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1 Can N₂O emissions offset the benefits from soil organic 2 carbon storage?

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ABSTRACT

40 To respect the Paris agreement targeting a limitation of global warming below 2°C by 2100, and possibly below
41 1.5 °C, drastic reductions of greenhouse gas emissions are mandatory but not sufficient. Large-scale deployment
42 of other climate mitigation strategies are also necessary. Among these, increasing soil organic carbon (SOC)
43 stocks is an important lever because carbon in soils can be stored for long periods and land management options
44 to achieve this already exist and have been widely tested. However, agricultural soils are also an important
45 source of nitrous oxide, (N₂O) a powerful greenhouse gas, and increasing SOC may influence N₂O emissions,
46 likely causing an increase in many cases, thus tending to offset the climate change benefit from increased SOC
47 storage. Here, we review the main agricultural management options for increasing SOC stocks. We evaluate the
48 amount of SOC that can be stored as well as resulting changes in N₂O emissions to better estimate the climate
49 benefits of these management options. We conclude that the climate mitigation induced by increased SOC
50 storage is generally overestimated if associated N₂O emissions are not considered, but is never fully offset. Some
51 options (e.g. biochar or non-pyrogenic amendment application) may even decrease N₂O emissions.

52

53 Key points:

- 54 • Carbon and nitrogen cycles in soil interact in numerous and complex ways and an impact of a land
55 management change on one cycle will generally influence the other.
- 56 • Several land management options designed to increase soil organic carbon stocks exist and have been
57 widely evaluated.
- 58 • Land management options to increase soil organic carbon also tend to increase nitrous oxide emissions
59 in some cases.
- 60 • We conclude that when increased N₂O emissions are taken into account, they partially offset the climate
61 benefits of increased organic carbon storage, but never negate them completely.

62

63 1. INTRODUCTION

64 The United Nations Framework Convention on Climate Change (UNFCCC) Paris Agreement adopted in
65 2015 aimed at keeping global warming below 2°C by 2100, and at possibly further limiting the temperature
66 increase to 1.5 °C. The Paris Agreement specified that the balance of anthropogenic greenhouse gas (GHG)
67 emissions and sinks should be attained by the second half of this century. This requires not only drastic
68 reductions in GHG emissions in the near future, but also net negative emission technologies because not all
69 emissions will be reducible to zero within this time scale^{1,2}. To a large extent, these negative emissions imply
70 land-based mitigation strategies³, mostly involving the production of organic matter by plant photosynthesis
71 coupled with carbon storage in living biomass and / or soil organic matter⁴. A pathway frequently discussed
72 known as bioenergy with carbon capture and storage (BECCS) comprises generating energy using biomass,
73 capturing the CO₂ evolved from this process and storing it in geological reservoirs. The deployment of BECCS
74 faces both technical challenges and most likely limitations due to high costs and adverse environmental
75 impacts^{5,6}. On the other hand, the net removal of atmospheric CO₂ taken up by plants in agricultural soils (i.e.,
76 carbon sequestration) has recently come under sharp focus as a more affordable and practical alternative,
77 potentially associated with positive economic outcomes and possibly applicable at large scale in managed
78 lands^{7,8}. The role of soils as a key component of the global carbon cycle is now recognized by the scientific
79 community and also by policymakers^{5,6}. Soils have never been harnessed at large scale for the purpose of
80 sequestering carbon, although they currently make up the largest reservoir of organic carbon in the terrestrial
81 biosphere, with a size of 1,500 Pg C to a depth of one meter⁹. However, the ecosystems which contain the largest
82 stocks of soil organic carbon (SOC) are unmanaged (comprising boreal forests, permafrost soils and wetlands),
83 whereas only soils from managed ecosystems, in particular agricultural soils, may be managed to increase SOC
84 stocks (i.e., carbon sequestration) Agriculture is also a key target sector for the reduction of methane (CH₄) and
85 nitrous oxide (N₂O) emissions¹⁰⁻¹². Yet, few countries have included agriculture in their nationally determined
86 contributions – a roadmap volunteered by national governments as part of the Paris Agreement to express their
87 efforts to reduce GHG emissions – because of potential trade-offs with food production and uncertainties on
88 achievable potentials¹³.

89 Recent emphasis on promoting SOC storage has resulted in international initiatives such as the “4 per mil”
90 initiative launched by France during the UNFCCC conference of the parties (COP) 21^{7,14}. It relies on the concept
91 that even a very small relative increase in SOC pools worldwide could offset a significant fraction of CO₂
92 emissions. Preliminary evaluation indicated that increasing global agricultural SOC stocks at an annual rate of

93 4‰ would result in a C sequestration potential of 2-3 Pg C yr⁻¹⁷. This may contribute significantly to the
94 objectives of the land sector to achieve the Paris agreement target¹⁵. Nevertheless, several studies have discussed
95 and criticized the feasibility of enriching soils at a rate of 4‰ over a sustained period of years¹⁶⁻¹⁸ because: (i) it
96 requires large amounts of new organic matter inputs, (ii) it requires large amounts of nutrients, (iii) it is difficult
97 to achieve this target rate in all agricultural systems, and (iv) it may be hampered by the climate change-induced
98 enhancement of SOC decomposition. Moreover, altered management practices may impact farmers' income and
99 imply trade-offs with food production¹⁷. Data from long-term experiments show that it is very difficult to
100 achieve the 4 per mil rate in temperate arable systems without drastic changes in management^{17,19}. Finally, the
101 annual rate of SOC increase generally levels off over time as the SOC pool increases and approaches a new
102 equilibrium level²⁰.

103 Nevertheless, concrete management options exist to increase SOC stocks such as cover crops, tillage
104 management, crop rotations, organic amendments, agroforestry and biochar amendments with effects depending
105 on local conditions²¹⁻²³. It must be noted that organic amendments may, in some case such as manures, be a
106 transfer of carbon from one terrestrial location to another rather than a net removal of carbon from the
107 atmosphere¹⁷. Nevertheless, well spatially distributed, organic amendments contribute to significant increase the
108 SOC pools at regional scale²⁴. These options have socio-economic impacts on farmers and land managers and
109 indirect effects on ecosystem services, through changes in crop yields, water consumption, nitrate leaching, and
110 CH₄ and N₂O emissions which have to be considered when evaluating the feasibility and the relevance of
111 implementing SOC storage options. The SOC storage potential of the various practices has been extensively
112 assessed in the recent scientific literature^{17,23,25-27}, and recently revised by IPCC in its 2019 report on climate
113 change and land¹². However, implications for the N cycle (in particular N₂O emissions), and other
114 biogeochemical cycles or crop yields have not been thoroughly documented so far²⁸⁻³⁰. Neither have been the
115 consequences of large-scale deployment of these measures, and constraints related to the nitrogen (N),
116 phosphorous (P) and potassium (K) cycles. These aspects are important because they determine the overall GHG
117 abatement efficiency of mitigation measures and set limits on their potential deployment. C and N cycles are
118 strongly interlinked (box. 1) in particular in soils and we assume that the deployment of land based mitigation
119 options to increase SOC may impact the N cycle and the associated N₂O emissions. A recent modelling study
120 suggests that measure to increase SOC sequestration might be offset by increased N₂O, depending on the crop
121 rotation and on the duration of the land management practices²⁸. A better understanding of such interactions is
122 necessary to evaluate the benefits of different management practices aimed at increasing SOC storage and to

123 predict the full GHG balance of each practice.

