Environmental Pollution 116 (2002) S127–S135

ENVIRONMENTAL POLLUTION

www.elsevier.com/locate/envpol

Spatial variability of soil organic carbon in grasslands: implications for detecting change at different scales

R.T. Conant*, K. Paustian

Natural Resource Ecology Laboratory, Colorado State University, Fort Collins, CO 80523-1499, USA

"Capsule": *The efficacy of soil sample collection techniques to characterize grassland soil carbon was found to precisely estimate soil carbon changes at the local to national scales.*

Abstract

Extensive data used to quantify broad soil C changes (without information about causation), coupled with intensive data used for attribution of changes to specific management practices, could form the basis of an efficient national grassland soil C monitoring network. Based on variability of extensive (USDA/NRCS pedon database) and intensive field-level soil C data, we evaluated the efficacy of future sample collection to detect changes in soil C in grasslands. Potential soil C changes at a range of spatial scales related to changes in grassland management can be verified ($\alpha = 0.1$) after 5 years with collection of 34, 224, 501 samples at the county, state, or national scales, respectively. Farm-level analysis indicates that equivalent numbers of cores and distinct groups of cores (microplots) results in lowest soil C coefficients of variation for a variety of ecosystems. Our results suggest that grassland soil C changes can be precisely quantified using current technology at scales ranging from farms to the entire nation. © 2001 Elsevier Science Ltd. All rights reserved.

Keywords: Soil carbon; Grasslands; USDA/NRCS pedon database; Soil sampling; Spatial variability

1. Introduction

Soil organic C is the largest C reservoir in many terrestrial ecosystems including grasslands, savannas, boreal forests, tundra, some temperate forests, and cultivated systems, comprising as much as 98% of ecosystem C stocks in some systems (Schlesinger, 1977). Globally, the amount of C stored in soil is equal to the amount stored in vegetation and in the atmosphere combined (Schimel, 1995). A substantial portion of C fixed by vegetation is transferred to the soil annually (Raich and Nadelhoffer, 1989), a portion of which is refractory material with long turnover times (Falloon and Smith, 2000; Paul et al., 1997); the rest decomposes relatively rapidly and is returned to the atmosphere as CO_2. Thus soil C is a large, relatively dynamic component of terrestrial C stocks.

Historically, intensive cultivation in the US has resulted in the transfer of about 1 Pg of soil organic matter to the atmosphere in the form of CO_2 (Kern, 1994). Soil organic matter (SOM) losses due to conversion of native grasslands to cultivated agriculture are both extensive and well documented (Davidson and Ackerman, 1993; Haas et al., 1957; Kern and Johnson, 1993; Schlesinger, 1986) and losses due to overgrazing and poor pasture management have also been observed (Abril and Bucher, 1999; Conant and Paustian, 2001; Fearnside and Barbosa, 1998). However, historical SOM losses can potentially be reversed, and atmospheric C sequestered, with improved agricultural management. In the United States, agricultural conservation practices such as reduced tillage, improved fertilizer management, elimination of bare fallowing, use of perennials in rotations, and use of cover crops can potentially sequester large amounts of atmospheric C (Paustian et al., 1997). Similarly, cultivated areas converted to well-managed permanent grassland, as pastures or rangelands, constitute potential C sinks. Within established pastures, soil C can be increased by eliminating soil disturbances and increasing primary production through improved grazing management, fertilization, sowing improved forage species and legumes, or irrigation (Conant et al., 2001).

While improved agricultural or pasture management can sequester C in soils (Sampson et al., 2000), there are several challenges associated with verifying changes in soil C. Many factors influence soil C, including temperature, precipitation, NPP, and soil physical char-

* Corresponding author. Fax: +1-970-491-1965.
E-mail address: conant@nrel.colostate.edu (R.T. Conant).

PII: S0269-7491(01)00265-2

acteristics (Parton et al., 1987), all of which are spatially variable. The result is substantial variability in soil C, with coefficients of variation as high as 20% even in a visually uniform cultivated field (Robertson et al., 1997). As variability increases, the minimum number of samples needed to detect a given level of change increases. Furthermore, short-term changes in soil C are usually small relative to the amount of C in soil. In grasslands, for example, soil C may be sequestered at a rate of around 0.46 Mg C ha^{-1} year^{-1} in surface horizons (Conant et al., 2001), against a background of 30–80 Mg C ha^{-1} in most temperate grassland soils (Conant et al., 2001).

Measurement and verification needs vary between broad-scale applications, such as national-level greenhouse gas inventories and design of government policy, versus project-based sequestration efforts that might be undertaken by private landowners or cooperatives at a more local scale. In the former instance, the main emphasis is likely to be on accurate quantification of carbon changes nationally, with less need for attribution of changes to specific localities and/or practices. In the latter, project-level, case, attribution to specific landowners and practices is likely to be important. In both instances, however, there are tradeoffs between the ability to detect change (precision) and the number of samples required (which is directly related to cost).

