Sensitivity of the Century model for estimating sequestered soil carbon using coarse- and fine-scale map data sources in north central Montana
by Ross Stanley Bricklemyer

A thesis submitted in partial fulfillment of the requirements for the degree of Master of Science in Land Resources and Environmental Sciences
Montana State University
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Abstract:
The use of agricultural best management practices, most notably the adoption of no-till systems, has become a potential technique to sequester (store) carbon in soils and help mitigate the effects of global warming. Efficient sampling designs and the use of process-based soil organic carbon (SOC) dynamics models are potential methods of monitoring and verifying soil carbon change. This research combined field-scale soil sampling and the use of the Century model to explore field-scale SOC variability and the effects of soil texture input data sources (STATSGO and SSURGO databases) on predicted SOC dynamics in north central Montana. Using soil-landscape associations for field stratification and sampling of micro sites for paired management comparisons was an efficient design for measuring SOC (CV = 8-13%). An optimal sampling design of 4 microsites by 2 cores or 3 microsites by 3 cores provided reliable detection of a tillage effect on SOC, given the magnitude of differences (1.3 to 5.1 t C ha^-1) and degree of variability measured. Including the effects of soil clay content as a covariant may provide unbiased estimations of the effects of tillage on SOC among sites, particularly for coarse scale comparisons. The Century model accurately predicted SOC content at five sites using site-specific soils data (10% deviation from measured values). Neither the STATSGO (1:250,000 scale) nor SSURGO (1:24,000 scale) databases adequately predicted soil textures, nor supplied adequate soil textural information for use in the Century model and so introduced potential error to field-specific predictions. Century proved to be sensitive to the effects of clay content when predicting the amount of SOC in a particular field; however the model was insensitive to the effects of soil texture on C sequestration as a result of no-till management. The methods used to measure SOC and the Century model proved to be useful tools for determining carbon stored due to no-till management. Additional research is needed to determine if a consistent relationship exists between soil texture and the effect of tillage on SOC and thus determine if adjustments are needed to the Century model's treatment of soil texture.

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SOIL CARBON USING COARSE- AND FINE-SCALE MAP DATA

SOURCES IN NORTH CENTRAL MONTANA

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APPROVAL

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Ross Stanley Bricklemyer

This thesis has been read by each member of the thesis committee and has been found to be satisfactory regarding content, English usage, format, citations, bibliographic style, and consistency, and is ready for submission to the College of Graduate Studies.

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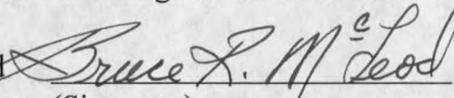
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Dedicated to my loving and supportive family

Ross Joseph, Helen Marie, and Scott Arnold Bricklemyer

Thank you for your support and patience.

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ABSTRACT

The use of agricultural best management practices, most notably the adoption of no-till systems, has become a potential technique to sequester (store) carbon in soils and help mitigate the effects of global warming. Efficient sampling designs and the use of process-based soil organic carbon (SOC) dynamics models are potential methods of monitoring and verifying soil carbon change. This research combined field-scale soil sampling and the use of the Century model to explore field-scale SOC variability and the effects of soil texture input data sources (STATSGO and SSURGO databases) on predicted SOC dynamics in north central Montana. Using soil-landscape associations for field stratification and sampling of microsites for paired management comparisons was an efficient design for measuring SOC (CV = 8-13%). An optimal sampling design of 4 microsites by 2 cores or 3 microsites by 3 cores provided reliable detection of a tillage effect on SOC, given the magnitude of differences (1.3 to 5.1 t C ha^{-1}) and degree of variability measured. Including the effects of soil clay content as a covariant may provide unbiased estimations of the effects of tillage on SOC among sites, particularly for coarse scale comparisons. The Century model accurately predicted SOC content at five sites using site-specific soils data (10% deviation from measured values). Neither the STATSGO (1:250,000 scale) nor SSURGO (1:24,000 scale) databases adequately predicted soil textures, nor supplied adequate soil textural information for use in the Century model and so introduced potential error to field-specific predictions. Century proved to be sensitive to the effects of clay content when predicting the amount of SOC in a particular field; however the model was insensitive to the effects of soil texture on C sequestration as a result of no-till management. The methods used to measure SOC and the Century model proved to be useful tools for determining carbon stored due to no-till management. Additional research is needed to determine if a consistent relationship exists between soil texture and the effect of tillage on SOC and thus determine if adjustments are needed to the Century model's treatment of soil texture.

REVIEW OF LITERATURE

Global Warming

Responding to mounting evidence of increasing atmospheric carbon dioxide

(CO_2) concentrations contributing to global warming, the nations of the world have come

together in the United Nations Framework Convention on Climate Change to begin

reducing atmospheric CO_2 concentrations (UNFCCC, 1994). The desirability of

sequestering carbon in terrestrial ecosystems to mitigate global warming has been

emphasized by recent political developments, most notably the Kyoto protocol (Masood

1997). If ratified, the Kyoto Protocol would have required the USA to reduce its net

carbon dioxide emissions to 7% below 1990 levels (UNFCCC, 1997). When negotiated

in 1997, Kyoto recognized direct CO_2 reductions and considered agricultural sinks (i.e.

offsets) only provisionally, as a means to meet target CO_2 reductions. In the subsequent

Conference of Parties (COP 6.5) in Bonn, Germany (July 2001), political agreement was

reached to recognize agricultural sinks as emissions offset and is worded as "application

of net-net accounting (net emissions or removals over the commitment period less net

removals in the base year) for agricultural activities (cropland management, grazing land

management and revegetation)" (COP 6.5: Bonn Agreement). More recently, at the

COP 7 meeting in Marrakech, Morocco (November 2001), emission offsets were

termed "emission removal units". The Kyoto Protocol has spawned the idea of emission

removal credits to offset greenhouse gas emissions which have become a topic of interest

in the production agriculture and agroeconomics sectors. The recent decision of the U.S. government to withdraw from the Kyoto Protocol does not exempt the USA. from addressing its CO_2 emission problem. The Bush administration has committed the USA to address the global carbon issue and U.S. policy includes emission reductions and emission removals (i.e. carbon sequestration) as a part of that effort. An 18% reduction in *greenhouse gas intensity,* which is a measure of emissions per unit gross domestic product (GDP), has been set as a target (Pianin 2002). National incentives, such as "green" payments for agricultural management changes and a potential market-based carbon credit trading system are being developed that will likely coincide with the 2008-2012 targeted emission reduction commitment period scheduled in Kyoto.

"And we will look for ways to increase the amount of carbon stored by America's farms and forests through a strong conservation title in the farm bill. I have asked Secretary Veneman to recommend new targeted incentives for landowners to increase carbon storage."

President George W. Bush - February 14, 2002

Greenhouse Gases

Gases that contribute to the greenhouse effect or the radiative forcing of the atmosphere include water vapor, carbon dioxide (CO_2), nitrous oxide (N_2O), methane(CH_4), ozone (O_3), chlorofluorocarbons (CFCs) and hydrochlorofluorocarbons

(HCFCs). Of these gases, water vapor, CO_2, N_2O, CH_4 and O_3 are naturally occurring, however it is generally accepted that anthropogenic sources of CO_2, N_2O and CH_4 emissions in addition to man-made gases such as CFCs and HCFCs, are resulting in a rapid increase in global temperatures (Levitus et al. 2001). According to a greenhouse gas (GHG) emissions inventory released by the U.S. Environmental Protection Agency (U.S. EPA), CO_2, CH_4, and N_2O comprised 98% of the U.S. greenhouse gases in 1999 (U.S. EPA 2001). Global warming potentials (GWP) of all greenhouse gases are standardized relative to CO_2 and are reported in units of teragrams of carbon dioxide equivalents (Tg CO_2 Eq.). The GWP of CO_2, CH_4, and N_2O are 1, 23, and 296, respectively, for a 100 yr time period (IPCC 2001). This means that one unit mass of N_2O has 296 times the atmospheric radiative forcing as one unit mass of CO_2. Total U.S. GHG emissions rose from 6,038 Tg CO_2 EQ to 6,746 Tg CO_2 EQ with an annual growth rate of 1.2% from 1990 to 1999. Individually, total CO_2 and N_2O emissions increased from 1990 to 1999, whereas total CH_4 emissions decreased slightly (U.S. EPA 2001).

Role of Agriculture

Greenhouse Gas Emissions

According to the EPA's greenhouse gas emissions inventory, agriculture plays a significant role in greenhouse gas emissions. In 1999, agriculture contributed 488 Tg CO_2 EQ, which is 7.2% of the total GHG emissions in the U.S.A. (U.S. EPA 2001). Most of

the agricultural emissions are associated with CH_4 and N_2O. Agriculture has direct and indirect sources of fossil fuel derived CO_2 emissions associated with fuel use, N fertilizer production and lime application (Lal et al. 1999). CO_2 emissions also occur through oxidation of soil organic matter in cultivated soils especially from organic (i.e. peat and muck) soils (Eve et al. 2002a). However, overall agricultural soils were a net sink for CO_2, sequestering 77 Tg CO_2 EQ in 1998 and 1999 (US EPA 2001).

The agriculture industry was responsible for 28% of the U.S. total methane emissions and 69% of U.S. total nitrous oxide emissions (US EPA 2001). Methane emissions were primarily from enteric fermentation and manure management in agriculture. Enteric fermentation is a natural process in ruminant digestion and, therefore the majority of CH_4 emissions are from dairy and beef cattle production. The management of livestock and poultry manure that promotes anaerobic decomposition is not only a significant source of methane; it is also, to a smaller degree, a source of nitrous oxide (US EPA 2001).

The single largest source of N_2O is agricultural soil management. Microbial processes of nitrification and denitrification naturally produce nitrous oxide in soils. The addition of nitrogen through synthetic and organic fertilizer use, manure application, and legume crop production enhances N_2O emissions. Soil management practices that affect the flux of N_2O and other GHGs include irrigation, drainage, tillage, fertilizer application, and fallowing of cropland. No-till management has been reported to emit less NO_2 as compared to more intensively tilled systems when spring thaw conditions on fallow are

considered (Lemke et al. 1999). In a controlled split-plot experiment in the Parkland

region of Alberta, annual losses of NO_2-N from intensively tilled management ranged from

0.1 to 4.0 kg N ha^{-1} with highest overall losses occurring on fallow ground followed by N

fertilizer treatments and pea residues (Lemke et al. 1999).

Soil Organic Carbon

Agriculture plays an important role in U.S. soil carbon dynamics. The U.S. land

area in cropland is approximately 170 million hectares or about 19% of the total U.S. land

area. The amount of historic soil organic carbon (SOC) lost from the U.S. cropland

carbon pool is estimated at 5000 Tg (Bruce et al. 1999). Therefore, there is a potential

to sequester, or store, nearly 5000 Tg of carbon from the atmosphere in U.S. cropland

alone if soil C levels were rebuilt to original levels. However, the economic potential to

sequester C in soil (the amount that could be sequestered at a plausible cost) is lower (45

to 100 Tg yr^{-1}) and is dependant on the market price of carbon relative to the cost

associated with implementing carbon sequestering practices (McCarl et al. 2002).

Land sinks have become an emission removal consideration because CO_2

consumed by plants and sequestered in soils is a large part of the global carbon cycle (Lal

et al. 1998). The global terrestrial carbon pool (2.1×10^6 Tg C) is approximately three

times larger than the atmospheric carbon pool (0.75×10^6 Tg C) (Flach et al. 1997).

Photosynthesis by plants converts CO_2 from the atmosphere into carbon-based organic

materials (i.e. stems, leaves, grain, roots, etc). The residual plant materials are converted

by biophysical reactions to soil organic matter, which is comprised of approximately 58%

carbon. Carbon sequestration in agricultural systems is carbon dioxide removed from the

atmosphere by plants and stored in soil via biological processes (McConkey et al. 1999).

Land management changes have the potential to sequester C and restore soil organic

carbon in the soil (Paustian et al. 1997a; Peterson et al. 1998). Management practices

that promote carbon sequestration in cropland include reduction of soil tillage and erosion,

crop residue management, increased cropping intensity, diversification of crop rotations,

and efficient fertilizer management (Campbell et al. 2000a,b; Liang et al. 1999; Lal et al.

1998; Peterson et al. 1998; Potter et al. 1997).

Terrestrial ecosystems might prove to be a net sink for atmospheric carbon,

however, the notion of significant reductions in atmospheric CO_2 as a result of agricultural

management changes has met some skepticism. Full accounting of CO_2 flux should be

considered when estimating annual SOC gains and developing carbon policy. Certain

changes in agricultural management increase the amount of SOC but there is a carbon

cost associated with all agricultural operations. The carbon cost associated with fuel use,

production and application of N fertilizer and lime, irrigation, and application of manure

should be discounted from their net contribution to carbon sequestration (Schlesinger

2000). In some cases, the C cost associated with the afore mentioned operations

significantly reduces the net storage of C (Schlesinger 2000). Additionally, a full

accounting of agriculturally related greenhouse gases should be included in the overall

potential of management practices to not only reduce atmospheric carbon dioxide but also

to reduce net greenhouse gas emissions (Robertson et al. 2000). Tillage in concert with the practice of summerfallow has been shown to release significant

amounts of nitrous oxide as compared to no-till management during the spring thaw in Alberta (Lemke et al. 1999).

Tillage Effects

Tillage promotes soil C losses due to increased erosion and microbial decomposition. Removal of grain and crop residues, and the practice of summerfallow reduces the overall amount of C input. Physical disturbance by tillage breaks down soil aggregates in the surface horizon, thus increasing the potential for soil erosion. Tillage also increases the potential for microbial decomposition rates due to greater bioavailability of organic matter and increasing aeration (Paustian et al. 1997a). After a tillage event, soil microorganisms rapidly metabolize soil organic matter and release CO_2 as a metabolic byproduct (Reicosky 1997). Continued tillage over many years has reduced the overall organic carbon content of soil by 20 to 50 percent (Tiessen et al. 1982; Rasmussen and Parton 1994; Lal et al. 1998).

It is generally accepted that reduction in soil disturbance in production agriculture promotes carbon storage. Not only can the reduction or elimination of tillage increase the amount of organic carbon in soil (Yang and Kay 2001, Paustian et al. 1998; Kern and Johnson 1993), but tillage also influences the distribution of C in the soil profile. In a comparison of chisel plowing versus moldboard plowing, Yang and Kay (2001) found no

significant difference in the total amount of SOC in the 0-20 cm depth. However, SOC

levels were greater in the 0-10 cm depth and less in the 10-20 cm depth with chisel

plowing compared to moldboard plowing whereas the latter had a more evenly distributed

SOC content from 0-20 cm. Conservation tillage, including zero till or no-till management

(NT) is considered a best management practice that promotes carbon sequestration and

increases soil organic carbon in fields that were previously managed with tillage. In 39

paired comparisons of no-till and convention tillage effects on soil C, no-till averaged 8%

greater SOC than conventional tillage in the mineral soil (Paustian et al. 1997b). Similarly,

organic C concentrations were higher in coarse- and fine-textured soils (13 and 44%,

respectively) under stubble mulch as compared to moldboard plowing, respectively.

Using the IPCC inventory method of estimating soil C change, conversion from

conventional tillage to no-till has been estimated to increase soil organic C at a rate of

0.20 Mg C ha^{-1} yr^{-1} for 20 yr in the mountain region of the United States (includes

Montana, Idaho, Colorado, Wyoming, Utah, Arizona and New Mexico) (Eve et al.

2002b). Reviews by Paustian et al. (1997a) and West and Marland (2002) estimated

that a conversion to no-till from conventional tillage in the U.S. could increase SOC by

0.3 and 0.34 Mg C ha^{-1} yr^{-1}, respectively, over a 20 yr period. In another review, Follet

(2001) estimated that initial C sequestration rates resulting from a change from

conventional tillage to no-till could be 0.3 to 0.6 Mg C ha^{-1} yr^{-1} for the U.S. Great Plains

and 0.1 to 0.5 Mg C ha^{-1} yr^{-1} for the Canadian prairies. In a modeling study of the

Canadian prairies, the Century model estimated C sequestration rates of approximately

0.13 Mg C ha^{-1}yr^{-1} (Smith et al. 2001). The adoption of no-till management carbon across the U.S. has the potential to sequester significant amounts of carbon, 275-763 Tg CO$_2$ Eq. annually (Lal et al.1999; Bruce et al.1999) which represents approximately 4 to 11% of the U.S. total GHG emissions in 1999. Management changes associated with the adoption of no-till agriculture commonly involve a concert of operational changes that contribute to increased soil carbon. Changes that commonly occur with adoption of no-till systems in semiarid regions are management of crop residues, varying crop rotations, fertilizer application, and increasing cropping intensity (McConkey 1999). Adequate fertilization, coupled with increasing cropping intensity, increases crop residues and thus carbon inputs, a major factor influencing SOC change in no-till systems (Campbell et al. 2001a). Decreasing fallow frequency in concert with fertilized wheat production showed carbon gains whereas unfertilized wheat showed little or no change after 10 to 30 years (Campbell et al. 2000a; Campbell et al. 2001b). The role of fertilizer use and it's contribution to NO2 emissions in agroecosystems is not well understood and is currently being debated (Lemke et al. 1999; Robertson et al.2000). Annual SOC gains due to increased cropping intensity and fertilization in semiarid southwestern Saskatchewan were measured at 0.32 Mg C ha^{-1}yr^{-1} for continuous wheat with nitrogen and phosphorus added (N+P), 0.28 Mg C ha^{-1}yr^{-1} for a wheat-lentil rotation (N+P), and 0.23 Mg C ha^{-1}yr^{-1} for a fallow-fall Rye-wheat rotation (N+P) (Campbell et al. 2000a).

Policies and Incentives for Soil C Sequestration

Agriculture can contribute to greenhouse gas reductions by emission reductions

and greenhouse gas removals. Emission reductions do not reduce the present amount

GHGs in the atmosphere; rather reductions refer to lowering the amount of GHGs that are

released into the atmosphere from agricultural activities such as fossil fuel use, livestock

production, and application of fertilizers. It is important to remember that significant levels

of GHGs are still emitted even after emission reductions. Greenhouse gas removals

actually reduce the present concentration of GHGs in the atmosphere relative to 'business

as usual', partially offsetting increases from other sources. Presently, biological sinks,

including carbon sequestered in agricultural soils, are the only method of removing

greenhouse gases such as CO_2. After a management change, soil carbon will increase

until the system reaches a new equilibrium (Fig. 1.1). At this point soil will no longer

sequester additional carbon provided that management practices continue unchanged.

This is when the issue of *permanence* and the risk associated with maintaining the

sequestered carbon stock becomes apparent. Fig. 1.1 models hypothetical gains in SOC

due to a change in management. Time, in this schematic represents, 10 to 30 yr. Carbon

that has been sequestered in agricultural soils can be a source of CO_2 with a return to the

previous conventional management system (Fig. 1.1). To deal with the maintenance risk of

the new carbon stock, policy makers must account for treatment of C sequestration

liability.

C sequestration is reversible
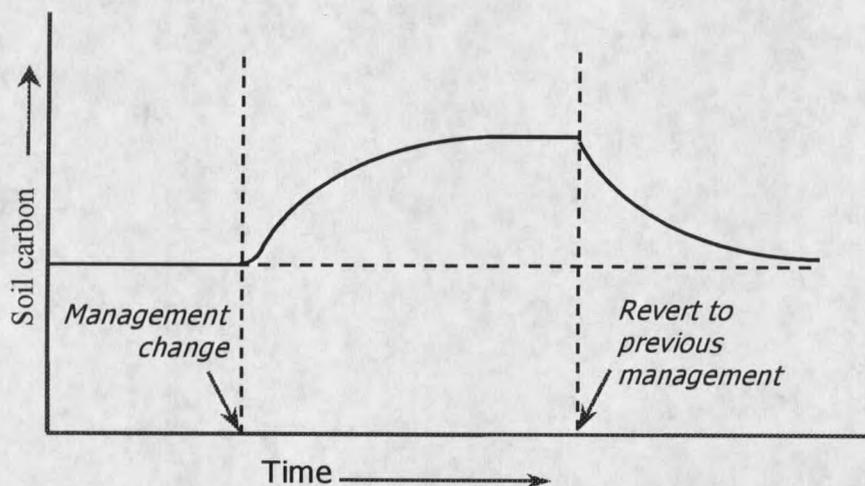
Figure 1.1. General schematic of carbon sequestration dynamics showing management changes in relationship to carbon storage (J. Bennett, pers. comm., 2002; graph adapted from Janzen et al. 1998).

Two scenarios addressing the permanence issue have been proposed. Discussion of the liability issue is easily understood if we present the treatment of sequestered carbon as a commodity or a service, understanding that carbon sequestered in agricultural soils is a service provided by farmers rather than a commodity to be sold. As a commodity (Fig. 1.2), farmers would assume the responsibility of sequestering and maintaining the carbon stock. In the case that the production system should fail, farmers could be forced to repurchase the removal credits defined in the contract. In this scenario, permanence risk is transferred to the farmer. Sequestering carbon as a service (i.e. carbon banking) would minimize the permanence risk to farmers (Fig 1.3). If removal units are treated as an emission storage service and the system should fail, the farmer forfeits only the added

value created by the carbon stored (Bennett 2002). Reversion to management previous to no-till is not anticipated once carbon contracts have been fulfilled. If ancillary long-term benefits (in addition to C storage) due to the adoption of best management practices occur, reversion to previous management would be unlikely regardless if C incentives are maintained. Whether carbon sequestration is treated as a commodity or a service, the policy must be efficient for successful implementation.

Designing efficient policies to sequester carbon in cropland is critical if significant amounts of greenhouse gases are to be reduced. There are three general designs that could feasibly be implemented to sequester carbon (Antle and Mooney 2001). Command-and-control policies could be used to force agricultural producers to use specific management practices. This system of regulation would be inefficient due to the heterogeneity of farm management practices and distribution of soil resources. It would not be feasible to prescribe mandatory management practices to individual producers that would fit their specific farm capital and soil resources to grow the crops suitable to their agricultural environment.

C sequestration as a Commodity
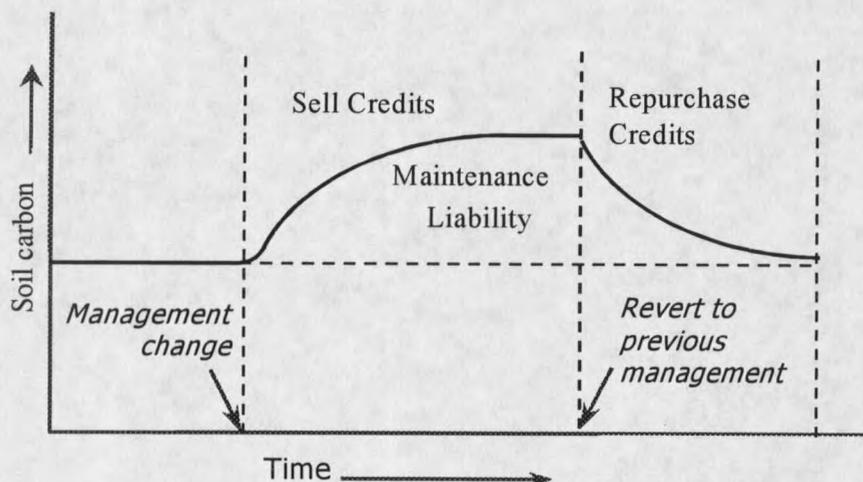
Figure 1.2. General schematic of carbon sequestration loss liability treated as a commodity with the farmer assuming the risk of maintaining the carbon stock (J. Bennett, pers. comm., 2002, graph adapted from Janzen et al. 1998).

C Sequestration as a Service
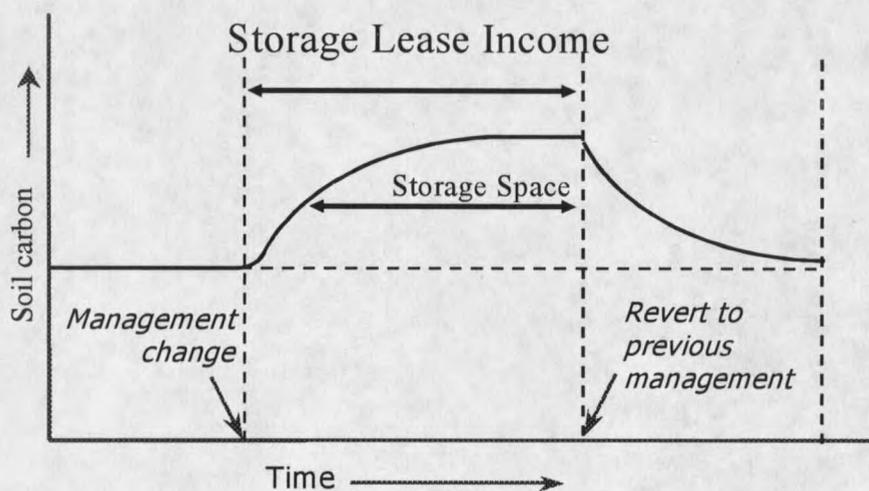
Figure 1.2. General schematic of carbon sequestration loss liability treated as a service with the service buyer assuming the risk of maintaining the carbon stock (J. Bennett, pers. comm., 2002, graph adapted from Janzen et al. 1998).