124 Here, we focus on the interactions between soil C and nutrient dynamics, and in particular on N dynamics
125 and N₂O emissions. The aims of the paper are to i) describe the mechanisms linking the C and N cycles in soils,
126 ii) assess how N₂O emissions may be affected by increased SOC pools as a land based mitigation option, iii)
127 review our knowledge on the other impacts of these practices.

128

129 2. INTERWINED SOIL CARBON AND NITROGEN CYCLES

130 Because C and N cycles are tightly coupled in soils, altering one will affect the other as shown in Fig. 1.
131 In soils the C and the N cycles are sometimes totally interdependent, in particular when both are in organic forms
132 but are sometimes uncoupled when C or N are present as minerals. Nevertheless they may still interact with each
133 other. This section summarizes the main mechanisms explaining how changes in the soil C cycle and SOC
134 sequestration interact with N cycle processes, and in particular N₂O emissions (Fig. 1, Box 1). The first reason
135 why soil C and N dynamics are interdependent is that both elements are stored predominantly as organic forms
136 in the soil, sometimes within the same compounds (amino acids, proteins, etc.), thus mineralisation generally
137 affects both. Moreover, the availability of mineral N in the soil controls a number of processes in both cycles and
138 *vice versa*. For instance, mineral N transformations depend on carbon availability and plant dry matter
139 production is limited by N availability. Nitrogen is needed to sustain photosynthesis and other physiological
140 processes³¹; therefore higher N availability would likely lead to greater primary productivity and inputs of plant-
141 derived organic matter to the soil³². On the other hand, higher N availability also tends to lower the allocation of
142 photosynthates to the root system³³. As root-derived C inputs contribute at least 2-3 times more than shoot-
143 derived C inputs to SOC storage^{34,35}, a high soil N availability could theoretically increase the plant biomass but
144 the plant biomass produced might not be transformed into SOC as efficiently because of a reduced amount of
145 root-derived C entering the soil³⁶. Soil organic matter (SOM) turnover (i.e., rate of mineralization and
146 transformation of SOM) also depends on the N availability for microorganisms. While a low mineral N
147 availability may limit the mineralization rate of plant residues and amendments^{37,38}, the combination of regular
148 inputs of fresh organic C with a low soil N availability can lead to positive priming effect, *i.e.* a higher rate of
149 SOM mineralization, and a lower SOC storage potential^{39,40}. Moreover, because of the relatively narrow range of
150 C:N ratios of SOM in mineral layers¹⁶ and because of the importance of soil microbial processing for building up
151 stable SOM⁴¹ in some ecosystems, large amounts of N are inevitably required to stabilize large amounts of
152 SOC^{42,43}. Because of this stoichiometric requirement, it might seem acceptable to maintain a high availability of

153 N in the soil by applying large amounts of mineral fertilizers. Such a strategy would however lead to potential N
154 losses, *e.g.* as N₂O emissions or nitrate leaching from soil, and further increase GHG emissions during fertilizer
155 production. Thus the modest increases in SOC resulting from N fertilizer applications up to sensible agronomic
156 rates are welcome in the context of C sequestration, but it would be counter-productive and inappropriate to
157 recommend higher rates of N application aimed at promoting an additional increase in soil C.

158 Input of N to terrestrial ecosystems by biological N₂ fixation is another example of a close link between
159 C and N resources. Root-associated or free-living N₂-fixing bacteria depend on the availability of organic C
160 resources for sustaining their heterotrophic needs, which may explain why N₂ fixation is only triggered when the
161 amount of soil mineral N is low. In particular, the energy cost of N₂ fixation represents between 5% and 23% of
162 daily photoassimilated C⁴⁴. The associated CO₂ losses by respiration may therefore decrease the amount of plant
163 C entering the soil. However, the consequence of this on the potential of SOC storage remains unclear. For
164 example, the presence of leguminous plants can result in lower belowground C inputs compared to gramineous
165 plants, leading to lower SOC concentrations^{45,46}. However, the net inputs of N to soils by leguminous plants have
166 been shown to correlate with a net accumulation of SOC, by providing the organic N required to stabilize an
167 additional amount of SOC in soils⁴⁷. Similarly, crop rotations that include leguminous plants appear store more
168 SOC than conventional monocultures⁴⁸, although this effect may be mainly due to longer periods of plant cover,
169 and to the presence of deeper root systems than to biological N₂ fixation itself⁴⁷. These feedbacks also depend on
170 which non-leguminous plants are associated⁴⁹ to the N₂-fixing plant, and may lead to contrasting results in terms
171 of SOC storage²⁴. Of course, obtaining N from legumes, where this is practicable, rather than from N fertilizer
172 does eliminate the GHG emissions associated with N fertilizer manufacture.

173 Fresh C inputs to the soil through root exudates or amendments may temporarily decrease or increase
174 soil pH, affecting the magnitude of N₂O emissions. Their consumption by microorganisms may also decrease the
175 local concentration of oxygen, leading to anaerobic conditions which are favourable to denitrification and N₂O
176 emissions⁵⁰. Furthermore, because organic materials generally act as electron donors in the denitrification
177 process and because soil organic matter content may lower the redox potential of the soil⁵¹, increasing the
178 amount of soil organic matter may also increase the activity of denitrifiers and therefore increase N₂O
179 emissions^{52,53}. These mechanisms likely explain why higher SOC contents in soils have indeed been shown to
180 correlate with larger N₂O emissions^{54,55}. N₂O emissions represent a particular case that illustrates how the soil N
181 cycle may be influenced by the C cycle. As a rule, net N₂O emissions from the soil at a given soil water-filled
182 pore space (WFPS) will usually be lower when the soil mineral N content is low and when soil pH is alkaline or

183 when C availability is reduced. Furthermore, because a low soil redox potential (< 400 mV)⁵² is required for
184 denitrification, N₂O emissions have been suggested to have their optimum at 70-80% WFPS, while prolonged
185 waterlogging conditions may result in complete nitrate reduction to N₂ instead of N₂O⁵⁶. Several mechanisms
186 can therefore explain why attempts to modify the soil C cycle may also affect N₂O emissions. On a longer time
187 scale, the build-up of SOC by various strategies may be expected to increase the retention of water and fertilizer-
188 N in the rooting zone through improved soil properties (eg.. water holding capacity, porosity, hydrophilicity) , in
189 a manner favourable for the denitrification to occur. This might trigger a higher primary production and
190 enhancing further SOC storage, but also increase the risk of N₂O emissions because of the increase in N sources
191 and the shift to soil environmental conditions more favourable to N₂O emissions. In the remainder of this paper
192 we consider possible interactions between increased SOC and changes in N₂O emission for a range of
193 management practices designed to increase SOC (Table 1).

194 3. HOW SOC STORING PRACTICES AFFECT N₂O EMISSIONS

195 *3.1 Balancing the Nitrogen inputs*

196 Since mineral N availability drives N₂O emissions as well as crop productivity and C inputs into the
197 soil^{57,58}, N fertilization should be carefully managed. A balance should be obtained between N inputs (including
198 fertilizers, manures and biological nitrogen fixation through symbiosis between N₂-fixing bacteria and some
199 plant species) and N exported in harvested products in order to reduce a N surplus that can be source of N₂O, but
200 without a major negative effect on crop productivity. This N surplus should ideally be zero, but it is actually
201 large and positive in many regions of the world, having intensive agriculture (e.g. parts of China, India, Europe,
202 North America), and negative in other regions (e.g. Africa)⁵⁹. Excess N associated with a positive surplus is a
203 major cause of N₂O emissions on farms, but also of nitrate leaching losses, part of which contributes to indirect
204 N₂O emissions if nitrate is denitrified within surface waters. Overall, N surplus is a strong driver of N₂O
205 emissions, especially when considering that the rate of emission is no longer linear for high N input⁶⁰. The
206 relatively low cost of mineral N fertilizers in developed countries compared to the price of agricultural products
207 incentivizes farmers to apply more N than recommended by good practices, as an 'insurance' against unforeseen
208 N losses due to climate variability. In some regions of the world, but not all, there is considerable potential to
209 lower agricultural N₂O fluxes in intensive farming by reducing the N surplus without affecting farmers' incomes
210 ⁶¹. Therefore, the use of mineral N to increase crop productivity may induce an increase of C input into the soil
211 but a complex balance must be found to avoid excessive N₂O emissions and N leaching.