In this paper we evaluate some limitations of sampling to quantify soil C changes in managed grasslands for broad-scale inventories/assessments and smaller-scale project-level quantification. First, we used a large compilation of soil data for the entire US, the USDA/NRCS (US Department of Agriculture/Natural Resource Conservation Service) pedon database (NSSC, 1997), to examine broad-scale soil C variability and the implications for detecting soil C changes at national, state, and county levels. Second, farm-level analysis, based on analysis of soil C variability within plots sampled to assess soil C levels in managed pastures, was used to evaluate the efficacy of different sampling schemes designed to maximize sensitivity to field-level changes in soil C following changes in land use or management. These spatially intensive sampling schemes could be used for benchmark site sampling, model validation, or to assess impacts of land management changes as part of a mitigation project. Results presented here are part of a larger effort to integrate quantitative measurements of changes in soil C with various tools to extrapolate and interpolate those measurements.

2. Materials and methods

2.1. NRCS PEDON database

The USDA/NRCS pedon database is a compilation of soil information collected during soil mapping or detailed soil investigations and assembled by the USDA/NRCS (NSSC, 1997). Each record, collected to 'represent the central concept of a soil series, the central concept of a map unit but not of a series, or to bracket a range of properties within a series or landscape' (NSSC, 1997), contains information about horizonation, geographic location, and many soil physical and chemical characteristics. Since soil C concentration and bulk density have been quantified only for a portion of the pedons (65% of those included in this study), we used soil C concentration rather than soil C content for this analysis and assumed that the coefficient of variation (%CV) for soil C content was equivalent to that for soil C concentration. Analysis of those data for which soil C concentration and bulk density were present showed that this assumption was valid at all scales (see below).

Data from the USDA/NRCS pedon database were queried to identify data points from uncultivated grassland-derived soils (Mollisols), which were the focus of this study. Uncultivated data points were distinguished from cultivated data points by the absence of an Ap horizon; only data from the surface 20 cm, regardless of horizon designation (A or B, but not Ap), were used for this analysis. Pedons containing organic horizons were excluded from the analysis. Various mixtures of total C, organic C, and inorganic C occur in the USDA/NRCS pedon database, but since organic C was the focus of this research, we only used data from the organic C field.

Not all data points within the USDA/NRCS pedon database are precisely georeferenced, but all contain the state and county designation. For this analysis, USDA/NRCS pedon data from uncultivated grasslands were summarized at the national, state, and county scales. Measured soil C variability coupled with the number of samples collected enabled us to evaluate sample size requirements to detect changes of a certain magnitude. Use of the USDA/NRCS pedon database in Nebraska for other purposes prompted selection of Nebraska for state-level analysis. Within Nebraska, Dundy County was chosen due to the large number of uncultivated grassland data points located there. Although these samples do not represent a random sample, soil C was normally distributed at all three scales.

Along with minimum, maximum, and mean soil C values for each scale, %CV was calculated at each scale as an indication of soil C variability. The relationship between %CV of samples analyzed and minimum detectible difference between initial samples (from the USDA/NRCS pedon database) and samples to be collected in the future (Sokal and Rohlf, 1981), was derived from the general relationship used to calculate the number of samples required to detect a difference of a give magnitude:

$$n = \frac{2 \times \left(Z_\alpha + Z_\beta\right)^2 \times \sigma^2}{\Delta^2} \qquad (1)$$

where Δ is the average change in soil C, σ^2 is the population variance, n is the number of samples, and Z_α and Z_β are critical values from the Z table. Since σ^2 is rarely known (Steel et al., 1997), sample variance is substituted for population variance and t values are used instead of Z values by using the correction factor in Eq. (2).

$$t_{\mathrm{corr}} = (\text{error d.f.} + 3)/(\text{error d.f.} + 1) \tag{2}$$

Solving for the average change in soil C results in Eq. (3).

$$\Delta = \frac{1.42 \times (Z_\alpha + Z_\beta) \times t_{\mathrm{corr}} \times \sigma}{n^{1/2}} \tag{3}$$

Since the %CV is equal to the standard deviation (σ) over the mean (μ), multiplying μ/μ gives the change (Δ) in soil C as a function of %CV, μ, and number of samples (4).

$$\Delta = \frac{1.42 \times (Z_\alpha + Z_\beta) \times t_{\mathrm{corr}} \times \%\mathrm{CV} \times \mu}{n^{1/2}} \tag{4}$$

This relationship, which enables estimation of the number of samples required to detect change of a certain magnitude, was evaluated at all three scales to examine implications for change detection with future re-sampling.

For illustrative purposes, we chose a soil C stock change of 2.3 Mg ha^{-1}, based on an average rate of soil C change due to improved pasture management of 0.46 Mg ha^{-1} year^{-1} (Conant et al., 2001) applied over a 5-year period. However, any other desired soil C stock change level could be specified and used to calculate the number of samples required to detect that change, at a given level of confidence. To express soil C stock changes in absolute terms (rather than in percent changes in mean values), we used the mean percent C values from the pedon database, at each spatial scale, and assumed a bulk density of 1.1 g soil cm^{-3} for grassland soils (based on bulk density data from the USDA/NRCS pedon database) and a depth of 20 cm.

2.2. Farm-level soil sampling

Soil samples were collected at four sites in a forest to pasture chronosequence and from one site that had been under pasture for 48 years following long-term cultivation (Table 1). Sites were chosen to assess soil C sequestration following conversion to pasture and how changes in pasture management influence soil C (Conant and Paustian, 1998). Slope, aspect, and soil series were uniform across sites, ensuring that land use and land use history were the primary factors influencing soil C content.

