THESIS

CONSTRAINTS AND CAPABILITIES OF NO-TILL DRYLAND AGROECOSYSTEMS FOR BIOENERGY PRODUCTION

Submitted by

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In partial fulfillment of the requirements

For the Degree of Master of Science

Colorado State University

Fort Collins, Colorado

Summer 2011

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ABSTRACT

CONSTRAINTS AND CAPABILITIES OF NO-TILL DRYLAND AGROECOSYSTEMS AS BIOENERGY PRODUCTION SYSTEMS

Crop residues are receiving attention as potential feedstocks for lignocellulosic biofuels. Sustainable residue harvest may be limited by soil erosion and the need to maintain soil organic carbon (SOC). Little attention has been given to the potential for residue harvest in the semi-arid Great Plains, largely due to assumptions of low production levels and the strong erosive forces of wind. Due to expanding interest in growing dedicated biofuel crops on marginal lands, these studies examined the capabilities and constraints of harvesting agricultural residues from dryland production systems in the semi-arid Great Plains.

The first study examined long-term production levels of grain and stover for wheat (*Triticum aestivum*), corn (*Zea mays*), and grain sorghum (*Sorghum bicolor*) at three no-till dryland cropping sites in wheat-corn-fallow and wheat-sorghum-fallow rotations, and evaluated the impact of stover removal on wind and water erosion, soil organic carbon dynamics, and future productivity. The Revised Universal Soil Loss Equation and the Wind Erosion Equation were used to simulate water and wind erosion under various levels of residue removal. The DAYCENT model was used to estimate changes in soil organic carbon, grain yields, and soil fertility, if 50% of corn and wheat stover were harvested each crop year. Model validation was performed by comparing long-term production rates and measured changes in soil organic carbon to model simulated output. Total aboveground biomass production for corn and sorghum, averaged over site and soil type, was 5550 ± 2810 kg ha^{-1} yr^{-1}, with average stover production of 2750 ± 1570 kg ha^{-1} yr^{-1} and 2800 ± 1570 kg ha^{-1} yr^{-1} for grain. The total aboveground annual biomass production of wheat across all sites averaged 5840 ± 2440 kg ha^{-1}. Wheat annual stover yields were 3940 ±1880 kg ha^{-1} and grain yields averaged 1950 ± 820 kg ha^{-1}. A 50% stover removal rate only slightly increased water erosion from 0.53 Mg ha^{-1} yr^{-1} (no removal) to a maximum of 1.4 Mg ha^{-1} yr^{-1}. Wind erosion was a bigger risk, with rates surpassing the tolerable erosion levels after removing 10 - 30% of corn stover, depending on site and soil landscape position. However, at all three sites, up to 80% of wheat straw could be harvested without surpassing tolerable erosion rates. Soil organic carbon (SOC) declined 6-9% after 96 years of simulating 50% removal of corn and wheat stover. Under 0% removal, SOC levels appeared relatively stable, with maximum declines of 2.0%. As SOC levels are very low in these dryland systems, these declines represent a very small net loss of SOC when compared to losses observed in more humid regions.

Under current wheat-corn-fallow management, virtually all stover must remain in order to control wind erosion and maintain soil organic carbon. However, if dedicated non-grain bioenergy crops were grown on an annual basis, there could be 2500-2700 kg ha^{-1} of harvestable biomass yearly while still retaining enough residues to maintain SOC. Total biomass production of a dedicated non-grain energy crop could be higher than the biomass production of the grain crops examined, namely because energy is not diverted to grain, and non-grain crops are not as sensitive to the timing of water deficits.

Replacement of the fallow period with a non-grain biomass crop could lower the amount of residue needed to control erosion. Elimination of the fallow period would likely reduce the amount of residue that must remain to maintain SOC, increasing the amount of biomass available for removal.

The second study uses the DAYCENT model to simulate variable responses in fertility, yield, and soil organic carbon within two field landscapes in eastern Colorado. Grain yield, soil fertility, and soil carbon were all impacted by stover harvest, but the magnitude and direction of responses were dependent on soil type. Yield declines as great as 1615 and 1382 kg ha^{-1} were simulated for corn and wheat, respectively. Declines in annual mineralization rates as high as 13 g N m^{-2} yr^{-1} were observed with stover harvest, but simulated changes in mineralization rates were highly variable between soil types, with net mineralization rates increasing with stover removal in some years. The impact of stover harvest on soil organic carbon varied with soil type and landscape position. Results are used to highlight the variable impacts stover harvest could have within one field or management unit, and demonstrate the need for landscape scale predictive models to assess the impact of stover removal on soil fertility and SOM dynamics and transfer processes models. Simulations characterized by an average soil type are not sufficient to account for the complexity of soils or the interactions and feedbacks of sediment, nutrient, and water transport that occur within agricultural fields. In addition to predictive model support systems, management of soil-specific responses to stover harvest will likely require adoption of precision agriculture technologies and practices such as variable rate harvesting and fertilization.

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CHAPTER 1: Constraints and Capabilities of No-Till Dryland Agroecosystems for Bioenergy Production

INTRODUCTION

As U.S. interest in biomass-based renewable energy rises, efforts are needed to evaluate the impacts of bioenergy on land use, water, and total agricultural production. Ensuring that the development of the biofuels industry does not impede future productivity of our soils and reduce ecosystem services is one of the great challenges before us. Because of the limitations and issues surrounding the sustainability of using food-based crops as energy feedstocks (Farrell et al., 2006), national legislation has been passed to promote the use of cellulosic materials such as perennial grasses (e.g., switchgrass, miscanthus), agricultural residues (e.g., corn stover, wheat straw), short-rotation woody crops, residues from forest harvesting, and other sources of lignocellulosic material (e.g., industrial waste streams) as feedstocks for ethanol production (Sissine, 2007). As the cellulosic energy platform emerges, consideration of regional and site-specific effects on land use, and soil and water resources is needed (Johnson, 2007).

Globally, the largest amount of grain production comes from corn, with approximately 820 Gg of grain, and similar amounts of stover produced annually (Somerville et al., 2010). Corn and other agricultural crop residues are being examined for their potential as a portion of the biomass energy base. Crop residues are the non-grain portion of a crop that remains in the field after grain harvest. However, crop residues, such as corn stover, play important roles in agricultural ecosystems and removal of residues for biofuel could result in removal of nutrients, increased erosion, reduced fallow efficiency, and losses of soil carbon (Graham et al., 2007; Johnson, 2007; Wilhelm et al., 2007; Wilhelm et al., 2004). Surface crop residues are a direct source of crop nutrients and continuous removal can result in depletion of soil fertility (Blanco-Canqui and Lal, 2009) and increased levels of sediment and nutrient runoff (Blanco-Canqui et al., 2009; Wilson et al., 2004). Studies have reported yield declines with stover removal, with results varying based on management, climate, and soil type (Wilhelm et al., 2004). While the degree of change in nutrient pools following stover removal depends on the amount of stover removed and site-specific characteristics, several studies have shown that stover removal can have negative impacts on soil properties and crop production (Blanco-Canqui and Lal, 2007; Blanco-Canqui et al., 2006). Blanco-Canqui and Lal (2009) hypothesized that the amount of stover required to maintain essential nutrients is higher than that required to control erosion. In areas where soil erosion is problematic, residues protect the soil surface from wind and raindrop impact. Water-limited regions need residues to help capture and store water, moderate soil temperature, and maintain infiltration rates (Peterson and Westfall, 2004; Unger, 1978).

Soil carbon maintenance may ultimately be the limiting factor to sustainable harvest rates. Soil organic carbon (SOC) is one measure of overall soil quality and productivity (Shukla et al., 2006). Several studies and model simulations suggest a linear relationship between the amount of residue returned to the soil and SOC levels. (Parton and Rasmussen, 1994; Paustian 1997). In many agricultural systems, the amount of crop residue needed to maintain and build SOC is greater than that needed to control water and wind erosion (Wilhelm et al., 2007). Although limiting, there are scenarios that may balance SOC maintenance and residue removal (Sheehan et al., 2003). The conversion of biomass to SOC is a complex process influenced by a variety of factors dependent on regional climate, soils, and precipitation rates. Management practices, crop rotations, and annual yields will also influence residue harvest rates.

Several studies have estimated the quantities of agricultural biomass potentially available for bioenergy feedstocks in the United States with limits set to control erosion and maintain soil organic carbon (Graham et al., 2007; Nelson et al., 2004; Perlack et al., 2005). Little attention has been given to the semi-arid Great Plains region because of the relatively low production levels, water limitations, and erosion risks. However, the Great Plains region could be called upon to play a role in feedstock production because of the extensive production of crops like corn, wheat, and sorghum that are being considered for use as biofuel feedstocks (Sarath et al., 2008).

Semi-arid lands make up approximately 17.7% of global terrestrial surface, receive between 200-800 mm of annual precipitation, and have an aridity index (Precipitation/Potential Evapotranspiration) of 0.20-0.50 (Lal, 2004). Semi-arid regions are prone to drought and erosion, making bioenergy crop production a tenuous undertaking. However, water-limited regions could potentially be used to grow bioenergy crops with drought resistance and high water use efficiencies (Somerville et al., 2010). The amount of water required to grow bioenergy crops is of increasing concern, with some research suggesting that large-scale development of biomass energy could exacerbate water shortages around the globe (Mulder et al., 2010). The water footprint of bioenergy crops is highly variable, and dependent on the production system and climatic conditions (Dominguez-Faus et al., 2009). Production of bioenergy crops with high water demands, or irrigated crop production, could have unintended impacts on water resources. Thermo chemical conversion methods for cellulosic biomass conversion allow room for diverse feedstocks with reduced water requirements (Stone et al., 2010). Marginal lands with suboptimal water availability or soil quality may be used to grow a proportion of biomass crops in order to minimize negative impacts on food and feed (Johnson, 2007).