212

213 *3.2 Reduced tillage/non tillage*

214 The effect of reduced tillage has attracted attention as a practice leading to increased SOC storage..
215 However, recent meta-analyses demonstrate only a small positive effect of no-tillage on SOC stocks in the
216 topsoil (0-30 cm layer) compared to conventional tillage, while it may vary widely across pedo-climatic
217 situations⁶²⁻⁶⁶. Moreover, it must be recognised that the largest impact of reducing tillage is a redistribution of
218 SOC toward the soil surface^{25,64-67}. As a consequence, data from field trials must be carefully examined to
219 distinguish between a genuine increase in SOC stocks in the surface soil layers from a simple change in the
220 vertical distribution of SOC concentration.

221 There has been considerable discussion as to whether the increased SOC in soil under zero tillage,
222 especially near the surface, might increase N₂O emissions, because : (i) increased organic matter content can
223 increase N₂O release⁶⁸, either because of increased energy supply to denitrifying organisms or because increased
224 biological activity utilises oxygen in soil, thus possibly leading to anoxic conditions at some microsites and ii)
225 reducing tillage can be associated in the short term with a less porous soil structure, conducive of anoxia⁶⁹ (Table
226 1). The different meta-analysis we compiled here (Fig. 2) suggest that N₂O emissions may offset the C storage in
227 no-till system when both fluxes are compared in CO₂ equivalents. However, there is conflicting evidence on
228 whether or not this risk is actually realised^{68,70}. Recent meta-analyses suggest that, in the majority of situations,
229 N₂O emissions are either unchanged or slightly decreased under zero or reduced tillage; the result will certainly
230 be influenced by soil type and local climate and weather conditions so it may not be possible to draw a
231 conclusion that is universally valid^{70,71}. Furthermore, in some studies, N₂O emissions were expressed on both an
232 area basis and a yield-scaled basis⁷⁰; because crop yields were slightly decreased under reduced tillage in some
233 environments, N₂O emissions per unit of grain (or other product) were sometimes increased compared to
234 conventional tillage.

235 *3.3 Erosion control - terracing*

236 Erosion control practices are able to maintain or increase SOC content at the plot scale⁷², although on a
237 larger scale whether erosion is net C sink or a net source is still debated⁷³⁻⁷⁵. Erosion control encompasses a wide
238 range of practices such as protecting the soil surface with cover crops or unharvested biomass (pruned fronds and
239 other plant residues), agroforestry, crop rotations, conservation tillage, or terracing on steep slopes. Some of

240 these practices are already addressed in other sections of this paper (3.2, 3.4 and 3.5), and the following focuses
241 on terracing.

242 Terracing is an ancient form of erosion control and a soil conservation method performed for thousands
243 of years in steep landscape regions^{76,77}. Despite its importance, studies focusing on quantifying soil erosion rates
244 and the resulting C fluxes and SOC stocks in terraced areas are limited, especially at regional scales. Generally,
245 terracing reduces soil erosion by reducing the slope gradient and length, and can decrease soil erosion rates by up
246 to 95%^{78,79}. It accordingly preserves SOC and nutrients. A meta-analysis on the ecosystem benefits of terracing
247 shows that, compared to unterraced slopes, soil in terraced slopes contains 28.1% and 41.7% more N and C,
248 respectively⁸⁰. However, the overall net effect of terracing on erosion depends on the terrace structure and
249 maintenance, crop type, soil conditions, crop management practices or agricultural machinery. To maximize its
250 positive effects, terracing needs to be combined with other soil conservation measures such as cover crops,
251 agroforestry, organic amendments or no-till^{81,82}. Furthermore, terraces need to be sustained, otherwise abandoned
252 terraces can become sources of substantial land degradation due to gully formation. This is the case in the
253 Mediterranean region where over 50% of the terraces have been abandoned^{77,80}.

254 The N₂O emissions associated with terracing are still poorly known. Terracing decreases the aggregate
255 breakdown and transport of soil by erosion, which would lead to reduced N₂O emissions. However, as stated
256 previously, N₂O emissions may increase with increased SOC. In addition, terracing tends to change the soil C:N
257 ratio⁸⁰ and this may change the N availability for nitrifying/denitrifying bacteria and thus affect N₂O emissions.
258 Finally, to fully estimate the effect of erosion control on N₂O budgets, it is important to measure emissions at the
259 catchment scale not only at the field scale. Since erosion control aims to avoid lateral losses of soil material
260 (containing various forms of N) ending up in rivers or in floodplains, it is necessary to combine measurements in
261 the terraced or unterraced fields with measurements and modelling on the fate of eroded N in floodplains and
262 rivers.

263

264 *3.4 Cover crops*

265 Planting cover crops is an effective management practice to increase SOC content. According to a
266 recent meta-analysis, it leads to SOC accumulation rates in the order of 1.18 t CO₂-eq. ha⁻¹ yr⁻¹ over 50 years⁸³ in
267 the topsoil, with a positive effect independent of tillage method, climatic zone or plant type (leguminous vs non-
268 leguminous). The impact of cover crops on SOC will depend on their duration and the frequency with which

269 they are included in a crop rotation, and this information is sometimes unclear in published reports of field
270 trials⁸³. The main driver of SOC storage seems to be the extra C input, as suggested by the high correlation
271 between rates of SOC stock change and the amounts of C returned to the soil by cover crop biomass⁸⁴. However,
272 there are limitations to the use of cover crops depending on cropping systems and climate conditions. For
273 example, in temperate regions they can be readily utilised during the winter period prior to sowing a spring-sown
274 crop, when the soil would otherwise be bare. But if only autumn-sown crops are grown, there is very limited
275 time between harvesting and the sowing of the subsequent crop.

276 The effects of cover crops on N₂O emissions are more variable and contrasted than those on SOC
277 changes (Table 1). Many factors influence the magnitude of N₂O emissions, such as the C:N ratio of cover crop
278 residues, their rate of decomposition, the extra inputs of fertilizer N sometimes applied to cover crops, whether
279 the residues are ploughed or left to decay on the soil surface. Current evidence points to a negative relationship
280 between N₂O emissions and the C:N ratio of residues⁵⁰. A low C:N ratio will increase the availability of soil N
281 for microbial transformations (e.g. nitrification and denitrification) whereas larger ratios will result in N
282 immobilization and deplete the soil inorganic N pool⁸⁵. Additional C inputs from cover crops may stimulate the
283 activity of denitrifier bacteria, which use these organic compounds as a source of energy⁸⁶. A meta-analysis⁸⁷
284 reported a significant increase in N₂O emission when leguminous cover crops were introduced. However,
285 another review⁸⁸ found out that the incorporation of either legume and non-legume cover crops tended to
286 increase N₂O emissions but the magnitude of the effect was not significant due to the high variability of data.
287 The effect of cover crops on N₂O emissions is therefore not yet fully understood and may well be highly site-
288 specific.