Secondly, incentive policies could 1) be used to encourage producers to engage in carbon sequestering practices through subsidies and 2) discourage carbon releasing practices using taxes. This method could also be an inefficient policy because it is based on management practices rather than reducing greenhouse gases. Adopting certain practices could reduce atmospheric CO_2 by storing carbon in the soil, however, those same practices may release other, more potent greenhouse gases as a result (i.e. "Leakage"). Some incentive-based policies are already being used to enhance agricultural environmental quality. These volunteer programs, the Conservation Reserve Program (CRP), Wetland Reserve Program (WRP), and Environmental Quality Incentives Program (EQUIP), offer payments and other financial support to farmers who participate.

The third potential policy for sequestering carbon is a market-based system for trading carbon. Economic analysis of carbon sequestration in Montana has shown that increasing SOC would be more efficient if conducted on a market-based "per tonne" of carbon payment rather than a governmental subsidy to producers on a "per acre" basis such as the CRP (Antle et al. 2001; Antle and Mooney 2002). Costs would also vary as a function of the types of management practices used. For example, estimated marginal cost for carbon sequestered on a per acre payment to producers to seed land into permanent grass ranges from \$34 to \$500 t^{-1} C, compared to \$12 to \$150 t^{-1} C in a system that would compensate farmers for adopting a continuous cropping system (Antle et al. 2001, 2002; Antle and Mooney 2002). The theory behind "per tonne" market-based carbon credits is that CO_2 emitters can purchase carbon that is being sequestered

for use as an offset against CO_2 emissions. Emitters would purchase carbon credits (emission removals) to meet emission reduction targets. The budding carbon credit trading system could be analogous to the sulfur dioxide trading that has been in practice in the USA since 1995 (Antle et al. 2001; Joskow et al. 1998; US EPA 2002). The agricultural industry is in a position to receive compensation for making land management changes, and increasing the amount of SOC in their fields. This appears to be a win-win scenario for agricultural producers and for reducing atmospheric carbon. The market for cropland carbon credits remains in its conceptual stages and a "cap-and-trade" policy limiting the amount of carbon dioxide emitted is needed to spur the development of this market. It is important to remember that carbon sequestration in agricultural soils is a temporary solution to managing atmospheric carbon. Carbon sequestration is a tool that can be used in the interim as industry transitions into less greenhouse gas intensive production.

Carbon Contract Design

The design of carbon credit contracts is linked to the policies and incentives for sequestering soil carbon discussed previously. Two types of contracts could be used to sequester, per-hectare (practice-based) and per-tonne (carbon-based) contracts (Antle et al. 2002). From a C storage perspective, the per-tonne contract is more efficient and would cost less than the per-hectare contract because the per-tonne contract would allow farmers to implement the most efficient production system for their particular goals (Antle

et al. 2002). Per-hectare contracts are likely to be used with government programs that specify so-called "best management practices" for soil C sequestration. For tradable emissions allowances, per-tonne contracts are more likely to be used because they specify the amount of C to be sequestered. Tradable emissions reduction contracts are likely to be denominated in standardized quantities of carbon such as 10,000 metric tonnes. Therefore, given the size of the typical farm and given that C accumulation rates are commonly less than one metric tonne per hectare per year, a large number of land units (fields) will have to be aggregated to make a single tradable contract.

To establish per-hectare or per-tonne contracts, it will be necessary for both buyers of carbon (government or private entities paying for carbon) and for sellers (farmers) to anticipate the amount of C that can be stored on a given land unit. These estimated rates of soil C accumulation will likely be determined from experimental data and from application of simulation models such as the Century model. Thus, it is important to establish how accurate a model like Century is for predicting soil C rates under field conditions.

After contracts are implemented, it will also be necessary to verify that the specified practices are being used or that the contracted amount of carbon is being accumulatied on the contracted land units. Costs for monitoring management practices for both a per-hectare and per-tonne contracts would be similar; however, the per-tonne contract would also require that a sampling procedure be implemented to verify that the specified amount of carbon was being accumulated. Note that it would not be necessary

to measure C on each square hectare of land, rather, a statistically valid sampling procedure could be used at specified time intervals to achieve a desired accuracy at the lowest cost (Antle et al. 2002). The cost of sampling and the desired accuracy will determine the spatial scale at which sampling will be conducted.

In summary, research is needed to provide information about the reliability of soil sampling procedures and simulation models to support the development of carbon contracts. If C contracts are developed for farm-specific credits, the focus of field research will likely be toward optimizing sampling designs and reducing analysis cost. If the contracting unit is much larger (e.x. county or agroecozone scales), then the research focus will likely shift toward uncertainty analyses of process-based models for estimating soil C change. This study focuses on site-specific C sampling optimization, validation the Century model for site-specific C modeling in Montana, and explores the use of existing soil databases for site-specific C modeling.

Century Model Description

In furthering understanding of the processes that govern soil carbon dynamics in terrestrial ecosystems, process-based predictive models have been developed to simulate carbon dynamics. The Century model is one such process-based model that estimates soil organic carbon changes based on macroenvironmental gradients, management, and soil and plant properties over long periods (>30 yrs) (Parton et al. 1987). Century simulates soil organic matter (SOM) formation, calculates decomposition rates based on first-order

kinetics that vary as a function of soil temperature and moisture, soil texture, and other variables. The model uses a monthly time step and includes both carbon and nitrogen flows. Input data required for Century includes monthly mean maximum and minimum air temperatures, monthly precipitation, soil texture (%sand, %silt, %clay), atmospheric and soil N inputs (eg. fertilizer N, and N fixation), land management practices (eg. crop rotations, crop types, tillage systems), and initial soil C and N content (Metherell et al. 1993; Parton et al. 1987). The Century model integrates a number of submodels including a soil organic matter, water balance, plant production, and nitrogen, phosphorus and sulfur submodels.

Soil Organic Matter Submodel

The Century model segregates soil organic matter into three fractions, 1) an active pool with short turnover time [1-5 yrs] 2) a slow pool with moderate turnover time [20-40 yrs], and 3) a recalcitrant or passive pool with long turnover time [200-1500 yrs]. Decomposition rates of SOM and plant residues are assumed to be microbially driven with a portion of carbon lost as CO_2 as a result of microbial respiration (Metherell et al. 1993; Parton et al. 1987). Products of decomposition enter either a surface microbial pool or one of the SOM pools with each pool having different potential maximum decomposition rates. The potential decomposition rate is reduced as a function of soil water content and temperature, and is potentially increased as a result of cultivation (Parton et al. 1987). The effect of soil water content on decomposition is calculated using

water budget and soil temperature models (Parton 1984). Soil texture influences the

decomposition turnover rate of the active SOM pool (fine textures decrease turnover

rate) and the efficiency of SOM stabilization in the slow pool (clay soils have greater

stabilization rates).

Decomposition is calculated by the general equation:

$$dCi/dt = k_i \times M_d \times T_d \times C_i \qquad \text{[Eqn 1]}$$

where C_i = carbon in the pool, i = 1) structural and 2) metabolic soil surface litter, 3)

structural and 4) metabolic soil litter, and 5) active, 6) slow, and 7) passive pools; k_i =

maximum decomposition rate (per week) for the ith pool (k_i^* = 0.076, 0.28, 0.094, 0.35,

0.14, 0.0038, and 0.00013); M_d = the effect of the ratio of precipitation to potential

evapotranspiration on decomposition; T_d = the effect of monthly average soil temperature

on decomposition (Parton et al. 1987). Decomposition rate modifiers (i.e. M_d, T_d, etc.)

are scaled from 0 to 1. Tillage and degree of anaerobic conditions also effect the rate of

SOM decomposition. Structural material decay rates are a function of the preceding and

also the lignin-to-N ratio of the structural material. In addition to the soil organic matter

pools, Century also estimates above and belowground litter pools and a surface microbial

pool. The model assumes the plant residue lignin-to-N ratio controls the split into the

structural and metabolic pools, where a higher ratio places more residues into the

structural pool which has a slower potential decomposition rate. The metabolic and

structural pools have assumed turnover rates of 0.1 to 1 and 1 to 5 years, respectively

(Metherell et al. 1993).

Each of the SOM pools, litter pools, and the surface microbial pools have decomposition rates that vary with respect to monthly soil temperature, and precipitation, litter quality and location (i.e. surface vs buried), soil texture and tillage disturbance (Parton et al. 1987). The active SOM pool is comprised of soil microbes and microbial by-products. The decomposition rate of the active pool (K^d_5) is a function of the silt and clay fraction (T), where

$$K^d_5 = K_5 . (1 - 0.75 \times T). \qquad [Eqn. 2]$$

Stabilization efficiency of active SOM to slow SOM (E_s) is also a function of T, where

$$E_s = (0.85 - 0.68 \times T) \qquad [Eqn. 3]$$

The slow pool consists of resistant plant material from the structural pool and stabilized microbial by-products from the active and surface microbe pools. The passive pool strongly resists decomposition and includes physically and chemically stabilized SOM. The fraction of the structural pool that is lignin influences the decomposition rate of the structural pool. The surface microbial pool is associated with surface litter decomposition. The turnover rate of the surface microbial pool is independent of soil texture and material is transferred directly into the slow SOM pool (Metherell et al. 1993).

Water Balance and Temperature Submodels

Century includes a water balance model that estimates water lost from evaporation and transpiration, water content of the soil, water associated with snowfall, and water flow between soil layers (Metherell et al. 1993; Parton et al. 1987). Water

loss from bare soil or evaporation is a function of standing dead and litter biomass, precipitation, and potential evapotransporation (PET). Potential evapotransporation is estimated using average monthly minimum and maximum air temperatures, and latitude. Other sources of water loss include interception loss (function of aboveground biomass, rainfall, and PET), and transpiration loss (function of live leaf biomass and PET). Interception and bare soil loss are calculated as an adjustment to monthly precipitation. Water added to the soil (net infiltration) is calculated as total monthly precipitation minus the loss associated with interception and bare soil evaporation. The form of precipitation is dependent upon monthly average air temperature (rain > 0°C and snow <0°C).

Water entering the soil is distributed amongst soil layers as a function water content and depth. The soil profile is divided into soil layers of 15 cm depth increments for the first 60 cm and then 30 cm increments below 60 cm. Flow between soil layers occurs after water is added to the top layer and excess water (greater than field capacity) is drained to the layer below. Water drainage is a function of field capacity, wilting point, soil texture, bulk density, and SOM content. Transpiration occurs after water has been added to and redistributed within the soil. Water can additionally be lost through leaching below the bottom soil layer and water that leaches this deep is no longer available for evapotranspiration. P, S, and labile sources of mineral N are leached through the soil layers as water is redistributed through saturated flow.

Average monthly soil temperature near the soil surface is calculated from monthly minimum and maximum soil temperatures. Soil temperatures are derived from monthly

minimum and maximum air temperatures and are modified by canopy biomass where greater canopy biomass decreases the maximum soil temperature and increases minimum soil temperature (Parton et al. 1987).

Plant Production Submodel

The Century model is capable of modeling carbon dynamics in grasslands, agricultural crops, forests and savannah (combination of the grassland and forest models including shading and nutrient competition factors) systems (Metherell et al. 1993). Both of the production models assume that maximum monthly plant production is governed by temperature and moisture and are adjusted lower if nutrient deficiencies occur (Metherell et al. 1993). In this study discussion is focused on the grassland/crop production model.

The grassland/crop production submodel simulates plant production dynamics for various herbaceous plant communities and crops by adjusting crop-specific growth parameters which in turn can be tailored to reflect specific growth patterns in defined environments (Metherell et al. 1993). This submodel has pools for live roots and shoots, and for standing dead plant material. Potential production of a particular crop is determined by a defined genetic maximum production and is adjusted as a function of soil temperature and water status, shading by dead vegetation and seedling growth, and nutrient limitations.

Allocation of carbon and production of roots and shoots is influenced by soil temperature where growth occurs when soil temperature is greater than 0°C, reaches

optimal growth at 20-25°C, and declines rapidly above 30-35°C (Metherell et al. 1993).

Biomass production, including roots, shoots and grain, is influenced by the soil water

status and is adjusted by a linear relationship of plant production to water status. Growth

is reduced when the moisture status is less than 0.8 when calculated by

$$\frac{\text{stored soil water in root zone} + \text{precipitation} + \text{irrigation}}{\text{potential evapotranspiration}}$$

Soil texture and thus soil water holding capacity determines the slope of the linear

relationship.

The effects of shading on crops grown from seedlings are a function of the amount

of standing live and dead vegetation. Scaling factors are invoked for partial or full

interception of light from canopy interference. Root growth is proportional to shoot

growth and reflects the dominant carbon allocation to roots in cereal crops and shoots in

root crops. In winter dormant crops such as winter wheat, the root to shoot ratio is held

constant throughout months that the average monthly soil temperature is less than 2°C.

The Century model takes into account that monthly crop growth is influenced

by nutrient limitations. Biomass production is constrained by the most limiting nutrient and

plant nutrient concentration limits are set independently for roots and shoots within defined

upper and lower limits. The Century model also incorporates a fertilizer function where

fertilizer amounts can be user-defined fixed amounts or automatically calculated for crop

production where the automated system can be set to achieve a particular level of the

crop's maximum production. The crop model also simulates the effects of harvest

(Metherell et al. 1993). At harvest, grain is removed from the system and the shoots can be removed from the system or transferred to standing dead and surface residue material. For grain crops, Century calculates a harvest index based on the genetic potential of the crop and water stress during the months of anthesis and grain fill. At the end of the growing season, the crop can be killed as in the case for cereal grains or shoots can regrow in the case of a pasture system. Additionally, the crop model simulates death of roots and shoots during the growing season and models the effects of grazing and fire.

The grassland/crop model has many options relating to soil management. Cultivation options allow the model to simulate the effects of soil disturbance by transferring roots, shoots, standing dead and surface litter into surface and soil litter and standing dead pools (Parton et al. 1987). It is possible to model various tillage systems ranging from a high disturbance system (moldboard plows and off set disks) to an advanced no-till (low disturbance drills and herbicide applications) or organically managed production system (green-manure additions and multiple tillage operations. Each of the cultivation options has a number of estimated parameters that simulate the effects of soil disturbance and mixing on the decomposition rates of soil organic matter in active, slow, passive and structural pools.

Nitrogen Submodel

The nitrogen submodel in Century has the same basic structure as that of carbon flow in the soil organic carbon submodel (Parton et al. 1987). The model assumes that

the carbon to nitrogen (C:N) ratios for structural pools is fixed at 150; however the C:N ratios of the metabolic pools are allowed to float as a function of incoming plant residue N content. The nitrogen content of the metabolic pool is allowed to vary as a function of the remaining N content of plant material that is not used for incorporation into the structural pool. C:N ratios of active, slow and passive SOM pools are allowed to float in a range from 15 to 3, 20 to 12, and 10 to 7, respectively, as mineral N in the surface mineral soil layer increases.

N flows through the model are assumed to be related to C flows and are a product of the C flow and the set C:N ratio for the given pool (Parton et al. 1987). Nitrogen immobilization and mineralization can occur as a function of the initial C:N ratio of the material entering a pool, the C:N ratio of the pool material is entering and the fraction of C lost as CO_2 through respiration. N inputs for the model include atmospheric deposition (N_a), and N_2 fixation through symbiotic and non-symbiotic means (N_f), which are both assumed to be controlled by annual precipitation (Eqn. 4, 5).

$$N_a = 0.21 + 0.0028 \times PPT \qquad \text{(Eqn. 4)}$$

$$N_f = -0.18 + 0.014 \times PPT \qquad \text{(Eqn. 5)}$$

Simulated N losses include ammonia volatilization, leaching, and NO_2 and N_2 volatilization resulting from nitrification and denitrification.

Century has been developed to simulate the effects of various soil management and land use practices. The effects of fertilizer use, irrigation, cultivation, grazing, and fire can be initialized in the model (Parton et al. 1987). Fertilizer can be added to the model

as fixed amounts or as an automatic function based on nutrient demand for a desired level

of production. Similar to fertilizer initiation, irrigation can be set in the model as a fixed

amount or as an automatic application when the available water in the root zone falls

below a specified available water holding capacity level. The amount of water added

through automatic application can be set to a nominated level, to reach field capacity, or

to reach field capacity plus potential evapotranspiration.

The Century model has been used extensively to model soil organic matter and

nitrogen dynamics in a wide array of environments, soil types, and management systems.

The Century model equations for carbon dynamics are reported to be sound based on

comparisons between field measurements and model estimates (Parton et al. 1982, 1987,

Monreal et al. 1997), however some researchers have observed shortcomings in Century

model estimations of grain yield, and thus carbon inputs in semiarid cropland, which in turn

effect SOC estimates (Campbell et al. 2001b). Century has been validated for use under

various soil, climatic, and agricultural practices. Parton et al. (1987) found that the model

adequately estimated soil C values in the Great Plains representing various soil textures

and climates ($r^2 = 0.88$, 0.92, and 0.92 for sandy, medium, and fine textured soils,

respectively). Century tended to overestimate C for fine textures and underestimate C for

sandy soils (Parton et al. 1987). In long-term plots in Sweden the model predicted SOC

values within 5 to 15% of measured values as a function of temperature and initial

composition of soil organic matter (Paustian et al. 1992).

Century has also been validated in eastern Canada where the model estimated

SOC to within 10% of measured values for 4 crop rotations in 3 soil climosequences (Monreal et al. 1997). Using Century to model short-term effects of management on SOM in Quebec, Vorony and Angers (1995) predicted soil microbial biomass to within 8% of measured values. In addition to the validation studies discussed above, carbon change due to cultivation and agricultural management have been modeled using the Century model in diverse climatic regions including the Argentine Pampa (Alvarez et al. 2001), Canadian prairies (Smith et al. 2001; Campbell et al. 2001b; Smith et al. 1997), the U.S. Great Plains (Parton and Rasmussen 1994), and European systems (Falloon and Smith 2002).

Scaling Issues

There is growing interest in using the Century model as a tool for estimating carbon dynamics in situations where a limited amount of field-level information is known for a particular field or farm (Zimmerman et al. 2003). Soil information can be obtained from existing databases at various scales of spatial resolution, which in turn have varying ranges in soil attributes. Existing soil databases such as the State Soil Geographic (STATSGO) at 1:250,000 scale and the countywide Soil Survey Geographic (SSURGO) at 1:24,000 scale, are generalizations of soil patterns across the landscape. Soil mapping units in both the STATSGO and SSURGO databases typically contain more than one defined soil series for our region (as many as 21 in STATSGO and 3 in SSURGO), with each mapping unit and soil series within the having its own soil attributes (NRCS 1995a;

NRCS 1995b). Both databases report a percent composition of each soil component per map unit, however neither database shows the spatial location of the soil components within the map unit. The Century model is initiated using single number variables that describe the soils, climate, and management in a particular scenario. Soil properties for each soil component are reported as a range in values. Therefore choosing representative values for use in the Century model can be important for accurate soil carbon estimations.

STATSGO and SSURGO databases, although found to be highly correlated (r = 0.98), have the potential for differences in soil attributes (Juracek and Wolock 2002). The degree of discrepancy between soil attributes at 1:250,000 scale (STATSGO), 1:24,000 scale (SSURGO), and site-specific differences has been found to be potentially substantial. For example, the soil attributes; clay percent, soil permeability, and hydrologic group were spatially and statistically analyzed for differences (Juracek and Wolock 2002). At small averaging areas (0.01km^2) in Kansas, differences in reported clay percent ranged from -45 to 58% with a standard deviation of 7. As the averaging area increased from 25 to 400 km^2, the range in difference was reduced (-11 to 4%) and the standard deviation decreased to 3 and 2 for the 25 and 400km^2 averaging areas, respectively. The correlation between the databases for percent clay also varied with landscape position. The correlation tended to have a negative relationship with most of the variability in clay % occurring in and around stream networks (-2.5 to -6.5%), meaning that SSURGO tended to have lower values for clay percent as compared to STATSGO. Differences in clay % values stabilized with distance from stream networks

with mean differences in the +/- 1% range. Although the agreement in the databases increased with averaging area and with distance up gradient from stream bottoms, the site-specific ranges are large and could result in substantial differences in model predicted SOC values for a specific farm field.

Justification of Research

Monitoring and verification of sequestered carbon will be a critical part of carbon credit contracts. States in the northern Great Plains, such as Montana, have land area in agriculture that is presumed too expansive for ground-based monitoring and verification to be cost-effective for individual fields, based on the number of samples needed to capture field scale SOC variability (Kaiser, 2000; K. Paustian pers comm 2001). One potential method of reducing monitoring and verification cost is the use of process-based data-driven models, such as Century, to predict soil organic carbon changes. Given that carbon trading will likely require model input data from existing soil taxonomic databases such as STATSGO and SSURGO, understanding the sensitivity of Century to variations in soil input data is critical. Management changes could result in significant differences in sequestered carbon estimates. Additionally, the accuracy of required soil input data could have significant effects on the validity and accuracy of sequestered carbon estimates.

SOIL ORGANIC CARBON VARIABILITY AND SAMPLING OPTIMIZATION IN DRYLAND WHEAT FIELDS IN NORTH CENTRAL MONTANA

Abstract

This research examined soil organic carbon (SOC) sampling variability in no-till and tilled dryland wheat fields in north central Montana. Paired sites were chosen such that soil properties, cropping system, landscape position, and climatic conditions were similar within each no-till/ till pair, but varied between pairs. Soil carbon analyses determined SOC content associated with each management practice. Organic carbon was determined by difference (total C – inorganic C). A LECO CNS-2000 analyzer was used for total carbon analysis and inorganic carbon was determined using a modified calcimeter method. Independent analysis of the 0 to 10 cm depth was used to quantify SOC sampling variability among cores, microsites, and fields. This inter-source variability was used to explore optimization of the number of microsites per field and the number of cores per microsite. An optimal sampling design of 4 microsites by 2 cores or 3 microsites by 3 cores provided reliable detection of a tillage effect on SOC, given the magnitude of differences (1.3 and 5.1 t C ha^{-1}) and degree of variability measured.

Introduction

If a carbon credit trading system emerges, accurate baseline carbon values in agricultural soils will be needed to determine the amount of carbon that has been sequestered (carbon gain) or decomposed (carbon loss) during the carbon contract period. Variability in soil organic carbon and the consistency of sampling procedures and protocols are important considerations in measuring soil carbon. Understanding SOC variability across soil landscapes requires understanding of the interactions of soil texture, slope, aspect, and other related soil properties. Optimal sampling designs for reducing SOC variability and monitoring SOC change over time should take these interactions into consideration. In the case of this study, depth to the Bk horizon (zone of secondary calcium carbonate accumulation) was treated as an indicator variable that reflects other factors which may influence SOC variability across the landscape.

A soil-landscape approach proposed by Dr. Thomas Keck (pers. comm. 2001) was used to refine the selection criteria for paired sampling to potentially reduce soil organic carbon variability. Identifying the same soil type across a field boundary provides a good starting point but soil type alone does not guarantee similar initial conditions with respect to SOC. There are several reasons for this:

1) Soils can often exhibit substantial short range spatial variability in selected soil properties even in apparently uniform environments.

2) Soil types (series) are based on generalized morphological characteristics. Their description as potentially mappable entities requires that ranges be given for

many different soil properties. For example, a typical range given for SOM in the surface horizon of an "aridic intergrade" of Mollisols is 1-3%.

3) Soil organic matter is generally not used as a diagnostic criterium in identifying soil series but rather it is an assigned attribute to the series.

4) Land use and management conditions are not accounted for in series descriptions or classifications, yet they can have a large effect on soil organic matter.

Soil organic carbon varies spatially as a function of short-range and landscape scale variability. The amount of variation in measured SOC values can be substantial with coefficients of variation of 20% reported in a uniformly managed 48-ha site in southwest Michigan (Robertson et al. 1997). Such variability could mask the slow accumulation of SOC over a 6- year period of management change. While the same taxonomic soil type may be accurately mapped on a ridgetop or in a swale, soils in these positions may have very different SOM levels and potentials for SOC change. Inherent differences due to topographic position can be accentuated by plowing in hilly terrain which enhances the transport of surface soil particles downslope, especially SOC (Gregorich et al. 1998). Reducing the influence of soil spatial variability was an important consideration in the research conducted for this thesis.

Monitoring and verification of agricultural soil carbon stocks will be critical to the development of a carbon credit trading system. Whether the system of carbon trading is based on a 'per tonne' of carbon change or based on the areal extent of a management

change, the variability associated with SOC needs to be understood so that modeling

efforts can take this variability into consideration. Assuming there is a definable

relationship between management practices and soil properties on the distribution of SOC

across a field, the accuracy of process-based models such as the Century model could

benefit from understanding C relationships with soil variability. The primary objective of

this study was to characterize soil organic carbon variability in dryland wheat fields for

tilled and no-till crop production systems in north central Montana. Field sampling was

designed to provide repeatable future sampling at exact locations by using 2x5 m sampling

plots, with multiple soil samples per plot, which will be referred to as microsites in this

study. It was hypothesized that variability within microsites would be less than that of the

variability among microsites and that the variability of SOC would be greater in the no-till

system than that of the tilled system. Differences at the management system level were

further hypothesized to be due to the soil homogenizing effect of tillage. Understanding

the variability of SOC sampling could support single-point-in-time measured differences in

soil organic carbon content under no-till and tilled management. The null hypothesis tested

that the variability of soil organic carbon within microsites is equal to the variability of

SOC among microsites in farm fields. A secondary objective of this study was to derive

an optimal sampling design for soil organic carbon when using a soil-landscape method of

site stratification. It was expected that the optimal combination of replications (i.e.

microsites) per field and cores per replication would be less than six microsites per field

with six cores per microsite. We asked two simple questions to determine an optimal

sampling design: 1) What was the minimum number of microsites required to consistently detect known differences in SOC; and, 2) Could the minimum number of microsites be reduced by taking multiple cores per microsite?