Dryland agricultural systems do not utilize limited supplies of irrigation water. Although dryland systems have lower production than irrigated systems, low land costs mean that dryland systems can produce less, yet still remain profitable. Additionally, sustainable biomass systems must minimize emission from land-use or management change (Searchinger et al., 2008). Semi-arid agrictulrual systems are carbon-poor due to low precipitation and production, as well as historic practices that have depleted carbon, and hence will not trigger large carbon emissions from management changes. Because of the extensive land area dedicated to dryland cropping in the semi-arid Great Plains, the potential of biomass production on these lands needs to be considered.

Unique roles of crop residues in dryland agricultural systems

Water capture and fallow efficiency

Areas of the Great Plains with limited precipitation are susceptible to low crop yields or total crop failures in years of erratic or low precipitation. Crop residues help promote water capture and conservation in water-limited systems. Many precipitation events are high intensity storms that exceed the soils' water infiltration capacity, resulting in water ponding and runoff. Soils are particularly susceptible to runoff if no soil cover exists due to the formation of surface crusts. A residue base can reduce water lost via water runoff by increasing hydraulic roughness, increasing the time to runoff initiation, reducing runoff velocity, obstructing runoff, and reducing velocity, thereby increasing potential infiltration rates (Alberts and Neibling, 1994). Crop residues can also have a positive impact on surface soil properties such as bulk density, porosity, sorptivity, and aggregation, that influence the potential for water infiltration and capture (Shaver et al., 2002). Shaver et al. (2003) found that for each 1000 kg ha^{-1} of residue addition, the bulk density of the surface 0-2.5 cm of soil was reduced by 0.01 g cm^{-3} and effective porosity was increased by 0.3%. Macro aggregation in the surface 2.5 cm of soil was also increased when greater amounts of residue biomass were returned to the soil surface.

Water capture is also important in increasing the amount of water stored during the fallow period. The practice of summer fallow was developed to decrease the risk of crop failure by increasing the probability of a successful wheat crop. In addition to increasing infiltration by preventing soil sealing, residues act as mulch to decrease evaporation from the soil surface after a rainfall event (Greb et al., 1970). Crop residues can help decrease soil temperatures and soil water loss, and maintain infiltration rates, thereby increasing fallow water storage (Peterson and Westfall, 2004). Standing crop residues reduce wind speed near the soil surface, and can therefore trap snow, an important fraction of the precipitation budget for the Central and Northern Great

Plains (Nielsen et al., 2005). Low productivity, coupled with rapid decay rates, often limits the amount of residue on the soil surface in dryland systems (Peterson and Westfall, 2004). Residue harvest could result in a reduction in fallow efficiency, water capture, and water storage.

Wind erosion

Residues are also needed to control wind erosion, the primary erosive force in the Great Plains. In areas where soil erosion is problematic, residues protect the soil surface from wind and also aid in formation of larger, more stable aggregates that are less vulnerable to wind erosion. Topsoil rich in organic matter is easily lost via erosion, decreasing system productivity. Factors affecting erosion are a function of climate, soil properties, topography, soil surface conditions, and management. Soil erosion is a common problem in the Great Plains, and can occur under even the best management regimes. Agricultural lands exhibit wide variations in soil types, soil erodibility, field slope and length, and climatic conditions. The Natural Resources Conservation Service (NRCS) of the United States Department of Agriculture (USDA) has established tolerable soil loss limits (referred to as T-values) on a county basis for all soil types in the United States. Soil loss tolerance values delineate the maximum rate of annual soil erosion that can occur with little or no long-term degradation of soil resources and/or loss of productivity. Factors considered in establishing soil-loss tolerance values include soil depth, physical properties, reduction of soil organic matter, loss of plant nutrients, and other considerations. Agreement on what is a tolerable level of soil loss is not a trivial matter and has been debated for decades (Sheehan et al., 2003). As noted by Mann et al., (2002), adequate erosion protection is not necessarily met by meeting the T-values used for erosion tolerance. However, the need to maintain adequate soil cover to control wind erosion is a constraint to stover collection (Nelson, 2002; Nelson et al., 2004).

No-till management has permitted successful cropping intensification in the semi-arid Great Plains by reducing the amount of summer fallow, resulting in higher levels of crop residue production than the traditional wheat-fallow system. Increased residue cover provides greater wind erosion control. For example, after a 12-year cropping period at three no-till dryland sites in eastern Colorado, average total residue levels in a more intense wheat-corn-fallow system were 32% higher than in the wheat-fallow system, and were generally adequate to control wind erosion (Cantero-Martinez et al., 2006). By comparison, in the wheat-fallow system, there was a negative trend in total residue amounts on the soil, demonstrating that even under no-till conditions, the traditional W-F management practice does not produce and retain enough biomass. While the amount of residue needed to control erosion and conserve water in semi-arid systems will vary based on climatic patterns, soil type, slope, and method of management, it is important to determine a target amount of residue to leave on the soil surface for wind erosion control The objectives of this study were: (i) to evaluate the long-term biomass production potential in no-till dryland cropping systems in the West Central Great Plains using wheat, corn and sorghum grain as model crops, (ii) to determine water and wind erosion constraints to residue harvest, and (iii) to examine the impact of stover removal on soil carbon dynamics and soil fertility. This study is based on results from a long-term field study of dryland cropping systems in the semi-arid Great Plains. The capabilities and constraints associated with growing crops on semi-arid lands were used as a framework to discuss the possibility of growing a hypothetical dedicated annual bioenergy crop in these systems.

MATERIALS AND METHODS

Overview of the Long-Term Study

The data used in this study are from 24 years of research conducted at three sites in the West Central Great Plains of eastern Colorado, for a total of 72 sites years. This study began in 1985 to identify sustainable no-till dryland cropping systems with improved precipitation use efficiency (Peterson et al., 1993). The experiment includes three variables: (1) climatic gradient, represented by site differences in potential evapotranspiration (PET), (2) soil type across landscape position, and (3) cropping intensity. The three sites in eastern Colorado are located near Sterling (40.37°N, 103.13° W), Stratton (39.18°N, 102.26°W), and Walsh (37.23°N, 102.14°W). An automated weather station at each site measured daily maximum and minimum air temperature, precipitation, mean relative humidity, total solar radiation, wind direction, and wind speed.

The study sites run along a north to south climatic gradient of increasing PET in eastern Colorado. Selected climatic properties, location, and elevation are shown in Table 1. All sites have approximately the same long-term annual precipitation, but increase in PET from north to south. The difference between annual precipitation and open pan evaporation represents the water deficit.

Along with climatic differences, each site has unique topography. Each site is laid out on a soil catena, or a sequence of different soil profiles that occur down a slope. Three distinct soil profiles exist at each site, consisting of a summit, side, and toe soil position. Soils within the catena differ due to the effects of topography on water movement, and each landscape position is comprised by a unique soil mapping unit with distinct soil textural classes. Soil classifications, as well as physical and hydraulic properties for the top two soil horizons by site and landscape position, are given in Table 2. Reported saturated hydraulic conductivity values were calculated using a pedotransfer function (Schaap et al., 2001). Site differences in slope and elevation create complicated water, erosion, and deposition patterns. Toeslope landscape positions generally have higher levels of carbon and nitrogen than summit and sideslope positions, with the Stratton toeslope containing approximately two times the total soil carbon and nitrogen than toeslope positions at Sterling and Walsh. This is largely due to sediment deposition and run-on water. Full details of the unique topography of each site can be found in Ascough et al., (2010).

Experimental Design

This experiment is a split-split-block design. Site location is the first block, and landscape position the sub-block. The experiment includes multiple cropping systems, all managed with no-till techniques. The cropping systems represent a continuum of increasing cropping intensity that decreases the frequency of summer fallow. Each cropping system phase was assigned randomly in strips within each block and extended over a topo-sequence of soils. All phases of each rotation were present in two replications each year. For example, in aWCF, every phase of the rotation (wheat, corn, and fallow phases) is replicated twice at each site, making a total of 6 plots for this system. Each experimental unit is an individual soil within a site and within a cropping system. All experimental units are 6.1 m wide, but vary in length between particular sites (185 to 305 m). Although a variety of cropping systems were evaluated in the long-term study, only the wheat-corn-fallow (WCF) and wheat-sorghum-fallow (WSF) cropping systems were examined in this study.

Biomass Production

Corn, sorghum, and winter wheat were evaluated for their long-term production of grain and stover. These crops were included in the long-term dryland cropping system study as grain crops, not biomass crops. Even though other annual species may be promising as biomass crops, the current evaluation illustrates the potential and limitations of biomass production in semi-arid environments using these three grain crops as models.

All systems in the long-term dryland cropping system study were managed with no-till techniques. Herbicides were chosen to permit crop rotations with no carryover problems (i.e., toxicity to the next crop). Nitrogen fertilizer was applied annually at planting in accordance with soil tests obtained from each soil and the crop present in a given year. Phosphorus was band-applied at planting of all crops at a rate of approximately 9.5 kg ha $^{-1}$. Total aboveground biomass samples were collected by hand, oven dried, separated into grain and stover (all harvested non-grain biomass), and weighed. Harvest index (dry mass grain yield/total aboveground biomass) was determined from the hand samples. In addition, grain yields were measured with a plot combine by harvesting an area approximately 1.2 m x 30 m in the center of each experimental unit. Grain samples were collected from the combine for each experimental unit to determine moisture content and test weight. The total biomass values were obtained by dividing grain yields from the combine by the ratio of grain mass to stover mass found in the hand-collected biomass sample.