289 One of the key points controlling cover crop effect on N₂O emissions is how often leguminous crops are
290 integrated within the crop rotation. Leguminous cover crops generally have a lower C:N ratio than non-
291 leguminous crops, and can fix substantial amounts of atmospheric N, reaching up to 0.1-0.2 t N ha⁻¹ yr⁻¹⁸⁹. These
292 rates may lead to a N surplus if all the leguminous cover crop biomass is incorporated. A recent study, using a
293 biogeochemistry model framework at European scale, estimated that systematic planting of N-fixing cover crops
294 may lead to a N surplus of about 0.04 t N ha⁻¹ yr⁻¹, compared to the use of non-legumes as cover crop²⁸. In this
295 scenario, the cumulative climate change mitigation effect of SOC sequestration was, on average, totally offset
296 after 50 years since the adoption of cover crops, due to enhanced N₂O emissions. While cover crops may induce
297 higher N₂O emissions, in particular if leguminous crop are extensively used, they can also reduce nitrate
298 leaching, by about 56% on average⁹⁰. This is beneficial for water quality and would be expected to lead to

299 decreased indirect N₂O emission through denitrification of nitrate entering surface water. Finally, another
300 indirect effect of leguminous cover crops on N₂O emissions will strongly depend on whether or not mineral N
301 fertilisation rates are reduced to take account of N provided by biological fixation. The meta-analysis we
302 compiled here indicate that additional N₂O emissions decrease the SOC storage benefit of cover crops, but do
303 not fully offset it (Fig. 2).

304

305 *3.5 Agroforestry*

306 Agroforestry systems include a diversity of practices ranging from complex associations found in
307 homegardens, multistrata systems or agroforests to simpler systems such as alley crops, silvopastoral systems,
308 riparian plantings, shelterbelts, windbreaks or hedgerows⁹¹. Despite this broad diversity, recent reviews and
309 meta-analyses consistently suggest that the conversion of arable land to agroforestry systems increases SOC
310 stocks⁹²⁻⁹⁴. In temperate regions, SOC accumulation rates are usually around 0.92 t CO₂-eq. ha⁻¹ yr⁻¹ in the
311 topsoil (0-30 cm)⁸³. They are highly dependent on local pedoclimatic conditions and on the type and design of
312 agroforestry systems (tree density, tree species, pruning management, etc), but rarely exceed 3.67 t CO₂-eq. ha⁻¹
313 yr⁻¹^{22,26}. However, the spatial distribution of SOC stocks in agroforestry systems is usually very heterogeneous,
314 with higher stocks under the tree canopy or along tree rows^{95,97}. Several mechanisms contribute to explain SOC
315 sequestration in agroforestry systems. The main one is probably being linked to higher organic inputs to the soil
316 compared to treeless agricultural land⁹⁸, including litterfall, pruning residues, and root inputs⁹⁹.

317 A recent synthesis of N₂O emissions under agroforestry compared to adjacent agricultural lands only found
318 minor differences in net emissions, with no clear overall direction of change⁹³. However, several authors found
319 increased N₂O emissions in agroforestry, related to a greater N supply through N₂-fixing trees¹⁰⁰⁻¹⁰³ or to the
320 incorporation of tree residues^{104,105}. By contrast, N₂O emissions are often reduced in silvo-arable systems and in
321 riparian buffers⁹³. Some authors suggest that concerns over N₂O emissions from N₂-fixing trees are unwarranted
322 since fluxes from soils planted with N₂-fixing trees are similar to those fertilized with mineral N¹⁰⁶. Furthermore,
323 the yield of crops in tropical agroforestry systems may be boosted as a result of higher N-inputs from trees. In
324 temperate regions where agroforestry systems are generally planted with non-legume trees, N₂O emissions are
325 often reduced⁹³, with several processes contributing to the trend. Increased nitrogen utilization at the plot scale
326 may be due to the presence of deep-rooted trees¹⁰⁷, which are capable of taking up nitrate-N that has leached
327 below crop rooting depth¹⁰⁸⁻¹¹⁰. This process can potentially reduce the amount of N available for nitrification
328 and denitrification, and thus reduce indirect N₂O emissions. Soil water content is often lower in agroforestry than

329 in treeless plots¹¹¹, due to a higher daily water consumption by trees and crops¹¹². A drier soil profile in
330 agroforestry systems could therefore lower N₂O emissions. In temperate silvoarable systems, tree rows are
331 usually uncropped and unfertilized. This reduction in the fertilized cropping area indirectly leads to lower N₂O
332 emissions per hectare. An obvious consequence of agroforestry, especially as tends to be practiced in temperate
333 regions, is that a smaller area of land is devoted to the agricultural crop being grown. So the impact of decreased
334 N₂O emissions may be different if expressed on an area basis compared to per unit of production.

335

336 *3.6 Non-pyrogenic organic amendments*

337 A literature review¹¹³ reported increases in SOC (sometimes expressed as stocks and sometimes as
338 concentration) after prolonged large applications of organic amendments under several different agro-climatic
339 conditions. These increases ranged from 20 to 90% of the initial total SOC after few years (3-60 years),
340 compared to unfertilized controls or treatments receiving only synthetic mineral N fertilisers, with most being in
341 the range 20- 45%. A meta-analysis¹¹⁴ based on 130 observations worldwide quantified the response of SOC
342 stocks to manure application over periods ranging from 3 to 82 years. The mean manure-C retention coefficient
343 defined as the average proportion of manure-C remaining in the soil was estimated at 12% for an average study
344 duration of 18 years. The authors finally estimated a relative SOC stock change factor of 26% which was also
345 related to cumulative manure inputs. Concerning Mediterranean cropping systems, and shorter durations, a meta-
346 analysis¹¹⁵ reported that the application of organic amendments increased SOC stocks by 23.5% with an average
347 SOC storage rate of 4.81 t CO₂-eq. ha⁻¹ yr⁻¹ calculated for an average duration of 7.9 years. From these meta-
348 analyses, it seems that there is a consensus that organic amendments lead, on average, to a relative increase of
349 SOC stocks in the top soils (roughly 20-30cm) of about 25% on a 20-year time frame (or 3 times the '4 per mil'
350 target). In one example¹⁷ where manure was applied annually at a high rate compared to what is usual in
351 agrosystems, the annual rate of SOC accumulation averaged 18‰ per year in the first 20 years, then declined to
352 6‰ per year after 40-60 years, and to only 2‰ per year after 80-100 years. However, from the perspective of
353 mitigating climate change, it is arguable whether any increase in SOC stocks resulting from applications of
354 manure or similar materials can be considered as C mitigation in the sense of either a transfer of C from
355 atmosphere to land or an avoided emission. Manure is generated in agricultural systems and is almost always
356 used in some way by application to soils, though often quite inefficiently. Thus, an increase in SOC stocks at a
357 given location mainly represents a transfer of C from one site to another as opposed to a net removal of
358 atmospheric carbon¹⁷. Local additional SOC storage may not represent a CO₂ sink, i.e. a net transfer of carbon

359 from the atmosphere to the soil at the landscape scale.