Materials and Methods

Site Selection

Two sites (Simpson and Ft. Benton) of a larger six-site data set were used for characterizing soil organic carbon sampling variability. Of the six sites, these were chosen because they differed strongly in clay content. Paired no-till (NT) and conventionally tilled (CT) agricultural fields were chosen to make inferential comparisons of SOC related to field management (Table 2.1, Fig. 2.1). Commonly used tillage implements in the study included tandem disc, chisel plough, cultivator, rod weeder, and hoe drill seeder. No-till in the study area was defined as chemical fallow followed by direct seeding into the previous crop's stubble.

Each paired site was required to meet the following criteria to minimize confounding factors that could mask the effects of tillage system.

1) Dryland fields were both in production in the 2001 growing season with the same type of wheat (i.e. spring or winter), and had similar cropping histories. This minimized inter-annual variation due to cropping type and history.

2) No-till fields had been managed without tillage for a minimum of six years to ensure they were fully representative no-till systems and to permit sufficient time

for SOC to change in respond to the management change.

3) NT/CT pairs were separated by an anthropogenic boundary (e.g. property boundary) rather than a naturally occurring boundary such as an abrupt topographic change. Close proximity minimized variability in climatic and soil properties.

4) Sampling sites were selected to have the same soil type and landscape position under both tillage practices. This increased the probability that soil properties were similar across management boundaries.

Table 2.1. Site descriptions for Simpson and Ft. Benton, MT, 2001.

	Simpson	Ft. Benton
County	Hill	Liberty
Latitude	48° 56' 09" N	48° 08' 19" N
Longitude	110° 13' 10" W	110° 56' 44" W
Slope Steepness	0 - 2 %	0 - 4 %
Land Form	Till plains	Till plains
MAP(cm)†	26	36
MAAT (∘C)‡	5.0	7.5
2001 crop	Spring wheat	Winter wheat
Years in no-till	7	6

† 30-yr mean annual precipitation (1961-1990). Source: Simpson NOAA weather station.
‡ 30-yr mean annual air temperature (1961-1990). Source: Ft. Benton NOAA weather station.

Once potential paired fields were identified, areas were located within fields having a high probability of uniform initial soil conditions with respect to soil organic matter. Site-specific ranges of selected soil properties and landscape attributes were used

as selection criteria for identifying paired samples across the field boundary separating

tillage systems. The combination of specific soil and landscape attributes is referred to

here as a soil-landscape association. Soil-landscape associations in the context of this

study can be considered as localized areas with a high probability of similar initial

conditions with respect to soil organic carbon (Dr. T. Keck, pers. comm., 2002).

Figure 2.1. Map of Montana showing locations of sampled sites. 1) Conrad W,
2) Simpson,3) Saint Johns, 4) Ft. Benton, 5) Chester, 6) Conrad E.

Two steps were required in defining the soil-landscape association at each site.

First, preliminary field sampling identified the specific combination of soil and landscape

attributes to be used as a soil-landscape association at each sample site. The

approximate areal extent of the association was then drawn on a base map. Secondly, a

30-meter grid was established on the map and points of intersection from the grid were randomly selected. An exploratory soil pit was excavated at corresponding point in each field. If the soils at the randomly selected sample point met the specific criteria, then it became a valid sample location. If not, then another point on the grid was randomly selected until six valid sample locations meeting the soil-landscape criteria were located on either side of the field boundary.

Landscape attributes of slope position, shape, steepness and aspect along with the local depositional environment were used to define the landscape component of a soil-landscape association. For the most part, landscape attributes were used to initially map out areas of potentially similar conditions. It is conceivable, however, that an initially selected sample point within the mapped area could be excluded because of landscape irregularities such as a pocket of secondary alluvial deposition within an otherwise intact glacial till landscape. An example landscape description for a specific site might be as follows: south-tending, linear backslopes with 2 to 4% slopes.

Soil attributes used to define the soil-landscape associations were soil texture and depth to accumulated calcium carbonate (Table 2.2). Turnover rate of soil organic carbon is a function of soil texture (Parton et al. 1994), and soil organic carbon content increases with clay content (Burke et al. 1989). The clay fraction of soil has been shown to reduce turnover rates of SOM decomposition through chemical (adsorption) and physical (aggregation) protection, thus increasing SOM (Burke et al. 1989).

Depth to CaCO$_3$ is an expression of the leaching environment and other soil properties related to soil texture and local climate conditions. Narrowing the site-specific range in depth to lime (accumulated calcium carbonate) for each soil-landscape association provided strict control in selecting paired samples with highly similar soil-site conditions. A typical soil component used for comparisons might be as follows: Kobase silty clay loam with clay loam textures throughout and a depth to accumulated calcium carbonate of 20 to 36 cm.

Table 2.2. Soil characteristics for sites at Simpson and Ft. Benton, MT, 2001, sampled for variability analysis (values are site-specific unless otherwise noted)

	Simpson	Ft. Benton
Soil Series	Telstad	Kobase
Soil Classification†	Fine-loamy, mixed, superactive, frigid Aridic Argiustolls	Fine, smectitic, frigid Torrertic Haplustepts
Depth to Lime (cm)	38-56	22-36
Depth to Lime (cm)†	18-50	24-60
Surface clay %	11-17	23-47
Surface clay %†	18-27	35-40
Surface Texture class	sandy loam	clay loam and clay
Surface Texture class†	loam	silty clay loam

† Reported by SSURGO database.

Variability within the soil properties used to define the soil-landscape association could affect SOC content significantly. The specific criteria for the NT/CT pairs minimized confounding effects so that differences in SOC content could be attributed to management practices and site-specific soil properties, rather than environmental gradients.

Soil Organic Carbon Sampling

Twelve microsites (six in each tillage system) randomly positioned within the

defined soil-landscape association were chosen for soil organic carbon sampling at each

site (Fig 2.2). The soil sampling scheme was adapted from the Canadian Prairie Soil

Carbon Balance Project (Ellert et al. 2001). Sample preparation and carbon analysis

procedures were adapted from Conant and Paustian (2002).
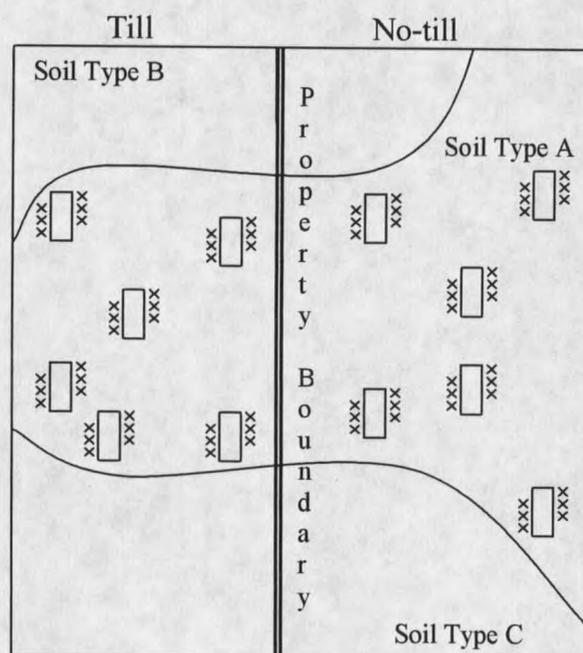
Figure 2.2. Schematic of sampling design. Boxes represent microsites and "x"s represent core configuration.

Each site was intensively sampled for soil organic carbon during the first two

weeks of September, 2001. Both sites were managed as a wheat-fallow system and

sampling occurred after the wheat crop was harvested at each site. A 2x5-m frame was

centered over the exploratory soil pit and directionally oriented so that core position #1

was the northeast corner of the frame. The frame served as a sampling template to ensure

consistent sample spacing at all microsites. Surface soil cores were taken by hand using

an aluminum bulk density ring (7.4-cm diameter x 10 cm deep). The ring provided a

consistent sample in loose, crumbly surface soil. Starting from the northeast corner (2,5),

soil samples were taken at the following positions around the frame: #1 (2,5), #2 (2,3), #3

(2,1), #4 (0,0), #5 (0,2), #6 (0,4) (Fig. 2.3).
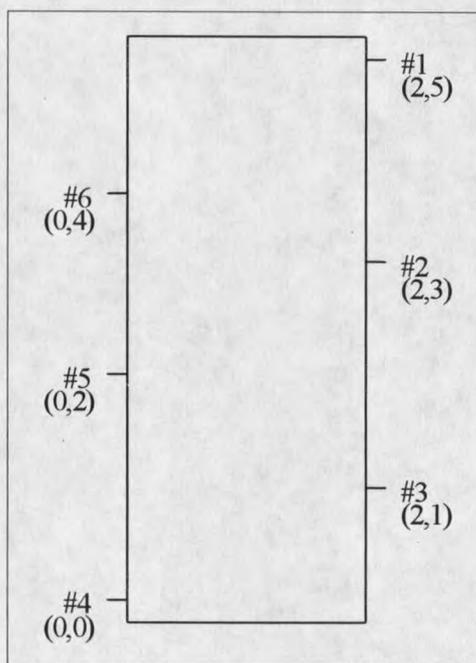
Figure 2.3. Sample microsite map showing locations of core samples (coordinate reference in parenthesis)

Sample Preparation

All samples were oven-dried at 40°C for 4 d, then weighed for bulk density

determination. Bulk density measurements were used to convert carbon concentrations to

an equivalent mass value and as a measure of sampling consistency.

Each sample was passed through a 2-mm sieve to remove rock fragments,

surface plant litter, and coarse root material. Visible surface litter and root material that

passed through the 2-mm sieve was removed by hand. Rock fragments, surface litter and

root material were removed, dried at 70°C for 24 hr and weighed separately to adjust

bulk density values. Surface litter was not analyzed for C content. Bulk soil samples were

sub-sampled (50 g for C analysis, 100 g for texture analysis and 100 g archived)

individually for analysis. Approximately 30 g of the C sub-sample was milled to fine

powder   (< 200 μm) in a ball mill (Pica Blender Mill Model 2601, Cianflone Scientific

Instruments Corp. Pittsburgh, PA) for C analysis.

Sample Analysis

Soil samples were analyzed for total C, inorganic C, total N, texture, and pH.

Total C and total N content of soil samples were measured by combustion using a LECO

C/N/S 2000 analyzer (LECO Corporation, St. Joseph, MI, USA). Total nitrogen was

measured to calculate C:N ratios as a check for credible organic carbon levels. Soil C:N

ratios have been measured in a range of 3:1 to 14:1 with a mean of 10:1 (Robertson et al.

1997). Soil C:N ratios outside that range could signify an inaccurate estimate of soil

inorganic carbon. Over- or underestimating soil inorganic C will result in erroneous

organic carbon values. Inorganic C was determined using a modified pressure calcimeter

method (Sherrod et al. 2002). Soil pH and texture were measured at the Montana State

University Soil Testing Laboratory using standard 1:1 (Eckert 1989) and modified

hydrometer methods (Gee and Bauder 1986), respectively. Using the measured bulk

density of the field samples, all soil organic carbon values were converted to and reported

on an equivalent mass basis according to methods described by Ellert and Bettany

(1995). Soil organic carbon (SOC) was calculated by difference given by the equation:

$$TotC - IC = SOC \quad [Eqn\ 1]$$

Where TotC = total soil carbon, IC = inorganic soil carbon, and SOC = soil organic

carbon.

Statistical Analysis

  Soil organic carbon and soil bulk density data were analyzed using analysis of

variance (ANOVA), analysis of covariance (ANCOVA), and variance component

analyses. ANOVA was run as a combined analysis including both sites. 'Site' and

'Tillage' treatments were considered fixed in the models. Data were analyzed using two

full models; 1) including individual core values as subsamples, and 2) using microsite

values calculated as the mean of the six cores associated with that microsite. Replication

occurs at the microsite level; therefore, microsite was used as the error term to test the

effects of site and tillage.

  Clay content is a driving variable in soil organic matter dynamics and clay

contents varied between and within sites; therefore, ANCOVA was used to determine if

measured differences in soil organic carbon and bulk density were increased by including

the effect of clay content as a covariate. ANCOVA was run as full and reduced (by site)

models using microsite values only because the covariant clay content was determined at the microsite level (microsites and cores were considered random factors). Variance component analysis was used to determine the relative contribution of core and microsite variability to the overall variance in both soil organic carbon and bulk density within tillage treatments. Variance component analysis estimates the magnitude of the variance component associated with the effect of a random factor (Neter et al. 1996). All statistical analyses were preformed using the SAS statistical software 8.02 (SAS, 2001).

The soil sampling optimization procedure consisted of randomly choosing microsites and cores for each microsite x core combination and running analysis of variance to determine the point at which significant differences in the tillage response were lost (P=0.10). For initial screening, 10 iterations were run for each microsite x core combination to remove combinations that did not meet our standards (all 10 iterations significant, P=0.10). The two optimal prospective microsite x core combinations (4x2 and 3x3) were subjected to a more robust set of 100 randomly selected iterations. A P-value of 0.10 was used for testing the significance of terms in all ANOVA and ANCOVA models.

Results

Sampling Variability

The full model ANOVA, including individual core values as subsamples,

determined that the site at Ft. Benton, MT (Site 4) had higher organic carbon content than

the site at Simpson, MT (Site 2) (P <0.01) and no-till fields had greater SOC content

than tilled fields (P <0.01) (Tables 2.3-2.4). Soil bulk density was greater at Simpson (P

<0.01); however, tillage did not have a significant effect on bulk density ($P = 0.62$)

(Tables 2.3 and 2.4). Coefficients of variation (CV) were lower for bulk density

compared to soil organic carbon (Table 4). The CV for SOC in no-till was less than in

tilled fields, while the %CV for BD was similar between tillage systems (Table 2.4).

Table 2.3. ANOVA mean squares for soil organic carbon (SOC) and soil bulk density (BD) in the 0 to 10 cm depth including individual cores as subsamples at Simpson and Ft. Benton, Montana, 2001.

Treatment	df	SOC Mean Squares	$P > F$	BD Mean Squares	$P > F$
				x10	
Site	1	2568.38	<0.01	37.90	<0.01
Tillage(S)	2	155.73	<0.01	0.62	0.62
Microsite(T*S)	20	2.78	<0.01	1.27	0.02
Cores	120	1.27		0.66	

Table 2.4. Soil organic carbon (SOC) content and soil bulk density (BD) values for the 0 to 10-cm soil depth at Simpson and Ft. Benton, Montana, 2001.

Tillage System	Simpson, MT				Ft. Benton, MT			
	SOC		BD		SOC		BD	
	t ha^{-1}	CV†	g cm^{-3}	CV	t ha^{-1}	CV	g cm^{-3}	CV
No-till	8.78*a*	8.3	1.33*a*	5.4	18.15*a*	8.4	1.24*a*	7.8
Tillage	6.92*b*	13.2	1.32*a*	6.2	14.44*b*	10.5	1.21*a*	7.7

† CV is coefficient of variation (standard deviation divided by mean) reported as a percentage.
Means followed by same letter do not differ ($P = 0.1$).

Tillage explained the majority of the variation in SOC after data were adjusted for covariant clay content (Table 2.5). Including clay as a covariate reduced site mean squares for SOC by 40-fold; however, the tillage mean squares were relatively unchanged (Table 2.5). The full model ANCOVA determined that the Ft. Benton site had significantly greater organic carbon than at Simpson ($P < 0.01$), and that no-till fields had significantly greater organic carbon than tilled fields ($P < 0.01$). Soil bulk density did not differ significantly between sites nor between tillage practices ($P = 0.17$ and 0.59, respectively) (Table 2.5). The reduced model ANCOVA (by site) showed that no-till fields at both Simpson and Ft. Benton had greater soil organic carbon than their corresponding tilled fields ($P = 0.10$ and <0.01, respectively) (Table 2.6). Soil bulk density was not affected by tillage at Simpson nor Ft. Benton ($P = 0.57$ and 0.33, respectively) (Table 2.6).

Table 2.5. ANOVA and ANCOVA mean squares for soil organic carbon (SOC) and soil bulk density (BD) at Simpson and Ft. Benton, Montana, 2001.

Treatment	df	SOC		BD	
		Mean Squares	$P > F$	Mean Squares	$P > F$
				x10	
		ANOVA (unadjusted for clay content)			
Site	1	428.06	<0.01	6.32	<0.01
Tillage(S)	2	25.95	<0.01	0.10	0.62
Microsite(T*S)	20	0.46		0.21	
		ANCOVA (adjusted for clay content)			
Site	1	10.62	0.01	0.45	0.17
Tillage(S)	2	23.34	<0.01	0.12	0.59
Microsite(T*S)	20	0.46		0.22	

Table 2.6. ANCOVA mean squares for soil organic carbon (SOC) and soil bulk density (BD) at Simpson and Ft. Benton, MT, 2001.

		Simpson, MT			
		SOC		BD	
	df	Mean Squares	$P > F$	Mean Squares	$P > F$
				x10	
Tillage	1	1.15	0.10	0.07	0.57
Microsite(T)	9	0.34		0.20	
		Ft. Benton, MT			
		SOC		BD	
	df	Mean Squares	$P > F$	Mean Squares	$P > F$
				x10	
Tillage	1	25.85	0.01	0.27	0.33
Microsite(T)	9	0.60		0.26	

Sampling Design Optimization

Variance component analysis of soil organic carbon and bulk density showed that the majority of the total variance occurred within microsites rather than between

microsites with the exception of SOC values in the tilled field at Simpson (Table 2.7).

This was especially evident for the no-till fields.

Table 2.7. Variance component estimations for soil organic carbon (SOC) and soil bulk density (BD) within sites at Simpson and Ft. Benton, Montana, 2001.

| | Simpson, MT | | | | Ft. Benton, MT | | | |
| | No-till | | Tillage | | No-till | | Tillage | |
	SOC	BD	SOC	BD	SOC	BD	SOC	BD
		x 100		x 100		x 100		x 100
Microsites	0.04	0.16	0.55	0.04	0.0	0.0	0.75	0.21
	(7)	(30)	(61)	(5)	(0)	(0)	(31)	(23)
Cores	0.50	0.38	0.33	0.65	3.1	0.92	1.7	0.69
	(93)	(70)	(39)	(95)	(100)	(100)	(69)	(77)

Note: Percentage of the total variance in parentheses.

Detection of a tillage effect on soil organic carbon became less frequent as random selections of microsites and cores were removed from the data set (Fig. 2.4). When only a single core per microsite was used to measure tillage-induced differences in soil organic carbon, six microsites per field was insufficient for both sites. However, if two cores per microsite were taken, four microsites were sufficient to consistently detect tillage-induced differences at both sites (Fig. 2.4). If three cores were taken, then three microsites were sufficient (Fig. 2.4). We reasoned that farmers and researchers would want to take as few total cores as possible to reduce costs associated with determining management-related changes in carbon, and so subjected the 4x2 and 3x3 combinations to more rigorous testing to see which was more consistent. We adopted a conservative level of significance for microsite x core combinations whereby all iterations had to be

significant ($P<0.10$) to meet our standard. Further testing showed that eight cores in a 4x2 configuration detected a tillage effect in 200 of 200 iterations (100 iterations per site) whereas nine cores in a 3x3 configuration detected a tillage effect in 199 of 200 iterations (100 iterations per site) (Table 2.8). The 3x3 combination had one non-significant iteration for the Simpson site ($P = 0.113$). Mean SOC, SE, CV and average P values were very similar for these two sampling combinations (Table 2.8). Mean implied change in carbon (ΔC), calculated as no-till C - tilled C, was similar within sites. The 3x3 combinations (2.3 and 3.5 t C ha^{-1}) tended to be slightly higher than the 4x2 combinations (2.0 and 3.3 t C ha^{-1}) for Simpson and Ft. Benton, respectively (Table 2.8). The range in 90 percent of ΔC values were noticeably lower for the 3x3 combinations, 1.2 and 2.2 t C ha^{-1} (Simpson and Ft. Benton, respectively) and the 4x2 combinations, 1.2 and 2.3 t C ha^{-1} (Simpson and Ft. Benton, respectively) suggesting the existence of some potential outlier values (Table 2.8). Sampling in a 4x2 or 3x3 design did not appear to change the distribution of implied C change (Fig. 2.5-2.6).

Simpson

Ft. Benton

🔲 Significant ■ Not Significant
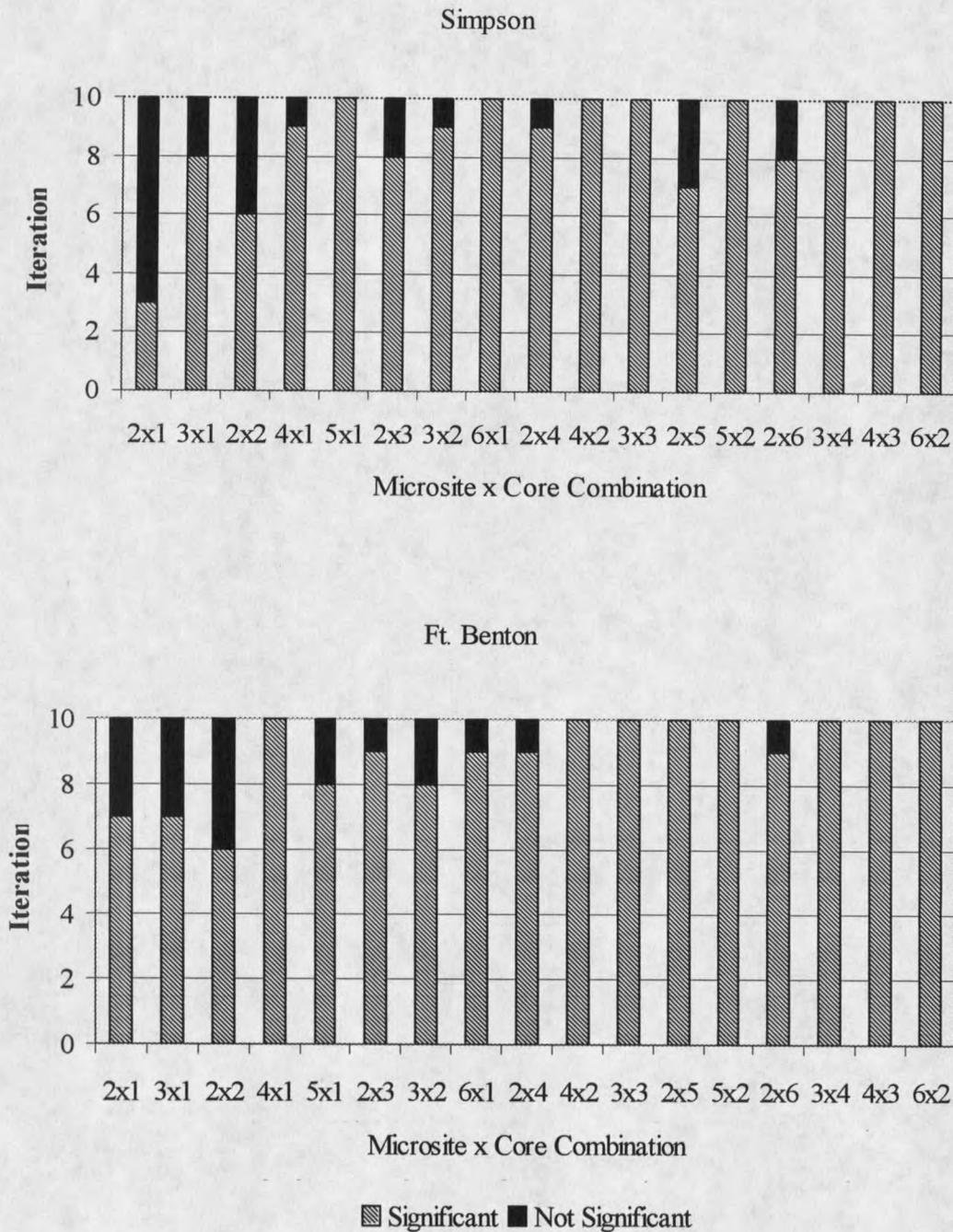
Figure 2.4. Significant vs. non-significant iterations for microsite x core combination ($P<0.10$) at Simpson and Ft. Benton, Montana, 2001. All combinations totaling more than twelve cores detected a tillage effect ($P<0.10$). Carbon change was 1.3 and 5.1 t C ha^{-1} at Simpson and Ft. Benton, respectively.

Table 2.8. Summary statistics of 100 ANOVA iterations of 4 x 2 and 3 x 3 microsite by core combinations in no-till (NT) and tilled (T) fields at Simpson and Ft. Benton, MT, 2001.

