Exhibit F

Soil Biology and Biochemistry 143 (2020) 107735

Contents lists available at ScienceDirect

Soil Biology and Biochemistry

journal homepage: http://www.elsevier.com/locate/soilbio

A meta-analysis of global cropland soil carbon changes due to cover cropping

Jinshi Jian[a,b], Xuan Du[c], Mark S. Reiter[a,d], Ryan D. Stewart[a,*]

[a] School of Plant and Environmental Sciences, Virginia Tech, Blacksburg, VA, USA
[b] Pacific Northwest National Laboratory-University of Maryland Joint Global Change Research Institute, 5825 University Research Court, Suite, 3500, College Park, MD, USA
[c] Department of Hydraulic Engineering, Yangling Vocational & Technical College, Yang Ling, Shaanxi, China
[d] Eastern Shore Agricultural Research and Extension Center, Virginia Tech, Painter, VA, USA

ARTICLE INFO

Keywords:
Cover crop
Soil organic carbon
Soil quality
Soil health
Meta-analysis
Database

ABSTRACT

Including cover crops within agricultural rotations may increase soil organic carbon (SOC). However, contradictory findings generated by on-site experiments make it necessary to perform a comprehensive assessment of interactions between cover crops, environmental and management factors, and changes in SOC. In this study, we collected data from studies that compared agricultural production with and without cover crops, and then analyzed those data using meta-analysis and regression. Our results showed that including cover crops into rotations significantly increased SOC, with an overall mean change of 15.5% (95% confidence interval of 13.8%–17.3%). Whereas medium-textured soils had highest SOC stocks (overall means of 39 Mg ha^{-1} with and 37 Mg ha^{-1} without cover crops), fine-textured soils showed the greatest increase in SOC after the inclusion of cover crops (mean change of 39.5%). Coarse-textured (11.4%) and medium-textured soils (10.3%) had comparatively smaller changes in SOC, while soils in temperate climates had greater changes (18.7%) than those in tropical climates (7.2%). Cover crop mixtures resulted in greater increases in SOC compared to mono-species cover crops, and using legumes caused greater SOC increases than grass species. Cover crop biomass positively affected SOC changes while carbon:nitrogen ratio of cover crop biomass was negatively correlated with SOC changes. Cover cropping was associated with significant SOC increases in shallow soils ($\leq$30 cm), but not in subsurface soils ($>$30 cm). The regression analysis revealed that SOC changes from cover cropping correlated with improvements in soil quality, specifically decreased runoff and erosion and increased mineralizable carbon, mineralizable nitrogen, and soil nitrogen. Soil carbon change was also affected by annual temperature, number of years after start of cover crop usage, latitude, and initial SOC concentrations. Finally, the mean rate of carbon sequestration from cover cropping across all studies was 0.56 Mg ha^{-1} yr^{-1}. If 15% of current global cropland were to adopt cover crops, this value would translate to 0.16 $\pm$ 0.06 Pg of carbon sequestered per year, which is ~1–2% of current fossil fuels emissions. Altogether, these results indicated that the inclusion of cover crops into agricultural rotations can enhance soil carbon concentrations, improve many soil quality parameters, and serve as a potential sink for atmosphere CO_2.

1. Introduction

Many woodlands and grasslands are being converted to cropland due to increasing world population and food production requirements. Between 1850 and 1980, at least 900 million hectare (Mha) of naturally vegetated lands were converted to croplands and pastures across the globe (Houghton, 1995), with the conversion process continuing today in many parts of the world (McGuire et al., 2001). Converting lands from natural vegetation to cropland leads to soil organic carbon (SOC) losses. For example, a large-scale study in Germany (Wiesmeier et al., 2013a,

Abbreviations: ha, hectare; Mha, million hectares; SD, Standard deviation; RR, Response ratio; CC(s), Cover crop(s); NC, No cover crops; BD, Bulk density [M L^{-3}]; SOC_{stock}, Soil organic carbon stock [M L^{-2}]; $SOC_{\%}$, Soil organic carbon concentration [M M^{-1}]; C_{rate}, Rate of change of soil organic carbon [M L^{-2} T^{-1}]; $C_{sequestration}$, Carbon sequestered in soil due to cover crop usage [M T^{-1}].

* Corresponding author.
E-mail address: ryan.stewart@vt.edu (R.D. Stewart).

https://doi.org/10.1016/j.soilbio.2020.107735
Received 14 September 2019; Received in revised form 17 January 2020; Accepted 25 January 2020
Available online 27 January 2020

2013b) showed that cropland soils stored lower amounts of SOC (90 Mg ha^{-1}) than either forest soils (98 Mg ha^{-1}) or grassland soils (118 Mg ha^{-1}). Based on a meta-analysis of 95 studies covering 322 sites with temperate climates, converting grassland to cropland led to mean SOC losses of 36% and converting from forest to cropland caused SOC to decrease by an average of 32% (Poeplau et al., 2011) Another meta-analysis conducted using 385 studies found that converting from primary forest to cropland yielded SOC losses of 25%–30% (Don et al., 2011).

Reversing carbon losses in cropland soils can be a means to sequestrate atmospheric carbon (Gattinger et al., 2012). Elevated SOC is also associated with improved soil health and fertility; therefore, increasing SOC may help to enhance agricultural productivity (Stewart et al., 2018; Van Eerd et al., 2014). Shifts in management practices, including the use of cover crops (CCs) within rotations, has been proposed as a way to increase SOC stocks (Kaye and Quemada, 2017). However, not all studies found CCs to result in SOC accumulation, with some demonstrating SOC losses after introduction of CCs (Bandick and Dick, 1999; Idowu et al., 2009; Ndiaye et al., 2000). Due to differences in climate and management, CCs may need to be used for decades in some systems to cause significant SOC increases (Poeplau and Don, 2015). Results also vary depending on soil texture and type, as some fine-textured soils can help to physically protect SOC from decomposition (Callesen et al., 2003; Krull et al., 2003), depending on factors such as mineralogy and amount of soil aggregation (Hassink and Whitmore, 1997; Schmidt et al., 2011).

Cover crop and rotation types can both affect SOC changes. For example, some studies found grass CCs, including cereal rye [*Secale cereal*] and annual ryegrass [*Lolium multiflorum*], cause greater SOC increases than leguminous CCs like cowpea [*Vigna unguiculate*] and hairy vetch [*Vicia villosa Roth*] (Mazzoncini et al., 2011; O'Dea et al., 2013). Other studies found greater SOC increases after legume CCs compared to grass CCs (Utomo et al., 1990), with mixtures often causing the greatest increases of all (Sainju et al., 2006). One reason for these incongruous results may result from differences in biomass and carbon:nitrogen (C:N) ratios between CC species. Legume CCs typically have low C:N ratios, due in part to their ability to fix atmospheric nitrogen, while grass CCs often have higher biomass but also higher C:N ratios. Crop rotations may also influence SOC accumulation from CCs, both by affecting CC planting and harvesting dates (thus varying the time for biomass accrual) and by altering soil properties such as nutrient availability, soil structure, and soil microbial properties (Bandick and Dick, 1999; Campbell et al., 1991; Sainju et al., 2006). However, it is not well understood whether and how cash crop rotations interact with CCs to alter SOC in agricultural soils.

To resolve uncertainties and discrepancies that can emerge from single-site studies, modern techniques such as meta-analysis have been used to compile and compare results from various investigations of CCs. For example, a meta-analysis for the Pampas region of Argentina showed that SOC significantly increased when CCs were grown in coarse- and fine-textured soils (Alvarez et al., 2017). Aguilera et al. (2013) found that practices combining external organic amendments with CCs caused significant increases in SOC concentrations. Blanco-Canqui et al. (2015) determined that introducing CCs resulted in an SOC increases of 0.1–1.0 Mg ha^{-1} yr^{-1}. While the rate of accumulation slowed through time, the results of that study suggested that more than a century may be needed for SOC concentrations to reach new equilibria. Despite generating such insights, however, most existing meta-analyses focused on a particular region and have not considered climatic influences and environmental factors in the response. These regional responses may therefore have limited applicability towards understanding SOC dynamics at global scales.

Meta-analysis has also not yet been used to comprehensively assess interactions between SOC and other soil properties (e.g., soil penetration resistance, soil nitrogen, soil microbial activity), even though individual studies identified different (and at times contradictory) parameter responses to CCs. For instance, CC introduction led to increases in soil aggregate stability (Marques et al., 2010; Stavi et al., 2012; Tesfahunegn et al., 2016) and water infiltration rates (Haruna et al., 2018), yet other studies demonstrated no effect or even decreased infiltration rates after introduction of CCs (Abdollahi and Munkholm, 2014; Steele et al., 2012). Likewise, some studies found that bulk density (BD) decreased after CC usage (Stavi et al., 2012; Spargo et al., 2008), while others found no effect (Blanco-Canqui et al., 2011, 2013; Jiang et al., 2007). To examine past and current practices that best quantify properties associated with soil health, Stewart et al. (2018) collected historical publications examining CC and no-till practices and integrated those data into a global soil health database called SoilHealthDB (Jian et al., 2019, 2020). That analysis determined that 13 out of 42 soil health indicators showed >10% difference from no cover crop (NC) controls in the first 1–3 years after CCs were introduced. Responsive parameters included soil aggregate stability, nitrogen mineralization rate, and microbial biomass nitrogen and carbon, whereas SOC did not show a consistent short-term response to CC. The question thus remains whether these changes in soil properties help to drive longer-term changes in SOC.

In this present study, we used the SoilHealthDB to further explore SOC dynamics after CCs. The study objectives were to: 1) quantify CC usage effects on SOC concentrations; 2) evaluate how climate type, CC species, and cash crop rotation affected SOC dynamics using meta-analysis; 3) identify possible mechanisms for SOC changes via correlation with other soil/agronomic variables; and 4) estimate carbon sequestration potential as CCs become applied to various extents within cropland across the globe.

2. Methods

2.1. Data collection

To gather data for analysis, we collected publications from three sources: (1) the "Research Landscape Tool" that compiled soil health-related publications and research projects into a searchable database (http://www.soilhealthinstituteresearch.org/); (2) cited papers from previous meta-analyses and review papers, specifically Poeplau and Don (2015), Alvarez et al. (2017), Sileshi (2009), and Gattinger et al. (2012); and (3) scholarship-focused search engines, including ISI Web of Science, Google Scholar, and the China National Knowledge Infrastructure (CNKI). We accessed the Research Landscape Tool (http://www.soilhealthinstituteresearch.org/Home/Search/Cover-Crop) on April 25, 2018, and found all peer-reviewed journal articles listed there under "cover crops". For search engines, we used the keywords "soil health" or "soil quality" and "cover crop" or "green manure" or "organic farm" to find relevant publications. We searched and downloaded more than 500 papers, and then used the following criteria to determine whether a publication would be included in this study: (1) experiments were conducted in the field or at a research station; (2) the publications reported comparisons between controls (e.g., NC control plots; baseline SOC) and treatments (i.e., with CCs); (3) the publications were either peer-reviewed journal articles, conference collections, theses, or dissertations; and (4) the publications were written in English or Chinese. With these constraints, 1195 comparisons were digitized from 131 studies across the globe. More than half (60%) of comparisons were from North America, with the remainder from the other five habitable continents (Fig. 1). Each study reported an average of 9 comparisons, and some comparisons in each study were not independent from one another. Following the method provided by Alvarez et al. (2017), we allocated a unique experiment ID to a comparison if the CC group, cash crop group, site, tillage, fertilization, soil depth, termination date, or rotation type were different from other comparisons (Fig. A1). Our methodology resulted in 581 independent experiments, of which 144 experiments included SD information (Fig. 1).

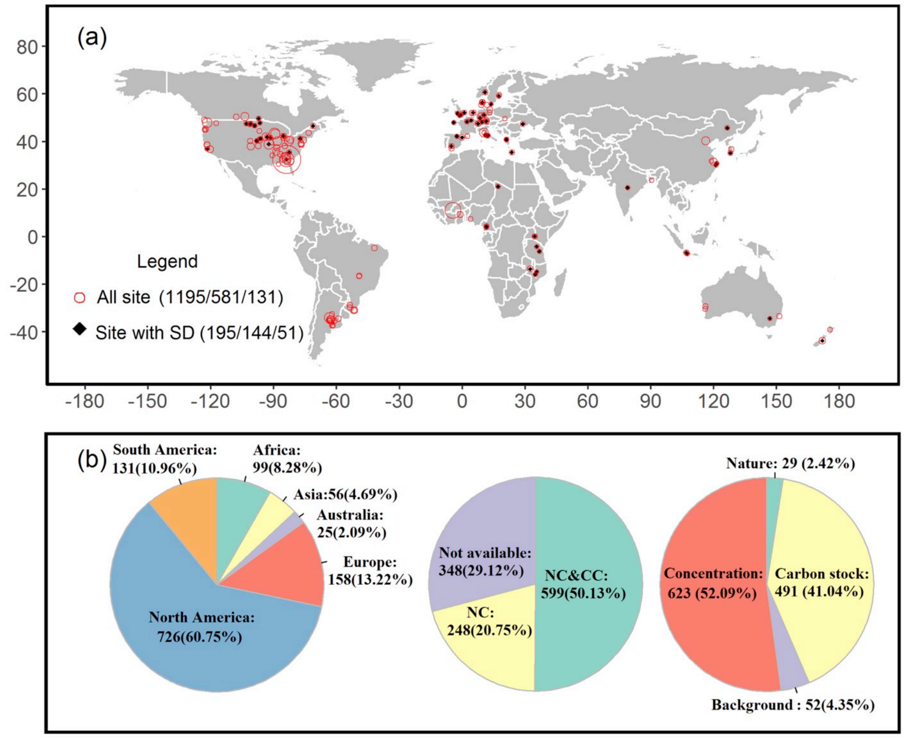


Fig. 1. (a) Site spatial distribution from cover crop studies compiled in the meta-analysis, which included 1195 pairwise comparisons from 131 papers and represented 581 independent experiments (All sites); of those, 195 comparisons from 51 papers reported standard deviations (Sites with SD), representing 144 independent experiments. (b) Breakdown of study comparisons by: (left) continent, (center) whether bulk density data was reported, and (right) type of carbon data reported. NC&CC = bulk density was reported for both control and cover crops; NC = bulk density was only reported for the control; Not available = bulk density was not reported; Carbon stock = carbon was reported as mass of carbon per area [M L^{-2}]; concentration = carbon was reported as mass of carbon per mass of soil [M M^{-1}]; background = background soil carbon was reported; nature = carbon concentration was compared against nearby soils under natural vegetation.

2.2. Data processing

Data were digitized from tables directly and from figures using the software *Data Thief* (version III, http://datathief.org/). After digitization, meta (background) data were extracted from publications and coded as being one of 38 types of information. For more details on these categories, please refer to Stewart et al. (2018).

Whenever latitude and longitude were not reported, we estimated that information based on the name and location of the site, using the website https://www.findlatitudeandlongitude.com/http://www.findlatitudeandlongitude.com/. Elevation was identified by latitude and longitude using https://www.freemaptools.com/elevation-finder.htm/http://www.freemaptools.com/elevation-finder.htm/ whenever it was not reported in the original source.

We applied several quality control tests to verify data quality. First, we checked original source material for all data reported in the meta-analyses/reviews to make sure that the information was translated correctly. Next, we used the following equation to convert any data reported as soil organic matter (SOM) to SOC (Gattinger et al., 2012; Nelson and Sommers, 1996).

$$\mathrm{SOC} = \mathrm{SOM}/1.72 \tag{1}$$

Once all data were collected and converted, we mapped sites by country to confirm that latitude and longitude were collected correctly.

The number of replications and standard deviation (SD) values were recorded from publications when possible. Most studies reported replication number (1076 out of 1195). When applicable, SD values were calculated from reported standard error (SE), coefficient of variation (CV), or confidence interval (CI) values according to the following equations:

$$\mathrm{SD} = \mathrm{SE} \times \sqrt{n} \tag{2}$$

$$\mathrm{SD} = \mathrm{CV} \times \mathrm{mean} \tag{3}$$

$$\mathrm{SD} = |\mathrm{CI} - \mathrm{mean}|/(2Z_{\alpha/2}) \times \sqrt{n} \tag{4}$$

where $Z_{\alpha/2} = 1.96$ when the significance level $\alpha = 0.05$.

For SOC data, only 195 out of 1195 comparisons (16%) reported SD information (Fig. 1). Considering only studies that reported SD, SE, CV, or CI, we determined that the ratio of the SD to the mean did not follow a normal distribution. We therefore used a bootstrap approach to resample the SD/mean with replacement 10,000 times. The mean ratio of SD/mean was 0.12 (with confidence interval of 0.10–0.14) for the control data and 0.13 (with confidence interval of 0.11–0.15) for the CC data (Fig. A4), which closely resembled findings of Luo et al. (2006). We thus assigned SD as 0.12 of the reported mean for all control data that did not report SD values, and as 0.13 of the reported mean for all CC data that did not report the SD values.

Only 599 comparisons (50.1%) reported BD for both the NC controls and CC treatments (Fig. 1b). A regression analysis on those comparisons showed that the BD of the CC treatments was highly correlated with the control BD (adjusted $R^2 = 0.92$; Fig. S5a). We thus estimated soil BD of CCs using the control BD for the 248 comparisons (20.80%, Fig. 1b) that only reported background BD. Finally, our analysis showed that BD was moderately correlated with $SOC_{\%}$ (adjusted $R^2 = 0.47$; Fig. A5b), so we estimated BD based on the SOC information for the 348 comparisons (29.10%, Fig. A5a) that did not include any BD measurements.

Carbon stocks were reported for 491 studies (41.0%), while carbon concentration was reported for 623 comparisons (52.1%; Fig. 1b). Converting carbon concentration values into carbon stock required the following equation:

$$SOC_{stock} = SOC_{\%} \times h \times BD \tag{5}$$

where SOC_{stock} represents soil organic carbon stock [M L^{-2}], $SOC_{\%}$ represents soil organic concentration [M M^{-1}], h represents the soil sampling depth [L], and BD represents soil bulk density [M L^{-3}].

2.3. Data analysis

After calculating SOC_{stock} for all observations, the carbon sequestration rate, C_{rate} [M L^{-2} T^{-1}], was calculated as:

$$C_{rate} = (SOC_{CC} - SOC_{NC})/y \tag{6}$$

where SOC_{CC} and SOC_{NC} are the respective soil carbon stocks [M L^{-2}] under CCs and NC controls, and y represents time after CC implementation [T]. All C_{rate} values were compiled together to determine the global mean rate of SOC change due to CCs. First, the normality of the distribution of C_{rate} values was tested using the Shapiro-Wilk test. Since that analysis suggested that data were not normally distributed ($p < 0.05$), we resampled using a bootstrapping approach to generate a normal distribution from data (Fig. A6).

The overall mean C_{rate} value [M L^{-2} T^{-1}] determined using this approach was then used to estimate total carbon sequestration, $C_{sequestration}$ [M T^{-1}], associated with incorporating CCs into agricultural rotations:

$$C_{sequestration} = C_{rate} \text{ x } A \text{ x } f \tag{7}$$

where A represents the global cropland area [L^2], and f is the proportion of cropland being managed with CCs [L^2 L^{-2}]. A was estimated as the mean of values reported in six studies found in the literature. Three values of f were analyzed: 1) $f = 0.08$, which represented the approximate amount of CCs currently planted in the United States (Tellatin and Myers, 2018); 2) $f = 0.15$, which represented the proportion of CCs that might be planted by 2040 assuming similar rates of adoption (Poeplau and Don, 2015; Tellatin and Myers, 2018); and 3) $f = 1.0$, which represented that total possible carbon that could be sequestered if all cropland was managed using CCs. For more details on how values for C_{rate}, A and f were selected, please refer to the Supplemental Information.

We also computed a response ratio (RR_{SOC}) for each pairwise comparison between SOC stocks in the CC treatment versus NC control:

$$RR_{SOC} = \ln (SOC_{CC} / SOC_{NC}) \tag{8}$$

as well as response ratios (RR_x) for all pairwise values reported for 32 different soil properties and agronomic variables:

$$RR_x = \ln (x_{cc}/x_{nc}) \tag{9}$$

where x_{cc} is the parameter value in the CC treatments and x_{nc} is the parameter value in the NC controls. Specific parameters analyzed included: 1) BD; 2) aggregate stability; 3) porosity; 4) penetration resistance; 5) infiltration rates; 6) saturated hydraulic conductivity; 7) erosion; 8) runoff; 9) leaching; 10) soil temperature; 11) soil water content; 12) available water holding capacity; 13) soil nitrogen; 14) phosphorus; 15) potassium; 16) pH; 17) cation exchange capacity; 18) electricity conductivity; 19) base saturation; 20) soil fauna; 21) fungal indicators; 22) other microbial indicators; 23) enzymatic assays; 24) mineralizable carbon; 25) mineralizable nitrogen; 26) N_2O gas emission; 27) burst test CO_2; 28) field-measured soil CO_2 efflux; 29) microbial biomass carbon; 30) microbial biomass nitrogen; 31) cash crop biomass not including yield (e.g., leaf, steam, root biomass); and 32) cash crop yield.

We next divided data into different categories to explore how climate, cash crop type(s), soil texture, CC type(s), and soil depth affected SOC dynamics under CCs. The climate type of each site was identified based on the Koppen climate classification (Kottek et al., 2006), with relevant categories including tropical, arid, temperate, and snowy climates. Cash crops were grouped into corn, soybean, wheat, other monoculture, corn-soybean rotation, corn-wheat-soybean rotation, and other rotations of more than two cash crops. The CCs were grouped as broadleaf, grass, legume, mixtures of two legumes, mixtures of a legume and a grass, and other mixtures of more than two CCs. Soil texture was grouped as being coarse (sand, loamy sand, and sandy loam), medium (sandy clay loam, loam, silt loam, and silt), or fine (clay, sandy clay, clay loam, silty clay, and silty clay loam) based on The Cornell Framework of Soil Health Manual (Moebius-Clune et al., 2016). Soil sampling depths were grouped into surface, i.e., ≤30 cm, and subsurface, i.e., >30 cm, depth increments (Fig. A2).

After grouping data, SOC_{stock} and C_{rate} values were compiled for the CC versus NC data in each category (reported here as means and SDs). RR_{SOC} values were used in a meta-analysis to compare SOC changes due to CCs for each category. Note that for ease of interpretation, RR_{SOC} values from the meta-analysis are presented as percent change, calculated as $100 * [e^{RRSOC} – 1]$.

Simple linear regression was used to analyze the relationship between RR_{SOC} and RR_x for the 32 different soil/agronomic variables recorded from the studies. Simple linear regression was also applied to explore the relationship between RR_{SOC} and many other environmental conditions. We specifically analyzed 16 environmental variables: 1) elevation; 2) latitude; 3) carbon to nitrogen ratio of CC biomass; 4) CC biomass returned to the field; 5) soil background total carbon content; 6) soil background pH; 7) clay content; 8) silt content; 9) sand content; 10) soil background BD; 11) duration of CCs; 12) years after CC implementation; 13) mean annual precipitation between 1960 and 2015 (MAP); 14) annual precipitation during the study period (Pannual); 15) mean annual temperature between 1960 and 2015 (MAT); and 16) annual temperature during the study period (Tannual). Note that the number of samples differed between different soil properties, agronomic variables, and environmental factors. Adjusted R^2 was used to evaluate the goodness of fit for the linear models used in this analysis.

All statistics were conducted using R (version 3.5.1, R Core Team, 2014). The meta-analysis was applied using '*metafor*' package, and the simple linear regressions were applied under R using the linear model (*lm*) function. We used '*ggmap*' to generate site spatial distribution (Kahle and Wickham, 2013).

3. Results

3.1. Carbon stocks under CCs versus NC controls

Cropland had greater SOC stocks when managed with CCs as compared to NC controls (Fig. 2). Soil carbon stocks were greater in regions characterized by a snowy climate (i.e., ≥1 month with average air temperature < -3 °C; Kottek et al., 2006) compared to temperate and tropic regions. Soil carbon stocks were similar across cash crop systems, except for the corn-wheat-soybean rotation, which had the largest SOC stocks. Likewise, medium-textured soils had the greatest SOC stocks, with mean SOC values of 39 Mg ha^{-1} under CC and 37 Mg ha^{-1} under NC. Soil carbon sequestration rates were similar across climates and cash crop systems; however, fine-textured soils showed larger carbon sequestration rates than the medium- and coarse-textured soils, and surface soils had larger sequestration rates than subsurface layers.

A meta-analysis was applied to analyze RR_{SOC} for different categories (Fig. 3). When cover crops were included in rotations, SOC increased by an average of 15.5% across all experiments (Fig. 3a); however, including only those experiments that reported SD in the original publication showed a 30% increase in SOC under CCs (Fig. 3b). When separated by categories, similar trends can be identified from both methods. Soil carbon stocks were significantly greater in CC treatments versus NC controls for all soil texture groups, but fine-textured soils showed larger increases than medium- and coarse-textured soils. Legume CCs, mixtures of two legumes, and mixtures of more than two other CCs showed significant increases in SOC relative to controls, while grass CCs or grass-legume mixtures did not show significant changes. Corn, wheat, and vegetables all showed significant carbon increases while soybeans, corn-soybean rotations, and corn-wheat-soybean rotations did not. Other rotations of two or more crops (i.e., those that did not include corn and soybeans) also showed significant SOC increases when using CCs. When separated into different soil sampling depths, surface soils showed

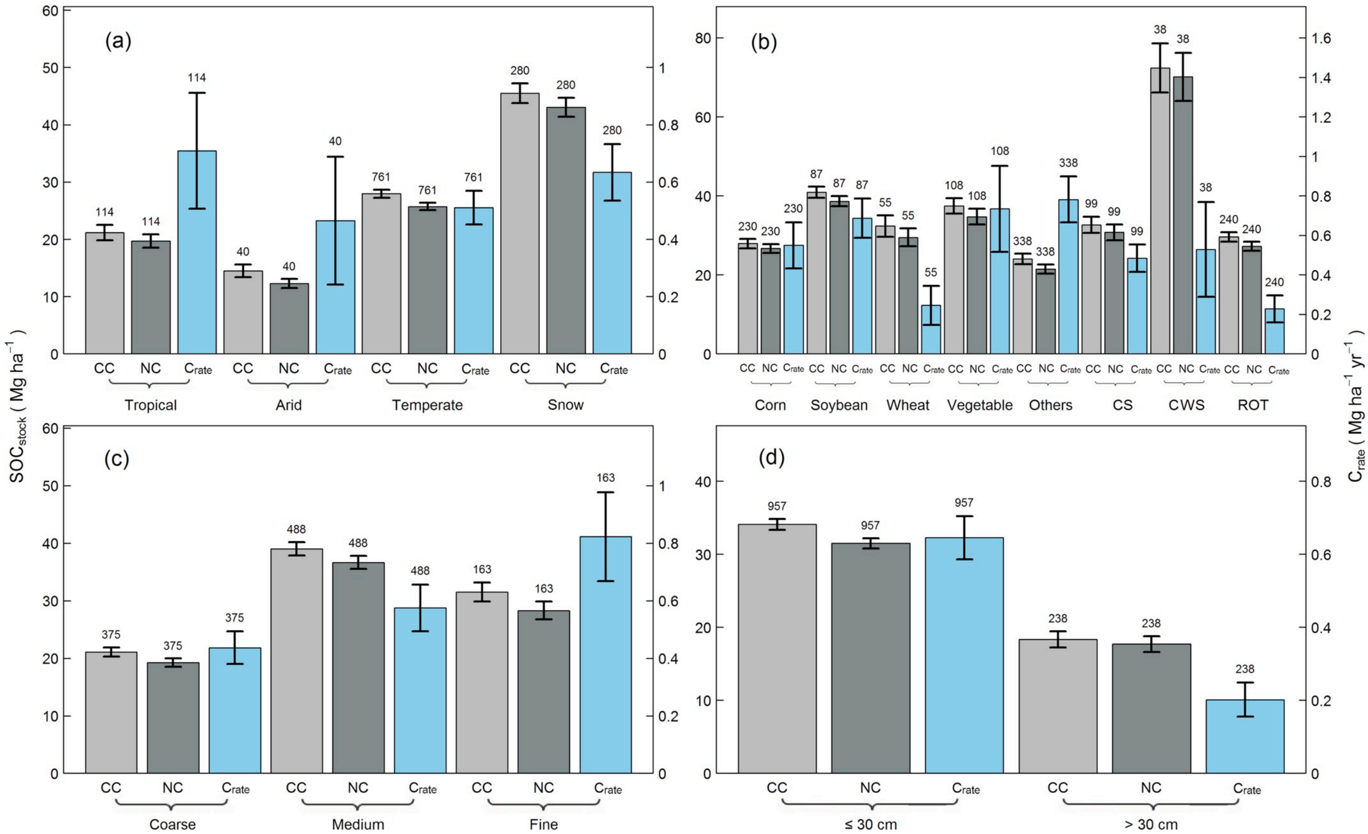


Fig. 2. Soil carbon stocks, SOC_{stock} (Mg ha^{-1}, y-axis on the left) and carbon sequestration rates, C_{rate} (Mg ha^{-1} yr^{-1}, y-axis on the right) in croplands under different (a) climatic regions, (b) cash crop types, (c) soil texture groups, and (d) soil sampling depth increments. Black bars represent standard errors and numbers above the bars are the number of pairwise comparisons. CC: cover crop; NC: no cover crop (i.e., control); C_{rate}: soil carbon sequestration rate; CS: rotation of corn and soybean; CWS: rotation of corn, soybean, and wheat; Other: other monoculture cash crop; ROT: other cash crop rotation.

significant carbon increases when CCs were used, whereas subsurface soils did not.

3.2. Carbon sequestration potential of CCs

The areal extent of global cropland was constrained from six different studies as $A = 1960 \pm 680$ Mha (Table A1). Using the overall mean C_{rate} value of 0.56 Mg ha^{-1} yr^{-1} (Fig. A6b), average global $C_{sequestration}$ values ranged from 0.09 ± 0.03 Pg yr^{-1} (for $f = 0.08$, which represented the current proportion of acreage managed using CCs in the United States) to 0.16 ± 0.06 Pg yr^{-1} (for $f = 0.15$) to 1.1 ± 0.4 Pg yr^{-1} (for $f = 1.0$, which represented all cropland being managed using CCs). Carbon sequestration values thus represented 0.5% on the low end to 16% on the high end of current fossil fuel emissions (Table 1).

3.3. Interactions between SOC changes and soil/agronomic variables

Other than SOC, many other physical, chemical, and biological soil properties can be affected by including CCs as a rotation. Linear regression between the RR_{SOC} (Equation (6)) and RR_x (Equation (7)) showed that SOC changes were negatively correlated with changes in BD, erosion, runoff, soil water content, and burst test CO_2 emissions, which are determined by rewetting air-dried soil to 50% water holding capacity and then measuring soil respiration for at least 2 h (Franzluebbers et al., 2000). SOC changes were positively correlated with changes in soil aggregation, porosity, soil nitrogen, phosphorus, potassium, cation exchange capacity, electrical conductivity, enzymatic assays, mineralizable carbon, mineralizable nitrogen, microbial biomass nitrogen, and biomass of cash crop yield ($p < 0.05$; Fig. 4). Correlations were highest between SOC and runoff (adjusted $R^2 = 0.86$), erosion (adjusted $R^2 = 0.47$), mineralizable carbon (adjusted $R^2 = 0.48$), mineralizable nitrogen (adjusted $R^2 = 0.22$), and soil nitrogen (adjusted $R^2 = 0.21$). SOC was not significantly correlated with other soil properties.

3.4. Interactions between soil carbon changes and environmental factors

Linear regression was applied to analyze the relationship between SOC response from CCs and sixteen other environmental variables (Fig. 5). Results showed that soil carbon stock had significant and positive correlations with annual temperature, years after CC implementation, and CC duration ($p < 0.05$ and slope > 0, labeled as orange in Fig. 5), and significant but negative correlations with latitude and soil background carbon stocks ($p < 0.05$ and slope < 0, labeled as blue in Fig. 5). Individual factors explained little of the total variability in SOC, with only annual temperature and soil background carbon having adjusted R^2 values > 0.05. All other factors were not significantly correlated with SOC changes caused by CCs ($p \geq 0.05$, labeled as pink in Fig. 5). It should be noted that CC biomass ($p = 0.15$) and CC biomass C:N ratio ($p = 0.24$) had the highest adjusted R^2 relationships with SOC changes, even though relationships were not significant. This result may be due to the small number of samples reported for biomass ($n = 28$) and C:N ratios ($n = 22$); which are much fewer than the number of observations reported for other factors.

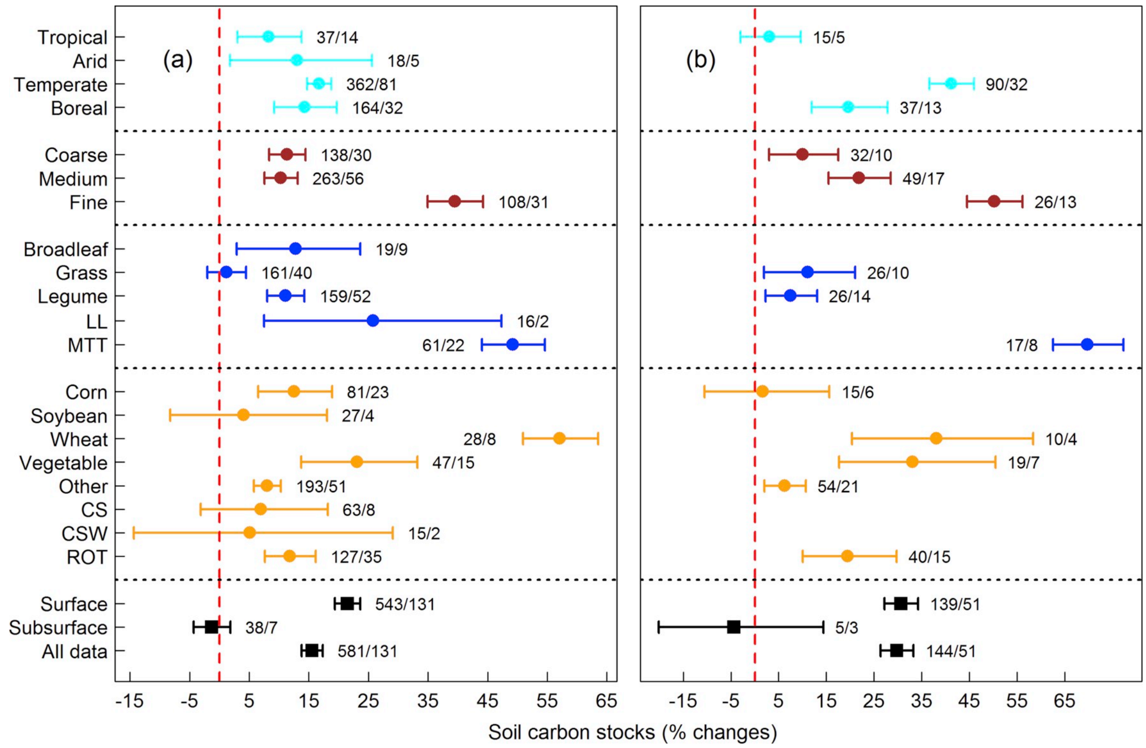


Fig. 3. Meta-analysis results showing change in soil carbon stocks due to implementation of cover crops in four climatic regions, three soil texture groups, six cover crop types, eight cash crop types, and two soil sampling depths. Panel (a) presents results from all comparisons; panel (b) presents meta-analysis results from comparisons that reported sufficient information to calculate standard deviations (SD). Circles or squares with error bars represent the overall mean RR_{SOC} values ± 95% confidence intervals (scaled to %). Categories whose 95% confidence intervals do not cross 0 (represented by the vertical red lines) have significant differences between cover crop treatments and controls. The number of experiments followed by the number of studies are listed for each category. Surface = samples collected from ≤30 cm depth; subsurface = samples collected from depths >30 cm. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

Table 1
Estimates of global soil carbon sequestration potential, $C_{sequestration}$, and % of current fossil fuel emissions, assuming three different levels of cover crop adoption: $f = 0.08$; $f = 0.15$; and $f = 1.0$. Note that C_{rate} was assumed to equal 0.56 Mg ha^{-1} yr^{-1}, A was assumed to equal 1960 ± 680 Mha, and current carbon emissions from fossil fuels were assumed to range from 9.0 to 11.0 Pg yr^{-1} (Jackson et al., 2017). The $C_{sequestration}$ values represent near-surface (≤30 cm depth) soils.

	$f = 0.08$	$f = 0.15$	$f = 1.0$
$C_{sequestration}$ (Pg yr^{-1}) = C_{rate} x A x f	0.09 ± 0.03	0.16 ± 0.06	1.1 ± 0.4
% of current fossil fuel emissions	0.52–1.3	0.98–2.5	6.5–16

4. Discussion

4.1. SOC changes due to cover crop usage

In this study we compiled data from 131 publications that compared SOC concentrations when CCs were included in rotations versus SOC in NC controls. In total, 1195 comparisons were included (Fig. 1). We first examined how different categorical environmental factors influenced SOC stocks and sequestration rates, specifically examining the role of climate, CC type, cash crop type, soil texture, and soil sampling depth (Fig. 2). We then used a meta-analysis to quantify SOC changes under CCs using RR_{SOC} as the response variable (Fig. 3). We also performed linear regressions to examine correlations between changes in SOC and continuous environmental factors (e.g., temperature, precipitation, CC biomass; Figs. 4) and 32 other soil/agronomic variables (Fig. 5). While previous studies examined mechanisms of SOC changes under CCs at various scales (Aguilera et al., 2013; Olson et al., 2014; Schmidt et al., 2017; Stavi et al., 2012; Tian et al., 2018; Poeplau and Don, 2015), this study for the first time analyzed how SOC dynamics correlated with other soil/agronomic variables.