360 Because organic amendments such as manures contain readily-decomposable N-rich compounds, there
361 is a significant risk that they may enhance N₂O emissions^{116–118}. Conversely, their use permits decreased use of
362 mineral N fertilizers, thereby saving N₂O emission from this source and fossil energy and the associated GHG
363 emissions from fertilizer manufacture. A further complicating factor in assessing the overall impact of manure
364 use is that indirect emissions due to storage or management are not negligible¹¹⁹. There are few reports in the
365 literature of long-term monitoring of N₂O emissions compared to data on SOC stock changes, primarily because
366 the former are much more difficult to measure. However, the effects of multiple types of organic amendments on
367 SOC storage and N₂O emissions have been evaluated in short-term experiments for various soil types, climates,
368 soil incorporation practices, and amendments types including crop residues, manure, composts of various origin
369 and maturation stages, and sewage sludge. A meta-analysis¹²⁰ concluded that the N₂O emission factors (EFs)
370 related to N inputs were mainly controlled by the C:N ratios of the added material, but that many other factors
371 influenced emission, such as soil properties (texture, drainage, SOC and N content), and climatic factors. For
372 instance, the authors observed that the EFs were on average 2.8 times greater in fine-textured soils compared to
373 coarse-textured, consistent with a previous meta-analysis⁵⁰. However, we should mention that the value of meta-
374 analyses is often limited due numerous controlling factors that are not always correctly reported in the papers
375 reviewed, and the general paucity of organic amendments characterisation in the literature. For instance, the two
376 meta-analyses mentioned in this paragraph only involved 28 to 38 individual journal articles.^{50,120} Another
377 approach is to compare organically managed soils with those managed without organic amendments¹²¹. Results
378 from such a comparison seem to indicate reduced N₂O emissions compared to situations relying totally on
379 mineral fertilizers, as show in Fig. 2. It should be noted that there is limited data from long-term studies on N₂O
380 emissions associated with additions of organic amendments; the data covers only a limited diversity of pedo-
381 climatic conditions, and especially the range of soil water filled pore space values explored.

382 *3.7 Biochar amendments*

383 Biochar (pyrolyzed organic matter amended to the soil) technology is considered by some authors to be
384 one of the methods with the highest potential to sequester carbon in soils compared to natural C cycle without
385 biochar production step⁴. The aim of biochar production from biomass pyrolysis is to produce recalcitrant
386 organic matter (i.e., charcoal and biomass-derived black C) which is then added to the soil. For this reason,
387 biochar can be considered as a negative emission technology different from other soil C sequestration methods⁵.

388 Biochar properties and effects on SOC stabilization strongly depend on the feedstock material and pyrolysis
389 conditions (e.g., maximum temperature, heating rates)^{122–126}, as well as biochar ageing and soil
390 properties^{127,128}. The efficiency of biochar for C sequestration is two-fold as compared to simply relying on soil
391 stabilization processes. First, slow pyrolysis for biochar production results in a much higher proportion of the
392 feedstock C bound in persistent molecular structures than through in situ stabilization by addition of unprocessed
393 organic matter to soil¹²⁹. With a slow pyrolysis at about 500°C, approximately 50% of the carbon contained in a
394 feedstock of Miscanthus or maize cobs ended up within the biochar and can therefore be assumed to be more
395 stable than carbon in the raw biomass¹³⁰. This compares with only 8-12% of straw residue returned to the field
396 being transformed into longer-lived SOM forms^{27,131}. Thus, pyrolysis is about four times more efficient than
397 SOM-formation processes to produce persistent C in soils. Second, field studies show that biochar has a longer
398 mean residence time in soils than SOM, i.e. >100 years¹³² vs. about 50 years for the latter¹³³. Combining effects
399 of the higher persistent-C yield with that of the longer mean residence time, biochar appears at least 8 times
400 more efficient at storing SOC than the return of non-pyrolysed residues. In meta-analyses, biochar amendment
401 tends to increase the soil organic carbon stocks by 40% but the studies used were generally short term (no more
402 than 4 years)¹³⁴. Nevertheless, this result must be considered with due care since it is not straightforward to
403 measure the effect of biochar, which is mainly C, on native SOC but one published study suggests that biochar
404 amendment increase total SOC including non-biochar C¹³⁵. In addition to the direct inputs of pyrolyzed biomass
405 to the soil, recent studies showed that biochar amendments could increase (positive priming), decrease (negative
406 priming) or have no effect on the mineralisation of native SOM. The biochar effect on the magnitude and
407 direction of priming is influenced by the incubation period and pyrolysis temperature^{122,123,136}. Positive priming,
408 which would cause destabilization of SOM, thus offsetting part of the increased SOC storage, could result from
409 the biochar affecting microbial biomass activity and enzyme production¹³⁷ through changes in availability of
410 organic substrates and nutrients, and modification of microorganism habitat associated with the great porosity
411 and large specific surface area of charcoal particles¹³⁷. Conversely, some studies showed that biochar-induced
412 negative priming, leading to further SOC storage in addition to direct biochar-C inputs, resulted from the
413 enhancement of organo-mineral interactions and soil aggregation with biochar^{138,139}, and a greater adsorption of
414 dissolved organic carbon onto biochar particles¹⁴⁰. In addition, biochar amendments have been shown to increase
415 soil water holding capacity, the availability of some nutrients (Ca²⁺, Mg²⁺ in particular) and to increase soil pH.
416 All of these mechanisms could further enhance crop productivity and biomass inputs into soil^{141–144}, with clearer
417 effects on crop yields in highly weathered tropical soils¹⁴⁵. Nevertheless, to process biomass into biochar,

418 transport it and incorporate it to the soil, some energy is needed (possibly produced by pyrolysis) and the related
419 GHG emissions associated with this process must be accounted for to calculate a full GHG balance for biochar-
420 Life cycle assessments (LCA) have shown that a positive balance can be obtained, illustrated by GHG reductions
421 up to 2.74 t CO₂ equivalent per ton of biochar amended on volcanic soils from Southern Chile¹⁴⁶. The balance
422 can also significantly increase when plant biomass production is accompanied by an efficient use of the
423 bioenergy produced during the pyrolysis process in order to maximize climate benefits from biochar production
424 followed by addition to soil, as shown by an LCA performed in Spain¹⁴⁷; implementation costs were also
425 decreased. One simulation study suggests that the maximum sustainable technical potential of biochar to
426 mitigate climate change, involving the widespread use of biochars, without threatening food security and
427 landscapes, could be a mitigation of 12% of current anthropogenic CO₂ emissions (1.8 Pg CO₂-C equivalent per
428 year)¹⁴⁸. The C:N ratio of SOM approximates 14¹⁴⁹ while that of biochar is generally higher than that of its
429 feedstock, i.e. generally > 50 for straw biochar and > 100 for wood biochar. It takes therefore at least five times
430 less N to stabilize organic C in the form of biochar than in the form of SOM. Beyond this critical observation,
431 biochar has other important interactions with the N cycle, notably: 1) volatilization and immobilization of N
432 during the pyrolysis process¹⁵⁰, 2) reduction of N₂O emissions after application to arable fields¹⁵¹, 3) reduction of
433 NO₃ and NH₄ leaching¹⁵². Emissions of N₂O from soils are in most cases substantially reduced by biochar
434 addition: a recent meta-analysis reported an average decrease of 38% across studies¹⁵¹. However, most
435 measurements are fairly short-term, the majority in this meta-analysis being <30 days. This effect appears
436 consistent when biochar is produced at over ~450C, so that the product is both high pH and high surface area
437 while containing very little labile C¹²⁵. The contribution of N₂O emissions attenuation with biochar was shown
438 to be negligible in the LCA performed in Southern Chile, compared to the climate change mitigation associated
439 to C storage¹⁴⁶. Furthermore, reductions in N₂O emission with biochar appears only significant for the first year
440 after application, which suggests that frequent applications are necessary to maintain such an effect. In view of
441 the large quantities of biochar usually applied in such studies, this may greatly limit the practical and/or
442 economical potential for using biochar as a method for decreasing N₂O emissions.