Simpson, MT

| | 4x2 | | | | | | | 3x3 | | | | | | |
| | SOC (t ha^{-1}) | | | | CV‡ | | | SOC (t ha^{-1}) | | | | CV | | |
	NT	T	ΔC†	SE	NT	T	P	NT	T	ΔC	SE	NT	T	P
Min	8.7	6.6	0.9 (1.4)	0.2	3.3	4.9	<0.01	8.7	6.6	0.9 (1.3)	0.10	3.1	5.4	<0.01
Mean	9.2	7.2	1.9	0.4	7.7	12.6	0.01	9.2	7.2	1.9	0.4	7.8	12.1	0.1
Max	9.7	7.9	2.9 (2.6)	0.5	12.7	17.3	0.05	9.8	7.9	3.2 (2.5)	0.6	12.6	17.8	0.1
Significant Iterations					100/100							99/100		

Ft. Benton, MT

| | 4x2 | | | | | | | 3x3 | | | | | | |
| | SOC (t ha^{-1}) | | | | CV | | | SOC (t ha^{-1}) | | | | CV | | |
	NT	T	ΔC	SE	NT	T	P	NT	T	ΔC	SE	NT	T	P
Min	17.1	13.1	2.1 (2.6)	0.2	3.3	4.5	<0.01	17.4	13.1	2.2 (2.6)	0.2	3.3	4.3	<0.01
Mean	18.1	14.4	3.7	0.5	7.5	10.4	<0.01	19.4	14.4	3.8	0.5	7.8	10.1	<0.01
Max	19.4	15.6	5.4 (4.9)	0.9	13.9	15.7	0.07	18.2	15.6	5.7 (4.8)	0.9	12.8	15.2	0.1
Significant Iterations					100/100							100/100		

† ΔC is implied change in soil organic carbon calculated as no-till C - tilled C for the entire dataset and do not necessarily correspond to reported minimum and maximum SOC values. Values in parenthesis correspond to minimum and maximum values for the center 90% of the data.

‡ CV is coefficient of variation reported as a percentage.

Simpson 4x2
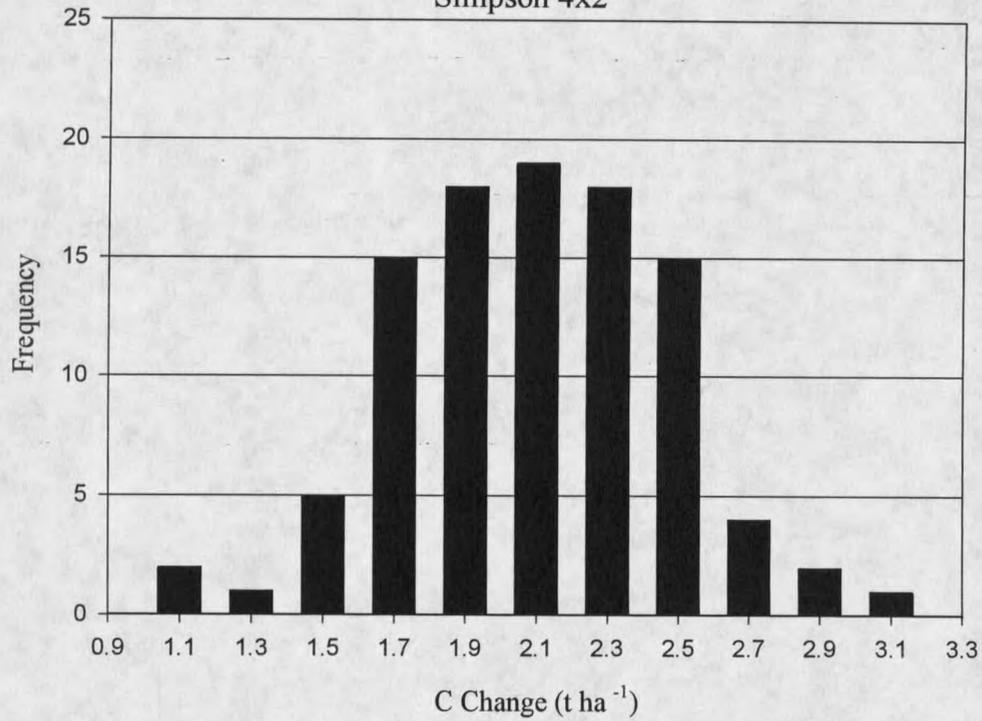
Simpson, MT 3x3
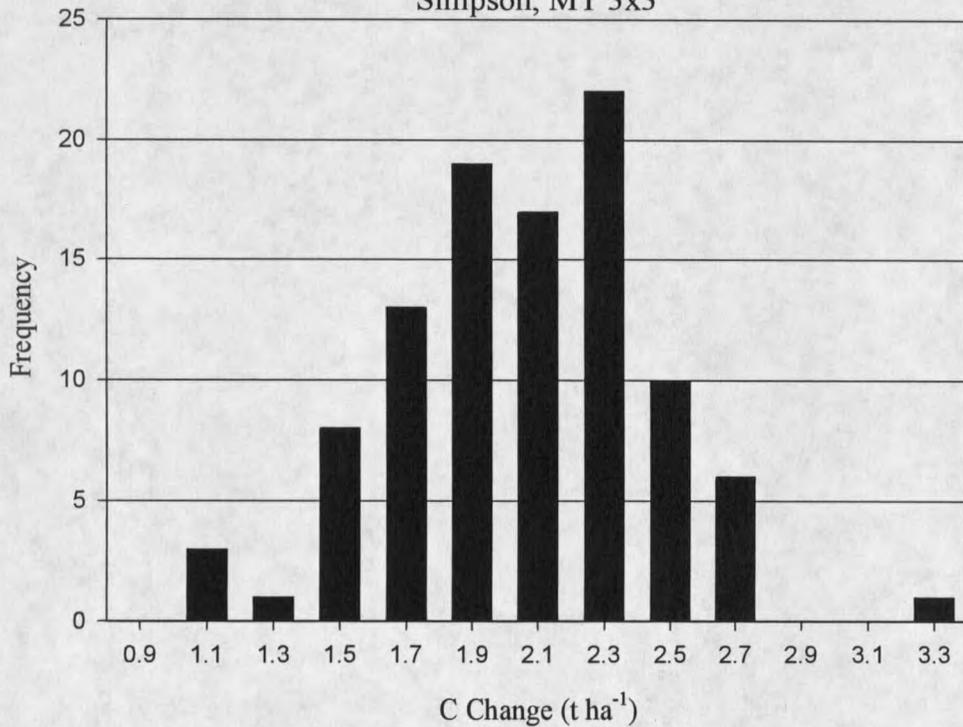
Figure 2.5. Distribution of implied C change (calculated as no-till C - tillage C) for 100 iterations of 4 microsites per field x 2 cores per microsite and 3 microsites per field x 3 cores per microsite at Simpson, MT, 2001.

Ft. Benton, MT 4x2

Frequency

C Change (t ha^{-1})

Ft. Benton, MT 3x3

Frequency

C Change (t ha^{-1})
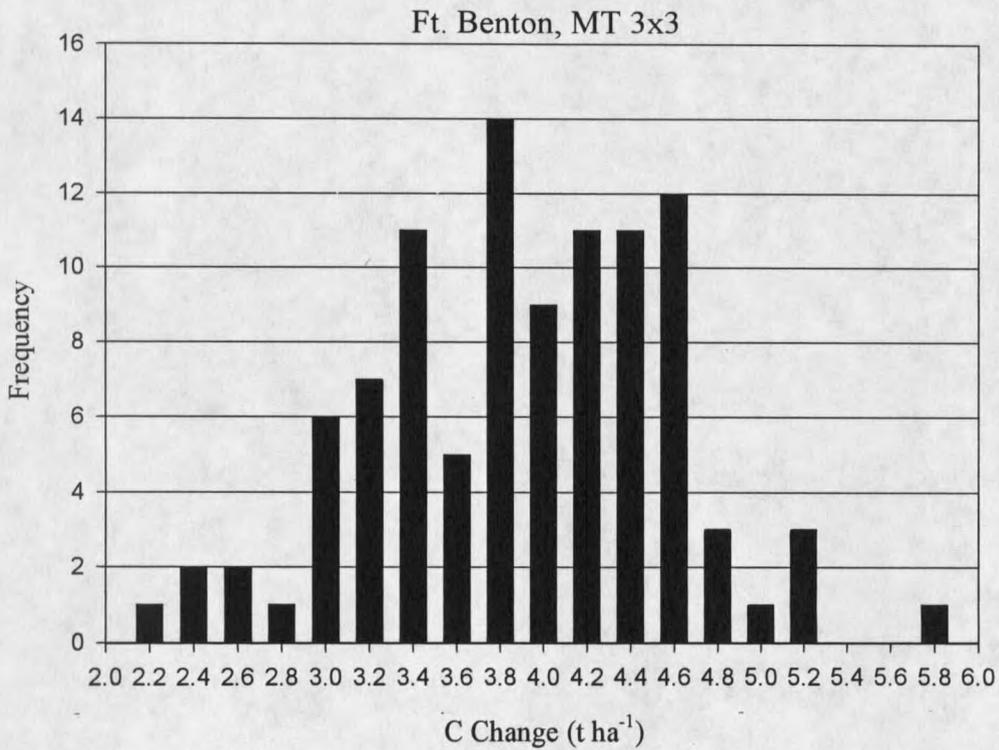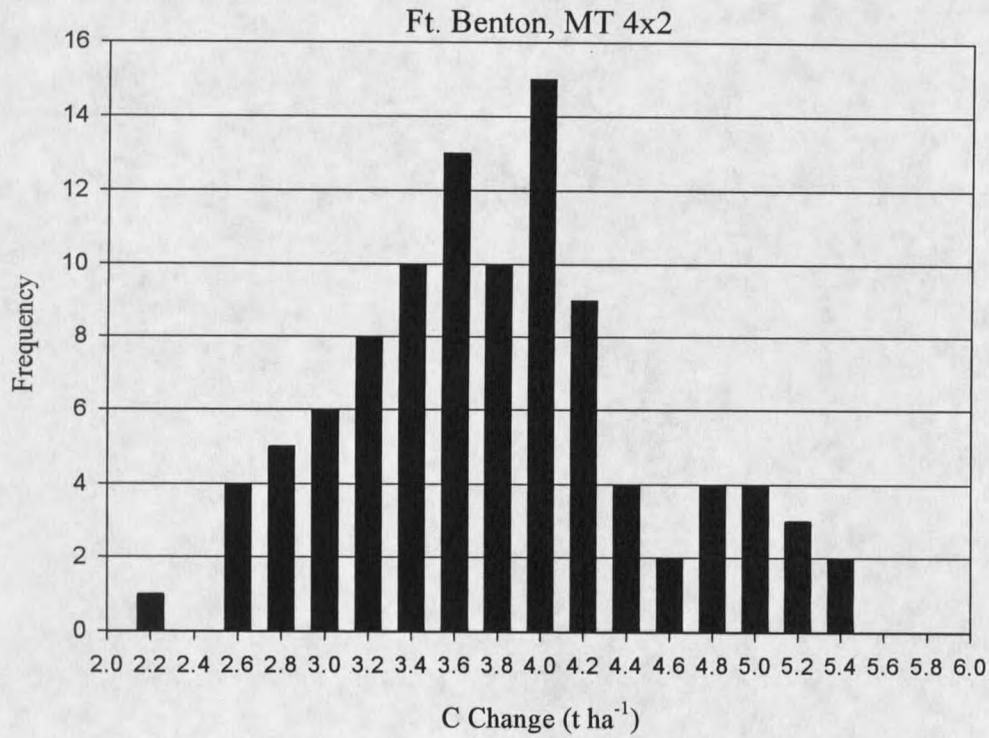
Figure 2.6. Distribution of implied C change (calculated as no-till C - tillage C) for 100 iterations of 4 microsites per field x 2 cores per microsite and 3 microsites per field x 3 cores per microsite at Ft. Benton, MT, 2001.

Discussion

Sampling Variability

The results of this study show that soil organic carbon is inherently variable related to inherent natural variability in soils and to the variability associated with soil sampling techniques. Bulk density did not differ due to tillage effect at either site, however differences occurred among microsites, which suggests that soil properties such as bulk density vary within soil-landscape associations independent of management practices. The lack of a significant tillage effect between tillage systems is consistent with results of Bauer and Black (1981) where stubble mulch and moldboard plow tillage systems were compared after 25yr of cultivation. Coefficients of variation for bulk density were 7 to 53% less than soil organic carbon, lower than the 61% difference found by Robertson et al. (1997) in a uniformly managed field with seemingly homogenous soil properties, confirming that soil organic carbon was more variable across a landscape compared to soil bulk density.

Based on variance component analysis, much of the variability occurred within microsites. The lower variability between microsites may be a function of the fine scale variability of SOC, the variability associated the sample collection methods, or a combination of both. It is expected that with a larger sample volume (i.e. cores bulked to microsite level vs. individual cores) that the variability in measurement values would decrease based on volume-variance relationships described by Wagenet (1985). Most of the variability in SOC occurred within microsites (core level variance component

analysis), yet differences among microsites existed, underscoring the notion that SOC can vary across a field even though great care was taken to sample similar soil-landscape associations and collect consistent samples. Stratifying by soil-landscape association likely reduced SOC variability in this study compared with previous studies. Soil organic carbon CV's ranged from 8 to 13%, lower than the 20% reported by Robertson et al. (1997), and the 12 to 19% reported by Conant and Paustian (2002). Though SOC appeared to be more variable within microsites, the expression of that variability was at the microsite level (shown in the sampling design optimization).

Accounting for differences in clay content may be an important consideration when determining the significance of management practices on SOC accumulation. The covariance analysis resulted in tillage variance remaining relatively unchanged while the variance associated with site was reduced 40-fold. Thus, determination of clay content may be critical when comparing SOC accumulation due to land management among sites differing in clay content. The importance of using a covariance analysis may become more important when making comparisons across larger areas (i.e. regional comparisons), because small differences in clay content between neighboring fields may be confounded with the interactions of translocation processes, soil movement, and tillage (B. McConkey, pers. comm. 2003). For example, an accumulation layer of translocated clay commonly occurs in the soils of this region. Soil movement through wind and water erosion has the potential to remove significant amounts of the soil surface, thus reducing the effective soil depth to the accumulated clay layer, particularly in a tillage based system.

As this process continues portions of the accumulated clay layer could be incorporated into the surface soil or exposed altogether, thus resulting in fine-scale differences in clay content within a field.

Sampling Design Optimization

The number of samples taken for soil organic carbon measurements has a direct influence on the precision of measurable SOC differences and the cost associated with acquiring and analyzing those samples. Our results concur that the sampling design is an important consideration. Concurrent with our hypothesis, results suggest that less than 36 cores in a 6 microsite x 6 core configuration will reliably detect a measurable tillage effect on SOC after 6 yr, given our initial soil and management conditions. The majority of variability associated with sampling procedures occurred within microsites (core level) for both soil bulk density and SOC, supporting that the overall number of cores per field is important to achieve reliable SOC measurements.

In this study, replication occurred at the microsite level, therefore microsites would be considered samples. It is possible to estimate the number of samples needed to detect a given magnitude of difference using an estimate of the population variance (i.e. sample variance) and including a t correction for that substitution (Steele and Torrie 1980). As an example, to detect a 1.0 t C ha^{-1} change due to the adoption of no-till for 10 yr, based on C sequestration rates of 0.06 to 0.12 t C ha^{-1} yr^{-1} found in the Brown and Dark Brown Chernozem soil groups of Canada (Smith et al. 2001), 6 and 9 samples (i.e.

microsites) per tillage treatment would have been required at Simpson and Ft. Benton, respectively, given the level of variability found at each respective site. However, reliable detection of the actual tillage effect (1.3 and 5.1 t C ha^{-1} at Simpson and Ft. Benton, respectively) was maintained when eight or nine cores (aggregated into 4 or 3 microsites, respectively) were used to sample similar soil-landscape associations. Based on results of the sampling optimization, a minimum of eight cores in a 4 microsite x 2 core or nine cores in a 3 microsite x 3 core configuration would be required to reliably detect a tillage effect on soil organic carbon differences after 6 yr within similar soil-landscape associations at Simpson and Ft. Benton. Either configuration could be used to efficiently measure C differences in fields of differing management practices and to monitor C change over time at benchmark sites.

Decisions as to the number of samples taken in an experiment are commonly based on time and resource allocations. As the number of microsites increases per field, the statistical power for determining differences increases (e.g. 3 x 3, n=3; 4 x 2, n=4), however the costs associated with sample analysis and georeferencing and installation of permanent sampling locations for repeated future sampling, also increase. Increasing the number of cores at each microsite provides a better pooled estimate of soil organic carbon at each microsite, however the time and labor cost also increases with additional cores. The cost associated with sample analysis would not increase by taking additional cores per microsite, because in an operational benchmark design, soil from all cores at a microsite could be bulked by depth in the field.

Conclusions

Four microsites with two cores per microsite or three microsites with three cores

per microsite were optimal sampling designs for measuring soil organic carbon differences

(6 to 7 yr) in fields managed with and without tillage at two sites in north central Montana.

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SENSITIVITY OF THE CENTURY MODEL TO SCALE-RELATED SOIL TAXONOMIC VARIATION IN SOIL TEXTURE

Abstract

To facilitate carbon credit trading, farmers must be able to verify changes in soil organic carbon (SOC) related to a change in management practice. It is likely that the use of predictive models will meet this criterion. It is uncertain if existing soil databases are sufficiently accurate to support a predictive model in providing field-specific measurement of SOC change. The objective of this study was to assess the sensitivity of Century model predictions of SOC change, resulting from farm management change, using existing soil databases. This research explored the influence of soil texture on the effect of tillage-induced changes in SOC storage, and the sensitivity of the Century model to the effects of soil texture input values from three resolutions. Six farm sites in north central Montana used to validate model predictions. Sites were paired tillage/no-till comparisons where historical cropping systems, and soil, landscape, and climatic conditions, were similar within each pair. The Century model accurately predicted SOC content using site-specific soils data (10% deviation from measured values). Neither the STATSGO nor SSURGO databases predicted soil textures adequately for use with Century. Century proved to be sensitive to the effects of clay content when predicting the amount of SOC in a particular field; however, the model was insensitive to the effects of soil texture on C sequestration as a result of land conversion to no-till management.

Introduction

Previous studies have shown that the Century model is a valid model for estimating soil organic carbon dynamics under various soil, climatic, and agricultural practices (Parton et al. 1987; Paustian et al. 1992; Monreal et al. 1997). Parton et al. (1987) found that the model adequately estimated soil C values in the Great Plains representing various soil textures and climates ($r^2 = 0.88$, 0.92, and 0.92 for sandy, medium, and fine textured soils, respectively). Century tended to overestimate C for fine textures and underestimate C for sandy soils (Parton et al. 1987). In long-term plots in Sweden the model predicted SOC values within 5 to 15% of measured values as a function of temperature and initial composition of soil organic matter (Paustian et al. 1992). Century has also been validated in eastern Canada where the model estimated SOC to within 10% of measured values for 4 crop rotations in 3 soil climosequences (Monreal et al. 1997). There is growing interest in using this model as a tool for estimating carbon dynamics in situations where a limited amount of site-specific information is known for a particular field or farm (Zimmerman et al. 2003).

The Century model is a process-based model that estimates soil organic carbon changes based on macroenvironmental gradients, management, and soil and plant properties over long periods (>30 yrs) (Parton et al. 1987). The model is initiated using specific variables describing the soils, climate, and management of a particular system. Century simulates soil organic matter (SOM) formation and calculates decomposition rates based on first-order kinetics that vary as a function of soil temperature and moisture,

soil texture, and other variables. The model uses a monthly time step and includes both carbon and nitrogen flows. Sensitivity of Century model predictions to differences in soil texture associated with soil taxonomic variation could have significant implications for the use of soils input data from various sources.

Studies have evaluated various aspects of the Century model for modeling SOM dynamics and validated the model for use in many environments, soil types, and management systems. Carbon change due to cultivation and agricultural management have been modeled using the Century model in the Argentine Pampa (Alvarez et al. 2001), Canadian prairies (Smith et al. 2001; Monreal et al. 1997; Campbell et al. 2001b; Smith et al. 1997), the U.S. Great Plains (Parton et al. 1987, 1994; Parton and Rasmussen 1994), Swedish agroecosystems (Paustian et al. 1992), and European systems (Falloon and Smith 2002) among others. The sensitivity of Century to some of the major input parameters was found to be highly variable and dependent upon site specific model run conditions. Century output values appeared to be most sensitive to temperature and native carbon inputs, where a 20% change in temperature and native carbon input values changed soil carbon content by 50 kg C ha^{-1} yr^{-1} in three major agricultural soil groups of Canada (Smith et al. 1997). Soil input variables from various sources and scales could have a similar effect.

Soils information can be obtained at various scales of spatial and soil-attribute resolutions. Existing soil databases such as the State Soil Geographic (STATSGO) database, and the Soil Survey Geographic (SSURGO) database are generalizations of

soil patterns across the landscape. Soil mapping units in both the STATSGO and SSURGO databases may contain more than one defined soil series, each with its own soil attributes. Soil properties for each soil series are reported as a range in values, therefore choosing a representative value for use in the Century model could be very important for accurate soil carbon estimations.

A soil-landscape approach proposed by Dr. Thomas Keck (pers. comm. 2001) was used to refine the selection criteria for paired sampling to potentially reduce soil organic carbon variability. Identifying the same soil type across a field boundary provides a good starting point but soil type alone does not guarantee similar initial conditions with respect to SOC. There are several reasons for this:

1) Soils can often exhibit substantial short range spatial variability in selected soil properties even in apparently uniform environments.

2) Soil types (series) are based on generalized morphological characteristics. Their description as potentially mappable entities requires that ranges be given for many different soil properties. For example, a typical range given for SOM in the surface horizon of an "aridic intergrade" of Mollisols is 1-3%.

3) Soil organic matter is generally not used as a diagnostic criterium in identifying soil series but rather it is an assigned attribute to the series.

4) Land use and management conditions are not accounted for in series descriptions or classifications, yet they can have a large effect on SOM.

Similar to other soil properties, soil organic carbon varies spatially. The amount

of variation in measured SOC values can be substantial with coefficients of variation of 20% reported in a uniformly managed 48-ha site in southwest Michigan (Robertson et al. 1997). Such variability could mask the slow accumulation of SOC over a 6-yr period of management change. The same taxonomic soil type may be accurately mapped on a ridgetop or in a swale but may have very different SOM levels and potentials for SOC change. Inherent differences due to topographic position are accentuated by plowing in hilly terrain which enhances the transport of surface soil particles downslope, especially SOC (Gregorich et al. 1998). Thus, reducing the influence of soil spatial variability was an important consideration for making the pair-wise comparisons of no-till and tillage based management.

STATSGO and SSURGO databases, although found to be highly correlated (R^2 = 0.98), have the potential for differences in soil attributes (Juracek and Wolock 2002). The degree of discrepancy between soil attributes at 1:250,000 scale (STATSGO), 1:24,000 scale (SSURGO), and site-specific differences has been found to be potentially substantial. For example, general soil attributes including clay percent, soil permeability, and hydrologic group were spatially and statistically analyzed for differences (Juracek and Wolock 2002). For small averaging areas (0.01 km^2) in Kansas, differences in reported clay percent ranged from -45 to 58% with a standard deviation of 7. As the averaging area increased from 25 to 400 km^2, the range in difference was reduced (-11 to 4%) and the standard deviation decreased to 3 and 2, respectively. The correlation between the databases also varied with landscape position with the most variability falling within stream

networks. Mean clay percent values stabilized with distance from stream networks with mean differences in the +/- 1% range. Although the agreement in the databases increased with the size of the averaging area and with distance away from stream bottoms, the site-specific ranges are large and could result in substantial differences in model predicted SOC values for a specific farm field. The objective of this study was to assess the sensitivity of Century model predictions of soil organic carbon change, resulting from farm management change, to variation in soil texture associated with using existing soil databases.

Methods

Site Selection

Paired no-till (NT) and conventionally tilled (CT) agricultural fields were chosen at six sites in north central Montana to make inferential comparisons of SOC related to field management (Fig. 3.1, Table 3.1). Commonly used tillage implements in the study included a tandem disc, chisel plough, cultivator, rod weeder, or hoe drill seeder. No-till in the study area was defined as chemical fallow followed by direct seeding into the previous crop's stubble. Soil organic carbon was sampled for validation of Century model predictions.
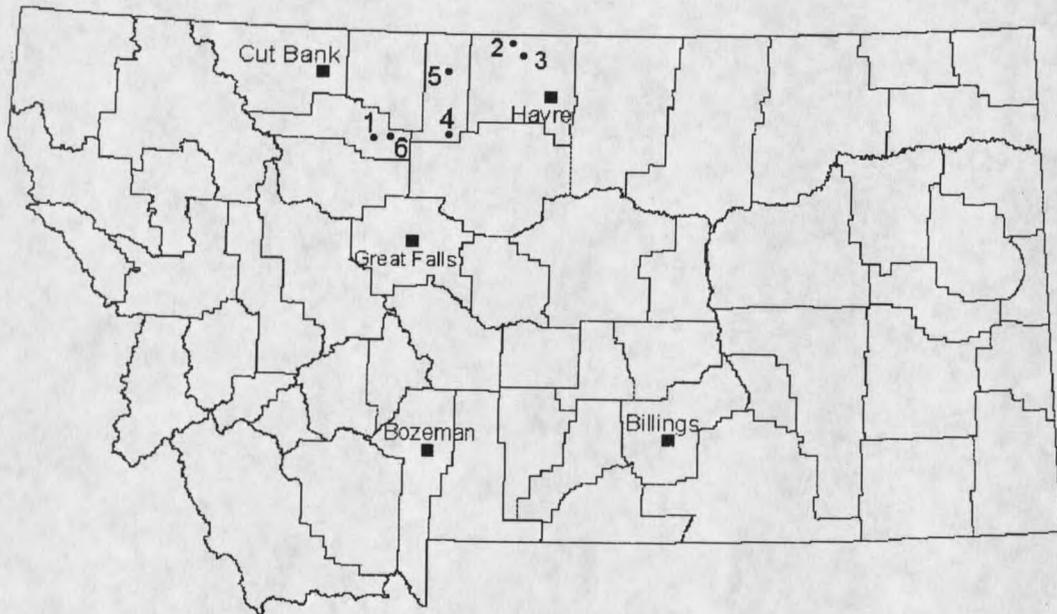
Figure 3.1. Map of Montana showing locations of sampled sites. 1) Conrad W,
2) Simpson,3) Saint Johns, 4) Ft. Benton, 5) Chester, 6) Conrad E.