Fields with different management histories were intensively sampled in order to detect differences between sites and to measure changes in soil C over time. Our sampling scheme was based on that used by the Canadian Prairie Soil Carbon Balance Project (Ellert et al., 2001), which was designed to maximize the ability to detect changes in soil C over time by ensuring that exact sample locations can be relocated, limiting horizontal variability of soil C. Within each field three microplots, each consisting of six regularly aligned soil cores, were sampled in early spring 1999 (Fig. 1a). Microplots were always oriented in the same direction. The location of the northeastern-most core was determined using differential GPS, and a relocatable Skotchmark EMS magnetic ball marker (3M Corporation, Austin, TX, USA) was buried at 2 m depth.

A Giddings hydraulic soil-coring rig was used to collect 6.5 cm diameter soil cores to a depth of 1 m; samples were not compacted during sample collection. Soil samples were returned to the laboratory and weighed, sieved to pass a 2-mm sieve, and ground to fine powder. Root material was removed by hand picking during sieving. Soil C concentrations were determined on individual samples using a LECO CHN-1000 analyzer (LECO Corporation, St. Joseph, MI, USA). Bulk density was calculated using volume of sample collected and the weight of soil in the sample; sample weight was corrected for soil moisture, root, and rock content. Samples were collected for several increments to a depth of 1 m, but results presented here are for the top 20 cm only.

To evaluate the variance components of field-level sampling using a microplot approach we compared our design of three microplots, with six cores each, with three alternative designs varying the number of microplots, but all having a total of 18 cores per site (Fig. 1). To generate the soil C contents for each design, the measured values for the 18 cores at each site were

Table 1
Characteristics of Virginia forest to pasture chronosequence field sites

Site No.	Current management	Historical management	Year of conversion
1	Forest	Forest	–
2	Intensively grazed pasture	Forest	1996
3	Intensively grazed pasture	Forest	1985
4	Intensively grazed pasture	Forest	1950
5	Intensively grazed pasture	Cultivation	1950

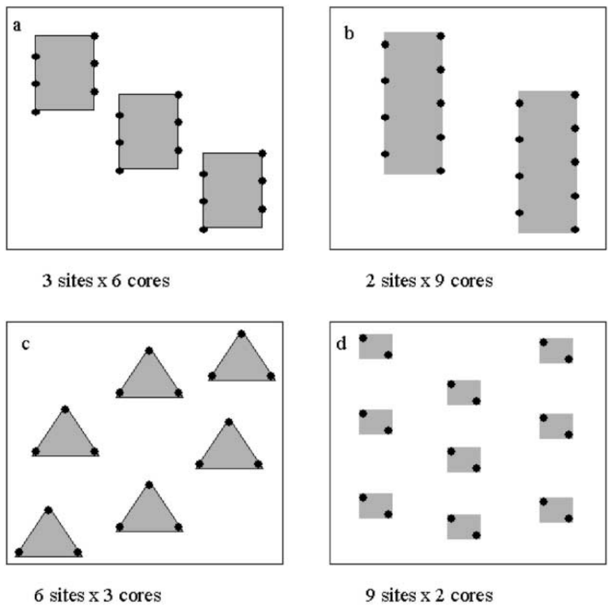

Fig. 1. Distribution of soil cores and microplots for each of the four different potential sampling schemes: (a) three microplots by six cores; (b) two microplots by nine cores; (c) six microplots by three cores; and (d) nine microplots by two cores.

randomly distributed across the 18 sample locations for 1000 unique combinations of each sampling scheme (i.e. 2×9, 3×6, etc.). The average%CV was calculated for the 1000 different configurations of each sampling scheme. The contribution of variance within and between microplots to total site variance was calculated using a random effect ANOVA (Steel et al., 1997). The total sum of squares was calculated for all cores from all microplots using the usual formulas for balanced ANOVA. The sum of squares between cores within a microplot was determined by subtracting the sum of squares between microplots from the sum of squares of all cores (Steel et al., 1997). The variance for each component was calculated as the individual sum of squares divided by the respective degrees of freedom (which were different for each configuration) and the proportion of total variance explained by within- and between-microplot variance was calculated by dividing component variance by total variance.

3. Results

3.1. NRCS PEDON database

General characteristics of soil C from the USDA/NRCS pedon database are shown in Table 2. More than 2700 samples from uncultivated grassland-derived soils in the USDA/NRCS pedon database contained soil C data (Table 2). Though the number of samples decreased dramatically for smaller sample areas, sampling density for Dundy County (0.49 samples per 1000 ha) was slightly greater than that at the state level (0.42 samples per 1000 ha) and much greater than at the national (0.012 samples per 1000 ha) level. The range of soil C values increased with area; the minimum was smallest and the maximum was largest at the national scale. Mean soil C was largest at the national-scale and smallest at the county-scale, reflecting the shift in the area of focus to the more arid grasslands in the state (Nebraska) and county (Dundy county in western Nebraska) selected (Table 2). Coefficient of variation decreased with scale and was lowest at the county-scale and highest at the national-scale (Table 2).

