more complex land use projects,
323 empirical models are less suited to capture interactions across multiple emission sources, and
324 may over- or under-credit projects where a practice has an influence on multiple emission
325 sources. There are relatively fewer projects targeting GHG mitigation on existing agricultural
326 lands, involving a broader suite of soil management practices, and early pilot-phase N₂O and
327 CH₄ reduction projects are only now being developed^{5,52}. Here, accurately quantifying C
328 sequestration and/or emission reductions is more challenging due to lower rates of change
329 relative to baseline conditions, thus requiring more sophisticated models and supporting research
330 infrastructure (Fig. 3).

331
332 Another challenge for projects on existing agricultural lands is obtaining and processing the
333 management activity data. For example, the Kenya Agriculture Carbon Project (KACP) involves
334 a total of 60,000 individual small-holder farmers⁷². In contrast to projects involving major land
335 cover changes, where remote sensing can provide much of the activity monitoring (e.g., retention
336 of forested land over time), such options are poorly-suited for monitoring crop type, fertilizer,
337 residue and water management, and organic matter amendments⁷³; for such practices the best
338 source of information are the land managers themselves (Fig. 3).

339
340 Thus another option is to engage land managers as information providers. Examples of this
341 approach are the Cool Farm Tool⁷¹, being used by farmers participating in low C supply chain
342 management, and the COMET-Farm tool, which allows farmers to compute full farm-scale GHG
343 budgets, for support of government-sponsored conservation initiatives and participation in

344 mitigation projects⁷⁴. Both tools provide web-based interfaces designed for non-specialists to
345 enter land management information; Cool Farm utilizes empirical emission factor-type models,
346 while COMET-Farm incorporates both empirical and process-based models. Such systems can
347 be used to integrate local knowledge on management practices with detailed soil and climate
348 maps, remote sensing and sophisticated models for emission calculations. Soon much of this
349 functionality could be deployed in mobile applications (Fig. 3), which would be particularly
350 advantageous in developing countries where existing infrastructure to collect and manage land
351 use data is weak⁷⁵.

352

353 [Fig. 3 about here]

354

355 **Quantifying uncertainties**

356 Inventories of soil C stock changes and net GHG fluxes using process-based models will always
357 have uncertainty due to lack of process understanding, inadequate parameterization, and
358 limitations associated with model inputs⁷⁶ (e.g., weather, management and soils data).

359 Empirical models generally rely on statistical analyses of measurement data to produce emission
360 factors, along with an estimated uncertainty¹⁴. However, empirical models can be biased if
361 measurements do not fully reflect the conditions for the agroecosystems in the project. Even with
362 the limitations in process-based understanding, process-based models are likely to provide the
363 most robust framework for estimating soil C stock and GHG flux changes in climate smart
364 agriculture programs⁷⁷.

365

366 Monitoring, reporting and verification (MRV) systems are a key element in a climate smart
367 agricultural program. While MRV systems place different levels of importance on uncertainty
368 depending on program type (see Box 1)⁷⁸, discounting payments based on the level of
369 uncertainty is likely to be part of programs with financial incentives, such as cap-and-trade.
370 Discounting encourages monitoring efforts to reduce uncertainty over time¹⁷. If discounting
371 payments for C sequestration and emission reduction practices with larger uncertainty is adopted
372 in climate smart agriculture programs, then more advanced methods with process-based models
373 will likely emerge as the preferred method due to less uncertainty. For example, uncertainty was
374 reduced by 24% when predicting national-scale C stock changes in the United States with
375 process-based models compared to empirically-derived factors⁷⁶.

376

377 Another consideration is that uncertainties in estimating C stock and GHG emissions with
378 process-based models are considerably larger for reporting by single individuals, particularly if
379 the amount of change on an individual farm is small⁷⁶. Aggregation of many farms into larger
380 projects will reduce uncertainties, which could be a viable approach for managing uncertainty
381 and reducing discounting of incentive payments.

382

383 Verification is an independent evaluation of estimated emissions intended to provide confidence
384 that the reported results are correct, but in practice, the requirements for verification are highly
385 variable across different GHG mitigation efforts, from essentially no requirements to annual
386 evaluations⁷⁸. Verification typically focuses on the accuracy of the estimates, and possibly the
387 most stringent approach is an independent set of measurements. Although independent data may
388 be less favored in terms of costs relative to alternatives, such as expert judgement⁷⁸, soil

389 monitoring networks deployed at national or regional scales could produce independent data for
390 evaluating model-based assessments of soil C stock changes and GHG emissions⁷⁹ and for model
391 bias adjustment, using empirically-based methods⁸⁰.