Each paired site was required to meet the following initial criteria to minimize

confounding factors that could mask the effects of a change in tillage system.

1) Sampling sites were selected to have the same soil type and landscape position

under both tillage practices. This provided an initial assessment that soil properties

and erosion potential were similar across management boundaries.

2) No-till fields had been managed without tillage for a minimum of six years to

ensure they were fully representative no-till systems and to permit sufficient time

for SOC to respond to the management change.

3) NT/CT pairs were separated by an anthropogenic boundary (e.g. property

boundary) rather than a naturally occurring boundary such as a waterway or an

Table 3.1. Site characteristics for Chester, Conrad East (E), Conrad West (W), Ft. Benton, Saint Johns, and Simpson, Montana, 2001. Values are site-specific unless otherwise noted.

	Chester	Conrad E	Conrad W	Ft. Benton	Saint Johns	Simpson
County	Liberty	Pondera	Pondera	Liberty	Hill	Hill
Latitude	48°42'04"N	48° 10' 07"N	48°10'32"N	48° 08' 19" N	48°50'32"N	48° 56' 09" N
Longitude	110°58'46"W	111° 38' 11"W	111°41'13"W	110° 56' 44" W	110°04' 57"W	110° 13' 10" W
Elevation (m)	1038	1040	1037	960	842	847
Years in NT	9	9	10	6	8	7
2001 crop	Spring Wheat	Winter Wheat	Spring Wheat	Winter Wheat	Spring Wheat	Spring Wheat
Crop Rotations	Cont W (no-till) W-F (till)	W-F	W-F	W-F	W-F	W-F
Tillage implements†	cult, rw	cult, sweeps, rw	cult, rw, harrow	cp, sweeps, rw	cult, cp, rw	cult, rw
Slope Steepness	0 - 3%	0 - 2%	0 - 2%	0 - 4 %	0 - 1%	0 - 2 %
Land Form	Till plains	Till plains	Till plains	Till plains	Till plains	Till plains
MAP (cm)‡	27	31	31	36	26	26
MAAT (°C)§	5.4	6.0	6.0	7.5	5.0	5.0

† cult = cultivator, rw = rod weeded, cp = chisel plow
‡ Mean Annual Precipitation estimated from the weather station nearest the site.
§ Mean Annual Air Temperature estimated from the weather station nearest the site.
Source: National Climate Data Center (NCDC) Historical listing of National Weather Service Cooperative Network.

abrupt topographic change. Close proximity minimized variability in climatic and soil properties.

4) Dryland fields produced (in the 2001 growing season) the same type of wheat (i.e. spring or winter), and had similar cropping histories. This minimized inter-annual variation due to cropping type and history.

Once potential paired fields were identified, areas were located within fields having a high probability of uniform initial soil conditions with respect to soil organic matter. Site-specific ranges of selected soil properties and landscape attributes were used as selection criteria for identifying paired samples across the field boundary separating tillage systems. The combination of specific soil and landscape attributes is referred to here as a soil-landscape association. Soil-landscape associations in the context of this study can be considered as localized areas with a high probability of similar initial conditions with respect to soil organic carbon (Dr. T. Keck, pers. comm., 2002).

Two steps were required in defining the soil-landscape association at each site. First, preliminary field sampling identified the specific combination of soil and landscape attributes to be used as a soil-landscape association at each sample site. The approximate areal extent of the association was then drawn on a base map. Secondly, a 30-m grid was established on the map and points of intersection from the grid were randomly selected. An exploratory soil pit was excavated at each corresponding point in the field. If soil and landscape attributes at the randomly selected sample point met the specific criteria, then it became a valid sample location. If not, then another point on the

grid was randomly selected until six valid sample locations meeting the soil-landscape

criteria were located on either side of the field boundary.

Landscape attributes of slope position, shape, steepness and aspect along with

the local depositional environment were used to define the landscape component of a soil-

landscape association. Landscape attributes were used initially to map out areas of

potentially similar conditions. It is conceivable, however, that an initially selected sample

point within the mapped area could be excluded because of landscape irregularities such

as a pocket of secondary alluvial deposition within an otherwise intact glacial till

landscape. An example landscape description for a specific site might be as follows:

south-tending, linear backslopes with 2 to 4% slopes.

Soil attributes used to define the soil-landscape associations were soil texture and

depth to accumulated calcium carbonate (lime) (Table 3.2). Turnover rate of soil organic

carbon is a function of soil texture (Parton et al. 1994), and soil organic carbon content

tends to increase with clay content (Burke et al. 1989). The clay fraction of soil has been

shown to reduce turnover rates of SOM decomposition through chemical (adsorption)

and physical (aggregation) protection, thus increasing SOM (Sorenson 1981; Tiessen et al

1982; Schimel et al. 1985).

Depth to lime is an expression of the leaching environment and other soil

properties related to soil texture and local climate conditions. Narrowing the site-specific

Table 3.2. Soil characteristics for Chester, Conrad East (E), Conrad West (W), Ft. Benton, Saint Johns, and Simpson, Montana, 2001. Values are for 0-20 cm depth and site-specific unless otherwise noted.

	Chester		Conrad E		Conrad W		Ft. Benton		Saint Johns		Simpson	
	No-till	Tilled	No-till	Tilled	No-till	Tilled	No-till	Tilled	No-till	Tilled	No-till	Tilled
Soil Series	Hillon		Kevin		Etheridge		Kobase		Telstad		Telstad	
Soil Classification†	Fine-loamy, mixed (calcareous), frigid Aridic Ustorthents.		Fine, smectitic, frigid Torrertic Argiustolls		Fine-loamy, mixed, frigid Aridic Argiustolls		Fine, smectitic, frigid Torrertic Haplustepts		Fine-loamy, mixed, frigid Aridic Argiustolls		Fine-loamy,mixed, superactive, frigid Aridic Argiustolls	
Texture Class [a]	cl		c		cl		c,cl		l		sl,l	
Texture Class†	l		cl		sicl		sicl		l		l	
Texture Class‡	fsl, l, sil, cl		sil, sicl, sic, cl, c		fsl, l, sicl, cl		fsl, l, cl		fsl, l, cl		fsl, l, cl	
Clay(%)	38-39		42-43		39-42		45-47		23-26		16-19	
Clay (%)†	5-35		18 -60		5-40		5-35		5-40		5-40	
Clay (%)‡	18-35		27-35		27-35		30-35		18-27		18-27	
Bulk Density (g/cm³)	1.27-1.28		1.32-1.36		1.36-1.40		1.29		1.46-1.47		1.45-1.51	
Bulk Density (g/cm³)†	1.1-1.45		1.1-1.45		1.15-1.40		1.15-1.45		1.15-1.45		1.15-1.45	
Bulk Density (g/cm³)‡	1.20-1.40		1.15-1.35		1.15-1.35		1.20-1.40		1.20-1.40		1.20-1.40	

[a] c = clay, cl = clay loam, fsl = fine sandy loam, l = loam, sic = silty clay, sicl = silty clay loam, sil = silt loam, sl = sandy loam
† Reported by STATSGO database.
‡ Reported by SSURGO database.

range in depth to lime for each soil-landscape association provided greater control in selecting paired samples with highly similar soil-site conditions. A typical soil component used for comparisons might be as follows: Kobase silty clay loam with clay loam textures throughout and a depth to lime of 20 to 36 cm.

Variability in the soil properties used to define the soil-landscape association could affect SOC content. The specific criteria for the NT/CT pairs minimized confounding effects so that differences in SOC content could be attributed to management practices and site-specific soil properties, rather than environmental gradients.

Soil Organic Carbon Sampling

Twelve microsites (six in both the tilled and no-till fields) randomly positioned within the defined soil-landscape association were chosen for soil organic carbon sampling at each site (Fig 3.2). The soil sampling scheme was adapted from the Canadian Prairie Soil Carbon Balance Project (Ellert et al. 2001), and sample preparation and C analysis procedures were adapted from Conant and Paustian (2002).
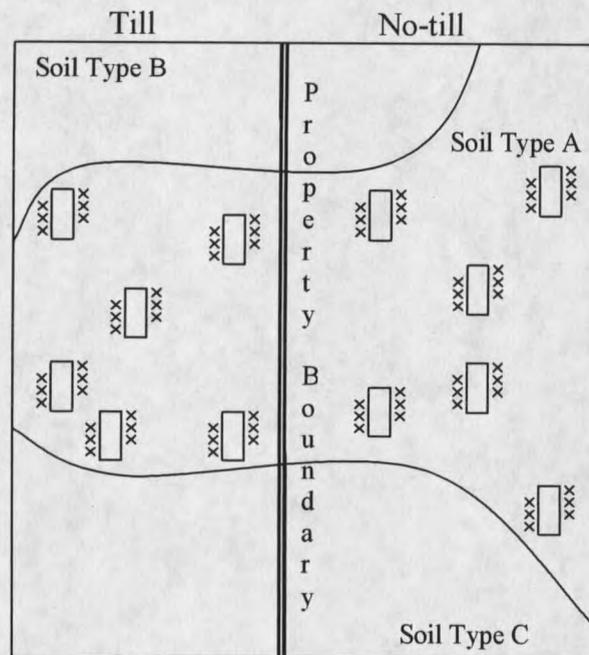
Figure 3.2. Schematic of sampling design. Boxes represent microsites and "x"s represent core configuration.

Each site was sampled for soil organic carbon during the first three weeks of September, 2001, after the wheat crop in the wheat/fallow rotation was harvested. A 2- x 5-m frame was centered over the exploratory soil pit and directionally oriented so that core position #1 was the northeast corner of the frame. The frame served as a sampling template to ensure consistent sample spacing at all microsites. Surface soil cores were taken by hand using an aluminum bulk density ring (7.4-cm diameter x 10 cm deep). The ring provided a consistent sample in loose, crumbly surface soil. Starting from the northeast corner (2,5), soil samples were taken at the following positions around the frame: #1 (2,5), #2 (2,3), #3 (2,1), #4 (0,0), #5 (0,2), #6 (0,4) (Fig. 3.3). Subsoil samples were taken in the exact positions as the surface soil samples using a Giddings

hydraulic press with a 5.4-cm dia. core. Subsoil samples were separated into 10-20, 20-

50, and 50-100 cm increments.
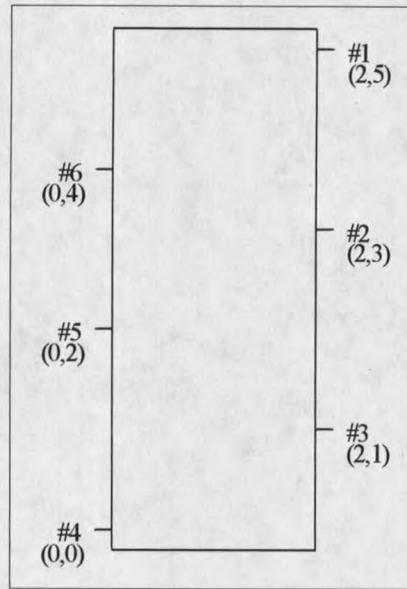
Figure 3.3. Sample microsite showing locations of cores samples (coordinate reference in parenthesis).

Sample Preparation

All samples were oven-dried at 40°C for 4 d, then weighed for bulk density

determination. Bulk density measurements were used to convert carbon concentrations to

an equivalent mass value and as a measure of sampling consistency.

Each sample was passed through a 2-mm sieve to remove rock fragments,

surface plant litter, and coarse root material. Visible litter and root material that passed

through the 2-mm sieve was removed by hand. Rock fragments, litter and root material

were dried at 70°C for 24 hr and weighed separately to adjust soil bulk density values.

Approximately 50 g of each bulk sample was composited by depth for each frame.

Composite samples were sub-sampled for analysis with 50 g for C analysis, 100 g for

texture analysis (0 to 10 and 10 to 20 cm depths only) and 100 g archived.

Approximately 30 g of the C sub-sample was milled to fine powder (< 200 μm) in a ball

mill (Pica Blender Mill Model 2601, Cianflone Scientific Instruments Corp. Pittsburgh,

PA) for C analysis.

Sample Analysis

Samples were analyzed for total C, inorganic C, total N, texture, and pH. Total

C and total N content of soil samples were measured by combustion using a LECO

C/N/S 2000 analyzer (LECO Corporation, St. Joseph, MI, USA). Inorganic C was

determined using a modified pressure calcimeter method (Sherrod et al. 2002). Soil pH

and texture were measured at the Montana State University Soil Testing Laboratory using

standard 1:1 (Eckert 1989) and modified hydrometer methods (Gee and Bauder 1986),

respectively. Using the measured bulk density of the field samples, all soil organic carbon

values were converted to and reported on an equivalent mass basis according to methods

described by Ellert and Bettany, 1995. Soil organic carbon (SOC) was calculated by

difference given by the equation:

$$TotC - IC = SOC \quad [Eqn\ 1]$$

Where TotC = total soil carbon, IC = inorganic soil carbon, and SOC = soil organic

carbon.

Soil samples were analyzed for total C, inorganic C, total N, texture, and pH.

Total C and total N content of soil samples were measured by combustion using a LECO

C/N/S 2000 analyzer (LECO Corporation, St. Joseph, MI, USA). Total nitrogen was

measured to calculate C:N ratios as a check for credible organic carbon levels. Soil C:N

ratios have been measured in a range of 3 to14:1 with a mean of 10:1 (Robertson et al.

1997). Soil C:N ratios outside that range could signify an inaccurate estimate of soil

inorganic carbon. Over- or underestimating soil inorganic C will result in erroneous

organic carbon values.

Statistical Analysis

Soil organic carbon data were analyzed using analysis of variance (ANOVA) for

all depths, and analysis of covariance (ANCOVA) for the 0 to 10 and 10 to 20 cm

depths. Clay content is a driving variable in soil organic matter dynamics. Clay contents

varied among and within sites; therefore, ANCOVA was used to determine if the

differences in soil organic carbon occurred when accounting for this covariant. Data were

analyzed using microsite values as replicates. 'Site' and 'Tillage' treatments were

considered fixed in the model. ANOVA and ANCOVA were run as full and reduced (by

site, by depth, and by site and depth) models. ANOVA and ANCOVA were also used

to determine if differences in soil organic carbon occurred when the data were pooled

into a 0-20 cm depth for comparison to Century model estimates. All statistical analyses

were preformed using the SAS statistical software 8.02 (SAS, 2001) and alpha = 0.1.

Century Model Initialization

The Century model was initialized for 5 of the 6 paired comparisons using present
and historical management information supplied by cooperating producers, soils data from
site-specific field samples and existing soil databases, and climate data from the National
Oceanic and Atmospheric Administration website. Crop growth parameters and tillage
effects on SOC are modifiable in Century. Crop growth parameters developed for
Saskatchewan, which has similar agricultural conditions as the study area, and default
tillage effects were used in this study. One site that was soil sampled (Conrad East) was
omitted because the no-till producer opted not to participate after soil samples were
taken. The data supplied by the cooperating producers included when the field was first
cultivated, management operations (by month), equipment used, crops grown (by year),
rotations (by year), fertilizer use (type, amount, date applied), and average yields (by
crop) for each field.

Soils data required for Century initialization include soil texture (% sand, silt, and
clay), bulk density, and drainage class. Paired fields were first modeled using site-specific
soils data (Table 2) and land management history. STATSGO and SSURGO soil data
were then used in place of site-specific soil data. Both STATSGO and SSURGO soils
information are reported as a range of high and low values, and all SSURGO data were
bound within the STATSGO range for clay % and bulk density in this study. Century soil
inputs spanned the full range of STATSGO data for the top 20 cm of soil (Century
assumes a depth of 20 cm).

Sensitivity Analysis

Holding the site-specific land management history constant, the Century model was initialized using the complete range of clay % and bulk density reported in STATSGO. It is important to note that STATSGO does report the relative % composition of the soil series in the mapping unit (Appendix B, Fig. B.1), however the database does not delineate how the soil series are distributed within the mapping unit For this reason, the largest range in texture and bulk density values were used to model the "worst-case" scenario. The model was re-initialized beginning with the lower limit of STATSGO data and proceeded by increasing clay content by 5% and decreasing soil bulk density proportionally (coarse-textured soils tend to have higher bulk density than fine-textured soils) to cover the complete STATSGO range in clay and bulk density for each field. All fields fell into the 'well-drained' drainage class; therefore, no adjustments were needed for that parameter. Soil water holding capacity is strongly linked to soil texture and the soil water status at wilting point (-15 bar) and field capacity (-0.33 bar) were estimated according to % sand, silt and clay, % organic matter, and soil bulk density in an equation developed by Rawls et al. (1982). The effect of erosion on soil organic carbon distribution was not included in model estimates. Century soil organic carbon estimates using both site-specific and STATSGO soils data were compared to measured C values.

Implied C change due to the adoption of no-till management for the field sites was calculated as the amount of organic carbon in the tilled fields subtracted from their paired

no-till plots. Prediction intervals were used to set a range in the distribution of predicted

values such that most observations will fall within that range with some degree of

confidence (Neter et al. 1996). Prediction intervals (95%) were built around the predicted

implied C change using the SigmaPlot 2000 v. 6.0 statistical software (SPSS Inc 2000).

The software used standard equations to calculate prediction intervals which are

described in detail by Neter et al. (1996). Results were compared to measured

differences to determine if Century predicted the measured implied change in carbon

including a 95% prediction interval.

Results

Soil Organic Carbon

Full model analysis (including all sites) of the 0-20 and 20-100 cm increments

showed that no-till had greater measured SOC in the 0 to 20 cm depth ($P<0.01$), but

SOC did not differ in the 20-100 cm depth ($P=0.12$) due to tillage (Table 3.3). In the 0

to 20 cm depth, mean squares for the site effect were reduced 6-fold and error mean

squares decreased 12% by adjusting for the clay covariant (ANCOVA), whereas tillage

means squares were unchanged (Table 3.3).

Despite the seemingly unimportant effect of clay on SOC in the full model,

analysis by site of the 0 to 20 cm increment showed that including clay as a covariant

influenced the significance of the tillage effect. Without adjusting for clay (ANOVA),

tillage had a significant effect on soil organic carbon at all sites except Conrad E (Table

3.4). Including clay as a covariant reduced error means squares at Chester, and had little effect at the other sites (Table 3.4). In addition to the non-significant tillage effect at Conrad E, ANCOVA also resulted in the loss of a significant tillage effect at Saint Johns (Table 3.4).

Table 3.3. Full model ANOVA and ANCOVA soil organic carbon mean squares for 0 to 20 and 20 to 100 cm depths at 6 sites in north central Montana, 2001.

	0 - 20 cm	
	Mean Squares	P
	ANOVA	
Site	310.6	<0.01
Tillage(S)	30.7	<0.01
Error	3.3	
	No-till	Till
	$t\ ha^{-1}$	
Mean SOC (SE)	24.2 (0.9)	21.3 (0.8)
	ANCOVA	
Site	50.9	<0.01
Tillage(S)	30.7	<0.01
Clay	25.4	<0.01
Error	2.9	
	No-till	Till
	$t\ ha^{-1}$	
Mean SOC (SE)	24.3 (0.3)	21.2 (0.3)
	20 - 100 cm	
	ANOVA	
Site	75.7	<0.01
Tillage(S)	11.4	0.12
Error	6.4	
	No-till	Till
	$t\ ha^{-1}$	
Mean SOC (SE)	18.0 (0.6)	17.6 (0.5)

In the 20 to 100 cm depth, tillage had a significant effect only at the two Conrad

sites (Table 3.4), however the trend for the tillage effect was not consistent. At Conrad

E, the tilled field had greater SOC than the no-tilled field; however, the relationship was

reversed and of similar magnitude at Conrad W (Table 3.5).

Across all sites, no prevalent relationship occurred between soil texture and C

sequestration rates with the adoption of no-till ($r^2 = 0.19$) (Fig. 3.4). Annualized rates of

C accumulation under no-till were used because the length of time since the adoption of

no-till was not consistent across all sites. There is a general trend of increasing C storage

with higher clay content, but the relationship is not strong. Annual precipitation can have a

significant impact on primary productivity and hence C inputs for the soil. Annual

precipitation since the adoption of no-till does not appear to have a strong relationship

with respect to the annualized rates of implied carbon change. Annual precipitation since

1996 has consistently been lower than mean annual precipitation at each site (Appendix

A, Figs A.1-A.4), and represents a minimum of half the years of no-till for each site.

There are likely many confounding variables that could be effecting C sequestration rates

in addition to soil texture.

Table 3.4. Reduced model (by site) mean squares for the 0 to 20 (ANOVA and ANCOVA) and 20 to 100 (ANOVA) cm depths at Chester, Conrad East (E), Conrad West (W), Ft. Benton, Saint Johns, and Simpson, Montana, 2001.

	Chester		Conrad E		Conrad W		Ft. Benton		Saint Johns		Simpson	
	Mean Squares	P>F	Mean Squares	P>F	Mean Squares	P>F	Mean Squares	P>F	Mean Squares	P>F	Mean Squares	P>F
0 to 20 cm												
ANOVA												
Tillage(S)	32.4	0.07	4.2	0.34	48.8	<0.01	76.0	<0.01	4.6	0.06	18.3	0.01
Error	8.0		4.1		1.3		3.6		1.0		1.9	
ANCOVA												
Clay	24.4	0.08	7.9	0.18	0.2	0.73	6.2	<0.01	0.1	0.77	0.1	0.88
Tillage(S)	38.9	0.03	2.9	0.39	42.8	<0.01	82.1	<0.01	3.3	0.12	10.1	0.05
Error	6.2		3.7		1.4		3.4		1.1		2.1	
20 to 100 cm												
ANOVA												
Tillage(S)	0.4	0.87	31.4	0.02	25.1	0.07	4.5	0.38	0.7	0.52	6.1	0.42
Error	13.3		4.1		5.7		5.3		1.7		8.5	

Table 3.5. Mean soil organic carbon values (0 to 20 and 20 to 100 cm) unadjusted (Mean) and adjusted (LS Mean) for clay content for Chester, Conrad East (E), Conrad West (W), Ft. Benton, Saint Johns, and Simpson, Montana, 2001.

	Chester		Conrad E		Conrad W		Ft. Benton		Saint Johns		Simpson	
	t ha^{-1}											
0 to 20 cm												
Mean	23.1		26.0		21.1		30.4		19.1		15.9	
SE	0.9		0.6		0.7		0.9		0.3		0.5	
LS Mean	22.3		24.1		20.8		27.8		21.6		19.9	
SE	0.6		0.8		0.7		1.0		1.0		1.5	
20 to 100 cm												
Mean	18.4		18.5		17.0		21.9		14.3		16.6	
SE	1.0		0.7		0.9		0.7		0.4		0.8	
0 to 20 cm												
	No-till	Till	No-till	Till	No-till	Till	No-till	Till	No-till	Till	No-till	Till
Mean	24.7*a*	21.4*b*	26.6*a*	25.4*a*	24.1*a*	20.1*b*	32.9*a*	27.8*b*	19.8*a*	18.5*b*	17.1*a*	14.6*b*
SE	1.1	1.2	0.9	0.8	0.4	0.5	1.0	0.5	0.4	0.4	0.4	0.7
LS Mean	24.0*a*	20.5*b*	24.6*a*	23.6*a*	22.6*a*	19.0*b*	30.5*a*	25.1*b*	22.6*a*	20.7*a*	20.9*a*	19.0*b*
SE	0.7	0.8	1.0	0.9	0.9	0.8	1.1	1.2	1.2	1.0	1.5	1.7
20 to 100 cm												
Mean	18.6*a*	18.3*a*	16.9*a*	20.1*b*	18.6*a*	15.3*b*	22.5*a*	21.2*a*	14.1*a*	14.6*a*	17.3*a*	15.9*a*
SE	2.0	0.5	0.7	1.0	1.4	0.9	1.1	0.8	0.5	0.5	0.9	1.4

Means followed by same letter did not differ ($P = 0.1$).