At 90% confidence levels, changes in soil C due to changes in grassland management over a 5–10 year period would be verifiable using random resampling with fewer than 470 samples in uncultivated grasslands at all scales (Fig. 2). Nationally, average changes in grassland C stocks on the order of 2.3 Mg C ha^{-1} can be detected by collecting 501 samples. Considerably fewer samples are required to verify changes of the same magnitude in Nebraska (224 samples), and fewer still for Dundy County (34 samples). Even smaller changes in soil C can be verified with relatively few samples at the county scale; for example, average changes of 1 Mg C ha^{-1} should be detectible with 90% confidence by collecting only 129 additional samples. While only 224 samples are required to detect an average change of 2.3 Mg C ha^{-1} at the state level, over 900 additional samples are required to detect an average change of 1 Mg C ha^{-1}. Nationally, detection sensitivity increases very slowly below 2 Mg C ha^{-1}, and 1176 samples are required to detect an average change of 1.5 Mg C ha^{-1}. A mean change of 1.5 Mg C ha^{-1} for the grassland area of 218 mha (Table 2) implies an increase of about 65 MMTC year^{-1} over 5 years, which is similar to the mean estimate of grassland C sequestration potential (70 MMTC year^{-1}) estimated by Follett et al. (2001).

Decreasing Type I error associated with remeasurement from 10 to 5% impacts the number of samples required to detect a difference of the same magnitude, particularly at the state and national-scales (Fig. 3). An additional 51 samples at the state-level and 166 samples at the national-level are required to detect a change of 2.3 Mg C ha^{-1} at 95% confidence than at 90% confidence.

3.2. Farm-level soil sampling

Coefficients of variation across all sampling design configurations and sites ranged from 13 to 24%. The largest %CV was for the 2×9 configuration at the recent forest conversion site (Site 2) and the smallest %CV was the 9×2 configuration at Site 3. For each configuration, %CV was largest at Site 2, which was recently converted from forest to pasture (Fig. 4). The forest sites had the second largest %CV under all

Table 2
Minimum, mean, and maximum soil C and %CV for all samples within uncultivated, grassland-derived soils at the national, state, and county scales[a]

Scale	Soil C (%)			CV (%)	Area (1000 ha)	No. samples
	Minimum	Mean	Maximum			
USA	0.05	1.78	14.25	63	218,000	2717
Nebraska	0.06	1.37	11.10	54	947	394
Dundy County	0.37	0.71	1.05	39	30.6	15

[a] Area (calculated using DISCover database; see Sampson et al., 2000) and number of samples from the USDA/NRCS pedon database are also shown.

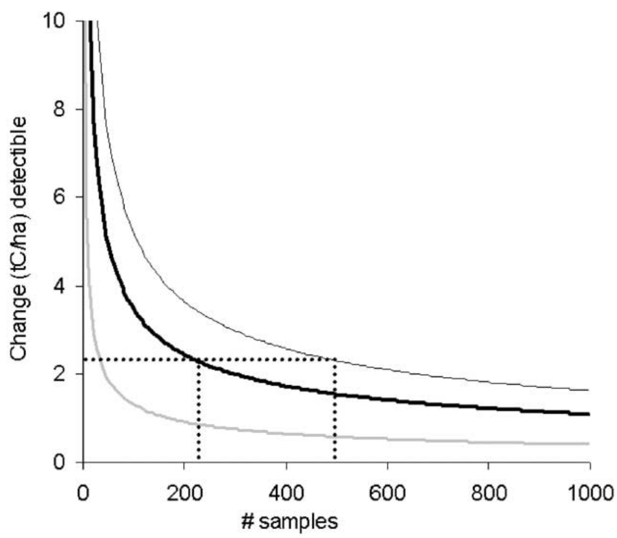

Fig. 2. Relationship between sampling intensity and magnitude of change in soil C detectible at the 90% confidence level by resampling for cultivated grasslands at the national, state, and county scales. The relationship is based on the USDA/NRCS pedon database.

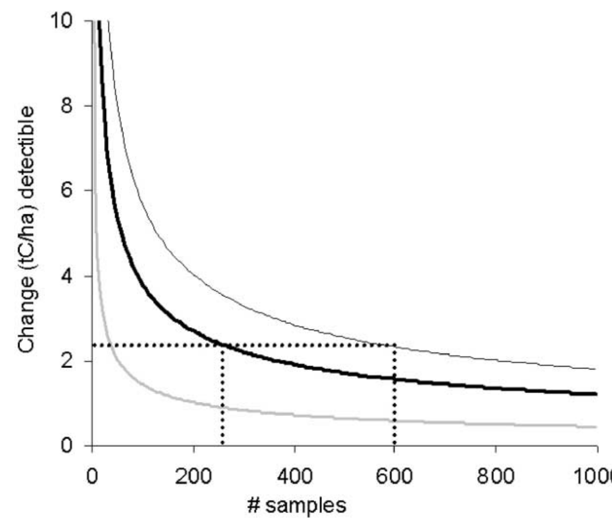

Fig. 3. Relationship between sampling intensity and magnitude of change in soil C detectible at the 95% confidence level by resampling for cultivated grasslands at the national, state, and county scales. The relationship is based on the USDA/NRCS pedon database.

configurations; %CV at the three older pasture sites was always lower (Fig. 4). The 2×9 configuration led to the largest %CV at all sites except for Site 4 where the 9×2 configuration had a slightly larger %CV. The 6×3 configuration had the lowest %CV at three sites (1, 2, and 4) and the 9×2 configuration resulted in the lowest %CV at the other two sites (3 and 5). Of the configurations evaluated, the 6×3 configuration resulted in the lowest %CV. However, since the 2×9 configuration had the lowest %CV for two of the sites, the optimal configuration appears to be somewhere between the 3×6 and the 2×9 configurations.