On some site-years, there were no measured biomass yields. For example, in site-years where extreme drought resulted in minimal grain yield, harvest was not performed and consequently no biomass data was collected. The reported biomass statistics for corn and sorghum do not include data from 2002 at Stratton and Sterling or 2008 at Walsh. No wheat biomass yields were measured at Stratton in 1987 and 2004 or at Walsh in 1988, 1996, 2002, and 2008.

Analysis of Variance (ANOVA) was done to determine the effects of site and soil position on stover, grain, and total biomass production. The general linear model (GLM) of the Statistical Analysis System (SAS version 9.2, SAS Institute Inc., 2009) was used for tests of main effects. Differences were recognized as significant at the $P < 0.05$ level. If site or slope were found to be significant, mean separation was done using Fisher’s Protected Least Significant Difference (LSD).

Water Erosion Modeling

Water erosion modeling was performed using the Revised Universal Soil Loss Equation (RUSLE2; version 1.26.6.4), an erosion model designed to predict long-term average annual soil losses (Mg ha^{-1} yr^{-1}) carried by runoff from a field with particular soil type, slope, cropping systems, and management practices (Renard, 1997). The RUSLE model is not designed to predict the erosion associated with single storm events, but rather predicts the long-term average annual soil loss from sheet and rill erosion. RUSLE is expressed as:

$$A = R \bullet K \bullet L \bullet S \bullet C \bullet P \qquad [1]$$

Where:

A = the computed spatial average soil loss and temporal soil loss per unit area (expressed in the units selected for K and for the period selected for *R*)

R = the rainfall-runoff erosivity factor – the rainfall erosion index plus a factor for any significant runoff from snowmelt

K = the soil erodibility factor – the soil loss rate per erosion index unit for a specified soil as measured on a standard plot, which is defined as a 72.6-ft length under identical conditions,

L= the slope-length factor – the ratio of soil loss from the field slope length to soil loss from a 72.6 ft length under identical conditions

S = the slope steepness factor, the ratio of soil loss from the field slope gradient to soil loss from a 9% slope under otherwise identical conditions

C = the cover-management factor – the ratio of soil loss from an area with specified cover and management to soil loss from an identical area in tilled continuous fallow, and

P = the support practice factor – the ratio of soil loss with a support practice such as contouring, strip-cropping, or terracing to soil loss with straight-row farming up and down the slope.

Modeling at each site was performed separately for each soil position so that landscape specific topography and soil types could be used as parameters in the model. Appropriate supporting database sets for climate, soils, and crop management zones were used, and can be found in Table 3. Management inputs and operations for a 3-year no-till cropping rotation of WCF (WSF at Walsh) were created (Table 4). The WSF simulation management input file was nearly identical to the WCF rotation, save that grain sorghum was planted in early June and harvested the first of October. In RUSLE, the user specifies what the average crop yields are during the simulation period. Long-term (24 year) averages were used as inputs at each soil position (see Tables 5 and 6). A 1% slope was used for the summit and toe positions at each site; a 4% slope was used for the sideslope position at Sterling and Stratton locations, and a 3% slope for the sideslope position at Walsh. Slope length for each soil position was set at 91.5 m (300 ft). To simulate the impact of residue removal on water erosion, a baling operation simulating 50% removal of stover was added at harvest in the management files.

Wind Erosion Modeling

The USDA's wind erosion equation (WEQ) was used to model wind erosion (Skidmore et al., 1970, 1979). The model is designed to predict long-term average annual soil losses from a field having specific characteristics. WEQ does not account for snow cover, seasonal changes in soil erodibility, or provide estimates of erosion from single storm events. The equation predicts the potential average annual soil loss (E) as a function of several variables:

$$E = f\,(I, K, C, L, V) \qquad [2]$$

Where:

I = the wind erodibility index, gauging soil susceptibility to detachment and transport by wind.

K = the soil ridge-roughness factor. K is dependent on the conditions of the field surface at a particular time, and is a measure of the effect of ridges and cloddiness made by tillage and planting implements.

C = the climactic factor. C represents the amount of erosive wind energy present at a particular location, and is expressed as a percentage of the C factor for Garden City, Kansas.

L = a function of wind direction, field length, and field width. L is the unsheltered distance wind travels across a field.

V = the vegetative cover factor, and is a measure of the cover provided by agricultural crops and residue. The level of protective cover is dependent on the type of crop and the orientation of the residues (standing or flat).

Long-term harvest yields across soil positions were used as model production inputs for each site. No-till operations were used as management inputs for all three sites. County level weather data built into the WEQ model was used as an abiotic driver. Weather location files used were from Akron, CO for the Sterling site, Burlington, CO for the Stratton site, and La Junta, CO for the Walsh site.

In WEQ, soils are placed into one of eight Wind Erodibility Groups (WEGs) based on the predominant soil textural class of the surface layer and the percent of dry soil aggregates >0.84 mm. The 4L WEG consists of calcareous loams, calcareous silt loams, calcareous clay loams, and calcareous silty clay loams. A Wind Erodibility Group (WEG) of 4L and a Wind Erodibility Index (I) of 86 was used for all three sites. Field width was set to 610 meters (2000 ft) with a ratio of 1.0 for the length/width of field. A climate factor of 80 was used at Sterling and Stratton. At

Walsh, the climate factor was 120, reflective of the more erosive winds at this southern location. To simulate the impacts of harvesting residues, a baling operation was added to the management plan immediately after harvest. The percent stover removal for each baling operation was varied from 0 - 100%.

Tolerable soil loss limits for the three sites were obtained from the Natural Resources Conservation Services' Web Soil Survey (http://websoilsurvey.nrcs.usda.gov). The Sterling and Stratton sites both have T values of 11 Mg ha^{-1} yr^{-1}. The tolerable level of annual erosion is less at Walsh, where the T-value is 9 Mg ha^{-1} yr^{-1}.

DAYCENT Modeling

The DAYCENT model was used to estimate changes in grain yield, soil fertility, and SOC with 50% residue harvest on a soil specific basis (three soil types at each experimental site). DAYCENT is the daily-time-step version of the CENTURY model. DAYCENT and CENTURY are generalized ecosystem models that simulate carbon, nitrogen, and phosphorus dynamics in grassland, forest, savanna, and agricultural ecosystems. The DAYCENT models decomposition, nutrient flows, soil water, and soil temperature are on a finer time scale than CENTURY, and has increased spatial resolution for soil layers. Both CENTURY and DAYCENT have been extensively tested in agricultural, grassland, and forest systems using observed data on soil organic matter, production levels, nutrient cycling, and trace gas fluxes (Del Grosso et al., 2002a; Del Grosso et al., 2002b; Kelly et al., 1997; Parton et al., 1994; Parton et al., 1998; Paustian et al., 1992; Stehfest et al., 2007).

Model Description

DAYCENT operates on a daily time step for nutrient cycling, water flow, and soil organic matter turnover, but uses a weekly time step submodel for plant production. Soil organic matter, total soil carbon, nitrogen (N), and N mineralization rates are calculated for the 0-20 cm soil depth. The soil organic matter (SOM) submodel computes the cycling of above- and below-ground dead plant material into active, slow, and passive soil organic matter pools. A SOM submodel estimates nutrient mineralization rates, which are dependent on dead plant material decomposition rates and turnover rates of soil organic matter pools. First order decomposition rate constants are modified by soil temperature and moisture, tillage management practices, soil texture, and litter quality. Potential plant production and nutrient uptake are calculated as a function of soil water stress, leaf area index, soil temperature, and incoming solar radiation. Actual plant production is limited according to soil nutrient availability. DAYCENT's land surface submodel simulates water flow and evapotranspiration for the plant canopy, litter, and soil profile, along with soil temperature (Parton et al., 1998). Full model descriptions can be found in Parton et al., (1987, 1994).

Model set-up

DAYCENT uses historical schedule files, generated by the user, along with soil and weather data, to predict crop yields. Detailed soil input variables include texture, bulk density, field capacity, wilting point, pH, and saturated hydraulic conductivity on a soil layer basis. Daily precipitation and daily maximum and minimum temperatures are used as weather driver inputs. Detailed soil files for each landscape position (summit, side, and toe) at each site were created from NRCS primary characterization and are available in Tables 1-6 of Appendix A.

To initialize the SOM and nutrient pools, two simulation periods were run prior to the start of the long-term experiment in 1985 to initialize SOC pools. From model year zero through model year 1934, a grazed native grassland system of 50% warm season grasses and 50% cool season grasses was simulated. For the 50-year period from 1935-1985, a winter wheat-fallow system was modeled using varieties with low to medium harvest indices and low levels of nitrogen inputs, simulating conventional practices in the region for that time period. In 1985, the

DAYCENT crop and vegetation files were changed to reflect the shift to no-till management. Long-term site specific records of annual planting and harvest dates were used, along with historical site-specific annual fertilizer rates. Historical fertilizer rates ranged from 2.2 - 11.3 g N m^{-2}, and were based on soil tests and expected crop yields. Planting dates ranged from September 14 to October 2 for wheat, with a median planting date of September 23. Harvest dates for wheat ranged from July 5 to July 15, with a median harvest date of July 12. Corn planting dates ranged from May 4 to June 1, with a median planting date of May 16th. Corn harvest dates ranged from October 7 to October 29, with a median harvest date of October 20.