Our analysis compared SOC stocks for CC treatments versus NC controls both in terms of absolute magnitudes (Fig. 2) and relative changes (i.e., Fig. 3). The former approach allowed us to compute the rate of carbon change, C_{rate}, whereas the latter quantified the total change, regardless of time elapsed. Results from these two approaches agreed for many factors, yet some discrepancies arose. As an example, tropical and snowy regions had higher C_{rate} values than temperate and arid climates (Fig. 2), while temperate regions had overall higher RR_{SOC} values than the tropics (Fig. 3). Of all cash crop rotations, wheat monocultures had the highest RR_{SOC} values but relatively low results for C_{rate}. These disparities may reflect variations in time of CC usage, as C_{rate} can change through time (Blanco-Canqui et al., 2015). Differences in sampling depths between studies could also be a factor, as approximately 20% of all observations came from deeper than 30 cm, where SOC changes were negligible (Figs. 2 and 3).

The meta-analysis showed that incorporation of CCs into rotations caused a mean SOC increase of 15.5% (when all comparisons were included) or 30.0% (when only comparisons with SD values were included; Fig. 3). Here we note that including calculated SD data is commonly used in meta-analyses that examine SOC dynamics (Aguilera et al., 2013; Alvarez et al., 2017; Don et al., 2011; Lesur-Dumoulin et al., 2017; Luo et al., 2006; Poeplau and Don, 2015; Sileshi, 2009; Tian et al., 2018; Tonitto et al., 2006). While the approach including all data provided a more conservative estimate of overall SOC changes due to CCs in the present study, it also resulted in higher estimates of SOC change under certain conditions (Fig. 3). The 15.5% mean change in SOC translates to a carbon sequestration value of C_{rate} = 0.56 Mg ha^{-1} yr^{-1} (Fig. A6), while the 30% mean change in SOC translates to C_{rate} = 1.05 Mg ha^{-1} yr^{-1}. Historical SOC losses when converting from natural

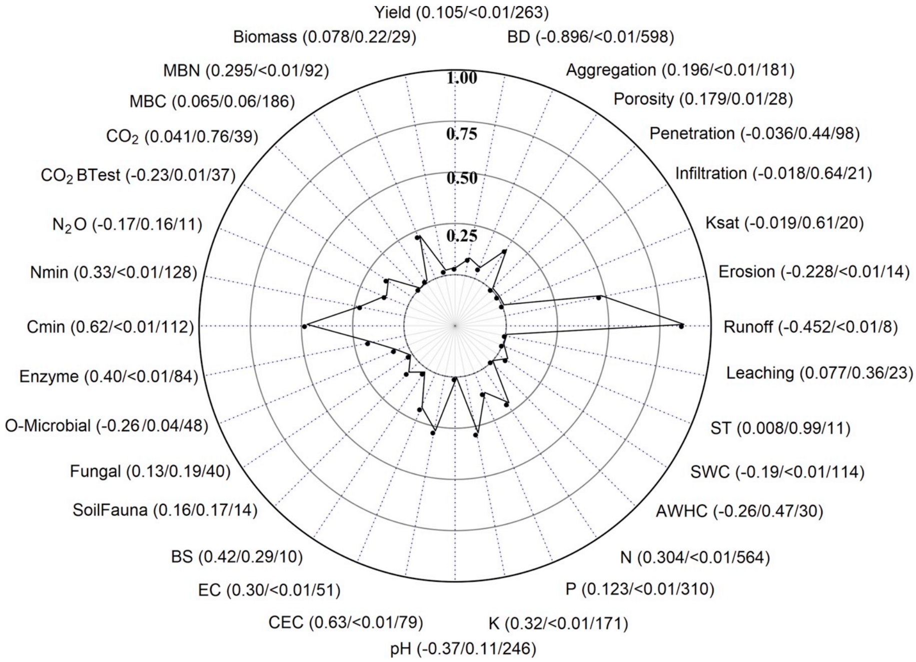


Fig. 4. Correlations between SOC changes due to cover crops and changes in other physical, chemical, biological properties of soil, cash crop biomass, and yield. The dots indicate adjusted R^2 values (each concentric circle represents an R^2 increment of 0.25). The values in the parentheses represent the slope/p-value of slope/number of samples of regression between the SOC response ratio (RR_{SOC}) and response of specific indicators (RR_x). NS means that the correlation was not significant ($p < 0.05$). Acronyms: BD – bulk density, Ksat – saturated hydraulic conductivity, ST – soil temperature, SWC – soil water content, AWHC – available water holding capacity, N – soil nitrogen, P – soil phosphorus, K – soil potassium, CEC – cation exchange capacity, EC – electricity conductivity, BS – base saturation, O-Microbial – other microbial indicator, Cmin – mineralizable carbon, Nmin – mineralizable nitrogen, CO_2BTest – CO_2 burst test, CO_2 – field-measured soil CO_2 efflux, MBC – microbial biomass carbon, MBN – microbial biomass nitrogen, Biomass – cash crop biomass not including yield (e.g., leaf, steam, root biomass).

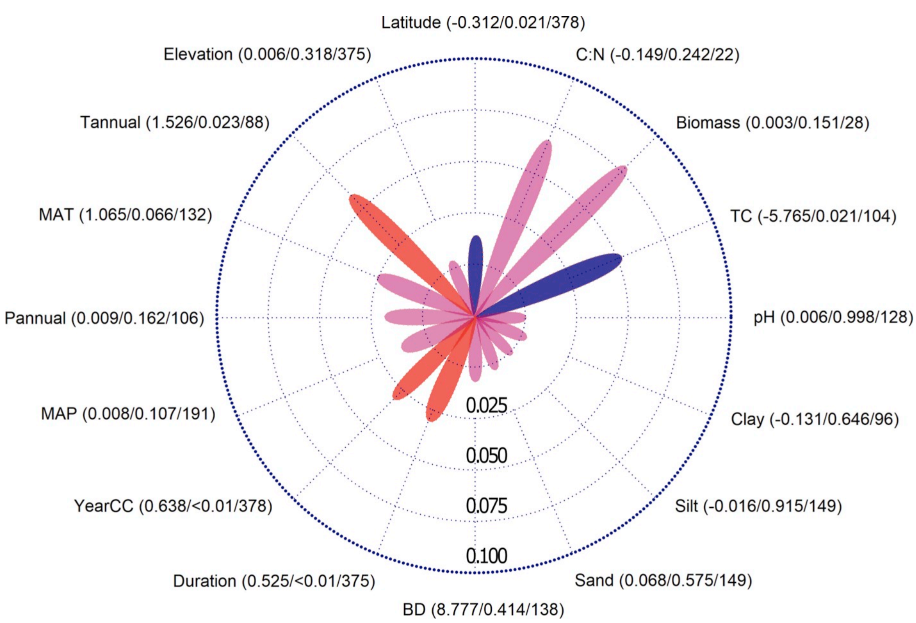


Fig. 5. Adjusted R^2 values from simple linear regression between SOC change under cover crops and interactions with 16 environmental factors. The blue color represents a significant negative correlation between a variable and the SOC response ratio (RR_{SOC}); the orange color represents a significant positive correlation between a variable and RR_{SOC}; and the pink color indicates no significant relationship between a variable and RR_{SOC} ($p > 0.05$). The adjusted R^2 values of 0, 0.025, 0.050, 0.075, and 0.100 are represented by the concentric circles. The values in the parenthesis represents the slope/p-value of slope/number of samples of the regression. Acronyms: MAP – mean annual temperature between 1960 and 2015, Pannual – annual precipitation during study period, MAT – mean annual temperature between 1960 and 2015, Tannual – annual temperature during study period, YearCC – years after cover crop implementation, BD – bulk density, TC – total soil carbon, Biomass – cover crop biomass, C:N – carbon to nitrogen ratio of cover crop biomass. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

vegetation to cropland were estimated to range from −25% to −36% (Don et al., 2011; Poeplau and Don, 2015), so SOC gains under CCs (assuming a 15.5% increase) may recover approximately one-quarter to one-third of this overall SOC loss.

The C_{rate} value of 0.56 Mg ha^{-1} yr^{-1} equals approximately 0.06–0.12 Pg yr^{-1} of sequestered carbon, under the assumption that approximately 8% of cropland is currently managed using CCs worldwide (Table 1). If all cropland were to adopt cover crops (i.e., $f = 1.0$), the $C_{sequestration}$ potential could be as large as 1.5 Pg yr^{-1}. This latter amount would account for 13–16% of annual carbon emissions from fossil fuel combustion, or approximately 1/2 of the terrestrial carbon sink (Stocker et al., 2013). Planting anywhere near 100% of global cropland with CCs is impractical due to numerous reasons, including expenses associated with planting and managing CCs (Zhou et al., 2017), potential water limitations (Reese et al., 2014), lack of suitable growing windows in certain crop rotations (Clark, 2008), and disruptions to cash crop planting and harvesting times. However, this number does provide an upper limit for the amount of atmospheric carbon that may be sequestered via CCs. At the more realistic adoption level of $f = 0.15$, carbon sequestered by CCs could represent ~1–2% of current yearly emissions from fossil fuel combustion.

4.2. Correlations between SOC accumulation and other factors

Our results also showed that different environmental and management factors affected SOC accumulation. For instance, SOC accumulation was found to vary by soil textural class. Medium-textured soils had the highest overall SOC stocks, including both CC and NC data (Fig. 2). Fine-textured soils resulted in the highest carbon increase after introduction of CCs, while medium and coarse-textured soils had lowest SOC increases with CCs (Fig. 3). These results could partially reflect study location, with many studies coming from the Midwestern U.S. (Fig. A7), a region that is characterized by organic-rich, medium-textured soils (Nachtergaele et al., 2010; Reynolds et al., 2000; Scharlemann et al., 2014). Having relatively high SOC stocks under NC conditions likely caused lower carbon sequestration rates relative to fine-textured soils that had relatively low initial SOC stocks.

These findings may also reflect the ability of some fine-textured soils to provide physical protection to SOC (Hassink et al., 1997; Zinn et al., 2005). Clays and silt-sized particles are more likely to form stable aggregates than sands (Gyawali and Stewart, 2019; Sollins et al., 1996), which can protect SOC by isolating it from microbial access (Puget et al., 2000; Six et al., 2004). Clay- and silt-sized particles provide the majority of sorbent surface area within soils and thereby affect SOC sorption and availability (Sollins et al., 1996). Clay particles can also alter microbial metabolism pathways and enzymatic activity, both of which can affect SOC (Huang et al., 1986).

At the same time, factors such as physical heterogeneity and root distributions can be more important than texture (Schmidt et al., 2011), and may help explain seemingly contradictory results in which coarse-textured soils have greater SOC changes than fine-textured soils. For instance, a meta-analysis in the Pampas region of Argentina found that SOC increased more in coarse-textured (+9%) than in fine-textured (+4%) soils (Alvarez et al., 2017). That particular study differed from ours in several other key aspects: 1) the two analyses included different number of studies; 2) Alvarez et al. (2017) focused on the Pampas region, which has a temperate climate, whereas our analysis collected results from many parts of the globe and considered four different climate types with associated effects on plant productivity; and 3) we separated soils into three texture groups (i.e., coarse, medium, and fine) following the Cornell Framework of Soil Health manual (Moebius-Clune et al., 2016), while Alvarez et al. (2017) grouped soils into two groups (i.e., coarse and fine) based on soil family information.

Our analysis also showed that SOC change under CCs was negatively correlated with total soil carbon content. This result may reflect an upper limit in the amount of carbon that can be stored in a soil matrix given the surrounding environmental conditions (e.g., soil temperature and soil moisture; Clark, 2008). SOC increases are often greatest in formerly degraded soils, e.g., soils that have experienced high erosion (Berhe et al., 2007). Therefore, condition of the physical substrate may be an important factor influencing SOC dynamics.

Runoff (adjusted $R^2 = 0.86$), mineralizable carbon (adjusted $R^2 = 0.48$), erosion (adjusted $R^2 = 0.47$), potassium (adjusted $R^2 = 0.29$), CEC (adjusted $R^2 = 0.28$), and mineralizable nitrogen (adjusted $R^2 = 0.22$) were indicators that correlated best with SOC change under CCs. SOC increases after CC introduction were associated with significant decreases in runoff and erosion (Fig. 4). Lower rates of runoff and erosion can reduce SOC losses from the field and thereby form a positive feedback (Berhe et al., 2007; Kaye and Quemada, 2017; Meyer et al., 1997). Likewise, root biomass, rhizo-deposits, and soil microbes are all important sources of SOC (Kutsch et al., 2009) and greater mass and activity of these groups under CCs may help to explain significant positive relationships seen with mineralizable carbon, mineralizable nitrogen, and potassium CCs (Araujo et al., 2012; Balakrishna et al., 2017). Enhanced soil nitrogen, phosphorus, and potassium content may have helped increase CC biomass by providing ample fertility from deeper within the soil profile. This process is especially important in coarse-textured or well-structured soils, where nutrients can experience rapid transport through the soil profile (Tremblay et al., 2012; Zhu et al., 2016).

The results in this study showed that soil carbon stock changes after CCs had significant positive correlations with annual temperature and precipitation (Fig. 5), matching observations reported in other studies. For example, in semiarid areas, SOC can increase when using CCs, but limited biomass production due to low precipitation means that accumulation can take longer than in wetter climates (Blanco-Canqui et al., 2013). At the same time, temperature is a key factor controlling plant growth and is often positively correlated with plant productivity (Churkina and Running, 1998), yet higher temperatures also typically cause higher decomposition rates (Lloyd and Taylor, 1994). The positive relationship between annual temperature and SOC changes after CCs indicates that rate of carbon accumulation from higher plant productivity exceeded any increases in decomposition rates. Our analysis also established that latitude was negatively correlated to changes in SOC_{stock} after cover cropping. Higher latitude locations have short growing seasons, which can reduce CC growth rates and limit plant residue inputs into soils (Mirsky et al., 2017).

When regressed against SOC changes, CC biomass and C:N ratios had relatively high R^2 values, with CC biomass having a positive correlation with SOC and C:N ratio having a negative one. However, these relationships were not significant (Fig. 5). Previous work has found that using CCs during fallow seasons can increase carbon returned to the soil via residue (O'Dea et al., 2013), with the rate of residue return scaling with aboveground biomass (Kuo et al., 1997). Other studies have also established that greater aboveground CC biomass can translate to more root biomass, enhanced rhizo-deposition, and greater diversity of soil microbes (Araujo et al., 2012; Balakrishna et al., 2017). All of these factors can increase the pool of belowground carbon (Kutsch et al., 2009). We thus speculate that the non-significant relationships determined by our study were likely influenced by the limited number of observations reported for each factor (N = 28 for biomass and N = 22 for C:N ratio).

Our analysis did reveal that SOC changes varied between different types of CC species (Fig. 3), which may indirectly reflect differences in CC biomass and C:N ratios. Specifically, legume and mixed CCs both caused significant SOC increases, while grass species did not. One possible reason for this particular outcome is that grass CCs often have high C:N ratios, which can increase the amount of time needed for biomass to be converted into SOC (Jani et al., 2016; Kaye and Quemada, 2017; O'Dea et al., 2013). High C:N ratios can also cause more carbon to be lost as respiration versus stabilized in soil, due to lower carbon use efficiency by decomposer organisms (Manzoni et al., 2012; 2008).

Perhaps as a result of these tradeoffs between maximum biomass and optimum C:N ratios in single-species CCs, mixtures of CCs provided the greatest overall increases in SOC. Similar results were reported by other studies; for example, Faé et al. (2009) found that CC mixtures led to greater SOC increases compared to single species CCs. Zhou et al. (2019) showed that increased plant species richness can reduce the litter C:N ratio and thus promote SOC accumulation. Likewise, Stavi et al. (2012) found that CC mixtures led to greater SOC concentrations (19.4 g kg^{-1}) than single-species CCs (15.9–17.6 g kg^{-1}) for farmland in Ohio. Using CC mixtures may therefore offer a reliable strategy for increasing SOC.

4.3. Comparison with other conservation agriculture practices

Cover crops represent only one conservation strategy for cropland management. It is well recognized that agro-forest systems usually have higher SOC compared with nearby cropland or pastures (Shi et al., 2018). Conservation tillage, including no-till, is another practice that can affect SOC concentrations in soil. No-till management often increases SOC near the soil surface (≤30 cm) compared with conventional tillage systems (Cooper et al., 2016). However, the increase in SOC under no-till has been challenged by recent analyses (Baker et al., 2007; Luo et al., 2010), which showed that SOC increases in soils can be offset by decreases in the subsurface layer. Further, few studies have performed factorial comparisons of cover cropping and tillage practices, limiting our ability to analyze interactions between these management strategies. Other cropland conservation management techniques include alley cropping and inter-seeding, diverse crop rotations, forage and biomass planting, contour farming, mulching, riparian herbaceous cover, and contour buffer strips. The ability of these other cropland conservation management strategies to increase SOC and sequester atmospheric carbon should be investigated in future studies.

4.4. Limitations and perspective

Most existing comparisons reported soil carbon concentration (i.e., $SOC_{\%}$) changes without reporting soil BD. Since BD is necessary to calculate SOC stocks, we had to estimate BD for many instances based on correlations developed using reported values (Fig. A5). As this process added additional uncertainty to the dataset, we recommend that soil carbon stock and BD should be reported in the future studies. Likewise, it is important that studies report SD values whenever possible.

Substantially fewer data were available for the subsurface layers (>30 cm) compared with the surface layers (≤30 cm). Our results suggested that CCs may not change SOC concentrations in subsurface soil layers; however, this conclusion may be due to the small sample size (only 38 experiments from 7 studies and only 5 experiments from 3 studies have SD information; Fig. 3). Because of this uncertainty, future experiments should strive to include samples from subsurface layers, which should help evaluate subsoil benefits of CCs in the future.

Comparisons collected in this study covered a wide time period (1960–2014) and included samples from various depths and sampling increments. We did not attempt to account for any sampling differences in this study. In addition, most comparisons were reported after less than 5 years of data collection, even though CC effects on SOC are often not detectable in the first few years after establishment, due to high spatial field variability or soil heterogeneity (Olson et al., 2014; Poeplau and Don, 2015). Future CC experiments should continue to collect data for mid- (e.g., 5–10 years) and long-term (e.g., >10 years) periods to the extent possible. Longer-term data are particularly important to understand maximum sequestration potentials in soils. The ~0.1–~1 Mg ha^{-1} yr^{-1} carbon sequestration rates found in this study (Fig. 3) and in others (e.g., Alvarez et al., 2017) suggest that soils can sequester 10–100 Mg ha^{-1} $century^{-1}$. However, those values approach or exceed existing carbon stocks in many existing systems (Fig. 2), making it probable that sequestration rates will diminish through time (Blanco-Canqui et al., 2015).

Finally, our estimated potential carbon sequestration values relied on a single mean value for SOC change that translated to all land planted with CCs. These estimates therefore represent crude approximations, as our analysis showed that SOC changes from CCs vary depending on numerous environmental and management factors. Future investigations should therefore seek to refine the initial estimates provided here, for example by incorporating global datasets for climate (Kottek et al., 2006), soil texture (Nachtergaele et al., 2010; Reynolds et al., 2000) and cropping systems (Thenkabail et al., 2013). Even so, it is important to consider that our analysis showed that CCs increase SOC concentrations in nearly all conditions. This general finding emphasizes the importance of efforts that encourage farmers to adopt these practices more widely, despite uncertainties regarding the exact amount of carbon that can be sequestered as a result.

5. Conclusion

In this study, we collected 1195 SOC comparisons between CC treatments and NC controls from 131 studies. Using this data, we conducted a meta-analysis to explore SOC changes under CCs and the interactions between SOC changes, soil properties, and environmental factors. Altogether, cover cropping caused a 15.5% increase in SOC (95% confidence interval of 13.8%–17.3%) in near-surface soils (i.e., ≤30 cm depth), indicating that inclusion of CCs into agricultural rotations can potentially increase soil carbon sequestration. As an example, under the reasonable assumption that 15% of worldwide cropland was to be managed using CCs, approximately 0.16 ± 0.06 Pg of carbon could be sequestered per year. These values represent ~1–2% of yearly emissions from fossil fuel combustion.

Subsurface soils (i.e., >30 cm) showed no significant change in SOC, possibly due to the limited number of samples reported for the subsurface. A regression analysis revealed that SOC increases were correlated with runoff, erosion, mineralizable carbon, mineralizable nitrogen, and soil nitrogen. Surrounding environmental conditions also affected SOC changes under CCs, but only explained a small amount of total variability. Similarly, CC biomass was positively correlated and C:N ratio was negatively correlated with changes in SOC, though these relationships were not significant. Based on sources of uncertainty identified in this study, we propose that future CCs studies should: 1) sample both the near surface (e.g., 0–10 cm) and the subsurface (e.g., 40–50 cm) layers of the soil profile; 2) maintain experiments over mid- (e.g., 5–10 year) to long-term (>10 years) periods; and 3) always report soil BD, so that SOC stock can be appropriately estimated.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

This material is based upon work supported by the Natural Resources Conservation Service, U.S. Department of Agriculture, under Conservation Innovation Grant 69-3A75-14-260. Jinshi Jian was partially supported by the U.S. Department of Energy, Office of Science, Biological and Environmental Research as part of the Terrestrial Ecosystem Sciences Program. Funding for this work was also provided in part by the Virginia Agricultural Experiment Station and the Hatch program of the National Institute of Food and Agriculture, U.S. Department of Agriculture.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.soilbio.2020.107735.

References

Manzoni, S., Taylor, P., Richter, A., Porporato, A., Ågren, G.I., 2012. Environmental and stoichiometric controls on microbial carbon-use efficiency in soils. New Phytologist 196, 79–91. https://doi.org/10.1111/J.1469-8137.2012.04225.X@10.1002/(ISSN)1469-8137(CAT)VIRTUALISSUES(VI)ECOLOGICALSTOICHIOMETRYGLOBALCHANGE.

Abdollahi, L., Munkholm, L.J., 2014. Tillage system and cover crop effects on soil quality:I. Chemical, mechanical, and biological properties. Soil Science Society of America Journal 78, 262. https://doi.org/10.2136/sssaj2013.07.0301.

Aguilera, E., Lassaletta, L., Gattinger, A., Gimeno, B.S., 2013. Managing soil carbon for climate change mitigation and adaptation in Mediterranean cropping systems: a meta-analysis. Agriculture, Ecosystems & Environment 168, 25–36. https://doi.org/10.1016/j.agee.2013.02.003.

Alvarez, R., Steinbach, H.S., De Paepe, J.L., 2017. Cover crop effects on soils and subsequent crops in the pampas: a meta-analysis. Soil and Tillage Research 170, 53–65. https://doi.org/10.1016/j.still.2017.03.005.

Araujo, A.S.F., Leite, L.F.C., De Freitas Iwata, B., De Andrade Lira, M., Xavier, G.R., Do Vale Barreto Figueiredo, M., 2012. Microbiological process in agroforestry systems. A review. Agronomy for Sustainable Development 32, 215–226. https://doi.org/10.1007/s13593-011-0026-0.

R Core Team, 2014. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria, 2013.

Baker, J.M., Ochsner, T.E., Venterea, R.T., Griffis, T.J., 2007. Tillage and soil carbon sequestration—what do we really know? Agriculture, Ecosystems & Environment 118, 1–5.

Balakrishna, A.N., Lakshmipathy, R., Bagyaraj, D.J., Ashwin, R., 2017. Influence of alley copping system on AM fungi, microbial biomass C and yield of finger millet, peanut and pigeon pea. Agroforestry Systems 91, 487–493. https://doi.org/10.1007/s10457-016-9949-4.

Bandick, A.K., Dick, R.P., 1999. Field management effects on soil enzyme activities. Soil Biology and Biochemistry 31, 1471–1479.

Berhe, A.A., Harte, J., Harden, J.W., Torn, M.S., 2007. The significance of the erosion-induced terrestrial carbon sink. BioScience 57, 337–346. https://doi.org/10.1641/B570408.

Blanco-Canqui, H., Mikha, M.M., Presley, D.R., Claassen, M.M., 2011. Addition of cover crops enhances No-till potential for improving soil physical properties. Soil Science Society of America Journal 75, 1471. https://doi.org/10.2136/sssaj2010.0430.

Blanco-Canqui, H., Holman, J.D., Schlegel, A.J., Tatarko, J., Shaver, T.M., 2013. Replacing fallow with cover crops in a semiarid soil: effects on soil properties. Soil Science Society of America Journal 77, 1026. https://doi.org/10.2136/sssaj2013.01.0006.

Blanco-Canqui, H., Shaver, T.M., Lindquist, J.L., Shapiro, C.A., Elmore, R.W., Francis, C. A., Hergert, G.W., 2015. Cover crops and ecosystem services: insights from studies in temperate soils. Agronomy Journal 107, 2449–2474. https://doi.org/10.2134/agronj15.0086.

Callesen, I., Liski, J., Raulund-Rasmussen, K., Olsson, M.T., Tau-Strand, L., Vesterdal, L., Westman, C.J., 2003. Soil carbon stores in Nordic well-drained forest soils-relationships with climate and texture class. Global Change Biology 9, 358–370. https://doi.org/10.1046/j.1365-2486.2003.00587.x.

Campbell, C.A., Biederbeck, V.O., Zentner, R.P., Lafond, G.P., 1991. Effect of crop rotations and cultural practices on soil organic matter, microbial biomass and respiration in a thin Black Chernozem. Canadian Journal of Plant Science 71, 363–376.

Churkina, G., Running, S.W., 1998. Contrasting climatic controls on the estimated productivity of global terrestrial biomes. Ecosystems 1, 206–215.

Clark, A., 2008. Managing Cover Crops Profitably. Diane Publishing.

Cooper, J., Baranski, M., Stewart, G., Nobel-de Lange, M., Bàrberi, P., Fließbach, A., Peigné, J., Berner, A., Brock, C., Casagrande, M., Crowley, O., David, C., De Vliegher, A., Döring, T.F., Dupont, A., Entz, M., Grosse, M., Haase, T., Halde, C., Hammerl, V., Huiting, H., Leithold, G., Messmer, M., Schloter, M., Sukkel, W., van der Heijden, M.G.A., Willekens, K., Wittwer, R., Mäder, P., 2016. Shallow non-inversion tillage in organic farming maintains crop yields and increases soil C stocks: a meta-analysis. Agronomy for Sustainable Development 36. https://doi.org/10.1007/s13593-016-0354-1.

Don, A., Schumacher, J., Freibauer, A., 2011. Impact of tropical land-use change on soil organic carbon stocks - a meta-analysis. Global Change Biology 17, 1658–1670. https://doi.org/10.1111/j.1365-2486.2010.02336.x.

Faé, G.S., Mark Sulc, R., Barker, D.J., Dick, R.P., Eastridge, M.L., Lorenz, N., 2009. Integrating winter annual forages into a no-till corn silage system. Agronomy Journal 101, 1286–1296. https://doi.org/10.2134/agronj2009.0144.

Franzluebbers, A.J., Haney, R.L., Honeycutt, C.W., Schomberg, H.H., Hons, F.M., 2000. Flush of carbon dioxide following rewetting of dried soil relates to active organic pools. Soil Science Society of America Journal 64, 613–623. https://doi.org/10.2136/sssaj2000.642613x.

Gattinger, A., Muller, A., Haeni, M., Skinner, C., Fliessbach, A., Buchmann, N., Mader, P., Stolze, M., Smith, P., Scialabba, N.E.-H., Niggli, U., 2012. Enhanced top soil carbon stocks under organic farming. Proceedings of the National Academy of Sciences 109, 18226–18231. https://doi.org/10.1073/pnas.1209429109.

Gyawali, A.J., Stewart, R.D., 2019. An improved method for quantifying soil aggregate stability. Soil Science Society of America Journal 83, 27–36. https://doi.org/10.2136/sssaj2018.06.0235.

Haruna, S.I., Nkongolo, N.V., Anderson, S.H., Eivazi, F., Zaibon, S., 2018. In situ infiltration as influenced by cover crop and tillage management. Journal of Soil and Water Conservation 73, 164–172. https://doi.org/10.2489/jswc.73.2.164.

Hassink, J., Whitmore, A.P., 1997. A model of the physical protection of organic matter in soils. Soil Science Society of America Journal 61, 131–139. https://doi.org/10.2136/sssaj1997.03615995006100010020x.

Hassink, J., Whitmore, A., Kubát, J., 1997. Size and density fractionation of soil organic matter and the physical capacity of soils to. European Journal of Agronomy 7, 189–199. https://doi.org/10.1016/S0378-519X(97)80025-6.

Houghton, R.A., 1995. Land-use change and the carbon cycle. Global Change Biology 1, 275–287.

Huang, P.M., Schnitzer, M., Stotzky, G., 1986. Influence of soil mineral colloids on metabolic processes, growth, adhesion, and ecology of microbes and viruses. In: Huang, P.M., Schnitzer, M. (Eds.), Interactions of Soil Minerals with Natural Organics and Microbes. Soil Science Society of America Special Publication, Madison, WI, pp. 305–428. https://doi.org/10.2136/sssaspecpub17.c10.

Idowu, O.J., Van Es, H.M., Abawi, G.S., Wolfe, D.W., Schindelbeck, R.R., Moebius-Clune, B.N., Gugino, B.K., 2009. Use of an integrative soil health test for evaluation of soil management impacts. Renewable Agriculture and Food Systems 24, 214–224. https://doi.org/10.1017/S1742170509990068.

Jackson, R.B., Le Quéré, C., Andrew, R.M., Canadell, J.G., Peters, G.P., Roy, J., Wu, L., 2017. Warning signs for stabilizing global CO2 emissions. Environmental Research Letters 12, 110202. https://doi.org/10.1088/1748-9326/aa9662.

Jani, A.D., Grossman, J., Smyth, T.J., Hu, S., 2016. Winter legume cover-crop root decomposition and N release dynamics under disking and roller-crimping termination approaches. Renewable Agriculture and Food Systems 31, 214–229. https://doi.org/10.1017/S1742170515000113.

Jian, J., Stewart, R., Du, X., 2019. SoilHealthDB [WWW document]. Figshare. https://doi.org/10.6084/m9.figshare.8292176.v8.

Jian, J., Du, X., Stewart D., R., 2020. A database for global soil health assessment. Scientific Data 7 (16), 1–8. https://doi.org/10.1038/s41597-020-0356-3.

Jiang, P., Anderson, S.H., Kitchen, N.R., Sadler, E.J., Sudduth, K.a., 2007. Landscape and conservation management effects on hydraulic Properties of a claypan-soil toposequence. Soil Science Society of America Journal 71, 803. https://doi.org/10.2136/sssaj2006.0236.

Kahle, D., Wickham, H., 2013. ggmap: spatial Visualization with ggplot2. R Journal 5, 144–161.

Kaye, J.P., Quemada, M., 2017. Using cover crops to mitigate and adapt to climate change. Rev. Agron. Sustain. Dev. 37 https://doi.org/10.1007/s13593-016-0410-x.

Kottek, M., Grieser, J., Beck, C., Rudolf, B., Rubel, F., 2006. Updated world map of the Köppen-Geiger climate classification. Meteorologische Zeitschrift 15, 259–263.

Krull, E.S., Baldock, J.A., Skjemstad, J.O., 2003. Importance of mechanisms and processes of the stabilisation of soil organic matter for modelling carbon turnover. Functional Plant Biology 30, 207–222.

Kuo, S., Sainju M., U., Jellum J., E., 1997. Winter Cover Crop Effects on Soil Organic Carbon and Carbohydrate in Soil. Soil Science Society of America Journal 61 (1), 145–152.

Kutsch, W.L., Bahn, M., Heinemeyer, A., 2009. Soil Carbon Dynamics: an Integrated Methodology. Cambridge University Press.

Lesur-Dumoulin, C., Malézieux, E., Ben-Ari, T., Langlais, C., Makowski, D., 2017. Lower average yields but similar yield variability in organic versus conventional horticulture. A meta-analysis. Agronomy for Sustainable Development 37. https://doi.org/10.1007/s13593-017-0455-5.

Lloyd, J., Taylor, J.A., 1994. On the temperature dependence of soil respiration. Functional Ecology 8, 315. https://doi.org/10.2307/2389824.

Luo, Y., Hui, D., Zhang, D., 2006. Elevated CO2 stimulates net accumulations of carbon and nitrogen in land ecosystems: a meta-analysis. Ecology 87, 53–63. https://doi.org/10.1890/04-1724.

Luo, Z., Wang, E., Sun, O.J., 2010. Can no-tillage stimulate carbon sequestration in agricultural soils? A meta-analysis of paired experiments. Agriculture, Ecosystems & Environment 139, 224–231. https://doi.org/10.1016/j.agee.2010.08.006.

Manzoni, S., Jackson, R.B., Trofymow, J.A., Porporato, A., 2008. The global stoichiometry of litter nitrogen mineralization. Science 321, 684–686. https://doi.org/10.1126/science.1159792.

Marques, M.J., García-Muñoz, S., Muñoz-Organero, G., Bienes, R., 2010. Soil conservation beneath grass cover in hillside vineyards under mediterranean climatic conditions (MADRID, Spain). Land Degradation & Development 21, 122–131. https://doi.org/10.1002/ldr.915.

Mazzoncini, M., Sapkota, T.B., Bàrberi, P., Antichi, D., Risaliti, R., 2011. Long-term effect of tillage, nitrogen fertilization and cover crops on soil organic carbon and total nitrogen content. Soil and Tillage Research 114, 165–174. https://doi.org/10.1016/j.still.2011.05.001.

McGuire, a.D., Sitch, S., Clein, J.S., Dargaville, R., Esser, G., Foley, J., Heimann, M., Joos, F., Kaplan, J., Kicklighter, D.W., Meier, R.a., Melillo, J.M., Moore, B., Prentice, I.C., Ramankutty, N., Reichenau, T., Schloss, a., Tian, H., Williams, L.J., Wittenberg, U., 2001. Carbon balance of the terrestrial biosphere in the Twentieth Century: analyses of CO 2 , climate and land use effects with four process-based ecosystem models. Global Biogeochemical Cycles 15, 183–206. https://doi.org/10.1029/2000GB001298.

Meyer, L.D., Dabney, S.M., Murphree, C.E., Harmon, W.C., Grissinger, E.H., 1997. Effect of cropland management practices on storm runoff and erosion. In: Proceedings of the Conference on Management of Landscapes Distributed by Channel Incision. The University of Mississippi, Oxford Campus, Mississippi, pp. 983–989.

Mirsky, S.B., Ackroyd, V.J., Cordeau, S., Curran, W.S., Hashemi, M., Reberg-Horton, S.C., Ryan, M.R., Spargo, J.T., 2017. Hairy vetch biomass across the eastern United States: effects of latitude, seeding rate and date, and termination timing. Agronomy Journal 109, 1510–1519. https://doi.org/10.2134/agronj2016.09.0556.

Moebius-Clune, B.N., Moebius-Clune, D.J., Gugino, B.K., Idowu, O.J., Schindelbeck, R.R., Ristow, A.J., van Es, H.M., Thies, J.E., Shayler, H.A., McBride, M.B., Kurtz, K.S.M., Wolfe, D.W., Abawi, G.S., 2016. Comprehensive Assessment of Soil Health: the Cornell Framework Manual, Third. ed. Cornell University, Ithaca, New York.

Nachtergaele, F., van Velthuizen, H., Verelst, L., Batjes, N.H., Dijkshoorn, K., van Engelen, V.W.P., Fischer, G., Jones, A., Montanarela, L., 2010. The harmonized world soil database. In: Proceedings of the 19th World Congress of Soil Science. Soil Solutions for a Changing World, Brisbane, Australia, pp. 1–6. August 2010. pp. 34–37.

Ndiaye, E.L., Sandeno, J.M., McGrath, D., Dick, R.P., 2000. Integrative biological indicators for detecting change in soil quality. American Journal of Alternative Agriculture 15, 26. https://doi.org/10.1017/S0889189300008432.

Nelson, D.W., Sommers, L.E., 1996. Total carbon, organic carbon, and organic matter. Methods Soil Anal. Part 3—Chem. Methods 961–1010.

Olson, K., Ebelhar, S.A., Lang, J.M., 2014. Long-term effects of cover crops on crop yields, soil organic carbon stocks and sequestration. Open Journal of Soil Science 4, 284–292. https://doi.org/10.4236/ojss.2014.48030.

O'Dea, J.K., Miller, P.R., Jones, C.A., 2013. Greening summer fallow with legume green manures: on-farm assessment in north-central Montana. Journal of Soil and Water Conservation 68, 270–282. https://doi.org/10.2489/jswc.68.4.270.

Poeplau, C., Don, A., 2015. Carbon sequestration in agricultural soils via cultivation of cover crops - a meta-analysis. Agriculture, Ecosystems & Environment 200, 33–41. https://doi.org/10.1016/j.agee.2014.10.024.

Poeplau, C., Don, A., Vesterdal, L., Leifeld, J., Van Wesemael, B., Schumacher, J., Gensior, A., 2011. Temporal dynamics of soil organic carbon after land-use change in the temperate zone - carbon response functions as a model approach. Global Change Biology 17, 2415–2427. https://doi.org/10.1111/j.1365-2486.2011.02408.x.

Puget, P., Chenu, C., Balesdent, J., 2000. Dynamics of soil organic matter associated with particle-size fractions of water-stable aggregates. European Journal of Soil Science 51, 595–605. https://doi.org/10.1111/j.1365-2389.2000.00353.x.

Reese, C.L., Clay, D.E., Clay, S.A., Bich, A.D., Kennedy, A.C., Hansen, S.A., Moriles, J., 2014. Winter cover crops impact on corn production in semiarid regions. Agronomy Journal 106, 1479–1488. https://doi.org/10.2134/agronj13.0540.

Reynolds, C.A., Jackson, T.J., Rawls, W.J., 2000. Estimating soil water-holding capacities by linking the Food and Agriculture Organization soil map of the world with global pedon databases and continuous pedotransfer functions. Water Resources Research 36, 3653–3662. https://doi.org/10.1029/2000WR900130.

