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Can N_2O emissions offset the benefits from soil organic carbon storage?

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ABSTRACT

To respect the Paris agreement targeting a limitation of global warming below 2°C by 2100, and possibly below 1.5 °C, drastic reductions of greenhouse gas emissions are mandatory but not sufficient. Large-scale deployment of other climate mitigation strategies are also necessary. Among these, increasing soil organic carbon (SOC) stocks is an important lever because carbon in soils can be stored for long periods and land management options to achieve this already exist and have been widely tested. However, agricultural soils are also an important source of nitrous oxide, (N_2O) a powerful greenhouse gas, and increasing SOC may influence N_2O emissions, likely causing an increase in many cases, thus tending to offset the climate change benefit from increased SOC storage. Here, we review the main agricultural management options for increasing SOC stocks. We evaluate the amount of SOC that can be stored as well as resulting changes in N_2O emissions to better estimate the climate benefits of these management options. We conclude that the climate mitigation induced by increased SOC storage is generally overestimated if associated N_2O emissions are not considered, but is never fully offset. Some options (e.g, biochar or non-pyrogenic amendment application) may even decrease N_2O emissions.

Key points:

- Carbon and nitrogen cycles in soil interact in numerous and complex ways and an impact of a land management change on one cycle will generally influence the other.
- Several land management options designed to increase soil organic carbon stocks exist and have been widely evaluated.
- Land management options to increase soil organic carbon also tend to increase nitrous oxide emissions in some cases.
- We conclude that when increased N_2O emissions are taken into account, they partially offset the climate benefits of increased organic carbon storage, but never negate them completely.

1. INTRODUCTION

The United Nations Framework Convention on Climate Change (UNFCCC) Paris Agreement adopted in 2015 aimed at keeping global warming below 2°C by 2100, and at possibly further limiting the temperature increase to 1.5 °C. The Paris Agreement specified that the balance of anthropogenic greenhouse gas (GHG) emissions and sinks should be attained by the second half of this century. This requires not only drastic reductions in GHG emissions in the near future, but also net negative emission technologies because not all emissions will be reducible to zero within this time scale[1,2]. To a large extent, these negative emissions imply land-based mitigation strategies[3], mostly involving the production of organic matter by plant photosynthesis coupled with carbon storage in living biomass and / or soil organic matter[4]. A pathway frequently discussed known as bioenergy with carbon capture and storage (BECCS) comprises generating energy using biomass, capturing the CO_2 evolved from this process and storing it in geological reservoirs. The deployment of BECCS faces both technical challenges and most likely limitations due to high costs and adverse environmental impacts[5,6]. On the other hand, the net removal of atmospheric CO_2 taken up by plants in agricultural soils (i.e., carbon sequestration) has recently come under sharp focus as a more affordable and practical alternative, potentially associated with positive economic outcomes and possibly applicable at large scale in managed lands[7,8]. The role of soils as a key component of the global carbon cycle is now recognized by the scientific community and also by policymakers[5,6]. Soils have never been harnessed at large scale for the purpose of sequestering carbon, although they currently make up the largest reservoir of organic carbon in the terrestrial biosphere, with a size of 1,500 Pg C to a depth of one meter[9]. However, the ecosystems which contain the largest stocks of soil organic carbon (SOC) are unmanaged (comprising boreal forests, permafrost soils and wetlands), whereas only soils from managed ecosystems, in particular agricultural soils, may be managed to increase SOC stocks (i.e., carbon sequestration) Agriculture is also a key target sector for the reduction of methane (CH_4) and nitrous oxide (N_2O) emissions[10–12]. Yet, few countries have included agriculture in their nationally determined contributions – a roadmap volunteered by national governments as part of the Paris Agreement to express their efforts to reduce GHG emissions – because of potential trade-offs with food production and uncertainties on achievable potentials[13].

Recent emphasis on promoting SOC storage has resulted in international initiatives such as the “4 per mil” initiative launched by France during the UNFCCC conference of the parties (COP) 21[7,14]. It relies on the concept that even a very small relative increase in SOC pools worldwide could offset a significant fraction of CO_2 emissions. Preliminary evaluation indicated that increasing global agricultural SOC stocks at an annual rate of

4‰ would result in a C sequestration potential of 2-3 Pg C yr^{-1}[17]. This may contribute significantly to the objectives of the land sector to achieve the Paris agreement target[15]. Nevertheless, several studies have discussed and criticized the feasibility of enriching soils at a rate of 4‰ over a sustained period of years [16–18] because: (i) it requires large amounts of new organic matter inputs, (ii) it requires large amounts of nutrients, (iii) it is difficult to achieve this target rate in all agricultural systems, and (iv) it may be hampered by the climate change-induced enhancement of SOC decomposition. Moreover, altered management practices may impact farmers' income and imply trade-offs with food production[17]. Data from long-term experiments show that it is very difficult to achieve the 4 per mil rate in temperate arable systems without drastic changes in management[17,19]. Finally, the annual rate of SOC increase generally levels off over time as the SOC pool increases and approaches a new equilibrium level[20].

Nevertheless, concrete management options exist to increase SOC stocks such as cover crops, tillage management, crop rotations, organic amendments, agroforestry and biochar amendments with effects depending on local conditions [21–23]. It must be noted that organic amendments may, in some case such as manures, be a transfer of carbon from one terrestrial location to another rather than a net removal of carbon from the atmosphere[17]. Nevertheless, well spatially distributed, organic amendments contribute to significant increase the SOC pools at regional scale[24]. These options have socio-economic impacts on farmers and land managers and indirect effects on ecosystem services, through changes in crop yields, water consumption, nitrate leaching, and CH_4 and N_2O emissions which have to be considered when evaluating the feasibility and the relevance of implementing SOC storage options. The SOC storage potential of the various practices has been extensively assessed in the recent scientific literature [17,23,25–27], and recently revised by IPCC in its 2019 report on climate change and land[12]. However, implications for the N cycle (in particular N_2O emissions), and other biogeochemical cycles or crop yields have not been thoroughly documented so far[28–30]. Neither have been the consequences of large-scale deployment of these measures, and constraints related to the nitrogen (N), phosphorous (P) and potassium (K) cycles. These aspects are important because they determine the overall GHG abatement efficiency of mitigation measures and set limits on their potential deployment. C and N cycles are strongly interlinked (box. 1) in particular in soils and we assume that the deployment of land based mitigation options to increase SOC may impact the N cycle and the associated N_2O emissions. A recent modelling study suggests that measure to increase SOC sequestration might be offset by increased N_2O, depending on the crop rotation and on the duration of the land management practices[28]. A better understanding of such interactions is necessary to evaluate the benefits of different management practices aimed at increasing SOC storage and to predict the full GHG balance of each practice.

Here, we focus on the interactions between soil C and nutrient dynamics, and in particular on N dynamics and N_2O emissions. The aims of the paper are to i) describe the mechanisms linking the C and N cycles in soils, ii) assess how N_2O emissions may be affected by increased SOC pools as a land based mitigation option, iii) review our knowledge on the other impacts of these practices.