392
393 Another approach to verification is to use atmospheric observations of trace gas concentrations
394 and inverse modeling to estimate fluxes between the atmosphere and land surface^{81,82}. This ‘top-
395 down’ modeling, utilizing a network of tower-based observations of CO₂ concentrations, was
396 used to verify ‘bottom-up’ inventory modeling based on observed management activities, in the
397 largely agricultural region of the central United States^{83,84}. Since atmospheric observations
398 integrate all CO₂ fluxes in the region, the inventory included a full assessment of all sources and
399 sinks. However, even with the fully integrated CO₂ flux, it is possible to statistically
400 disaggregate individual sources as part of the analysis, such as contributions from soil C pools to
401 the regional flux⁸⁵. Satellite-based measurements are providing a new source of atmospheric
402 trace gas data that can be used to estimate land surface fluxes with inverse modeling
403 frameworks^{86,87}. While atmospheric observations and satellite imagery may become a standard
404 for verifying regional inventories in the future, the methods need further testing in the near term
405 before deploying operational systems.

406

407 **Conclusions and way forward**

408 Climate change and GHG mitigation require an ‘all of the above’ approach⁸⁸, where all reduction
409 measures that are feasible, cost-effective and environmentally sustainable should be pursued.

410 For soils, a variety of management practices and technologies are known to reduce emissions and
411 promote C sequestration, most of which also provide environmental co-benefits. Impediments to

412 more aggressively implementing agricultural soil GHG mitigation strategies to date are primarily
413 the feasibility of cost-effectively quantifying and verifying soil mitigation activities⁸⁹.
414 Overcoming these barriers therefore translates into: i) increasing the acceptance of soil
415 management within compliance and voluntary C markets, ii) reducing costs to governments for
416 providing environmental-based subsidies, and iii) meeting demands of consumers for ‘low
417 carbon’ products.

418

419 Reducing and managing uncertainties are key to both improved predictive models and decision-
420 support tools and the design of effective policies that promote soil-based GHG mitigation. To
421 advance these efforts, several research and development priorities are apparent (Fig 3). First,
422 support for research site networks of soil flux (N₂O, CH₄) and soil C measurements⁹⁰
423 encompassing a wide variation in management, as well as ‘on-farm’ soil C monitoring
424 networks⁷⁹ needs to be strengthened, in coordination with basic research (e.g., on SOM
425 stabilization processes, N₂O and CH₄ microbiology, plant-microbe interactions, plant breeding
426 and root phenotyping) to advance process understanding, develop new mitigation practices and
427 fill gaps for underrepresented soil/climate/management systems. High quality data generated
428 from consistent measurement protocols is critical for evaluating and improving models. These
429 efforts may benefit from development of new sensor technologies enabling cheaper and quicker
430 soil measurements⁹¹. While multiple competing models are needed, both to spur innovation and
431 because no single model will be best in all situations, model development will benefit from
432 greater collaboration and cross-model testing among developers, moving towards a more open-
433 source, community development approach⁹². Large geospatial databases of soil biophysical
434 properties and climate variables are critical to accurately quantify soil processes across the

435 landscape (Fig. 3). High resolution soil maps exist in most developed countries (and increasingly
436 in developing countries⁹³), and if made publically available⁹⁴, would greatly improve capabilities
437 for modeling GHG emission at scale.

438
439 Finally, realising the potential for climate change mitigation through global soil management
440 requires understanding cultural, political and socioeconomic contexts, and the ways in which
441 widespread, sustained changes in practice can be successfully achieved within it^{95,96}. As such,
442 there needs to be greater level of engagement with the land users themselves, who will be the
443 ones implementing practices that abate GHG emissions and sequester C. Engagement means
444 both education and outreach, highlighting the links between agriculture and GHGs and utilizing
445 innovative strategies⁷⁵ (Fig. 3) to involve stakeholders in gathering and using their local
446 knowledge of how the land is being used now and how it might best be used in the future,
447 establishing a new paradigm for climate-smart soil management.