For the spin-up and base periods, one hundred years of repeating daily climatic data from the Vegetation/Ecosystem Modeling and Analysis Project (VEMAP; resolution 0.5 degrees latitude x 0.5 degrees longitude) and the DAYMET model (resolution 1 km^2) was used from the centroid of the counties of the experimental sites (Kittel et al., 2004; Thornton et al., 1997). For the period from 1985-2008, climatic data collected from the on-site weather stations at each experimental location was used. As wind erosion was determined to be a concern at Walsh even without stover harvest, thus eliminating the possibility of harvesting any stover sustainably, we did not simulate the impacts of harvesting stover on soil carbon at this site.

No model equations were adjusted, and non-site specific parameters were left unchanged, save for one crop parameter (HIMAX) that controls the maximum harvest index,which was adjusted to better represent long-term average observed harvest index values. Otherwise, parameter constants controlling crop growth were left unchanged, as the model has been tested in dryland conditions with wheat and corn (Del Grosso et al., 2002a). Crop files with parameter values can be found in Tables 7-10 of Appendix A.

In the long-term W-C-F rotations, all phases of the rotation are present in any given cropping year. Accurate schedule files were created for two of the three rotation phases, W-C-F and C-F-W, in order to have more years of simulated grain yields to compare with observed grain yields. Each schedule file was created for the 24 year period of 1985-2008. First, simulations were run with a schedule file reflecting actual management (no residue removal). Separate simulations were run for the three landscape positions (summit, sideslope, toeslope) at each site. This was done to more accurately reflect site specific soil and topographic parameters. Model output from each soil position was compared with observed yields. To compare observed and simulated stover yields, we assumed a carbon concentration of 0.40 for both wheat and corn stover.

Neutron attenuation was used to measure soil water content in 0-30 cm increments throughout the entire soil profile at planting and harvest at each site and cropping system. To validate the DAYCENT water submodel, ten years of soil water data were compared with model simulated soil water content (1988-1998).

Sampling for total carbon occurred numerous times during the 24-year experiment, but not always to the depth of 20 cm, the increment simulated by DAYCENT. Four years of observed total soil carbon (0-20 cm) were compared to simulated total soil carbon for (1986, 1997, 2001, and 2006). Methods of carbon sample preparation and analysis can be found in Sherrod et al., (2003). Observed carbon is reported as the mean of two replicates taken from the same treatment. In some years, samples were obtained from more than one phase of the same treatment (i.e., from W-C-F, and C-F-W). In these cases, the mean of both phases is used for comparison with simulated results. In addition, DAYCENT reports simulated total soil carbon on a monthly basis. Because field sampling for observed carbon always occurred between May and September, total annual simulated soil carbon is reported as the mean of modeled soil carbon from May – September. The combination of sites, soil type, crop rotation phase, and year of carbon determination gave a total of 24 individual comparisons of modeled and observed SOC data points.

After validating simulations with observed data, new schedule files were created for three different scenarios: (1) 50% removal of corn stover at harvest, (2) 50% removal of wheat stover at harvest, and (3) 50% removal of both corn and wheat stover. The first two scenarios result in stover harvest once every three years, while the third scenario harvests stover two out of every three years. The new schedule files were identical to the base schedule files, save for the addition of a stover harvest event. To project the long-term impacts of stover harvest, the 24-year schedule files were repeated as blocks for a total of 96 years. Observed weather data from 1985-2008 was repeated in the long-term projections so that changes in yield, carbon, or fertility due to management shifts (i.e., stover harvest) would be clearly distinct from weather-driven changes. Hence, model simulations do not simulate shifts in weather or atmospheric CO_2 concentration.

Fallow Efficiency, grain yield, and nitrogen mineralization rates

To calculate fallow efficiency, three numbers are needed: (i) total water in the soil profile at the start of the fallow period, (ii) water in the soil profile at the end of the fallow period, and (iii) total precipitation that occurred in the interim. The percentage of total precipitation that was stored during the fallow period (i.e., fallow efficiency) is found by subtracting water in the soil profile at the start of the fallow period from soil profile water at the end of the fallow period and then dividing by the total precipitation received during fallow. We used soil profile water output from DAYCENT's water sub-model and observed precipitation data to calculate fallow efficiency under 0% and 50% stover removal scenarios.

The long-term impacts of stover removal on soil fertility and grain yields were also simulated with DAYCENT. Grain yields (g C m^{-2}) and net annual mineralization rates (g N m^{-2} yr^{-2}) were examined with and without stover removal for three different soil types at each site. Results by soil type for each site were compiled to obtain responses on a site average basis.

Model Evaluation Statistics

Two model evaluation statistics were calculated to quantify DAYCENT model performance: (1) Relative error (RE), which shows the bias of the predicted mean relative to the observed mean, and (2) a normalized objective function (NOF), based on the root mean square error (RMSE), which shows the average deviation between predicted and observed values, regardless of sign. Relative error is expressed as:

$$RE = \frac{(\overline{\mathrm{P}} - \overline{\mathrm{O}})}{\overline{\mathrm{O}}} * 100 \qquad [3]$$

Where $\overline{P}$ is the predicted mean and $\overline{O}$ is the observed mean. Relative error is an arithmetic average over the duration of data, showing long-term bias. A negative RE value indicates a model bias toward underestimation, while a positive value indicates a model bias towards overestimation. Relative error is useful to pick out long-term trends in over or under prediction. While this simple interpretation is useful for guiding model improvement, bias alone is not sufficient to explain model errors. For example, a relative error value near zero could be the result of very small model errors in all situations or of large errors in some years that cancel out years of over and under prediction. The root mean square error (RMSE) eliminates the problem of compensation between over and under prediction by averaging the squared differences, hence weighting large differences more heavily. The RMSE is often more convenient to work with than the mean square error, because the RMSE has the same units as the response variable. The RMSE was calculated as:

$$RMSE = \sqrt{\frac{\sum_{\mathrm{i}=1}^{\mathrm{n}} (\mathrm{P} - \mathrm{O})^2}{\mathrm{n}}} \qquad [4]$$

Where P_i is the *i*th predicted value, O_i is the *i*th observed value, and *n* is the number of data pairs. The NOF (unitless) was calculated as:

$$NOF = \frac{\mathrm{RMSE}}{\bar{\mathrm{O}}} \qquad [5]$$

where RMSE and $\overline{O}$ are as previously defined. An NOF of zero indicates a perfect fit between experimental data and simulated results, while an NOF of less than 1 can be interpreted as a simulation error of less than 1 standard deviation around the experimental mean.

RESULTS AND DISCUSSION

Biomass Production

The long term, dryland cropping systems study conducted in the Central Great Plains of eastern Colorado provides a unique dataset for evaluation of biomass production over a long time period, at multiple locations, and for multiple crops. The combination of 24 years of dryland crop production data shows a large range in average total biomass production depending on crop species, soil position, and location. The importance of geographic, climatic, and soil variability is clearly illustrated in the long-term dataset.

The effects of location and soil type on the means and standard deviations of stover, grain, and total biomass production are summarized for individual crops in Tables 1-5 and 1-6 and Fig 1. Corn stover, grain, and total biomass production was significantly higher at Stratton than at Sterling or Walsh, but there were no significant differences in production between Sterling and Walsh. Wheat stover production at Stratton was significantly higher than at Sterling and Walsh, with no significant differences between Sterling and Walsh. Wheat grain production was significantly higher at Stratton than at Sterling, but there was no significant difference in grain production between Stratton and Walsh, or between Sterling and Walsh. Total biomass production for wheat was significantly higher at Stratton than at Sterling and Walsh, with no significant difference between total biomass production at Sterling and Walsh.

Total biomass production for corn and sorghum, averaged over site and soil type, was 5550 kg ha^{-1} yr^{-1}. Total corn/sorghum biomass production was divided into average stover and grain yields of 2750 kg ha^{-1} yr^{-1} and 2800 kg ha^{-1} yr^{-1}, respectively (Table 5). The total annual biomass production of wheat across all sites averaged 5800 kg ha^{-1} yr^{-1} (Table 6). Wheat annual stover yields were 3900 kg ha^{-1} yr^{-1} and grain yields averaged 1950 kg ha^{-1} yr^{-1}

At the Sterling location, average total biomass production was 5500 kg ha^{-1} yr^{-1} for corn and 5700 kg ha^{-1} yr^{-1} for wheat. Average corn stover production was 2700 kg ha^{-1} yr^{-1}, with grain production averaging 2800 kg ha^{-1} yr^{-1}. Average wheat stover production at Sterling was 3800 kg ha^{-1} yr^{-1}. For corn, the stover production level, averaged over site and soil type was 5550 kg ha^{-1} yr^{-1}. Total corn biomass production was divided into average stover and grain yield of 2750 kg ha^{-1} yr $^{-1}$ and 2800 kg ha^{-1} yr $^{-1}$, respectively (Table 5).

At the Stratton location, average total biomass production was 6200 kg ha^{-1} yr^{-1} for corn and 6550 kg ha^{-1} yr^{-1} for wheat. Average corn stover and grain production was 3100 kg ha^{-1} yr^{-1}. Average wheat stover production at Stratton was 4500 kg ha^{-1} yr^{-1}, and average grain production was 2100 kg ha^{-1} yr^{-1}.