2. INTERWINED SOIL CARBON AND NITROGEN CYCLES

Because C and N cycles are tightly coupled in soils, altering one will affect the other as shown in Fig. 1. In soils the C and the N cycles are sometimes totally interdependent, in particular when both are in organic forms but are sometimes uncoupled when C or N are present as minerals. Nevertheless they may still interact with each other. This section summarizes the main mechanisms explaining how changes in the soil C cycle and SOC sequestration interact with N cycle processes, and in particular N_2O emissions (Fig. 1, Box 1). The first reason why soil C and N dynamics are interdependent is that both elements are stored predominantly as organic forms in the soil, sometimes within the same compounds (amino acids, proteins, etc.), thus mineralisation generally affects both. Moreover, the availability of mineral N in the soil controls a number of processes in both cycles and *vice versa*. For instance, mineral N transformations depend on carbon availability and plant dry matter production is limited by N availability. Nitrogen is needed to sustain photosynthesis and other physiological processes[31]; therefore higher N availability would likely lead to greater primary productivity and inputs of plant-derived organic matter to the soil[32]. On the other hand, higher N availability also tends to lower the allocation of photosynthates to the root system[33]. As root-derived C inputs contribute at least 2-3 times more than shoot-derived C inputs to SOC storage[34,35], a high soil N availability could theoretically increase the plant biomass but the plant biomass produced might not be transformed into SOC as efficiently because of a reduced amount of root-derived C entering the soil[36]. Soil organic matter (SOM) turnover (i.e., rate of mineralization and transformation of SOM) also depends on the N availability for microorganisms. While a low mineral N availability may limit the mineralization rate of plant residues and amendments[37,38], the combination of regular inputs of fresh organic C with a low soil N availability can lead to positive priming effect, *i.e.* a higher rate of SOM mineralization, and a lower SOC storage potential[39,40]. Moreover, because of the relatively narrow range of C:N ratios of SOM in mineral layers[16] and because of the importance of soil microbial processing for building up stable SOM[41] in some ecosystems, large amounts of N are inevitably required to stabilize large amounts of SOC[42,43]. Because of this stoichiometric requirement, it might seem acceptable to maintain a high availability of

N in the soil by applying large amounts of mineral fertilizers. Such a strategy would however lead to potential N losses, *e.g.* as N_2O emissions or nitrate leaching from soil, and further increase GHG emissions during fertilizer production. Thus the modest increases in SOC resulting from N fertilizer applications up to sensible agronomic rates are welcome in the context of C sequestration, but it would be counter-productive and inappropriate to recommend higher rates of N application aimed at promoting an additional increase in soil C.

Input of N to terrestrial ecosystems by biological N_2 fixation is another example of a close link between C and N resources. Root-associated or free-living N_2-fixing bacteria depend on the availability of organic C resources for sustaining their heterotrophic needs, which may explain why N_2 fixation is only triggered when the amount of soil mineral N is low. In particular, the energy cost of N_2 fixation represents between 5% and 23% of daily photoassimilated C[44]. The associated CO_2 losses by respiration may therefore decrease the amount of plant C entering the soil. However, the consequence of this on the potential of SOC storage remains unclear. For example, the presence of leguminous plants can result in lower belowground C inputs compared to gramineous plants, leading to lower SOC concentrations[45,46]. However, the net inputs of N to soils by leguminous plants have been shown to correlate with a net accumulation of SOC, by providing the organic N required to stabilize an additional amount of SOC in soils[47]. Similarly, crop rotations that include leguminous plants appear store more SOC than conventional monocultures[48], although this effect may be mainly due to longer periods of plant cover, and to the presence of deeper root systems than to biological N_2 fixation itself[47]. These feedbacks also depend on which non-leguminous plants are associated[49] to the N_2-fixing plant, and may lead to contrasting results in terms of SOC storage[24]. Of course, obtaining N from legumes, where this is practicable, rather than from N fertilizer does eliminate the GHG emissions associated with N fertilizer manufacture.

Fresh C inputs to the soil through root exudates or amendments may temporarily decrease or increase soil pH, affecting the magnitude of N_2O emissions. Their consumption by microorganisms may also decrease the local concentration of oxygen, leading to anaerobic conditions which are favourable to denitrification and N_2O emissions[50]. Furthermore, because organic materials generally act as electron donors in the denitrification process and because soil organic matter content may lower the redox potential of the soil[51], increasing the amount of soil organic matter may also increase the activity of denitrifiers and therefore increase N_2O emissions[52,53]. These mechanisms likely explain why higher SOC contents in soils have indeed been shown to correlate with larger N_2O emissions[54,55]. N_2O emissions represent a particular case that illustrates how the soil N cycle may be influenced by the C cycle. As a rule, net N_2O emissions from the soil at a given soil water-filled pore space (WFPS) will usually be lower when the soil mineral N content is low and when soil pH is alkaline or when C availability is reduced. Furthermore, because a low soil redox potential (< 400 mV)[52] is required for denitrification, N_2O emissions have been suggested to have their optimum at 70-80% WFPS, while prolonged waterlogging conditions may result in complete nitrate reduction to N_2 instead of N_2O [56]. Several mechanisms can therefore explain why attempts to modify the soil C cycle may also affect N_2O emissions. On a longer time scale, the build-up of SOC by various strategies may be expected to increase the retention of water and fertilizer-N in the rooting zone through improved soil properties (eg.. water holding capacity, porosity, hydrophilicity) , in a manner favourable for the denitrification to occur. This might trigger a higher primary production and enhancing further SOC storage, but also increase the risk of N_2O emissions because of the increase in N sources and the shift to soil environmental conditions more favourable to N_2O emissions. In the remainder of this paper we consider possible interactions between increased SOC and changes in N_2O emission for a range of management practices designed to increase SOC (Table 1).

3. HOW SOC STORING PRACTICES AFFECT N_2O EMISSIONS

3.1 Balancing the Nitrogen inputs

Since mineral N availability drives N_2O emissions as well as crop productivity and C inputs into the soil[57,58], N fertilization should be carefully managed. A balance should be obtained between N inputs (including fertilizers, manures and biological nitrogen fixation through symbiosis between N_2-fixing bacteria and some plant species) and N exported in harvested products in order to reduce a N surplus that can be source of N_2O, but without a major negative effect on crop productivity. This N surplus should ideally be zero, but it is actually large and positive in many regions of the world, having intensive agriculture (e.g. parts of China, India, Europe, North America), and negative in other regions (e.g. Africa)[59]. Excess N associated with a positive surplus is a major cause of N_2O emissions on farms, but also of nitrate leaching losses, part of which contributes to indirect N_2O emissions if nitrate is denitrified within surface waters. Overall, N surplus is a strong driver of N_2O emissions, especially when considering that the rate of emission is no longer linear for high N input[60]. The relatively low cost of mineral N fertilizers in developed countries compared to the price of agricultural products incentivizes farmers to apply more N than recommended by good practices, as an 'insurance' against unforeseen N losses due to climate variability. In some regions of the world, but not all, there is considerable potential to lower agricultural N_2O fluxes in intensive farming by reducing the N surplus without affecting farmers' incomes [61]. Therefore, the use of mineral N to increase crop productivity may induce an increase of C input into the soil but a complex balance must be found to avoid excessive N_2O emissions and N leaching.

3.2 Reduced tillage/non tillage

The effect of reduced tillage has attracted attention as a practice leading to increased SOC storage.. However, recent meta-analyses demonstrate only a small positive effect of no-tillage on SOC stocks in the topsoil (0-30 cm layer) compared to conventional tillage, while it may vary widely across pedo-climatic situations[62–66]. Moreover, it must be recognised that the largest impact of reducing tillage is a redistribution of SOC toward the soil surface[25,64–67]. As a consequence, data from field trials must be carefully examined to distinguish between a genuine increase in SOC stocks in the surface soil layers from a simple change in the vertical distribution of SOC concentration.

There has been considerable discussion as to whether the increased SOC in soil under zero tillage, especially near the surface, might increase N_2O emissions, because : (i) increased organic matter content can increase N_2O release[68], either because of increased energy supply to denitrifying organisms or because increased biological activity utilises oxygen in soil, thus possibly leading to anoxic conditions at some microsites and ii) reducing tillage can be associated in the short term with a less porous soil structure, conducive of anoxy[69] (Table 1). The different meta-analysis we compiled here (Fig. 2) suggest that N_2O emissions may offset the C storage in no-till system when both fluxes are compared in CO_2 equivalents. However, there is conflicting evidence on whether or not this risk is actually realised[68,70]. Recent meta-analyses suggest that, in the majority of situations, N_2O emissions are either unchanged or slightly decreased under zero or reduced tillage; the result will certainly be influenced by soil type and local climate and weather conditions so it may not be possible to draw a conclusion that is universally valid[70,71]. Furthermore, in some studies, N_2O emissions were expressed on both an area basis and a yield-scaled basis[70]; because crop yields were slightly decreased under reduced tillage in some environments, N_2O emissions per unit of grain (or other product) were sometimes increased compared to conventional tillage.

3.3 Erosion control - terracing

Erosion control practices are able to maintain or increase SOC content at the plot scale[72], although on a larger scale whether erosion is net C sink or a net source is still debated[73–75]. Erosion control encompasses a wide range of practices such as protecting the soil surface with cover crops or unharvested biomass (pruned fronds and other plant residues), agroforestry, crop rotations, conservation tillage, or terracing on steep slopes. Some of these practices are already addressed in other sections of this paper (3.2, 3.4 and 3.5), and the following focuses on terracing.

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

The N_2O emissions associated with terracing are still poorly known. Terracing decreases the aggregate breakdown and transport of soil by erosion, which would lead to reduced N_2O emissions. However, as stated previously, N_2O emissions may increase with increased SOC. In addition, terracing tends to change the soil C:N ratio[80] and this may change the N availability for nitrifying/denitrifying bacteria and thus affect N_2O emissions. Finally, to fully estimate the effect of erosion control on N_2O budgets, it is important to measure emissions at the catchment scale not only at the field scale. Since erosion control aims to avoid lateral losses of soil material (containing various forms of N) ending up in rivers or in floodplains, it is necessary to combine measurements in the terraced or unterraced fields with measurements and modelling on the fate of eroded N in floodplains and rivers.

3.4 Cover crops

Planting cover crops is an effective management practice to increase SOC content. According to a recent meta-analysis, it leads to SOC accumulation rates in the order of 1.18 t CO_2-eq. ha^{-1} yr^{-1} over 50 years[83] in the topsoil, with a positive effect independent of tillage method, climatic zone or plant type (leguminous vs non-leguminous). The impact of cover crops on SOC will depend on their duration and the frequency with which they are included in a crop rotation, and this information is sometimes unclear in published reports of field trials[83]. The main driver of SOC storage seems to be the extra C input, as suggested by the high correlation between rates of SOC stock change and the amounts of C returned to the soil by cover crop biomass[84]. However, there are limitations to the use of cover crops depending on cropping systems and climate conditions. For example, in temperate regions they can be readily utilised during the winter period prior to sowing a spring-sown crop, when the soil would otherwise be bare. But if only autumn-sown crops are grown, there is very limited time between harvesting and the sowing of the subsequent crop.

The effects of cover crops on N_2O emissions are more variable and contrasted than those on SOC changes (Table 1). Many factors influence the magnitude of N_2O emissions, such as the C:N ratio of cover crop residues, their rate of decomposition, the extra inputs of fertilizer N sometimes applied to cover crops, whether the residues are ploughed or left to decay on the soil surface. Current evidence points to a negative relationship between N_2O emissions and the C:N ratio of residues[50]. A low C:N ratio will increase the availability of soil N for microbial transformations (e.g. nitrification and denitrification) whereas larger ratios will result in N immobilization and deplete the soil inorganic N pool [85]. Additional C inputs from cover crops may stimulate the activity of denitrifier bacteria, which use these organic compounds as a source of energy[86]. A meta-analysis[87] reported a significant increase in N_2O emission when leguminous cover crops were introduced. However, another review[88] found out that the incorporation of either legume and non-legume cover crops tended to increase N_2O emissions but the magnitude of the effect was not significant due to the high variability of data. The effect of cover crops on N_2O emissions is therefore not yet fully understood and may well be highly site-specific.

One of the key points controlling cover crop effect on N_2O emissions is how often leguminous crops are integrated within the crop rotation. Leguminous cover crops generally have a lower C:N ratio than non-leguminous crops, and can fix substantial amounts of atmospheric N, reaching up to 0.1-0.2 t N ha^{-1} yr^{-1}[89] . These rates may lead to a N surplus if all the leguminous cover crop biomass is incorporated. A recent study, using a biogeochemistry model framework at European scale, estimated that systematic planting of N-fixing cover crops 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 crop[28]. In this scenario, the cumulative climate change mitigation effect of SOC sequestration was, on average, totally offset after 50 years since the adoption of cover crops, due to enhanced N_2O emissions. While cover crops may induce higher N_2O emissions, in particular if leguminous crop are extensively used, they can also reduce nitrate leaching, by about 56% on average[90]. This is beneficial for water quality and would be expected to lead to decreased indirect N_2O emission through denitrification of nitrate entering surface water. Finally, another indirect effect of leguminous cover crops on N_2O emissions will strongly depend on whether or not mineral N fertilisation rates are reduced to take account of N provided by biological fixation. The meta-analysis we compiled here indicate that additional N_2O emissions decrease the SOC storage benefit of cover crops, but do not fully offset it (Fig. 2).

3.5 Agroforestry

Agroforestry systems include a diversity of practices ranging from complex associations found in homegardens, multistrata systems or agroforests to simpler systems such as alley crops, silvopastoral systems, riparian plantings, shelterbelts, windbreaks or hedgerows[91]. Despite this broad diversity, recent reviews and meta-analyses consistently suggest that the conversion of arable land to agroforestry systems increases SOC stocks[92–94]. In temperate regions, SOC accumulation rates are usually around 0.92 t CO_2-eq. ha^{-1} yr^{-1} in the topsoil (0-30 cm)[83]. They are highly dependent on local pedoclimatic conditions and on the type and design of agroforestry systems (tree density, tree species, pruning management, etc), but rarely exceed 3.67 t CO_2-eq. ha^{-1} yr^{-1} [22,26]. However, the spatial distribution of SOC stocks in agroforestry systems is usually very heterogeneous, with higher stocks under the tree canopy or along tree rows[95,97]. Several mechanisms contribute to explain SOC sequestration in agroforestry systems. The main one is probably being linked to higher organic inputs to the soil compared to treeless agricultural land[98], including litterfall, pruning residues, and root inputs[99].

A recent synthesis of N_2O emissions under agroforestry compared to adjacent agricultural lands only found minor differences in net emissions, with no clear overall direction of change[93]. However, several authors found increased N_2O emissions in agroforestry, related to a greater N supply through N_2-fixing trees[100–103] or to the incorporation of tree residues[104,105]. By contrast, N_2O emissions are often reduced in silvo-arable systems and in riparian buffers[93]. Some authors suggest that concerns over N_2O emissions from N_2-fixing trees are unwarranted since fluxes from soils planted with N_2-fixing trees are similar to those fertilized with mineral N[106]. Furthermore, the yield of crops in tropical agroforestry systems may be boosted as a result of higher N-inputs from trees. In temperate regions where agroforestry systems are generally planted with non-legume trees, N_2O emissions are often reduced[93], with several processes contributing to the trend. Increased nitrogen utilization at the plot scale may be due to the presence of deep-rooted trees[107], which are capable of taking up nitrate-N that has leached below crop rooting depth [108–110]. This process can potentially reduce the amount of N available for nitrification and denitrification, and thus reduce indirect N_2O emissions. Soil water content is often lower in agroforestry than in treeless plots[111], due to a higher daily water consumption by trees and crops[112]. A drier soil profile in agroforestry systems could therefore lower N_2O emissions. In temperate silvoarable systems, tree rows are usually uncropped and unfertilized. This reduction in the fertilized cropping area indirectly leads to lower N_2O emissions per hectare. An obvious consequence of agroforestry, especially as tends to be practiced in temperate regions, is that a smaller area of land is devoted to the agricultural crop being grown. So the impact of decreased N_2O emissions may be different if expressed on an area basis compared to per unit of production.

3.6 Non-pyrogenic organic amendments

A literature review[113] reported increases in SOC (sometimes expressed as stocks and sometimes as concentration) after prolonged large applications of organic amendments under several different agro-climatic conditions. These increases ranged from 20 to 90% of the initial total SOC after few years (3-60 years), compared to unfertilized controls or treatments receiving only synthetic mineral N fertilisers, with most being in the range 20- 45%. A meta-analysis[114] based on 130 observations worldwide quantified the response of SOC stocks to manure application over periods ranging from 3 to 82 years. The mean manure-C retention coefficient defined as the average proportion of manure-C remaining in the soil was estimated at 12% for an average study duration of 18 years. The authors finally estimated a relative SOC stock change factor of 26% which was also related to cumulative manure inputs. Concerning Mediterranean cropping systems, and shorter durations, a meta-analysis[115] reported that the application of organic amendments increased SOC stocks by 23.5% with an average SOC storage rate of 4.81 t CO_2-eq. ha^{-1} yr^{-1} calculated for an average duration of 7.9 years. From these meta-analyses, it seems that there is a consensus that organic amendments lead, on average, to a relative increase of 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' target). In one example[17] where manure was applied annually at a high rate compared to what is usual in agrosystems, the annual rate of SOC accumulation averaged 18‰ per year in the first 20 years, then declined to 6‰ per year after 40-60 years, and to only 2‰ per year after 80-100 years. However, from the perspective of mitigating climate change, it is arguable whether any increase in SOC stocks resulting from applications of manure or similar materials can be considered as C mitigation in the sense of either a transfer of C from atmosphere to land or an avoided emission. Manure is generated in agricultural systems and is almost always used in some way by application to soils, though often quite inefficiently. Thus, an increase in SOC stocks at a given location mainly represents a transfer of C from one site to another as opposed to a net removal of atmospheric carbon [17]. Local additional SOC storage may not represent a CO_2 sink, i.e. a net transfer of carbon from the atmosphere to the soil at the landscape scale.

Because organic amendments such as manures contain readily-decomposable N-rich compounds, there is a significant risk that they may enhance N_2O emissions[116–118]. Conversely, their use permits decreased use of mineral N fertilizers, thereby saving N_2O emission from this source and fossil energy and the associated GHG emissions from fertilizer manufacture. A further complicating factor in assessing the overall impact of manure use is that indirect emissions due to storage or management are not negligible [119]. There are few reports in the literature of long-term monitoring of N_2O emissions compared to data on SOC stock changes, primarily because the former are much more difficult to measure. However, the effects of multiple types of organic amendments on SOC storage and N_2O emissions have been evaluated in short-term experiments for various soil types, climates, soil incorporation practices, and amendments types including crop residues, manure, composts of various origin and maturation stages, and sewage sludge. A meta-analysis[120] concluded that the N_2O emission factors (EFs) related to N inputs were mainly controlled by the C:N ratios of the added material, but that many other factors influenced emission, such as soil properties (texture, drainage, SOC and N content), and climatic factors. For instance, the authors observed that the EFs were on average 2.8 times greater in fine-textured soils compared to coarse-textured, consistent with a previous meta-analysis [50]. However, we should mention that the value of meta-analyses is often limited due numerous controlling factors that are not always correctly reported in the papers reviewed, and the general paucity of organic amendments characterisation in the literature. For instance, the two meta-analyses mentioned in this paragraph only involved 28 to 38 individual journal articles.[50,120]. Another approach is to compare organically managed soils with those managed without organic amendments [121]. Results from such a comparison seem to indicate reduced N_2O emissions compared to situations relying totally on mineral fertilizers, as show in Fig. 2. It should be noted that there is limited data from long-term studies on N_2O emissions associated with additions of organic amendments; the data covers only a limited diversity of pedo-climatic conditions, and especially the range of soil water filled pore space values explored.

3.7 Biochar amendments

Biochar (pyrolyzed organic matter amended to the soil) technology is considered by some authors to be one of the methods with the highest potential to sequester carbon in soils compared to natural C cycle without biochar production step[4]. The aim of biochar production from biomass pyrolysis is to produce recalcitrant organic matter (i.e., charcoal and biomass-derived black C) which is then added to the soil. For this reason, biochar can be considered as a negative emission technology different from other soil C sequestration methods[5].

Biochar properties and effects on SOC stabilization strongly depend on the feedstock material and pyrolysis conditions (e.g., maximum temperature, heating rates)[122–126], as well as biochar ageing and soil properties[127,128].The efficiency of biochar for C sequestration is two-fold as compared to simply relying on soil stabilization processes. First, slow pyrolysis for biochar production results in a much higher proportion of the feedstock C bound in persistent molecular structures than through in situ stabilization by addition of unprocessed organic matter to soil [129]. With a slow pyrolysis at about 500°C, approximately 50% of the carbon contained in a feedstock of Miscanthus or maize cobs ended up within the biochar and can therefore be assumed to be more stable than carbon in the raw biomass[130]. This compares with only 8-12% of straw residue returned to the field being transformed into longer-lived SOM forms[27,131].Thus, pyrolysis is about four times more efficient than SOM-formation processes to produce persistent C in soils. Second, field studies show that biochar has a longer mean residence time in soils than SOM, i.e. >100 years[132] vs. about 50 years for the latter[133]. Combining effects of the higher persistent-C yield with that of the longer mean residence time, biochar appears at least 8 times more efficient at storing SOC than the return of non-pyrolysed residues. In meta-analyses, biochar amendment tends to increase the soil organic carbon stocks by 40% but the studies used were generally short term (no more than 4 years)[134]. Nevertheless, this result must be considered with due care since it is not straightforward to measure the effect of biochar, which is mainly C, on native SOC but one published study suggests that biochar amendment increase total SOC including non-biochar C[135]. In addition to the direct inputs of pyrolyzed biomass to the soil, recent studies showed that biochar amendments could increase (positive priming), decrease (negative priming) or have no effect on the mineralisation of native SOM. The biochar effect on the magnitude and direction of priming is influenced by the incubation period and pyrolysis temperature[122,123,136]. Positive priming, which would cause destabilization of SOM, thus offseting part of the increased SOC storage, could result from the biochar affecting microbial biomass activity and enzyme production[137] through changes in availability of organic substrates and nutrients, and modification of microorganism habitat associated with the great porosity and large specific surface area of charcoal particles[137]. Conversely, some studies showed that biochar-induced negative priming, leading to further SOC storage in addition to direct biochar-C inputs, resulted from the enhancement of organo-mineral interactions and soil aggregation with biochar[138,139], and a greater adsorption of dissolved organic carbon onto biochar particles[140]. In addition, biochar amendments have been shown to increase soil water holding capacity, the availability of some nutrients (Ca^{2+}, Mg^{2+} in particular) and to increase soil pH. All of these mechanisms could further enhance crop productivity and biomass inputs into soil[141–144], with clearer effects on crop yields in highly weathered tropical soils[145]. Nevertheless, to process biomass into biochar, transport it and incorporate it to the soil, some energy is needed (possibly produced by pyrolysis) and the related GHG emissions associated with this process must be accounted for to calculate a full GHG balance for biochar. Life cycle assessments (LCA) have shown that a positive balance can be obtained, illustrated by GHG reductions up to 2.74 t CO_2 equivalent per ton of biochar amended on volcanic soils from Southern Chile[146]. The balance can also significantly increase when plant biomass production is accompanied by an efficient use of the bioenergy produced during the pyrolysis process in order to maximize climate benefits from biochar production followed by addition to soil, as shown by an LCA performed in Spain[147] ; implementation costs were also decreased. One simulation study suggests that the maximum sustainable technical potential of biochar to mitigate climate change, involving the widespread use of biochars, without threatening food security and landscapes, could be a mitigation of 12% of current anthropogenic CO_2 emissions (1.8 Pg CO_2 -C equivalent per year)[148]. The C:N ratio of SOM approximates 14[149] while that of biochar is generally higher than that of its feedstock, i.e. generally > 50 for straw biochar and > 100 for wood biochar . It takes therefore at least five times less N to stabilize organic C in the form of biochar than in the form of SOM. Beyond this critical observation, biochar has other important interactions with the N cycle, notably: 1) volatilization and immobilization of N during the pyrolysis process[150], 2) reduction of N_2O emissions after application to arable fields[151], 3) reduction of NO_3 and NH_4 leaching[152]. Emissions of N_2O from soils are in most cases substantially reduced by biochar addition: a recent meta-analysis reported an average decrease of 38% across studies[151]. However, most measurements are faily short-term, the majority in this meta-analysis being <30 days . This effect appears consistent when biochar is produced at over ~450C, so that the product is both high pH and high surface area while containing very little labile C [125]. The contribution of N_2O emissions attenuation with biochar was shown to be negligible in the LCA performed in Southern Chile, compared to the climate change mitigation associated to C storage[146]. Furthermore, reductions in N_2O emission with biochar appears only significant for the first year after application, which suggests that frequent applications are necessary to maintain such an effect. In view of the large quantities of biochar usually applied in such studies, this may greatly limit the practical and/or economical potential for using biochar as a method for decreasing N_2O emissions.

A way to capitalize on the positive interactions between biochar properties and the N cycle is through the development of biochar-based fertilizers made by mixing biochar with mineral or organic sources of nutrients[153]. This method aims at reducing nitrate losses and N_2O emissions, and at increasing N use efficiency. Moreover, biochar tends to adsorb mineral N and the mixing with a nutrient-rich material prevents potential N deficiency created by field application of large amounts of raw biochar. Some biochar structures have been successfully loaded with nitrate ions through co-composting, which could greatly increases the fertilisation value of the product[154]. Producing biochar fertilizers requires the development of appropriate technologies. For example, mixing raw biochar - a high pH product- with manure and slurries can result in large amounts of NH_3 being volatilized. Therefore, biochar acidification is generally required when making biochar fertilizers from organic feedstocks. However, biochar is also a strong sorbent for NH_3[155], which may be captured from the atmosphere during the pyrolysis process and made available to plants later. This is a promising technology to abate anthropogenic emissions of NH_3[155] as well as directly reduce NH_3 volatilization from soils[150]. In conclusion, pending proper technology, biochar may be intimately mixed with N sources and applied each year as a fertilizer to maximize reductions in both N_2O emissions and nitrate leaching, while sequestering C in a structure requiring little N. However, further studies are needed to validate the scant results currently available.

3.8 Overview of the current evidence

Fig. 2 summarises published data on rates of change in SOC and rates of emission of N_2O resulting from four prominent sets of practices designed to increase SOC, namely agroforestry, cover crops, no-tillage and organic amendment. All fluxes are expressed in CO_2 equivalents, using a global warming potential value integrated over 100 years and including global warming potential (GWP) of 298 for N_2O as recommend by last IPCC report[156]. GWP is the time-integrated radiative forcing induced by a pulse emission of a given component (here N_2O), relative to a pulse emission of an equal mass of CO_2[156]. The data on Fig. 2 were taken from several meta-analyses and review papers. Here, we did not re-analyse the data gathered by such meta-analysis, but rather presented the mean effect size from each study converted in CO_2 equivalents. When results were given for the whole experiment duration, we divided by the duration of the experiment to obtain the mean annual SOC storage/ N_2O emissions (see supplementary information for detailed methods). The data in Fig. 2 is based on over 700 measurements of SOC change and 200 measurements of N_2O. Even allowing for some papers being cited in more than one meta-analysis, this is a large body of data and, to our knowledge, has not previously been assembled in this way. At first sight it appears that SOC increases produced by the four sets of treatments varied widely from -0.52 ±0.46 to -6.74±1.21 t CO_2-eq. ha^{-1} yr^{-1}, the negative sign representing accumulation of SOC, i.e. transfer of C from atmosphere to soil. However, this wide range is somewhat deceptive as the two largest values are from very specific situations. The value of -6.74±1.21 t CO_2-eq. ha^{-1} yr^{-1} for agroforestry (Kim *et al*, 2016) is from 34 sets of data for systems with a particularly high tree density (see legend to Fig. 2); the other two meta-analyses for agroforestry, based on >200 datasets, give values of less than half this at around -3 t CO_2-eq. ha^{-1} yr^{-1}. For cover crops the majority of data, based on 186 datasets, lead to mean rates of C accumulation in the range of -1.2 to -2.0 t CO_2-eq. ha^{-1} yr^{-1}. The highest value observed for cover crop was 3.67 t CO_2-eq. ha^{-1} yr^{-1} but it specifically refers to cover crops included between the wide rows of Mediterranean woody crops (olive, almond and vineyards) where the soil would otherwise be bare[157]. This is clearly an important management change within this environment but represents a much greater input of plant material than, say, cover crops grown during winter within temperate arable cropping systems. For both agroforestry and cover crops, and even ignoring the two sets of data for SOC increases that are especially large, it appears from the summarized data in Fig. 2 that SOC increases resulting from these two changes in management considerably outweigh increased N_2O emissions when both are expressed on a CO_2-eq basis and these management changes can be expected to beneficial for climate change mitigation. For no-tillage the situation is different: the relatively small rates of SOC accumulation are approximately equal to the increases in N_2O emission when both are expressed on a CO_2-eq basis, so there appears to be no overall climate change benefit. For organic amendments the results indicate that N_2O emissions are decreased and thus reinforce the SOC benefit, though this is based on a very limited amount of data and also, as discussed earlier, it is questionable whether SOC increases from addition of organic amendments can be fully regarded as climate change mitigation. It should be noted that most of the studies are performed over a few years and assessment of GHG balance in the long term, especially for N_2O, are still missing[28]. Some practices were too little documented or with not enough information to be compared with the others (e.g. , biochar or erosion control).

4. DISCUSSION AND OUTLOOK

Overall, the meta-analysis we gathered here (Fig. 2) suggest that, with the exception of reduced tillage practices, increased N_2O emissions are not sufficient to invalidate the GHG abatement potential achieved by SOC sequestration strategies. Some sequestration strategies (e.g, biochar or non-pyrogenic organic amendment application) may even generate win-win situations through a decrease in N_2O emissions, although the experimental evidences are still scant. In addition, the economic impacts and large-scale effects of the options examined here warrant further assessment. Some practices may affect crop yields or farmers' income, depending on pedo-climatic conditions and the details and practicalities of the cropping systems. For instance, conservation practices, and especially no-till may slightly decrease crop yields under temperate climates but be beneficial in drier conditions[158]. Similarly, the yield of arable crops is usually reduced in agroforestry systems in temperate regions[159] but in more arid climates, crops perform better[160]. Nevertheless, for agroforestry, trees produce timber, firewood, honey, fruits, etc. that are also a source of incomes for the farmers and may lead to greater overall sustainability. Beyond yield impacts, some socio-cultural or economic factors come into play that may hinder the adoption of carbon sequestration practices. For example, in the United States, the cost of carbon capture through Natural Resources Conservation Service programs is estimated at US \$ 32-442 per tonne of CO_2, with an average of US \$ 183[161]. A carbon price much higher than the present value (around US \$ 10 as a global average[162]) would be necessary to promote carbon sequestration practices, as well as a regulation to direct the financial flow of industrial and energy emitters to the agricultural sector.

To be deployed at large scale and to enter emission trading systems, the GHG fluxes of each change in agricultural practices should be estimated precisely. Various models may be used to account for scale or leakage effects such as indirect land-use changes [163]. The methods currently available include data-driven approaches based on worldwide measurement networks[164], statistical or empirical flux-upscaling models[57,164], process-based models and, lastly, integrated assessment models (IAM)[165]. Process-based models include a representation of N cycling processes, which are an essential tool in assessing and predicting the terrestrial N cycle and N_2O fluxes in response to multi-factor global changes. Such models have been used to estimate N_2O emissions from natural and agricultural soils at various scales, from field to global level via the integration of a prognostic N cycle into different land surface models[166]. As an example, Fig. 3 shows the results of simulations by various models at global scale. Most of N_2O emissions from cropland are due to the use of mineral fertilizers (Fig. 3a) and are mainly located in USA, Europe, India and China. They may be used to quantify carbon sequestration in soil minus the N_2O emission trade-off at global scale, based on ensemble runs as was initiated in the global N_2O Model Inter-Comparison Project[167].

IAMs focus on the interactions between the economic activities and earth system responses and are vital for estimating what socioeconomic changes would be needed to reduce GHG emissions across sectors and increase biospheric C sinks[168]. Until recently, most IAMs did not explicitly take into account SOC restoration practices[169]. A recent study that did include them found that soils could be a sink of 3.5 $GtCO_2$-eq/yr by 2050 under a carbon price of 190 USD/tCO_2[13]. This carbon mitigation option, if achievable in practice, would reduce the burden of climate stabilization for all sectors of the economy, including agriculture. In addition, SOC increases are often correlated with higher crop yields and contribute to a range of other environmental benefits and increased sustainability of agricultural systems. Practices designed to increase SOC can offer a win-win solution *vis a vis* food security, by mitigating food calorie losses resulting from the application of emission reduction targets (e.g. through decreased applications of mineral fertilizers) and reducing undernourishment.

Finally, many of the practices reviewed (Table 1) here may be combined on a given field: for example, no-till can be combined with cover crops, organic amendments, or agroforestry. Such combinations have been little tested in practice and in particular synergetic effects between them have not been evaluated in depth[170,171]. Conversely, they may come with trade-off, antagonistic or synergistic effects regarding SOC storage rates, as well as N_2O emissions or other impacts and these needs to be identified and quantified. Furthermore, proper assessment of carbon sequestration measures raises classical GHG accounting issues, such as double counting, improper setting of system boundaries and counterfactual scenarios[169]. Although further research is still needed to quantify the potential of SOC sequestration options on a local to regional basis, it appears that their potential to mitigate climate change, even when factoring in N_2O emissions is still significant and that they deserve further consideration in climate stabilization scenarios. Including the state-of-the-art knowledge reviewed here on the effectiveness of such measures in land system or integrated assessment models could be a prime target to assess 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 N_2O 564 emissions.

Management Practice	**Effect on soil C stocks**	**Effect on N_2O emissions (*a priori*)**
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)

Figure legends

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

Figure 2: Estimation of the SOC storage and N_2O emissions of land-based mitigation options expressed in CO_2 equivalents. Negative values indicate a net reduction in GHG emissions in terms of CO_2 equivalents, while positive values show a net increase of CO_2 equivalent emissions. All values refer to the difference between the land-based mitigation option in question and a "control" land (e.g. no-tillage vs conventional tillage). For agroforestry, the control land is cropland and different types of agroforestry systems were considered. NB: In Kim et al. (2016) the majority of soil C storage data comes from intercropping, improved fallows and rotational woodlots, which are systems with high tree density. This could partially explain the very high estimation of soil C storage found in Kim et al. (2016) compared to other papers. Organic amendments do not include biochar. The control used for comparison with organic amendments is an experiment managed with inorganic fertilizers. For cover crops meta-analysis, Vicente-Vicente et al. (2016) only consider Mediterranean woody crops (olive, almond and vineyards), which could also explain the large soil C rates estimated. Uncertainty is given as standard error (SE) for every paper. If it was provided as a confidence interval (CI) or standard deviation (SD) it has been adequately transformed to unify the units. (*Reviews; ** For these meta-analysis the values reported in the graph have been recalculated as the weighted mean across all experiments, from the database provided by the authors, because the values coming from the papers could not be used as they were reported as a percentage only).[121,157,172,173]

Figure 3: Spatial and latitudinal patterns of contributions of fertilizer (a) and manure (b) on cropland soil N_2O emissions obtained during the global N_2O Model Intercomparison Project[167]. Average over the 2006-2015 period.

[Box 1: Intertwined soil C and N cycles. Primary producers fix atmospheric CO_2 through photosynthesis and produce biomass that subsequently enters the soil *via* rhizodeposition and litter production or *via* organic amendments or deposition due to erosion or dissolved organic carbon (DOC) leaching into the subsoil. Organic C entering soil is further processed by soil organisms into organic by-products and subsequent CO_2 losses during mineralization. SOC can also be lost through fire, or displaced by erosion/deposition and lateral transfer of DOC. The net difference between C inputs and outputs determines how much organic C is stored in the soil. A part of this carbon is stabilized for decades to centuries through several mechanisms such as interactions with the soil mineral matrix, chemical recalcitrance or protection within aggregates. Nitrogen can enter the soil *via* atmospheric deposition or biological N_2 fixation, or as mineral or organic fertilizers. Nitrogen can leave the soil through plant uptake, leaching or gaseous emissions. The critical N pool sustaining plant growth is mineral N (ammonium (NH_4^+) and nitrate (NO_3^-)), which either originates from the mineralization of soil organic matter and ammonification as well as from mineral fertilizers inputs. Specific chemotrophic microorganisms called nitrifiers can oxidize ammonium, first into nitrite (NO_2^-) and then into nitrate. The reverse process called dissimilatory nitrate reduction can occur in anaerobic conditions[174]. Heterotrophic denitrifying communities can also use nitrate and reduce it to N_2. In each of these processes, nitrous oxide (N_2O) can be generated[175]. Net N_2O emissions from the soil will usually be lower when the amount of soil mineral N is low 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 NH_3, or as other gaseous forms of N oxides that can be deposited and contribute to indirect N_2O formation.]

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Figure

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Box

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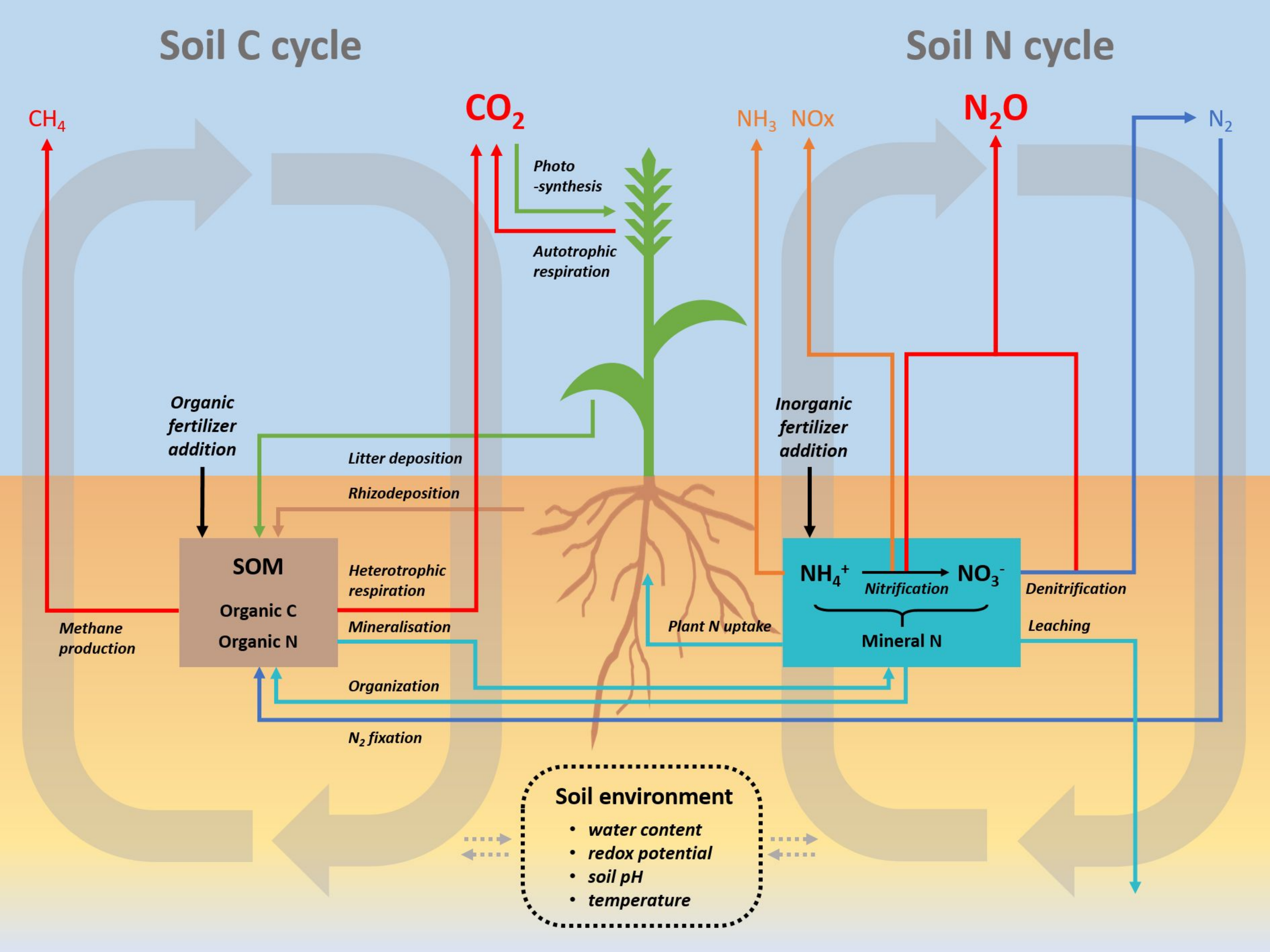
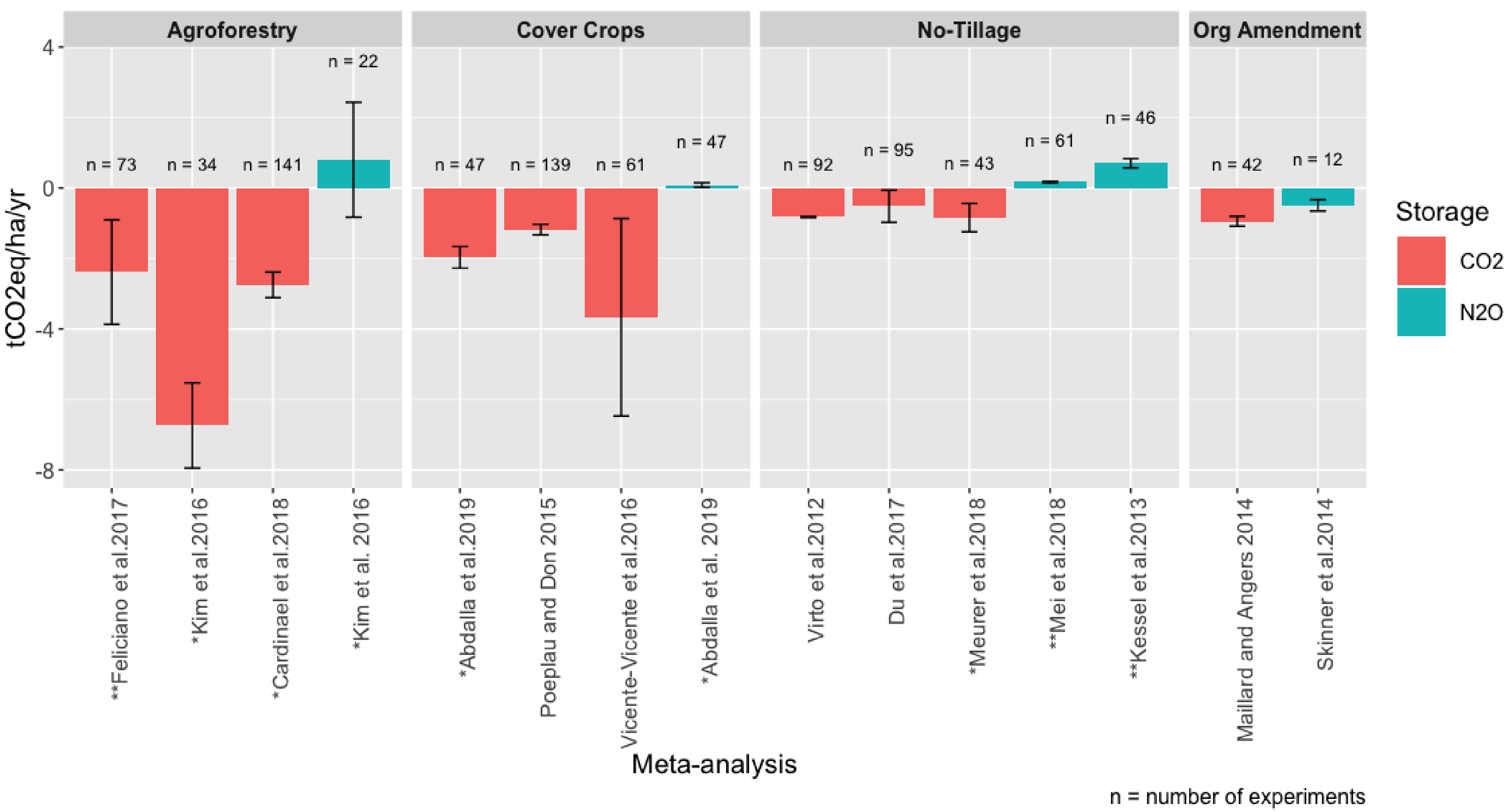
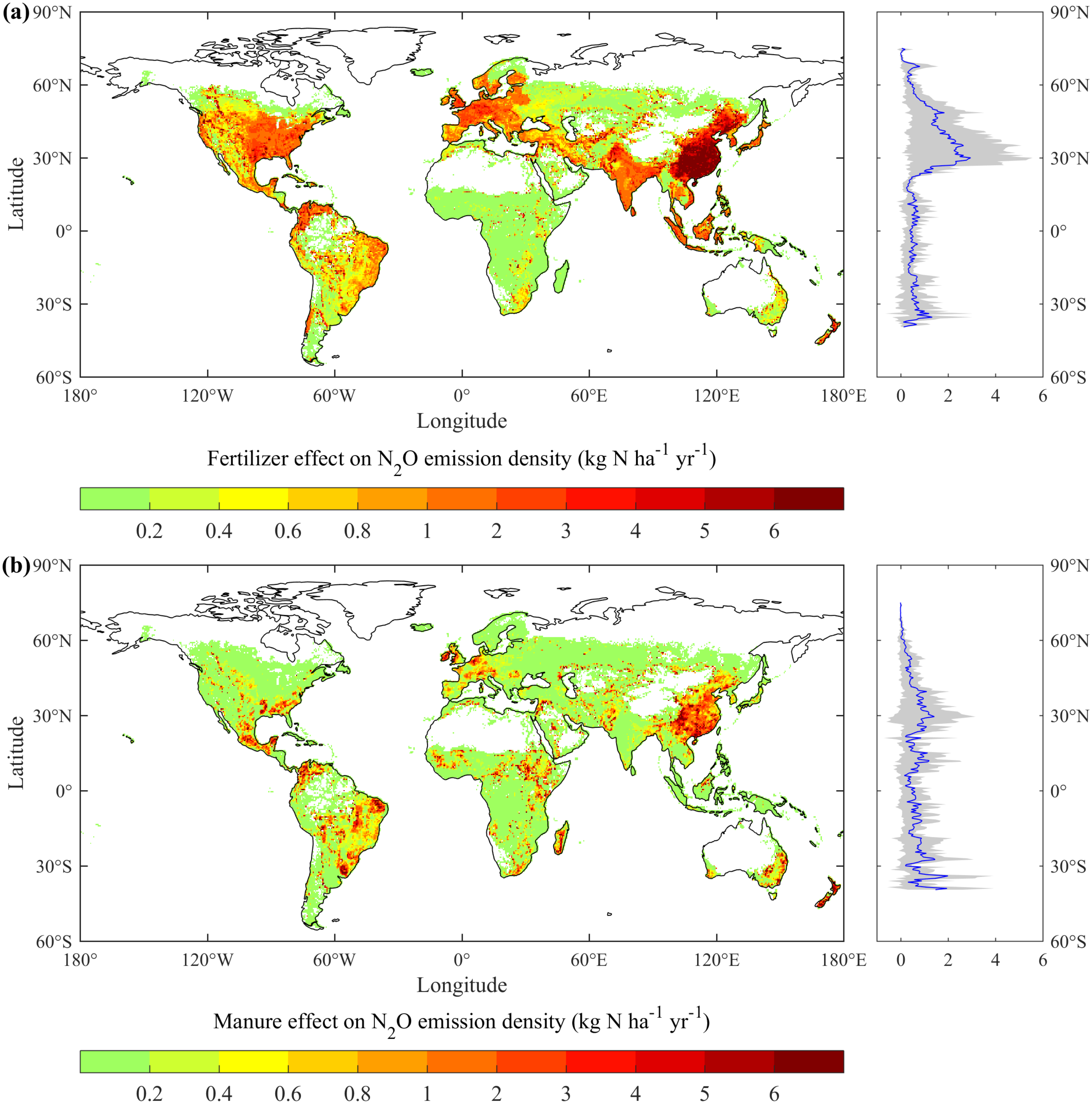


Soil C cycle
Soil N cycle
CH_4
CO_2
NH_3
NOx
N_2O
N_2
Photo
-synthesis
Autotrophic
respiration
Organic
fertilizer
addition
Litter deposition
Rhizodeposition
Inorganic
fertilizer
addition
SOM
Organic C
Organic N
Heterotrophic
respiration
Methane
production
Mineralisation
Organization
N_2 fixation
Plant N uptake
NH_4^+
Nitrification
NO_3^-
Mineral N
Denitrification
Leaching
Soil environment
water content
redox potential
soil pH
temperature



Agroforestry
Cover Crops
No-Tillage
Org Amendment
4
0
-4
-8
tCO2eq/ha/yr
n = 73
n = 34
n = 141
n = 22
n = 47
n = 139
n = 61
n = 47
n = 92
n = 95
n = 43
n = 61
n = 46
n = 42
n = 12
**Feliciano et al.2017
*Kim et al.2016
*Cardinael et al.2018
*Kim et al. 2016
*Abdalla et al.2019
Poeplau and Don 2015
Vicente-Vicente et al.2016
*Abdalla et al. 2019
Virto et al.2012
Du et al.2017
*Meurer et al.2018
**Mei et al.2018
**Kessel et al.2013
Maillard and Angers 2014
Skinner et al.2014
Meta-analysis
Storage
CO2
N2O
n = number of experiments


(a)
90°N
60°N
30°N
0°
30°S
60°S
Latitude
180°
120°W
60°W
0°
60°E
120°E
180°E
Longitude
0
2
4
6
Fertilizer effect on N_2O emission density (kg N ha^{-1} yr^{-1})
0.2
0.4
0.6
0.8
1
2
3
4
5
6
(b)
90°N
60°N
30°N
0°
30°S
60°S
Latitude
180°
120°W
60°W
0°
60°E
120°E
180°E
Longitude
0
2
4
6
Manure effect on N_2O emission density (kg N ha^{-1} yr^{-1})
0.2
0.4
0.6
0.8
1
2
3
4
5
6