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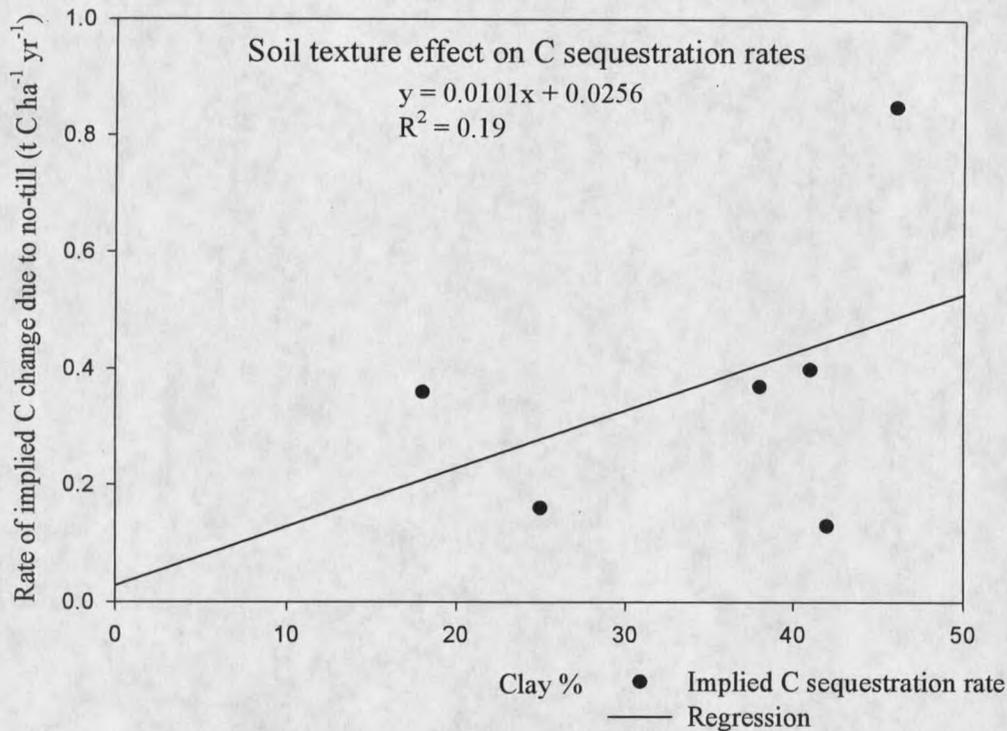
Figure 3.4. Regression of soil clay content on the annualized rate of implied C change due to the adoption of no-till at six sites in north central Montana, 2001.

Analyzing by sampling depths (0-10, 10-20, 20-50 and 50-100 cm) across all sites, tillage had a significant effect in the 0-10 and 10-20 cm depths ($P<0.01$ for both ANOVA and ANCOVA, respectively) (Table 3.6). Tillage effect was significant for the 20 to 50 cm depth ($P = 0.04$ for ANOVA) and not significant at the 50-100 ($P = 0.28$ for ANOVA) cm depth (Table 3.6). Adjusting for clay, mean squares for site were reduced 4-fold in the 0-10 cm depth and nearly 10-fold for the 10-20 cm depth (Table 3.6). Mean squares for tillage remained unchanged in the 0-10 cm depth, but were reduced by nearly half in the 10-20 cm depth (Table 3.6). Error mean squares were unchanged (Table 3.6).

Examining the sampling depth analysis by site revealed mixed results for tillage in the 0 to 10 and 10 to 20-cm depths. Without adjusting for clay, no-till fields had greater SOC than tilled fields at 3 of 6 sites (0 to 10 cm), Conrad W, Ft. Benton, and Simpson (Table 3.7). No-till remained greater than tilled after adjusting for clay at 3 of 6 sites, but Saint Johns and Simpson were interchanged (Table 3.7).

Soil organic carbon results in the 10 to 20 cm depth were more consistent among sites and tillage systems, but were inconsistent below 20 cm. No-till was significantly greater than till at the Conrad E and Chester sites only (10 to 20 cm), regardless if means were adjusted for clay (Table 3.7). Below 20 cm, no consistent tillage effect was detected (Table 3.8).

Table 3.6. Reduced model (by depth) ANOVA and ANCOVA mean squares for 0 to 10, 10 to 20, 20 to 50 , and 50 to 100 cm depths at 6 sites in north central Montana, 2001.

	0-10 cm		10-20 cm		20-50 cm		50-100 cm	
	Mean Squares	$P{>}F$	Mean Squares	$P{>}F$	Mean Squares	$P{>}F$	Mean Squares	$P{>}F$
				ANOVA				
Site	89.5	<0.01	70.4	<0.01	29.8	<0.01	28.3	<0.01
Tillage(S)	11.7	<0.01	8.3	<0.01	5.7	0.04	4.1	0.3
Error	1.2		1.7		2.4		3.2	
		ANCOVA						
Site	24.1	<0.01	7.4	<0.01				
Tillage(S)	11.6	<0.01	4.5	<0.01				
Clay	2.3	0.16	0.8	0.5				
Error	1.2		1.7					

Table 3.7. ANOVA and ANCOVA soil organic carbon means (t ha^{-1}) for the 0 to 10 and 10 to 20 cm depths at Chester, Conrad East (E), Conrad West (W), Ft. Benton, Saint Johns, and Simpson, Montana, 2001.

	Chester		Conrad E		Conrad W		Ft. Benton		Saint Johns		Simpson	
	No-till	Till	No-till	Till	No-till	Till	No-till	Till	No-till	Till	No-till	Till
	ANOVA											
	0 - 10 cm											
Mean	11.9*a*	11.5*a*	12.4*a*	12.7*a*	12.8*a*	10.1*b*	17.7*a*	14.2*b*	10.0*a*	9.6*a*	8.8*a*	7.0*b*
SE	0.7		0.6		0.4		0.3		0.3		0.3	
	10 - 20 cm											
Mean	12.9*a*	9.9*b*	14.3*a*	12.7*b*	11.4*a*	10.1*b*	15.1*a*	13.6*a*	9.8*a*	9.0*a*	8.3*a*	7.6*a*
SE	0.7		0.5		0.4		0.7		0.4		0.4	
	ANCOVA											
	0 - 10 cm											
LS Mean	11.9*a*	11.4*a*	12.2*a*	12.9*a*	12.9*a*	10.1*b*	17.9*a*	14.0*b*	10.7*a*	8.8*b*	8.3*a*	7.4*a*
SE	0.7		0.6		0.4		0.3		0.3		0.4	
	10 - 20 cm											
LS Mean	12.8*a*	9.9*b*	14.3*a*	12.6*b*	11.3*a*	10.2*a*	15.3*a*	13.6*a*	9.7*a*	9.0*a*	8.4*a*	7.6*a*
SE	0.8		0.4		0.5		0.8		0.4		0.4	

Means followed by same letter did not differ ($\alpha = 0.1$).

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Table 3.8. ANOVA soil organic carbon means (t ha^{-1}) for 20 to 50 and 50 to 100 cm depths at Chester, Conrad East (E), Conrad West (W), Ft. Benton, Saint Johns, and Simpson, Montana, 2001.

	Chester		Conrad E		Conrad W		Ft. Benton		Saint Johns		Simpson	
	20 - 50 cm											
	No-till	Till	No-till	Till	No-till	Till	No-till	Till	No-till	Till	No-till	Till
Mean	9.3a	8.9a	10.6a	12.7b	11.6a	9.2b	13.3a	12.4a	9.2a	8.9a	9.1a	9.5a
SE	0.9		0.6		0.6		0.7		0.4		0.5	
	50 - 100 cm											
Mean	9.4a	9.5a	6.3a	7.4a	8.0a	6.1a	9.2a	8.9a	4.9a	5.6a	8.2a	6.4a
SE	1.0		0.6		0.8	0.6	0.5		0.4		1.0	

Means followed by same letter do not differ ($P = 0.1$).

<u>Century Model Sensitivity Analysis</u>

The Century model tended to overestimate soil organic carbon as compared to measured values. Modeled SOC estimates of soil organic carbon in the 0-20 cm soil depth at each site deviated from measured values with an average difference of +10% and a range of -1% to +28% (Table 3.9). On average, model estimates for the two tillage systems were +9% (-1% to +28%) and +11% (1% to 18%) for no-till and tilled, respectively, supporting that the Century model is not biased toward a particular tillage system (Table 3.9).

The accuracy of Century model SOC estimates using STATSGO and SSURGO database information were dependant upon whether reported database ranges in clay content included the site-specific values. STATSGO reported clay contents fell short of measured clay values at Chester, Ft. Benton and the tilled field at Conrad W, but encapsulated the measured clay values at Conrad W (no-till field), Saint Johns, and Simpson (refer to Table 3.2 for STATSGO values). SSURGO ranges in clay content did not include measured values at Chester, Conrad W, Ft. Benton and the tilled field at Simpson, but did include measured values at Saint Johns and the no-till field at Simpson (refer to Table 3.2 for SSURGO values). Ranges in estimated SOC content using STATSGO and SSURGO soil data included the measured SOC values at Chester (tilled field only), Conrad W, Saint Johns, and Simpson (with the exception of the tilled field using SSURGO data). Largely due to shortcomings of the STATSGO and SSURGO databases, model predictions of SOC were lower than measured values at Chester (no-till

field only) and Ft. Benton (Table 3.9, Figs. 3.5-3.6). Estimated soil organic carbon values from Century were similar to measured values at Saint Johns and Simpson (Table 3.9, Figs. 3.7-3.8); however the model tended to over-estimate measured values at Conrad W (Table 3.9, Fig. 3.9).

Estimates of SOC from Century for no-till and tilled systems are sensitive to the effect of clay content based on the range of modeled SOC values. Estimated SOC for the upper limit of reported clay content was 2.3 to 2.7 times greater than SOC estimates for the lower clay limit at the 5 sites modeled using STATSGO database ranges in clay content (Figs 3.5-3.9).

Model estimates showed inconsistent results when predicting the implied carbon change (no-till C - tilled C) using STATSGO soil data. Either as a reflection of shortcomings of the soils database or an apparent insensitivity of Century to the effect of clay content on SOC gains through no-till adoption, Century estimates (including a 95% prediction interval) did not predict the measured implied change in C associated with the adoption of no-till at any of the 5 sites (Figs. 3.10-3.14). Century underestimated implied C change at Chester and Simpson (Figs. 3.10-3.11); however, implied C change was over-estimated at Conrad W and Saint Johns (Figs. 3.12-3.13). Century estimates were lower than the measured value at Ft. Benton as a function of both a shortcoming in STATSGO data and an apparent insensitivity of the models to tillage effect on SOC as a function soil clay content (Fig. 3.14).

Table 3.9. Measured and predicted 0 to 20 cm SOC values (using site-specific and soil databases for soils inputs) for no-till and tilled fields at, Simpson, Saint Johns, Ft. Benton, Chester, and Conrad West (W), Montana, 2001.

	Chester		Conrad W		Ft. Benton		Saint Johns		Simpson	
	No-till	Till	No-till	Till	No-till	Till	No-till	Till	No-till	Till
	t ha^{-1}									
Mean Measured SOC	24.7	21.4	24.1	20.1	32.9	27.8	19.8	18.5	17.1	14.6
SE	1.1	1.2	0.4	0.5	1.0	0.5	0.4	0.4	0.4	0.7
95% CI	23.5-29.9	22.7-20.1	23.7-24.5	19.6-20.6	31.9-33.9	27.3-28.3	19.4-20.2	18.1-18.9	16.7-17.5	13.9-15.3
Site-specific soils data										
Predicted SOC	25.3	23.6	30.7	24.4	36.7	32.2	19.7	19.4	17.4	14.8
Difference from measured SOC	+2%	+10%	+28%	+21%	+12%	+16%	-1%	+5%	+2%	+1%
SSURGO soils data used for input data										
Predicted range in SOC with varying input data	16-24	15-22	23-27	19-22	26-30	24-27	18-22	16-20	17-22	15-19
STATSGO soils data used for input data										
Predicted range in SOC with varying input data	10-23	9-21	13-29	9-24	13-29	11-26	12-28	10-26	10-27	10-26

Figure 3.5. Predicted SOC values (Century model) for no-till and tilled fields using STATSGO reported clay content in 5% increments at Chester, Montana, 2001.
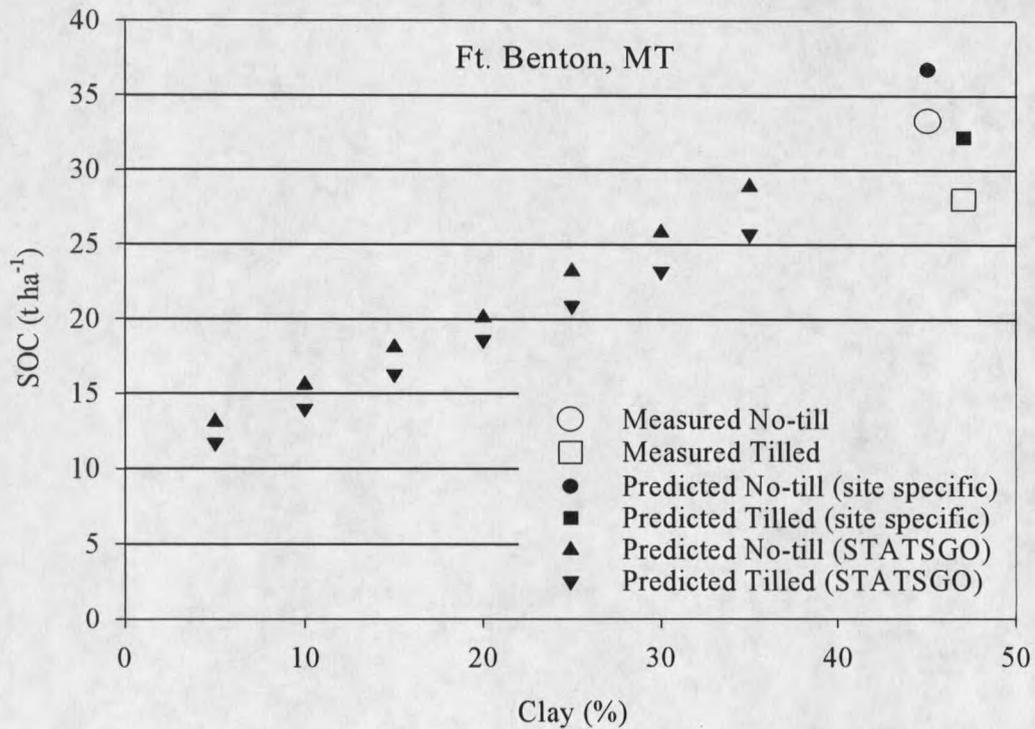
Figure 3.6. Predicted SOC values (Century model) for no-till and tilled fields using STATSGO reported clay content in 5% increments at Ft. Benton, Montana, 2001.
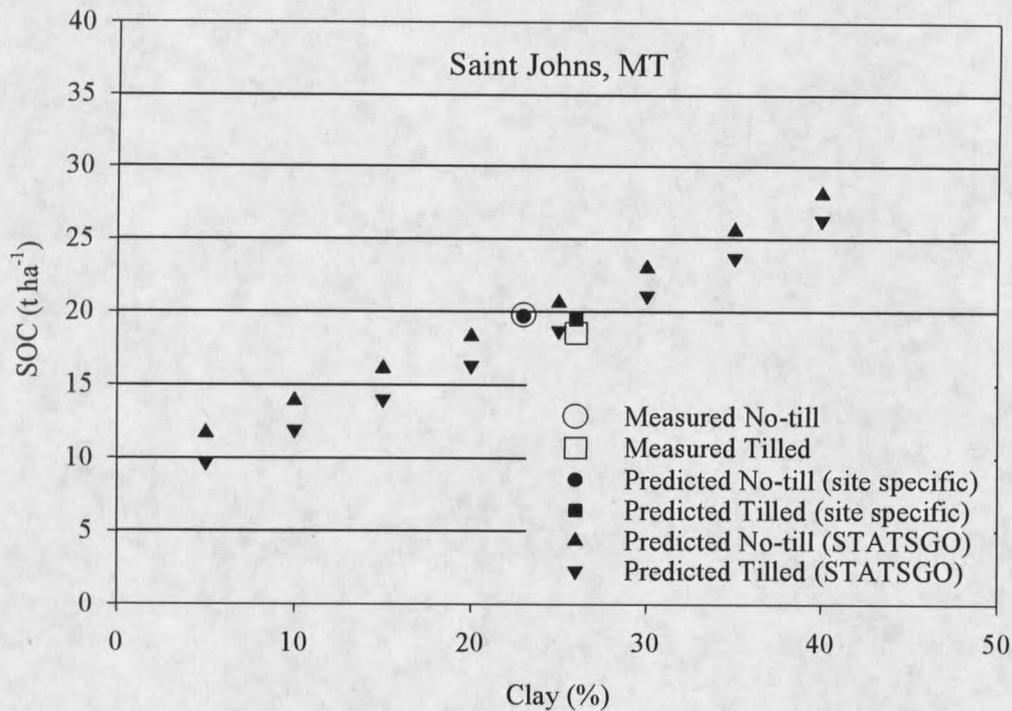
Figure 3.7. Predicted SOC values (Century model) for no-till and tilled fields using STATSGO reported clay content in 5% increments at Saint Johns, Montana, 2001.
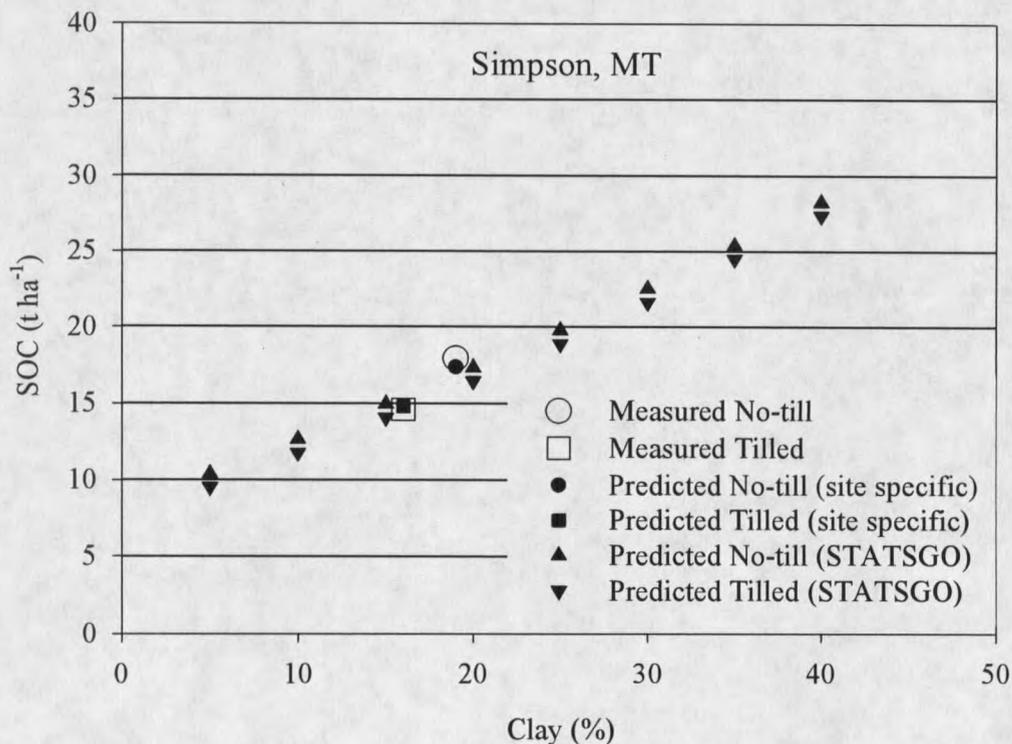
Figure 3.8. Predicted SOC values (Century model) for no-till and tilled fields using STATSGO reported clay content in 5% increments at Simpson, Montana, 2001.
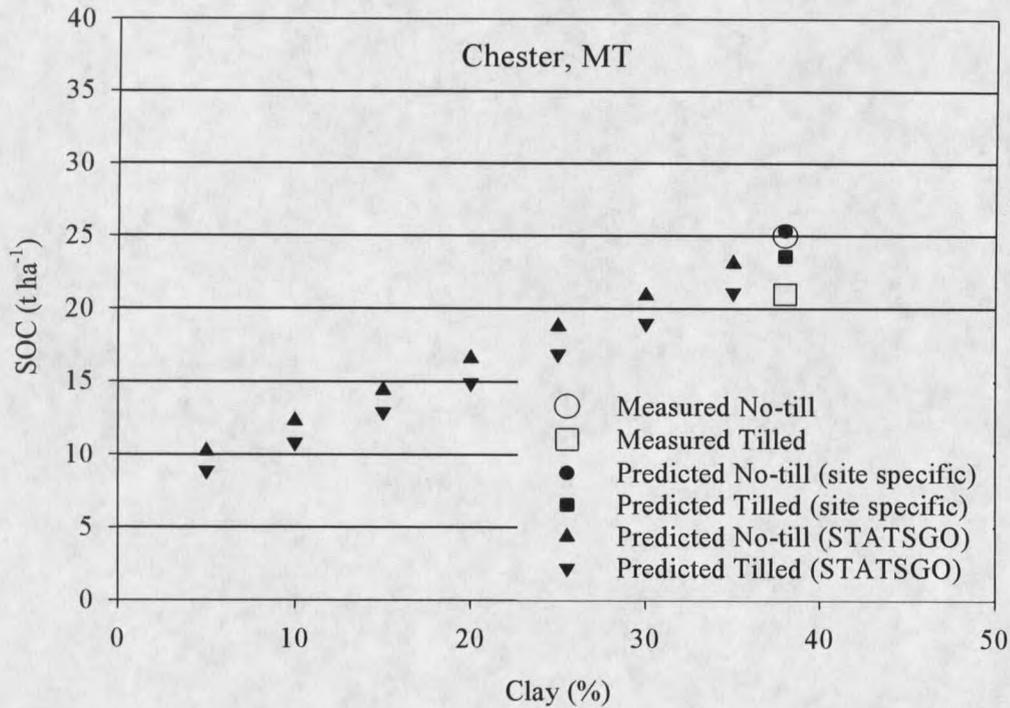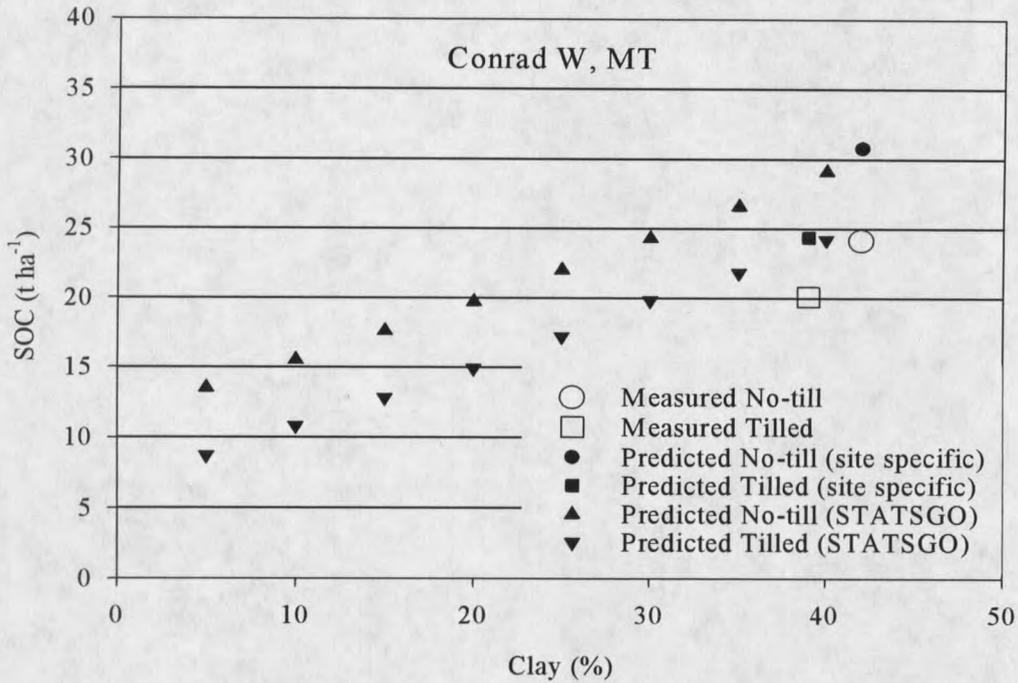
Figure 3.9. Predicted SOC values (Century model) for no-till and tilled fields using STATSGO reported clay content in 5% increments at Conrad W, Montana, 2001.
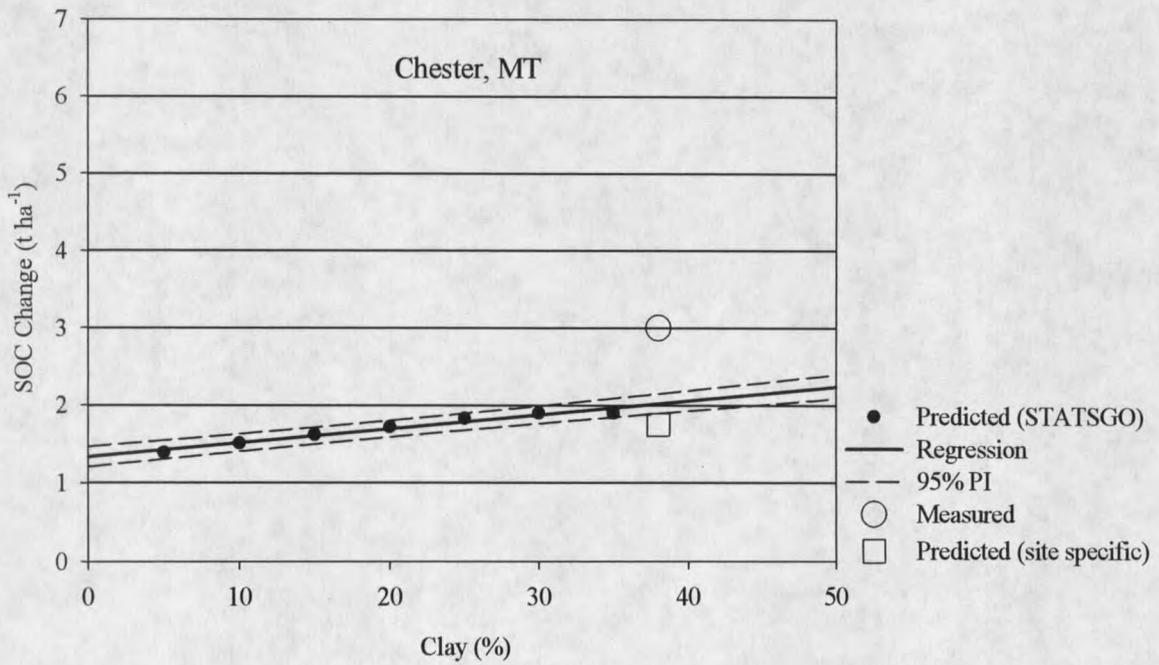