443 A way to capitalize on the positive interactions between biochar properties and the N cycle is through
444 the development of biochar-based fertilizers made by mixing biochar with mineral or organic sources of
445 nutrients¹⁵³. This method aims at reducing nitrate losses and N₂O emissions, and at increasing N use efficiency.
446 Moreover, biochar tends to adsorb mineral N and the mixing with a nutrient-rich material prevents potential N
447 deficiency created by field application of large amounts of raw biochar. Some biochar structures have been

448 successfully loaded with nitrate ions through co-composting, which could greatly increase the fertilisation value
449 of the product¹⁵⁴. Producing biochar fertilizers requires the development of appropriate technologies. For
450 example, mixing raw biochar - a high pH product- with manure and slurries can result in large amounts of NH₃
451 being volatilized. Therefore, biochar acidification is generally required when making biochar fertilizers from
452 organic feedstocks. However, biochar is also a strong sorbent for NH₃¹⁵⁵, which may be captured from the
453 atmosphere during the pyrolysis process and made available to plants later. This is a promising technology to
454 abate anthropogenic emissions of NH₃¹⁵⁵ as well as directly reduce NH₃ volatilization from soils¹⁵⁰. In
455 conclusion, pending proper technology, biochar may be intimately mixed with N sources and applied each year
456 as a fertilizer to maximize reductions in both N₂O emissions and nitrate leaching, while sequestering C in a
457 structure requiring little N. However, further studies are needed to validate the scant results currently available.

458 *3.8 Overview of the current evidence*

459 Fig. 2 summarises published data on rates of change in SOC and rates of emission of N₂O resulting from four
460 prominent sets of practices designed to increase SOC, namely agroforestry, cover crops, no-tillage and organic
461 amendment. All fluxes are expressed in CO₂ equivalents, using a global warming potential value integrated over
462 100 years and including global warming potential (GWP) of 298 for N₂O as recommended by last IPCC report¹⁵⁶.
463 GWP is the time-integrated radiative forcing induced by a pulse emission of a given component (here N₂O),
464 relative to a pulse emission of an equal mass of CO₂¹⁵⁶. The data on Fig. 2 were taken from several meta-
465 analyses and review papers. Here, we did not re-analyse the data gathered by such meta-analysis, but rather
466 presented the mean effect size from each study converted in CO₂ equivalents. When results were given for the
467 whole experiment duration, we divided by the duration of the experiment to obtain the mean annual SOC
468 storage/ N₂O emissions (see supplementary information for detailed methods). The data in Fig. 2 is based on
469 over 700 measurements of SOC change and 200 measurements of N₂O. Even allowing for some papers being
470 cited in more than one meta-analysis, this is a large body of data and, to our knowledge, has not previously been
471 assembled in this way. At first sight it appears that SOC increases produced by the four sets of treatments varied
472 widely from -0.52 ± 0.46 to -6.74 ± 1.21 t CO₂-eq. ha⁻¹ yr⁻¹, the negative sign representing accumulation of SOC,
473 i.e. transfer of C from atmosphere to soil. However, this wide range is somewhat deceptive as the two largest
474 values are from very specific situations. The value of -6.74 ± 1.21 t CO₂-eq. ha⁻¹ yr⁻¹ for agroforestry (Kim *et al*,
475 2016) is from 34 sets of data for systems with a particularly high tree density (see legend to Fig. 2); the other two
476 meta-analyses for agroforestry, based on >200 datasets, give values of less than half this at around -3 t CO₂-eq.
477 ha⁻¹ yr⁻¹. For cover crops the majority of data, based on 186 datasets, lead to mean rates of C accumulation in the

478 range of -1.2 to -2.0 t CO₂-eq. ha⁻¹ yr⁻¹. The highest value observed for cover crop was 3.67 t CO₂-eq. ha⁻¹ yr⁻¹
479 but it specifically refers to cover crops included between the wide rows of Mediterranean woody crops (olive,
480 almond and vineyards) where the soil would otherwise be bare¹⁵⁷. This is clearly an important management
481 change within this environment but represents a much greater input of plant material than, say, cover crops
482 grown during winter within temperate arable cropping systems. For both agroforestry and cover crops, and even
483 ignoring the two sets of data for SOC increases that are especially large, it appears from the summarized data in
484 Fig. 2 that SOC increases resulting from these two changes in management considerably outweigh increased
485 N₂O emissions when both are expressed on a CO₂-eq basis and these management changes can be expected to
486 beneficial for climate change mitigation. For no-tillage the situation is different: the relatively small rates of SOC
487 accumulation are approximately equal to the increases in N₂O emission when both are expressed on a CO₂-eq
488 basis, so there appears to be no overall climate change benefit. For organic amendments the results indicate that
489 N₂O emissions are decreased and thus reinforce the SOC benefit, though this is based on a very limited amount
490 of data and also, as discussed earlier, it is questionable whether SOC increases from addition of organic
491 amendments can be fully regarded as climate change mitigation. It should be noted that most of the studies are
492 performed over a few years and assessment of GHG balance in the long term, especially for N₂O, are still
493 missing²⁸. Some practices were too little documented or with not enough information to be compared with the
494 others (e.g. , biochar or erosion control).

495

496 4. DISCUSSION AND OUTLOOK

497 Overall, the meta-analysis we gathered here (Fig. 2) suggest that, with the exception of reduced tillage
498 practices, increased N₂O emissions are not sufficient to invalidate the GHG abatement potential achieved by
499 SOC sequestration strategies. Some sequestration strategies (e.g. biochar or non-pyrogenic organic amendment
500 application) may even generate win-win situations through a decrease in N₂O emissions, although the
501 experimental evidences are still scant. In addition, the economic impacts and large-scale effects of the options
502 examined here warrant further assessment. Some practices may affect crop yields or farmers' income, depending
503 on pedo-climatic conditions and the details and practicalities of the cropping systems. For instance, conservation
504 practices, and especially no-till may slightly decrease crop yields under temperate climates but be beneficial in
505 drier conditions¹⁵⁸. Similarly, the yield of arable crops is usually reduced in agroforestry systems in temperate
506 regions¹⁵⁹ but in more arid climates, crops perform better¹⁶⁰. Nevertheless, for agroforestry, trees produce timber,

507 firewood, honey, fruits, etc. that are also a source of incomes for the farmers and may lead to greater overall
508 sustainability. Beyond yield impacts, some socio-cultural or economic factors come into play that may hinder the
509 adoption of carbon sequestration practices. For example, in the United States, the cost of carbon capture through
510 Natural Resources Conservation Service programs is estimated at US \$ 32-442 per tonne of CO₂, with an
511 average of US \$ 183¹⁶¹. A carbon price much higher than the present value (around US \$ 10 as a global
512 average¹⁶²) would be necessary to promote carbon sequestration practices, as well as a regulation to direct the
513 financial flow of industrial and energy emitters to the agricultural sector.

514 To be deployed at large scale and to enter emission trading systems, the GHG fluxes of each change in
515 agricultural practices should be estimated precisely. Various models may be used to account for scale or leakage
516 effects such as indirect land-use changes¹⁶³. The methods currently available include data-driven approaches
517 based on worldwide measurement networks¹⁶⁴, statistical or empirical flux-upscaling models^{57,164}, process-based
518 models and, lastly, integrated assessment models (IAM)¹⁶⁵. Process-based models include a representation of N
519 cycling processes, which are an essential tool in assessing and predicting the terrestrial N cycle and N₂O fluxes
520 in response to multi-factor global changes. Such models have been used to estimate N₂O emissions from natural
521 and agricultural soils at various scales, from field to global level via the integration of a prognostic N cycle into
522 different land surface models¹⁶⁶. As an example, Fig. 3 shows the results of simulations by various models at
523 global scale. Most of N₂O emissions from cropland are due to the use of mineral fertilizers (Fig. 3a) and are
524 mainly located in USA, Europe, India and China. They may be used to quantify carbon sequestration in soil
525 minus the N₂O emission trade-off at global scale, based on ensemble runs as was initiated in the global N₂O
526 Model Inter-Comparison Project¹⁶⁷.