Coefficients of variation for the actual 6×3 configurations ranged from 12 to 19% and were lower than the average of the 1000 random combinations of the actual data at all five sites. Based on the statistics of minimum detectible difference, the range of %CVs obtained from field sampling indicates that between 14 and 28 samples would be required to detect a change of 2.3 Mg C ha^{-1}.

The relative distribution of variance components (i.e. variance within versus between microplots) as a function of the sampling configuration (i.e. 2×9, 3×6, etc.) was fairly similar across all sites (Fig. 4). The maximum amount of variation contributed by cores was greatest with the fewest number of cores at all sites. Under the 2×9 configuration, within microplot variance contributed the maximum portion of total variance (about 70%). As the number of cores per microplot decreased, the portion of total variance due to within micrplot variance decreased to 56–57% for the 3×6 configuration and then to 45–47% for the 6×3 configuration. The portion of variance within microplots at Sites 2, 3, and 4 was lowest for the 6×3 configuration, but lowest for the 9×2 configuration at the other two sites. Minimum within microplot variance as a potion of total site variance was 37%, occurring under the 9×2 configuration and Sites 1 and 5. Averages for all sites indicate that equivalent amounts variability were contributed by microplots and cores for the 6×3 configuration.

4. Discussion

Our results suggest that changes in soil C brought about by changes in pasture management can be quantified precisely using current technology at scales ranging