Figure 3.10. Implied carbon change (no-till C - tilled C), with 95% prediction interval for modeled values, using STATSGO reported clay content at Chester, Montana, 2001.
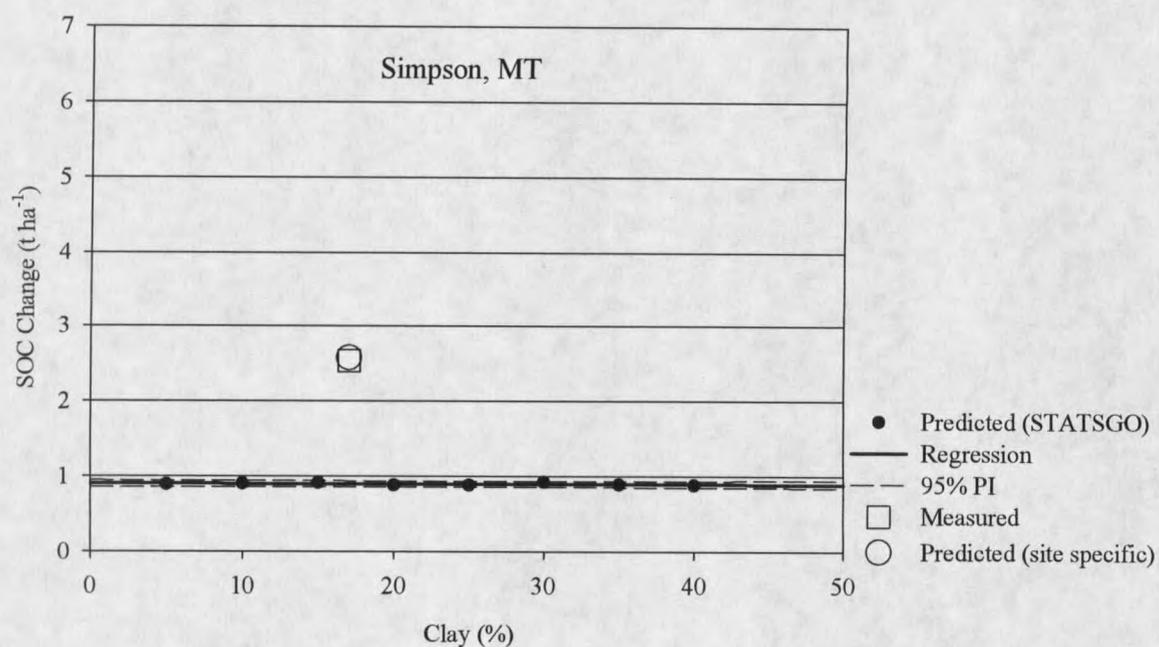
Figure 3.11. Implied carbon change (no-till C - tilled C), with 95% prediction interval for modeled values, using STATSGO reported clay content at Simpson, Montana, 2001.
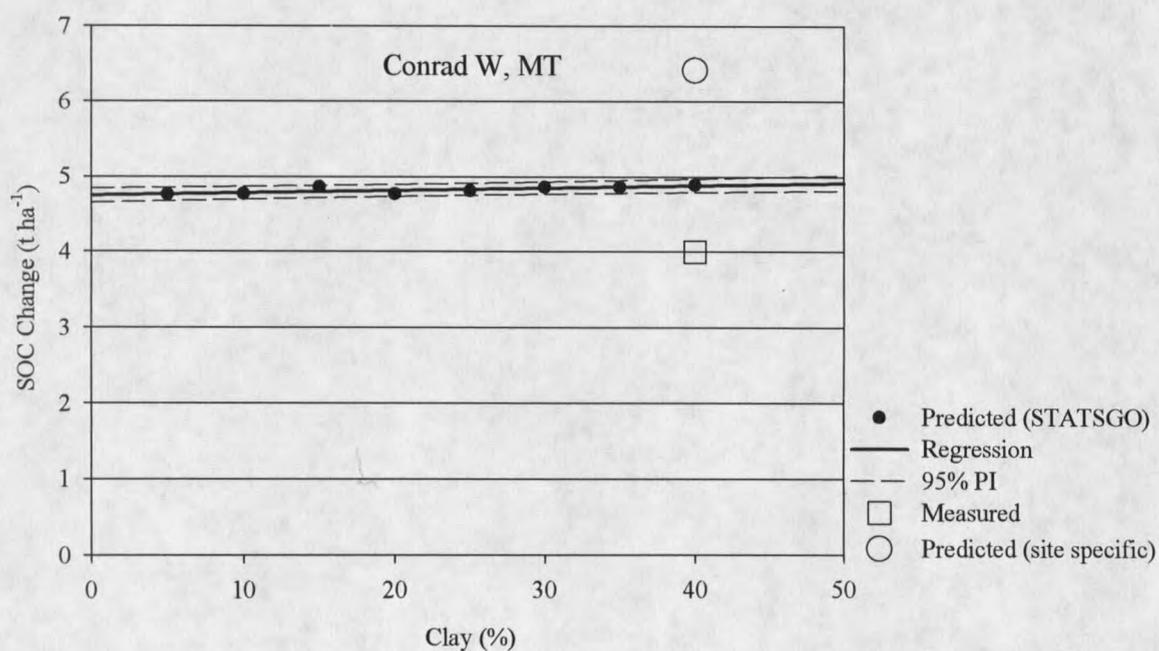
Figure 3.12. Implied carbon change (no-till C - tilled C), with 95% prediction interval for modeled values, using STATSGO reported clay content at Conrad W, Montana, 2001.
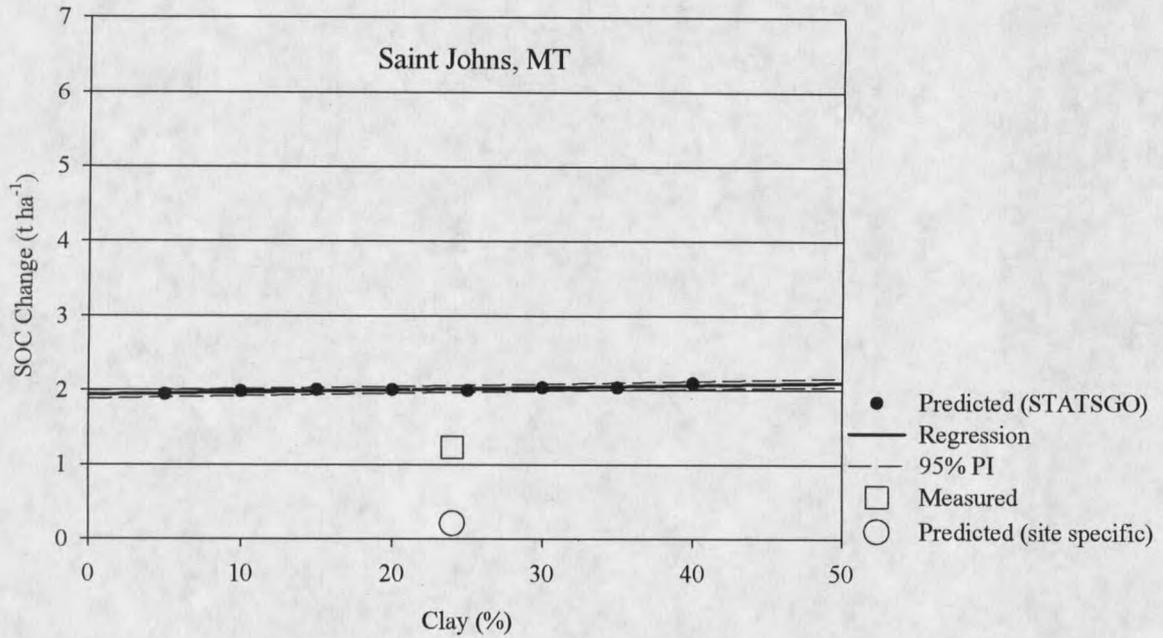
Figure 3.13. Implied carbon change (no-till C - tilled C), with 95% prediction interval for modeled values ,using STATSGO reported clay content at Saint Johns, Montana, 2001.
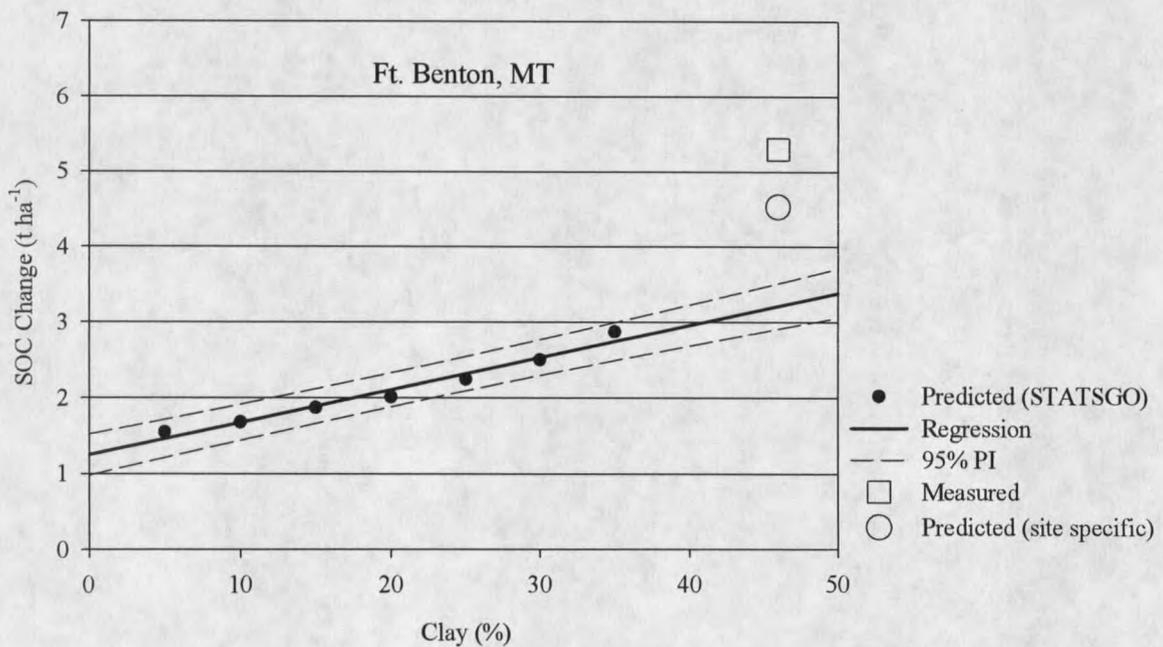
Figure 3.14. Implied carbon change (no-till C - tilled C), with 95% prediction interval for modeled values, using STATSGO reported clay content at Ft. Benton, Montana, 2001.

Discussion

Soil Organic Carbon

Significant increases in soil organic carbon due to the adoption of no-till

management and increased cropping intensity have been documented in other studies

(Paustian et al. 1997,1998; West and Marland 2002; Campbell et al. 1996, Halvorson et

al. 2002). Tillage commonly does not have an effect on SOC below the depth of

plowing, which tends to be less than 20 cm in the Northern Plains (Campbell et al. 2000a;

Campbell et al. 200b). Though SOC differences occurred in the 20 to 50-cm depth at

two sites in this study, the tillage effect on SOC conflicted, where the tilled field had

greater SOC than the no-till field at Conrad E and the reverse occurred at Conrad W.

Reliable, unbiased baseline carbon values will be a critical component to a

market-based carbon trading system. Likewise, it would be equally important to

determine; 1) if significant gains in soil carbon occur, and 2) the rate of C accumulation

that occurs with the adoption of best management practices, such as the elimination of

tillage (i.e. no-till), in major agricultural regions. This would particularly be of interest if a

practice-based (i.e. per-acre payments) approach were to be taken for carbon trading.

The data reported in this study support that clay content was an important covariant with

respect to soil organic carbon measurements and should be considered when determining

differences among sites. Including clay as a covariant was not as critical for determining

differences among tillage systems.

Covariate effects of clay content on the significance of SOC content under no-till management occurred in both full and reduced models, however the effects were small as compared to comparisons among sites. Adjusting for differences in clay content (i.e. ANCOVA) when analyzing SOC values among and within sites (i.e. tillage effects) may have reduced bias (i.e. soil texture) for estimates of soil organic carbon values and may have provided more reliable determination of true differences in soil organic carbon in paired field site comparisons.

Significant tillage effects on SOC differences within the top 20 cm (0 to 10, 10 to 20 or 0 to 20 cm depths) occurred at all sites when including clay as a covariant. In the aggregated 0-20 cm analysis, ANCOVA showed differences at 4 of 6 sites. Conrad E and Saint Johns were the exceptions. When including a covariant in the analysis, mean responses are a function of both the treatment and the value of the covariant (Neter et al. 1996). ANCOVA adjusts mean values to account for the influence of the covariant, thus ANCOVA means differed from ANOVA means. In the 0-20 cm depth, ANCOVA mean differences in SOC between tillage practices converged by 0.2, 0.4, and 0.6 t ha^{-1} at Conrad E, Conrad W, and Simpson, respectively, with no change in significance as compared to ANOVA. For the same depth at Chester, Ft. Benton, and Saint Johns, mean SOC differences diverged by 0.2, 0.3, and 0.6 t ha^{-1}, respectively, however significant difference determined by ANOVA at Saint Johns was overturned. The more detailed 0 to 10 and 10 to 20-cm analysis showed a tillage effect at Saint Johns in the 0-10 cm depth and at Conrad W in the 10-20 cm depth where ANOVA had previously

determined no difference. ANCOVA had the opposite effect at the Simpson site in the 0 to 10-cm depth. Accounting for the clay effect in the 0-10 cm depth caused mean SOC differences to diverge at Chester, Conrad W, Conrad E, Ft. Benton, and Saint Johns by, 0.1, 0.1, 0.4, 0.4 and 1.5 t ha^{-1}, respectively and to converge at Simpson by 0.9 t ha^{-1}. In the 10 -20 cm depth, mean SOC differences converged at Chester (0.1 t ha^{-1}), Saint Johns (0.1 t ha^{-1}), and Conrad W (0.2 t ha^{-1}), causing a significant difference at Conrad W to be overturned, and diverged at the remaining sites by 0.1 to 0.2 t ha^{-1} as a result of accounting for clay percentage differences. Including clay as a covariant to remove the effect of % soil clay on SOC content provided reliable estimates of the effects of management at 6 sites in north central Montana.

The use of clay as a covariant for paired management comparisons of SOC content at the field level may not be as critical as for comparisons at a coarser scale (B. McConkey, pers. Comm., 2003). Textural differences at the field level may be a function of the interactions of translocation processes, soil movement, and tillage. For example, an accumulation layer of translocated clay commonly occur in the soils of this region. Soil movement through wind and water erosion has the potential to remove significant amounts of the soil surface, thus reducing the effective soil depth to the accumulated clay layer. As this process continues, particularly in a tilled fallow scenario, portions of the accumulated clay layer could be incorporated into the surface soil or exposed altogether, thus resulting in fine-scale differences in clay content within a field. Clay content would remain a consideration, however including clay content as a covariant may not be critical.

The magnitude of measured SOC in the 0-10, 10-20 and 0-20 cm depths are consistent with other dryland agriculture research. The management systems in this study were primarily adequately fertilized wheat-fallow systems. The magnitude of implied SOC increase due to 6-10 years of no-till ranged from 1.2 to 5.3 t C ha^{-1}, which was consistent with the 2 to 5 t C ha^{-1} increase with 7-11 years of no-till in Saskatchewan (Campbell et al. 1996, 2001b).

North-central Montana and southwestern Saskatchewan are in the same agroecoregion (Padbury et al. 2002), thus comparison of results within this agroecosystem is especially relevant. Annual SOC gains were estimated using the measured implied C change (no-till C - tilled C) and averaging over years in no-till. The annual C gains in the 0-10, 10-20 and 0-20 cm depths were similar to or less than that reported from southwest Saskatchewan, with the exception of Ft. Benton. Annual C accumulation due to the adoption of no-till ranged from 0.13 to 0.4 t C ha^{-1} yr^{-1} at 5 of 6 sites, with a much higher rate of 0.87 t C ha^{-1} yr^{-1} at Ft. Benton. At Simpson, Chester, and Conrad W, annual C gains were 0.34, 0.36, and 0.4 t C ha^{-1} yr^{-1}, after 7, 9, and 10 years of no-till, which is equivalent to rates measured in southwestern Saskatchewan (Campbell et al. 1996, 2001b). No-till management sequestered soil carbon at rates of 0.37 to 0.52 t C ha^{-1} yr^{-1} near Swift Current, Sask. in fertilized wheat-fallow fields over 7 to 11 years (Campbell et al. 1996, 2001b). The Conrad E and Saint Johns sites had annual gains of 0.13 and 0.15 t C ha^{-1} yr^{-1} after 9 and 8 years of no-till, respectively, which is lower than Campbell et al. (2001b) findings. The Saint Johns site had one of the

lowest mean annual precipitation values which would tend to limit net primary production, C inputs into the soil, and hence, C gains. Additionally, beginning in 1996, annual precipitation across all sites was only 72 to 90% of the 30- yr average (Appendix A, Figs. A.1-A.4), which would also contribute to lower sequestration rates.

Modeled annual C gains were calculated using the predicted implied C change (no-till C - tilled C) from the Century model and averaging over years in no-till. Modeled estimates ranged from 0.04 to 0.75 t C ha^{-1} yr^{-1} for 5 of the 6 sites (Conrad E could not be modeled due to lack of management data). Agreements between measured and modeled annual gains were mixed among sites. For example, modeled C gains were equivalent to the measured value at Simpson, greater than measured at Conrad W, and modeled C gains were lower than measured at Chester, Ft. Benton and Saint Johns. Modeled C gains deviated from measured by 0.03 to 0.23 t C ha^{-1} yr^{-1}.

Though the Ft. Benton site had greater annual C gains than has been reported in other parts of the same agroecoregion, site characteristics were different than the other sites in this study and those differences tend to favor greater C storage. The Ft. Benton site had the highest clay content of all the sites and had been under no-till for the shortest time (6 yr). Clay soils accumulate more carbon under no-till (Campbell et al. 1996) and C storage tends to be greatest in the first decade of no-till adoption (Paustian et al. 1997). Additionally, the 30-yr average (1961-1990) annual precipitation was greater at Ft. Benton than any other site, which would favor increased net primary production, increased C inputs to the soil, and hence, greater C storage. Nevertheless, the rate of

change at Ft. Benton is higher than typically found in other uncontrolled field experiments in this agroecozone.

It is difficult to show causality in paired observational comparisons because there are potentially more confounding variables than can be accounted for in such a study. Although due diligence was used in establishing these paired sites, causality must be inferred with caution. Paired comparisons certainly measure a difference, but additional unknowns limit the extent of SOC change that can be attributed simply to change in tillage system. Additionally, by excluding the carbon stored in the accumulation of plant material on the surface of no-till fields, the no-till vs tilled comparison may be biased toward the tilled field due to annual incorporation of crop residues into the soil. However, taken in the aggregate, no-till systems in this study increased SOC in the 0 to 20-cm soil depth, across all sites, by an average of 2.9 t C ha^{-1} over a 6 to 10 yr period.

Century Sensitivity Analysis

The dependability of Century estimates was largely dependant on the accuracy of input data for soil properties. The lack of both precision and accuracy in STATSGO and SSURGO database information limited model results. STATSGO and SSURGO database ranges of clay content did not encompass the measured clay values in one or both of the tillage treatments at 5 or 7 site/tillage combinations, respectively, which had a profound effect on the accuracy of model estimates. The overall average deviation of Century model SOC estimates from measured values was +10% when using site-specific

soils data. When using STATSGO and SSURGO soils data, in only 7 of 10 (STATSGO) and 6 of 10 (SSURGO) cases did the range in model estimates include measured values, despite the large range in predicted values. When STATSGO included the measured clay values, the model performed well (5 of 5 modeled SOC ranges included measured values); however, when STATSGO did not include measured clay values, only 2 of 5 of the ranges in modeled SOC estimates encompassed the measured values. A similar trend occurred when using SSURGO soils data (3 of 3 when measured clay was included in SSURGO and 4 of 7 otherwise). Though SOC estimates were well short of the measured values when the databases fell short, the slope of the prediction line suggests that if the databases included the measured clay value, modeled estimates were on target.

For each site and tillage system, as clay content increased SOC estimates from Century also increased. There was a 2.5-fold increase in the amount of carbon predicted from the low clay values to the high clay values. This supports that Century is sensitive to the effects of soil texture when predicting soil organic carbon. Century calculates a decomposition rate of the active soil organic matter pool and the stabilization efficiency of the active SOM pool to the slow pool as functions of soil silt and clay fractions (Parton et al. 1994).

Field studies have been inconclusive in determining the effect soil texture has on C sequestration with the adoption of no-till management. The results of this study did not show a meaningful relationship of soil texture with annualized implied rate of C change due to adopting no-till ($r^2 = 0.19$). In a review of 11 long-term paired conventional and no till

experiments, Paustian et al. (1997) also did not observe an apparent pattern of C sequestration related to the effects of soil texture to a depth of 30 cm. Similarly, data from approximately 140 field sites across major agricultural regions of Canada have not shown a definite relationship of soil texture on C storage with the elimination of tillage in the short-term (3 years) (B. McConkey, unpublished data, 2003). Additionally, Sims and Nielsen (1986) did not find a significant correlation between clay and SOC content in 52 cultivated grassland A horizons in Montana. The Sims and Nielsen (1986) study, however, included substantially larger ranges in elevation and precipitation (670-1463 m and 28-76 cm, respectively) than the present study (840-1040 m, and 26-36 cm), which could have masked a relationship between clay and SOC. Soil organic carbon research from long-term plot studies in southwest Saskatchewan did show that soil texture influences the amount of soil carbon gained in response to no-till management in the 0 to15-cm soil depth (Campbell et al. 1996). After 11 to 12 years of no-till, the amount of soil carbon was greater in no-tilled fields than the tilled control by 0, 1.6 and 3.9 t ha^{-1} in a sandy loam (~10% clay), silt loam (~23% clay), and clay (42% clay) soil, respectively. The relationship of soil texture on C accumulation with the adoption of no-till may not be a major factor in short-term C gains (3 to 5 yr) where confounding factors such as drought, erosion, and destructive weather events may limit biomass production (B. McConkey, pers comm. 2003). In the long-term (10+ yr), soil texture effects may become more evident.

Assuming a relationship exists between soil texture and the effect of tillage on soil organic carbon, Century estimates were not sensitive to the effects of soil texture (i.e. clay%) on soil carbon change in response to the adoption of no-till. Century did not predict difference the amount of implied carbon change due to the adoption of no-till at the 5 sites modeled in this study with respect to the wide range in clay percentages. Model results for the five sites in Montana showed little difference in the amount of carbon stored in coarse-textured soils (5% clay) compared to fine-textured soils (35-40% clay), with the exception of the Ft. Benton site. For example, the difference in predicted soil C change per year at 5% and 35 or 40 % clay content ranged from 0 to 0.06 t C ha^{-1} yr^{-1} across all sites, except Ft. Benton. In this study, the time under no-till management was less than 11-12 years (6 to 10 years); however, Paustian et al. (1997), citing unpublished data from C.A. Campbell, found that SOC increased (0 to 15-cm depth) more rapidly in the first 6 yr (15%) of no-till than the remaining 7 yr (4%) of a 13-yr study in Saskatchewan. The relationship of nearly 0.12 t C ha^{-1} increase per 1% increase in clay content observed by Campbell et al. (1996) was not observed in Century results for the Montana sites. For example, Century estimated implied carbon gains due to the adoption of no-till from 1.0 to 4.6 t ha^{-1} at 5% clay, and gains of 0.9 to 5.0 t ha^{-1} at 35 to 40% clay.

The insensitivity of Century to soil texture effects on C sequestration with the adoption of no-till management is consistent with research by Smith et al. (2001) when modeling C dynamics across Canada, where Century showed no influence of soil texture

on C change in response to changes in tillage management. Using Century, Smith et al.

(2001) determined soil carbon sequestration coefficients for a wheat-fallow system in

sandy loam (0.052 t C ha^{-1} yr^{-1}), loam (0.052 t C ha^{-1} yr^{-1}) and clay loam (0.049 t C ha^{-1}

yr^{-1}) soils in the Brown Chernozem soil group, similar to the soils in the Montana study

area. The insensitivity of the model to soil texture effects on C sequestration under no-till

management is also consistent with the Century model's assumed direct influence of tillage

disturbance on C storage. The Century model treats the influence of tillage as a relative

increase in the potential rate of decomposition and is independent of soil texture in the

model. Texture sensitivity in Century occurs as indirect secondary effects through the

water balance, productivity, and SOM decomposition submodels. Soil texture may not

have a strong influence on the tillage effect on C sequestration in these types of systems.

If the assumption that a strong relationship exists between soil texture and the effect of

tillage on soil organic carbon is not true, then the research presented here further validates

the Century model for estimating soil organic carbon dynamics in semi-arid dryland

agricultural in north central Montana.

It is hoped that by using process-based soil organic matter models, such as the

Century model, that there would be a better understanding of SOM dynamics and that

models could reduce the cost associated with monitoring and verifying C change for a

carbon-trading effort. The research used to develop Century has broadened the

knowledge of the processes and mechanisms that govern C dynamics. However, by

using Century to quantify C change for carbon trading, there is the potential for substantial

economic consequences. If the model were to underestimate the amount of carbon

sequestered during a carbon-contract period, then the farmer may not receive appropriate

compensation for their effort. Equally, if the model were to overestimate the increase of

soil carbon during a contract period, the farmer may be over-compensated and the buyer

of the carbon credit may not receive the full benefit of the C that has been purchased. To

quantify the potential economic consequences, an uncertainty and cost/benefit analysis of

both measured C change over time and modeled estimates is needed, which is beyond the

scope of this research.

Due to potential economic consequences resulting from incorrect estimations of

sequestered C, questions have arisen as to whether soil sampling for SOC should occur

both prior to, and at the conclusion of, a carbon-trading contract. Some experts claim

that soil sampling for C on a regional- or national-scale effort could potentially be cost

prohibitive from both a monetary and C emissions standpoint. Still, others argue that with

efficient sampling techniques, measuring C is a viable option and could strengthen the

validity and support for carbon contracts; thus the parties involved with such an agreement

may be more willing to continue participation and to receive benefits from a national C-

sequestration effort. Additionally, potential new technological advances with in situ SOC

measurements will likely further reduce the cost associated with field measurements.

Additional research is needed on both the effect of soil texture on C sequestration

with the adoption of no-till, and the Century model's sensitivity to soil texture on modeled

tillage effects, to determine if adjustments to the model would be necessary. This research

was not an exhaustive look at the effects of soil texture on C sequestration nor Century's

predictive capabilities as influenced by soil texture. The results presented here are for a

relatively homogenous agroecoregion in north-central Montana

Largely, it was shortcomings of the STATSGO and SSURGO soils databases

that limited the effectiveness of Century. There was no advantage to using the finer-scale

SSURGO data over the coarse-scale STATSGO data. Both databases suffered from the

same basic limitations and were equally poor at encompassing the "true" soil textures

measured in this study. Results of using either the 1:250,000 scale STATSGO or the

1:24,000 scale SSURGO data for Century soil texture and soil bulk density inputs were

equally poor for predicting carbon change due to the adoption of no-till in north central

Montana.

Shortcomings of the STATSGO and SSURGO soil databases could present

issues for developing carbon contracting units. If contracting units for C credits were to

coincide with field-scale C estimations, the confidence of C estimates using Century with

either soils database would likely not meet standards of C buyers. Field-scale sources of

soils data would be needed and methods of collection optimized to reduce costs. If C

contract units encompassed relatively homogenous argoecozones, a "representative soil"

may be defined for that particular region. This representative soil may be initially

delineated using STATSGO soil mapping units; however, sampling to test the accuracy of

the STATSGO soil attributes for the agroecozone would be needed to increase the

confidence of STATSGO soil data and soil carbon estimates.

Conclusions

Including soil clay content was an important consideration when comparing soil organic carbon content among sites; however the clay covariant was not as critical for comparisons between tillage systems. Both the STATSGO (1:250,000 scale) and the SSURGO (1:24,000 scale) soil databases did not adequately reflect the soil textures across the six sites sampled in this study. The Century model was sensitive to the effects of soil texture when predicting the amount of SOC in fields managed with and without tillage; however the model did not show that soil texture changes the amount of SOC accumulated over a 6 to 10 year period of no-till management. From a modeling standpoint, neither the STATSGO nor the SSURGO databases provided adequate field-scale soil texture information for use in the Century model, thus site-specific soil information is recommended for use with the Century model for modeling C dynamics at the field/farm level.

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EPILOGUE

General Summary

The research presented in this thesis will lend support to future measurement, prediction, and general understanding of soil organic carbon sequestration in semiarid dryland agricultural systems. Reliable measurement of soil organic carbon change associated with a shift in tillage management can be difficult, for SOC varies spatially and the degree of variability can be substantial. Confounding variables such as climate, topographic position, erosion potential, and soil texture also vary spatially and can greatly influence SOC change across small distances. A soil-landscape association method of field stratification addressed these confounding variables and reduced SOC measurement variability; however, due to natural variation in soil texture, differences in soil texture still occurred between tillage treatments at some sites. Differences in soil textures may confound determination of the tillage effect, particularly at regional and larger areas. A measure of soil texture (i.e. % clay) was added as a covariant in statistical analyses. Including percent clay as a covariant provided estimates of SOC under no-till and tilled management adjusted for differences in clay content, and reliable determination of tillage effects.

The effect of soil textural variation at field-, 1:24,000 (SSURGO), and 1:250,000 (STATSGO) scales on the predictive capability of the Century model was explored.

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Both the SSURGO and STATSGO databases were limited in their accuracy of predicting

soil textures at the sampled fields. Ranges in clay percentage reported by SSURGO

included the measured values for 3 of 10 fields and STATSGO included 5 of 10. Due to

the differences in scale, the width of the STATSGO ranges in clay % were 2 to 7 times as

wide as the SSURGO ranges. The shortcomings of these soils databases had a profound

effect on the accuracy of Century model predictions using SSURGO and STATSGO

soils data to run the model.

Both soil texture and the scale at which soil texture data were obtained had an

effect on the accuracy of soil organic carbon values predicted by the Century model.

Using field-scale soil textures and site-specific management data, Century accurately

predicted soil organic carbon at five sites in north central Montana to within an average of

10 % (range of -1 to +28%) of measured values. Largely due to shortcomings of the

SSURGO and STATSGO databases, model predictions of SOC were lower than

measured values at Chester (no-till field only) and Ft. Benton. Estimated soil organic

carbon values from Century were similar to measured values at Saint Johns and Simpson;

however the model tended to over-estimate measured SOC values at Conrad W.

Soil organic carbon estimates from Century for the management systems in this

study were sensitive to the effect of clay content based on the range of modeled soil

organic carbon values. Estimated SOC for the upper limit of reported clay content was

2.3 to 2.7 times greater than SOC estimates for the lower clay limit at the 5 sites modeled

using STATSGO database ranges in clay content. Conversely, Century was largely

insensitive to the effects of soil texture on the potential soil carbon change in response to the adoption of no-till. Century did not predict the amount of implied carbon change in response to the adoption of no-till at the 5 sites modeled in this study over a wide range in clay percentages. Model results for the five sites in Montana showed little difference in the amount of carbon stored in coarse-textured soils (5% clay) compared to fine-textured soils (35-40% clay), with the exception of the Ft. Benton site. The insensitivity of Century to a soil textural effect on C storage under no-till management assumes that a strong relationship exists between soil texture and the effect of tillage on soil organic carbon. This relationship is currently not well understood.

Additional analysis of the Century model's sensitivity to soil textural input variables is needed to determine if adjustments to the model would be necessary. This research was not an exhaustive look at the effects of soil texture on Century's predictive capabilities. Largely, it was shortcomings of the SSURGO and STATSGO soils databases that limited the effectiveness of Century. The model was sensitive to the effects of soil texture when predicting the amount of SOC in fields managed with and without tillage; however the model was not sensitive to the effects of soil texture on the ability of a particular soil texture to accumulate SOC over a 6 to 10 year period of no-till management. From a modeling standpoint, neither the SSURGO nor the STATSGO databases provided adequate soil textural information for use in the Century model, thus site-specific soil information is recommended for use with the Century model.

Looking Back

Research is a learning experience from the initial idea to well past the completion of a thesis or other publication. Reviewing the entire process of this research is as daunting a task now as the first few organizational meetings were in the beginning. Much research and progress has been made in the last few years pertaining to sampling methods and modeling techniques directed at soil organic carbon change resulting from changes in soil management. The learning curves associated with soil sampling and carbon analysis were aggressive personal climbs; additionally, overcoming the vertical challenge of learning a model as complex as the Century model was no menial task.

The soil-landscape association method of field stratification was effective at locating sites that were similar with respect to soil properties and the potential of soil organic carbon change. Excavating exploratory soil pits at each prospective microsite location proved time consuming and somewhat inefficient, costing 6 to 8 hours per site to complete, depending on the soil variability at the site. When this point of the stratification process has been reached, an exploratory soil core may prove just as effective at obtaining the necessary soil attributes needed to define the soil-landscape association, assuming that compaction is taken into consideration when measuring depth to lime. Using an exploratory core at potential microsites would translate into a more rapid turnover time for microsite selection and reduce field time by approximately 3 to 4 hours per site.

The soil organic carbon sampling variability study could have been more robust had more than two sites been used in the analysis. This portion of the thesis was a secondary objective and could have been set up to include more sites had it been more of a primary objective. The results of the study are encouraging to the development of efficient soil organic carbon sampling designs for the semi-arid dryland agriculture in Montana. In both the sampling variability study and the Century sensitivity study, the measurement of carbon in surface residues and coarse plant litter (>2 mm) in the soil would reduce the potential bias created by not including these fractions in the total carbon stock, particularly in the no-till fields.

The Century sensitivity study could have been more conclusive, 1) had more sites been soil sampled, better representing the full range of soil textures reported by STATSGO, for comparison to predicted values, and 2) if carbon change had been directly measured rather than implying the change in paired no-till / tilled comparisons. Sampling cropland that represents the full range of soil textures reported by STATSGO and SSURGO would be highly unlikely due to the efficacy of sandy soils to support crops in semi-arid dryland agriculture, and the logistics of locating paired comparisons. The difficulty of using paired comparisons lies in locating paired fields that meet the criteria necessary for good comparisons, and finding twice as many producers who are interested in participating (i.e. one producer per tillage treatment). Paired comparisons certainly measures a difference, but causality must be implied with caution, for there can be many confounding factors that were unintentionally overlooked. Long-term measurements on

experimental plots provide better treatment comparisons, however no-till farmers familiar

with university research were interested in what has happened to soil carbon since they

have adopted no-till, and paired comparisons are required to determine such a change.

Looking Ahead

Much soil carbon research has been done using small plot experimental designs.

Small plot designs are easily managed, and are run by scientists. The research using small

plots has certainly contributed much to the soil organic carbon knowledge-base. I believe

the next step is to begin involving the 'end-user' in soil carbon research. Soil carbon

research has been focused on the processes of soil carbon dynamics, and determining

potential rates of change associated with various management options. Including farmers

in an applied experimental design at a field-scale could shed light on a regional potential of

carbon storage. This proposed research would have farmers manage the experimental

plots using their equipment, and making decisions based on farm sustainability rather than

strict experimental designs. Through a grant from the Consortium for Agricultural Soils

Mitigation of Greenhouse Gases (CASMGS), this proposed research is underway in

north central Montana. A split-block experimental design is set up to test the effects of

tillage management and cropping intensity on carbon sequestration in semi-arid dryland

small-grain production. Cropping intensity will be increased by including a pulse crop of

the participating producer's choice. At the end of the study we will be able to compare

results to those found by small plot experiments and to begin making estimations of the

potential of Montana dryland agriculture to sequester carbon and help mitigate greenhouse gases.

The design of the CASMGS project offers experimental control benefits as well as additional research questions to be asked in addition to that asked in the research presented in this thesis. First, the split-block design allows for careful control of the management systems within each site as compared to the uncontrolled management comparisons in the thesis. Second, repeated measurements over time will allow for a 'true' determination of carbon change due to no-till rather than an implied change, and will provide a better comparison to Century model estimates. The addition of pulse crops in rotation will shed light into the plausibility of increasing both cropping intensity and soil carbon storage in a semi-arid environment. Pulse crops also provide additional soil nitrogen through N-fixation which may reduce fertilizer requirements yet maintain both crop productivity and provide the nitrogen required for soil organic matter formation. Lastly, nitrous oxide emissions will be compared among treatments. This will allow measurement of greenhouse gas "leakage" from the system for a net greenhouse gas accounting. Benefits of the research possibilities offered in the CASMGS project will be realized for many years to come.

Coupled with the CASMGS project is another avenue of soil carbon research that is just beginning to surface. Remote sensing has been used for many applications ranging from forestry management to land cover and land use change detection. Remote sensing products, in particular satellite imageries, have the potential to monitor and verify

carbon sequestering practices. For example, Landsat ETM+ satellite imagery and logistic regression have been used to document no-till and conventional till management in north central Montana (Bricklemyer et al. 2002). Satellite imagery may provide cost efficient means of estimating soil, crop, and management inputs for use in the Century model as well as providing verification that certain management practices have or have not been used on particular fields.

The topic of soil texture effects on C sequestration with the adoption of no-till has not been resolved. Research in southwestern Saskatchewan supports that there is a strong effect of soil texture on carbon gains under long-term no-till, whereas research across a larger area of the Canadian prairies and the U.S. Great Plains has not observed a strong, meaningful relationship of C storage and clay content. This issue should be researched in more detail. The research may entail measurements in a paired design similar to that used in this thesis research and/or a rigorous small plot study using amended soils encompassing a wide range of soil textures. The effect of soil texture on C storage with the elimination of tillage may be a regional effect that can be tailored into the treatment of soil textures in the Century model.

In the interest of a full accounting of the potential of agricultural systems to sequester carbon and help mitigate the greenhouse effect, organic systems should be explored. The number of acres managed as organic systems are growing every year and some organic producers are claiming to increase soil organic matter at a rapid rate. Implementation of the bench mark sampling design in fields recently converted to organic

management would begin to look at carbon dynamics under these intensively managed systems. Additionally, there has been some discussion about inorganic carbon increases through biochemical processes below the tillage layer in tillage-based systems; this should be explored further to determine if the same processes occur in no-till systems.

Technology is rapidly evolving in our world. As soil scientists, we should embrace technology and it's potential to help meet our needs. Faster, more accurate, and efficient means of measuring soil carbon are needed. Today we have machines that can measure wheat protein "on-the-go" to help in the precision agriculture effort. "On-the-go" quantitative soil carbon measurements could propel the fledgling carbon-credit market. Additionally, probability based statistical designs could be developed to target soil sampling in carbon contract units.

Equally as import as the potential of carbon sequestration in agricultural systems to reduce greenhouse gases to help mitigate the greenhouse effect are the "ancillary" benefits of agricultural best management practices. These "ancillary" benefits include reduced soil erosion, increased productivity, and increased biodiversity in agoecosystems. The elimination of tillage, better stubble management, increased cropping intensity, and diverse crop rotations have the potential to conserve our soil resources and propel us toward a more sustainable agricultural

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APPENDICES

APPENDIX A

ANNUAL SITE PRECIPITATION

Annual Precipitation for Chester, Montana
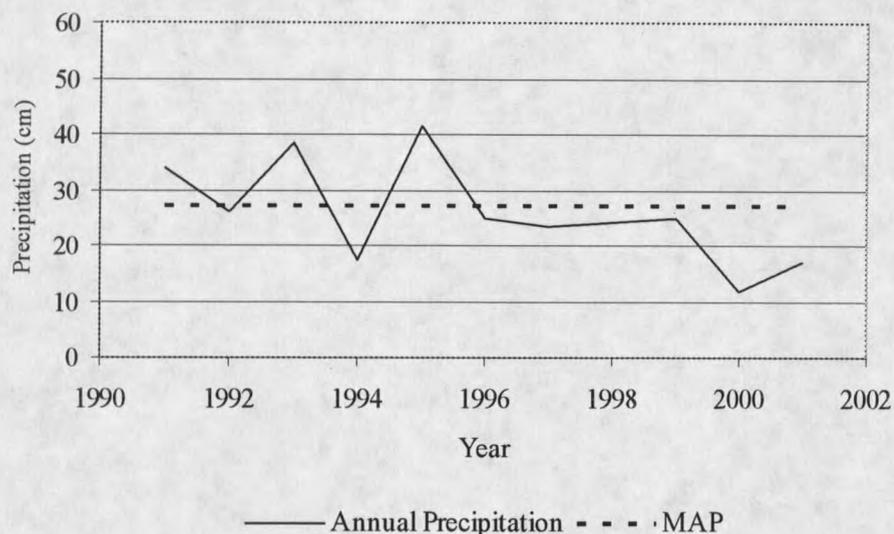

————— Annual Precipitation - - - - ·MAP

Figure A.1. Annual precipitation (1991-2001) as compared to mean annual precipitation (MAP) for weather station nearest the Chester site. Data source: National Climate Data Center (NCDC) Historical listing of National Weather Service Cooperative Network.
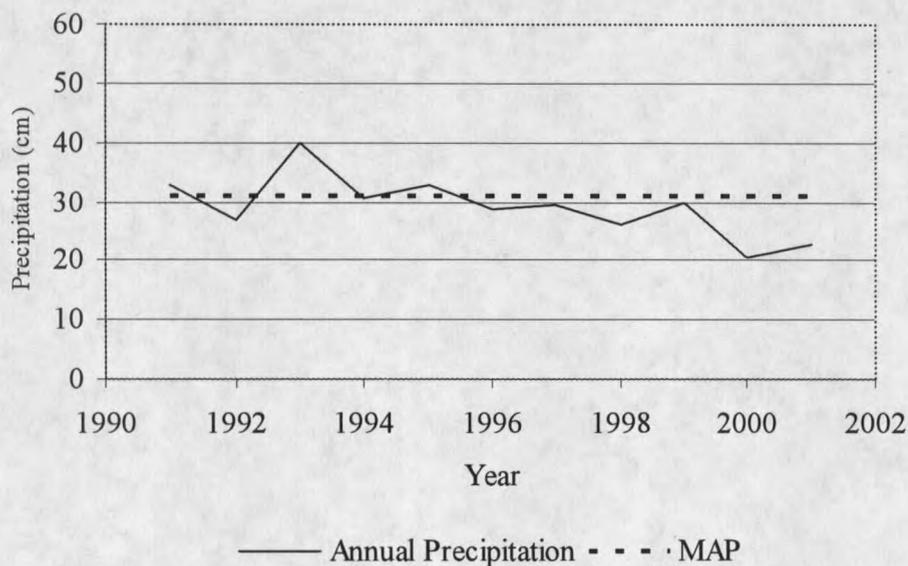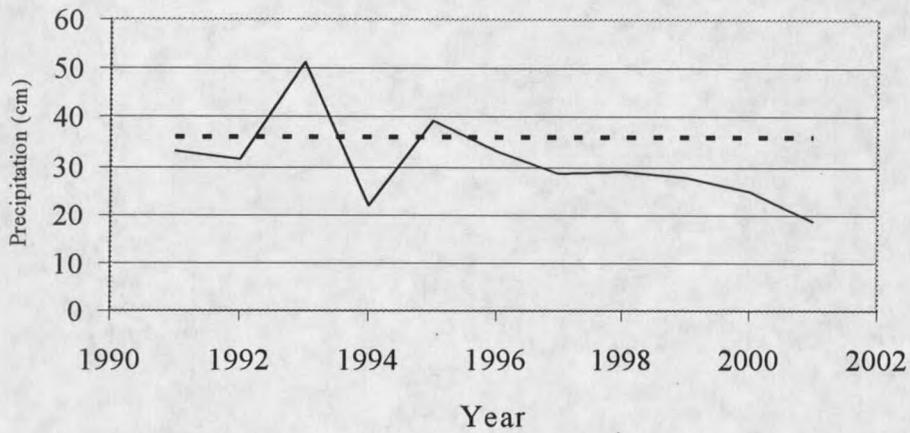
Annual Precipitation for Conrad, Montana

————— Annual Precipitation - - - - ·MAP

Figure A.2. Annual precipitation (1991-2001) as compared to mean annual precipitation (MAP) for weather station nearest the Conrad E and Conrad W sites. Data source: National Climate Data Center (NCDC) Historical listing of National Weather Service Cooperative Network.
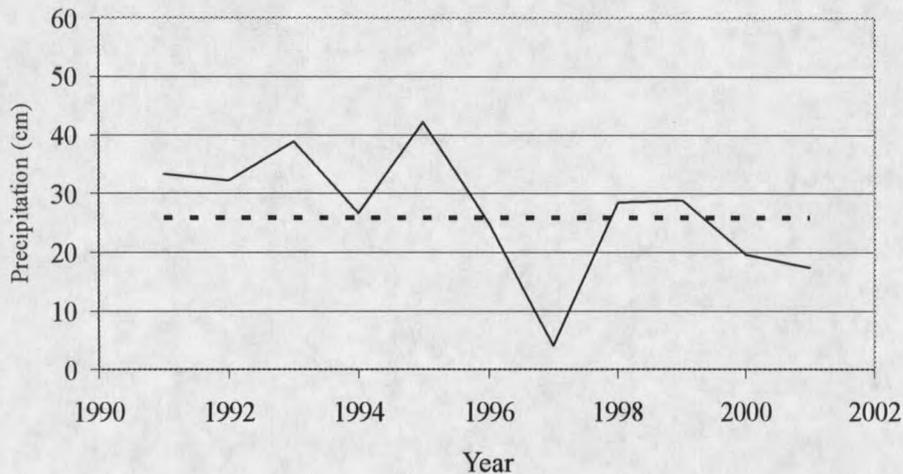
Annual Precipitation for Ft. Benton, Montana

———— Annual Precipitation - - - ·MAP

Figure A.3. Annual precipitation (1991-2001) as compared to mean annual precipitation (MAP) for weather station nearest the Ft. Benton site. Data source: National Climate Data Center (NCDC) Historical listing of National Weather Service Cooperative Network.

Annual Precipitation for Simpson and St. Johns, Montana

———— Annual Precipitation - - - ·MAP

Figure A.4. Annual precipitation (1991-2001) as compared to mean annual precipitation (MAP) for weather station nearest the Simpson and St. Johns sites. Data source: National Climate Data Center (NCDC) Historical listing of National Weather Service Cooperative Network.
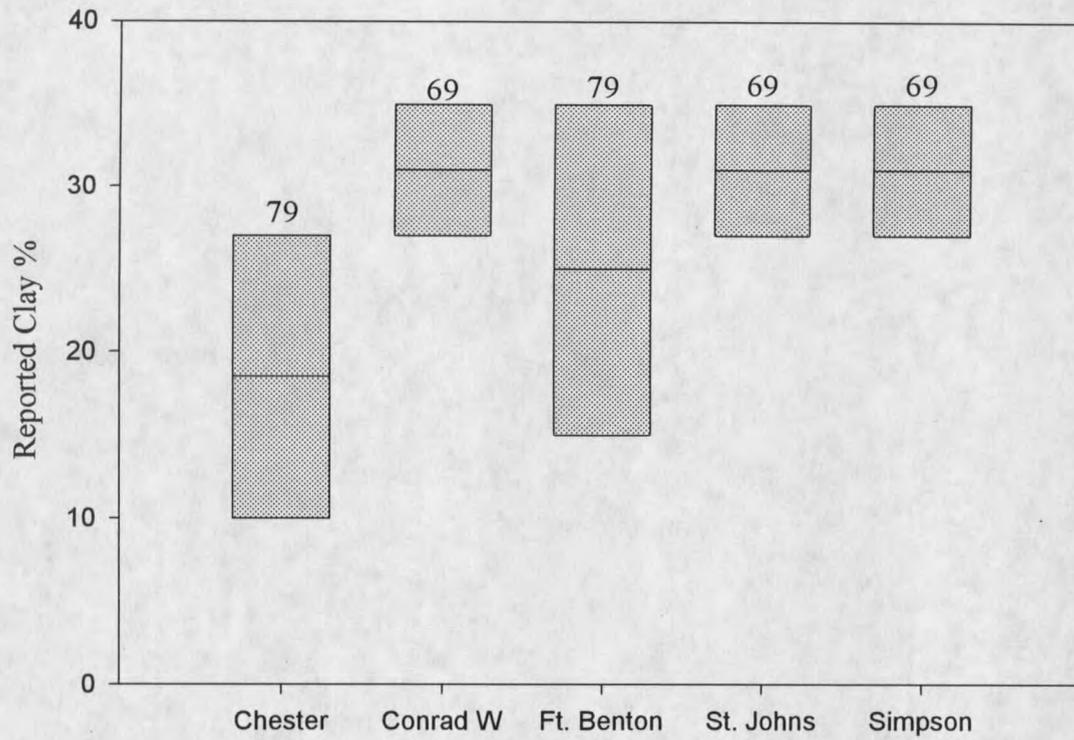
APPENDIX B

STATSGO SOIL COMPOSITION

Figure B.1. STATSGO reported clay % corresponding to the three most prevalent soil series at Chester, Conrad W, Ft. Benton, St. Johns, and Simpson, Montana. Values at the top of boxes indicate the % composition of soils within each mapping unit that are included in the afore mentioned range of clay %. Lines within boxes represent mean values.