527 IAMs focus on the interactions between the economic activities and earth system responses and are vital
528 for estimating what socioeconomic changes would be needed to reduce GHG emissions across sectors and
529 increase biospheric C sinks¹⁶⁸. Until recently, most IAMs did not explicitly take into account SOC restoration
530 practices¹⁶⁹. A recent study that did include them found that soils could be a sink of 3.5 GtCO₂-eq/yr by 2050
531 under a carbon price of 190 USD/tCO₂¹³. This carbon mitigation option, if achievable in practice, would reduce
532 the burden of climate stabilization for all sectors of the economy, including agriculture. In addition, SOC
533 increases are often correlated with higher crop yields and contribute to a range of other environmental benefits
534 and increased sustainability of agricultural systems. Practices designed to increase SOC can offer a win-win
535 solution *vis a vis* food security, by mitigating food calorie losses resulting from the application of emission
536 reduction targets (e.g. through decreased applications of mineral fertilizers) and reducing undernourishment.

537

538 Finally, many of the practices reviewed (Table 1) here may be combined on a given field: for example,
539 no-till can be combined with cover crops, organic amendments, or agroforestry. Such combinations have been
540 little tested in practice and in particular synergetic effects between them have not been evaluated in depth^{170,171}.
541 Conversely, they may come with trade-off, antagonistic or synergistic effects regarding SOC storage rates, as
542 well as N₂O emissions or other impacts and these needs to be identified and quantified. Furthermore, proper
543 assessment of carbon sequestration measures raises classical GHG accounting issues, such as double counting,
544 improper setting of system boundaries and counterfactual scenarios¹⁶⁹. Although further research is still needed
545 to quantify the potential of SOC sequestration options on a local to regional basis, it appears that their potential
546 to mitigate climate change, even when factoring in N₂O emissions is still significant and that they deserve further
547 consideration in climate stabilization scenarios. Including the state-of-the-art knowledge reviewed here on the
548 effectiveness of such measures in land system or integrated assessment models could be a prime target to assess
549 their impacts at global scale.

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562 ITEMS:

563 Table 1: Summary of the effects of management practices on soil organic carbon (SOC) storage and N₂O
 564 emissions.

Management Practice	Effect on soil C stocks	Effect on N₂O emissions (<i>a priori</i>)
Reduced tillage / zero tillage	Reduced C loss/Increased C inputs to soils when associated with a reduced weed management	Promote denitrification (anaerobiosis)
Erosion control (contour plowing, terracing)	Reduced C loss	Unclear
Addition of non pyrogenic organic amendments (compost, manure, crop residues)	Increased C input but in some cases (e.g. manure) rather a transfer from one terrestrial location to another than a transfer of C from atmosphere to soil	Enhanced denitrification rate (via anaerobiosis and the supply of electron donors), and soil N availability
Use of cover crops	Reduced C loss/increased C input	Decreased denitrification because of N uptake by plants; may be compensated for by N inputs from BNF
Biochar	Increased C input	Decreased nitrification due to adsorption of mineral N with biochar.
Agroforestry	Increased C input, reduced C loss, increased aggregate stability	Decreased denitrification (lower soil moisture, increased soil porosity, increased nitrogen uptake), except for N ₂ -fixing trees (increasing soil available N)

565 Figure legends

566 Figure 1: A schematic representation of C-N interactions in the terrestrial ecosystem. Note that biological
567 nitrogen fixation and denitrification are process performed by microorganisms that also need C as substrate.

568 Figure 2: Estimation of the SOC storage and N₂O emissions of land-based mitigation options expressed in CO₂
569 equivalents. Negative values indicate a net reduction in GHG emissions in terms of CO₂ equivalents, while
570 positive values show a net increase of CO₂ equivalent emissions. All values refer to the difference between the
571 land-based mitigation option in question and a “control” land (e.g. no-tillage vs conventional tillage). For
572 agroforestry, the control land is cropland and different types of agroforestry systems were considered. NB: In
573 Kim et al. (2016) the majority of soil C storage data comes from intercropping, improved fallows and rotational
574 woodlots, which are systems with high tree density. This could partially explain the very high estimation of soil
575 C storage found in Kim et al. (2016) compared to other papers. Organic amendments do not include biochar. The
576 control used for comparison with organic amendments is an experiment managed with inorganic fertilizers. For
577 cover crops meta-analysis, Vicente-Vicente et al. (2016) only consider Mediterranean woody crops (olive,
578 almond and vineyards), which could also explain the large soil C rates estimated. Uncertainty is given as
579 standard error (SE) for every paper. If it was provided as a confidence interval (CI) or standard deviation (SD) it
580 has been adequately transformed to unify the units. (*Reviews; ** For these meta-analysis the values reported in
581 the graph have been recalculated as the weighted mean across all experiments, from the database provided by the
582 authors, because the values coming from the papers could not be used as they were reported as a percentage
583 only).^{121,157,172,173}

584 Figure 3: Spatial and latitudinal patterns of contributions of fertilizer (a) and manure (b) on cropland soil N₂O
585 emissions obtained during the global N₂O Model Intercomparison Project¹⁶⁷. Average over the 2006-2015
586 period.

587

588 **[Box 1: Intertwined soil C and N cycles.** Primary producers fix atmospheric CO₂ through
589 photosynthesis and produce biomass that subsequently enters the soil *via* rhizodeposition and litter production or
590 *via* organic amendments or deposition due to erosion or dissolved organic carbon (DOC) leaching into the
591 subsoil. Organic C entering soil is further processed by soil organisms into organic by-products and subsequent
592 CO₂ losses during mineralization. SOC can also be lost through fire, or displaced by erosion/deposition and
593 lateral transfer of DOC. The net difference between C inputs and outputs determines how much organic C is
594 stored in the soil. A part of this carbon is stabilized for decades to centuries through several mechanisms such as
595 interactions with the soil mineral matrix, chemical recalcitrance or protection within aggregates. Nitrogen can
596 enter the soil *via* atmospheric deposition or biological N₂ fixation, or as mineral or organic fertilizers. Nitrogen
597 can leave the soil through plant uptake, leaching or gaseous emissions. The critical N pool sustaining plant
598 growth is mineral N (ammonium (NH₄⁺) and nitrate (NO₃⁻)), which either originates from the mineralization of
599 soil organic matter and ammonification as well as from mineral fertilizers inputs. Specific chemotrophic
600 microorganisms called nitrifiers can oxidize ammonium, first into nitrite (NO₂⁻) and then into nitrate. The reverse
601 process called dissimilatory nitrate reduction can occur in anaerobic conditions¹⁷⁴. Heterotrophic denitrifying
602 communities can also use nitrate and reduce it to N₂. In each of these processes, nitrous oxide (N₂O) can be
603 generated¹⁷⁵. Net N₂O emissions from the soil will usually be lower when the amount of soil mineral N is low
604 and when soil pH is alkaline^{55,175} but also when C and oxygen availability are reduced^{56,175}. N can also be lost as
605 NH₃, or as other gaseous forms of N oxides that can be deposited and contribute to indirect N₂O formation.]