Sainju, U.M., Singh, B.P., Whitehead, W.F., Wang, S., 2006. Carbon supply and storage in tilled and nontilled soils as influenced by cover crops and nitrogen fertilization. J. Environ. Qual. 35, 1507. https://doi.org/10.2134/jeq2005.0189.
Scharlemann, J.P.W., Tanner, E.V.J., Hiederer, R., Kapos, V., 2014. Global soil carbon: understanding and managing the largest terrestrial carbon pool. Carbon Management 5, 81–91. https://doi.org/10.4155/cmt.13.77.
Schmidt, M.W.I., Torn, M.S., Abiven, S., Dittmar, T., Guggenberger, G., Janssens, I.A., Kleber, M., Kögel-Knabner, I., Lehmann, J., Manning, D.A.C., Nannipieri, P., Rasse, D.P., Weiner, S., Trumbore, S.E., 2011. Persistence of soil organic matter as an ecosystem property. Nature. https://doi.org/10.1038/nature10386.
Schmidt, H.P., Pandit, B.H., Cornelissen, G., Kammann, C.I., 2017. Biochar-based fertilization with liquid nutrient enrichment: 21 field trials covering 13 crop species in Nepal. Land Degradation & Development 28, 2324–2342. https://doi.org/10.1002/ldr.2761.
Shi, L., Feng, W., Xu, J., Kuzyakov, Y., 2018. Agroforestry systems: meta-analysis of soil carbon stocks, sequestration processes, and future potentials. Land Degradation & Development. https://doi.org/10.1002/ldr.3136.
Sileshi, G., 2009. In: Evidence for Impact of Green Fertilizers on Maize Production in Sub-saharan Africa.
Six, J., Bossuyt, H., Degryze, S., Denef, K., 2004. A history of research on the link between (micro)aggregates, soil biota, and soil organic matter dynamics. Soil and Tillage Research. https://doi.org/10.1016/j.still.2004.03.008.
Sollins, P., Homann, P., Caldwell, B.A., 1996. Stabilization and destabilization of soil organic matter: mechanisms and controls. Geoderma 74, 65–105. https://doi.org/10.1016/S0016-7061(96)00036-5.
Spargo, J.T., Alley, M.M., Follett, R.F., Wallace, J.V., 2008. Soil carbon sequestration with continuous no-till management of grain cropping systems in the Virginia coastal plain. Soil and Tillage Research 100, 133–140. https://doi.org/10.1016/j.still.2008.05.010.
Stavi, I., Lal, R., Jones, S., Reeder, R.C., 2012. Implications of cover crops for soil quality and geodiversity in a humid-temperate region in the midwestern USA. Land Degradation & Development 23, 322–330. https://doi.org/10.1002/ldr.2148.
Steele, M.K., Coale, F.J., Hill, R.L., 2012. Winter annual cover crop impacts on No-till soil physical properties and organic matter. Soil Science Society of America Journal 76, 2164. https://doi.org/10.2136/sssaj2012.0008.
Stewart, R.D., Jian, J., Gyawali, A.J., Thomason, W.E., Badgley, B.D., Reiter, M.S., Strickland, M.S., 2018. What we talk about when we talk about soil health. Agric. Environ. Lett. 5–9. https://doi.org/10.2134/ael2018.06.0033.
Stocker, T.F., Qin, D., Plattner, G.-K., Tignor, M., Allen, S.K., Boschung, J., Nauels, A., Xia, Y., Bex, V., Midgley, P.M., 2013. Climate Change 2013: the Physical Science Basis.
Tellatin, S., Myers, R.L., 2018. Cover crop impacts on US cropland carbon sequestration. Journal of Soil and Water Conservation 73, 117A–121A. https://doi.org/10.2489/jswc.73.5.117A.
Tesfahunegn, G.B., Tamene, L., Vlek, P.L.G., Mekonnen, K., 2016. Assessing farmers' Knowledge of weed species , crop type and soil management practices in relation to soil quality status in mai-negus catchment, northern Ethiopia. Land Degradation & Development 27, 120–133.
Thenkabail, P.S., Mariotto, I., Gumma, M.K., Middleton, E.M., Landis, D.R., Huemmrich, K.F., 2013. Selection of hyperspectral narrowbands (HNBs) and composition of hyperspectral twoband vegetation indices (HVIs) for biophysical characterization and discrimination of crop types using field reflectance and Hyperion/EO-1 data. IEEE J. Selected Topics Appl. Earth Observ. Remote Sens. 6, 427–439.
Tian, D., Xiang, Y., Wang, B., Li, M., Liu, Y., Wang, J., Li, Z., Niu, S., 2018. Cropland abandonment enhances soil inorganic nitrogen retention and carbon stock in China: a meta-analysis. Land Degradation & Development. https://doi.org/10.1002/ldr.3137.
Tonitto, C., David, M.B., Drinkwater, L.E., 2006. Replacing bare fallows with cover crops in fertilizer-intensive cropping systems: a meta-analysis of crop yield and N dynamics. Agriculture, Ecosystems & Environment 112, 58–72. https://doi.org/10.1016/j.agee.2005.07.003.
Tremblay, N., Bouroubi, Y.M., Bélec, C., Mullen, R.W., Kitchen, N.R., Thomason, W.E., Ebelhar, S., Mengel, D.B., Raun, W.R., Francis, D.D., Vories, E.D., Ortiz-Monasterio, I., 2012. Corn response to nitrogen is influenced by soil texture and weather. Agronomy Journal 104, 1658–1671. https://doi.org/10.2134/agronj2012.0184.
Utomo, M., Frye, W.W., Blevins, R.L., 1990. Sustaining soil nitrogen for corn using hairy vetch cover crop. Agronomy Journal 82, 979. https://doi.org/10.2134/agronj1990.00021962008200050028x.
Van Eerd, L.L., Congreves, K.A., Hayes, A., Verhallen, A., Hooker, D.C., 2014. Long-term tillage and crop rotation effects on soil quality, organic carbon, and total nitrogen. Canadian Journal of Soil Science 94, 303–315. https://doi.org/10.4141/cjss2013-093.
Wiesmeier, M., Hübner, R., Barthold, F., Spörlein, P., Geuß, U., Hangen, E., Reischl, A., Schilling, B., von Lützow, M., Kögel-Knabner, I., 2013a. Amount, distribution and driving factors of soil organic carbon and nitrogen in cropland and grassland soils of southeast Germany (Bavaria). Agriculture, Ecosystems & Environment 176, 39–52. https://doi.org/10.1016/j.agee.2013.05.012.
Wiesmeier, M., Prietzel, J., Barthold, F., Spörlein, P., Geuß, U., Hangen, E., Reischl, A., Schilling, B., von Lützow, M., Kögel-Knabner, I., 2013b. Storage and drivers of organic carbon in forest soils of southeast Germany (Bavaria) - implications for carbon sequestration. Forest Ecology and Management 295, 162–172. https://doi.org/10.1016/j.foreco.2013.01.025.
Zhou, X.V., Larson, J.A., Boyer, C.N., Roberts, R.K., Tyler, D.D., 2017. Tillage and cover crop impacts on economics of cotton production in Tennessee. Agronomy Journal 109, 2087–2096. https://doi.org/10.2134/agronj2016.12.0733.
Zhou, G., Xu, S., Ciais, P., Manzoni, S., Fang, J., Yu, G., Tang, X., Zhou, P., Wang, W., Yan, J., Wang, G., Ma, K., Li, S., Du, S., Han, S., Ma, Y., Zhang, D., Liu, J., Liu, S., Chu, G., Zhang, Q., Li, Y., Huang, W., Ren, H., Lu, X., Chen, X., 2019. Climate and litter C/N ratio constrain soil organic carbon accumulation. Nat. Sci. Rev. https://doi.org/10.1093/nsr/nwz045.
Zhu, B., Yi, L., xia, Xu, shui, H., Guo, L., mei, Hu, gao, Y., Zeng, Z. hai, Chen, F., Liu, Z. yong, 2016. Non-leguminous winter cover crop and nitrogen rate in relation to double rice grain yield and nitrogen uptake in Dongting Lake Plain, Hunan Province, China. J. Integr. Agric. 15, 2507–2514. https://doi.org/10.1016/S2095-3119(16)61331-X.
Zinn, Y.L., Lal, R., Resck, D.V.S., 2005. Texture and organic carbon relations described by a profile pedotransfer function for Brazilian Cerrado soils. Geoderma 127, 168–173. https://doi.org/10.1016/j.geoderma.2005.02.010.

Update

Soil Biology and Biochemistry

Volume 161, Issue , October 2021, Page

DOI: https://doi.org/10.1016/j.soilbio.2021.108389

Soil Biology and Biochemistry 161 (2021) 108389

Contents lists available at ScienceDirect

Soil Biology and Biochemistry

journal homepage: www.elsevier.com/locate/soilbio

Corrigendum to “A meta-analysis of global cropland soil carbon change from cover cropping” [Soil Biol. Biochem. 143 (2020) 107735]

Jinshi Jian[a,b], Xuan Du[c], Mark S. Reiter[a,d], Ryan D. Stewart[a,*]

[a] *School of Plant and Environmental Sciences, Virginia Tech, Blacksburg, VA, USA*
[b] *Pacific Northwest National Laboratory-University of Maryland Joint Global Change Research Institute, 5825 University Research Court, Suite, 3500, College Park, MD, USA*
[c] *Department of Hydraulic Engineering, Yangling Vocational & Technical College, Yang Ling, Shaanxi, China*
[d] *Eastern Shore Agricultural Research and Extension Center, Virginia Tech, Painter, VA, USA*

The authors regret that Figure A5b (Supplementary Data) included an error. The originally published figure was labelled as presenting the relationship between bulk density and soil organic carbon concentration (%), but the figure actually showed bulk density versus soil organic carbon stocks (Mg/ha). Figure A5b has been revised to present bulk density versus soil organic carbon concentrations (%), as this relationship was the basis for the analyses presented in the manuscript. The correct data have an *adj*R^2 value of 0.20, whereas *adj*$R^2 = 0.47$ was reported in the original text. All other analyses remain unchanged.

The authors would like to apologise for any inconvenience caused.

DOI of original article: https://doi.org/10.1016/j.soilbio.2020.107735.
* Corresponding author.
E-mail address: ryan.stewart@vt.edu (R.D. Stewart).

https://doi.org/10.1016/j.soilbio.2021.108389

Available online 12 August 2021

Exhibit G

Agroforest Syst (2012) 86:105–111
DOI 10.1007/s10457-012-9573-x

Agroforestry for biomass production and carbon sequestration: an overview

Shibu Jose · Sougata Bardhan

Received: 10 September 2012 / Accepted: 26 September 2012 / Published online: 4 October 2012

Abstract Ever since the Kyoto Protocol, agroforestry has gained increased attention as a strategy to sequester carbon (C) and mitigate global climate change. Agroforestry has been recognized as having the greatest potential for C sequestration of all the land uses analyzed in the Land-Use, Land-Use Change and Forestry report of the IPCC; however, our understanding of C sequestration in specific agroforestry practices from around the world is rudimentary at best. Similarly, while agroforestry is well recognized as a land use practice capable of producing biomass for biopower and biofuels, very little information is available on this topic. This thematic issue is an attempt to bring together a collection of articles on C sequestration and biomass for energy, two topics that are inextricably interlinked and of great importance to the agroforestry community the world over. These papers not only address the aboveground C sequestration, but also the belowground C and the role of decomposition and nutrient cycling in determining the size of soil C pool using specific case studies. In addition to providing allometric methods for quantifying biomass production, the biological and economic realities of producing biomass in agroforestry practices are also discussed.

S. Jose (✉) · S. Bardhan
The Center for Agroforestry, School of Natural Resources, University of Missouri, 203 Anheuser Busch Natural Resources Building, Columbia, MO 65211, USA
e-mail: joses@missouri.edu

Keywords Soil carbon · Coffee agroforestry · Cacao agroforestry · Bioenergy · Biofuels · Allometric equations · Biomass crops

Introduction

Global climate change and energy security are two key issues that are at the forefront of environmental discussions the world over. Although they bring up unique challenges, global warming and energy security are inextricably interlinked. Increasing concentration of atmospheric carbon dioxide (CO_2) is considered the predominant cause of global climatic change. It is believed that agricultural and forestry practices can partially mitigate increasing CO_2 concentration by sequestering carbon (C). Similarly, alternative agricultural practices where biomass crops are cultivated can impact CO_2 levels not only by sequestering C, but also by replacing fossil fuel with the biomass produced. Agroforestry, like many other land use systems, offers great potential for sequestering C and producing biomass for biofuels.

Ever since the Kyoto Protocol, agroforestry has gained increased attention as a strategy to sequester C from both developed and developing nations. The available estimates of C stored in agroforestry range from 0.29 to 15.21 Mg C/ha/year above ground, and 30–300 Mg C/ha up to 1 m depth in the soil (Nair et al. 2010). Since the industrial revolution, atmospheric CO_2 has increased by more than 40 %, from 280 ppm in 1,750 to about 392 ppm in 2012 and at the current rate it is expected to surpass 400 ppm by 2015 (Hutchinson et al. 2007; Tans 2012). In the three years from 2010 to 2012, CO_2 emissions increased at an alarming rate of 2 ppm/year or 4.4 Pg C/year. While agroforestry has been recognized as having the greatest potential for C sequestration (Fig. 1) of all the land uses analyzed in the Land-Use, Land-Use Change and Forestry report of the IPCC (2000), our understanding of C sequestration in specific agroforestry practices from around the world is rudimentary at best.

The incorporation of trees or shrubs on farms or pastures can increase the amount of C sequestered compared to a monoculture field of crop plants or pasture (Sharrow and Ismail 2004; Kirby and Potvin 2007). In addition to the significant amount of C stored in aboveground biomass, agroforestry systems can also store C belowground. While most studies report aboveground C sequestration, belowground C and soil C are often not reported from agroforestry systems. The soil C pool, comprising about 2,500 Gt, is one of the largest C pools and is larger than the atmospheric pool (760 Gt) (Lal 2004). The extent of soil C is dependent on a delicate balance between litter and rhizodeposition and the release of C due to decomposition and mineralization. Several other factors such as quality of C input, climate, soil physical and chemical properties further determine the rate of decomposition and thus stabilization of soil organic C in a particular ecosystem. Since modernization of agriculture in the 19th-century, soil carbon pool has gradually depleted because of several factors such as deforestation, intensive cropping and biomass removal, soil erosion, and unsustainable agricultural practices. Most of the decline in soil organic matter has been observed in regions under intensive crop production such as continuous row cropping or monocropping. Depletion of soil C has been documented to result in decreased productivity, poor soil physical and chemical properties, and negative secondary environmental impacts. It has been well documented that conversion of degraded agricultural soils into agroforestry systems can rebuild soil productivity.

One of the commodities agroforestry is well suited to producing is biomass for biopower and biofuels (Jose et al. 2012). Heavy reliance on foreign based fossil fuels has sparked an interest in domestic renewable energy sources in many countries. For example, in 2003 the Biomass Research and Development Technical Committee (BRDTC), established by US Congress in 2000, envisioned a goal of a 30 % replacement of US petroleum consumption with biofuels by 2030 (DOE; US Department of Energy 2003). Currently, petroleum products supply about 36 % of US energy consumption, while biomass and biofuels provide 4.3 % of total US energy consumption (EIA; Energy Information Administration 2011).

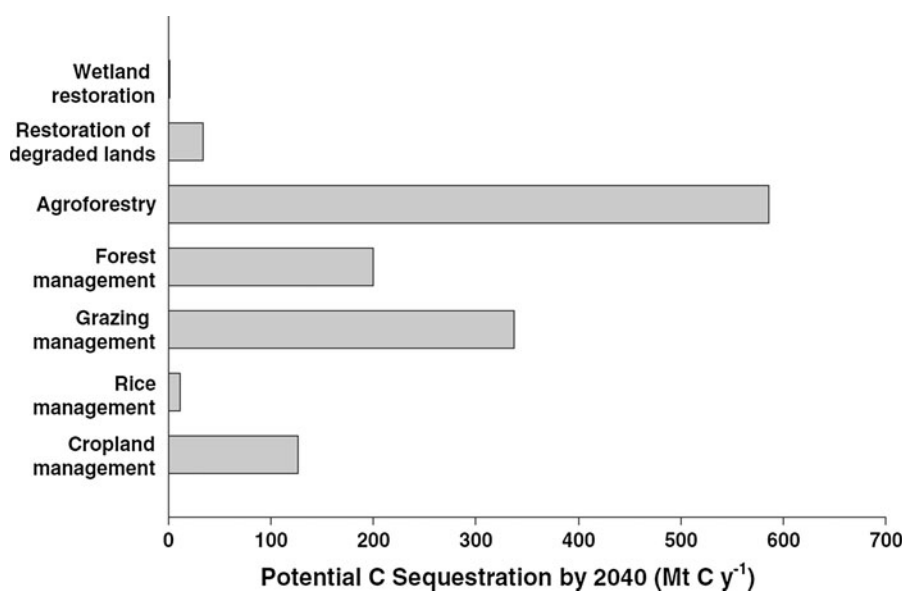


Fig. 1 Carbon sequestration potential of different land use systems by 2040 (adapted from IPCC 2000). Agroforestry offers the greatest potential because of the large extent of area (630 × 10^6 ha) available worldwide for agroforestry adoption

While a study conducted by the US Department of Energy concluded that achieving this goal is possible, the report stated significant expansion of perennial biofuel crop production would be necessary (DOE 2011). Although agroforestry offers a solution to avoid the food versus fuel debate by combining food production and biomass for energy on the same piece of land (Henderson and Jose 2010; Holzmueller and Jose 2012), very little information is available on this topic.

If agroforestry is to be used in C sequestration schemes such as the clean development mechanisms (CDM), better information is required about above and belowground C stocks, soil C, and areas under agroforestry practices. Although there have been a number of publications recently and in the past about the C sequestration potential of agroforestry systems, the information is widely dispersed (but, see Kumar and Nair 2011). The objective of this thematic issue is to compile several original research articles from North America, South America, and Africa that investigate C sequestration and biomass production potential of specific agroforestry practices.

Quantifying carbon sequestration

The first two papers examined C sequestration in silvopastoral systems. Dube et al. investigated the carbon (C) sequestration potentials of three predominant ecosystems in Patagonia in Chile: *Pinus ponderosa*-based silvopastoral systems (SPS), pine plantations (PPP) and natural pasture (PST). Silvopastoral systems are highly efficient in increasing productivity for both plants and animals as mutually optimum conditions for growth and development are created in a properly managed silvopastoral system. Plants gain benefits through nutrient cycling by addition of manure in the system and partial shade from the canopy while animals enjoy ideal temperature and humidity under the tree canopy. In their study, Dube et al. observed higher aboveground tree C, belowground tree C, and soil organic C stock in the silvopastoral system compared to the other systems. Silvopastoral systems also had more favorable air temperature and soil moisture parameters.

Ermson et al. conducted a similar study in the southeastern US where they explored the effect of grazing and forage enhancement on total soil C (TSC), soil nitrogen(N), and phosphorus (P) dynamics in a goat (*Capra aegagrus hircus*)—loblolly pine (*Pinus taeda* L.) silvopasture system on a Kipling silt loam soil (fine, smectitic, thermic, Typic Paleudalfs) in Alabama from 2006 to 2010. In this study however, silvopasture plots were characterized by low initial pH, low TSC, and the soils were deficient in N and P. Four years after tree thinning and 3 years of grazing in June 2010, the silvopasture treatment still exhibited low soil pH (<6) and TSC levels of less than 20 g/kg. The authors speculated that in the long-term, grazing without additional soil management practices may still improve soil fertility through nutrient recycling and C sequestration and thereby making the goat-loblolly silvopasture system both environmentally and economically sustainable.

The next two papers investigated C sequestration in coffee agroforests, one in Guatemala and another in Costa Rica. Schmitt-Harsh observed that coffee agroforests in Guatemala stored somewhere between 74.0 and 259.0 Mg C/ha with a mean of 127.6 Mg C/ha. The average carbon stocks of coffee agroforests were significantly lower than estimated for the mixed dry forests (198.7 Mg C/ha); however, individual tree and soil C pools were not significantly different suggesting that shade trees played an important role in facilitating C sequestration and soil conservation in these systems. This research demonstrates the importance of conservation-based production systems such as coffee agroforests in sequestering C alongside natural forest systems.

Hãger attempted to unravel the relationship between species composition, diversity, and C storage in coffee agroforests of Costa Rica. Total C stocks were 43 % higher on organic farms than on conventional farms ($P < 0.05$) and although vegetation structure was different, there was no difference in species diversity between organic and conventional farms. Combined effect of farm type, species richness, species composition and slope explained 83 % of the variation in total C storage across all farms. Organic coffee agroforestry farms may contribute to GHG mitigation and biodiversity conservation in a synergistic manner which has implications for the effective allocation of resources for conservation and climate change mitigation strategies in the agricultural sector.

There are three papers included in this thematic issue that provide unique perspectives on soil C sequestration and its interrelationship with organic matter decomposition and nutrient cycling. The first paper by Kim examined how an intercropping system with a nitrogen (N)-fixing tree (*Gliricidia*) and maize could help mitigate climate change through enhanced soil C sequestration in sub-Saharan Africa while dealing with GHG emissions from the soil. Using data from Makumba et al. (2007), the author estimated that 67.4 % of the sequestered soil C (76 Mg C/ha in 0–2 m soil depth) was lost from the system as CO_2 during the first 7 years of intercropping. An annual net gain 3.5 Mg C/ha/year was estimated from soil C sequestered and lost. The author further observed that if N_2O emission was reduced as well, the overall mitigation benefit achieved from the intercropping system would be larger. These results suggest that field measurements and modeling of CO_2, N_2O and CH_4 emissions should be taken into account while estimating C sequestration in agroforestry systems.

Gaiser et al. tested Leucaena (*Leucaena leucocephala* (L), *Senna siamea* (S) and maize (M) residue addition on soil organic matter accumulation under sub-humid tropical conditions in Benin, West Africa. On an *Imperata cylindrica* (I) dominated grass fallow, a total amount of 30 Mg/ha dry matter was applied within 18 months. Changes in the light and heavy soil organic C fraction (LF and HF) and in the total soil organic C content (LF + HF) in the topsoil were observed. All organic materials increased the proportion of the LF fraction in the soil significantly. The increase in HF was 39–51 % of the increase in total organic C, depending on the source of the organic material. The potential of the tested organic materials to increase total soil organic C content (including all soil organic C fractions) was in the order L > S > M > I, whereas the order of the HF fraction was L = S > I > M. Cation exchange capacity of the newly formed heavy soil organic C was highest with L and lowest with M. Ranking of the transformation efficiency of applied plant residues into the heavy soil organic C fraction was I > L = S > M. Transformation efficiency of the residues could neither be explained by lignin nor lignin/N ratio, but rather by extractable polyphenols. The results show that accumulation of the HF fraction in tropical soils is feasible through the application of large quantities of plant residues, but depends strongly on the quality of the organic matter added.

Zaia et al. evaluated the impact of plant litter deposition in cacao agroforestry systems on soil C, N, P and microbial biomass in Bahia, Brazil. They studied five cacao agroforestry systems of different ages under two different soils (Oxisol and Inceptisol). Overall, the average stocks of organic C, total N and total organic P for 0–50 cm soil depth were 89072, 8838 and 790 kg/ha, respectively. At this soil depth the average stock of labile organic P was 55.5 kg/ha. Microbial biomass was mostly dominated in the 0–10 cm soil depth, with a mean of microbial biomass C of 286 kg/ha, microbial biomass N of 168 kg/ha and mineralizable N of 79 kg/ha. The dynamics of organic P in these cacao agroforestry systems were not directly associated with organic C dynamics in soils, in contrast to the N dynamics.

Ecosystem models that can estimate plant and soil C stocks can be an invaluable tool for estimating C sequestration potential of agroforestry systems at larger scales. The CO_2FIX model has been used to estimate the dynamics of C stocks and flows for a variety of ecosystems around the world (Schelhaas et al. 2004). However, this model has not been tested using empirical data from agroforestry systems. Kaonga et al. tested the validity of the CO_2FIX model in predicting the aboveground and soil C stock using empirical data from 7-year-old *Leucaena* woodlots at Msekera, Zambia. They also assessed the impact of converting a degraded agricultural land to woodlots on C stocks. Measured above and belowground tree C stocks and increment of aboveground biomass differed significantly among different species. Measured stem and total aboveground tree C stocks in the *Leucaena* woodlots ranged from 17.1 to 29.2 and from 24.5 to 55.9 Mg/ha, respectively. Measured soil organic carbon (SOC) stocks at 0–200 cm depth in *Leucaena* stands ranged from 106.9 (*L. diversifolia*) to 186.0 Mg/ha (*L. leucocephala*). Although, modeled stem and branch C stocks closely matched measured stocks, the soil module of CO_2FIX could not predict the soil C accurately. The authors concluded that inadequate long-term empirical data on climate, litter quality, litter quantity, and tree growth, and the transient nature of SOC stocks that were disturbed in recent decades were most likely reasons for the inconclusive results from the model.

Udawatta and Jose synthesized the available information to estimate C sequestration under agroforestry systems in the US. They estimated that 530 Tg/year could be sequestered by four major agroforestry practices which could help offset current US emission rate of

1,600 Tg C/year from burning fossil fuel (coal, oil, and gas) by 33 %. These authors estimated C sequestration potential for silvopasture, alley cropping, and windbreaks in the US as 464, 52.4, and 8.6 Tg C/year, respectively. According to them, riparian buffers could sequester an additional 4.7 Tg C/year while protecting water quality. While acknowledging the need for accurate area estimates under different agroforestry practices in the US, they also emphasized the need for long-term data, standardized protocols for C quantification and monitoring, predictive models to understand long-term C sequestration, and site-specific agroforestry design criteria to optimize C sequestration.

The paper by Nair elaborated on the need for rigorous and consistent procedures to measure the extent of C sequestration in agroforestry systems. The author accurately pointed out that the current methods of estimating C varied widely and the estimations were based on several assumptions. According to him, large-scale global models based on such measurements and estimations were more likely to result in serious under- or overestimations of C in agroforestry practices. The author reveals several erroneous assumptions, operational inadequacies and inaccuracies commonly found in the current literature. He provides several practical recommendations for researchers that include using accurate description of the methods and procedures among others. This would help other researchers to examine the datasets and incorporate them in larger databases and help agroforestry earn its deserving place in mainstream efforts.

Estimating biomass production

Allometric equations are commonly used in estimating biomass production by trees in agroforestry systems. However, these equations are most often derived from forest grown trees that are different in their growth form from those open-grown trees in agroforestry configurations. This can introduce errors in estimating not only biomass production potential, but C sequestration as well. It is imperative that species specific allometric equations for different agroforestry practices must be developed in order to overcome this serious weakness in agroforestry research. There are two papers in this thematic issue that provide allometric equations for estimating aboveground biomass for trees in agroforestry.

Tamang et al. conducted their study in Florida, USA, to develop biomass equations for cadaghi (*Corymbia torelliana*) trees in various aged windbreaks. Trees were destructively sampled based on diameter at breast height (DBH) and crown biomass was estimated using randomized branch sampling (RBS) while trunk biomass was measured by taking disks every 1.5 m along the stem. Separate nonlinear equations were developed for crown, trunk and whole tree biomass estimation using DBH and height as predictors. The study found that DBH alone was sufficient to predict aboveground biomass while the inclusion of height provided more accurate results. Using their equation the authors recorded a total biomass per 100 m windbreak length to be between 166 and 26,605 kg. They concluded that fast-growing cadaghi could provide landowners higher returns from biomass or carbon trade to offset the cost of land occupied by the windbreaks.

Kuyah et al., on the other hand, developed new allometric equations using remotely sensed crown area and/or tree height as predictor of aboveground biomass. These equations corresponded well with the data obtained from destructive sampling with about 85 % of the observed variation in aboveground biomass explained by crown area. Addition of height and wood density as second predictor variables improved model fit by 6 and 2 % and lowered the relative error by 7 and 2 %, respectively. Total estimated aboveground biomass carbon was measured at 20.8 t C/ha, which was about 19 % more than the amount estimated using DBH as predictor. These results confirm that the new allometric equations using crown area could be a better predictor of aboveground biomass and can be used as an important tool for predicting carbon stock in such systems.

The last two papers explore biomass production potential of two temperate alley cropping systems, one from Canada and another from the US. As pointed out by Holzmueller and Jose (2012), alley cropping is one of the most suitable agroforestry practices for growing biomass for biopower and biofuels. Cardinael et al. examined short rotation willow production in the alleys of 21-year-old trees on marginal land in Guelph, Canada. As a control, the same willow clones were established on an adjacent piece of land without established trees (conventional willow system). They quantified carbon pools, fine root and leaf biomass inputs, and clone yields in both the intercropping and conventional monocropping systems. Willow biomass yield was significantly higher in the agroforestry field (4.86 odt ha^{-1}/year) compared to the conventional field (3.02 odt ha^{-1}/year). Clonal differences in biomass were also apparent with clones SV1 and SX67 with the highest yields and 9,882–41 with the lowest. Willow fine root biomass in the top 20 cm of soil was significantly higher in the intercropping system (3,062 kg/ha) than in the conventional system (2,536 kg/ha). Soil organic carbon was also significantly higher in the agroforestry field (1.94 %) than in the conventional field (1.82 %).

Susaeta et al. assessed the economics of loblolly pine (*Pinus taeda* L.)—switchgrass (*Panicum virgatum*) alley cropping in the southern US. Assuming a price range of switchgrass between \$15 and \$50 Mg^{-1} and yield of 12 Mg ha^{-1}/year, loblolly pine monoculture would be the most profitable option for landowners instead of intercropping if the price of switchgrass was below \$30 Mg^{-1}. However, when switchgrass prices were $\geq$\$30 Mg^{-1}, landowners would be financially better off adopting intercropping if competitive interaction between crops were minimal. Various assumptions were used in their analysis ranging from no competition between species for resources and reduced loblolly pine productivity due to competition with switchgrass to reduced productivity of both species due to competition for nutrients, water and light. Findings also suggested that the optimal system would depend on the competitive interactions between switchgrass and loblolly pine crops, and the expected prices for each crop.

Conclusion

Research findings from around the world have clearly demonstrated that agroforestry offers unique opportunities to increase C stocks in the terrestrial biosphere. Agroforestry could play a substantial role in reducing atmospheric concentration of CO_2 by (1) storing C in above and belowground biomass and in soil, and (2) growing biomass for biopower and biofuels and thereby replacing fossil fuel. Agroforestry could also protect existing C stocks if improved fallows and similar agroforestry practices could provide food and fuelwood, thereby reducing the rate of deforestation. Despite widespread recognition of agroforestry's potential for C sequestration and biomass production, our understanding of these topics is limited. There is still a lack of quantitative information from specific systems. This thematic issue is an attempt to bring together several original research articles from North America, South America, and Africa that investigate C sequestration and biomass production potential of specific agroforestry practices. While there are issues related to inconsistencies in methodologies, lack of soil C estimates and GHG emissions from soil, and reliable large-scale C estimates for different agroforestry practices, it is apparent that the research community is aggressively generating the much needed data at different scales. This will definitely help quantify agroforestry's role in C sequestration and biomass production and contribute meaningfully to global climate change mitigation efforts.

References

DOE (2011) US billion-ton update: biomass supply for a bioenergy and bioproducts industry. Perlack RD and Stokes BJ (Leads), ORNL/TM-2011/224. Oak Ridge National Laboratory, Oak Ridge, TN. p 227

DOE (US Department of Energy) (2003) Roadmap for agriculture biomass feedstock supply in the United States. US Department of Energy, DOE/NE-ID-1129

EIA (Energy Information Administration) (2011) Annual energy review 2010. US energy information administration, office of energy statistics, US Department of Energy, Washington, DC. http://www.eia.gov/totalenergy/data/annual/index.cfm, Accessed 20 Sep 2012

Henderson DE, Jose S (2010) Biomass production potential of three short rotation woody crop species under varying nitrogen and water availability. Agrofor Syst 80:259–273

Holzmueller EJ, Jose S (2012) Bioenergy crops in agroforestry systems: potential for the US North central region. Agrofor Syst 85:305–314

Hutchinson JJ, Campbell CA, Desjardins RL (2007) Some perspectives on carbon sequestration in agriculture. Agric For Meteorol 142:288–302

IPCC (2000) Land use, land-use change, and forestry. Cambridge University Press, Cambridge, UK, p 375 A special report of the IPCC

Jose S, Gold MA, Garrett HE (2012) The future of agroforestry in the United States. Pp. 217–246. In: Nair PKR, Garrity D (eds) Agroforestry: The future of global land use. Springer, The Netherlands, p 541

Kirby KR, Potvin C (2007) Variation in carbon storage among tree species: implications for the management of a small-scale carbon sink project. For Ecol Manag 246:208–221

Kumar BM, Nair PKR (2011) Carbon sequestration potential of agroforestry systems: opportunities and challenges. Springer, The Netherlands, p 307

Lal R (2004) Soil carbon sequestration impacts on global climate change and food security. Science 304:1623–1627

Makumba W, Akinnifesi FK, Janssen B, Oenema O (2007) Long-term impact of a gliricidiamaize intercropping system on carbon sequestration in southern Malawi. Agric Ecosyst Environ 11:237–243

Nair PKR, Nair VD, Kumar BM, Showalter JM (2010) Carbon sequestration in agroforestry systems. Adv Agron 108: 237–307

Schelhaas MJ, van Esch PW, Groen TA, de Jong BHJ, Kanninen M, Liski J, Masera O, Mohren GMJ, Nabuurs GJ, Palosuo T, Pedroni L,Vallejo A, Vilén T (2004) CO2FIX V 3.1 – A modelling framework for quantifying carbon sequestration in forest ecosystems, Wageningen, Alterra, Alterra-rapport 1068, 122 p

Sharrow SH, Ismail S (2004) Carbon and nitrogen storage in agroforests, tree plantations, and pastures in western Oregon, USA. Agrofor Syst 60:123–130

Tans P (2012) Concentrations of CO_2 in the earth's atmosphere (parts per million) derived from in situ air measurements at the Mauna Loa observatory, Hawaii. ftp://ftp.cmdl.noaa.gov/ccg/co2/trends/co2_mm_mlo.txt; Accessed 24 Sep 2012

Exhibit H

MR. YAWEN HUANG (Orcid ID : 0000-0001-8854-6005)

PROF. WEI REN (Orcid ID : 0000-0003-0271-486X)

DR. DAFENG HUI (Orcid ID : 0000-0002-5284-2897)

DR. JIAN YANG (Orcid ID : 0000-0001-9121-3308)

Article type : Primary Research Articles

Xiongxiong Bai (ORCID: 0000-0001-5104-2946)

Article type: Primary Research Articles

Responses of soil carbon sequestration to climate smart agriculture practices: A meta-analysis

Keywords

Soil organic carbon, biochar, cover crop, conservation tillage, climate, meta-analysis

Running title

Climate-smart agriculture and C sequestration

Xiongxiong Bai[1, 2#], Yawen Huang[1#], Wei Ren[1*], Mark Coyne[1], Pierre-Andre Jacinthe[3], Bo Tao[1], Dafeng Hui[4], Jian Yang[5], and Chris Matocha[1]

[1] Department of Plant and Soil Sciences, University of Kentucky, Lexington, KY 40546, USA.

[2] College of Life Sciences, Henan Normal University, Xinxiang, Henan 453007, China.

[3] Indiana University Purdue University Indianapolis, Indianapolis, IN 46202, USA.

[4] Department of Biological Sciences, Tennessee State University, Nashville, TN 37209, USA.

[5] Department of Forestry and Natural Resources, University of Kentucky, Lexington, KY 40546, USA.

[#] Xiongxiong Bai and Yawen Huang equally contribute to this work.

* Corresponding author: Wei Ren, wei.ren@uky.edu

Abstract

Climate-smart agriculture (CSA) management practices (e.g., conservation tillage, cover crops, and biochar applications) have been widely adopted to enhance soil organic carbon (SOC) sequestration and to reduce greenhouse gas emissions while ensuring crop productivity. However, current measurements regarding the influences of CSA management practices on SOC sequestration diverge widely, making it difficult to derive conclusions about individual and combined CSA management effects and bringing large uncertainties in quantifying the potential of the agricultural sector to mitigate climate change. We conducted a meta-analysis of 3,049 paired measurements from 417 peer-reviewed articles to examine the effects of three common CSA management practices on SOC sequestration as well as the environmental controlling factors. We found that, on average, biochar applications represented the most effective approach for increasing SOC content (39%), followed by cover crops (6%) and conservation tillage (5%). Further analysis suggested that the effects of CSA management practices were more pronounced in areas with relatively warmer climates or lower nitrogen fertilizer inputs. Our meta-analysis demonstrated that, through adopting CSA practices, cropland could be an improved carbon sink. We also highlight the importance of considering local environmental factors (e.g., climate and soil conditions and their combination with other management practices) in identifying appropriate CSA practices for mitigating greenhouse gas emissions while ensuring crop productivity.

1. Introduction

Soil organic carbon (SOC) is a primary indicator of soil health and plays a critical role in food production, greenhouse gas balance, and climate mitigation and adaptation (Lorenz & Lal, 2016). The dynamic of agricultural SOC is regulated by the balance between carbon inputs (e.g., crop residues and organic fertilizers) and outputs (e.g., decomposition and erosion) under long-term constant environment and management conditions. However, this balance has been dramatically altered by climate change, which is expected to enhance SOC decomposition and weaken the capacity of soil to sequester carbon (Wiesmeier *et al*., 2016). Generally, agricultural soils contain considerably less SOC than soils under natural vegetation due to land conversion and cultivation (Hassink, 1997; Poeplau & Don, 2015), with a potential to sequester carbon from the atmosphere through proper management practices (Lal, 2018). Therefore, it is crucial to seek practical approaches to enhance agricultural SOC sequestration without compromising the provision of ecosystem services such as food, fiber or other agricultural products.

Climate-smart agriculture (CSA) has been promoted as a systematic approach for developing agricultural strategies to ensure sustainable food security in the context of climate change (FAO, 2013). One of the major objectives of CSA is to reduce greenhouse gas emissions and enhance soil carbon sequestration and soil health (Campbell *et al*., 2014; Lipper *et al*., 2014). The key for sequestering more carbon in soils lies in increasing carbon inputs and reducing carbon outputs. Frequently recommended approaches for SOC sequestration include adding cover crops into the crop rotation, applying biochar to soils, and minimizing soil tillage (i.e., conservation tillage). In recent decades, these management practices have been applied in major agricultural regions globally, and a large number of observations/measurements have been accumulated (e.g., Chen *et al.*, 2009; Spokas *et al.*, 2009; Clark *et al.*, 2017).

Several mechanisms have been proposed to explain the positive effects of CSA management practices on SOC sequestration. For example, conservation tillage reduces soil disturbance and the soil organic matter decomposition rate (Salinas-Garcia *et al.*, 1997) and promotes fungal and earthworm biomass (Lavelle, 1999; Briones & Schmidt, 2017), thereby improving SOC stabilization (Liang & Balser, 2012). Cover crops provide additional biomass inputs from above- and belowground (Blanco-Canqui *et al.*, 2011), increase carbon and nitrogen inputs, and enhance the biodiversity of agroecosystems (Lal, 2004). Moreover, cover crops can promote soil aggregation and structure (Sainju *et al*., 2003), therefore indirectly reduce carbon loss from soil erosion (De Baets *et al*., 2011). Biochar amendments affect SOC dynamics through two pathways: (1) improving soil aggregation and physical protection of aggregate-associated SOC against microbial attack; (2) increasing the pool of recalcitrant organic substrates resulting in a low SOC decomposition rate and substantial negative priming (Zhang *et al.*, 2012; Du *et al.*, 2017a, Weng *et al.*, 2017).

Although these CSA management practices have been widely used to enhance soil health (e.g., Thomsen & Christensen, 2004; Denef *et al.*, 2007; Fungo *et al.*, 2017; Weng *et al.*, 2017), their effects on SOC sequestration are variable and highly dependent on experiment designs and site-specific conditions such as climate and soil properties (Poeplau & Don, 2015; Abdalla *et al.*, 2016; Liu *et al*., 2016; Paustian *et al.,* 2016). The potential to sequester soil carbon varies greatly among CSA practices, which has not been well addressed. Some studies even suggested negative effects of CSA management practices on SOC (e.g., Tian *et al*., 2005; Liang *et al*., 2007). Also, most prior quantitative research focused on the effects of a single CSA practice on SOC (e.g., Poeplau & Don, 2015; Abdalla *et al*., 2016; Liu *et al*., 2016), very few studies estimated the combined effects of diverse CSA and conventional management practices. Some recent studies reported that a combination of cover crops and conservation tillage could significantly increase SOC compared to a single management practice (Blanco-Canqui *et al*., 2013; Ashworth *et al*., 2014; Higashi *et al*., 2014; Duval *et al*., 2016). For example, Sainju *et al.* (2006) suggested that soil carbon sequestration may increase 0.267 Mg C ha^{-1} yr^{-1} under a combination of no-till and cover crop practices, where the latter was a mixed culture of hairy vetch (*Vicia villosa*) and rye (*Secale cereale*); in contrast, a carbon loss of 0.967 Mg C ha^{-1} yr^{-1} occurred when only no-till was used. Agegnehu *et al.* (2016) reported that 1.58% and 0.25% more SOC were sequestered in the mid-season and end-season, respectively, under conservation tillage when biochar was also applied. These findings highlight the importance of quantitatively evaluating the combined effects of multiple CSA management practices (including the combination of CSA and conventional management practices) on SOC sequestration under different climate and soil conditions.

This study aims to fill the above-mentioned knowledge gap through a meta-analysis to simultaneously examine the effects of three widely used CSA management practices (i.e., conservation tillage [no-till, NT; and reduced tillage, RT], cover crops, and biochar) on SOC sequestration (Fig. 1). Our scientific objectives were to: (1) evaluate and compare the effects of conservation tillage, cover crops, and biochar use on SOC; (2) examine how environmental factors (e.g., soil properties and climate) and other agronomic practices (e.g., nitrogen fertilization, residue management, irrigation, and crop rotation) influence SOC in these CSA management environments.

[Insert Figure 1]

2. Materials and methodology

2.1. Data collection

We extracted data from 417 peer-reviewed articles (297 for conservation tillage, 64 for cover crops, and 56 for biochar) published from 1990 to May 2017 (Data S1). Among all publications, 113 for conservation tillage, 32 for cover crops, and 7 for biochar were conducted in the U.S. All articles were identified from the Web of Science. The search keywords were "soil organic carbon" and "tillage" for conservation tillage treatments; "soil organic carbon" and "cover crop" for cover crop treatments; and "soil organic carbon" and "biochar" for biochar treatments. All selected studies meet the following inclusion criteria: (1) SOC was measured in field experiments (to estimate the potential of biochar to increase soil carbon, we also included soil incubation and pot experiments with regard to biochar use); (2) observations were conducted on croplands excluding orchards and pastures; (3) ancillary information was provided, such as experiment duration, replication, and sampling depth; and (4) other agronomic management practices were included besides the three target management practices in this study. We considered conventional tillage as the control for NT and RT. Experiments that eliminated any tillage operation were grouped into the NT category, and experiments using tillage with lower frequency or shallower till-depth or less soil disturbance in comparison to the paired conventional tillage (e.g., moldboard plow and chisel plow) were grouped into the RT category. Likewise, "no cover crop" and "no biochar" were treated as control experiments relative to cover crop and biochar treatments, respectively. We only considered studies that viewed cover crops as treatments and fallow (or weeds) as controls.

Soil organic carbon data were either derived from tables or extracted from figures using the GetData Graph Digitizer software v2.26 (http://getdata-graph-digitizer.com/download.php). Other related information from the selected studies was also recorded, including location (i.e., longitude and latitude), experiment duration, climate (mean annual air temperature and precipitation), soil properties (texture, depth, and pH), and other agronomic practices (crop residues, nitrogen fertilization, irrigation, and crop rotation). The study durations were grouped into three categories: short (≤5 years), medium (6-20 years), and long term (>20 years). Climate was grouped according to the aridity index published by UNEP (1997) as either arid (≤ 0.65) or humid (> 0.65). Study sites were grouped into cool (temperate and Mediterranean climates) and warm zones (semitropical and tropical climates) (Shi *et al.*, 2010). Soil texture was grouped as silt loam, sandy loam, clay and clay loam, loam, silty clay and silty clay loam, and loamy sand according to the USDA soil texture triangle. Soil depth was grouped as 0-10 cm, 10-20 cm, 20-50 cm, and 50-100 cm. Soil pH was grouped as acidic (< 6.6), neutral (6.6-7.3), and alkaline (> 7.3). Crop residue management was grouped as "residue returned" and "residue removed." We only included those studies that used the same residue management in the control and treatment groups. Similarly, nitrogen fertilization was grouped into no addition, low (1-100 kg N ha^{-1}), medium (101-200), and high levels (> 200). Irrigation management was grouped as irrigated or rainfed. Crop sequence was grouped as rotational or continuous crops (including crop-fallow systems). We also estimated the response of SOC in the whole-soil profiles (from the soil surface to 120 cm, with an interval of 10 cm) to CSA management practices.

The standard deviation (SD) of selected variables, an important input variable to the meta-analysis, was computed as SD = SE$\times\sqrt{n}$, where SE is the standard error and *n* is the number of observational replications. If the results of a study were reported without SD or SE, SD was calculated based on the average coefficient of variation for the known data. Publication bias was analyzed by the method of fail-safe number, which suggests that the meta-analysis can be considered robust if the fail-safe number is larger than 5*k+10 (where k is the number of observed studies) (Rothstein *et al.*, 2006).

2.2. Meta-analysis

A random-effect model of meta-analysis was used to explore environmental and management variables that might explain the response of SOC to CSA management practices. The data analysis was performed in R (R Development Core Team 2009). The response ratio (RR) was defined as the ratio between the outcome of CSA management practices and that of the control group. The logarithm of RR ($\ln RR$) was calculated as the effect size of each observation (Hedges *et al.*, 1999, Equation (1)):

$$\ln RR = \ln\left(\overline{X_t}/\overline{X_c}\right) = \ln \overline{X_t} - \ln \overline{X_c} \quad (1)$$

where $\overline{X_t}$ and $\overline{X_c}$ are SOC values in the treatment and control groups, respectively. The variance (v) of $\ln RR$ was computed as:

$$\nu = \frac{S_t^2}{n_t \overline{X_t^2}} + \frac{S_c^2}{n_c \overline{X_c^2}} \quad (2)$$

where S_t and S_c are the standard deviations of the treatment and control groups, respectively, while n_t and n_c are the sample sizes of the treatment and control, respectively.

The weighting factor (w), as the inverse of the variance, was computed for each observation to obtain a final weighting factor (w'), which was then used to calculate the mean effect size (RR_{++}). The equations were:

$$w = 1 / \nu \quad (3)$$

$$w' = w / n \quad (4)$$

$$RR_{++} = \frac{\sum_i \ln RR'_i}{\sum_i w'_i} \quad (5)$$

where $\ln RR' = w' \ln RR$ is the weighted effect size, n is the total number of observations per study, and i is the ith observation.

The 95% confidence intervals (CI) of $\ln RR_{++}$ were computed to determine statistical significance. The comparison between treatment and control was considered significant if the 95% CIs did not overlap zero (vertical lines in the graphs). The percent change was transformed [($e^{RR_{++}}$-1) ×100%] to explain the response of the estimated CSA management practices.

3. Results

3.1 SOC responses to conservation tillage, cover crops, and biochar

Biochar applications enhanced SOC storage by 39% (28% in the field and 57% in incubation and pot experiments, Fig. S1), representing the most effective practice, followed by cover crops (6%) and conservation tillage (5%) (Fig. 2). Cover crop species had a pronounced positive effect on SOC sequestration (Fig. S1), ranging from 4% for non-leguminous cover crops to 9% for leguminous cover crops. When investigating different types of conservation tillage, NT and RT had similar effects on SOC (approximately 8% increase). All results were statistically significant (Fig. 2). Theoretically, the combination of CSA management practices may result in greater or lesser effects on soil sequestration compared to single CSA management practice. However, if synergistic effects were the prevalent interactions, this combination might potentially enhance carbon accumulation (e.g., over 50% increase in SOC), which is subject to further investigation in field experiments. Across the whole dataset we compiled, the SOC varied widely in each CSA treatment (Fig. S2). We calculated the distribution of the data points (the ratio of SOC of each treatment to that of the corresponding control, i.e., NT/RT vs. conventional tillage, cover crops vs. no cover crop, and biochar use vs. non-biochar; Fig. S2). Most of the studies used in this meta-analysis reported positive responses of SOC to NT, RT, cover crops, and biochar treatment (60%, 65%, 68%, and 91%, respectively). The SOC change rates were 0.38±0.71 Mg ha^{-1} yr^{-1} (n=56) and -0.29±0.79 Mg ha^{-1} yr^{-1} (n=30) in NT and RT systems, respectively (Fig. S3). We did not calculate SOC sequestration rates for other treatments (i.e., cover crops and biochar) due to the lack of some ancillary information (e.g., bulk density).

[Insert Figure 2]

3.2 Effects of CSA management practices in different climate zones

Overall, CSA management practices sequestered more SOC in arid areas than in humid areas (Fig. 3a). Biochar and cover crops increased 12% (38% vs. 26%) and 3% (9% vs. 6%) more SOC in arid areas, respectively, compared to humid areas. In comparison, the NT-induced SOC uptake was slightly higher in arid areas than that in humid areas (9% and 8%, respectively). However, the RT-induced SOC increment in arid areas was two times greater than that in humid areas. Our further analysis suggested that CSA management practices significantly increased SOC in both cool and warm climate zones with diverse responses (Fig. 3b). For example, in warm areas, biochar applications only increased SOC by half of the enhancement observed in cool areas. Cover crops increased SOC by 15% in warm areas, three times larger than that in cool areas. In warm areas, NT increased SOC by 15% compared to 8% in cool areas. Reduced tillage increased SOC by 7% and 6% in warm and cool areas, respectively.

[Insert Figure 3]

3.3 Effects of CSA management practices with different soil properties

The effects of CSA management practices on SOC were strongly influenced by soil texture (Fig. 4). Biochar applications increased SOC by 63, 62%, and 52% in silty clay and silty clay loam soils, loam soils, and loamy sand soils, respectively. While relatively lower soil carbon uptakes under biochar applications were found in clay loam and clay soils (32%), silt loam soils (35%), and sandy loam soils (34%). Cover crops increased SOC by 4%, 6%, 7%, and 6% in clay loam and clay soils, silt loam soils, loam soils, and sandy loam soils, respectively. No-till increased SOC by 16% in silty clay and silty clay loam soils, compared to 12% in sandy loam soils and 7% in loamy sand soils. Reduced tillage increased SOC by 21%, 7%, and 15% in silty clay and silty clay loam soils, loam soils, and loamy sand soils, respectively. Overall, cover crops sequestered more carbon in coarse-textured soils than in fine-textured soils. In contrast, NT and RT increased SOC more in fine-textured soils than in coarse-textured soils. No obvious relationship was found between biochar use and soil textures.

[Insert Figure 4]

The positive effects of CSA management practices on SOC decreased with soil depth (Fig. 5). Biochar significantly increased SOC by 41% and 14% in the 0-10 cm and 0-30 cm soil layers, respectively (Table S1). Cover crops significantly increased SOC by 9%, 3%, and 9% in the 0-10 cm, 10-20 cm, and 20-50 cm depth ranges, respectively. Further analysis showed that cover crops could increase SOC (5%) in the entire 0-70 cm soil profile (Table S1). Both NT and RT could significantly increase SOC most at 0-10 cm depth (22% and 17%, respectively). Although reduced SOC was observed in the 10-20 cm and 20-50 cm soil layers (-4% and -10%, respectively), NT could still enhance SOC sequestration in the entire soil profile up to 120 cm (Table S1). In comparison, RT could increase SOC in the 0-70 cm soil profile (Table S1) although decreased soil carbon (not statistically significant) was observed in the 10-50 cm soil layer (Fig. 5).

[Insert Figure 5]

All CSA management practices except RT positively influenced the SOC pool regardless of soil pH. The management-induced SOC uptake was generally higher in alkaline soils than in acid soils (Fig. 6). Biochar use increased SOC by 65%, 35%, and 28% in alkaline, neutral, and acid soils, respectively. Cover crops increased SOC by 15% in neutral soils, followed by alkaline (9%) and acid soils (6%). No-till increased SOC by 6% in acid soils and 13% in alkaline soils. The SOC increased by RT was greater in alkaline soils (9%) than acid soils (6%), but RT had no significant influence on SOC in neutral soils.

[Insert Figure 6]

3.4 Combined effects of experiment duration and other agronomic practices

The CSA management practices are generally applied together with other agronomic practices such as residue return, nitrogen fertilizer use, and irrigation. These agronomic practices may interact with the CSA management practices with positive or negative effects on the capacity of soils to sequester carbon. In this study, we considered experiment duration and four other agronomic practices, including residue return, nitrogen fertilization, irrigation, and crop sequence, to quantify these effects.

Our results demonstrated that the influences of three CSA management practices on SOC varied with experiment duration. Biochar amendments significantly increased SOC by 45% and 36% in short-term and medium-term experiments, respectively. Cover crops significantly increased SOC by 5%, 11%, and 20% in the short-term, medium-term, and long-term experiments, respectively (Fig. 7). No-till significantly increased SOC by 13% in the long-term experiments, followed by medium-term (7%) and short-term (6%). Reduced tillage increased SOC by 12% in long-term studies, followed by medium-term (9%) and short-term experiments (3%). The average durations differed in each group (Table S2), which may influence the effect of CSA management practices on SOC. When excluding short and medium experiment durations (≤ 20 years) and shallow sampling (< 20 cm), RT significantly increased SOC by 14%, while NT had no significant effect on SOC (Fig. S4).

[Insert Figure 7]

When crop residues were returned, conservation tillage and cover crops significantly increased SOC: 9% for NT, 6% for cover crops, and 5% for RT (Fig. 8). However, if crop residues were removed, neither cover crops nor RT had a significant effect on SOC, although there was a significant increase in SOC under NT (5%).

[Insert Figure 8]

Our results suggested that nitrogen fertilizer use could alter the magnitude of soil carbon uptake induced by CSA management practices. Biochar boosted the most SOC among CSA management practices regardless of nitrogen fertilizer levels, with the strongest effects under the low-level nitrogen inputs, followed by the high-level (38%), medium-level (29%), and no nitrogen fertilizer use (27%) (Fig. 9). Cover crops increased SOC by 6% under both low-level and medium-level nitrogen inputs, slightly higher than that under the high-level nitrogen fertilizer use (3%). No-till tended to sequester more soil carbon when nitrogen fertilizer input was relatively lower (11%, 8%, and 6% for low-level, medium-level, and high-level nitrogen fertilization, respectively). While RT increased SOC by 13% at the medium-level nitrogen fertilizer rate, approximately two times larger than those under the low-level and high-level nitrogen fertilizer use (Fig. 9).

[Insert Figure 9]

When investigating the irrigation effects, our results suggested that biochar markedly stimulated SOC increases in irrigated croplands (49%), three times higher than those under rainfed condition. Similarly, NT increased SOC by 15% in irrigated croplands, twice as much soil carbon as that in rainfed croplands. Cover crops increased SOC by 7% and 4% in irrigated and rainfed croplands, respectively. In contrast, the RT-induced SOC increase was 16% under the rainfed condition, 5% higher than that in irrigated croplands (Fig. 10a).

The CSA management practices significantly promoted SOC uptakes in both rotational and continuous cropping systems (Fig. 10b). Specifically, biochar amendments enhanced SOC by 52% in rotational cropping systems, much higher than that in the continuous cropping system (31%). While SOC uptakes induced by NT and RT showed no obvious differences in the rotational and continuous cropping systems (9% and 8% vs. 8% and 7%). Cover crops increased SOC by 4% in rotational cropping systems, lower than that in continuous cropping systems (8%).

[Insert Figure 10]

3.5 Combinations of CSA management practices

Our results demonstrated that combining different CSA management practices might significantly enhance SOC sequestration. In warm regions, SOC increased by 13% with the combination of conservation tillage and cover crops (Fig. 11). In loamy sand and sandy clay loam soils, associated SOC uptakes increased to 31% and 21%, respectively. A similar effect was also observed in medium-term experiments. However, in clay soils, the combination of cover crops and conservation tillage significantly decreased SOC by 19%.

[Insert Figure 11]

4. Discussion

4.1 Effects of CSA management practices on SOC

Common approaches for enhancing SOC focus on increasing carbon inputs, decreasing losses, or simultaneously affecting both inputs and losses. All CSA management practices discussed here, i.e., biochar, cover crops, and conservation tillage, increase soil carbon sequestration to different extents. For example, SOC enhancement by biochar applications can reach up to 40% (Liu *et al.*, 2016), while conservation tillage and cover crops increase SOC by only 3-10% (Luo *et al.*, 2010; Abdalla *et al.*, 2016; Du *et al.*, 2017b; Zhao *et al.*, 2017) and ~10% (Aguilera *et al.*, 2013), respectively. Our results agree with these earlier findings: biochar use increased SOC by 39%, followed by cover crops (6%) and conservation tillage (5%). The discrepancies among various CSA management practices in enhancing SOC fundamentally lie in their functional mechanisms. Biochar addition, with a low turnover rate, contributes directly to soil carbon storage and indirectly decreases native SOC decomposition rates by negative priming (Wang *et al.*, 2016). Cover crops are green manure that increases carbon inputs to the soil and subsequent SOC (Poeplau & Don, 2015). Conservation tillage practices may not necessarily add carbon; their contribution is primarily accomplished by protecting SOC from decomposition and erosion (Six *et al.*, 2000; Lal, 2005). Additionally, all three CSA management practices can potentially improve soil properties, thereby stimulating more carbon inputs from residue return and rhizodeposition due to promoted plant growth, and reducing carbon losses via decreasing leaching and erosion. However, the effectiveness of these practices on SOC sequestration and the mechanisms involved vary with environmental factors and other agronomic practices.

4.2 Environmental control in CSA management practices

Environmental factors such as climate and soil properties may influence carbon inputs to the soil and affect the processes that regulate carbon loss, considering that all CSA practices are implemented in site-specific climate and soil conditions. The effects of CSA management practices on SOC could be biased by environmental factors.

4.2.1 Climate variability

Climate is one of the major driving forces that regulate SOC distribution. On average, SOC accumulation is greater than decomposition in wet areas than in dry and warm regions (Jobbágy & Jackson, 2000). Soil carbon is positively related to precipitation and negatively correlated with temperature (Rusco *et al.*, 2001), with the former correlation tending to be stronger (Martin *et al.*, 2011; Meersmans *et al.*, 2011). High precipitation is usually associated with abundant growth and high rates of carbon inputs to soils (Luo *et al.*, 2017), while low temperatures may remarkably reduce microbial activity, resulting in low rates of organic matter decomposition and measurable amounts of SOC accumulation (Castro *et al.*, 1995; Garcia *et al.*, 2018). Biochar applications result in greater SOC accumulation in arid/cool areas than in humid/warm environments (Fig. 3), probably due to the porous structure and the capacity of biochar to promote greater soil water retention (Karhu *et al.*, 2011; Abel *et al.*, 2013). It is not clear why biochar has a greater impact on SOC accrual in cool regions. A possible explanation is that high soil temperatures may promote biochar decomposition and oxidation (Cheng *et al.*, 2008).

Cover crops and NT increased SOC with no significant difference between aridity conditions (Table 1), although they performed better at storing SOC in arid areas (Fig. 3a). This result suggests that arid-region soils have a high potential to store carbon when using proper management practices (Tondoh *et al.*, 2016). In addition, cover crops and NT can enhance carbon sequestration more in warm areas than in cool areas. Temperature could affect the establishment and growth of cover crops (Akemo *et al.*, 2000). In warm areas, cover crops may develop well and potentially capture more carbon dioxide (CO_2) from the atmosphere, thus providing more carbon inputs into soils after they die (e.g., Bayer *et al.*, 2009).

Tillage results in the breakdown of macroaggregates and the release of aggregate-protected SOC (Six *et al.*, 2000; Mikha & Rice, 2004). Tillage-induced SOC decomposition usually proceeds at higher rates in warm than in cool areas. Implementing NT, with minimal soil disturbance, protects SOC from decomposition. As a result, SOC increases can be more significant in warm conditions considering the relatively higher baseline of the decomposition rate compared to that in cool areas.

[Insert Table 1]

4.2.2 Soil properties

Soil organic carbon is strongly correlated with clay content, with an increasing trend toward more SOC in fine-textured soils (Stronkhorst & Venter, 2008; Meersmans *et al.*, 2012). The SOC mineralization rate probably diminishes as clay concentrations increase (Sainju *et al.*, 2002). Clay minerals can stabilize SOC against microbial attack through absorption of organic molecules (Ladd *et al.*, 1996). By binding organic matter, clay particles help form and stabilize soil aggregates, imposing a physical barrier between decomposer microflora and organic substrates and limiting water and oxygen available for decomposition (Dominy *et al.*, 2002).

Biochar use and cover crops promote carbon sequestration for all soil texture types. Such an enhancement of SOC does not vary significantly with soil texture (Table 1). The ability of conservation tillage to enhance SOC, however, differs with soil texture (Fig. 4). Conservation tillage merely reduces soil disturbance and normally does not add extra materials to soils. It can be inferred that the effect of conservation tillage on SOC is more texture-dependent than the other two management practices. Biochar is a carbon-rich material with a charged surface, organic functional groups, and a porous structure, which can potentially increase soil aggregation and cation exchange capacity (Jien & Wang, 2013). Similarly, cover crops directly provide carbon inputs to soils, and their root development and rhizodeposition can also benefit soil structure. These benefits are embedded in the source of biochar and cover crops *per se*. Thus, the effectiveness of biochar and cover crops in increasing SOC may depend on their properties other than soil texture.

Soil depth may potentially influence the effects of the CSA practices on SOC (Baker *et al.*, 2007). The CSA practices were most beneficial to SOC accumulation in surface soils. For example, NT increased SOC by 7% in the 0-3 cm soil layer (Abdalla *et al.*, 2016) and by 3% at the 40 cm depth (Luo *et al.*, 2010). Our findings suggested that CSA practices can enhance SOC sequestration in the entire soil profile, although the positive effects vary with soil depths (Table S1). Conventional tillage breaks soil aggregates and increases aeration and thus enhances soil organic matter mineralization (Cambardella & Elliott, 1993). Conventional tillage also incorporates residues into deeper soil layers, resulting in a more uniform distribution of SOC (albeit at lower concentrations) in the soil profile (Sainju *et al.*, 2006; Plaza-Bonilla *et al.*, 2010). In contrast, conservation tillage keeps residues at the soil surface and reduces their degree of incorporation into soil (Franzluebbers *et al.*, 1995). Nevertheless, positive effects of NT on SOC have been found in a deep soil profile (0-60 cm, Liu *et al.*, 2014). As noted, in the 10-50 cm soil layer, the effect of cover crops on SOC was found to be the greatest among all the CSA management practices we discussed (Fig. 5). This is perhaps because much of the crop and cover crop root growth occurs in the surface soil (e.g., Box & Ramsuer, 1993; Sainju *et al.*, 1998) and the generally greater contribution of roots to SOC than aboveground biomass (Balesdent & Balabane, 1996; Allmaras *et al.*, 2004).

Soil pH is recognized as a dominant factor governing the soil organic matter turnover rate, although its mode of impact is still unclear (Van Bergen *et al.*, 1998). Soil pH affects selective presentation or metabolic modification of specific components (e.g., lignin-cellulose, lipids) during decomposition (Kemmitt *et al.*, 2006) and therefore abiotic factors (e.g., carbon and nutrient availability) and biotic factors (e.g., the composition of the microbial community). Also, soil pH can change the decomposition rate of crop residues and SOC via its effect on SOC solubility and indirectly by altering microbial growth, activity, and community structure (Pietri & Brookes, 2009; Wang *et al.*, 2017). The levels of soluble organic carbon may increase with increasing acidity (Willett *et al.*, 2004; Kemmitt *et al.*, 2006). Motavalli *et al.* (1995) suggested that increased soil acidity would cause greater soil organic matter accumulation due to reduced microbial mineralization; however, this was challenged by Kemmitt *et al.* (2006) who found no significant trend in SOC in response to pH changes. In this study, most CSA management practices resulted in greater increases in SOC in neutral or alkaline soils compared to acid soils.

4.3 CSA and other agronomic practices

Crop residues provide substantial amounts of organic matter and may influence the effect of CSA practices on SOC. Residue retention changes the formation of soil macroaggregates (Benbi & Senapati, 2010), promoting SOC preservation and accumulation (Six *et al.*, 2002). Residue cover protects the soil surface from direct impact by raindrops (Blanco-Canqui *et al.*, 2014). In addition, crop residues provide organic substrates to soil microorganisms that can produce binding agents and promote soil aggregation (Guggenberger *et al.*, 1999). Conversely, residue removal reduces carbon input to the soil system and ultimately decreases SOC storage (Manna *et al.*, 2005; Koga & Tsuji, 2009). This suggests that the amount of carbon inputs predominantly controls changes in SOC stocks (Virto *et al.*, 2012). For the conditions of cover crops and NT, enhancing SOC was significantly greater with residue return than with residue removal. Our study suggests that changes in SOC did not differ with residue management in RT (Table 1), although a slightly greater increase in SOC occurred with residue retention than with residue removal (Fig. 8). This unexpected result is likely due to the limited number of observations with residue removal. Another possible reason is that the interaction between residue management and soil type may lead to various responses in SOC stocks. For example, residue removal increased SOC by 3.6% while residue retention had no effect on SOC in clay and clay loam soils. The decomposition of crop residues involves complex processes, which are controlled by multiple biogeochemical and biophysical conditions.

Nitrogen fertilization noticeably increases SOC stock but with diminishing returns. For example, Blanco-Canqui *et al.* (2014) indicate that nitrogen fertilizer increases SOC when the nitrogen fertilization rate is below 80 kg N ha^{-1}, above which it reduces aggregation and then decreases SOC stocks. Nitrogen fertilization can stimulate biological activity by altering carbon/nitrogen ratios, thereby promoting soil respiration and decreasing SOC content (Mulvaney *et al.*, 2009); however, excessive nitrogen addition may reduce soil fungi populations, inhibit soil enzyme activity, and decrease CO_2 emissions (Wilson & Al Kazi, 2008). These findings suggest that nitrogen fertilization enhances the positive effect of CSA management practices on SOC, likely through increased plant biomass production (Gregorich *et al.*, 1996). However, nitrogen addition complicates the effects of biochar on SOC (Fig. 9). Nitrogen fertilizer may affect biochar stability and the response of native SOC decomposition to biochar addition (Jiang *et al.*, 2016). Positive (Bebber *et al.*, 2011; Jiang *et al.*, 2014) and negative (Pregitzer *et al.*, 2008) effects of nitrogen on SOC mineralization rates have been reported. These contrasting effects could be an alleviation of microbial nitrogen limitations (Jiang *et al.*, 2016) and changes in the microbial decomposer community toward more efficient carbon-users (Janssens *et al.*, 2010). A possible explanation of the various responses of nitrogen rate in biochar-modified soils is that either inadequate or excessive nitrogen addition may inhibit microbial activity to some extent, whereas medium-level nitrogen fertilization rates benefit microbes the most, which needs to be confirmed in future research.

Aridity can limit plant growth and crop residue return and ultimately compromise SOC accumulation (Moreno *et al.*, 2006). Jien and Wang (2013) suggest that CSA management practices can potentially enhance soil water retention by improving soil porosity and erosion control. Irrigation ensures sufficient water for plant growth, resulting in more biomass production than in rainfed conditions (Shipitalo *et al.*, 1990; Chan, 2004; Capowiez *et al.*, 2009; Swanepoel *et al.*, 2016). The crop root density is much higher in irrigated conditions compared to rainfed conditions (Jobbágy & Jackson, 2000), leading to higher organic matter input. Thus, CSA management practices in combination with irrigation could further increase SOC content.

Rotational cropping potentially provides high carbon input to soils. Compared to continuous cropping systems, crops in rotational cropping systems have a greater belowground allocation of biomass (Van Eerd *et al.*, 2014), resulting in more inputs of crop residue to the soil system. Enhancing rotation complexity can benefit carbon sequestration (West & Post, 2002). The present analysis suggests that all CSA practices can prominently increase SOC sequestration regardless of the crop rotation system. Biochar addition increased SOC more in rotational cropping systems than in continuous cropping systems, while cover crops increased SOC more in continuous systems (Fig. 10). This is likely because cover crops increased the diversity of the original continuous systems, resulting in larger percentage changes in SOC content compared to rotational systems. Cover crop species introduce large uncertainties because the quantity and quality of cover crop residues may vary greatly with species. Residues with a high carbon/nitrogen ratio probably increase the amount of SOC (Duong *et al.*, 2009). The growth period of legume cover crops may be longer in continuous than in rotational cropping systems, thus providing more organic matter and nitrogen input to the soil. Ultimately, these processes would increase SOC stocks.

The effect size of combined cover crops and conservation tillage was generally less than 11% (the sum of the effect size of cover crops and conservation tillage). However, in sandy clay loam and loamy sand soils, the sum of the effect size was 21% and 31%, respectively. Coarse-textured soils are not carbon-saturated and have great potential for carbon uptake. Cultivated land tends to suffer from SOC degradation, and SOC accumulation could quickly increase upon initiating farming practices due to high carbon inputs to the soil system (Vieira *et al.*, 2009). For example, in sandy loam soils, Higashi *et al.* (2014) showed that SOC increased by 22% with a combination of cover crops and NT. These results may be attributed to the stability of soil water-stable aggregates when cover crops are grown in sandy clay loam soils (McVay *et al.*, 1989), given that aggregate stability has been linked to protection of SOC from mineralization (Unger, 1997). The combination of cover crops and conservation tillage significantly decreased SOC in clay soils. The reason for this unexpected result may be due to the limited number of study sites where this combination of treatments was evaluated (few data points in our meta-analysis) but also to the diverse methods (e.g., burning) by which the cover crop biomass was managed (Tian *et al.*, 2005).

4.4 Uncertainty analysis and prospects

Our meta-analysis, based on 3,049-paired comparisons from 417 peer-reviewed articles, quantitatively analyzed SOC changes as influenced by major CSA management practices and associated environmental factors and other agronomic practices. The publication bias analysis suggested that most results in this study are robust (Table S3). The accuracy and robustness of metadata analysis depend highly on both the data quality and quantity. A detailed statement of the experimental conditions will provide more information for in-depth analysis. Future CSA research also requires standardized field management, for example, the definitions and names of different conservation tillage methods should be uniform across studies to facilitate classification research.

To the best of our knowledge, this study made the first attempt to examine synergistic effects when two or more CSA management practices are used together. Although our results present the positive effects of CSA management on soil carbon storage, especially when multiple management practices are adopted collectively, each practice may have constraints regarding enhancing soil carbon sequestration. The SOC benefit of CSA management practices strongly depends on environmental factors and other agronomic practices. Therefore, the choice of proper practices is potentially highly region-specific. Our results imply that CSA may have great potential for climate change mitigation as the combination of conservation tillage, cover crops, and biochar can theoretically enhance SOC by 50%. However, field experiments are still needed to support this claim. In addition, some CSA management practices may promote nitrous oxide or methane emissions (e.g., Six *et al.*, 2004; Spokas & Reicosky, 2009; Kessel *et al.*, 2013; Huang *et al.,* 2018), which, to some extent, would offset their benefit on climate change mitigation. Therefore, evaluating the CSA effects should also include non-CO_2 greenhouse gases such as nitrous oxide and methane. We call for field experiments that can fully examine key indicators (such as soil carbon and greenhouse gases) in response to single and combined CSA management practices.

Additionally, incorporating cover crops into current cropping systems could potentially alter conventional rotations. For example, cover crops in herbaceous crop rotations can substitute bare fallows or commercial crops. We only considered studies that treated cover crops as treatments and fallow (or weeds) as controls in this study. In comparison to bare fallows, cover crops can enhance soil health and quality (Jarecki & Lal, 2003). The benefits of cover crops include uptakes and stores of soil nutrients between seasons when they are susceptible to leaching (Doran & Smith, 1987). However, the substitution of commercial crops could reduce the productivity of the system, which has climatic implications related to the opportunity cost of the extra land required (e.g., Balmford *et al.*, 2018; Searchinger *et al.*, 2018). Thus, future studies should further address these potential side effects caused by land use change.

Materials producing biochar may have other uses or fates, and the biochar-making processes may produce CO_2 (e.g., Llorach-Massana *et al.*, 2017), although biochar addition is an effective way to sequester SOC. These uncertainties, to some extent, can offset the benefits of biochar for climate change mitigation through SOC sequestration (Powlson *et al.*, 2008). The carbon footprint of biochar production depends on production technology and the types of feedstocks (Meyer *et al.*, 2017). Mukherjee and Lal (2014) found that "carbon dioxide emissions from biochar-amended soils have been enhanced up to 61% compared with unamended soils." However, with a low carbon footprint, each ton of biochar could sequester 21 to 155 kg of equivalent CO_2 (Llorach-Massana *et al.*, 2017). Matovic (2011) also suggested that 4.8 Gt C yr^{-1} would be sequestered if 10% of the world's net primary production were converted into biochar, "at 50% yield and 30% energy from volatiles." To fully understand the net impacts of biochar on climate mitigation, future studies should stress the carbon footprint in the lifecycle of biochar.

It is essential to realistically examine the effects of CSA management practices on SOC and greenhouse gases at multiple scales from plot and field levels to regional and global scales. Therefore, future CSA research is expected to include varied climate and geographic conditions, address more biogeochemical and hydrological processes, and apply diverse methods such as the data-model fusion approach. For example, modeling studies have attempted to investigate regional cropland SOC dynamics as influenced by multiple global environmental changes while considering more traditional and less CSA practices (e.g., Molina *et al.,* 2017; Nash *et al.,* 2018; Ren *et al.,* 2012, 2018). In the future, ecosystem models need to be improved to incorporate multiple common CSA management practices. Additional model evaluations are needed to quantify the potential of cropland carbon sequestration by adopting multiple CSA practices at broad scales as new data become available from suggested field experiments and observations.

Acknowledgements

This work was supported by the National Institute of Food and Agriculture, U.S. Department of Agriculture (NIFA-USDA Hatch project 2352437000) and the National Science Foundation under Cooperative Agreement No.1355438. Bai acknowledges the support of the China Scholarship Council. We thank Elisa D'Angelo for the comments and suggestions on the manuscript. We also thank three anonymous reviewers and editor for comments that improved this manuscript. The authors declare no conflict of interest.

References

Abdalla, K., Chivenge, P., Ciais, P., & Chaplot, V. (2016). No-tillage lessens soil CO_2 emissions the most under arid and sandy soil conditions: results from a meta-analysis. *Biogeosciences*, 13(12), 3619-3633. doi: 10.5194/bg-13-3619-2016

Abel, S., Peters, A., Trinks, S., Schonsky, H., Facklam, M., & Wessolek, G. (2013). Impact of biochar and hydrochar addition on water retention and water repellency of sandy soil. *Geoderma*, 202-203(2), 183-191.

Agegnehu, G., Bass, A. M., Nelson, P. N., & Bird, M. I. (2016). Benefits of biochar, compost and biochar-compost for soil quality, maize yield and greenhouse gas emissions in a tropical agricultural soil. *Science of The Total Environment*, 543, 295-306. doi: 10.1016/j.scitotenv.2015.11.054

Aguilera, E., Lassaletta, L., Gattinger, A., & Gimeno, B. S. (2013). Managing soil carbon for climate change mitigation and adaptation in Mediterranean cropping systems: a meta-analysis. *Agriculture Ecosystems & Environment*, 168, 25-36. doi: 10.1016/j.agee.2013.02.003

Akemo, M. C., Regnier, E. E., & Bennett, M. A. (2000). Weed suppression in spring-sown rye (*Secale cereale*)–pea (*Pisum sativum*) cover crop mixes. *Weed Technology*, 14(3), 545-549.

Allmaras, R. R., Linden, D. R., & Clapp, C. (2004). Corn-residue transformations into root and soil carbon as related to nitrogen, tillage, and stover management. *Soil Science Society of America Journal*, 68(4), 1366-1375.

Ashworth, A. J., Allen, F. L., Wight, J. P., Saxton, A. M., & Tyler, D. D. (2014). Long-term Soil organic carbon changes as affected by crop rotation and bio-covers in no-till crop systems. In A. E. Hartemink & K. McSweeney (Eds.), *Soil Carbon* (pp. 271-279).

Baker, J. M., Ochsner, T. E., Venterea, R. T., & Griffis, T. J. (2007). Tillage and soil carbon sequestration—What do we really know? *Agriculture Ecosystems & Environment*, 118(1), 1-5.

Balesdent, J., & Balabane, M. (1996). Major contribution of roots to soil carbon storage inferred from maize cultivated soils. *Soil Biology & Biochemistry*, 28(9), 1261-1263.