At Walsh, the most southern location, average total biomass production was 5000 kg ha^{-1} yr^{-1} for sorghum and 5400 kg ha^{-1} yr^{-1} for wheat. Average sorghum grain and stover yields were 2500 kg ha^{-1} yr^{-1} and 2400 kg ha^{-1} yr^{-1}, respectively. Wheat stover production averaged 3500 kg ha^{-1} yr^{-1}. Average wheat grain production was 1900 kg ha^{-1}.

Annual variability in yield for wheat and corn/sorghum is given in Fig. 2. This figure illustrates the significant variability in production levels that can occur in grain-based dryland cropping systems of the Central Great Plains. In some years, there is large biomass production, suggesting that dryland farms in the Central Great Plains may be good sources of biomass. However, in other years, production is low and below expected levels of sustainable biomass harvest.

The species differences observed illustrate that sustainable biomass production must consider the crop species and crop rotations as important variables. Wheat has larger amounts of stover production than corn, while corn has greater grain yield. Corn and sorghum total biomass production is similar to that of wheat, but a larger percentage of the total biomass is grain compared to stover. Consideration of how total biomass is divided among grain and stover is important, especially if dual purpose grain and biomass harvests are considered. The species, varieties, and hybrids in the long-term experiment are grain crops that were not grown to maximize biomass. We are using these crops as model biomass crops to illustrate the range in biomass production and sensitivity to climate and location, but other studies should consider the potential of dedicated biomass crops in these environments. For all crop species, stover, grain, and biomass production was significantly higher at the toeslope position than at the summit or sideslope. There were no significant differences between production levels at the summit and sideslope. Toeslope soils, which can gain run-on water, soil, and nutrients, generally have thicker top soil and higher soil organic matter content. Summit and sideslope soils are shallower and less productive due to erosion and run-off. Evaluation of biomass production potential over a soil variable improves the value of this dataset by allowing extrapolation of biomass production to soils with a range in production potential. Models used to predict biomass production potentials should consider variable production potential by soil type.

These biomass production levels are low relative to those for the Midwest and other more humid environments that have been considered as sources of agricultural biomass. This is primarily due to differences in annual precipitation. Agricultural production has been economically sustainable in these environments as a function of relatively low land costs compared to land costs in more productive areas. The relatively low levels of biomass production in the dryland environments of the Central Great Plains raises concern about the ecological sustainability of biomass harvest. Under an assumption that 5 Mg ha^{-1} yr^{-1}of stover are needed to maintain soil organic carbon (Wilhelm et al., 2007), there would be little or no biomass left for sustainable harvest in these environments. However, the 5 Mg ha^{-1} assumption is based on data and conditions from no-till systems in Corn Belt environments and may not be applicable to semi-arid, dryland cropping systems.

Wind Erosion Modeling

Modeled wind erosion rates without simulated biomass harvest were 5.2, 4.3, and 8.1 Mg ha^{-1} yr^{-1} for the Sterling, Stratton, and Walsh locations, respectively (Fig. 3). Tolerable soil loss limits at the three sites are 11 (Sterling, Stratton) and 9 Mg ha^{-1} yr^{-1} (Walsh). While no measurements of wind erosion have been made at these sites, it has generally been observed that no-till management is very effective at controlling wind erosion. Cantero-Martinez et al. (2006) showed that crop residue levels at Sterling and Stratton averaged 2.7 and 4.0 Mg ha^{-1} yr^{-1}, respectively, which are levels generally accepted to provide adequate wind erosion. However, at Walsh, the same study found average crop residue levels to be marginal for soil erosion protection. The impact of climate on wind erosion was evident in simulations at Walsh, where erosion rates without removal were 8.1 Mg ha^{-1} yr^{-1}, just below the T-value of 9 Mg ha^{-1} yr^{-1}.

Wind erosion greatly increased at all three sites with simulated corn or sorghum stover harvest (Fig. 3). For example, at Sterling, where simulated annual erosion in a wheat-corn-fallow rotation was 5.2 Mg ha^{-1} yr^{-1} without residue removal, baling only 20% of the corn stover increased annual erosion to 13.7 Mg ha^{-1} yr^{-1}, well above the T-value for that site (11 Mg ha^{-1} yr^{-1}). At Stratton, a removal rate of 20% of corn stover resulted in levels of erosion just below the T-value of 11 Mg ha^{-1} yr^{-1}; erosion rates exceed the T-value by increasing the stover removal rate to 30%, with an erosion rate of 14.1 Mg ha^{-1} yr^{-1}. At the Walsh location, removing only 10% of sorghum stover pushed erosion rates above the T-value. Across all experimental sites, wheat produced substantially more residue than the minimum required for maintaining T-values. Up to 80% of wheat residue at Sterling and Stratton could be removed without surpassing tolerable erosion rates. These modeling results agree with work on the same sites that found residue levels present on the soil surface under more intensive no-till cropping systems were generally adequate to control erosion by wind, except at the high PET site, where residue levels were marginal for adequate wind erosion control (Cantero-Martinez et al., 2006). However, it should be noted that large variability in residue levels occurs among years, as differences in annual precipitation and temperatures cause variations in crop production and subsequent residue levels.

Leaving a vegetative-cover on the soil surface is one of the most effective practices for controlling wind erosion. However, residues vary significantly in their effectiveness in preventing wind erosion. The level of erosion protection provided by vegetation depends on how much dry matter it contains, its texture, whether it is living or dead, and its orientation (standing or flat). In estimating the Vegetative Cover Factor (V), WEQ converts the dry weights of growing crops or crop residues into equivalent quantities of 10-inch-long stalks of small grain lying flat in rows 10 inches apart, or small grain equivalent (SGe). The SGe describes the amount of vegetative protective cover on a field surface that a particular agricultural crop residue provides. Large, randomly distributed corn stalks lying flat on the soil surface are much less effective than fine-strawed, upright wheat stubble; hence wheat has a much higher SGe. For example, even though there are comparable amounts of dry matter left on the soil surface after removing 50% of wheat or corn stover (average across sites: 1680 kg ha^{-1} for wheat; 1260 kg ha^{-1} for sorghum; 1460 kg ha^{-1} for corn), the protection provided by the wheat and sorghum is greater than that of corn, with sorghum falling between the two. After harvesting 50% of wheat straw at Sterling, there was still 4360 kg ha^{-1} of SGe dry matter. Removing 50% of the corn stover left only 960 kg ha^{-1} of SGe, a mere 22% of the SGe provided by wheat (Fig. 4). Similarly, the estimated percent ground cover after harvesting half of the stover is 56% for wheat, 48% for sorghum, and only 37% for corn. The timing of corn and sorghum harvest also contributes to negative impacts of removing corn and sorghum stover. In Colorado, the critical erosion period is from November through April. It is a period when the soil surface conditions and erosive winds result in the greatest potential for erosion.

Tolerable erosion rates were maintained when 70-80% of wheat residues were removed (Fig. 3), but rose above tolerable levels when the SGe levels dropped below 1680 kg ha^{-1} (approx.

85% removal rate). This suggests that there is a critical amount of SGe that must remain to prevent wind erosion. These modeling results are based on long-term average yields as a model input, but with fluctuating yield numbers, the percent of stover that could be removed while still maintaining an acceptable amount of SGe will vary from year to year.

It is important to note that wind erosion also represents a loss of sediment-bound SOC. The eroded soil is enriched in SOC. This is particularly true in no-till systems, where surface soils are stratified, as there is little or no opportunity for vertical re-distribution of surface SOC. As a simple example, soils at Sterling are around 0.8% carbon. Annual erosion rates of 10 Mg^{-1} ha^{-1} yr^{-1} (still below the recommended T-value) could result in losses of approximately 80 g C m^{-2} yr^{-1}.

Water Erosion Modeling

Soil loss levels due to water erosion in the wheat-corn-fallow and wheat-corn-sorghum systems were well below recommended T-values for all sites and slopes. With no residue removal, average annual erosion rates ranged from 0.09 - 0.53 Mg ha^{-1} yr^{-1}. Adding a baling operation immediately after grain harvest resulted in nominal increases in annual erosion levels (Table 7). Not surprisingly, the side soil positions were most prone to soil water erosion, but even these levels never rose above 1.30 Mg ha^{-1} yr^{-1} when 50% of residues were collected at harvest. RUSLE simulations suggest that water erosion is not the limiting constraint to stover collection in these semi-arid Great Plains environments. However, as noted earlier, the model is not designed to predict erosion associated with intense single storm events. A large proportion of annual precipitation in eastern Colorado occurs as high intensity summer thunderstorms, which can induce runoff and soil erosion. High intensity rainstorms have the potential to substantially increase erosion rates via high amounts of runoff. A single rainstorm of high intensity can result in sediment loss with residue removal rates of 50% under no-till conditions (Blanco-Canqui et al., 2009). Removing crop residue can result in a shorter time to runoff initiation and greater runoff volume. Hence, removing enough residues to substantially reduce soil cover could leave soils vulnerable to sediment runoff during high intensity rainfall events.

While no-till agriculture may help reduce runoff and associated soil loss compared to other management systems, most of the reductions in erosion are due to the crop residues left on the soil surface (Wilson et al., 2004). Previous work by Norvell et al., (2008) at Sterling and Stratton reported runoff rates as high as 54% of the precipitation during a high intensity rain event during periods of low crop residue cover. Using historical rainfall intensity data coupled with field measurements of runoff, estimates of runoff for different climatic and management conditions ranged from 8 mm yr^{-1} with good surface cover during drought, to 80 mm yr^{-1} with low soil protection and above average precipitation. Using sediment concentrations from 66 observed runoff events, the authors estimated annual rates of soil erosion to range from 1 to 9 Mg ha^{-1} yr^{-1}, depending on climatic conditions and cover. In years of above average precipitation and low cover, soil erosion rates were estimated at 9.3 and 4.5 Mg ha^{-1} yr^{-1} for Sterling and Stratton, respectively. While RUSLE does an adequate job of estimating long term average rates of water erosion, these production systems have higher potential annual water erosion rates than the long-term average rates predicted by the model. For example, even at the soil position most prone to water erosion (Stratton side slope), reducing average crop yields by 50%, which effectively reduces the percent soil cover to 25-35% throughout the 3-yr WCF rotation, results in a simulated annual soil loss rate of only 2.4 Mg ha^{-1}.