448

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715 **Figure text.**

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

731

732 Fig. 2. Global potential for agricultural-based GHG mitigation, relating average per ha net GHG
733 reduction rates and potential area (in Mha) of adoption (note log-scales). Unless otherwise
734 noted, estimates are from Smith et al.¹⁸ based on cropland and grassland area projections for
735 2030. Ranges in total Pg CO₂eq yr⁻¹ represent varying adoption rates as a function of C pricing
736 (\$20, \$50, and \$100 per Mg CO₂_eq), to a maximum technical potential, i.e., full
737 implementation of practices on the available land base. Multiple practices are aggregated for

738 cropland (e.g. improved crop rotations and nutrient management, reduced tillage) and grazing
739 land (e.g., grazing management, nutrient and fire management, species introduction) categories.
740 Practices that increase net soil C stocks and/or reduce emissions of N₂O and CH₄ are combined
741 in each practice category. The portion of projected mitigation from C stock increase (ca. 90% of
742 the total technical potential) would have a limited time span of 20-30 years, whereas non-CO₂
743 emission reduction could, in principle, continue indefinitely¹⁸. Estimates for biochar application
744 from Woolf et al.⁶⁶ represent a technical potential only, but based on a full life cycle analysis
745 applicable over a 100 year time span. Although global estimates of the potential impact of
746 enhanced root phenotypes for crops have not been published, a first-order estimate of ~1 Pg
747 CO₂eq yr⁻¹ is shown, using as an analog, global average C accrual rates (0.23 Mg C ha⁻¹ yr⁻¹) for
748 cover crops²⁴, applied to 50% of the cropland land area used by Smith et al.¹⁸.

749
750 Fig. 3. Expanding the role for agricultural soil GHG mitigation will require an integrated
751 research support and implementation platform. Targeted basic research on soil processes (a few
752 examples of priority areas shown here), expanding measurement/monitoring networks and
753 further developing global geospatial soils data can improve predictive models and reduce
754 uncertainties. Ongoing advances in information technology and complex system and ‘Big Data’
755 integration, offer the potential to engage a broad-range of stakeholders, including land managers,
756 to ‘crowd-source’ local knowledge of agricultural management practices through web-based
757 computer and mobile apps, and help drive advanced model-based GHG metrics. This will
758 facilitate implementation of climate-smart soil management policies, via cap-and-trade systems,
759 product supply chain initiatives for ‘low-carbon’ consumer products, national and international

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

762

763 **[BOX 1]**

764 **Implementation strategies for soil GHG mitigation**

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

767 1) Regulation/taxation: Direct regulatory measures to reduce soil GHGs at the entity scale are
768 likely politically unfeasible and costly. Taxation of N fertilizer, already used in parts of the US
769 and Europe to reduce nitrate pollution, could function as an indirect tax to reduce N₂O emissions.

770 2) Subsidies: Targeted government payments/subsidies for implementing GHG-reducing
771 practices is emerging as a policy alternative. For example, US Dept. of Agriculture programs
772 are including GHG mitigation as a conservation goal and provisions in the EU Common
773 Agricultural Policy link subsidy payments to ‘cross compliance’ measures that include
774 maintenance of soil organic matter stocks⁹⁷. A more direct link to soil GHG emissions follows
775 from a recent decision to include cropland and grassland in EU commitments under the Kyoto
776 Protocol⁹⁸.

777 3) Supply chain initiatives: Major food distributors are targeting sustainability metrics, including
778 low GHG footprints, as a consumer marketing strategy⁹⁹, setting performance standards for
779 contracted agricultural producers, including requiring field-scale monitoring of production
780 practices and quantification of GHG emissions.

781 4) Cap and trade (C&T): In a C&T system, emitters are subject to an overall emissions level or
782 ‘cap’, in which permitted emissions decrease over time. Emitters can stay below the capped
783 levels by reducing their own emissions and/or by purchasing surplus permits from capped
784 entities that have exceeded their required reductions. Both compliance and voluntary markets
785 can function as C&T systems¹⁰⁰. Within many C&T systems, a limited amount of emission

786 reductions (termed ‘offsets’) can be provided by non-capped entities. Inclusion of agricultural
787 activities as offset providers has been growing, particularly within voluntary markets. To
788 maintain the integrity of emission caps, key criteria for offset providers include demonstrating
789 *additionality*, i.e., insuring that reductions result from project interventions and not simply
790 business-as-usual trends, avoiding *leakage*, i.e., unintended emission increases elsewhere as a
791 consequence of the project activities, and providing for *permanence* (e.g., that increased soil C
792 storage, credited as a CO₂ removal, is maintained long-term).

793 **[End BOX 1]**

794