606

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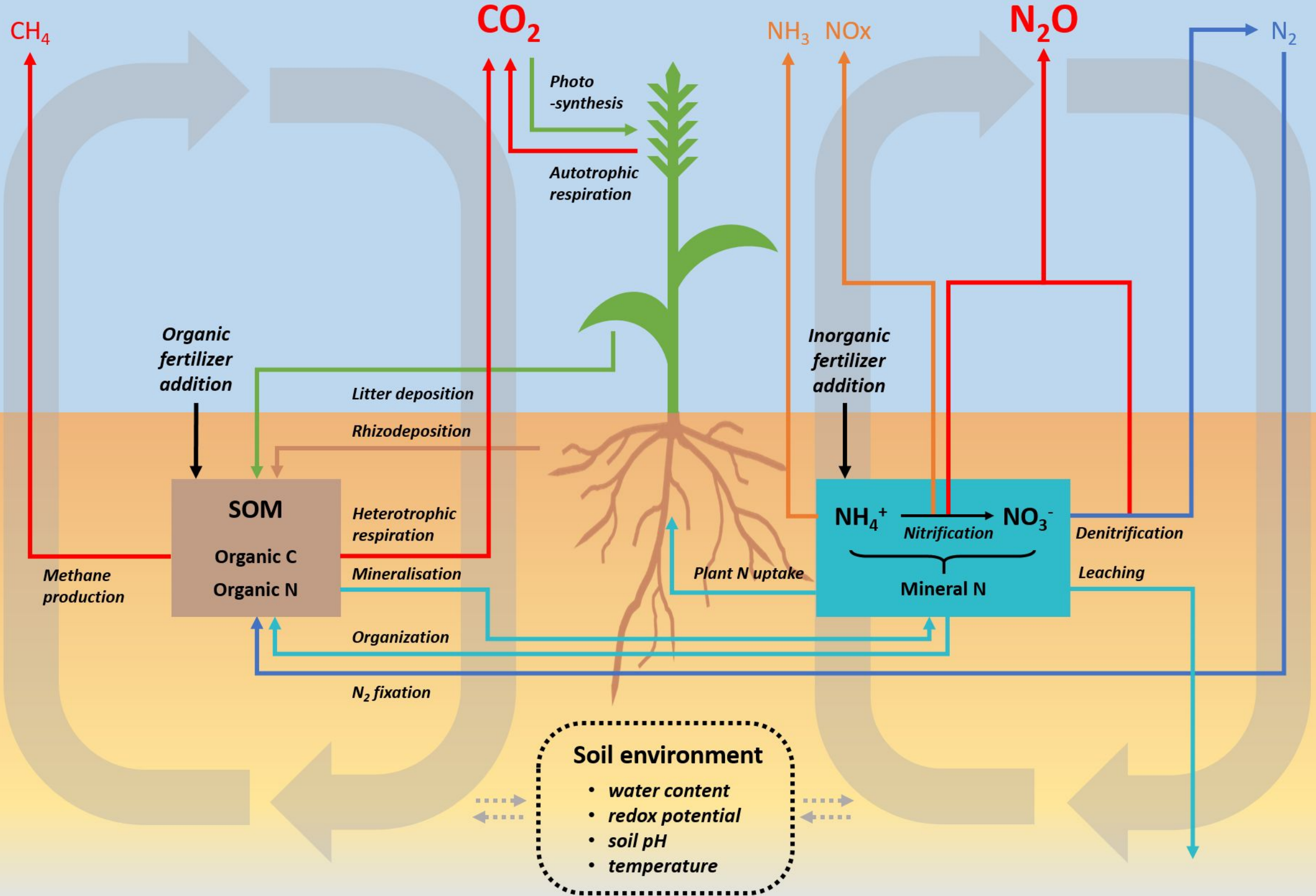
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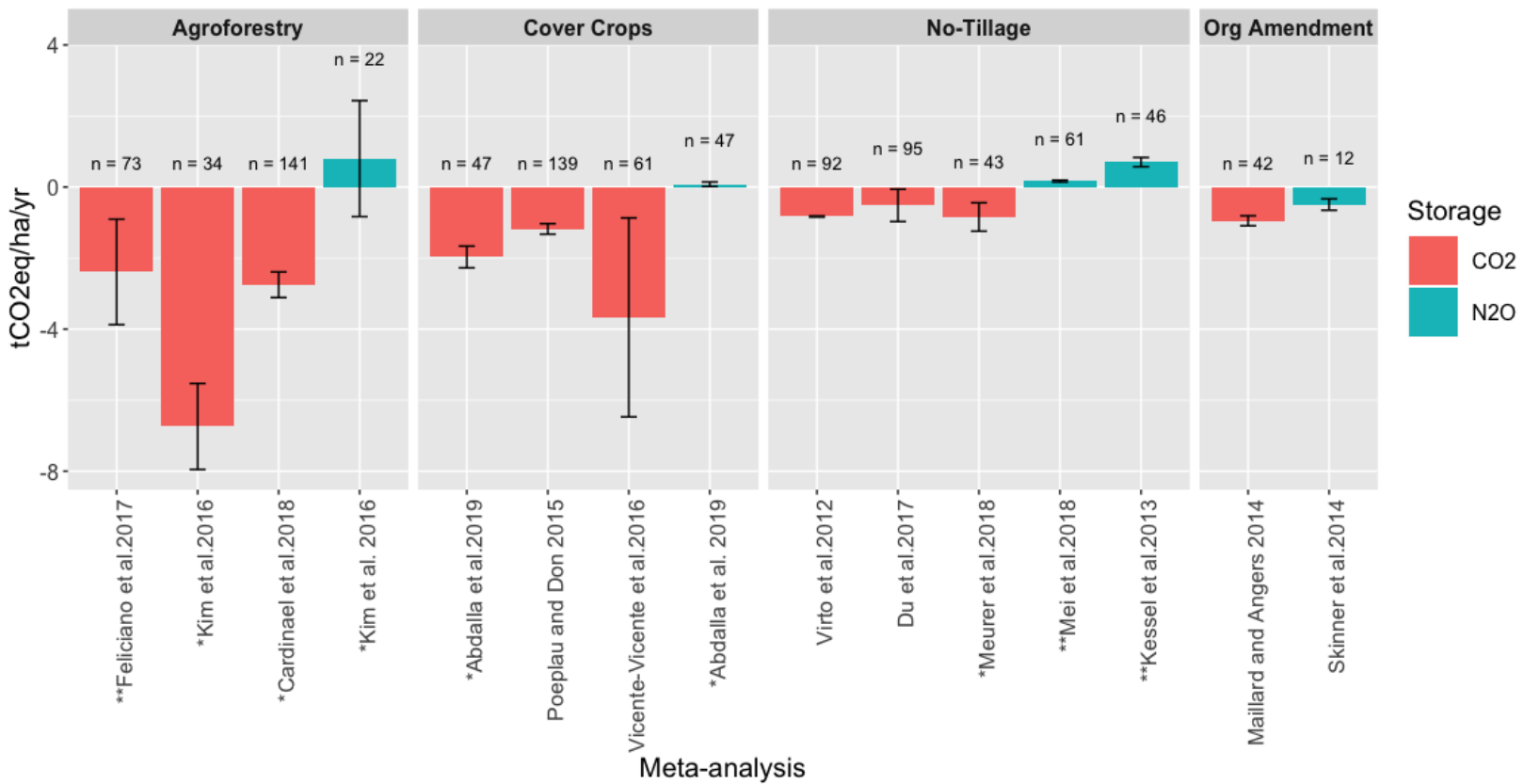
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Soil C cycle

Soil N cycle





n = number of experiments