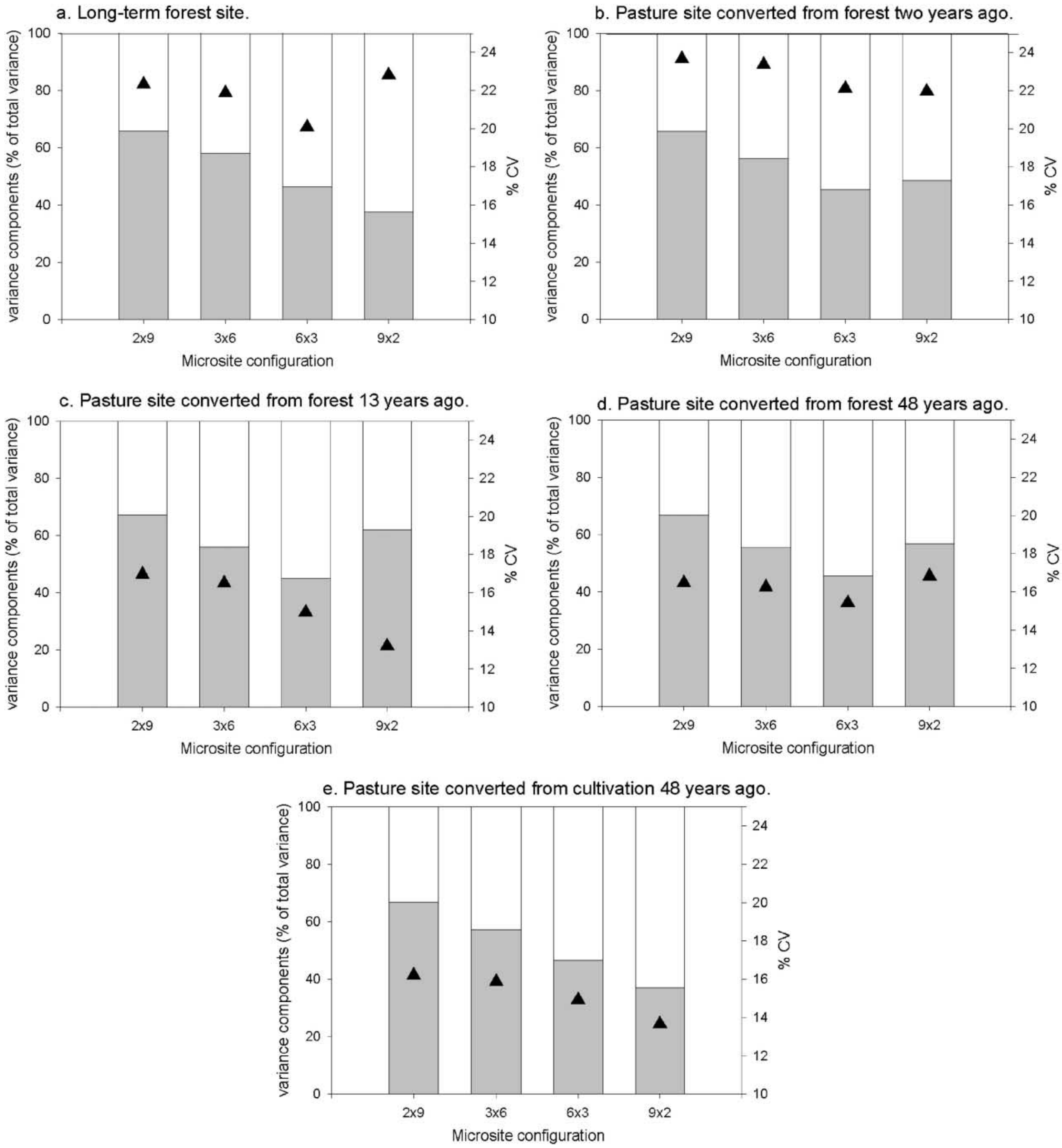

Fig. 4. Coefficient of variation (triangles; scale on right), and portion of total soil C variability due to within (shaded portion of bars) and between (unshaded portion) microplot variability (scale on left) for (a) long term forest site, (b) a site recently converted from forest and under well-managed pasture for 2 years, (c) a site converted from forest and under well-managed pasture for 13 years, (d) a site converted from forest and under well-managed pasture for 48 years, and (e) a site recently converted from cultivation and under well-managed pasture for 48 years.

from individual farms to the entire nation. As with any sampling design, a tradeoff exists between the number of samples required and the desired precision, and the number of samples collected and analyzed is directly related to costs associated with verification. This work demonstrates that small changes in soil C at a very fine scale, such as an individual farm or conglomeration of a few farms, can be detected with careful sampling, yet these methods are clearly impractical at the national scale. Our research also suggests, however, that the number of samples required to detect changes in soil C likely to occur over a 5-year period is not unduly large at the state and national level. Similar results should be expected for other land-use changes.

Further stratification of samples to quantify soil C spatial covariation with driving variables such as soil type, temperature, precipitation, or species composition/ crop rotation could greatly increase the power of the analysis by enabling more detailed interpolation, but would require more samples to detect changes of the same magnitude. Our use of the USDA/NRCS pedon database as a baseline measured the ability to detect total changes in soil C only; this method does not distinguish between contributions from different pasture management practices. Furthermore, there is no experimental control so that changes in soil C due to CO_2 fertilization, increased N deposition, climate change, or any other aspect of global change cannot be distinguished from changes due directly to changes in management. This work does, however, suggest that we might be able to detect management-induced changes soil C in uncultivated grasslands soils at different scales using the USDA/NRCS pedon database as a reference.

Our assessment using the USDA/NRCS pedon database required four assumptions that may decrease the certitude and limit the ability to extrapolate our results. First, since a portion of the data points in the USDA/NRCS pedon database contain no information on bulk density, we performed our analysis without considering variation in bulk density or management-related changes in bulk density. Though not as large as the variation in soil C concentration, in a visually uniform cultivated field bulk density had a %CV of 8% (Robertson et al., 1997). The %CV for bulk density for farm-level sampling in Virginia was 6% (Conant and Paustian, 2001). Coefficients of variation for soil C content at the county and national scales was within 1% of that for soil C concentration at those scales, while the %CV at the state level was 14% less for the soil C content than for soil C concentration. Therefore, utilizing results based on soil C concentration seems to result in a somewhat conservative estimate of coefficients of variation at the state scale and reasonable estimates at the county and national levels. Second, we selected Nebraska and Dundy County in part because of the large portion of grassland pedons, resulting in more samples collected per unit area and, thus, a lower %CV. Also, Dundy County contains more USDA/NRCS pedon data points than surrounding counties. Therefore, these state- and county-scale results may not be representative of results from other states or counties. Third, we treated the pedon data as if they were randomly located in uncultivated grassland soils, though this is not likely; this assumption could lead to underestimates of samples required to detect changes of a given magnitude if %CV were to be larger for randomly located samples. Finally, we assumed that the various rates of response to changes in grassland management would be normally distributed, though it is unclear whether this is true.

Project-, farm- or field-level changes in soil C due to changes in management can be detected using standard soil sampling protocols. Though time demands associated with microplot survey and layout favors establishments of few microplots with more cores, our results suggest that an intensive sampling scheme somewhere between the 3×6 and 2×9 configurations (such as 4×5 or 5×4) will optimize sampling to detect changes over time across a variety of sites. Soil C %CVs from these forest/pasture sites in Virginia agree with soil C and other soil organic matter parameters measured in a variety of intensively sampled fields elsewhere (Bragato and Primavera, 1998; Mollitor et al., 1980; Mottonen et al., 1999; Robertson et al., 1997) and suggest that this may be near the lower limit for %CV in soil C content.

Our estimate that between 14 and 28 samples are required to detect a change of 2.3 Mg C ha^{-1} may be confounded since these results are from non-random microplots selected, in part, for their uniformity. While %CV fell well within the range of those observed elsewhere (see above), many potential sources of variation, such as soil inclusions, variable slopes, variation in land use history, etc. are not accounted for in this analysis. Therefore, our inferences about future sample sizes would not necessarily apply to areas with divers slopes, soils, and land use histories. These sources of variation could be examined with a stratified or randomized sampling strategy. Nevertheless, our results indicate that sampling design can be optimized at the field-scale to detect small changes in soil C using equivalent numbers of microplots and cores within a site. Furthermore, using data from these sites in a space for time substitution to examine the impacts of changes in land use on soil C, we were able to detect significant ($P<0.05$) differences between treatments that were as small as 3.59 Mg C ha^{-1}, but means that differed by 2.05 Mg C ha^{-1} were not significantly different (Conant and Paustian, 2001). This type of intensive sampling is well-suited for benchmark sites that can be resampled over time, sample collection from experimental sites designed to assess impacts of various types of management, or sites used for model validation.