Soil erosion via water is unlikely to be the constraining factor to stover harvest if good soil cover is maintained. Norvell et al., (2008) estimated that with good soil cover, erosion rates would drop to 2.4 and 0.9 Mg ha^{-1} for above and below average precipitation at Sterling, and 1.1 and 1.4 Mg ha^{-1} for above and below average precipitation at Stratton. These erosion rates are relatively low and show the value of management practices that protect the soil surface from soil erosion. However, care must be taken to ensure that a proper residue base is maintained and the erosion associated with high intensity rainfall events does not become unsustainable.

DAYCENT Modeling

Model validation - crop yields

Overall, observed and simulated yields typically followed the same pattern at Sterling and Stratton (Fig. 5). RMSE values for wheat grain were 14 and 34 g m^{-2} yr^{-1} at Sterling and Stratton, respectively. RMSE values for corn were slightly higher, with an RMSE of 41 g m^{-2} yr^{-1} at Sterling and 38 g m^{-2} yr^{-1} at Stratton (Table 8). The observed yield variation was primarily driven by variation in the amount and timing of precipitation. The relatively large NOF values were mainly due to overestimation of the model in years of drought, and underestimation in years of good precipitation, where simulated yields appeared to be limited by nitrogen. In addition, DAYCENT cannot account for reductions in yields via insect, hail, or herbicide damage.

The only year that the model was unable to accurately model C in harvested grain was 2002, a year of crop failure due to severe drought with approximately 250 mm of total precipitation. The failure of the model to simulate years of low production due to exceptionally low amounts of precipitation (<250 mm) was also noted by Alvaro-Fuentes et al., (2009). Overall, the model was able to accurately simulate year to year trends in grain yields and associated carbon in grain. Comparison of observed and simulated C in harvested grain, averaged over years, shows that the model does very well in simulating long-term average yields for both stover and grain (Fig. 6). Relative error trends in simulated wheat and corn grain yield were not consistent across sites, but results for corn showed lower overall relative error than wheat. Relative error at Sterling was -5.4% for wheat grain, and 2.4% for corn grain. At Stratton, relative error was 8.6% for wheat grain and -4.1% for corn grain.

For the purposes of this simulation, it was important that long-term observed and modeled stover yields track closely in order to have confidence in the inferences of stover removal. As with grain, the model does well in simulating long-term average stover yields. Relative error for long-term average stover was quite low at Stratton (-0.6 % for wheat stover and 0.0% for corn stover). Relative error for stover at Sterling was higher for wheat (-12.6%), but corn again had a low relative error of -0.7%. RMSE values at both sites for corn and wheat stover were quite high compared to those of grain (RMSE = 52 for wheat and 42 for corn). The timing and duration of water stress can impact harvest index, and the depression in harvest index can be severe if there is a limited supply of water. In water-stressed systems, harvest index shows large variations from year to year. For example, observed harvest indices at Sterling ranged from 0.18 – 0.57 for wheat, and from 0.14 – 0.63 for corn. DAYCENT was not able to simulate the low harvest indices that occur during water stress, and hence underestimated stover production in water-stressed years.

It should be noted that simulation runs on a landscape position basis tended to over or under estimate stover and grain yield, depending on soil position. For example, yields at the toe position were underestimated, while yields at the summit were typically overestimated. It was determined that this was due to the fact that DAYCENT cannot simulate soil, water, and nutrient movement from one landscape position to the next (see thesis chapter 2).

Model validation – soil water

Total simulated soil profile water at the summit position for both sites fit well with observed data (RE=0.7% for Sterling and 1.3% for Stratton). Soil water at the toe position was strongly underestimated, with a relative error of -27.7% and -29.0% for Sterling and Stratton, respectively. It has been observed that runoff from the side-slope position at these sites results in run-on water at the toe-slope position. DAYCENT is not set up as a spatially distributed model and therefore, does not account for these landscape water dynamics. As previously stated, DAYCENT predictions are better when averaged over landscape positions than for site specific predictions.

Model validation – soil carbon

Soil organic carbon (SOC) was measured in four different years during the field experiments: 1986, 1997, 2001, and 2006. Simulated and observed SOC correlated very well on a site average basis (RMSE = 24; NOF=0.01; Fig. 7). Consistent with the results for yield and soil water comparisons, a pattern of overestimation of SOC in soils at the summit position and under estimation at toeslope position was found (see thesis chapter 2). This is not surprising, considering that water and wind erosion can deposit carbon and nutrients at the toe soil position and that increased water in the toeslope position results in higher crop yields. While DAYCENT did not simulate the spatial dynamics well, simulation of SOC on a landscape average basis closely matched observed values.

Predicted effects of stover harvest on SOC

The impact of 50% removal of corn and/or wheat stover was projected out until 2080 by using the same 24-year management and weather files. The model predicted a decline in SOC for the no-stover removal scenario during the first 5 years of the simulation (Fig. 8). This decline is a continuation of the decline in SOC occurring during the previous 50-years of wheat-fallow management and a lag in the stabilization of SOC levels after adoption of no-till practices in 1986. However, with 0% residue removal, SOC levels show very little long-term fluctuation apart from the expected inter-annual variability due to annual variations in yield and climate, and do not appear to be increasing in SOC.

Soil organic carbon declined 8.6% and 9.5% after 96 years of simulating 50% removal of both corn and wheat stover at Sterling and Stratton, respectively. Removing wheat stover (once every three years) for the 96 years of simulation reduced carbon levels by 5.8% at Sterling and

5.9% at Stratton. Similar results were obtained for removal of corn stover, with reductions of 5.4% and 6.3% over the course of 96 years. After 50-60 simulation years at Sterling, reductions in SOC due to removal of wheat and corn stover appear to stabilize, and are 5-6% below the no removal scenario for the remainder of the simulation. At Stratton, SOC levels take approximately 65 years to reach a new equilibrium level, which fluctuates between 9-10% below SOC levels without stover harvest. After 96 years of simulated stover harvest, carbon levels are approximately 125 and 250 g C m^{-2} lower at Sterling and Stratton, respectively.

It is interesting to note that the model does not predict increasing levels of SOC under the no-till WCF system. These longer-term model simulations show that conversion to a no-till management system with more intensive production of crop residues resulted in a relatively fast stabilization of SOC levels. No-till practices in dryland environments are ideal for conserving soil and water, yet the potential for carbon sequestration is not well demonstrated. Previous work in these no-till systems showed increases in SOC in the 0-5 cm depth after 12 years of continuous cropping, but no significant differences between the W-F and W-C-F cropping systems. The continuous cropping system contains no fallow, and returns more stover on an annual basis than the W-F or W-C-F cropping systems. Bowmen et al., (1999) also found that in no-till dryland systems, SOC was increased in the 0-5 cm depth under continuous cropping compared to W-F, but found no significant differences in carbon between rotations with a fallow period. Concentrations of SOC and N are typically higher within the 0-5 or 0-10 cm depth in of intensively cropped no-till systems, but these studies do not report the impact of no-till on the total soil profile SOC.

Fallow Efficiency

The fallow water storage efficiencies under 0 and 50% removal were calculated for seven simulated periods of fallow from 1985-2008. To account for water movement over the three landscape positions, simulated runoff water from the summit was added to profile water on the sideslope, and runoff from the sideslope was added to the toeslope. Using this method, simulated fallow efficiencies at Stratton ranged from 16 - 30% with no stover removal. These numbers are well within the expected range of reported mean fallow efficiencies in no-till dryland systems (McGee et al., 1997).

Surprisingly, DAYCENT showed no noticeable net change in fallow efficiency with 50% stover harvest. The simulated amount of water lost via evaporation increased, but water lost via runoff only increased in one year (Fig. 9). As part of the water sub model, DAYCENT treats water that falls on crop residue as being intercepted and evaporated off of the crop residue, never infiltrating the soil. In several years, increased evaporation is offset by less interception. It seems unlikely that there would be no net change of stored water with stover removal.

Work in the southern Great Plains found that soil water storage progressively increased with increasing application rates of straw mulch during fallow (Greb et al., 1967). A straw-mulch study in the Central and Northern Great Plains reported precipitation storage increased from 16% with no mulch to 37% with 6720 kg ha^{-1} of mulch (Greb et al., 1970). In the Great Plains, the residue amounts at the time of maximum residue accumulation are in the 2200 – 5600 kg ha^{-1} range (Peterson and Westfall, 2004). In our long-term field studies, average stover yields ranged from around 2500 – 3000 kg ha^{-1} for corn to 3500 - 4500 kg ha^{-1} for wheat. Removing 50% of this stover on an annual basis would decrease the residue base, and likely reduce fallow efficiency. The ability of the water sub model to accurately simulate changes in soil water storage with stover harvest should be more closely examined if DAYCENT is widely adopted as a tool to infer the impacts of stover harvest in water-limited regions.

Soil Fertility Impacts

Long-term predicted crop grain yields were strongly decreased with 50% removal of corn and wheat stover (Fig. 10). Negative changes in yield as high as 45% were simulated on a soil type basis. Predicted yields at Stratton were more impacted by stover harvest than yields at Sterling. Yields during the first 24 years of the simulation show a less severe decline than yields during the remainder of the 96 year simulation.

The decline in simulated yields can be attributed to changes in net N mineralization rates (Fig. 11). N mineralization rates are a function of temperature, precipitation, C/N ratio of soil organic matter, pH, and soil texture. In any given year, if we simulated a situation of excess available nitrogen, only 1-2% yield declines were observed with residue removal. These small simulated declines are likely due to small changes in soil temperature (approx. 1-2° C) or changes in water loss via evaporation. The magnitude of change in annual N mineralization with stover harvest is quite different between sites, but at both sites, an initial increase in mineralization rates lasting approximately 18-20 years can be observed. Increased rates of mineralization could be due to both increases in soil temperatures and decreased N immobilization.

Previous in situ nitrogen mineralization experiments at Sterling and Stratton found that total net N mineralization in the W-C-F cropping system was half that of the W-F system (Kolberg et al., 1999). The authors attributed this to greater accumulation of crop residue carbon on the soil surface and surface layers in the more intensive W-C-F system, and hence greater N immobilization. Nitrogen immobilization rates are notoriously high for nitrogen-poor material such as cereal straw. The same study also found that the rate of nitrogen fertilization influences soil N mineralization in the W-C-F rotation, with approximately 0.2 kg ha^{-1} total net soil N mineralization from mid-April to September expected for each 1 kg ha^{-1} of nitrogen applied. While DAYCENT simulated net mineralization rates are a bit lower than observed in-situ rates, Kelly et al. (2000) also found that observations of net N mineralization tended to be higher than simulated values because plant uptake is considered in the model and not included in field measurements of mineralization rates.

A strong decline in mineralization rates can be observed after approximately 20 simulation years. Because simulations were run in 24-year ‘blocks’ with the same climatic inputs, we were able to compare changes in mineralization between blocks. The first 24-year block shows an increase in mineralization rates; mineralization rates strongly decline in the second 24-year block, and by the third 24-year block, mineralization rates stabilize at a new lower level. Yield declines stabilize at virtually the same time as N mineralization.

Yield declines also contribute to simulated losses of soil carbon with stover harvest, although yield reduction is not the primary driver of simulated SOC declines. Assuming a 1:1 ratio for grain and stover, a 10% reduction in grain yield means 10% less stover is produced (and a farmer would then be harvesting 50% of a smaller value). However, stover declines represent a very small carbon loss when compared with harvesting 50% of total stover. We simulated a 20% increase in fertilizer rates, enough to bring yields back to, or above, levels without stover harvest. Preventing yield declines mitigated SOC declines with stover harvest by approximately 2%, but did not halt the decline.

These results are not surprising, as continually removing stover depletes both available N and C. The magnitude of yield decline was surprising. In dryland systems, yields are low compared to those in well-watered regions, and hence, even a small decline can have a large impact on percentage yield decline. Decreased N mineralization rates with stover harvest will require farmers to compensate for yield loss with increased fertilization. Anticipating dynamic N mineralization responses to stover harvest will be a difficult task, as yield and N mineralization responses will be both soil and site specific (Figs. 10 and 11).

Potential harvestable stover with dedicated bioenergy crops

If the current wheat-corn-fallow production systems are continued, there would be very little or no stover available for harvest. Wind erosion modeling demonstrated that virtually no corn stover can be removed without exceeding tolerable wind erosion rates, but up to 80% of wheat stover can be harvested. We assumed that 100% of corn stover must remain to provide adequate protection from wind erosion. Total average wheat stover production at Sterling and Stratton was 3796 and 4474 kg ha^{-1}, respectively. Controlling for wind erosion, only 80% of wheat stover can be harvested. In the current management system, wheat is present only once every three years. The need to control wind erosion would leave only 3000 kg ha^{-1} of harvestable wheat stover at Sterling and 3600 kg ha^{-1} at Stratton (1000 and 1200 kg ha^{-1}, respectively, if presented on an annualized basis (Fig. 12).

If the target or goal is to prevent any losses in SOC, there will be virtually no collectable stover in a W-C-F system. Within 5 years of conversion to no-till management, levels of SOC at both Sterling and Stratton appear to stabilize (Fig. 8). With projection of the current management practices (i.e., no stover harvest) for nearly 100 years, there is virtually no change in SOC, indicating that residue return rates with no stover harvest are generally adequate to maintain SOC. All stover would have to be retained to prevent small declines in SOC. On an annualized basis, this is equal to 2200 kg ha^{-1} of stover retained at Sterling and 2500 kg ha^{-1} at Stratton (Fig. 12).

The stover return rates required to maintain SOC in dryland systems are low compared to those reported for more productive systems. Larson et al., (1972) estimated that 6 Mg ha^{-1} yr^{-1} of cornstalks was required to prevent losses of organic matter on a soil in Iowa. Under no or conservation tillage, Johnson et al. (2006) estimated that 5.25 Mg ha^{-1} of stover was needed in a continuous corn system to maintain carbon, and 7.90 Mg ha^{-1} of corn stover was needed in a no-till corn-soybean rotation. According to these simulations, in no-till dryland systems, approximately 2.2 – 2.5 Mg ha^{-1} yr^{-1} are required to maintain SOC, much less than in the Corn Belt.

An alternate scenario could exist in which dedicated non-grain bioenergy crops, such as winter triticale, are grown. A review of cropping systems in the Great Plains found that systems containing forages generally had greater water and precipitation use efficiencies than systems that did not contain forages (Nielsen et al., 2006). Because energy is not diverted to grain production, overall biomass yields of a non-grain biomass crop could be higher and more consistent than those of grain crops. Grain crops require precipitation at critical stages whereas yields of non-grain biomass crops are less sensitive to precipitation timing (Nielsen et al., 2009). Growing forage crops on an annual basis could allow increases in annual production of biomass because it diminishes the risk of crop failure inherent in systems containing grain crops.

Nielsen et al. (2006) examined the dry matter production of a three year rotation of dryland silage corn, foxtail millet, and winter triticale for seven years (1998 – 2004) in the Central Great Plains. Precipitation during the study period was highly variable, with several years of below average rainfall. Dry matter production of winter triticale ranged from 730 – 10,630 kg ha^{-1}, averaging 3920 kg ha^{-1}. Linking production functions with historical precipitation records gave estimated average winter triticale dry matter yields of 5370 kg ha^{-1}yr^{-1}. The authors determined that dry matter production of at least 4000 kg ha^{-1} would be expected to occur in 75% of years for winter triticale.

In the WCF cropping system examined in this study, total average biomass production of wheat was approximately 5600 kg ha^{-1} at Sterling and 6500 kg ha^{-1} at Stratton. On average, total biomass production of corn was 5500 kg ha^{-1} at Sterling and 6200 kg ha^{-1} at Stratton. In a WCF rotation, with a crop present two out of every three years, adding total biomass production for wheat and corn at each site results in an annualized total biomass of 3700 kg ha^{-1} yr^{-1}at Sterling and 4200 kg ha^{-1} yr^{-1} at Stratton. In a continuous cropping system at these dryland sites that contained both grain and forage crops, Peterson et al., (2004) found that annualized biomass yields under continuous cropping were approximately 1000 kg ha^{-1} greater than biomass yields from three year rotations that contained fallow. Based on these long-term observations, we could project that in a production system with a non-grain biomass crop, average annual production levels may be 4700 and 5200 kg ha^{-1} at Sterling and Stratton, respectively. The potential for a dedicated biomass crop in these ecosystems must still consider the required amounts of residue return for sustainability. Model simulations for the WCF system indicated that approximately 2200 and 2500 kg ha^{-1} of residue must remain on an annual basis to maintain SOC, which is greater than the amount of residue required to prevent unacceptable levels of wind erosion. Subtracting off residue required to maintain SOC leaves an average of 2500 and 2700 kg ha^{-1} of biomass that could be sustainably harvested on an annual basis (Fig. 12).

The estimated stover needed to maintain SOC and prevent untolerable rates of soil erosion is reported for a WCF system. Elimination of the fallow period would likely reduce the amount of residue that must remain to maintain SOC, and perhaps build SOC. As noted by Sherrod et al. (2003), continuous cropping minimizes the opportunity for accelerated rates of SOC oxidation, and more closely matches perennial systems in which nutrient immobilization and mineralization processes are balanced, resulting in minimum nutrient loss and maximum accumulation of organic matter. Sherrod et al. (2003) found that cropping systems that eliminated the summer fallow maximized the amount of soil SOC and total N. Building SOC in dryland no-till systems may only be possible with increased cropping intensity via elimination of summer fallow, as inclusion of fallow has a negative influence on SOC in the central Great Plains (Bowman et al., 1999). Fallow is also the period with the highest erosion risk, as there is no growing crop to attenuate wind speeds and protect the soil surface. Replacement of the fallow period with a non-grain biomass crop could lower the amount of residue needed to control erosion. As every crop has a different protective nature against wind erosion, the amount of stover needed for wind protection would vary based on the biomass crop, but replacing corn with a biomass crop with a higher SGe would also lower the amount of residue required to control wind erosion.

The amount of biomass that could be collected from dedicated dryland biomass cropping systems in the semiarid Great Plains may be close to the amounts that can be collected in high production areas. Johnson et al., (2007) reported that average non-grain biomass yield in 2005 for corn was 6.95 Mg ha^{-1}, substantially higher than could ever be achieved in dryland systems. However, they also estimate that at least 5.25 Mg ha^{-1} of this biomass must be retained, leaving only 1.7 Mg ha^{-1} of harvestable biomass. This is less than the average amount of biomass estimated to be available on an annual basis in a dryland cropping system with dedicated production of a biomass crop. Thus, it is found that harvest of agricultural residues in drylands of the semiarid Great Plains is not feasible with current cropping systems. However, there is potential for these systems to be viable sources of biomass if cropping systems were converted to dedicated bioenergy cropping.

CONCLUSIONS

If agriculture is to be involved in meeting the increasing biofuel demand, production of biofuel feedstocks will likely require conversion of existing crop land to grow dedicated biofuel crops. Shifting agricultural production towards biofuel/bioenergy crops with heavy water requirements could have negative impacts on already limited water resources (Stone et al., 2010). Producing non-grain biomass for bioenergy in the semi-arid Great Plains would require no additional water input or competition for water resources. Wind erosion can be controlled with proper management (i.e., no-till techniques, leaving a minimum residue base). While small changes in soil carbon were observed with simulated stover harvest, harvesting biomass from systems with inherently low rates of soil carbon minimizes the magnitude of soil carbon that will be lost or depleted. If dedicated non-grain crops were grown, approximately 2.5 – 2.7 Mg ha^{-1} of biomass could be harvested, while still retaining enough residues to maintain soil carbon. Eliminating summer fallow and intensification of production could be possible if the systems are not producing grain crops. Elimination or reduction in periods of summer fallow would likely reduce minimum residue return rates, as soil cover during fallow is greatly reduced and increases erosion risks. Additionally, during fallow, carbon inputs to the system are virtually zero, but decomposition of SOM continues, resulting in carbon losses. While crop yields are much lower in dryland systems than in more humid regions of the country, the Great Plains should not be discounted in the inventories of potential biomass producing regions. The performance of growing non-grain dedicated biomass crops in semi-arid regions should be examined.

LITERATURE CITED

Alberts E.E., Neibling W.H. (1994). Influence of crop residues on water erosion Lewis Publishers, Chelsea, Mich.

Alvaro-Fuentes J., Lopez M.V., Arrue J.L., Moret D., Paustian K. (2009). Tillage and cropping effects on soil organic carbon in Mediterranean semiarid agroecosystems: Testing the Century model. Agriculture Ecosystems & Environment 134:211-217. DOI: 10.1016/j.agee.2009.07.001.

Ascough J.C., Andales A.A., Sherrod L.A., McMaster G.S., Hansen N.C., DeJonge K.C., Fathelrahman E.M., Ahuja L.R., Peterson G.A., Hoag D.L. (2010). Simulating landscape catena effects in no-till dryland agroecosystems using GPFARM. Agricultural Systems 103:569-584. DOI: 10.1016/j.agsy.2010.06.005.

Blanco-Canqui H., Lal R. (2007). Soil and crop response to harvesting corn residues for biofuel production. Geoderma 141:355-362. DOI: 10.1016/j.geoderma.2007.06.012.

Blanco-Canqui H., Lal R. (2008). No-tillage and soil-profile carbon sequestration: An on-farm assessment. Soil Science Society of America Journal 72:693-701. DOI: 10.2136/sssaj2007.0233.

Blanco-Canqui H., Lal R. (2009). Corn Stover Removal for Expanded Uses Reduces Soil Fertility and Structural Stability. Soil Science Society of America Journal 73:418-426. DOI: 10.2136/sssaj2008.0141.

Blanco-Canqui H., Lal R., Post W.M., Owens L.B. (2006). Changes in long-term no-till corn growth and yield under different rates of stover mulch. Agronomy Journal 98:1128-1136. DOI: 10.2134/agronj2006.0005.

Blanco-Canqui H., Stephenson R.J., Nelson N.O., Presley D.R. (2009). Wheat and Sorghum Residue Removal for Expanded Uses Increases Sediment and Nutrient Loss in Runoff. Journal of Environmental Quality 38:2365-2372. DOI: 10.2134/jeq2009.0021.

Bowman R.A., Vigil M.F., Nielsen D.C., Anderson R.L. (1999). Soil organic matter changes in intensively cropped dryland systems. Soil Science Society of America Journal 63:186-191.

Cantero-Martinez C., Westfall D.G., Sherrod L.A., Peterson G.A. (2006). Long-term crop residue dynamics in no-tilt cropping systems under semi-arid conditions. Journal of Soil and Water Conservation 61:84-95.

Del Grosso S., Ojima D., Parton W., Mosier A., Peterson G., Schimel D. (2002a). Simulated effects of dryland cropping intensification on soil organic matter and greenhouse gas exchanges using the DAYCENT ecosystem model. Environmental Pollution 116:S75-S83.

Del Grosso S.J., Ojima D.S., Parton W.J., Mosier A.R., Peterson G.A. (2002b). Regional assessment of net greenhouse gas fluxes from agricultural soils in the USA Great Plains under current and improved management Millpress Science Publishers, Rotterdam.

Farrell A.E., Plevin R.J., Turner B.T., Jones A.D., O'Hare M., Kammen D.M. (2006). Ethanol can contribute to energy and environmental goals. Science 311:506-508. DOI: 10.1126/science.1121416.

Graham R.L., Nelson R., Sheehan J., Perlack R.D., Wright L.L. (2007). Current and potential US corn stover supplies. Agronomy Journal 99:1-11. DOI: 10.2134/argonj2005.0222.

Greb B.W., Smika D.E., Black A.L. (1967). Effect of straw mulch rates on soil water storage during summer fallow in great plains. Soil Science Society of America Proceedings 31:556-&.

Greb B.W., Smika D.E., Black A.L. (1970). Water conservation with stubble mulch fallow. Journal of Soil and Water Conservation 25:58-&.

Johnson J.M.-F., Coleman Mark D., Gesch Russ, Jaradat Adbullah, Mitchell Rob, Reicosky Don, Wilhelm W.W. (2007). Biomass-bioenergy crops in the United States: a changing paradigm. The American Journal of Plant Science and Biotechnology Vol. 1(1): 1-28

Johnson J.M.F., Allmaras R.R., Reicosky D.C. (2006). Estimating source carbon from crop residues, roots and rhizodeposits using the national grain-yield database. Agronomy Journal 98:622-636. DOI: 10.2134/agronj2005.0179.

Kelly R.H., Parton W.J., Hartman M.D., Stretch L.K., Ojima D.S., Schimel D.S. (2000). Intra-annual and interannual variability of ecosystem processes in shortgrass steppe. Journal of Geophysical Research-Atmospheres 105:20093-20100.

Kelly R.H., Parton W.J., Crocker G.J., Grace P.R., Klir J., Korschens M., Poulton P.R., Richter D.D. (1997). Simulating trends in soil organic carbon in long-term experiments using the century model. Geoderma 81:75-90.

Kittel T.G.F., Rosenbloom N.A., Royle J.A., Daly C., Gibson W.P., Fisher H.H., Thornton P., Yates D.N., Aulenbach S., Kaufman C., McKeown R., Bachelet D., Schimel D.S., Participants V. (2004). VEMAP phase 2 bioclimatic database. I. Gridded historical (20th century) climate for modeling ecosystem dynamics across the conterminous USA. Climate Research 27:151-170.

Kolberg R.L., Westfall D.G., Peterson G.A. (1999). Influence of cropping intensity and nitrogen fertilizer rates on in situ nitrogen mineralization. Soil Science Society of America Journal 63:129-134.

Lal R. (2004). Carbon sequestration in dryland ecosystems. Environmental Management 33:528-544. DOI: 10.1007/s00267-003-9110-9.

Mann L., Tolbert V., Cushman J. (2002). Potential environmental effects of corn (Zea mays L.) stover removal with emphasis on soil organic matter and erosion. Agriculture Ecosystems & Environment 89:149-166.

Mulder K., Hagens N., Fisher B. (2010). Burning Water: A Comparative Analysis of the Energy Return on Water Invested. Ambio 39:30-39. DOI: 10.1007/s13280-009-0003-x.

Nelson R.G. (2002). Resource assessment and removal analysis for corn stover and wheat straw in the Eastern and Midwestern United States - rainfall and wind-induced soil erosion methodology. Biomass & Bioenergy 22:349-363.

Nelson R.G., Walsh M., Sheehan J.J., Graham R. (2004). Methodology for estimating removable quantities of agricultural residues for bioenergy and bioproduct use. Applied Biochemistry and Biotechnology 113:13-26.

Nielsen D.C., Unger P.W., Miller P.R. (2005). Efficient water use in dryland cropping systems in the Great Plains. Agronomy Journal 97:364-372.

Nielsen D.C., Vigil M.F., Benjamin J.G. (2006). Forage yield response to water use for dryland corn, millet, and triticale in the central Great Plains. Agronomy Journal 98:992-998. DOI: 10.2134/agronj2005.0356.

Nielsen D.C., Vigil M.F., Benjamin J.G. (2009). The variable response of dryland corn yield to soil water content at planting. Agricultural Water Management 96:330-336. DOI: 10.1016/j.agwat.2008.08.011.

Norvell K., Hansen N.C., Westfall D.G., Ahuja L.R. (2008). Runoff and Erosion Estimates for Great Plains Dryland Agroecosystems., Proceedings of AGU Hydrology Days pp. 79-87.

Parton W.J., Rasmussen P.E. (1994). Long-term effects of crop management in wheat-fallow .2. Century model simulations. Soil Science Society of America Journal 58:530-536.

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

Parton W.J., