Achieving Soil Organic Carbon Sequestration with Conservation Agricultural Systems in the Southeastern United States

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Conservation management of degraded land has the potential to build soil fertility, restore soil functions, and mitigate greenhouse gas emissions as a consequence of surface soil organic matter accumulation. Literature from the southeastern United States was reviewed and synthesized to: (i) quantitatively evaluate the magnitude and rate of soil organic C (SOC) sequestration with conservation agricultural management; (ii) evaluate how conservation management affects surface SOC accumulation and its implications on ecosystem services; and (iii) recommend practical soil sampling strategies based on spatial and temporal issues to improve the detection of statistically significant SOC sequestration. Soil organic C sequestration was 0.45 ± 0.04 Mg C ha^{-1} yr^{-1} (mean ± standard error, n = 147, 20 ± 1 cm depth, 11 ± 1 yr) with conservation tillage compared with conventional tillage cropland. Establishment of perennial pastures sequestered 0.84 ± 0.11 Mg C ha^{-1} yr^{-1} (n = 35, 25 ± 2 cm depth, 17 ± 1 yr). Stratification of SOC with depth was common under conservation agricultural management and appears to be integrally linked to abatement of soil erosion, improvement in water quality, and SOC sequestration. Sampling of conservation management systems should ideally occur repeatedly with time in controlled and replicated experiments, but there is also an urgent need for chronosequence and paired-field surveys of SOC on working farms in the region to validate and expand the scope of inference of experimental results. Landowners in the southeastern United States have great potential to restore soil fertility and mitigate greenhouse gas emissions with the adoption of and improvement in conservation agricultural systems (e.g., continuous no-till, high-residue crop rotations, high organic matter inputs).

Abbreviations: NT, no-till; SOC, soil organic carbon.

Soil contains a vast reservoir of terrestrial C as organic matter (1500 Pg), which is both a source and sink of greenhouse gases (Schlesinger and Andrews, 2000). Agricultural land occupies 40% of the land area in the United States (National Agricultural Statistics Service, 2007) and therefore controls a significant amount of the photosynthetic sink of CO_2 from the atmosphere, as well as the respiration source of CO_2 from plant biomass and soil organisms. The C cycle can be either positively or negatively affected by agricultural activities, depending on management choices.

Agricultural soils provide a number of ecosystem services vital to the human population (Lal, 2008). Two key services among many are (i) production of food, feed, fiber, and fuel, and (ii) C sequestration and moderation of the climate. Unfortunately, historical exploitation of agricultural soils with unsustainable practices has led to degraded soils in need of restoration (Lal, 2004). Soils of the southeastern United States are a good example of the land trauma that occurred following pioneer cultivation and expansion (Triplett and Dick, 2008), which led to soil fertility decline, soil structural degradation, and vast removal of soil from the landscape via erosion. For example, the average soil loss on the 17 Mha of land in the Southern Piedmont Major Land Resource Area has been nearly 20 cm since 1700

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(Trimble, 1974). Restoration of soils in the southeastern United States has been possible with the reintroduction of planted forests, perennial pastures, and conservation tillage management of crops to much of the land (Hendrickson et al., 1963; Bruce et al., 1995; Richter et al., 1999; Franzluebbers, 2005).

The southeastern United States is a region of high potential productivity based on favorable climatic conditions, e.g., mild winters, hot summers, and plentiful precipitation. Soil acidity, low water-holding capacity, and low nutrient-supplying capacity are conditions that limit productivity but that can be overcome with management to increase soil organic matter. Recent reviews of available data established the value of conservation agricultural management systems to sequester SOC in the southeastern United States (Franzluebbers, 2005; Causarano et al., 2006).

The objectives of this review of literature in the southeastern United States are to: (i) assess whether recently reported data would affect the estimated SOC sequestration with conservation agricultural management in the southeastern United States compared with a previous review (Franzluebbers, 2005); (ii) summarize SOC depth distributions, how they are affected by management, and what their implications on ecosystem services might be; and (iii) address spatial and temporal issues of soil sampling frequency and intensity to obtain more robust estimates of SOC sequestration with conservation agricultural management.

SOIL ORGANIC CARBON SEQUESTRATION WITH CONSERVATION AGRICULTURAL MANAGEMENT

Conservation Tillage

Since the review of Franzluebbers (2005), an additional 51 comparisons of SOC on conventional- and conservation-tillage cropland in the southeastern United States have been reported (Table 1). Characteristics of these recent data (Table 1) did not vary significantly from those reported earlier (Franzluebbers, 2005), e.g., the duration of the tillage management system was 11 ± 1 yr (mean ± standard error, $n = 51$) compared with 10 ± 1 yr ($n = 96$), the soil sampling depth was 21 ± 1 cm compared with 19 ± 1 cm, and the SOC sequestration rate was 0.50 ± 0.09 Mg C ha^{-1} yr^{-1} compared with 0.42 ± 0.05 Mg C ha^{-1} yr^{-1}. Therefore all data were pooled to create a regional summary of SOC sequestration with conservation tillage.

From 147 comparisons across eight states, SOC sequestration with conservation tillage compared with conventional tillage was 0.45 ± 0.04 Mg C ha^{-1} yr^{-1} (mean ± standard error) in a sampling depth of 20 ± 1 cm and experiment duration of 11 ± 1 yr (Table 2). The mean value increased only 0.03 Mg C ha^{-1} yr^{-1} from that reported earlier (Franzluebbers, 2005), so there was consistency in the central tendency. Data were sorted by states to test if there were differences (t-test) due to general characteristics (e.g., climate, soils, or cultural practices) unique to each state. Some experimental differences occurred among states, e.g., the duration of experimentation was greater in South Carolina than in Georgia or North Carolina ($P \leq 0.001$) and the soil sampling depth was greater in Maryland than in Mississippi or North Carolina ($P \leq 0.001$). The yearly difference in SOC between tillage systems was significantly lower in Texas (0.36 ± 0.04 Mg C ha^{-1} yr^{-1}) than in Alabama (0.67 ± 0.11 Mg C ha^{-1} yr^{-1}) ($P = 0.01$) or Georgia (0.58 ± 0.13 Mg C ha^{-1} yr^{-1}) ($P = 0.08$); however, it was greater in Texas than in Maryland (0.16 ± 0.11 Mg C ha^{-1} yr^{-1}) ($P = 0.04$). It is possible that the drier climate in Texas (625 mm in Weslaco to 993 mm in College Station) may have limited the SOC sequestration rate with conservation tillage compared with the wetter climates in Alabama (1391 mm in Belle Mina to 1652 mm in Brewton) and Georgia (1146 mm in Fort Valley to 1308 mm in Griffin). Similarly, the colder climate in Maryland (~13°C) may have limited the SOC sequestration rate with conservation tillage compared with the warmer climates in Alabama (11.9°C in Crossville to 18.2°C in Brewton) and Georgia (16.0°C in Griffin to 18.9°C in Fort Valley). These climate-dependent responses were consistent with an earlier review of how conservation tillage affected SOC sequestration rates across the United States and Canada (Franzluebbers and Steiner, 2002). The lack of statistical difference in mean characteristics among other states was probably as dependent on the low number of comparisons within states as on the actual levels. Observations in Alabama, Georgia, and Texas accounted for 66% of the data. There is a need for more SOC data comparisons between conventional and conservation tillage in others states of the region.

The large variation in the yearly difference in SOC between tillage systems (i.e., the SOC sequestration rate, mean ± standard deviation of 0.45 ± 0.52 Mg C ha^{-1} yr^{-1}) was probably due to several widely varying experimental factors, including the length of experimentation, the size of plots, the number and frequency of soil samples, the depth of sampling, and the type of analytical approach, among others. A large, regionally coordinated study of SOC under conventional- and conservation-tillage systems would be useful to limit potential analytical variations. In addition, crop management variables were also widely different, including crop types, crop rotations, fertilization type, amount, and frequency, and pest pressures and control measures. There is still a need to conduct many more studies in the region to understand how these cultural variables might affect the SOC sequestration rate.

The overall mean SOC sequestration rate across experimental and cultural differences represents a broad perspective for the region as a whole. Similar to that reported earlier (Franzluebbers, 2005), the SOC sequestration rate in studies with cover crops (0.55 ± 0.06 Mg C ha^{-1} yr^{-1}, $n = 87$) was greater ($P < 0.01$) than without cover crops (0.30 ± 0.05 Mg C ha^{-1} yr^{-1}, $n = 60$). Clearly, site-specific and cultural-specific effects on the SOC sequestration rate need to be better understood so that the process of sequestration can be manipulated for greater benefit to farmers and society.

Since there were a relatively large number of comparisons of SOC between conventional- and conservation-tillage systems in the region ($n = 147$), a density distribution was constructed to assess the likelihood of achieving particular SOC sequestration rates (Fig. 1). The SOC sequestration

Table 1. Soil organic C (SOC) and associated site characteristics from recent studies investigating tillage systems (CT, conventional tillage; NT, no-till) in the southeastern United States. These data are in addition to those reported earlier in Franzluebbers (2005).

Location	Taxonomic classification	Soil texture†	Cropping system‡	Duration	Depth	SOC under CT	SOC under NT	Reference
				yr	cm	—— Mg ha^{-1} ——		
Belle Mina, AL	Typic Paleudult	SiL	CO/RY-CO-CN/RY (inorganic)	10	20	37.4	40.1	Sainju et al. (2008)
Bella Mina, AL	Typic Paleudult	SiL	CO/RY-CO-CN/RY (poultry litter)	10	20	43.7	43.7	Sainju et al. (2008)
Belle Mina, AL	Rhodic Paleudult	SiL	CO	9	6	7.2	8.4	Truman et al. (2003)
Headland, AL	Plinthic Kandiudult	LS	CO/RG-PN/O	3	20	21.6	23.3	Siri-Prieto et al. (2007)
Shorter, AL	Typic Paleudult	LS	CO/BO+RY-CN/WL+CC	2.5	30	23.5	26.2	Terra et al. (2005)
Shorter, AL	Typic Paleudult	LS	CO/BO+RY-CN/L+CC (with manure)	2.5	30	29.1	32.6	Terra et al. (2005)
AL Coastal Plain	Kandiudults–Haploxeralfs	LS	CO/RY-PN/RY	13.3	20	18.0	20.1	Causarano et al. (2008)
AL Piedmont	Kanhapludults	SCL	CO	10	20	19.6	25.4	Causarano et al. (2008)
Bartow, GA	Plinthic Kandiudult	SL	CO/RY	2	15	17.6	19.3	Sainju et al. (2007)
Tifton, GA	Plinthic Kandiudult	LS	CO/RY-CO/RY-PN/RY	2	15	17.7	17.0	Sainju et al. (2007)
Fort Valley, GA	Plinthic Paleudult	SL	CO-GS	7	30	21.4	24.2	Sainju et al. (2006)
Fort Valley, GA	Plinthic Paleudult	SL	CO/RY-GS/RY	7	30	22.6	27.2	Sainju et al. (2006)
Fort Valley, GA	Plinthic Paleudult	SL	CO/HV-GS/HV	7	30	23.1	26.7	Sainju et al. (2006)
Fort Valley, GA	Plinthic Paleudult	SL	CO/RY+HV-GS/RY+HV	7	30	23.9	27.9	Sainju et al. (2006)
Watkinsville, GA	Typic Kanhapludult	SL	CO/RY-CN/RY-ML/RY-GS/RY-SB/CC-CN/CC	7	20	27.1	32.7	Franzluebbers et al. (2007)
Watkinsville, GA	Typic Kanhapludult	SL	GS/RY (ungrazed)	3	30	40.6	51.6	Franzluebbers and Stuedemann (2008)
Watkinsville, GA	Typic Kanhapludult	SL	GS/RY (grazed)	3	30	46.5	49.5	Franzluebbers and Stuedemann (2008)
Watkinsville, GA	Typic Kanhapludult	SL	WT/ML (ungrazed)	3	30	45.6	45.1	Franzluebbers and Stuedemann (2008)
Watkinsville, GA	Typic Kanhapludult	SL	WT/ML (grazed)	3	30	42.8	46.6	Franzluebbers and Stuedemann (2008)
GA Coastal Plain	Plinthic Kandiudults	LS	CO-PN/WT-SB/RY	15.3	20	16.7	21.5	Causarano et al. (2008)
GA Piedmont	Kanhapludults	SL	GS/WT-SB-CO/O	8.3	20	26.7	27.5	Causarano et al. (2008)
Harmony, NC	Typic Kanhapludult	SCL-L	CN (silage)/BL (silage)	7	20	33.5	36.6	Franzluebbers and Brock (2007)
Harmony, NC	Typic Kanhapludult	SCL-L	CN (silage)/RY	7	20	33.5	33.1	Franzluebbers and Brock (2007)
Harmony, NC	Typic Kanhapludult	SCL-L	CN (silage)/BL-SG/RY	7	20	33.5	39.1	Franzluebbers and Brock (2007)
NC Coastal Plain	Quartzipsamments–Kandiudults	LS	CO-CO/RY-SB/WT	7.5	20	18.2	33.6	Causarano et al. (2008)
NC Piedmont	Typic Kanhapludults	SCL	PN/WT-CO/WT-SB/CC	12.7	20	25.0	32.6	Causarano et al. (2008)
Florence. SC	Typic Kandiudult	LS	CO-WT/SB-CO	25	7.6	10.1	20.3	Bauer et al. (2006)
Florence. SC	Typic Kandiudult	LS	CO-WT/SB-CO	24	15	20.6	31.4	Novak et al. (2007)
SC Coastal Plain	Kandiudults–Kanhapludults	LS	CO-SB	16	20	24.8	25.3	Causarano et al. (2008)
SC Piedmont	Typic Kanhapludults	SCL	CO-SB/WT-ML	14.7	20	22.2	26.5	Causarano et al. (2008)
College Station, TX	Udifluventic Haplustept	SiCL	WT (0N)	20	30	36.5	37.8	Dou and Hons (2006)
College Station, TX	Udifluventic Haplustept	SiCL	WT (2N)	20	30	37.5	40.8	Dou and Hons (2006)
College Station, TX	Udifluventic Haplustept	SiCL	GS-WT/SB (0N)	20	30	35.5	44.3	Dou and Hons (2006)
College Station, TX	Udifluventic Haplustept	SiCL	GS-WT/SB (2N)	20	30	35.4	49.2	Dou and Hons (2006)
College Station, TX	Udifluventic Haplustept	SiCL	WT/SB (0N)	20	30	35.8	42.3	Dou and Hons (2006)
College Station, TX	Udifluventic Haplustept	SiCL	WT/SB (2N)	20	30	35.6	46.5	Dou and Hons (2006)
College Station, TX	Udifluventic Haplustept	SiCL	GS	20	15	15.1	19.5	Wright and Hons (2005)
College Station, TX	Udifluventic Haplustept	SiCL	WT	20	15	22.2	23.7	Wright and Hons (2005)
College Station, TX	Udifluventic Haplustept	SiCL	SB	20	15	14.1	22.2	Wright and Hons (2005)
College Station, TX	Udifluventic Haplustept	SiCL	GS-WT/SB	20	15	17.4	25.1	Wright and Hons (2004)
College Station, TX	Udifluventic Haplustept	SiCL	WT/SB	20	15	18.7	22.9	Wright and Hons (2004)
College Station, TX	Udifluventic Haplustept	SiCL	SB	20	15	15.1	20.5	Wright and Hons (2004)
Weslaco, TX	Typic Calciustoll	SCL	CO/CN	9	30	40.2	41.8	Zibilske et al. (2002)
VA Coastal Plain	Aquic Hapludult	SL	CO-WT/SB	14	15	15.9	21.9	Spargo et al. (2008)
VA Coastal Plain	Aquic Hapludult	SL	CO-WT/SB (with biosolids)	9	15	20.2	22.9	Spargo et al. (2008)
VA Coastal Plain	Typic Hapludult	LfS	CO-WT/SB	3	15	14.1	13.3	Spargo et al. (2008)
VA Coastal Plain	Typic Hapludult	LfS	CO-WT/SB (with biosolids)	9	15	16.7	19.1	Spargo et al. (2008)
VA Coastal Plain	Typic Hapludult	LfS	CO-WT/SB	11	15	17.2	19.3	Spargo et al. (2008)
VA Coastal Plain	Typic Hapludult	LfS	CO-WT/SB (with biosolids)	11	15	20.2	29.9	Spargo et al. (2008)
VA Coastal Plain	Hapludults	SL	CO/WT-SB	13.3	20	20.4	31.4	Causarano et al. (2008)
VA Piedmont	Kanhapludults–Kandiudults	L	SB-CO/WT	9	20	27.7	29.9	Causarano et al. (2008)
Mean ± SE (n = 51)				11 ± 1	21 ± 1	25.6 ± 1.4	30.2 ± 1.4	

† L, loam; LfS, loamy fine sand; LS, loamy sand; SCL, sandy clay loam; SiCL, silty clay loam; SL, sandy loam; SiL, silt loam.

‡ BL, barley; BO, black oat; CC, crimson clover; CN, corn; CO, cotton; GS, grain sorghum; HV, hairy vetch; ML, millet; O, oat; PN, peanut; RG, ryegrass; RY, rye; SB, soybean; WT, wheat; 0N, no N fertilizer; 2N, recommended N fertilizer input.

Table 2. Mean ± standard error of soil organic C (SOC) characteristics under conventional and no-till systems from studies by state (data from Table 1 and previously reported in Franzluebbers [2005]).

Property	Alabama	Georgia	Maryland	Mississippi	N. Carolina	S. Carolina	Texas	Virginia	Mean
Comparisons, no.	27	31	13	6	15	9	38	8	147
Duration of comparison, yr	9 ± 1	8 ± 1	14 ± 2	8 ± 1	7 ± 1	17 ± 2	13 ± 1	10 ± 1	11 ± 1
Soil depth, cm	21 ± 1	21 ± 1	23 ± 1	15	15 ± 1	15 ± 1	22 ± 1	16 ± 1	20 ± 1
SOC with conventional tillage, Mg ha^{-1}	22.4 ± 1.5	25.5 ± 1.7	39.2 ± 4.4	16.4 ± 1.7	21.5 ± 1.7	20.7 ± 2.2	28.5 ± 1.9	19.0 ± 1.5	25.5 ± 0.9
SOC with no-till, Mg ha^{-1}	26.6 ± 1.5	28.8 ± 1.9	40.6 ± 4.3	19.5 ± 1.1	24.5 ± 2.1	25.4 ± 2.2	33.0 ± 1.8	23.5 ± 2.3	29.2 ± 0.9
Difference in SOC between tillage systems, Mg ha^{-1}	4.1 ± 0.6	3.3 ± 0.5	1.5 ± 0.9	3.1 ± 0.9	3.0 ± 1.1	4.7 ± 1.2	4.5 ± 0.5	4.4 ± 1.5	3.7 ± 0.3
Yearly difference in SOC between tillage systems, Mg ha^{-1} yr^{-1}	0.66 ± 0.11	0.59 ± 0.13	0.16 ± 0.11	0.37 ± 0.10	0.40 ± 0.14	0.27 ± 0.06	0.36 ± 0.04	0.36 ± 0.13	0.45 ± 0.04
Ratio of SOC with no-till to conventional tillage, kg kg^{-1}	1.21 ± 0.03	1.15 ± 0.03	1.05 ± 0.02	1.22 ± 0.06	1.14 ± 0.06	1.28 ± 0.10	1.19 ± 0.02	1.23 ± 0.07	1.18 ± 0.02

rate averaged 0.45 Mg C ha^{-1} yr^{-1} and the median rate was 0.35 Mg C ha^{-1} yr^{-1}. The cumulative density function in Fig. 1 suggested that there was an 82% chance of achieving a value of at least 0.10 Mg C ha^{-1} yr^{-1} (a value at least marginally different from zero), a 63% chance of achieving a value of at least 0.25 Mg C ha^{-1} yr^{-1}, a 32% chance of achieving a value of at least 0.50 Mg C ha^{-1} yr^{-1}, and a 15% chance of achieving a value of at least 0.75 Mg C ha^{-1} yr^{-1}. This probability density function based on previously collected data should be viewed as a guide for future observations, assuming that the observations are randomly associated and not a function of a unique controlling factor. This could be tested with a coordinated sampling campaign across the region.

The rate of SOC sequestration with conservation tillage in the southeastern United States was similar to the rate reported for the central United States (0.48 ± 0.59 Mg C ha^{-1} yr^{-1}, Johnson et al., 2005), but higher than rates reported for the southwestern United States (0.30 ± 0.21 Mg C ha^{-1} yr^{-1}, Martens et al., 2005), the northwestern United States and western Canada (0.27 ± 0.19 Mg C ha^{-1} yr^{-1}. Liebig et al., 2005), and eastern Canada (−0.07 ± 0.27 Mg C ha^{-1} yr^{-1}, Gregorich et al., 2005). From 93 paired comparisons around the world, SOC sequestration with no-till (NT) compared with conventional tillage was 0.48 ± 0.13 Mg C ha^{-1} yr^{-1} (West and Post, 2002). In a review of literature projected to a 20-yr period, Six et al. (2004) reported SOC sequestration rates of 0.22 and 0.10 Mg C ha^{-1} yr^{-1} under humid and dry climates, respectively. Although limited in data, N_2O emission would have reduced these rates by 0.07 Mg C ha^{-1} yr^{-1} based on equivalent global warming potential. Mean comparisons in different regions of the United States and the world indicate that regionally defined differences in SOC sequestration rates with conservation tillage probably do exist, but that there is also a need for more thorough characterization and evaluation.

The overwhelming majority of data suggests that conservation tillage in the southeastern United States will lead to significantly positive organic C sequestration in the upper 20 cm of the soil. Still greatly lacking in the region are data on how conservation agriculture affects other important greenhouse gases, i.e., N_2O and CH_4. This research must be undertaken.

Pasture Management

Establishment of perennial pasture following conventionally tilled cropland can also lead to significantly positive SOC sequestration (Table 3). Three recent studies reported SOC content following grass establishment to provide an additional 23 observations to the 12 observations reported previously in the southeastern United States (Franzluebbers, 2005). Across the 35 observations, SOC sequestration with pasture establishment had a mean ± standard error of 0.84 ± 0.11 Mg C ha^{-1} yr^{-1}. The cumulative density function in Fig. 2 suggested that there was an 87% chance of achieving a value of at least 0.10 Mg C ha^{-1} yr^{-1}, an 81% chance of achieving a value of at least 0.25 Mg C ha^{-1} yr^{-1}, a 66% chance of achieving a value of at least 0.50 Mg C ha^{-1} yr^{-1}, a 48% chance of achieving a value of at least 0.75 Mg C ha^{-1} yr^{-1}, and a 30% chance of achieving a value of at least 1.00 Mg C ha^{-1} yr^{-1}.

The type of pasture management can have a significant influence on SOC sequestration. In a 5-yr study on 'Coastal' bermudagrass [*Cynodon dactylon* (L.) Pers.] in Georgia, SOC accumulated with a moderate cattle stocking rate (~6 steers ha^{-1}) compared with no cattle grazing, but declined at a high stocking rate (Fig. 3). In contrast, surface-residue C declined at all cattle stocking rates. Evidence for SOC sequestration with organic fertilization (i.e., poultry litter) compared with inorganic fertilization has not been convincing in Georgia (Franzluebbers and Stuedemann, 2009). Soil retention of C applied as manure in warm, moist environments was estimated as 7 to 8%, while retention in cool, dry environments was estimated as 11 to 23% (Franzluebbers and Doraiswamy, 2007). In Virginia, orchardgrass (*Dactylis glomerata* L.) pastures with management-intensive grazing sequestered SOC at a rate of 0.61 Mg C ha^{-1} yr^{-1} (time-weighted estimates based on 14 ± 11 yr) compared with the prevailing extensive grazing (Conant et al., 2003). There is a great need to conduct further studies to determine the rate of SOC sequestration under the diversity of pasture management systems used in the region.

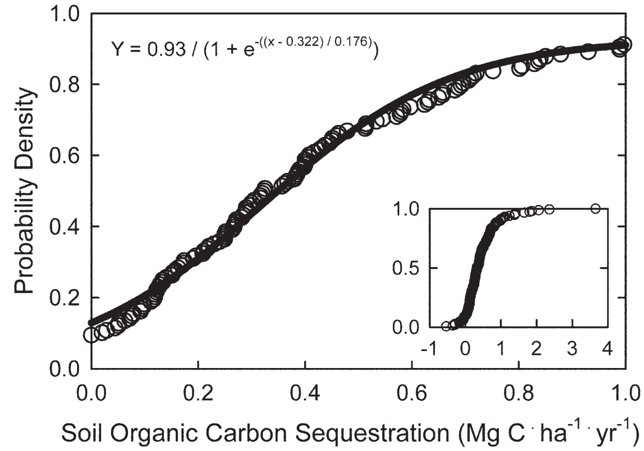

Fig. 1. Density of soil organic C sequestration estimates with conservation-tillage cropland in the southeastern United States (*n* = 147). Inset shows all data, while the main graph shows the range of soil organic C values encompassing approximately the middle 80% of observations.

STRATIFICATION OF SOIL ORGANIC CARBON WITH DEPTH

Conservation agricultural systems in the southeastern United States typically develop a highly stratified vertical distribution of SOC with time (Fig. 4). Significant changes in SOC

Table 3. Rate of soil organic C (SOC) sequestration following grass establishment from recent studies in the southeastern United States. These data are in addition to those reported earlier in Franzluebbers (2005).

Location	Taxonomic classification	Experimental conditions	Duration	SOC sequestration	Reference
			yr	Mg C ha^{-1} yr^{-1}	
Farmington, GA	Typic Kanhapludult	unharvested 'Coastal' bermudagrass with inorganic fertilizer, 30-cm depth	12	0.54	Franzluebbers and Stuedemann (2009)
Farmington, GA	Typic Kanhapludult	grazed Coastal bermudagrass at low pressure with inorganic fertilizer, 30-cm depth	12	1.15	Franzluebbers and Stuedemann (2009)
Farmington, GA	Typic Kanhapludult	grazed Coastal bermudagrass at high pressure with inorganic fertilizer, 30-cm depth	12	0.73	Franzluebbers and Stuedemann (2009)
Farmington, GA	Typic Kanhapludult	hayed Coastal bermudagrass with inorganic fertilizer, 30-cm depth	12	0.24	Franzluebbers and Stuedemann (2009)
Farmington, GA	Typic Kanhapludult	unharvested Coastal bermudagrass with low broiler litter fertilization, 30-cm depth	12	0.52	Franzluebbers and Stuedemann (2009)
Farmington, GA	Typic Kanhapludult	grazed Coastal bermudagrass at low pressure with low broiler litter fertilization, 30-cm depth	12	1.64	Franzluebbers and Stuedemann (2009)
Farmington, GA	Typic Kanhapludult	grazed Coastal bermudagrass at high pressure with low broiler litter fertilization, 30-cm depth	12	1.13	Franzluebbers and Stuedemann (2009)
Farmington, GA	Typic Kanhapludult	hayed Coastal bermudagrass with low broiler litter fertilization, 30-cm depth	12	0.39	Franzluebbers and Stuedemann (2009)
Farmington, GA	Typic Kanhapludult	unharvested Coastal bermudagrass with high broiler litter fertilization, 30-cm depth	12	1.33	Franzluebbers and Stuedemann (2009)
Farmington, GA	Typic Kanhapludult	grazed Coastal bermudagrass at low pressure with high broiler litter fertilization, 30-cm depth	12	1.42	Franzluebbers and Stuedemann (2009)
Farmington, GA	Typic Kanhapludult	grazed Coastal bermudagrass at high pressure with high broiler litter fertilization, 30-cm depth	12	0.90	Franzluebbers and Stuedemann (2009)
Farmington, GA	Typic Kanhapludult	Hayed Coastal bermudagrass with high broiler litter fertilization, 30-cm depth	12	–0.06	Franzluebbers and Stuedemann (2009)
AL Coastal Plain	Kandiudults–Haploxeralfs	grazed (2) and hayed (1) pastures, 20-cm depth	27 ± 12	0.60	Causarano et al. (2008)
AL Piedmont	Kanhapludults	grazed (1) and hayed (2) pastures, 20-cm depth	33 ± 6	0.34	Causarano et al. (2008)
GA Coastal Plain	Plinthic Kandiudults	grazed pastures (3), 20-cm depth	17 ± 6	1.13	Causarano et al. (2008)
GA Piedmont	Kanhapludults	grazed pastures (3), 20-cm depth	24 ± 15	0.33	Causarano et al. (2008)
NC Coastal Plain	Quartzipsamments–Kandiudults	grazed (1) and hayed (1) pastures, 20-cm depth	15 ± 7	1.12	Causarano et al. (2008)
NC region	Typic Kanhapludults	grazed pastures (2), 20-cm depth	14 ± 4	0.92	Causarano et al. (2008)
SC Coastal Plain	Kandiudults–Kanhapludults	grazed (2) and hayed (1) pastures, 20-cm depth	20	0.08	Causarano et al. (2008)
SC Piedmont	Typic Kanhapludults	grazed pastures (3), 20-cm depth	23 ± 6	1.03	Causarano et al. (2008)
VA Coastal Plain	Hapludults	grazed (2) and hayed (1) pastures, 20-cm depth	27 ± 3	0.51	Causarano et al. (2008)
VA Piedmont	Kanhapludults–Kandiudults	grazed pastures (3), 20-cm depth	27 ± 12	0.82	Causarano et al. (2008)
Charlottesville, VA	Typic Hapludult	rotationally grazed orchardgrass/clover, 50-cm depth	29 ± 24	0.15	Conant et al. (2004)
Mean ± SE (*n* = 23)			17 ± 1	0.74 ± 0.10	

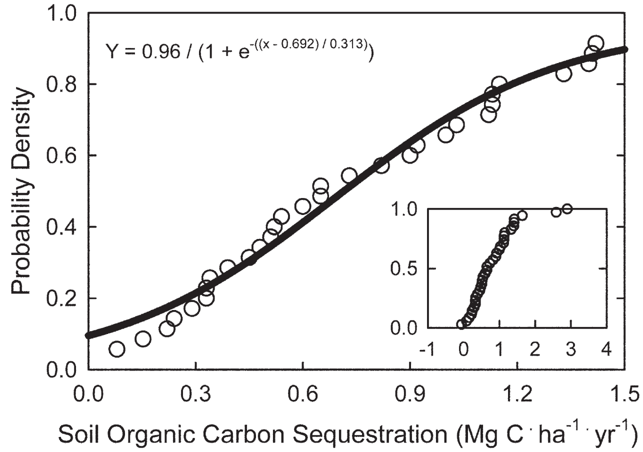

Fig. 2. Density of soil organic C sequestration estimates with pasture establishment in the southeastern United States (*n* = 35). Inset shows all data, while the main graph shows the range of soil organic C values encompassing approximately the middle 80% of observations.

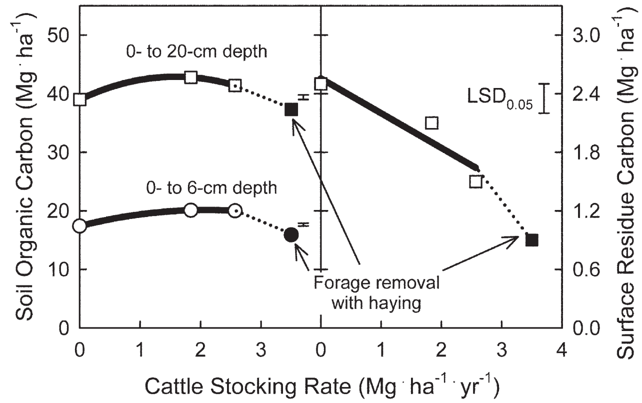

Fig. 3. Soil organic C and surface residue C as a function of cattle stocking rate on bermudagrass in Georgia (data from Franzluebbers et al., 2001).

that do occur with conservation agricultural systems in the region are usually limited to the surface 30 cm of soil and even more typically to the surface 15 cm of soil. The ratio of SOC at 0 to 15 cm to that at 15 to 30 cm in the example in Fig. 4 increased from 2.4 at initiation to 3.1 at the end of 5 yr to 3.6 at the end of 12 yr.

High surface SOC has important impacts on several soil functions. Enrichment of the soil surface with organic C is ecologically essential, because the soil surface is the interface that (i) receives much of the fertilizers and pesticides applied to agricultural land, (ii) receives the intense impact of rainfall, and (iii) partitions the fluxes of gases and water into and out of the soil (Franzluebbers, 2002a). Therefore, organic-C-enriched surface soil fosters productivity, regulates terrestrial water flow, sequesters C from the atmosphere, cycles nutrients through biological activity, filters and denatures pollutants, and creates a biologically active and diverse warehouse of soil microorganisms. Soil organic C allows plant roots and soil biota to reconfigure the soil matrix into a stable structure with permanent channels (biopores), a process that is important in achieving high water infiltration. As an example, the positive effect of a high stratification ratio of SOC (an index of biophysical changes in the soil structure) on water infiltration was demonstrated in a controlled infiltration experiment on a Typic Kanhapludult, whereby water infiltration increased 27% with doubling of the organic C uniformly throughout a 12-cm depth of soil and increased more than 200% when organic C was concentrated in the surface 3 cm of the soil (Franzluebbers, 2002b). In field studies with small-plot rainfall simulations (4.6–5.5 m^2) in Mississippi and Ohio (Rhoton et al., 2002), soil loss was inversely related to the calculated SOC stratification ratio (Franzluebbers, 2008). Other field-plot and water-catchment studies have documented the positive influence of conservation tillage and pasture management on soil stabilization and the avoidance of water and nutrient runoff (reviewed in Franzluebbers, 2008). Very few of these studies provided measurements of the depth distribution of SOC, however, which would have probably been highly stratified under conservation systems and uniformly distributed with depth under inversion tillage systems (i.e., bare surface). The assumed surface accumulation and high stratification ratio of SOC under conservation management systems are further supported by observations in the following.

In a tillage × cropping system experiment in Watkinsville, GA, the stratification ratio of SOC (0–6/12–20 cm) was initially 3.7 when a long-term pasture was terminated, and then became widely divergent throughout the subsequent 3 yr of the study (Franzluebbers and Stuedemann, 2008). The stratification ratio of SOC was 0.9 and 3.8 under conventional tillage and NT, respectively, at the end of 1 yr, 1.1 and 3.8 at the end of 2 yr, and 1.2 and 3.9 at the end of 3 yr. Similar values for the stratification ratio of SOC (0–5/15–30 cm) were reported for four measurements during 30 yr of experimentation with moldboard plowing (1.7 ± 0.2), NT management (3.4 ± 0.1), and grass sod (4.0 ± 0.2) in Kentucky (Diaz-Zorita and Grove, 2002).

In a survey of agricultural land uses in Alabama, Georgia, South Carolina, North Carolina, and Virginia, the stratification ratio of SOC (0–5/12.5–20 cm) averaged 1.4 with conventional-tillage cropland and reached a plateau of 2.8 within 10 yr on conservation-tillage cropland and a plateau of 4.2 with perennial pasture (Fig. 5). This survey included a wide diversity of cropping histories and soil types, which may have contributed to the wide variation observed. In a survey of cropland fields on three different soil types in the Virginia Coastal Plain, the stratification ratio of SOC (0–2.5/7.5–15 cm) was linearly related to the number of years of continuous NT (initially 1.5 following conventional tillage and increasing to 3.6 with 14 yr of NT) (Spargo et al., 2008).

Stratification of SOC with depth may also be predictive of terrestrial C storage with conservation agricultural systems in the southeastern United States. In the land use survey by Causarano et al. (2008), the stratification ratio of SOC (0–5/12.5–20 cm) was related to the total stock of SOC in the surface 20-cm depth (Fig. 6). This relationship indicates that the majority of C stored under conservation management in these Ultisols and Alfisols of the region occurred within the surface 5 cm. More data will be needed to extend the applicability of this relationship throughout the region. When using only the surface 2.5 cm of soil for

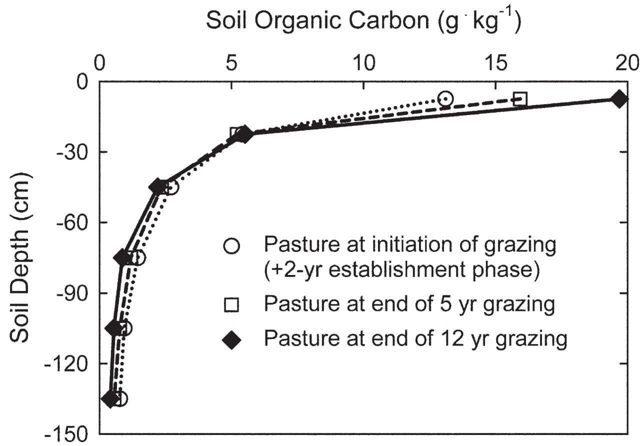

Fig. 4. Soil organic C depth distribution as a function of duration of pasture management (data from Franzluebbers and Stuedemann, 2009).

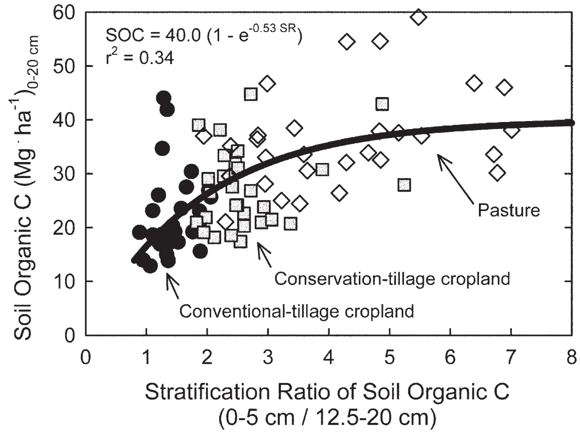

Fig. 6. Stock of soil organic C to a depth of 20 cm as a function of the stratification ratio of soil organic C, independent of land management (data from Causarano et al., 2008).

calculation of the stratification ratio, there was little relationship between the stratification ratio and the stock of SOC in the surface 15 cm of conservation-tilled Coastal Plain soils in Virginia (Spargo et al., 2008). Taking these data together suggests that significant accumulation (if not the majority) of SOC occurs within the surface 5 cm with conservation management.

SAMPLING ISSUES FOR SOIL ORGANIC CARBON DETERMINATION

Sampling issues have been raised in various studies when assessing SOC sequestration in agricultural systems. These issues, including bulk density determination, depth of sampling, frequency of sampling, and stratified sampling, are addressed here to provide recommendations based on observations and synthesis of the literature to balance practicality and scientific rigor.

Bulk Density

Bulk density and SOC concentration are two key properties needed to calculate SOC stocks and changes with time. Bulk density has not always been determined in older studies reporting SOC concentration with different management. Bulk density can be estimated through pedotransfer functions, most typically from soil organic matter and soil texture (clay, silt, and sand contents) (Manrique and Jones, 1991; De Vos et al., 2005; Kätterer et al., 2006; Benites et al., 2007). The relationship between bulk density and SOC in Typic Kanhapludults from Georgia is shown in Fig. 7. Despite the apparent limited sampling domain, the relationship is, in fact, similar to that reported for forested soils in Belgium ($BD = 1.78 - 0.173\ LOI^{1/2}$, where BD is bulk density and LOI is loss-on-ignition) (De Vos et al., 2005). Therefore in the absence of bulk density measurements, SOC can be used to predict bulk density (note that the greater the SOC concentration, the lower the bulk density—an inverse relationship that limits the growth of SOC stock and clearly indicates the need to either measure bulk density directly or estimate bulk density for a soil type with an adjustment for SOC concentration). In a survey of land uses in the Coastal Plain and Southern Piedmont (including three Haploxeralfs, three Hapludalfs, three Quartzipsamments, eight Hapludults, three Paleudults, 22 Kandiudults, and 45 Kanhapludults), soil bulk density predicted by the relationship with SOC in Fig. 7 was 6.1 ± 6.9% (mean ± standard deviation) lower than actual measurements of bulk density (Causarano et al., 2008). Using predicted bulk density, SOC sequestration with conservation tillage compared with conventional tillage would have changed from 5.1 ± 6.8 to 4.2 ± 6.4 Mg ha^{-1}

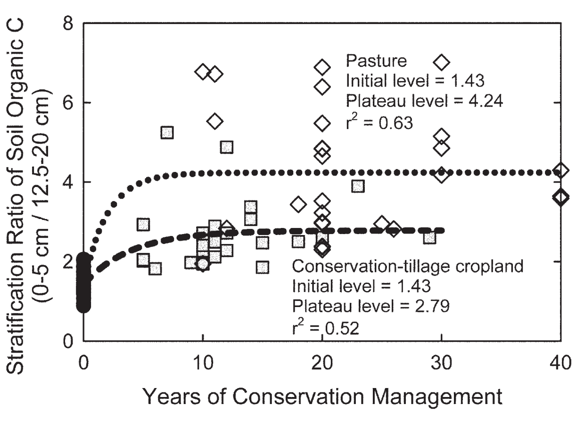

Fig. 5. Stratification ratio of soil organic C as a function of years of conservation management (data from Causarano et al., 2008). Diamonds are pasture, squares are conservation-tillage cropland, and filled circles are conventional-tillage cropland.

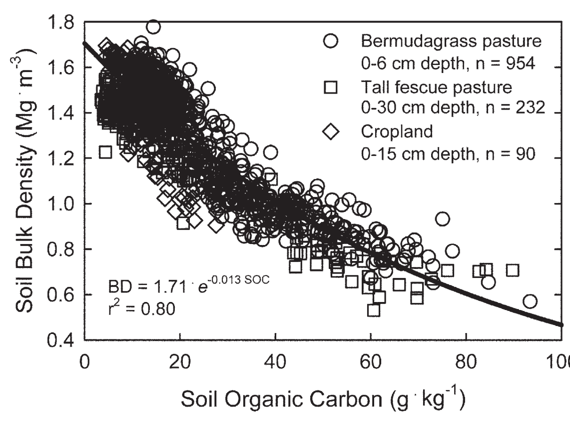

Fig. 7. Relationship of soil bulk density to soil organic C concentration among Typic Kanhapludults from three studies in Georgia (data from Franzluebbers et al., 1999, 2000a, 2001).

Table 4. Comparative analysis of how residual variations in soil bulk density and soil organic C (SOC) concentration affect the calculation of the stock of SOC.

Sampling depth	Bulk density		SOC			Variation in SOC stock due to	
	Mean	CV	Mean conc.	CV	Mean stock	Bulk density	SOC conc.
cm	Mg m^{-3}	%	g kg^{-1}	%	Mg ha^{-1}		
Cropland study, end of 4 yr (Franzluebbers et al., 1999)							
0–2.5	1.09	6	18.6	16	5.1	0.3	0.8
2.5–7.5	1.51	5	10.2	18	7.7	0.4	1.4
7.5–15	1.60	3	7.2	24	8.7	0.2	2.1
Pasture and cropland survey, 6–50 yr (Franzluebbers et al., 2000b)							
0–5	1.11	6	28.5	13	15.8	1.0	2.0
5–12.5	1.57	4	9.1	20	10.7	0.4	2.1
12.5–20	1.62	5	5.9	33	7.1	0.4	2.4
Pasture study, end of 5 yr (Franzluebbers et al., 2001)							
0–3	1.05	6	37.7	9	11.8	0.7	1.1
3–6	1.47	2	15.7	9	6.9	0.2	0.6
6–12	1.56	2	13.1	10	12.2	0.3	1.2
12–20	1.57	2	7.6	15	9.6	0.2	1.5
Integrated crop–livestock study, end of 3 yr (Franzluebbers and Stuedemann, 2008)							
0–3	1.12	6	26.2	11	8.8	0.6	1.0
3–6	1.38	8	15.5	13	6.4	0.5	0.8
6–12	1.48	7	11.4	11	10.1	0.7	1.1
12–20	1.55	6	9.1	23	11.2	0.7	2.6
20–30	1.53	6	6.2	25	9.5	0.6	2.4
Mean ± SE	1.41 ± 0.05	5 ± 1	14.8 ± 2.4	17 ± 2	9.5 ± 0.7	0.5 ± 0.1	1.5 ± 0.2

and with perennial pasture compared with conventional tillage would have changed from 14.3 ± 10.5 to 12.8 ± 8.8 Mg ha^{-1}. Predictions would have been reasonably close to actual determinations.

How does SOC stock vary as a function of natural variations in bulk density and SOC concentration? The answer may depend to some extent on the techniques used to measure the two properties. Using core samples (typically five to eight cores, 4-cm diameter, various depths) to estimate bulk density and dry combustion of finely ground and homogenized soil from composited core samples to estimate the SOC concentration, the stock of SOC varied three times more from residual variations in SOC concentration than from residual variations in bulk density (Table 4). The coefficient of variation averaged 5% for bulk density determinations and 17% for SOC concentration. This comparative analysis suggests that random variation in SOC concentration should be of greater concern than bulk density. Obtaining the best estimate of SOC concentration, therefore, requires not only homogenizing samples once samples are collected, but also obtaining the most representative sample from the plot or field. It should be noted that in this evaluation, bulk density was measured from mass and volume of whole soil samples (150–1500-g samples), while SOC concentration was measured from <1-g subsamples. These differences in sample size should be expected to influence variability.

Spatial and Temporal Variation

Obtaining the best representative soil sample from the field to determine SOC sequestration requires spatial and temporal considerations. Spatially, SOC can vary laterally within a field due to landscape features (i.e., change in slope, soil type, textural class, etc.; Terra et al., 2005; Novak et al., 2009) and management (i.e., field borders, equipment traffic, plant species, animal behavior, etc.). Soil organic C concentration also varies predictably with depth (i.e., greater at the soil surface and declining to a minimum value throughout most of the soil profile) (Fig. 4).

On a variably eroded, upland landscape dominated by Cecil–Madison–Pacolet soils (fine, kaolinitic, thermic Typic Kanhapludults), SOC and N concentrations from individual cores (4-cm diam., 0–8-cm depth) within 0.25-m^2 areas (12 cores from five separate areas) had coefficients of variation of 19 ± 10%, while coefficients of variation from 159 samples collected at 30-m distances within a 12-ha field were 23 ± 2% (Franzluebbers, 1999). These variations were of the same magnitude as those reported in other studies (Table 4). Compositing more subsamples to represent a field (if zones of potential variation are not previously known) should be considered a better option than collecting numerous single-point samples for individual analysis. Better representation of the management system of interest should be a priority.

Expected variation in SOC within a field or plot can be effectively addressed by subdividing the field or plot for sampling. In grazed bermudagrass pastures in Georgia, individual paddocks were divided into three zones based on expected differences in SOC within a paddock due to animal behavior, i.e., concentrated dung and urine deposition around shade and water sources (Franzluebbers and Stuedemann, 2009). The coefficient of variation in SOC concentration was reduced by half when zonal samples within a paddock were averaged rather than analyzed separately (Table 5). These data indicate that experimental error can be reduced by investing in a sampling approach that accounts for expected variations in SOC.

Another approach often used in assessing SOC sequestration in minimally replicated or large, unreplicated field designs is to collect pseudo-replicates to increase the power of analysis. If field variation can be logically blocked a priori among contrasting systems, pseudo-replication can increase experimental power and facilitate separation of potential differences in SOC. In a cropped water-catchment study with only two replications of two management systems, dividing each water catchment into similar sampling zones was effective in partitioning variation due to common landscape features, as well as to increase experimental power without artificially reducing random variation (Franzluebbers et al., 2007). Pseudo-replication should be avoided if it simply reduces variation among samples (i.e., if samples are inherently dependent, not independent). Increasing the number of soil samples can be useful to reduce the minimum detectable difference (Yang et al., 2008). Pseudo-replication would be ideally suited to get better estimates from on-farm assessments (e.g., strip plots or side-by-side fields) that can be sampled with delineation of expected differences due to soil type, textural class, topography, etc. (VandenBygaart et al., 2002).

Soil organic C sequestration could be expected to occur not only near the soil surface where large depositions of plant and animal residues combined with frequently cold or dry conditions would limit their decomposition, but also deeper in the soil profile where soil remains undisturbed and low available nutrients might limit decomposition of root deposits. The few data that are available, however, do not support statistically significant deep-profile SOC sequestration in the southeastern United States. On eroded cropland converted to a pine plantation in South Carolina, organic C sequestration was 0.95 Mg C ha^{-1} yr^{-1} in the forest-floor litter layer, 0.04 Mg C ha^{-1} yr^{-1} in the 0- to 15-cm soil depth, and unchanged or tending to decline at a depth of 15 to 60 cm (Richter et al., 1999). Organic C from profiles of major soil types throughout the southeastern United States was greater under forest and grass than under crops at a depth of 0 to 25 cm, but was not different among land uses at depths below the plow layer (Fig. 8). Although management details were lacking from this survey report from several soils throughout the southeastern United States, the data did suggest a nonsignificant trend for progressively diminishing SOC sequestration deeper in the profile under perennial grass compared with annual crops.

Detailed soil characterization with time would probably be necessary to detect significant changes in SOC with depth in management systems having vigorous deep-rooting capability. To provide some perspective, roots of bermudagrass at the end of 3 yr of experimentation were 3.3 ± 1.7, 0.9 ± 0.5, and 0.9 ± 0.5 Mg ha^{-1} at depths of 0 to 30, 30 to 60, and 60 to 90 cm, respectively (Adams et al., 1966). Assuming that 50% of the root biomass accumulated within a year, 43% of the root biomass was organic C (Franzluebbers et al., 1994), and 24% of the organic C could be retained as SOC following a year of decomposition (Franzluebbers et al., 1998), then it would have taken 9, 32, and 31 yr to achieve SOC accumulation of 1.5 Mg ha^{-1} (a minimum detectable limit reported in Table 4) at depths of 0 to 30, 30 to 60, and 60 to 90 cm, respectively. Much larger minimum detectable differences can be expected, however; e.g., values of 10.3 ± 3.0 Mg ha^{-1} were reported for three studies in Ontario and Illinois at a depth of 40 to 50 cm (Yang et al., 2008). Therefore, overcoming inherent variability to detect significant changes in SOC deeper in the profile may require an unusually long evaluation period or more intensive sampling protocols.

As noted in Table 5, SOC concentration declines with depth and the coefficient of variation often increases with depth. These effects result in an increasingly larger difference in SOC needed for the detection of significance; e.g., the least significant difference increased an average of 0.6 Mg ha^{-1} for each additional 30 cm of soil depth analyzed (Franzluebbers and Studemann, 2005). Reports from other ecological regions have also found greater random variation in SOC (and therefore none or marginally significant differences between management systems) when sampling below the plow layer (Deen and Kataki, 2003; Carter, 2005; Dolan et al., 2006; Yang et al., 2008). In contrast, SOC at the end of 28 yr of NT in Indiana was greater than with conventional tillage at a depth of 0 to 30 cm and to a cumulative depth of 100 cm, despite observation of a reverse tillage effect at the 30- to 50-cm depth (Gál et al., 2007). To overcome experimental limitations of increasing variability with lower C concentration at depth, numerous soil samples (Yang et al., 2008) and frequent sampling events throughout time have been suggested (Franzluebbers and Stuedemann, 2005). Results from single-point-in-time measurements could be either verified with multiple samplings or improved by obtaining a more robust temporal estimate of SOC sequestration, rather than a rate assumed from a homogenous starting point or from relative comparison with another land use (e.g., NT vs. conventional tillage, not knowing if conventional tillage were in an assumed steady-state condition).

Table 5. Comparative analysis of soil organic C concentration and its coefficient of variation as affected by subsampling within a paddock at the end of 12 yr of cattle grazing bermudagrass in Georgia (data from Franzluebbers and Stuedemann, 2009).

Sampling depth	Soil organic C conc.					
	Subsampling within paddocks (experimental units = 54)				Pooling subsamples (experimental units = 18†)	
	Zone 1 (near shade)	Zone 2 (midway)	Zone 3 (far from shade)	CV	Paddock mean	CV
cm	g kg^{-1}			%	g kg^{-1}	%
0–15	23.8	18.5	20.0	20	20.7	9
15–30	5.6	5.1	5.9	43	5.6	23
30–60	2.2	2.0	2.4	45	2.2	19
60–90	0.8	0.8	1.0	39	0.9	16
90–120	0.6	0.5	0.6	93	0.6	48
120–150	0.3	0.4	0.5	60	0.4	42

† Averaged across three zones within each paddock.

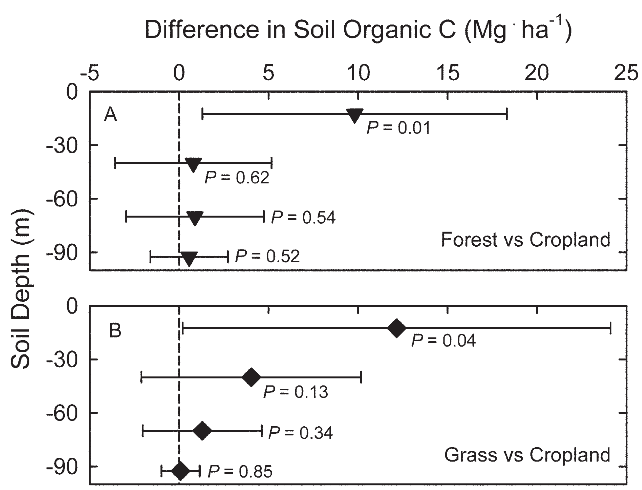

Fig. 8. Relative difference in soil organic C between (a) forest and cropland and (b) grass and cropland as affected by depth in the profile (data from McCracken, 1959).

Frequent sampling of SOC could reduce the detection limit for SOC sequestration. Sampling the surface 30 cm of soil at initiation and at the end of 5 yr of pasture management resulted in a least significant difference of 1.1 Mg ha^{-1} yr^{-1} (Franzluebbers and Stuedemann, 2005). Annual sampling of the surface 6 cm of soil in the same systems resulted in a least significant difference of only 0.3 Mg ha^{-1} yr^{-1} (Franzluebbers et al., 2001). Regression analysis of SOC determination during each of 7 yr in the middle of long-term NT experiments resulted in significantly positive C sequestration estimates in Georgia (0.81 Mg ha^{-1} yr^{-1}; Franzluebbers et al., 2007) and in North Carolina (0.55 Mg ha^{-1} yr^{-1}; Franzluebbers and Brock, 2007). For an experiment expected to last ≥10 yr, a reasonable strategy would be to collect soil every 2 to 4 yr to obtain at least four points for regression analysis with time, while not sampling too frequently to avoid unnecessarily high research cost and exhaustion.

On-farm surveys can be a valuable approach to evaluate practically achievable SOC sequestration; however, sampling designs should be robust enough to overcome potentially distracting assumptions (e.g., uniform starting conditions, peculiar management influences, subtle differences in landscape features, etc.), which cannot be readily verified under such uncontrolled conditions. Replication of fields under similar management should be a high priority. Sampling a single field with pseudo-replication results in a limited scope of inference. The SOC survey reported by Causarano et al. (2008) illustrates an experimental design that maximized the scope of inference by selecting paired long-term land use conditions within a general area (county) and repeating this approach in three different counties in each of five different states across two different Major Land Resource Areas (Piedmont and Coastal Plain). To get a good representation of each field sampled, eight cores were composited. An inclination might have been to collect multiple samples (pseudo-replicates) from within each field. Although this would have produced a better estimate of the field mean, it would have contributed nothing to experimental variation and separation of land use means.

CONCLUSIONS

A preponderance of evidence suggests that conservation agricultural systems in the southeastern United States will sequester a significant amount of organic C in the surface 20 cm of soil. Both conservation-tillage cropland and perennial pastures exhibit high SOC sequestration. Whether significant subsurface SOC sequestration can be achieved within a human generation in this warm, humid climate remains debatable. Sufficient data to support deep-profile SOC sequestration are lacking. Sampling strategies for future research endeavors should consider spatial and temporal opportunities to increase experimental power to detect statistically significant SOC sequestration. With 45 Mha of agricultural land in the southeastern United States, 28 Tg C yr^{-1} could be reasonably calculated (0.45 Mg C ha^{-1} yr^{-1} on 26 Mha of cropland and 0.86 Mg C ha^{-1} yr^{-1} on 19 Mha of pastureland) as potentially sequestered in soil organic matter. This highly significant amount of SOC sequestration is vital to improve the soil fertility of these historically degraded lands and to offset greenhouse gas emissions in the region.

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