5. Conclusions

This research evaluated two methods of detecting changes in soil C driven by changes in agricultural management: (1) regional (county-, state-, or national-level) analysis relying on previously collected data and (2) field-level sampling. Statistically derived expectations about our ability to detect changes with future resampling were assessed for each method. Our results show that detection of changes in soil C likely to occur over a 5–10 year period can be accomplished at broad scales (county, state, or national). Sample-based verification of changes in soil C content can accurately be carried out at any scale, but require (1) antecedent measurements to establish initial soil C content, (2) an a priori sampling method designed to achieve the desired precision and spatial resolution, and (3) understanding of the tradeoff between precision and costs associated with sampling.

Two-tiered sampling, combining intensive field-or farm-level sampling and diffuse, geo-referenced regional or national sampling, could form the basis of a national network capable of quantifying national-level changes in soil C. Intensive sites could be used to assess the impacts of various management practices, while diffuse sampling, including remeasurements over 5–10 years, can be used to verify aggregate changes in soil C stocks. Both could be used for model verification at different scales. Previous discussions on the feasibility of national scale monitoring and verification of soil C sequestration have, in some instances, implied the necessity of intensive sampling over large areas. For example, Subak (2000) estimates a total cost of one billion dollars for a national soil C monitoring program for the first commitment period (2008–2012) specified in the Kyoto Protocol, employing a sampling density of one sample per 20 acres of US cropland (0.12 samples ha^{-1}), with the conclusion that 'these costs are high compared with the potential value of carbon offsets.' However, we believe that such assessments ignore existing data and overestimate the sampling density required to detect modest changes in soil C.

Previous soil C research has produced a wealth of information that can be exploited to refine our ability to detect changes in soil C. While collection and analysis of randomly located soil samples can produce precise estimates of changes in soil C over time, broadly stratified national or regional soil sampling, coupled with model-based quantification of soil C changes (e.g. Eve et al., 2000, 2001; Paustian et al., 2001), existing data, and sampling targeted to verify impacts of different management practices, offers an alternative likely to decrease sampling intensity (i.e. intensive sampling is only required for model validation and benchmark sites) yet increase certainty and applicability of results.

Acknowledgements

Discussions with N.A. Scott, K.R. Tate, G.R. Smith, and M. Sperow aided significantly in the development of this project. Thanks to participants of the Canadian Prairie Soil Carbon Balance Project (B.G. McConkey, in particular) for assisting in implementation of the intensive farm-level sampling method. The comments of N.A. Scott, B.G. McConkey, and one anonymous reviewer on an earlier version of this manuscript improved the manuscript. J. Zumbrunnen's statistical assistance was instrumental in completion of both the USDA/NRCS pedon and farm-level analyses. This paper was presented at the USDA Forest Service Southern Global Change Program sponsored Advances in Terrestrial Ecosystem: Carbon Inventory, Measurements, and Monitoring Conference held 3–5 October 2000 in Raleigh, North Carolina.

References

Abril, A., Bucher, E.H., 1999. The effects of overgrazing on soil microbial community and fertility in the Chaco dry savannas of Argentina. Applied Soil Ecology 12, 159–167.

Bragato, G., Primavera, F., 1998. Manuring and soil type influence on spatial variation of soil organic matter properties. Soil Science Society of America Journal 62 (5), 1313–1319.

Conant, R.T., Paustian, K., 1998. Carbon sequestration in pastures. In: Orlic, M. (Ed.), The Ruminant Livestock Efficiency Program Annual Conference. US Environmental Protection Agency, Athens, GA, pp. 32–40.

Conant, R.T., Paustian, K., 2001. Potential soil carbon sequestration in overgrazed grassland ecosystems. Global Biogeochemical Cycles (in preparation).

Conant, R.T., Paustian, K., Elliott, E.T., 2001. Grassland management and conversion into grassland: Effects on soil carbon. Ecological Applications 11 (2), 343–355.

Davidson, E.A., Ackerman, I.L., 1993. Change in soil carbon inventories following cultivation of previously untilled soils. Biogeochemistry 20, 161–193.

Ellert, B.H., Janzen, H.H., McConkey, B.G., 2001. Measuring and comparing soil carbon storage. In: Lal, R., Kimble, J.M., Follett, R.F., Stewart, B.A. (Eds.), Assessment Methods for Soil Carbon. CRC Press, Boca Raton, pp. 131–1465.

Eve, M.D., Paustian, K., Follett, R.F., Elliott, E.T., 2000. A national inventory of changes in soil carbon from national resources inventory data. In: Lal, R., Kimble, J.M., Follett, R.F., Stewart, B.A. (Eds.), Assessment Methods for Soil Carbon. CRC Press, Boca Raton, FL, pp. 593–610.

Eve, M.D., Paustian, K., Follett, R.F., Elliott, E.T., 2001. An inventory of carbon emissions and sequestration in US cropland soils. In: Lal, R., McSweeny, K. (Eds.), Soil Management for Enhancing Carbon Sequestration. SSSA Special Publ, Madison, WI, pp. 51–65.