Balmford, A., Amano, T., Bartlett, H., Chadwick, D., Collins, A., Edwards, D., . . . Smith, P. (2018). The environmental costs and benefits of high-yield farming. *Nature Sustainability*, 1(9), 477.

Bayer, C., Dieckow, J., Amado, T. J. C., Eltz, F. L. F., & Vieira, F. C. B. (2009). Cover crop effects increasing carbon storage in a subtropical no-till sandy Acrisol. *Communications in Soil Science and Plant Analysis*, 40(9-10), 1499-1511.

Bebber, D. P., Watkinson, S. C., Boddy, L., & Darrah, P. R. (2011). Simulated nitrogen deposition affects wood decomposition by cord-forming fungi. *Oecologia*, 167(4), 1177-1184.

Benbi, D. K., & Senapati, N. (2010). Soil aggregation and carbon and nitrogen stabilization in relation to residue and manure application in rice–wheat systems in northwest India. *Nutrient Cycling in Agroecosystems*, 87(2), 233-247.

Blanco-Canqui, H., Ferguson, R. B., Shapiro, C. A., Drijber, R. A., & Walters, D. T. (2014). Does inorganic nitrogen fertilization improve soil aggregation? Insights from two long-term tillage experiments. *Journal of Environmental Quality*, 43(3), 995-1003. doi: 10.2134/jeq2013.10.0431

Blanco-Canqui, H., Holman, J. D., Schlegel, A. J., Tatarko, J., & Shaver, T. M. (2013). Replacing fallow with cover crops in a semiarid soil: effects on soil properties. *Soil Science Society of America Journal*, 77(3), 1026-1034. doi: 10.2136/sssaj2013.01.0006

Blanco-Canqui, H., Mikha, M. M., Presley, D. R., & Claassen, M. M. (2011). Addition of cover crops enhances no-till potential for improving soil physical properties. *Soil Science Society of America Journal*, 75(4), 1471-1482. doi: 10.2136/sssaj2010.0430

Box, J., & Ramsuer, E. (1993). Minirhizotron wheat root data: comparisons to soil core root data. *Agronomy Journal*, 85(5), 1058-1060.

Briones, M. J. I., & Schmidt, O. (2017). Conventional tillage decreases the abundance and biomass of earthworms and alters their community structure in a global meta-analysis. *Global Change Biology*, 23(10), 4396-4419. doi: 10.1111/gcb.13744

Cambardella, C., & Elliott, E. (1993). Carbon and nitrogen distribution in aggregates from cultivated and native grassland soils. *Soil Science Society of America Journal*, 57(4), 1071-1076.

Campbell, B. M., Thornton, P., Zougmoré, R., Asten, P. V., & Lipper, L. (2014). Sustainable intensification: what is its role in climate smart agriculture? *Current Opinion in Environmental Sustainability*, 8(8), 39-43.

Capowiez, Y., Cadoux, S., Bouchant, P., Ruy, S., Roger-Estrade, J., Richard, G., & Boizard, H. (2009). The effect of tillage type and cropping system on earthworm communities, macroporosity and water infiltration. *Soil & Tillage Research*, 105(2), 209-216.

Castro, M. S., Steudler, P. A., Melillo, J. M., Aber, J. D., & Bowden, R. D. (1995). Factors controlling atmospheric methane consumption by temperate forest soils. *Global Biogeochemical Cycles*, 9(1), 1-10.

Chan, K. Y. (2004). Impact of tillage practices and burrows of a native Australian anecic earthworm on soil hydrology. *Applied Soil Ecology*, 27(1), 89-96.

Chen, H. Q., Marhan, S., Billen, N., & Stahr, K. (2009). Soil organic-carbon and total nitrogen stocks as affected by different land uses in Baden-Wurttemberg (southwest Germany). *Journal of Plant Nutrition and Soil Science*, 172(1), 32-42. doi: 10.1002/jpln.200700116

Cheng, C. H., Lehmann, J., Thies, J. E., & Burton, S. D. (2008). Stability of black carbon in soils across a climatic gradient. *Journal of Geophysical Research Biogeosciences*, 113(G2), 50-55.

Clark, K. M., Boardman, D. L., Staples, J. S., Easterby, S., Reinbott, T. M., Kremer, R. J., . . . Veum, K. S. (2017). Crop yield and soil organic carbon in conventional and no-till organic systems on a claypan soil. *Agronomy Journal*, 109(2), 588-599. doi: 10.2134/agronj2016.06.0367

De Baets, S., Poesen, J., Meersmans, J., & Serlet, L. (2011). Cover crops and their erosion-reducing effects during concentrated flow erosion. *Catena*, 85(3), 237-244.

Denef, K., Zotarelli, L., Boddey, R. M., & Six, J. (2007). Microaggregate-associated carbon as a diagnostic fraction for management-induced changes in soil organic carbon in two Oxisols. *Soil Biology & Biochemistry*, 39(5), 1165-1172. doi: 10.1016/j.soilbio.2006.12.024

Dominy, C., Haynes, R., & Van Antwerpen, R. (2002). Loss of soil organic matter and related soil properties under long-term sugarcane production on two contrasting soils. *Biology and Fertility of Soils*, 36(5), 350-356.

Doran, J., & Smith, M. (1987). Organic matter management and utilization of soil and fertilizer nutrients. *Soil fertility and organic matter as critical components of production systems*, 53-72.

Du, Z. L., Angers, D. A., Ren, T. S., Zhang, Q. Z., & Li, G. C. (2017b). The effect of no-till on organic C storage in Chinese soils should not be overemphasized: a meta-analysis. *Agriculture Ecosystems & Environment*, 236, 1-11. doi: 10.1016/j.agee.2016.11.007

Du, Z. L., Zhao, J. K., Wang, Y. D., & Zhang, Q. Z. (2017a). Biochar addition drives soil aggregation and carbon sequestration in aggregate fractions from an intensive agricultural system. *Journal of Soils and Sediments*, 17(3), 581-589. doi: 10.1007/s11368-015-1349-2

Duong, T., Baumann, K., & Marschner, P. (2009). Frequent addition of wheat straw residues to soil enhances carbon mineralization rate. *Soil Biology & Biochemistry*, 41(7), 1475-1482.

Duval, M. E., Galantini, J. A., Capurro, J. E., & Martinez, J. M. (2016). Winter cover crops in soybean monoculture: effects on soil organic carbon and its fractions. *Soil & Tillage Research*, 161, 95-105. doi: 10.1016/j.still.2016.04.006

FAO. (2013). Climate-smart Agriculture: Sourcebook. Food and Agriculture Organization of the United Nations (FAO), Rome, Italy.

Franzluebbers, A., Hons, F., & Zuberer, D. (1995). Tillage and crop effects on seasonal soil carbon and nitrogen dynamics. *Soil Science Society of America Journal*, 59(6), 1618-1624.

Fungo, B., Lehmann, J., Kalbitz, K., Thiongo, M., Okeyo, I., Tenywa, M., & Neufeldt, H. (2017). Aggregate size distribution in a biochar-amended tropical Ultisol under conventional hand-hoe tillage. *Soil & Tillage Research*, 165, 190-197. doi: 10.1016/j.still.2016.08.012

Garcia, C., Nannipieri, P., & Hernandez, T. (2018). Chapter 9 - The future of soil carbon. In C. Garcia, P. Nannipieri & T. Hernandez (Eds.), *The Future of Soil Carbon* (pp. 239-267): Academic Press.

Gregorich, E., Liang, B., Ellert, B., & Drury, C. (1996). Fertilization effects on soil organic matter turnover and corn residue C storage. *Soil Science Society of America Journal*, 60(2), 472-476.

Guggenberger, G., Frey, S. D., Six, J., Paustian, K., & Elliott, E. T. (1999). Bacterial and fungal cell-wall residues in conventional and no-tillage agroecosystems. *Soil Science Society of America Journal*, 63(5), 1188-1198.

Hassink, J. (1997). The capacity of soils to preserve organic C and N by their association with clay and silt particles. *Plant and Soil*, 191(1), 77-87.

Hedges, L. V., Gurevitch, J., & Curtis, P. S. (1999). The meta-analysis of response ratios in experimental ecology. *Ecology*, 80(4), 1150-1156.

Higashi, T., Yunghui, M., Komatsuzaki, M., Miura, S., Hirata, T., Araki, H., . . . Ohta, H. (2014). Tillage and cover crop species affect soil organic carbon in Andosol, Kanto, Japan. *Soil & Tillage Research*, 138, 64-72. doi: 10.1016/j.still.2013.12.010

Huang, Y., Ren, W., Wang, L., Hui, D., Grove, J. H., Yang, X., ... & Goff, B. (2018). Greenhouse gas emissions and crop yield in no-tillage systems: A meta-analysis. *Agriculture, Ecosystems & Environment*, 268, 144-153.

Janssens, I., Dieleman, W., Luyssaert, S., Subke, J. A., Reichstein, M., Ceulemans, R., . . . Matteucci, G. (2010). Reduction of forest soil respiration in response to nitrogen deposition. *Nature Geoscience*, 3(5), 315.

Jarecki, M. K., & Lal, R. (2003). Crop management for soil carbon sequestration. *Critical Reviews in Plant Sciences*, 22(6), 471-502. doi: 10.1080/07352680390253179

Jiang, X., Cao, L., & Zhang, R. (2014). Changes of labile and recalcitrant carbon pools under nitrogen addition in a city lawn soil. *Journal of Soils and Sediments*, 14(3), 515-524.

Jiang, X., Haddix, M. L., & Cotrufo, M. F. (2016). Interactions between biochar and soil organic carbon decomposition: effects of nitrogen and low molecular weight carbon compound addition. *Soil Biology and Biochemistry*, 100, 92-101.

Jien, S. H., & Wang, C. S. (2013). Effects of biochar on soil properties and erosion potential in a highly weathered soil. *Catena*, 110(110), 225-233.

Jobbágy, E. G., & Jackson, R. B. (2000). The vertical distribution of soil organic carbon and its relation to climate and vegetation. *Ecological Applications*, 10(2), 423-436.

Karhu, K., Mattila, T., Bergström, I., & Regina, K. (2011). Biochar addition to agricultural soil increased CH_4 uptake and water holding capacity - Results from a short-term pilot field study. *Agriculture Ecosystems & Environment*, 140(1-2), 309-313.

Kemmitt, S. J., Wright, D., Goulding, K. W., & Jones, D. L. (2006). pH regulation of carbon and nitrogen dynamics in two agricultural soils. *Soil Biology and Biochemistry*, 38(5), 898-911.

Kessel, C., Venterea, R., Six, J., Adviento-Borbe, M. A., Linquist, B., & Groenigen, K. J. (2013). Climate, duration, and N placement determine N_2O emissions in reduced tillage systems: a meta-analysis. *Global Change Biology*, 19(1), 33-44.

Koga, N., & Tsuji, H. (2009). Effects of reduced tillage, crop residue management and manure application practices on crop yields and soil carbon sequestration on an Andisol in northern Japan. *Soil Science & Plant Nutrition*, 55(4), 546-557.

Ladd, J. N., Foster, R. C., Nannipieri, P., & Oades, J. (1996). Soil structure and biological activity. *Soil Biochemistry*, 9, 23-78.

Lal, R. (2004). Soil carbon sequestration to mitigate climate change. *Geoderma*, 123(1), 1-22.

Lal, R. (2005). Soil erosion and carbon dynamics. *Soil & Tillage Research*, 81(2), 137-142.

Lal, R. (2018). Digging deeper: A holistic perspective of factors affecting soil organic carbon sequestration in agroecosystems. *Global Change Biology*, 24(8):3285-3301. doi: 10.1111/gcb.14054

Lavelle, P., Brussaard, L., & Hendrix, P. (1999). *Earthworm Management in Tropical Agroecosystems*. Biddles Ud, Guildford and King's Lynn, UK.

Liang, A. Z., Zhang, X. P., Fang, H. J., Yang, X. M., & Drury, C. F. (2007). Short-term effects of tillage practices on organic carbon in clay loam soil of northeast China. *Pedosphere*, 17(5), 619-623. doi: 10.1016/s1002-0160(07)60073-3

Liang, C., & Balser, T. C. (2012). Warming and nitrogen deposition lessen microbial residue contribution to soil carbon pool. *Nature Communications*, 3, 1222. doi: 10.1038/ncomms2224

Lipper, L., Thornton, P., Campbell, B. M., Baedeker, T., Braimoh, A., Bwalya, M., . . . Henry, K. (2014). Climate-smart agriculture for food security. *Nature Climate Change*, 4(12), 1068–1072.

Liu, E. K., Teclemariam, S. G., Yan, C. R., Yu, J. M., Gu, R. S., Liu, S., . . . Liu, Q. (2014). Long-term effects of no-tillage management practice on soil organic carbon and its fractions in the northern China. *Geoderma*, 213, 379-384. doi: 10.1016/j.geoderma.2013.08.021

Liu, S. W., Zhang, Y. J., Zong, Y. J., Hu, Z. Q., Wu, S., Zhou, J., . . . Zou, J. W. (2016). Response of soil carbon dioxide fluxes, soil organic carbon and microbial biomass carbon to biochar amendment: a meta-analysis. *Global Change Biology Bioenergy*, 8(2), 392-406. doi: 10.1111/gcbb.12265

Llorach-Massana, P., Lopez-Capel, E., Peña, J., Rieradevall, J., Montero, J. I., & Puy, N. (2017). Technical feasibility and carbon footprint of biochar co-production with tomato plant residue. *Waste Management*, 67, 121-130.

Lorenz, K., & Lal, R. (2016). Soil organic carbon: an appropriate indicator to monitor trends of land and soil degradation within the SDG framework. Dessau-Roßlau, Germany.

Luo, Z., Feng, W., Luo, Y., Baldock, J., & Wang, E. (2017). Soil organic carbon dynamics jointly controlled by climate, carbon inputs, soil properties and soil carbon fractions. *Global Change Biology*, 23(10), 4430-4439. doi: 10.1111/gcb.13767

Luo, Z. K., Wang, E. L., & Sun, O. J. (2010). Can no-tillage stimulate carbon sequestration in agricultural soils? A meta-analysis of paired experiments. *Agriculture Ecosystems & Environment*, 139(1-2), 224-231. doi: 10.1016/j.agee.2010.08.006

Manna, M. C., Swarup, A., Wanjari, R. H., Ravankar, H. N., Mishra, B., Saha, M. N., . . . Sarap, P. A. (2005). Long-term effect of fertilizer and manure application on soil organic carbon storage, soil quality and yield sustainability under sub-humid and semi-arid tropical India. *Field Crops Research*, 93(2-3), 264-280. doi: 10.1016/j.fcr.2004.10.006

Martin, M. P., Wattenbach, M., Smith, P., & Meersmans, J. (2011). Spatial distribution of soil organic carbon stocks in France. *Biogeosciences*, 8(5), 1053-1065.

Matovic, D. (2011). Biochar as a viable carbon sequestration option: Global and Canadian perspective. *Energy*, 36(4).

McVay, K., Radcliffe, D., & Hargrove, W. (1989). Winter legume effects on soil properties and nitrogen fertilizer requirements. *Soil Science Society of America Journal*, 53(6), 1856-1862.

Meersmans, J., Martin, M. P., De Ridder, F., Lacarce, E., Wetterlind, J., De Baets, S., . . . Bispo, A. (2012). A novel soil organic C model using climate, soil type and management data at the national scale in France. *Agronomy for Sustainable Development*, 32(4), 873-888.

Meersmans, J., Van, W. B., Goidts, E., Van, M. M., De, B. S., & De, R. F. (2011). Spatial analysis of soil organic carbon evolution in Belgian croplands and grasslands, 1960-2006. *Global Change Biology*, 17(1), 466-479.

Meyer, S., Genesio, L., Vogel, I., Schmidt, H.-P., Soja, G., Someus, E., . . . Glaser, B. (2017). Biochar standardization and legislation harmonization. *Journal of Environmental Engineering and Landscape Management*, 25(2), 175-191.

Mikha, M. M., & Rice, C. W. (2004). Tillage and manure effects on soil and aggregate-associated carbon and nitrogen. *Soil Science Society of America Journal*, 68(3), 809-816.

Molina, L. G., Moreno Pérez, E. D. C., & Pérez, A. B. (2017). Simulation of soil organic carbon changes in Vertisols under conservation tillage using the RothC model. *Scientia Agricola*, 74(3), 235-241.

Moreno, F., Murillo, J., Pelegrín, F., & Girón, I. (2006). Long-term impact of conservation tillage on stratification ratio of soil organic carbon and loss of total and active $CaCO_3$. *Soil & Tillage Research*, 85(1), 86-93.

Motavalli, P., Palm, C., Parton, W., Elliott, E., & Frey, S. (1995). Soil pH and organic C dynamics in tropical forest soils: evidence from laboratory and simulation studies. *Soil Biology and Biochemistry*, 27(12), 1589-1599.

Mukherjee, A., & Lal, R. (2014). The biochar dilemma. *Soil Research*, 52(3), 217-230.

Mulvaney, R., Khan, S., & Ellsworth, T. (2009). Synthetic nitrogen fertilizers deplete soil nitrogen: a global dilemma for sustainable cereal production. *Journal of Environmental Quality*, 38(6), 2295-2314.

Nash, P. R., Gollany, H. T., Liebig, M. A., Halvorson, J. J., Archer, D. W., & Tanaka, D. L. (2018). Simulated soil organic carbon responses to crop rotation, tillage, and climate change in North Dakota. *Journal of Environmental Quality*, 47(4), 654-662.

Paustian, K., Lehmann, J., Ogle, S., Reay, D., Robertson, G. P., & Smith, P. (2016). Climate-smart soils. *Nature*, 532(7597), 49-57.

Pietri, J. A., & Brookes, P. (2009). Substrate inputs and pH as factors controlling microbial biomass, activity and community structure in an arable soil. *Soil Biology and Biochemistry*, 41(7), 1396-1405.

Plaza-Bonilla, D., Cantero-Martinez, C., & Alvaro-Fuentes, J. (2010). Tillage effects on soil aggregation and soil organic carbon profile distribution under Mediterranean semi-arid conditions. *Soil Use and Management*, 26(4), 465-474. doi: 10.1111/j.1475-2743.2010.00298.x

Poeplau, C., & Don, A. (2015). Carbon sequestration in agricultural soils via cultivation of cover crops—a meta-analysis. *Agriculture, Ecosystems & Environment*, 200, 33-41. doi: 10.1016/j.agee.2014.10.024

Powlson, D. S., Riche, A. B., Coleman, K., Glendining, M. J., & Whitmore, A. P. (2008). Carbon sequestration in European soils through straw incorporation: limitations and alternatives. *Waste Management*, 28(4), 741-746.

Pregitzer, K. S., Burton, A. J., Zak, D. R., & Talhelm, A. F. (2008). Simulated chronic nitrogen deposition increases carbon storage in Northern Temperate forests. *Global Change Biology*, 14(1), 142-153.

Ren, W., Tian, H., Tao, B., Huang, Y., & Pan, S. (2012). China's crop productivity and soil carbon storage as influenced by multifactor global change. *Global Change Biology*, 18(9), 2945-2957.

Ren W. (2018) Towards an Integrated Agroecosystem Modeling Approach for Climate-Smart Agricultural Management. In: Wendroth, O., R.J. Lascano, and L. Ma (Eds.) Bridging among disciplines by synthesizing soil and plant processes. *Advances in Agricultural Systems Modeling*, Volume 8: 127-144, ASA, CSSA, SSSA, Madison, WI 53711-5801, USA.

Rothstein, H. R., Sutton, A. J., & Borenstein, M. (Eds.) (2006). *Publication Bias in Meta-Analysis: Prevention, Assessment and Adjustments*. John Wiley & Sons.

Rusco, E., Jones, R., & Bidoglio, G. (2001). Organic matter in the soils of Europe: Present status and future trends. European Soil Bureau, Soil and Waste Unit, Institute for Environment and Sustainability, JRC Ispra Institute, Joint Research Centre European Commission, Italy.

Sainju, U., Singh, B., & Whitehead, W. (2002). Long-term effects of tillage, cover crops, and nitrogen fertilization on organic carbon and nitrogen concentrations in sandy loam soils in Georgia, USA. *Soil & Tillage Research*, 63(3), 167-179.

Sainju, U., Whitehead, W., & Singh, B. (2003). Cover crops and nitrogen fertilization effects on soil aggregation and carbon and nitrogen pools. *Canadian Journal of Soil Science*, 83(2), 155-165.

Sainju, U. M., Singh, B. P., & Whitehead, W. F. (1998). Cover crop root distribution and its effects on soil nitrogen cycling. *Agronomy Journal*, 90(4), 511-518.

Sainju, U. M., Singh, B. P., Whitehead, W. F., & Wang, S. (2006). Carbon supply and storage in tilled and nontilled soils as influenced by cover crops and nitrogen fertilization. *Journal of Environmental Quality*, 35(4), 1507-1517. doi: 10.2134/jeq2005.0189

Salinas-Garcia, J., Hons, F., & Matocha, J. (1997). Long-term effects of tillage and fertilization on soil organic matter dynamics. *Soil Science Society of America Journal*, 61(1), 152-159.

Searchinger, T. D., Wirsenius, S., Beringer, T., & Dumas, P. (2018). Assessing the efficiency of changes in land use for mitigating climate change. *Nature*, 564(7735), 249.

Shi, J., Luo, Y. Q., Fan, Z., & Ping, H. (2010). The relationship between invasive alien species and main climatic zones. *Biodiversity & Conservation*, 19(9), 2485-2500.

Shipitalo, M. J., Edwards, W. M., Dick, W. A., & Owens, L. B. (1990). Initial storm effects on macropore transport of surface-applied chemicals in no-till soil. *Soil Science Society of America Journal*, 54(6), 1530-1536.

Six, J., Conant, R., Paul, E. A., & Paustian, K. (2002). Stabilization mechanisms of soil organic matter: implications for C-saturation of soils. *Plant and Soil*, 241(2), 155-176.

Six, J., Elliott, E. T., & Paustian, K. (2000). Soil macroaggregate turnover and microaggregate formation: a mechanism for C sequestration under no-tillage agriculture. *Soil Biology & Biochemistry*, 32(14), 2099-2103.

Six, J., Ogle, S. M., Conant, R. T., Mosier, A. R., & Paustian, K. (2004). The potential to mitigate global warming with no-tillage management is only realized when practised in the long term. *Global Change Biology*, 10(2), 155-160.

Spokas, K., Koskinen, W., Baker, J., & Reicosky, D. (2009). Impacts of woodchip biochar additions on greenhouse gas production and sorption/degradation of two herbicides in a Minnesota soil. *Chemosphere*, 77(4), 574-581.

Spokas, K. A., & Reicosky, D. C. (2009). Impacts of sixteen different biochars on soil greenhouse gas production. *Annals of Environmental Science*, 3, 179-193.

Stronkhorst, L., & Venter, A. (2008). Investigating the soil organic carbon status in South African soils and the relationship between soil organic carbon and other soil chemical properties. ARC-ISCW Report No. Agricultural Research Council, Pretoria.

Swanepoel, C. M., Laan, M. V. D., Weepener, H. L., Preez, C. C. D., & Annandale, J. G. (2016). Review and meta-analysis of organic matter in cultivated soils in southern Africa. *Nutrient Cycling in Agroecosystems*, 104(2), 107-123.

Thomsen, I. K., & Christensen, B. T. (2004). Yields of wheat and soil carbon and nitrogen contents following long-term incorporation of barley straw and ryegrass catch crops. *Soil Use and Management*, 20(4), 432-438.

Tian, G., Kang, B. T., Kolawole, G. O., Idinoba, P., & Salako, F. K. (2005). Long-term effects of fallow systems and lengths on crop production and soil fertility maintenance in West Africa. *Nutrient Cycling in Agroecosystems*, 71(2), 139-150.

Tondoh, J. E., Ouédraogo, I., Bayala, J., Tamene, L., Sila, A., Vågen, T. G., & Kalinganiré, A. (2016). Soil organic carbon stocks in semi-arid West African drylands: implications for climate change adaptation and mitigation. *Soil,* Copernicus GmbH, 1-41 p.

UNEP. (1997). World Atlas of Desertification, 2nd edn (eds Middleton N, Thomas D), Edward Arnold, London.

Unger, P. W. (1997). Management-induced aggregation and organic carbon concentrations in the surface layer of a Torrertic Paleustoll. *Soil & Tillage Research*, 42(3), 185-208.

Van Bergen, P. F., Nott, C. J., Bull, I. D., Poulton, P. R., & Evershed, R. P. (1998). Organic geochemical studies of soils from the Rothamsted Classical Experiments—IV. Preliminary results from a study of the effect of soil pH on organic matter decay. *Organic Geochemistry*, 29(5–7), 1779-1795.

Van Eerd, L. L., Congreves, k. A., Hayes, A., Verhallen, A., & Hooker, D. C. (2014). Long-term tillage and crop rotation effects on soil quality, organic carbon, and total nitrogen. *Canadian Journal of Soil Science*, 94(3), 303-315.

Vieira, F. C. B., Bayer, C., Zanatta, J. A., Mielniczuk, J., & Six, J. (2009). Building up organic matter in a subtropical Paleudult under legume cover-crop-based rotations. *Soil Science Society of America Journal*, 73(5), 1699-1706. doi: 10.2136/sssaj2008.0241

Virto, I., Barré, P., Burlot, A., & Chenu, C. (2012). Carbon input differences as the main factor explaining the variability in soil organic C storage in no-tilled compared to inversion tilled agrosystems. *Biogeochemistry*, 108(1-3), 17-26.

Wang, J., Xiong, Z., & Kuzyakov, Y. (2016). Biochar stability in soil: meta-analysis of decomposition and priming effects. *Global Change Biology Bioenergy*, 8(3), 512-523.

Wang, X., Butterly, C. R., Baldock, J. A., & Tang, C. (2017). Long-term stabilization of crop residues and soil organic carbon affected by residue quality and initial soil pH. *Science of The Total Environment*, 587, 502-509.

Weng, Z., Van Zwieten, L., Singh, B. P., Tavakkoli, E., Joseph, S., Macdonald, L. M., . . . Cowie, A. (2017). Biochar built soil carbon over a decade by stabilizing rhizodeposits. *Nature Climate Change*, 7(5), 371-376. doi: 10.1038/nclimate3276

West, T. O., & Post, W. M. (2002). Soil organic carbon sequestration rates by tillage and crop rotation. *Soil Science Society of America Journal*, 66(6), 1930-1946.

Wiesmeier, M., Poeplau, C., Sierra, C. A., Maier, H., Frühauf, C., Hübner, R., . . . Hangen, E. (2016). Projected loss of soil organic carbon in temperate agricultural soils in the 21st century: effects of climate change and carbon input trends. *Scientific Reports*, 6, 32525.

Willett, V. B., Reynolds, B. A., Stevens, P. A., Ormerod, S. J., & Jones, D. L. (2004). Dissolved organic nitrogen regulation in freshwaters. *Journal of Environmental Quality*, 33(1), 201-209.

Wilson, H. M., & Alkaisi, M. M. (2008). Crop rotation and nitrogen fertilization effect on soil CO_2 emissions in central Iowa. *Applied Soil Ecology*, 39(3), 264-270.

Zhang, A., Bian, R., Pan, G., Cui, L., Hussain, Q., Li, L., . . . Han, X. (2012). Effects of biochar amendment on soil quality, crop yield and greenhouse gas emission in a Chinese rice paddy: a field study of 2 consecutive rice growing cycles. *Field Crops Research*, 127, 153-160.

Zhao, X., Liu, S. L., Pu, C., Zhang, X. Q., Xue, J. F., Ren, Y. X., . . . Zhang, H. L. (2017). Crop yields under no-till farming in China: A meta-analysis. *European Journal of Agronomy*, 84, 67-75. doi: 10.1016/j.eja.2016.11.009

Table 1. Between-group variability (Q_M) of the variables controlling the effects of climate-smart agriculture management practices on soil organic carbon.

Variables	No-till		Reduced tillage		Cover crop		Biochar	
	df	Q_M	df	Q_M	df	Q_M	df	Q_M
Duration	2	12.14**	2	13.69**	2	26.19***	1	0.04
Aridity index	1	0.13	1	10.99***	1	0.04	1	5.73*
Mean annual air temperature	1	16.32***	1	0.47	1	55.99***	1	6.48*
Soil texture	5	20.98***	5	32.15***	4	3.58	5	9.65
Soil depth	3	210.69***	3	73.38***	2	17.38***	-	-
Soil pH	2	9.8**	2	3.52	2	9.05*	2	28.64***
Residue	1	6.56*	1	0.04	1	4.07*	-	-
Nitrogen fertilization	3	7.62	3	11.43*	2	0.89	2	7.22*
Irrigation	1	9.61**	1	0.92	1	0.16	1	1.7
Crop rotation	1	1.72	1	0.26	1	19.43***	1	4.53*

Statistical significance of Q_M: * $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$.

Figure captions

Figure 1. Relationship between climate-smart management practices and soil processes. "+" means a positive feedback or promotion effect; "-" means a negative feedback or inhibition function; and "?" means the effect is unclear. Blue, black, and red show the effect of cover crops, conservation tillage, and biochar on the soil environment, processes, and pools, respectively. SOC: soil organic carbon.

Figure 2. Comparison of climate-smart management vs. their controls for the entire dataset. The number in parentheses represents the number of observations. Error bars represent 95% confidence intervals. SOC: soil organic carbon; NT: no-till; RT: reduced tillage.

Figure 3. Comparison of climate-smart management vs. their controls for subcategories of climate zone (a: the climate zones were divided by aridity index; b: the climate zones were

divided by mean annual air temperature). The number in parentheses represents the number of observations. Error bars represent 95% confidence intervals. SOC: soil organic carbon; NT: no-till; RT: reduced tillage.

Figure 4. Comparison of climate-smart management vs. their controls for subcategories of soil textures. The number in parentheses represents the number of observations. Error bars represent 95% confidence intervals. SOC: soil organic carbon; NT: no-till; RT: reduced tillage.

Figure 5. Comparison of climate-smart management vs. their controls for subcategories of soil depth. The number in parentheses represents the number of observations. Error bars represent 95% confidence intervals. SOC: soil organic carbon; NT: no-till; RT: reduced tillage. The average depths of each categorical group were presented in supplementary files (Table S4-S7).

Figure 6. Comparison of climate-smart management vs. their controls for subcategories of soil pH. The number in parentheses represents the number of observations. Error bars represent 95% confidence intervals. SOC: soil organic carbon; NT: no-till; RT: reduced tillage.

Figure 7. Comparison of climate-smart management vs. their controls for subcategories of experiment duration. The number in parentheses represents the number of observations. Error bars represent 95% confidence intervals. SOC: soil organic carbon; NT: no-till; RT: reduced tillage.

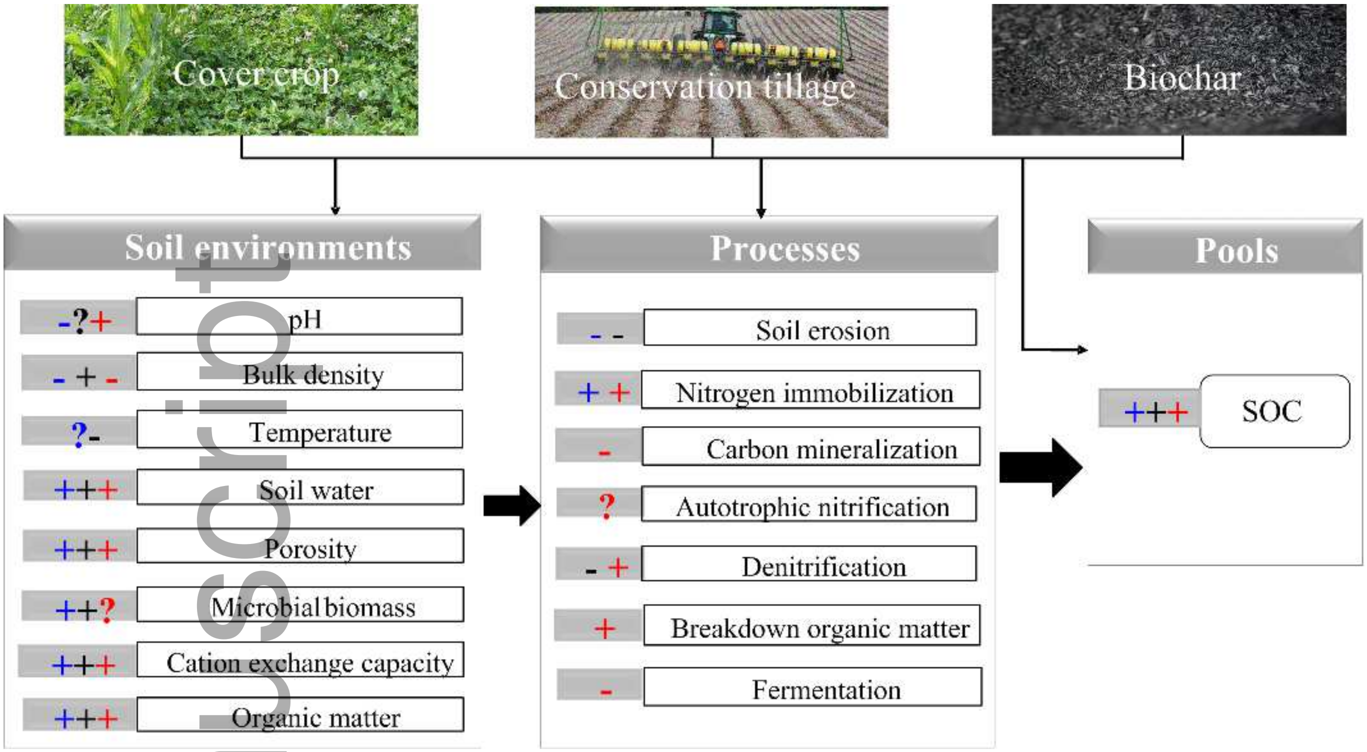
Figure 8. Comparison of climate-smart management vs. their controls for subcategories of crop residues. The number in parentheses represents the number of observations. Error bars represent 95% confidence intervals. SOC: soil organic carbon; NT: no-till; RT: reduced tillage.

Figure 9. Comparison of climate-smart management vs. their controls for subcategories of nitrogen fertilizer use. The number in parentheses represents the number of observations. Error bars represent 95% confidence intervals. Low, medium, and high levels of nitrogen fertilizer use represent 1-100, 101-200, and >200 kg N ha^{-1}, respectively. SOC: soil organic carbon; NT: no-till; RT: reduced tillage.

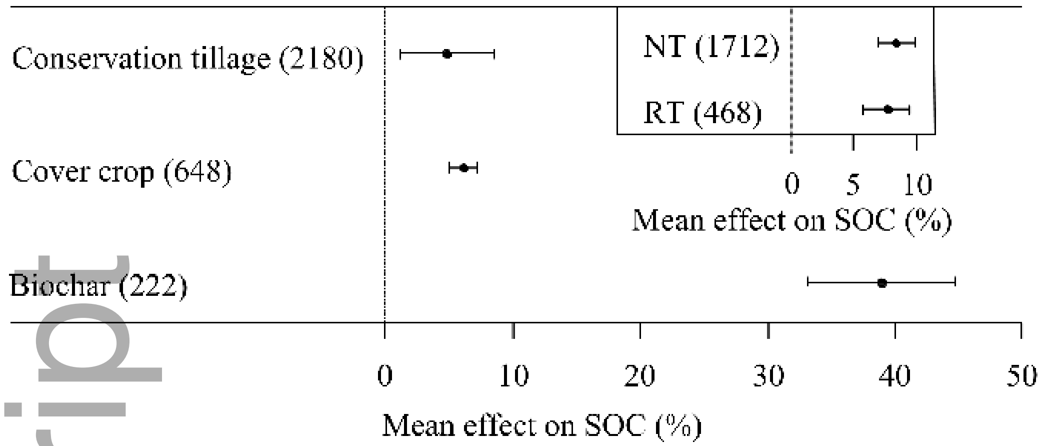
Figure 10. Comparison of climate-smart management vs. their controls for subcategories of water management (a) and cropping systems (b). The number in parentheses represents the

number of observations. Error bars represent 95% confidence intervals. SOC: soil organic carbon; NT: no-till; RT: reduced tillage.

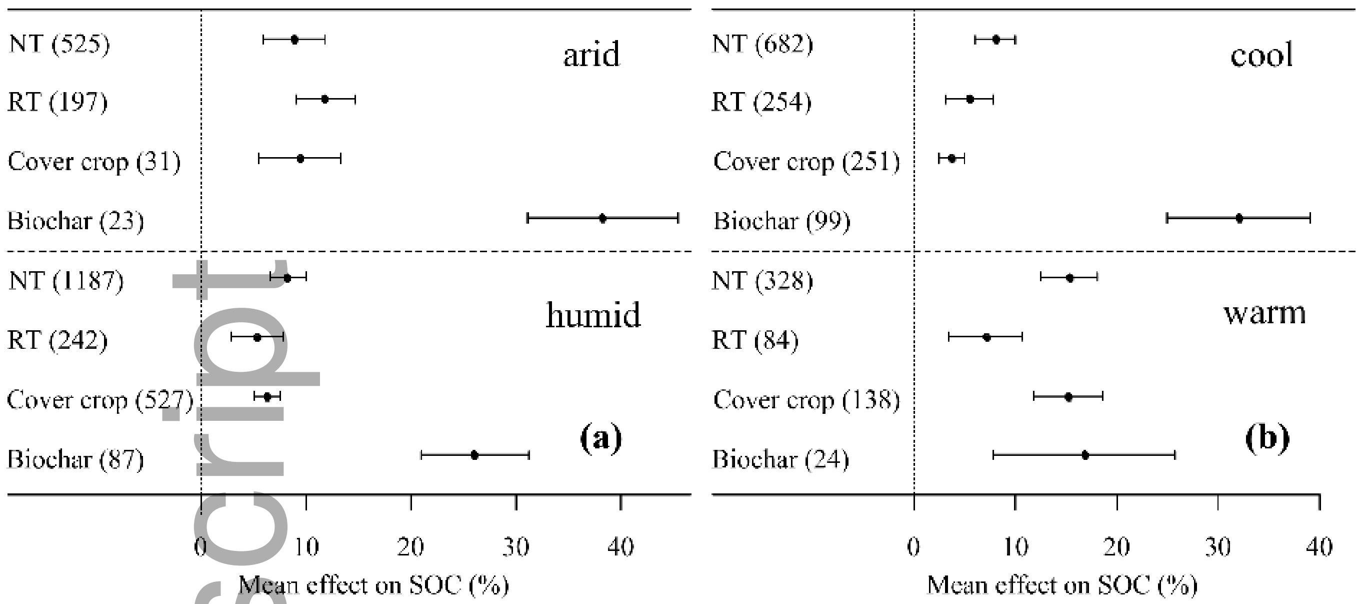
Figure 11. The effect size of combined conservation tillage and cover crops for different subcategories. The number in parentheses represents the number of observations. Error bars represent 95% confidence intervals. The vertical solid line represents 11%, which is the theoretical sum of the effect sizes of conservation tillage and cover crops. SOC: soil organic carbon.



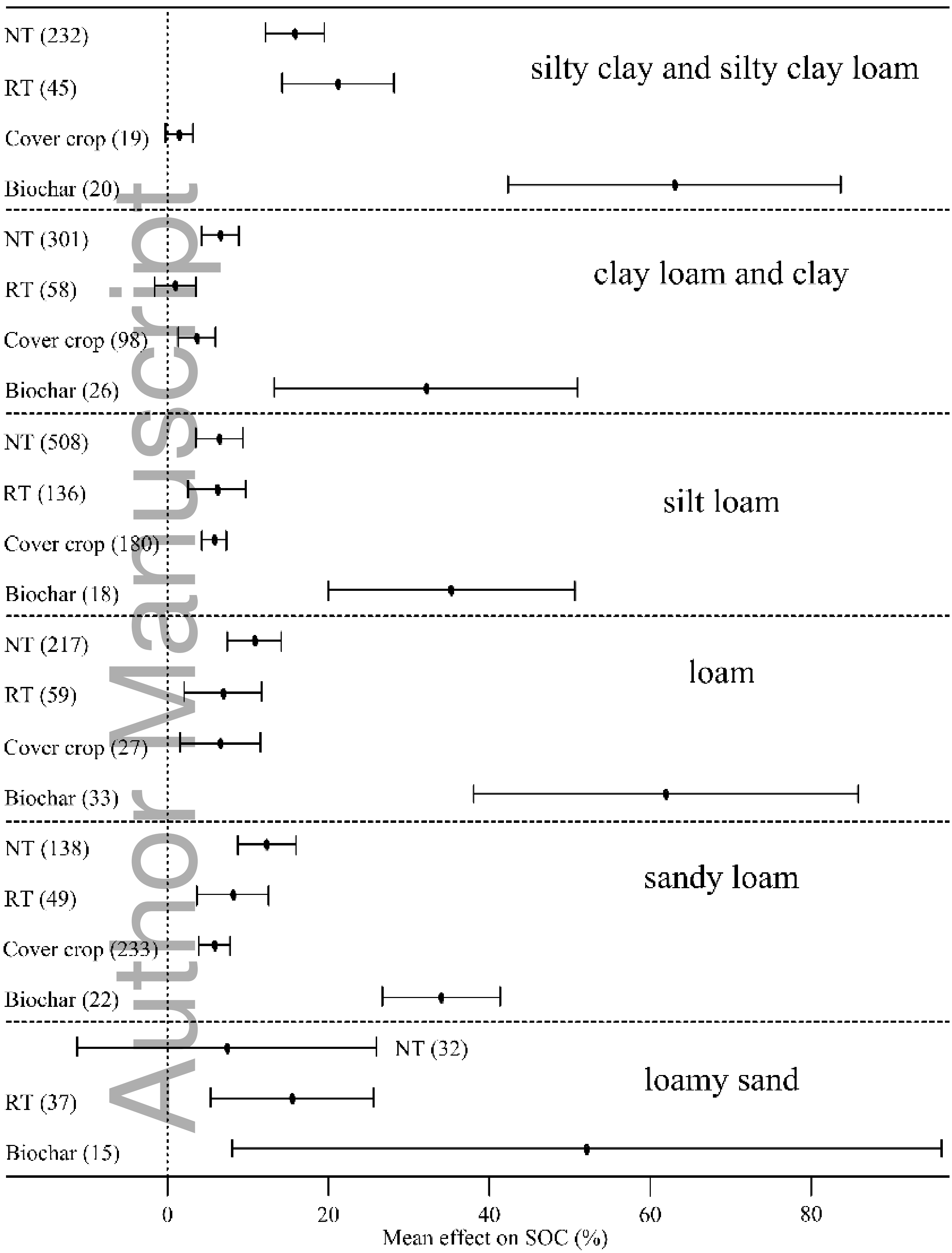
gcb_14658_f1.tif



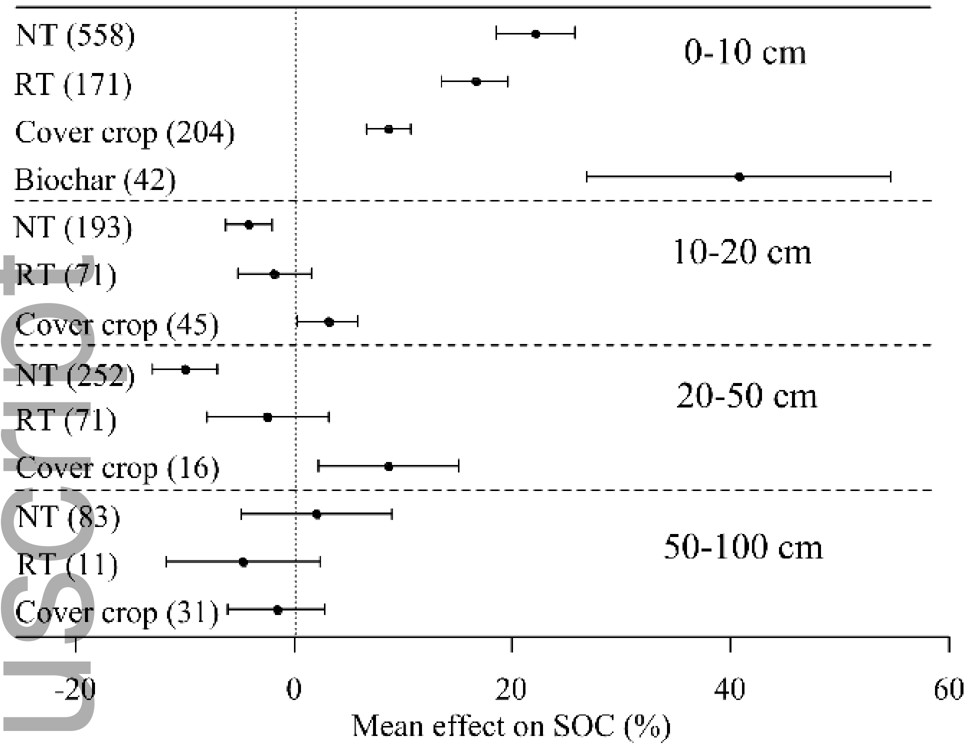
gcb_14658_f2.tif



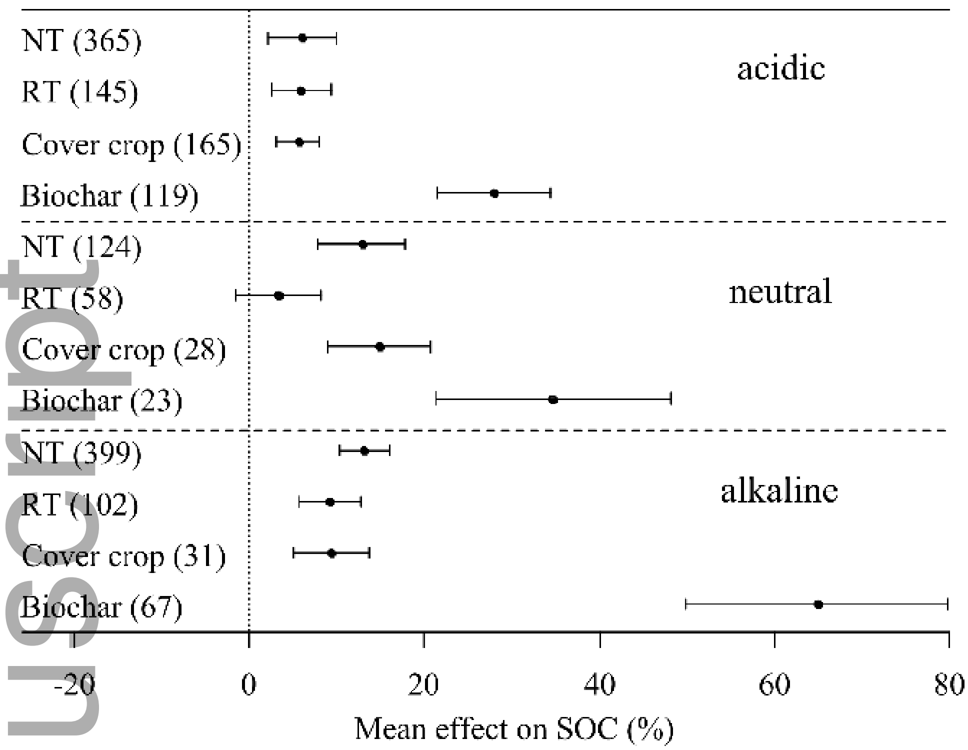
gcb_14658_f3.tif



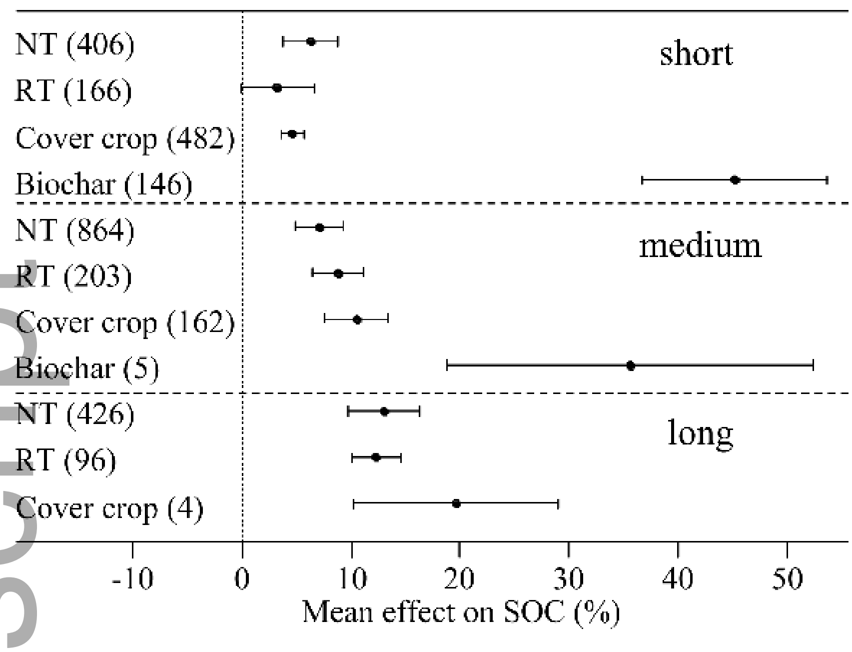
gcb_14658_f4.tif



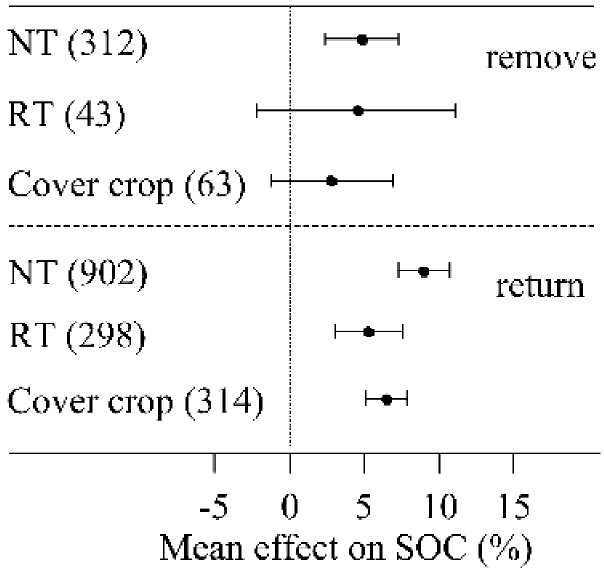
gcb_14658_f5.tif



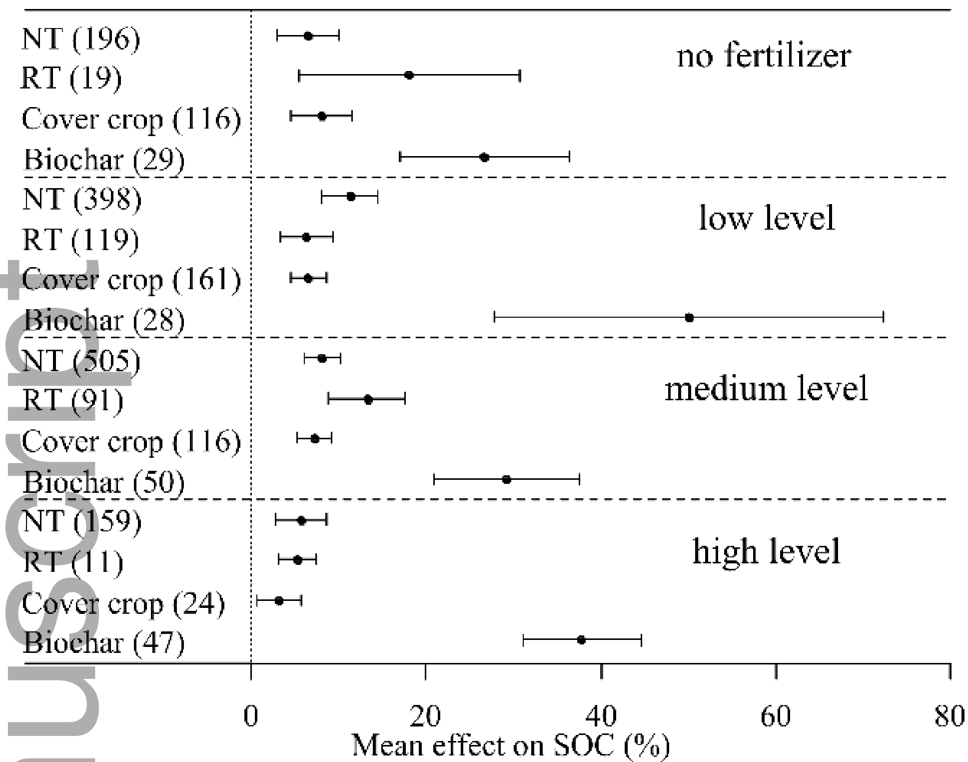
gcb_14658_f6.tif



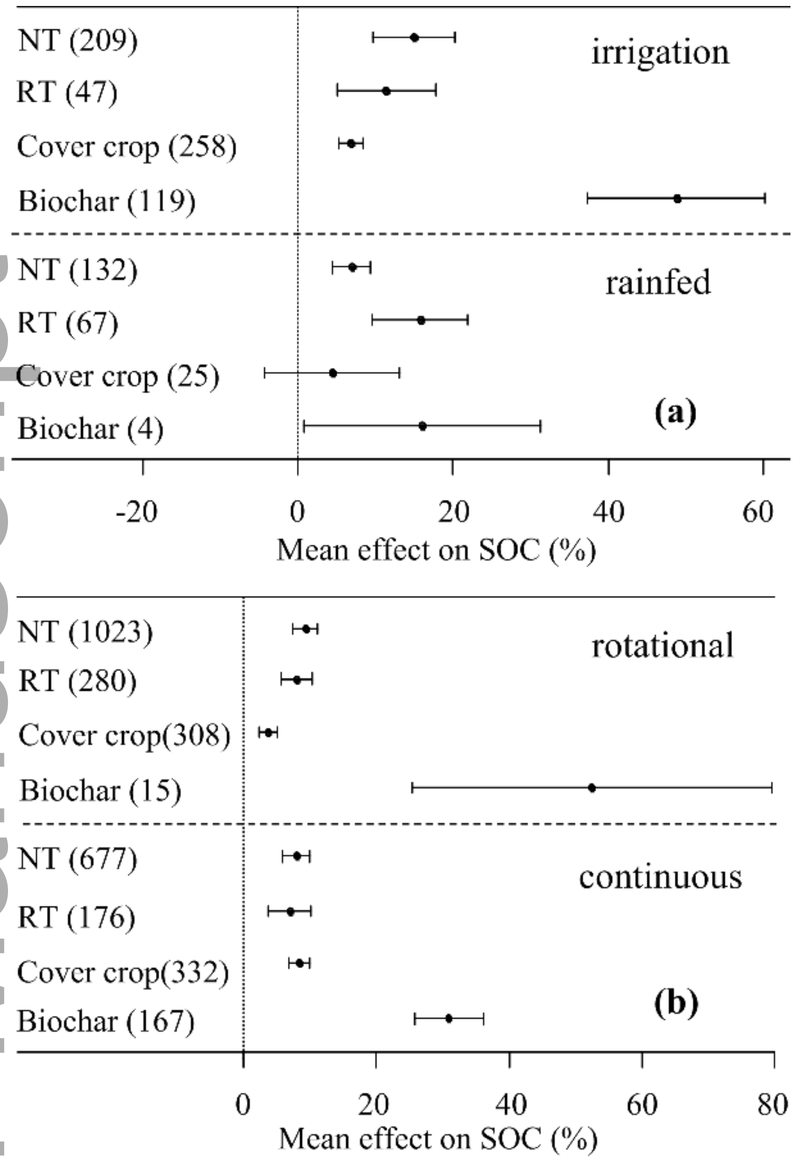
gcb_14658_f7.tif



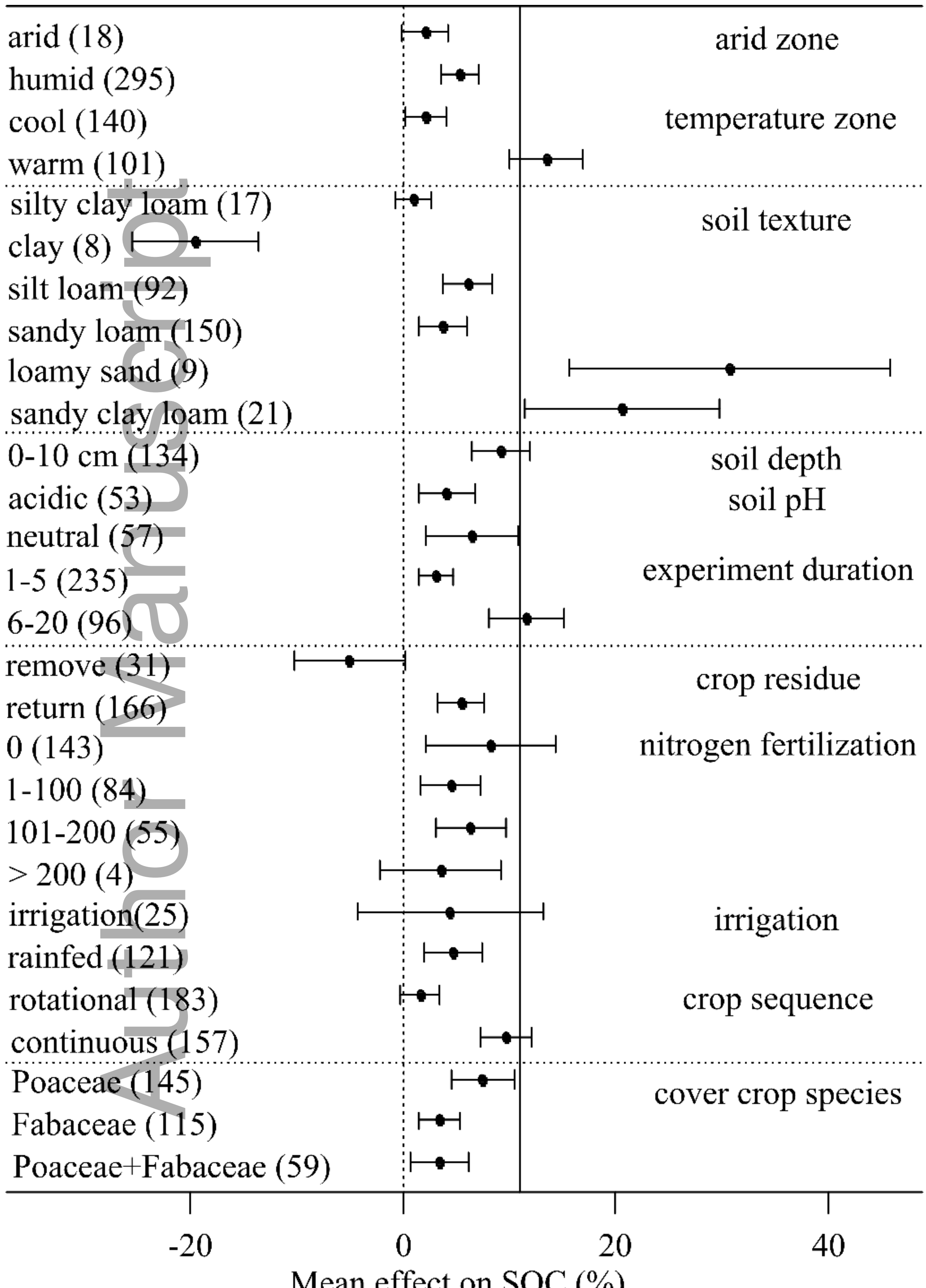
gcb_14658_f8.tif



gcb_14658_f9.tif



gcb_14658_f10.tif



gcb_14658_f11.tif

Exhibit I

Nutrient Cycling in Agroecosystems **60:** 1–8, 2001.

Nitrous oxide emissions from an irrigated soil as affected by fertilizer and straw management

X. Hao*, C. Chang, J.M. Carefoot, H.H. Janzen & B.H. Ellert
Lethbridge Research Centre, Agriculture and Agri-Food Canada, PO Box 3000, Lethbridge, AB Canada T1J 4B1
*(*Corresponding author; e-mail: haoxy@em.agr.ca)*

Key words: greenhouse gas, N_2O flux, straw and fertilizer management, tillage

Abstract

Nitrous oxide (N_2O) emission from farmland is a concern for both environmental quality and agricultural productivity. Field experiments were conducted in 1996–1997 to assess soil N_2O emissions as affected by timing of N fertilizer application and straw/tillage practices for crop production under irrigation in southern Alberta. The crops were soft wheat (*Triticum aestivum* L.) in 1996 and canola (*Brassica napus* L.) in 1997. Nitrous oxide flux from soil was measured using a vented chamber technique and calculated from the increase in concentration with time. Nitrous oxide fluxes for all treatments varied greatly during the year, with the greatest fluxes occurring in association with freeze-thaw events during March and April. Emissions were greater when N fertilizer (100 kg N ha^{-1}) was applied in the fall compared to spring application. Straw removal at harvest in the fall increased N_2O emissions when N fertilizer was applied in the fall, but decreased emissions when no fertilizer was applied. Fall plowing also increased N_2O emissions compared to spring plowing or direct seeding. The study showed that N_2O emissions may be minimized by applying N fertilizer in spring, retaining straw, and incorporating it in spring. The estimates of regional N_2O emissions based on a fixed proportion of applied N may be tenuous since N_2O emission varied widely depending on straw and fertilizer management practices.

Introduction

Agricultural soils are recognized as an important source of atmospheric nitrous oxide (N_2O), a gas contributing to the 'enhanced greenhouse effect' (Mosier and Schimel, 1991). Nitrous oxide also participates in reactions which destroy stratospheric ozone, resulting in a higher UV-B intensity (Cicerone, 1987). In addition to these environmental concerns, N_2O emission also affects crop production as it represents a loss of plant-available N which reduces fertilizer efficiency (Eichner, 1990).

Nitrous oxide is produced from plant-available soil N during nitrification and denitrification (Sahrawat and Keeney, 1986). The emission of N_2O from soil follows an irregular pattern during the year, often depending on agricultural practices, soil properties, and climatic conditions (Jarvis et al., 1991; Ramos, 1996; Henault et al., 1998; Lemke et al., 1998a).

Application of fertilizer N could potentially increase N_2O emissions by supplying more plant-available N for N_2O production (Loro et al., 1997; Mulvaney et al., 1997). However, emission of N_2O from N-fertilized cropland varies considerably, ranging from 0.001% to 6.84% of applied N (Eichner, 1990). Past research has concentrated primarily on the effects of various chemical forms of N fertilizer on N_2O emissions (e.g., Clayton et al., 1997; Mulvaney et al., 1997). Other studies considered how fertilizer rate, placement and application method affected N_2O emissions (Henault et al., 1998; Skiba et al., 1993; Mulvaney et al., 1997).

This study considers N_2O emissions from irrigated soils in a semi-arid region. Under irrigation, crop yields must be higher than dry land farming to cover the added costs. These yields are achieved through a combination of higher moisture levels and increased fertilization, both of which could affect N_2O emis-

Table 1. Treatments used in 1996 – 1997 study

Treatment # Nrate&season_straw_tillage	N Fertilizer Rate	N Fertilizer Time	Straw-Tillage
	(kg N ha^{-1})		
1 Fert0_NS_Fall	0		No straw[b]-Fall plow[c]
2 Fert0_NS_DS	0		No straw- Direct seed
3 Fert0_S_Fall	0		Straw-Fall plow
4 Fert100f_NS_Fall	100	Fall[a]	No straw-Fall plow
5 Fert100f_NS_DS	100	Fall	No straw-Direct seed
6 Fert100f_NS_Spring	100	Fall	No straw-Spring plow[c]
7 Fert100f_S_Fall	100	Fall	Straw-Fall plow
8 Fert100f_S_Spring	100	Fall	Straw-Spring plow
9 Fert100s_NS_Fall	100	Spring[a]	No straw-Fall plow
10 Fert100s_S_Fall	100	Spring	Straw-Fall plow

[a] N fertilizer applied on 30 October 1996 and 5 May 1997, for fall and spring applications, respectively.
[b] Straw was baled and removed from the field on 15 October 1996.
[c] Field plowed on 31 October 1996 and 12 May 1997 for fall and spring plow, respectively.

sions. Higher yields under irrigation mean more crop residue is produced. Both the amount of residue and its management (fall or spring incorporation or direct seeding) could also affect N_2O emissions. Often management involves straw removal, and in some areas removal has increased to satisfy the demand for livestock bedding and raw material to produce fiber and energy. Although straw removal may adversely affect the quality and long-term sustainability of agricultural land (Dormaar and Carefoot, 1998), the effect of straw management on N_2O emissions from irrigated land has not been studied. Furthermore, recent trends toward increased use of conservation tillage may also have implications for N_2O emissions. For example, higher N_2O emissions have been reported with zero-till when compared to conventional tillage (Aulakh et al., 1984; MacKenzie et al., 1997; Palma et al., 1997).

The objectives of this study were to investigate the effects of the timing of N fertilizer application (either spring or fall), straw removal, and tillage practices (fall plow, spring plow or direct seed) on N_2O emission under irrigated conditions. Such information will be imperative to develop management practices that minimize N losses, including N_2O emissions, from agricultural land.

Materials and methods

Nitrous oxide emissions were measured in a long-term irrigated residue management experiment initiated in 1986 on a Dark Brown Chernozemic soil (Typic Haploboroll) at Lethbridge Alberta ($49^0$42' N, $112^0$48' W) (Carefoot et al., 1994). The cropping sequence from 1986 to 1996 was wheat–wheat–oats, including soft white spring wheat (*Triticum aestivum* L.) and spring seeded oats (*Avena sativa* L.). To help manage an infestation of wild oats, canola (*Brassica napus* L.) was used instead of oats in 1997.

In the fall of 1986, a factorial experiment was set up with five straw-tillage treatments (straw-fall plow, straw-spring plow, no straw-fall plow, no straw-spring plow or no straw-direct seed), four N fertilizer application rates (0, 50, 100 or 200 kg N ha^{-1}) and two N fertilizer application times (fall or spring) using a randomized complete block design with four replications. 'No straw' indicates removal by baling of threshed straw but not the standing stubble (typically 15 to 17 cm tall) remaining after grain harvest. Nitrogen fertilizer in the form of NH_4NO_3 was broadcast onto the soil surface. Soil properties have been described in detail by Carefoot et al. (1994) and by Dormaar and Carefoot (1998).

The effects of fertilizer timing and tillage-straw management on N_2O emission were studied for selected treatments in 1996–1997 (Table 1). Soft wheat (cv. AC Reed) was harvested on 12 October 1996

Table 2. Average N_2O emission, water-filled porosity and temperature during 1996 – 1997

Treatment[a]	Daily flux[b]	Water-filled porosity[b]	Soil temperature[b]
	g N ha^{-1} d^{-1}	m^3 m^{-3}	°C
1 Fert0_NS_Fall	1.18b[c]	0.353d	5.73ab
2 Fert0_NS_DS	1.60b	0.494a	5.09ab
3 Fert0_S_Fall	5.23ab	0.348d	5.90a
4 Fert100f_NS-Fall	15.64a	0.337d	5.10ab
5 Fert100f_NS_DS	5.74ab	0.486ab	4.56ab
6 Fert100f_NS_Spring	4.60b	0.441bc	4.69ab
7 Fert100f_S_Fall	8.55ab	0.333d	4.97ab
8 Fert100f_S_Spring	2.50b	0.427c	5.57ab
9 Fert100s_NS_Fall	9.47ab	0.345d	5.55ab
10 Fert100s_S_Fall	4.34b	0.351d	4.28b

[a] Treatments are defined in Table 1.
[b] Average over the entire experimental period.
[c] Means followed by different letters indicate significant differences at the 0.05 probability level, according to the Tukey test (SAS Institute, 1990).

and straw was baled for the required treatments (treatments 1, 2, 4, 5, 6, and 9 in Table 1). Fertilizer rates of 0 and 100 kg N ha^{-1} were selected. The 100 kg N ha^{-1} represents the current recommended N fertilizer rate for irrigated fields in southern Alberta (McKenzie and Kryzanowski, 1993). For the fertilized treatments, NH_4NO_3 was broadcast on 30 October 1996 for fall-applied treatments or on 5 May 1997 for spring-applied treatments.

The direct-seeded treatments were not tilled. The other treatments were tilled with a moldboard plow (to 20 cm depth) followed by cultivation with a disc, either on 31 October 1996 for the fall tillage treatments or on 12 May 1997 for the spring tillage treatments. Canola (cv. Tobin) was seeded on 21 May, harvested on 7 August and the residue was baled on 12 August 1997. Nitrous oxide fluxes were measured from 6 November 1996 to 9 September 1997.

Nitrous oxide fluxes from soil were measured on four replicate plots for each treatment, using a vented chamber (Hutchinson and Mosier, 1981) modified to permit separation of the chamber cover from the base (Chang et al., 1998). One base collar was installed in each plot where it remained for the entire experimental period, except during tillage, seeding or harvesting operations. The volume of the chamber was 1604 cm^3 (9.2 cm high by 14.9 cm in diam.) and its cross-sectional area was 174 cm^2. Fluxes were measured at weekly intervals (between 0700 and 0830 h) by attaching the chamber covers to the base collars and collecting air samples from the chamber head space. For each chamber, 5 ml of headspace air was drawn through a septum into 10 ml polypropylene syringes at 0, 10, 20, 30, and 60 min after the soil was covered. Immediately after air sampling, the syringe needle was stuck into a rubber stopper to prevent gas exchange. On the same day, air samples were analyzed for N_2O using a gas chromatograph (Varian 3600, Varian Instruments, Walnut Creek, CA) equipped with an electron capture detector. During winter months, snow was brushed off the soil surface before the chamber cover was attached to the base.

For each chamber, the flux was calculated by fitting a second order polynomial equation (SAS Institute, 1990) to the five successive N_2O concentrations versus time. Fluxes estimated from single interval measurements may not be accurate (Anthony et al., 1995), because the pattern of N_2O accumulation during 60 min is usually curvilinear (Hutchinson and Mosier, 1981; Chang et al., 1998). This likely reflects the change in concentration gradient as N_2O accumulates in the chamber headspace. The flux of N_2O was calculated by taking derivatives of the second order polynomials and converting them into g ha^{-1} d^{-1}. For each plot, cumulative N_2O emission was calculated by summing the products of the measurement interval (i.e., days between measurements) and the mean flux for that interval (i.e., arithmetic mean of fluxes measured at the start and the end of the interval). Soil temperatures (2.5 cm depth), estimated from thermocouples (Digital Omega HH – 25C, Omega Technology, Stamford, CT), were recorded when the air samples were collected from the chambers. When the soil was not frozen, soil moisture content in the 0-15 cm layer was determined by time domain reflectometry using three-wire probes similar to the design of Zegelin et al. (1989) and a cable tester (1502C, Tektronix, Beaverton OR). The ratio of water- to air-filled porosity was calculated based on the volumetric moisture content and bulk density (Table 2). Since bulk density (BD) data were not collected during 1996-1997 cropping season, BD data collected in spring 1995 (Domaar and Carefoot, 1998) were used in the calculation. Climatic data were obtained from the Lethbridge Research Centre meteorological station located less than 300 m away from the experimental plots.

ANOVA was conducted on the arithmetic mean daily N_2O fluxes for the entire 1996–1997 experimental period (SAS Institute, 1990). Since the ANOVA was significant at a probability level of 0.05, the multi-range Tukey test was performed to assess treatment differences. The contrast method (via. the GLM procedure, SAS Institute, 1990) was also used to evaluate the influence of key management variables.

Results and discussion

Environmental conditions

During the study period (November 1996 through September 1997), the total precipitation (393 mm) was slightly above the long-term average (379 mm) for the 11-month period. Most precipitation occurred in May (96 mm) and June (101 mm) 1997 (Figure 1a). Air temperatures were highly variable during winter and spring when warm 'Chinook' winds blow in southern Alberta (Grace and Hobbs, 1986). For example, between 19 and 22 December 1996, mean daily air temperature changed from 1.2 to $-27.2\ ^0$C (Figure 1a).

The average soil temperature (2.5 cm depth) for all treatments followed similar patterns to air temperature, but the variations were much smaller (Figure 1b). Among the different treatments, water-filled porosity was affected only by the tillage operations (Figure 1c and Tables 2 and 3). Fall plow had the lowest while direct seeding had the highest water-filled porosity throughout the experimental period.

Temporal trends

The average N_2O daily flux for unfertilized and 100 kg ha^{-1} N fertilized treatments fluctuated greatly, ranging from as low as -5 (uptake by soil) to 18 for the unfertilized and 63 g N ha^{-1} d^{-1} for the N

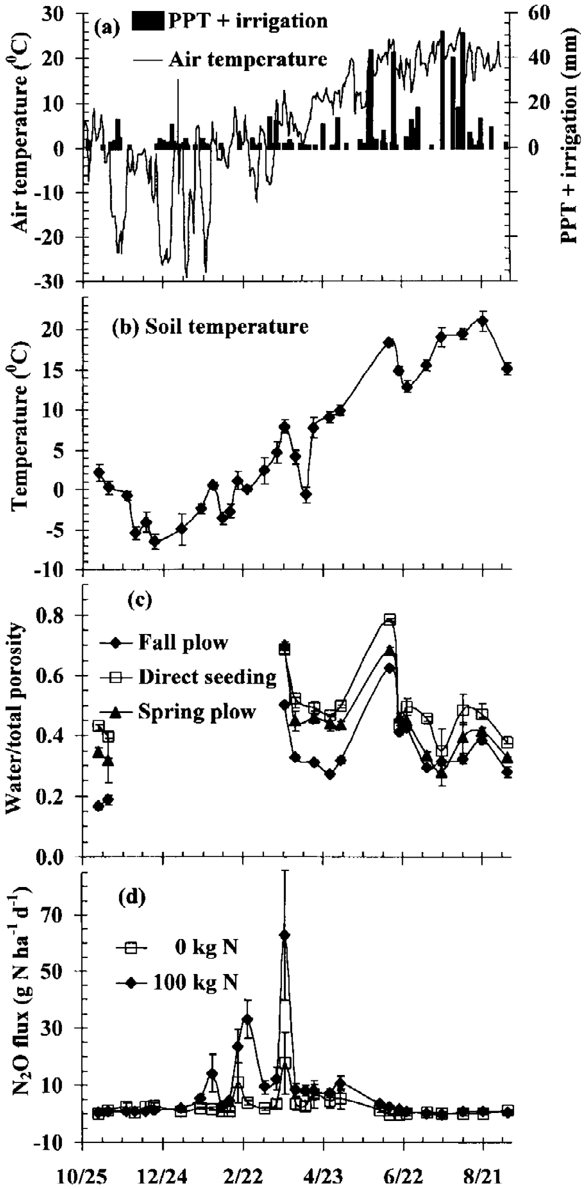


Figure 1. Mean N_2O fluxes in relation to environmental conditions during 1996 – 1997. (a) Daily air temperature and precipitation + irrigation, (b) Soil temperature at 2.5 cm, (c) water-filled porosity in the 0–15 cm soil layer and (d) mean daily N_2O flux patterns for unfertilized and fertilized treatments.

fertilized treatment during the year (Figure 1d). Maximum daily fluxes occurred in early spring (February to early April) as observed by other researchers in western Canada (Nyborg et al., 1997; Chang et al., 1998; Lemke et al., 1998b). The N_2O fluxes increased whenever the soil temperature increased (Figure 1b and 1d), perhaps associated with freeze-thaw events

Table 3. Effect of management practices on N_2O flux, soil moisture and surface temperature

Effect	Contrast	Mean difference[b]		
		N_2O flux	Water-filled porosity	Soil temperature
		(g $ha^{-1}d^{-1}$)	m^3 m^{-3}	°C
N fertilizer				
Zero vs. 100 kg in fall	1+2+3 vs. 4+5+6+7+8+9+10[a]	−4.47*	−0.0136	0.630**
100 kg in fall vs. spring	4+7 vs. 9+10	4.77*	0.0098	0.116
Straw vs. no straw				
Zero N fertilizer	3 vs. 1	4.14	0.0051	0.172
100 kg N fertilizer	7+8+10 vs. 4+5+6+9	−3.56§	−0.0312**	−0.003
Plowing time				
Fall vs. spring/DS	4+7 vs. 5+6+8	8.66**	−0.1135**	0.279
DS vs. spring	5 vs. 6	2.24	0.0452**	−0.132

[a] Treatment numbers are defined in Table 1.
[b] Average over the entire experimental period.
§, *, ** Significant at 0.1, 0.05 and 0.01 probability levels, respectively.

during this time of year (Chen et al., 1995). The daily N_2O flux was very low from late July to the end of September. This flux decline was probably related to the decrease in available N content caused by crop uptake and leaching over the growing season (Xu et al., 1998). Relatively high water-filled porosity in late June (due to above normal precipitation) and in late July and early August (due to irrigation on 17 July, 25 July, and 1 August 1997) failed to produce appreciable N_2O fluxes.

Influence of fertilizer timing

Nitrous oxide emission was affected by the timing of N fertilizer application (Figure 2). Using contrast comparison, fall fertilizer application produced significantly greater N_2O emissions than spring fertilizer application (Table 3). Fall application provided a longer period with available N and moisture conditions favorable for denitrification and N_2O production during the freeze-thaw cycles in the early spring. Carefoot et al. (1994) and Nyborg et al. (1990) attributed the low recovery of fall-applied N fertilizer to denitrification during winter and spring. The N_2O fluxes from spring fertilized plots were lower than fall fertilized plots because the fertilizer was applied in early May after the maximum N_2O flux had occurred. Previous research seldom considered fertilizer timing (fall vs. spring application), but this study clearly demonstrates a significant influence on N_2O emissions. Despite the reduced frequency of N_2O flux measurements during late May to early June, previously published work suggests that large fluxes do not typically occur during this time of year in this region (Chang et al., 1998; Lemke et al., 1998a). While N_2O emissions from spring fertilized plots were greater than emissions from unfertilized plots (Figure 2a and 2c), the differences were not statistically significant.

Effect of straw removal

The effect of straw removal on N_2O emission depended on fertilizer treatment (Figures 2 and 3). When no fertilizer was applied and plots were plowed in the fall, N_2O emission from the ‘straw-removed’ treatment was about 25% that of the treatment where straw was incorporated into the soil (Figure 3a and 3c). The lower N_2O emission associated with straw removal, also observed by Zhengping et al. (1991), could be linked to several factors. Annual straw removal eventually depletes soil organic C and N (Dormaar and Carefoot, 1998), and decreases the amount of N potentially available for nitrification and denitrification. The availability of organic C as an electron donor for denitrification also influences N_2O production in soil (Sahrawat and Keeney, 1986). On one hand, the annual addition of fresh crop residue to soil stimulates denitrification (Burford and Bremner, 1975; Groffman, 1985) by providing readily available C for denitrifying bacteria (with adequate NO_3^- and moisture). Decomposition of crop residue consumes oxygen, which also stimulates denitrification and N_2O emission. On the other hand, addition of fresh wheat residue with a high C:N ratio could cause N immobilization and reduce

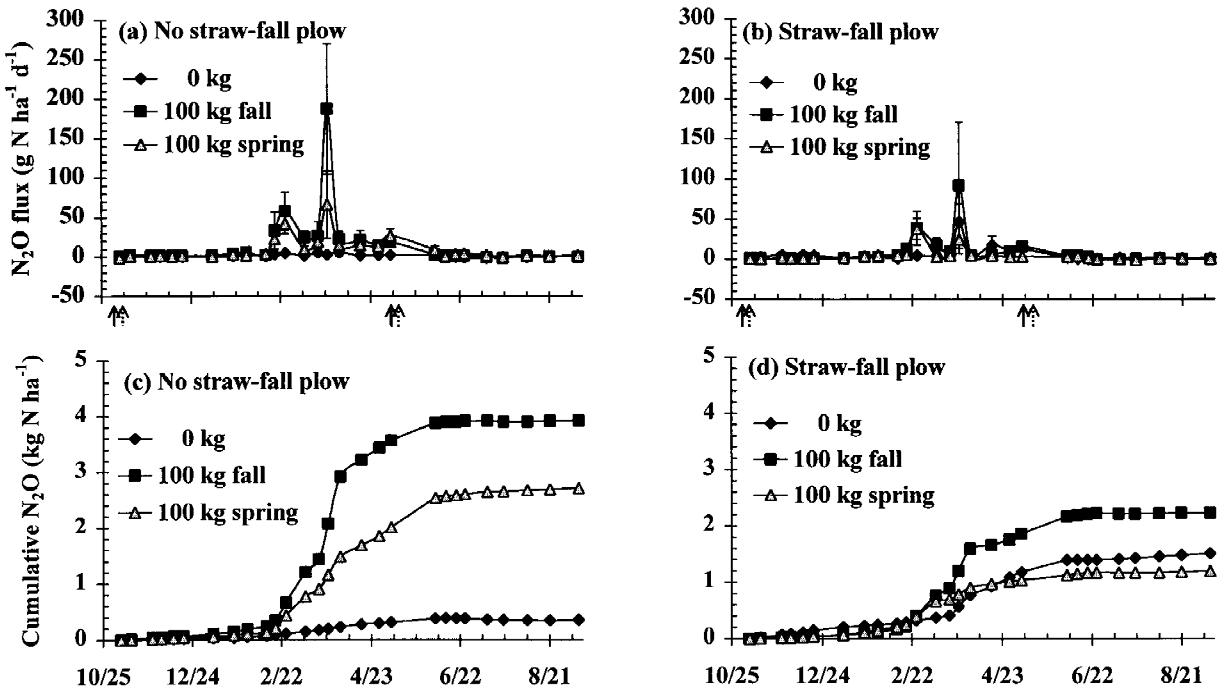


Figure 2. Effect of N fertilizer application timing during 1996–1997 on mean daily N_2O flux (a and b) and on cumulative N_2O emissions (c and d) measured on the fall plow treatments where straw was removed (a and c) and not removed (b and d). (Solid arrows point to the dates of fertilizer application and dotted arrows point to the dates of tillage operation.)

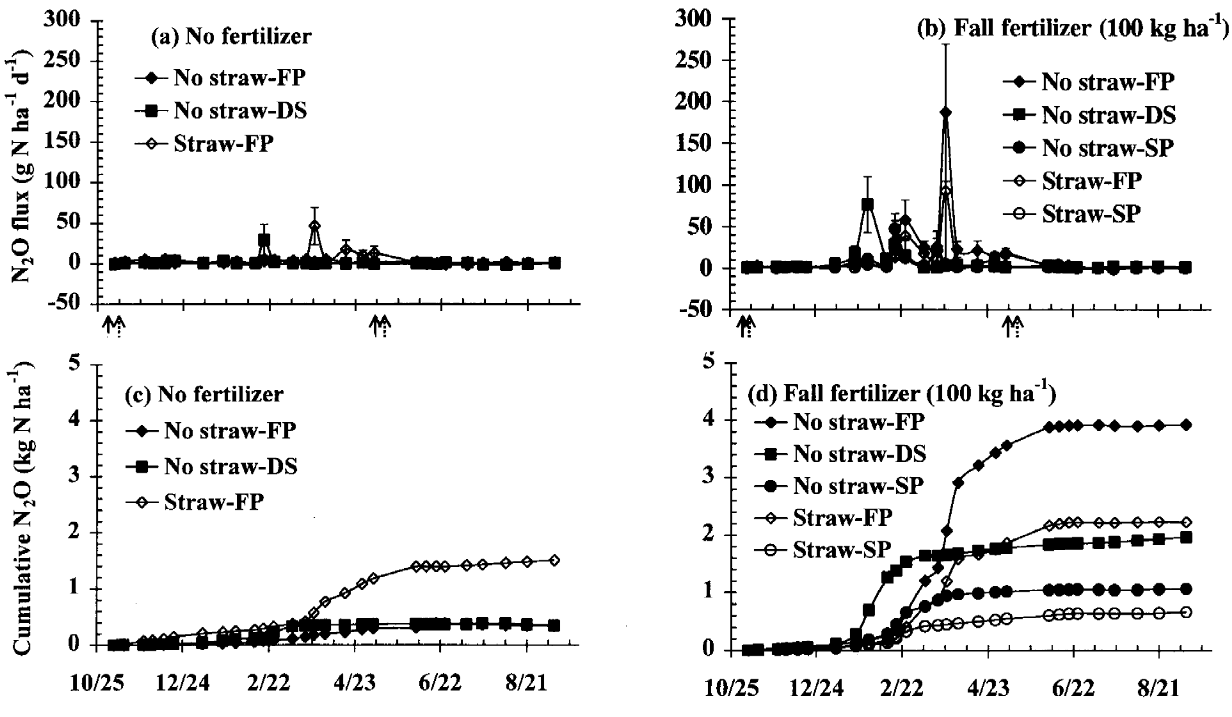


Figure 3. Effect of straw-tillage practices during 1996–1997 on mean daily N_2O flux (a and b) and cumulative N_2O emissions (c and d) measured on treatments with zero (a and c) and 100 kg N fertilizer applied in the fall (b and d). (Solid arrows point to the dates of fertilizer application and dotted arrows point to the dates of tillage operation.)

the N available for denitrification and nitrification, thus reducing N_2O production in the short term. The amount of N_2O produced reflects the combined effect of all these processes.

When 100 kg N ha^{-1} fertilizer was applied, however, the treatments with straw incorporated into the soil for both fall and spring tillage had about half the N_2O emission of those plots where straw was removed (Figures 2 and 3 and Table 3). In contrast to unfertilized treatments, straw incorporation caused much lower N_2O emissions compared to straw removed treatments. If a fixed proportion of straw N is converted to N_2O after incorporation into the soil (IPCC: Intergovernmental Panel on Climate Change, 1997), then more N_2O should be emitted from treatments with straw retained (especially at higher rates of N fertilization where straw N inputs are more appreciable). Since this was not observed, even in an experiment which had been in place for over 10 years, it appears that the proportion of straw N converted to N_2O is not the same as, and may be considerably less than, the proportion of fertilizer N converted to N_2O.

Effect of tillage

The effects of tillage were only studied for plots where fertilizer was applied in the fall. Spring plow produced the lowest average daily N_2O flux followed by direct seeding (but the difference was not significant). However, fall plow did produce significantly higher N_2O emissions (Figure 3 and Table 3) than the other two tillage treatments.

Several factors contribute to the higher N_2O emission from fall plow. First, tilling the soil in the fall incorporates the applied N fertilizer with the topsoil and thereby increases the amount of N available in soil for nitrification and denitrification in the winter and early spring. Fall plow also stimulates N mineralization, which further increases the amount of N available in soil as reported earlier for this field by Carefoot and Janzen (1997). Second, tilling the soil in the fall incorporates straw residue into soil and its decomposition consumes oxygen and makes the soil more anaerobic. Although the water-filled porosity was lower for fall plow, localized anaerobic sites could occur due to the decomposition of fresh straw residue. This is possible even for plots with straw removed, since baling does not remove all residue (especially standing stubble, leaves and chaff). Third, soil temperature was also higher for fall plow than spring plow or direct seeding treatments over the winter and spring months. Higher temperature increases denitrification and nitrification because biological activity is temperature-dependent.

The slightly higher N_2O emission from direct seeding over spring plowing probably reflects the greater water-filled porosity in these soils during spring. Although some researchers found that direct seeding produced higher N_2O emissions than conventional tillage (Aulakh et al., 1984; Hilton et al., 1994; MacKenzie et al., 1997; Palma et al., 1997), their N_2O emission data were collected during the growing season. The timing of conventional tillage operations in these experiments was not studied. Our study suggests that the timing of tillage operations (fall or spring) may have a greater influence on N_2O emissions than the tillage method (e.g., spring soil disturbance by direct seeding or pre-seeding tillage).

Implications

Nitrogen fertilizer is essential to grain production throughout the Canadian Prairies, especially on irrigated cropland. Many farmers in this area apply N fertilizer in the fall after harvest, when there is less demand on labor and machinery and better fertilizer prices. But this study shows that fall N fertilization can lead to higher N_2O emissions compared to those from spring fertilization or from unfertilized soil. Thus, N fertilizer should be applied in the spring.

The removal of straw as a raw material for production of fiber and energy reduces the amount of organic matter and nutrients returned back to soil. The results of this study provide an additional argument for limiting this practice. Removal of straw from fertilized soil increased N_2O emissions.

Tillage also had an impact on N_2O emission. Tillage effects are not only related to the method of tillage, but also when the tillage operation was conducted. Spring plow resulted in the lowest N_2O emission among the three tillage practices studied.

The N_2O emission from agricultural land has been estimated based upon the amount of N added to the soil without giving consideration to management practices (IPCC: Intergovernmental Panel on Climate Change, 1997). The IPCC assumes 1.25% of applied N (from fertilizer application or straw N addition) is lost as N_2O, implying a linear relationship between N_2O loss and the amount of N returned to the soil. As demonstrated in this study, N_2O emissions from agricultural land also depend on the timing of fertilizer application (fall vs. spring) and the tillage/straw management practices. Thus estimates of N_2O emissions based on the 'fraction of applied N' approach may be tenuous.

Acknowledgments

The authors gratefully acknowledge the dedicated technical assistance of Greg Travis, Elaine Nakonechny, Tony Curtis and Brett Hill in carrying out the field and laboratory work for this study. We also thank the AARI for their financial support.

References

Anthony WH, Hutchinson GL & Livingston GP (1995) Chamber measurement of soil-plant-atmosphere gas exchange: linear vs diffusion-based flux models. Soil Sci Soc Am J 59: 1308–1310

Aulakh MS, Rennie DA & Paul EA (1984) Gaseous nitrogen losses from soils under zero-till as compared with conventional-till management systems. J Environ Qual 13: 130–136

Burford JR & Bremner JM (1975) Relationship between the denitrification capacities of soils and total water soluble and readily decomposable soil organic matter. Soil Biol Biochem 7: 389–394

Carefoot JM, Janzen HH & Lindwall CW (1994) Crop residue management for irrigated cereals on the semi-arid Canadian prairies. Soil Tillage Res 32: 1–20

Carefoot JM & Janzen HH (1997) Effect of straw management, tillage timing and timing of fertilizer nitrogen application on the crop utilization of fertilizer and soil nitrogen in an irrigated cereal rotation. Soil Tillage Res 44: 195–210

Chang C, Cho cm & Janzen HH (1998) Nitrous oxide emission from long-term manured soils. Soil Sci Soc Am J 62: 677–682

Chen Y, Tessier S, MacKenzie AF & LaverdiPre MR (1995) Nitrous oxide emission from an agricultural soil subject to different freeze-thaw cycles. Agric Ecosys Environ 55: 123–128

Cicerone RJ (1987) Changes in stratospheric ozone. Science 237: 35–42

Clayton H, McTaggart IP, Parker J & Swan L (1997) Nitrous oxide emissions from fertilised grassland: A 2-year study of the effects of N fertilizer form and environmental conditions. Biol Fertil Soils 25: 252–260

Dormaar JF & Carefoot JM (1998) Effect of straw management and nitrogen fertilizer on selected soil properties as potential soil quality indicators of an irrigated Dark Brown Chernozemic soil. Can J Soil Sci 78: 511–517

Eichner MJ (1990) Nitrous oxide emissions from fertilized soils: Summary of available data. J Environ Qual 19: 272–280

Grace B & Hobbs EH (1986) The climate of the Lethbridge agricultural area: 1902-1985. LRS Mimeo Rep 3, Agric Can Res Stn, Lethbridge, AB Canada

Groffman PM (1985) Nitrification and denitrification in conventional and no-tillage soils. Soil Sci Soc Am J 49: 329–334

Heaney DJ, Nyborg m, Solberg ED, Malhi SS & Ashworth J (1992) Over winter nitrate loss and denitrification potential of cultivated soils in Alberta. Soil Biol Biochem 24: 877–884

Henault C, Devis X, Page S, Justes E, Reau R & Germon JC (1998) Nitrous oxide emission under different soil and land management conditions. Biol Fertil Soils 26: 199–207

Hilton BR, Fixen PE & Woodard HJ (1994) Effects of tillage, nitrogen placement, and wheel compaction on denitrification rates in the corn cycle of a corn-oats rotation. J Plant Nutr 17: 1341–1357

Hutchinson GL & Mosier AR (1981) Improved soil cover method for field measurement of nitrous oxide fluxes. Soil Sci Soc Am J 45: 311–316

Intergovernmental Panel on Climate Change (1997) Revised 1996 IPCC guidelines for national greenhouse gas inventories: Reference manual. Chapter 4 Agriculture. Organization for Economic Corporation and Development (OECD) and the International Energy Agency (IEA) Paris

Jarvis SC, Barraclough D, Williams J & Rook AJ (1991) Patterns of denitrification loss from grazed grassland: Effect of N fertilizer inputs at different sites. Plant Sci 131: 77–88

Lemke RL, Izaurralde RC, Malhi SS, Arshad MA & Nyborg m (1998a) Nitrous oxide emissions from agricultural soils of the Boreal and Parkland regions of Alberta. Soil Sci Soc Am J 62: 1096–1102

Lemke RL, Izaurralde RC & Nyborg m (1998b) Seasonal distribution of nitrous oxide emission from soils in the Parkland region. Soil Sci Soc Am J 62: 1320–1326

Loro PJ, Bergstrom DW & Beauchamp EG (1997) Intensity and duration of denitrification following application of manure and fertilizer to soil. J Environ Qual 26: 706–713

McKenzie RH, & Kryzanowski L K (1993) Fertilizing irrigated grain and oilseed crops. Agdex 100/541-1 Alberta Agriculture Edmonton, Alberta, Canada

MacKenzie AF, Fan MX & Cadrin F (1997) Nitrous oxide emission as affected by tillage, corn-soybean-alfalfa rotations and nitrogen fertilization. Can J Soil Sci 77: 145–152

Mosier AR & Schimel DS (1991) Influence of agricultural nitrogen on atmospheric methane and nitrous oxide. Chem Ind 23: 874–877

Mulvaney RL Khan SA & Mulvaney CS (1997) Nitrogen fertilizers promote denitrification. Biol Fertil Soils 24: 211–220

Nyborg M, Malhi SS & Solberg ED (1990) Effect of date of application on the fate of ^{15}N-labelled urea and potassium nitrate. Can J Soil Sci 70: 21–31

Nyborg m, Ladlaw JW, Solberg ED & Malhi SS (1997) Denitrification and nitrous oxide emissions from a Black Chernozemic soil during spring thaw in Alberta. Can J Soil Sci 77: 153–160

Palma RM, Rimolo m, Saubidet MI & Conti ME (1997) Influence of tillage systems on denitrification in maize-cropped soils. Biol Fertil Soils 25: 142–146

Ramos C (1996) Effect of agricultural practices on the nitrogen losses to the environment. Fertil Res 43: 183–189

Sahrawat KL & Keeney DR (1986) Nitrous oxide emission from soils. Adv Soil Sci 4: 103–148

SAS Institute (1990) SAS/STAT User's guide. 4th edn SAS Institute Inc, Cary, NC

Skiba U, Smith KA & Fowler D (1993) Nitrification and denitrification as sources of nitric oxide and nitrous oxide in a sandy soil. Soil Biol Biochem 25: 1527–1536

Thornton FC & Valente RJ (1996) Soil emissions of nitric oxide and nitrous oxide from no-till corn. Soil Sci Soc Am J 60: 1127–1133

Xu C, Shaffer MJ & Al-Kaisi m (1998) Simulating the impact of management practices on nitrous oxide emissions. Soil Sci Soc Am J 62: 736–742

Zegelin SJ, White I & Jenkins DR (1989) Improved field probes for soil water content and electrical conductivity measurement using time domain reflectometry. Water Resour Res 25: 2367–2374

Zhengping W, Liantie L, van Cleemput O & Baert L (1991) Effect of urease inhibitors on denitrification in soil. Soil Use Man 7: 230–232

Exhibit J

Nutrient Cycling in Agroecosystems **52:** 151–163, 1998.

Nitrous oxide emissions from agricultural fields during winter and spring thaw as affected by management practices

C. Wagner-Riddle & G.W. Thurtell
Department of Land Resource Science, University of Guelph, Guelph, Ontario, Canada, N1G 2W1

Key words: flux-gradient method, gaseous losses from manure applications, spring thaw N_2O emissions, nitrogen cycle

Abstract

Highest rates of N_2O emissions from fertilized as well as natural ecosystems have often been measured at spring thaw. But, it is not clear if management practices have an effect on winter and spring thaw emissions, or if measurements conducted over several years would reveal different emission patterns depending on winter conditions. In this study, we present N_2O fluxes obtained using the flux-gradient approach over four winter and spring thaw periods, spanning from 1993 to 1996, at two locations in Ontario, Canada. Several agricultural fields (bare soil, barley, soybean, canola, grass, corn) subjected to various management practices (manure and nitrogen fertilizer addition, alfalfa ploughing, fallowing) were monitored. Nitrous oxide emissions from these fields from January to April over four years ranged between 0 and 4.8 kg N ha^{-1}. These thaw emissions are substantial and should be considered in the nitrous oxide budgets in regions where thaw periods occur. Our study indicates that agricultural management can play a role in mitigating these emissions. Our data show that fallowing, manure application and alfalfa incorporation in the fall lead to high spring emissions, while the presence of plants (as in the case of alfalfa or grass) can result in negligible emissions during thaw. This presents an opportunity for mitigation of N_2O emissions through the use of over-wintering cover crops.

Introduction

Uncertainties in the global nitrous oxide budget (Watson et al., 1990) have not only been related to the lack of N_2O flux measurement programmes, but also to the high spatial and temporal variability of fluxes. Because agriculture is estimated to contribute 80% of the anthropogenic N_2O emissions to the atmosphere (Isermann, 1994), a need for continuous N_2O flux measurements under various agricultural management practices has been identified (Mosier et al., 1996).

Highest rates of N_2O emissions from agricultural and natural ecosystems have often been measured at spring thaw (Goodroad and Keeney, 1984a; Goodroad et al., 1984; Cates and Keeney, 1987; Flessa and Dörsch, 1995; Wagner-Riddle et al., 1997). Flessa and Dörsch (1995) observed frost-induced releases at four agricultural sites, and suggested that this could be a general phenomenon for soils in temperate and boreal climates. No significant difference in the thaw-induced emission from extensively and intensively managed plots was observed during that study. In a study of two maize fields managed at two nitrogen levels, Goodroad et al. (1984) also did not observe a significant effect of management on spring thaw N_2O emission.

Field and laboratory studies have demonstrated that freeze/thaw cycles induce high N_2O production in soils (Goodroad and Keeney, 1984b; Christensen and Tiedje, 1990). With the intention of simulating typical emissions from freeze/thaw cycles for regions of continental climate, Chen et al. (1995) observed increased emissions with increased number of freeze/thaw cycles applied to soil cores. Because field N_2O fluxes have not been monitored over several winter and spring thaws, it is unclear if field conditions would always be conducive to increased emissions with increased number of freeze/thaw cycles.

Among the methods available for N_2O flux monitoring, micrometeorological methods are ideally suited for continuous monitoring, providing spatially integrated fluxes from large areas with minimum disturbance of atmospheric conditions (Denmead and Raupach, 1993). In addition, the ability to provide hourly N_2O fluxes over periods when diurnal soil temperatures cycle between freezing and thawing is important in measurement programmes aimed at quantifying N_2O emissions due to winter and spring thaw.

In this study, we present N_2O fluxes obtained using the flux-gradient method over four winter and spring thaw periods, spanning from 1993 to 1996. Our objective was to quantify N_2O emissions from several agricultural fields, and identify management practices (manure and nitrogen fertilizer addition, alfalfa ploughing, and fallowing) that contribute to increased emissions. The effect of winter conditions (snow cover, freeze/thaw cycles), and soil conditions (nitrate and ammonium concentration, and temperature) on winter and spring thaw N_2O emissions are discussed.

Material and methods

Nitrous oxide fluxes were measured at the Elora Research Station (43°49'N, 80°35'W, Conestogo silt loam), Ontario, Canada, from the end of March 1993, until the end of April 1996. In addition, N_2O fluxes were measured at the Arkell Research Station (43°30'N, 80°15'W, Burford loam), Ontario, Canada, from October 1995 to the end of April 1996. The soil characteristics for Elora were 32% sand, 52% silt, 16% clay, pH (H_2O)=7, 3.7% total organic C, and 0.3% total organic N. At Arkell, the soil was composed of 38% sand, 47% silt, 16% clay, pH (H_2O)=7.4, 2.1% total organic C and 0.2% total organic N.

Four 1-ha plots were monitored starting in 1993 at Elora: Plot (1) a bare soil fallowed for the second consecutive year to which fertilizer had last been added in September 1991, followed by a barley (*Hordeum vulgare* L.) crop fertilized with 75 kg N ha^{-1} of ammonium nitrate during each of May 1994 and May 1995; Plot (2) a bare soil fallowed for the second consecutive year to which liquid dairy cattle manure was applied in August 1992 (60 kg N ha^{-1}) and May 1993 (90 kg N ha^{-1}), followed by a soybean (*Glycine max* L.) crop during each of 1994 and 1995; (3) an alfalfa (*Medicago sativa* L.) plot established in May 1992, cut twice in the summers of 1992 and 1993 and ploughed down in October 1993, followed by a canola (*Brassica napus*) crop fertilized with 100 kg N ha^{-1} of ammonium nitrate during each of May 1994 and May 1995; and (4) an established 20-year old stand of Kentucky bluegrass (*Poa pratensis L.*) fertilized with 50 kg N ha^{-1} in May 1992 and May 1995. Two additional 2-ha plots were monitored at Elora starting in January 1994: Plot (5) a spring-ploughed corn crop receiving 100 Kg N ha^{-1} in ammonium nitrate in May 1994, followed by a no-till corn crop receiving 100 kg N ha^{-1} as anhydrous ammonia in June 1995; Plot (6) a fall-ploughed corn plot receiving 100 Kg N ha^{-1} in ammonium nitrate during each of May 1994 and May 1995. Crop residues were left on the soil surface after harvest, except for the barley plot where part of the straw was removed, and for fall-ploughing of corn and alfalfa. A summary of plots and management treatments is given in Table 1.

At Arkell, we measured N_2O fluxes from four 1-ha plots previously cropped with wheat (*Triticum aestivum* L.), consisting of: Plot (1) wheat stubble; Plot (2) oats no-till planted through wheat stubble; Plot (3) red clover no-till planted through wheat stubble; and Plot (4) control plot (wheat stubble ploughed-down). Plots 1, 2 and 3 received a fall application of liquid swine manure (75 kg N ha^{-1}), while plot 4 did not receive any treatments. Cover crop establishment was poor in plots 2 and 3, so that results from these plots were treated as replicates of plot 1, that is, as wheat stubble plots.

Nitrous oxide fluxes from each plot were calculated using the flux-gradient method:

$$\text{Flux} = -K \frac{\partial C}{\partial z}$$

where K (m^2 s^{-1}) is the eddy diffusivity of N_2O, and $\partial C/\partial z$ is the concentration gradient. Concentration gradients were estimated using a finite concentration difference, ΔC (ng N_2O m^{-3}), occurring over a vertical distance Δz(m). The eddy diffusivity compatible with Δz was estimated for each plot using a wind profile method as described by Wagner-Riddle et al. (1996). Cup anemometers were placed at four heights above each plot, and only hourly wind speeds larger than 1.5 m s^{-1} were considered for the calculation of eddy diffusivity.

The hourly concentration difference (ΔC in parts per trillion) over a height difference Δz (m) was measured using a Tunable Diode Laser Trace Gas Analyzer, TDLTGA (Edwards et al., 1994). Three

Table 1. Summary of plots and management practices monitored for N_2O flux at the locations of Elora and Arkell

Plot	Year	Description	Management practices
Elora			
1	1993	fallow, bare soil	last N fertilization September 1991
	1994, 1995	barley	75 kg N ha^{-1} NH_4NO_3 in May
2	1993	fallow + manure	liquid dairy cattle manure in August 1992 (60 kg N ha^{-1}) and May 1993 (90 kg N ha^{-1})
	1994, 1995	soybeans	-
3	1993	alfalfa	plough-down in September
	1994, 1995	canola	100 kg N ha^{-1} NH_4NO_3 in May
4	1993	grass	50 kg N ha^{-1} NH_4NO_3 in May
	1994	grass	-
	1995	grass	50 kg N ha^{-1} NH_4NO_3 in May
5	1994	spring-ploughed corn	100 kg N ha^{-1} anhydrous ammonia in May
	1995	no-till corn	100 kg N ha^{-1} NH_4NO_3 in May
6	1994, 1995	fall-ploughed corn	100 kg N ha^{-1} NH_4NO_3 in May
Arkell			
1	1995, 1996	wheat stubble	liquid swine manure (75 kg N ha^{-1}) in Oct
2	1995, 1996	oats no-till on wheat stubble	liquid swine manure (75 kg N ha^{-1}) in Oct
3	1995, 1996	red clover no-till on wheat stubble	liquid swine manure (75 kg N ha^{-1}) in Oct
4	1995, 1996	fall-ploughed wheat stubble	-

TDLTGA units were used during this study, one for each of: plots 1 to 4 at Elora, plots 5 and 6 at Elora, and plots 1 to 4 at Arkell. The wavenumber for the N_2O absorption line for both units used at Elora was 2233.333 cm^{-1}, and for the unit used at Arkell was 2236.2235 cm^{-1}. Air was drawn alternately every 5 s from two heights, typically spaced at 0.40 m above the plot surface, and then directed to a centrally located TDLTGA via approximately 70 m of tubing. The height of the lower air intake was 0.25 m for the bare soil plots, and a height within the range of 1.3 to 3 times the crop height for the vegetated plots. A site valve was used to select the plot to be sampled during each consecutive hour. The setup of sample air intakes, tubing, valves and pump are described in detail by Wagner-Riddle et al. (1997). During each measurement hour, an average N_2O concentration difference between intake heights was obtained for the monitored plot. With four 1-ha plots sampled sequentially (plots 1 to 4, at Elora and Arkell), this sampling scheme resulted in six hourly concentration differences for each plot during each measurement day. For the larger 2-ha plots cropped with corn, sequential sampling involved switching between 2 plots (plots 5 and 6 at Elora) resulting in twelve hourly concentration differences for each plot during each measurement day. Only concentration gradients measured when the wind direction at the adjacent weather station allowed for a fetch-to-height ratio of at least 50:1 (horizontal distance to height of measurement ratio) were used in the flux calculations. Due to the variable positioning of the sample intakes in the various plots this criteria resulted in a different number of total hourly or daily flux measurements in each plot.

Hourly N_2O fluxes were calculated using the concentration gradient and the eddy diffusivity as described above. For hourly fluxes, the TDLTGA has a resolution of approximately ± 10 ppt for the N_2O concentration gradient, which combined with an average eddy diffusivity of 0.05 m^2 s^{-1} would result in an error of approximately ± 2 ng m^{-2} s^{-1}. The resolution was further improved by averaging hourly fluxes to obtain daily mean fluxes.

Snow depth accumulated on each plot was measured with a ruler once every week, or as necessary after snowfalls. Hourly soil temperatures at 1, 10 and 20 cm were measured in each plot, except for 1996, using copper-constantan thermocouples encased in epoxy filled 20 cm-long copper tubing following procedure by Berard and Thurtell (1990). Rainfall and air temperatures were measured at the weather station located at the Elora research station. For 1996, hourly soil temperatures at 5, 10 and 20 cm were recorded at a weather station located at approximately 15 km from the Elora site and 2 km from the Arkell site.

Soil samples were collected from the Ap horizon during the fall of each year. Five soil cores (5 cm i.d. by 5 cm) were collected weekly at randomly selected locations at 2.5 cm below the soil surface within each plot. Moisture content was determined gravimetrically with 15 g moist soil. Moist soil (25 g) was extracted with 50 mL 0.5 *M* K_2SO_4 solution. Extract solutions were filtered, then frozen until analyzed for NH_4^+ and (NO_3^- + NO_2^-) (Tel and Heseltine, 1990).

Results and discussion

Several freeze/thaw periods occurred during January to April in 1994 and 1996, while 1995 presented one freeze/thaw cycle in January, followed by an extended cold period and a short thaw in March (Figures 1A, 2A and 3A). In 1993, the measurement period only started at the end of March, being limited to one final thaw period, and therefore, not discussed here in detail. During the freeze periods from 1994 to 1996, when the soil temperature was below 0 °C and the surface was covered by snow, daily N_2O fluxes from all plots, except the ploughed alfalfa, were small (<10 ng m^{-2} s^{-1}) (Figures 1, 2 and 3). Similar small fluxes were observed before the January freeze, that is during October to December in 1993, 1994 and 1995 (data not shown).

For the ploughed-down alfalfa, emissions were high throughout the fall of 1993 (data not shown), and the winter of 1994 (Figure 1B), increasing when air temperature increased, even though both air and soil temperatures were still below 0 °C. For example, during the freeze period from day 1 to 49 in 1994, daily fluxes averaged 38.8 ng m^{-2} s^{-1} (Table 2), with notable increases (>100 ng m^{-2} s^{-1}) during day 48 and 49, when daily mean air temperature was still averaging below 0 °C. Emissions from ploughed alfalfa continued high during the first thaw period in 1994 (day 50–53), presenting the highest average of all plots (Table 2) for that period. While N_2O emissions from alpine and subalpine snowpacks were less than 1 ng m^{-2} s^{-1} (Sommerfeld et al., 1993), Van Bochove et al. (1996) have estimated winter emissions of the order of 50 ng m^{-2} s^{-1} from a field under 60 cm of snow that had previously been cropped with barley and fertilized with 25 kg N ha^{-1}. Although we observed comparable emissions from the ploughed alfalfa plot, the other fields monitored showed fluxes of only 1 to 10 ng m^{-2} s^{-1}.

For most plots monitored, daily N_2O fluxes averaged less than 20 ng m^{-2} s^{-1} during the first thaw in 1994, 1995 and 1996 (Figures 1–3, Tables 2–4), with the exceptions of ploughed-down alfalfa in 1994, ploughed corn stubble plot in 1994 and 1995, and the ploughed-down wheat stubble and the wheat stubble that had received an application of liquid swine manure in the fall of 1995. The latter emissions at 90 ng m^{-2} s^{-1} for day 17 to 20 in 1996 were comparable to the ploughed alfalfa in 1994 (71.6 ng m^{-2} s^{-1}).

As subsequent freeze/thaw cycles occurred after the first thaw during all years, the soil temperature at 1 and 10 cm depth sequentially decreased and increased (Table 2–4). Average temperatures for the thaw periods did not always increase above 0 °C, but N_2O flux averages clearly increased during these periods. The exception was the over-wintering alfalfa (not ploughed) in 1993 (data not shown) and the grass plot during all years, which did not show a clear N_2O emission increase during any thaw periods.

The timing of emission peaks was coincidental for all plots presenting emission episodes, particularly in 1994 for fallow, manured fallow, ploughed alfalfa, standing and ploughed corn stubble (Figure 1B and C). Note that data are missing for the corn plots for most of March 1994 (days 60–90). The effect of the May 1993 manure addition to the fallow plot was still evident in the spring of 1994 with larger amplitude in the daily fluxes of the manured plot (up to 614 ng m^{-2} s^{-1}) when compared to bare soil (up to 370 ng m^{-2} s^{-1}). An exception to the timing pattern was the earlier decrease in emissions from the ploughed alfalfa and ploughed corn stubble, on day 105 when compared to day 115 for the other plots. This is evident when averages for the final thaw period (day 98–120) are compared among plots (Table 2).

In contrast to 1994, only two freeze/thaw periods were observed in 1995. As well, the final thaw period in 1995 was characterized by an initial sharp increase in temperature followed by a relatively cool period when the soil temperatures did not drop below 0 °C, and rainfall was low. Nitrous oxide emissions then decreased sharply after only 5 days during the final thaw

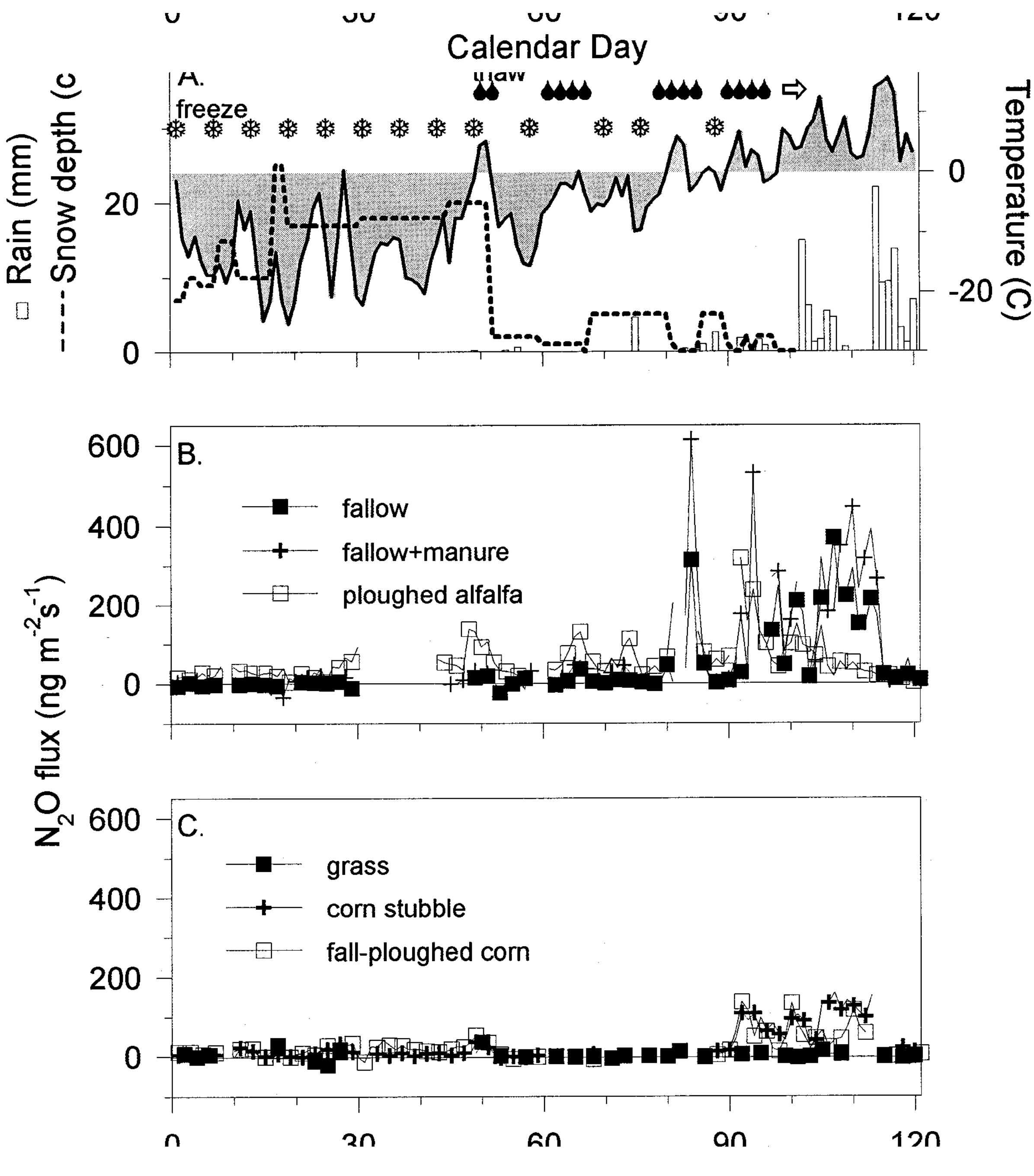


Figure 1. A. Snow depth on the ground, daily rainfall and daily air temperature recorded at Elora during January to April 1994. Freeze periods with air temperature below 0 °C and significant snow cover are indicated with the symbol ❄, and thaw periods with air temperature above 0 °C (including hourly values) and decreasing snow cover due to melting are indicated with the symbol 💧. The start of the final thawing period is indicated by the symbol ⇒. Daily N_2O flux values measured for the same period over B. fallow soil, fallow soil that received liquid dairy cattle manure in 1992 and 1993, and an alfalfa crop ploughed down in 1993, and C. bluegrass last fertilized in 1992, corn stubble, and corn stubble ploughed-down in 1993. Only every second data point measured is shown. Note that N_2O flux data are missing from day 70 to 85 for corn plots due to equipment failure.

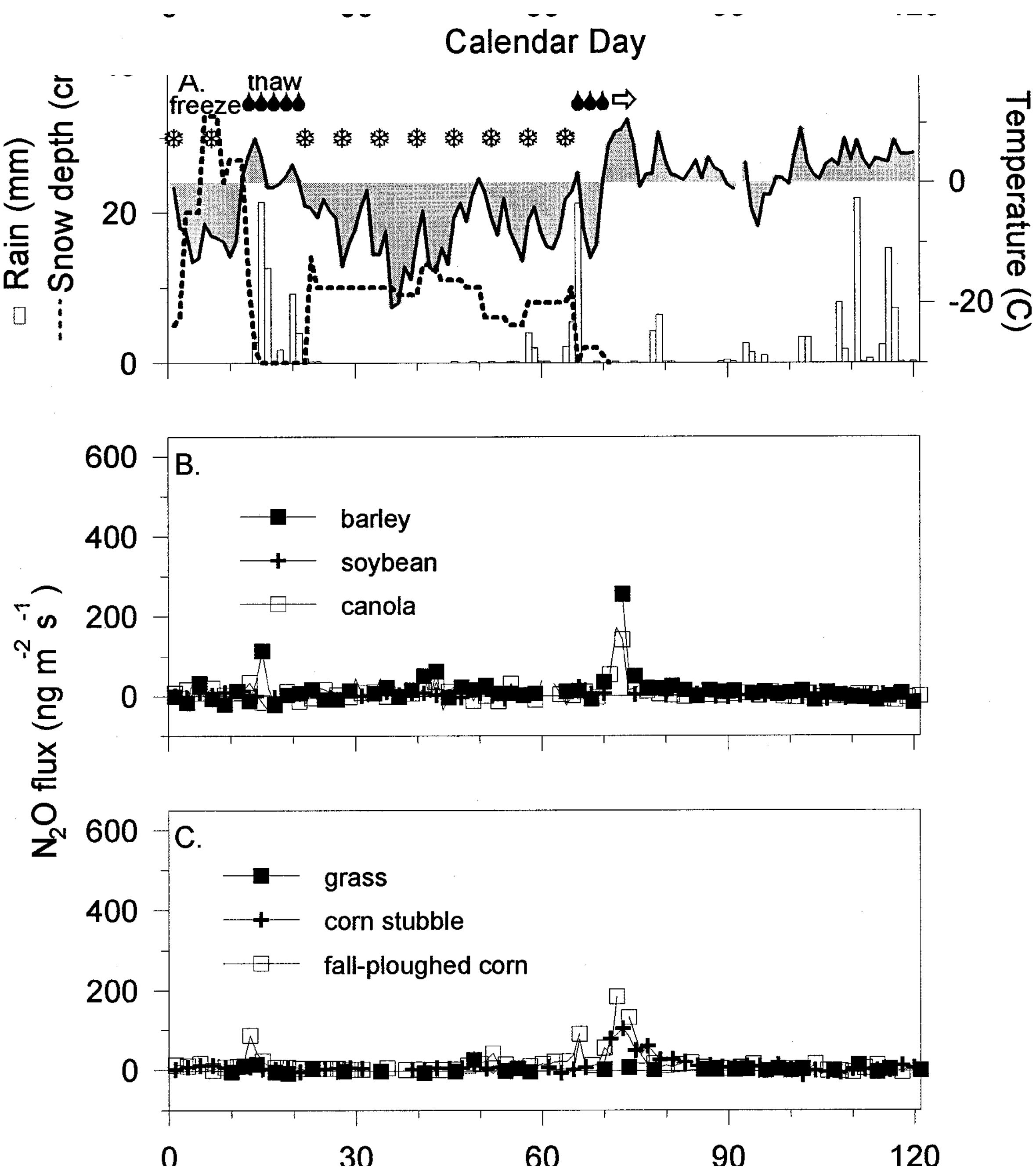


Figure 2. A. Snow depth on the ground, daily rainfall and daily air temperature recorded at Elora during January to April 1995. Freeze periods with air temperature below 0 °C and significant snow cover are indicated with the symbol ⊛, and thaw periods with air temperature above 0 °C (including hourly values) and decreasing snow cover due to melting are indicated with the symbol 🌢. The start of the final thawing period is indicated by the symbol ⇒. Daily N_2O flux values measured for the same period over B. plots cropped with barley, soybeans, and canola in 1994, and C. bluegrass last fertilized in 1994, corn stubble, and corn stubble ploughed-down in 1994. Only every second data point measured is shown.

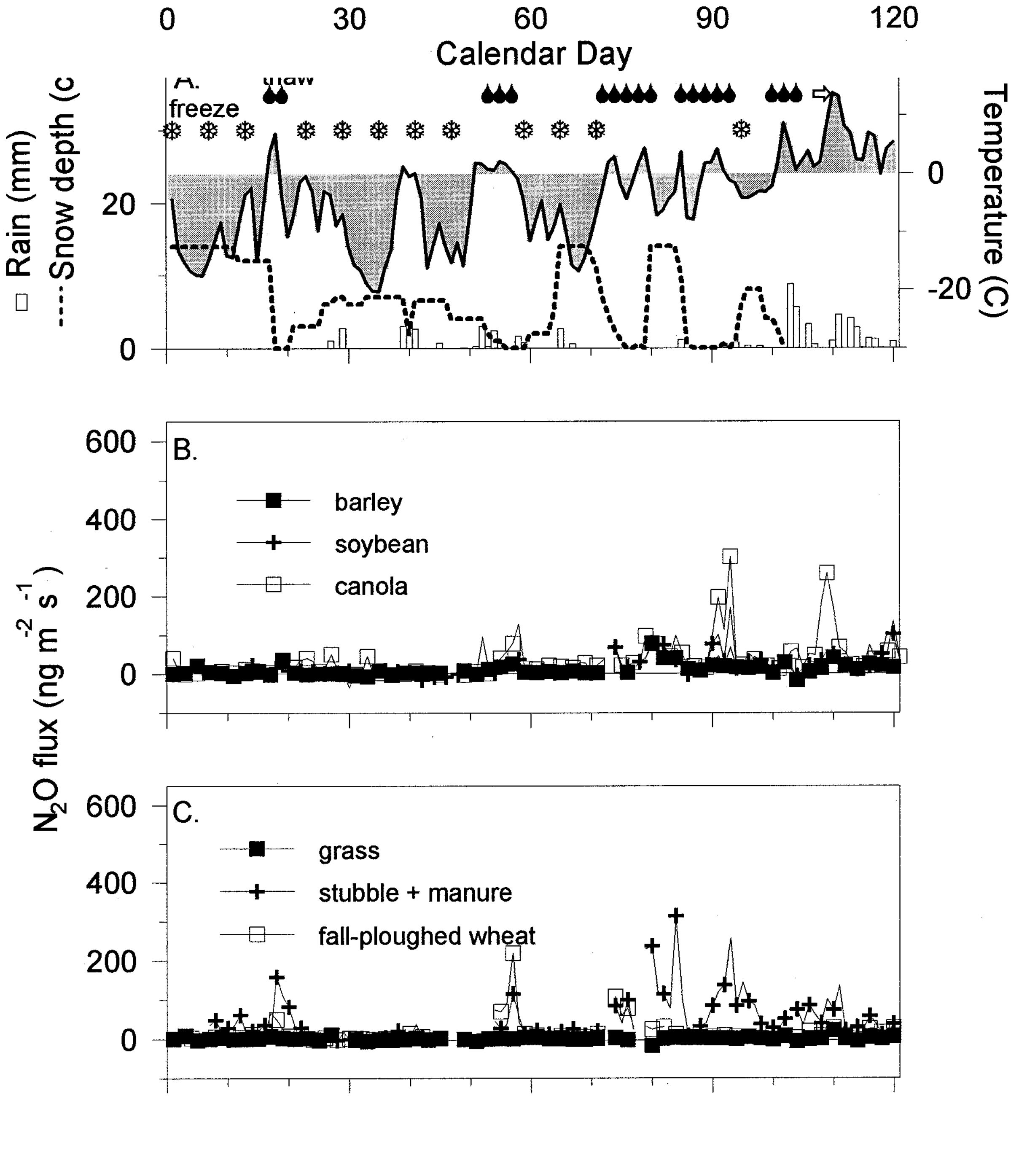


Figure 3. A. Snow depth on the ground, daily rainfall and daily air temperature recorded at Elora during January to April 1996. Freeze periods with air temperature below 0 °C and significant snow cover are indicated with the symbol ⊛, and thaw periods with air temperature above 0 °C (including hourly values) and decreasing snow cover due to melting are indicated with the symbol 💧. The start of the final thawing period is indicated by the symbol ⇒. Daily N_2O flux values measured for the same period over B. plots cropped with barley, soybeans, and canola in 1995, and C. bluegrass last fertilized in 1994, wheat stubble that received liquid swine manure in 1995, and wheat stubble ploughed-down in 1995. Only every second data point measured is shown.

Table 2. Mean N_2O flux values during freezing and thawing periods from January to April 1994 measured over fallow, fallow that received manure in 1993, fall-ploughed alfalfa, grass, corn stubble and fall-ploughed corn plot. All plots except grass refer to residues from the previous year's crop. Standard deviation of N_2O flux means, and number of hourly flux values averaged over each period are given in brackets. Thawing periods are characterized as temporary, and as a final thawing period. Mean temperature is given for each riod for air and for soil depths of 1 and 10 cm under fallow soil. Total N_2O–N loss from January to April 1994 is given in kg N ha^{-1}

	Period									
	First freeze	First thaw	Second freeze	Second thaw	Third freeze	Third thaw	Fourth freeze	Fourth thaw	Final Thaw	
Day of year	1–49	50–53	54–60	61–67	68–78	79–85	86–89	90–97	98–120	
Temperature (°C)										
air	-13.2	-0.3	-10.9	-2.8	-5.2	0.7	-0.7	1.9	6.8	
1 cm	-2.0	-1.4	-5.2	-1.1	-1.9	0.1	0.2	1.7	6.6	
10cm	-1.4	-1.1	-4.7	-1.3	-1.7	-0.3	0.1	1.1	6.1	
					N_2O Flux (ng m^2 s^{-1})					Total loss (kg N ha^{-1})
Plots										
fallow	0.9	2.1	7.9	14.2	5.6	78.3	17.8	57.5	149.0	2.627
	(10.4, 113)	(22.2, 14)	(5.73, 20)	(13.3, 26)	(5.1, 43)	(118.1, 25)	(22.3, 14)	(46.8, 27)	(104.9, 96)	
fallow+manure	8.2	19.6	9.5	36.1	16.3	265.8	20.1	164.8	198.2	4.837
	(13.6, 80)	(16.2, 16)	(18.6, 13)	(18.3, 19)	(14.6, 37)	(247.8, 18)	(22.7, 12)	(185.4, 23)	(144.1, 85)	
ploughed alfalfa	38.8	71.6	16.6	75.1	53.4	107.7	57.8	153.6	55.0	3.792
	(30.3, 156)	(33.7, 20)	(20.7, 21)	(38.2, 29)	(28.4, 53)	(77.0, 17)	(16.6, 14)	(94.7, 32)	(40.3, 117)	
grass	3.4	18.5	-2.1	-0.9	-0.8	4.1	-1.8	4.4	4.8	0.209
	(11.8, 67)	(25.3, 6)	(1.6, 11)	(0.4, 7)	(2.6, 15)	(5.4, 13)	(-, 3)	(2.7, 12)	(7.1, 41)	
corn stubble	7.7	11.3	-1.2		4.5		9.7	71.3	85.1	1.666
	(8.8, 363)	(12.9, 37)	(2.2, 56)		(-, 7)		(3.6, 10)	(32.6, 69)	(51.6, 167)	
ploughed corn stubble	15.4	25.4	-2.5		-7.4		6.0	69.8	47.2	1.334
	(12.0, 378)	(18.6, 42)	(2.3, 55)		(-, 8)		(2.6, 15)	(43.8, 70)	(38.2, 167)	

period, that is, on day 75 N_2O fluxes had decreased to values smaller than 50 ng m^{-2} s^{-1} (Figures 2B and C). Consequently, N_2O flux averages during the final thaw period were lower in 1995, when compared to 1994 and 1996, with total loss of N_2O–N during the January to April period lower for most plots during 1995 (Table 3). A direct comparison of the effect of years is possible for the corn stubble and ploughed corn stubble plots. While in 1994 total N_2O–N losses from January to April for these plots were, respectively, 1.666 and 1.334 kg N ha^{-1} (Table 2), the losses over the same period in 1995 were 0.523 and 0.924 kg N ha^{-1}. Note also that flux data for corn are missing for the third and second thaw in 1994, due to equipment problems. If average N_2O fluxes from the fourth thaw period are assumed for these periods, N_2O–N total losses for 1994 are underestimated by approximately 0.27 kg N ha^{-1}. Therefore, an increased number of freeze/thaw cycles in 1994 when compared to 1995 (four versus two) resulted in an approximate doubling of N_2O emissions.

The 1996 winter and spring seasons presented a total of five cycles between freezing and thawing temperatures (Figure 3A and Table 4). While emissions increased and decreased during each consecutive freeze/thaw cycle up to day 80, emissions continued high during the fourth freeze when air temperature decreased to -5.3 °C, snow depth was larger than 10 cm, but soil temperatures stayed around the 0 °C level (Table 4). Similar results were observed for the fifth freeze cycle that occurred between day 95 and 99. Daily N_2O fluxes from all plots in 1996 were not as high as recorded for the two fallow plots in 1994, but total N_2O–N losses for the wheat stubble that received liquid swine manure in the fall of 1995 were comparable (Tables 2 and 4). In addition, soybean and canola plots also recorded higher N_2O emissions in 1996 when compared to 1995.

Table 3. Same as for Table 2 but for January to April 1995 over barley, soybean, canola, grass, corn, and ploughed corn stubble plots. Mean soil temperatures at 1 and 10 cm measured under the barley plot

	Period					
	First freeze	First thaw	Second freeze	Second thaw	Final Thaw	
Day of year	1–12	13–21	22–65	66–70	71–120	
Temperature (° C)						
air	-8.2	2.2	-8.1	-5.9	3.0	
1 cm	0.0	2.1	-1.0	-1.3	3.3	
10 cm	0.2	2.2	-0.5	-1.0	3.2	
			N_2O Flux (ng m^2 s^{-1}			Total loss (kg N ha^{-1})
Plots						
barley	8.0 (20.9, 37)	18.7 (39.3, 39)	8.7 (20.3, 149)	11.1 (16.5, 17)	16.1 (39.3, 212)	0.828
soybean	4.2 (6.0, 45)	1.3 (8.3, 18)	2.5 (6.0, 118)	11.4 (7.7, 17)	2.6 (8.1, 129)	0.197
canola	9.8 (6.3, 60)	-3.3 (21.2, 40)	8.9 (15.4, 183)	8.2 (12.0, 22)	10.9 (31.7, 229)	0.585
grass	0.0 (8.0, 13)	1.8 (8.4, 28)	-1.2 (8.8, 60)	6.4 (7.7, 8)	1.0 (3.9, 105)	0.026
corn	7.0 (3.5, 111)	5.5 (5.5, 49)	2.5 (4.9, 295)	22.8 (28.8, 23)	11.9 (23.1, 357)	0.523
ploughed corn stubble	9.4 (6.3, 104)	21.5 (27.3, 64)	10.1 (11.8, 241)	42.1 (38.1, 21)	14.4 (32.5, 323)	0.924

Table 4. Same as for Table 2 but for January to April 1996 over barley, soybean, canola, grass, wheat stubble with fall-manure, and ploughed wheat stubble plots. Mean soil temperature at 5 and 10 cm measured under grass at the weather station

	Period											
	First freeze	First thaw	Second freeze	Second thaw	Third freeze	Third thaw	Fourth freeze	Fourth thaw	Fifth freeze	Fifth thaw	Final Thaw	
Day of year	1–16	17–20	21–51	52–58	59–71	72–80	81–84	85–94	95–99	100–104	105–120	
Temperature (°C)												
air	-11.1	-0.5	-8.6	0.7	-9.2	0.1	-5.3	-0.7	-3.1	2.9	5.4	
5 cm	-5.2	-0.4	-5.0	-0.4	-3.1	-1.1	-0.2	0.0	-0.2	1.5	4.6	
10 cm	-4.5	-0.5	-4.5	-0.4	-2.6	-1.0	-0.1	0.0	0.0	1.0	3.9	
						N_2O Flux (ng m^2 s−1)						Total loss (kg N ha^{-1})
Plots												
barley	6.5 (6.9, 71)	11.5 (18.1, 21)	1.1 (4.4, 143)	30.3 (29.9, 30)	6.7 (3.5, 61)	34.3 (33.5, 16)	40.9 (9.5, 22)	22.5 (19.9, 51)	18.1 (3.4, 22)	9.1 (17.4, 21)	19.6 (11.3, 75)	0.897
soybean	7.8 (6.8, 50)	13.2 (6.2, 18)	1.4 (6.4, 137)	16.0 (14.3, 24)	5.4 (3.2, 49)	37.6 (27.7, 25)	56.0 (21.1, 20)	46.0 (54.3, 40)	27.2 (13.7, 21)	6.0 (8.9, 15)	36.6 (31.3, 75)	1.196
canola	9.6 (10.4, 50)	17.8 (12.9, 21)	8.0 (17.9, 147)	45.7 (46.4, 30)	19.2 (5.8, 66)	47.3 (29.4, 29)	74.9 (21.9, 22)	86.4 (95.8, 48)	33.4 (13.0, 17)	41.7 (24.5, 16)	78.3 (71.0, 81)	2.343
grass	1.5 (2.7, 67)	3.3 (2.9, 22)	0.0 (3.7, 129)	0.6 (1.5, 23)	1.1 (2.2, 59)	-0.6 (8.1, 22)	3.8 (2.8, 21)	2.8 (2.9, 51)	-0.2 (5.9, 22)	0.1 (4.7, 23)	3.7 (6.1, 76)	0.084
stubble + manure	28.4 (15.9, 52)	90.0 (61.6, 18)	8.7 (11.6, 102)	56.3 (34.8, 18)	18.0 (5.9, 58)	119.3 (92.7, 24)	156.1 (107.1, 15)	90.1 (72.1, 47)	80.0 (45.9, 21)	45.7 (22.3, 11)	50.0 (34.5, 79)	3.156
ploughed wheat stubble	6.6 (3.2, 41)	36.9 (25.7, 21)	3.1 (4.0, 101)	99.0 (69.0, 18)	9.4 (4.9, 61)	55.6 (32.6, 21)	24.4 (10.3, 13)	9.1 (7.0, 45)	3.9 (1.4, 19)	6.2 (2.3, 15)	18.9 (8.6, 75)	1.163

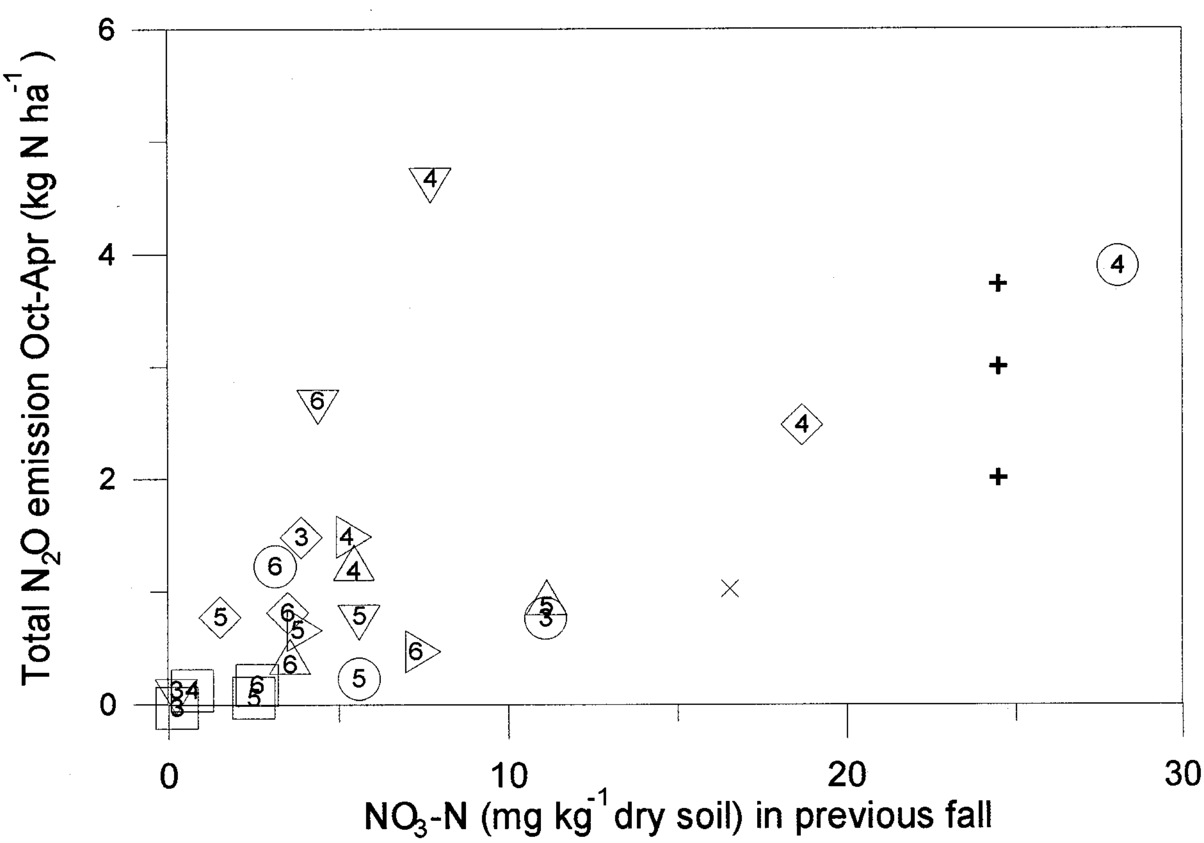


Figure 4. Total N_2O emission accumulated between January and April of each studied year for all plots studied as a function of the soil nitrate concentration in the previous fall. Soil sampling dates varied between September and November. Data points for Elora are shown with: ◇ plot 1 (fallow/barley), ○ plot 2 (manure/soybean), ▽ plot 3 (ploughed alfalfa/canola), □ plot 4 (grass), ▹ plot 5 (spring-ploughed corn/no-till), and △ plot 6 (fall-ploughed corn). Data points for Arkell 1996 are shown with: + plot 1, 2, 3 (wheat stubble + manure) and × plot 4 (ploughed stubble). Labels on each data point indicate the winter and spring thaw year at Elora (3=93, 4=94, 5=95 and 6=96).

From our results it is clear that both freeze/thaw cycles and management practices play a role in the magnitude of winter and spring emissions. The total N_2O–N emissions from October to April in each studied year were correlated to the nitrate concentration in the soil measured during the previous fall (r=0.70, Figure 4). For vegetated plots, such as grass, or alfalfa before ploughing in 1993, soil nitrate concentrations in the fall were less than 5 mg kg^{-1} dry soil and total N_2O–N emissions less than 0.2 kg N ha^{-1}. In the other extreme, the fallow and manured plots presented NO_3–N soil concentrations in the fall that were higher than 20 mg kg^{-1} dry soil, contributing to the larger than 2 kg N ha^{-1} of N_2O–N loss between January and April of the following year. The only plot that did not fit this pattern very well was plot 3, where an alfalfa crop was ploughed down in the fall of 1993. Although NO_3–N soil concentrations were less than 10 mg kg^{-1} dry soil, N_2O–N losses during October to April in 1994 and 1996 exceeded 2 kg N ha^{-1}. Rapid cycling between nitrogen in the organic form and NO_3 could explain the low NO_3 detected in the soil solution. Surprisingly, organic N from alfalfa ploughing in 1993 still seemed to be affecting N_2O emissions two years after incorporation into the soil.

It is believed that increased emissions due to freeze/thaw cycles are a combination of physical release of N_2O, and N_2O production in the surface soil, and N_2O diffusion from the subsurface soils (Goodroad and Keeney, 1984b). Christensen and Christensen (1991) showed that organic matter becomes available for denitrification when microbes are subjected to a killing freeze and aggregates are disintegrated due to freeze/thaw cycles. Flessa and Dörsch (1995) concluded that the marked increase in N_2O emissions was very likely due to increased denitri-

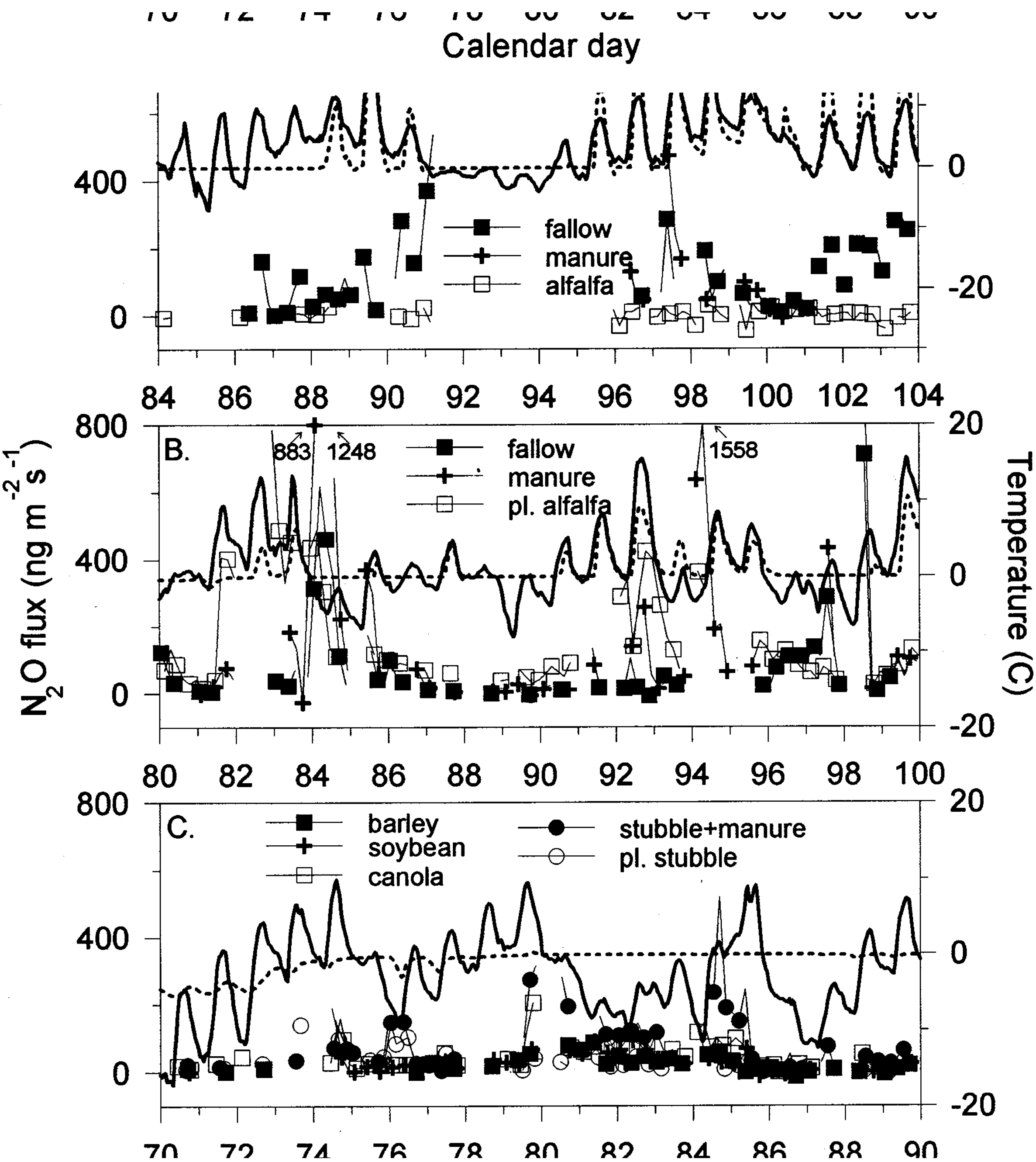


Figure 5. Hourly air (solid line) and soil temperatures at 1 cm depth (or 5 cm for 1996) (dotted line) for the fallow or barley residue plot (or weather station for 1996), and hourly N_2O fluxes for A. fallow, manured fallow, and alfalfa crop in 1993; B. fallow, manured fallow, and ploughed alfalfa in 1994; and C. plots previously cropped with barley, soybeans, canola, and wheat (including manure stubble and ploughed-down stubble), in 1996. Labels for C. indicate values that are off-scale. Only every second data point measured is shown.

fication activity in the uppermost thawed soil layer. During their study year, peak emissions were observed during the first freeze/thaw cycle, and citing Melin and Nommik (1983) they suggested that the suppression of nitrous oxide reductase at low temperatures could explain the initial peak observed.

In contrast, we observed N_2O fluxes peaking at the third or fourth freeze/thaw cycle, during some years, respectively in 1994 and 1996. Upon close examination of the hourly N_2O fluxes and air and soil temperatures, it can be seen that maximum N_2O fluxes occurred frequently during periods of cooling or minimum temperatures (Figure 5). Typically, a three to four-day period of increasing air temperatures, with soil temperatures increasing above freezing, followed by a cold period when soil temperatures returned to 0 °C, resulted in increased fluxes. Peaks occurred during the cooling phase of the diurnal temperature wave, as on day 91 (5:00 h) in 1993 (Figure 5A), days 84 (6:00 h) and 94 (7:00 h) in 1994 (Figure 5B), and days 76 (9:00 h) and 79 (21:00 h) in 1996 (Figure 5C). But also on the warming phase of the diurnal temperature wave, as on days 97 (10:00 h) in 1993, days 85 (10:00 h), 94 (7:00 h), 97 (13:00 h) and 98 (13:00 h) in 1994, and day 84 (17:00 h) in 1996. Two possible mechanisms might be playing a role for this lack of coupling between the soil temperature and the occurrence of N_2O emission episodes: (1) a shortage of substrate for denitrification (carbon and/or nitrate) may be limiting the production of N_2O as temperatures increase on the ascending phase of the temperature wave; and (2) a suppression of nitrous oxide reductase at low temperatures (Melin and Nommik, 1983) may be delaying the reduction of N_2O produced during the ascending phase of the temperature wave. This second mechanism may be also allowing for the accumulation of N_2O once temperatures decrease and the reduction of N_2O to N_2 stops on the descending part of the temperature wave. It is also obvious that level of nitrate in soils was limiting N_2O production by denitrification, since plots with low nitrate levels, such as alfalfa in 1993 (Figure 5A) did not present any increased fluxes. In addition, Figure 5 clearly shows that emissions occur after a change in hourly temperature from above freezing to below freezing, or vice-versa, but that these cycles do not always result in high emissions even in plots with high nitrate levels.

Conclusions

Nitrous oxide emissions from agricultural fields from January to April over four years in Ontario, Canada, ranged between 0 and 4.8 kg N ha^{-1}. Compared to yearly estimates ranging between 0.5 and 4.1 kg N ha^{-1} based on fertilizer applications (Wagner-Riddle et al., 1997), these thaw emissions are substantial and should be considered in the nitrous oxide budgets in regions where thaw periods occur.

While other studies have measured high emissions during soil thaw, our study indicates that agricultural management can play a role in mitigating these emissions. Our data show that fallowing, manure application and alfalfa incorporation in the fall lead to high spring emissions, while the presence of plants (as in the case of alfalfa and grass) can result in negligible emissions during thaw. This presents an opportunity for mitigation of N_2O emissions through the use of over-wintering cover crops.

Emissions from plots cultivated with crops that received nitrogen fertilizer additions at planting time were much smaller than those resulting from fall applications of manure or alfalfa on uncropped plots, but it is not clear from our results if a fall nitrogen application on a vegetated plot would also result in high thaw emissions.

Hourly fluxes monitored in this study indicated a lack of coupling between the N_2O emission peaks and soil temperature maxima. In addition, the temporal variability of N_2O fluxes on an hourly basis highlighted the importance of continuous monitoring for the quantification of seasonal emission totals.

Acknowledgements

This research project was funded by grants from the Environmental Research Program of the Ontario Ministry of Environment and Energy, the Natural Sciences and Engineering Research Council of Canada, the Ontario Ministry of Agriculture, Food and Rural Affairs, and by research contracts through the Green Plan Initiative/Agriculture Canada. The technical assistance of Mr. R. Sweetman was invaluable for the conduction of the field experiments. The assistance of Ms. Adriana Furon in data analysis was greatly appreciated.

References

Berard R & Thurtell GW (1990) Soil temperature measurements. In: Goel NS & Norman JM (eds) Instrumentation for Studying Vegetation Canopies for Remote Sensing in Optical and Thermal Infra-red Regions, pp 293–300. Glasgow, UK: Remote Sensing Reviews 5, Harwood Academic

Cates RL & Keeney DR (1987) Nitrous oxide production throughout the year from fertilized and manured maize fields. J Environm Qual 16: 443–447

Chen Y, Tessier S, MacKenzie AF & Laverdière MR (1995) Nitrous oxide emission from an agricultural soil subjected to different freeze-thaw cycles. Agric Ecosys Environm 55: 123–128

Christensen S & Christensen BT (1991) Organic matter available for denitrification in different soil fractions: effect of freeze/thaw cycles and straw disposal. J Soil Sci 42: 637–647

Christensen S & Tiedje JM (1990) Brief and vigorous N_2O production by soil at spring thaw. J Soil Sci 41: 1–4

Denmead OT & Raupach MR (1993) Methods for measuring atmospheric gas transport in agricultural and forest systems. In: Harper LA et al. (eds) Agricultural Ecosystem Effects on Trace Gases and Global Climate Change, pp 19–44. ASA Spec Publ 55. Madison, Wisc: ASA, CSSA, and SSSA

Edwards GC, Neumann HH, den Hartog G, Thurtell GW & Kidd GE (1994) Eddy correlation measurements of methane fluxes using a tunable diode laser at the Kinosheo Lake tower site during the Northern Wetlands Study (NOWES). J Geophys Res 99: 1511–1517

Flessa H & Dörsch P (1995) Seasonal variation of N_2O and CH_4 fluxes in differently managed arable soils in southern Germany. J Geophys Res 100: 23115–23124

Goodroad LL & Keeney DR (1984a) Nitrous oxide emission from forest, marsh, and prairie ecosystems. J Environm Qual 13: 448–452

Goodroad LL & Keeney DR (1984b) Nitrous oxide emissions from soils during thawing. Can J Soil Sci 64: 187–194

Goodroad LL, Keeney DR & Peterson LA (1984) Nitrous oxide emissions from agricultural soils in Wisconsin. J Environm Qual 13: 557–561

Isermann K (1994) Agriculture's share in the emission of trace gases affecting the climate and some cause-oriented proposals for sufficiently reducing this share. Environ Pollut 83: 95–111

Melin J & Nommik H (1983) Denitrification measurements in intact soil cores. Acta Agri Scand 33: 145–151

Mosier AR, Duxbury JM, Freney JR, Heinemeyer O & Minami K (1996) Nitrous oxide from agricultural fields: assessment, measurement and mitigation. Plant and Soil 181: 95–108

Sommerfeld RA, Mosier AR & Musselman RC (1993) CO_2, CH_4 and N_2O flux through a Wyoming snowpack and implications for global budgets. Nature 361: 140–142

Tel DA & Heseltine C (1990) The analyses of KCl soil extracts for nitrate, nitrite and ammonium using a TRAACS 800 analyzer. Commun Soil Sci Plant Anal 21: 1681–1688

Van Bochove E, Jones HG, Pelletier F & Prévost D (1996) Emission of N_2O from agricultural soil under snow cover: a significant part of N budget. Hydrol Proc 10: 1545–1549

Wagner-Riddle C, Thurtell GW, King KM, Kidd GE & Beauchamp EG (1996) Nitrous oxide and carbon dioxide fluxes from a bare soil using a micrometeorological approach. J Environ Qual 25: 898–907

Wagner-Riddle C, Thurtell GW, Kidd GK, Beauchamp EG & Sweetman R (1997) Estimates of Nitrous Oxide Emissions from Agricultural Fields over 28 months.Can J Soil Sci 77: 135–144

Watson RT, Rodhe H, Oeschger H & Siegenthaler U (1990) Greenhouse gases and aerosols. In: Houghton JT et al. (eds) Climate Change: The IPCC Scientific Assessment, pp 7–40. Cambridge Univ Press, Cambridge, UK