Falloon, P.D., Smith, P., 2000. Modeling refractory soil organic matter. Biology and Fertility of Soils 30, 388–398.

Fearnside, P.M., Barbosa, R.I., 1998. Soil carbon changes from conversion of forest to pasture in Brazilian Amazonia. Forest Ecology and Management 108, 147–166.

Follett, R.F., Kimble, J.M., Lal, R., 2001. The potential of US grazing lands to sequester soil carbon. In: Follett, R.F., Kimble, J.M., Lal, R. (Eds.), The Potential of US Grazing Lands to Sequester Soil Carbon. CRC Press, Chelsea, MI, pp. 401–430.

Haas, H.J., Evans, C.E., Miles, E.F., 1957. Nitrogen and Carbon Changes in Great Plains Soils as Influenced by Cropping and Soil Treatments. Technical Bulletin No. 1164, USDA.

Kern, J.S., 1994. Spatial patterns of soil organic matter in the contiguous United States. Soil Science Society of America Journal 58, 439–455.

Kern, J.S., Johnson, M.G., 1993. Conservation tillage impacts on national soil and atmospheric carbon levels. Soil Science Society of America Journal 53, 200–210.

Mollitor, A.V., Leaf, A.L., Morris, L.A., 1980. Forest soil variability on Northeastern flood plains. Soil Science Society of America Journal 44, 617–620.

Mottonen, M., Jarvinen, E., Hokkanen, T.J., Kuuluvaninen, T., Ohtonen, R., 1999. Spatial distribution of soil erosterol in the organic layer of a mature Scots pine (*Pinus sylvestris* L.) forest. Soil Biology and Biochemistry 31, 503–516.

NSSC, 1997. National Soil Survey Laboratory Characterization Data. USDA-NRCS, Lincoln, NE.

Parton, W.J., Schimel, D.S., Cole, C.V., Ojima, D.S., 1987. Analysis of factors controlling soil organic matter levels in Great Plains grasslands. Soil Science Society of America Journal 51 (5), 1173–1179.

Paul, E.A., Follett, R.F., Leavitt, S.W., Halvorson, A., Peterson, G.A., Lyon, D.J., 1997. Radiocarbon dating for determination of soil organic matter pool sizes and dynamics. Soil Science Society of America Journal 61 (4), 1058–1067.

Paustian, K., Collins, H.P., Paul, E.A., 1997. Management controls on soil carbon. In: Paul, E.A., Paustian, K., Elliot, E.T., Cole, C.V. (Eds.), Soil Organic Matter in Temperate Agroecosystems. CRC Press, Boca Raton, pp. 15–49.

Paustian, K., Brenner, J., Killian, K., Cipra, J., Williams, S., Kautza, T., Bluhm, G., 2001. State-level analyses of C sequestration in agricultural soils. In: Kimble, J.M., Lal, R., Follett, R.F., Stewart, B.A. (Eds.), Agricultural Practices and Policies for Carbon Sequestration in Soil. CRC Press, Boca Raton, FL (in press).

Raich, J.W., Nadelhoffer, K.J., 1989. Belowground carbon allocation in forest ecosystems: global trends. Ecology 70 (5), 1346–1354.

Robertson, G.P., Klingensmith, K.M., Klug, M.J., Paul, E.A., Crum, J.R., Ellis, B.G., 1997. Soil resources, microbial activity, and primary production across an agricultural ecosystem. Ecological Applications 7 (1), 158–170.

Sampson, R.N., Scholes, R.J., Cerri, C., Erda, L., Hall, D.O., Handa, M., Hill, P., Howden, M., Janzen, H., Kimble, J., Lal, R., Marland, G., Minami, K., Paustian, K., Read, P., Sanchez, P.A., Scoppa, C., Solberg, B., Trossero, M.A., Trumbore, S., Van Cleemput, O., Whitmore, A., Xu, D., Burrows, B., Conant, R., Liping, G., Hall, W., Kaegi, W., Reyenga, P., Roulet, N., Skog, K.E., Smith, G.R., Wang, Y., 2000. Additional human-induced activities—article 3.4. In: Watson, R.T. et al. (Eds.), Land Use, Land-Use Change, and Forestry. Cambridge University Press, Cambridge, pp. 180–281.

Schimel, D.S., 1995. Terrestrial ecosystems and the carbon cycle. Global Change Biology 1, 77–91.

Schlesinger, W.H., 1977. Carbon balance in terrestrial detritus. Annual Review of Ecology and Systematics 8, 51–81.

Schlesinger, W.H., 1986. Changes in soil carbon storage and associated properties with disturbance and recovery. In: Trabalka, J.R., Reichle, D.E. (Eds.), The Changing Carbon Cycle: A Global Analysis. Springer Verlag, New York, pp. 194–220.

Sokal, R.R., Rohlf, F.J., 1981. Biometry. W.H. Freeman and Co, New York.

Steel, R.G.D., Torrie, J.H., Dickey, D.A., 1997. Principles and Procedures of Statistics: A Biometrical Approach. McGraw-Hill, New York.

Subak, S., 2000. Agricultural soil carbon accumulation in North America: considerations for climate policy. Global Environmental Change 10, 185–195.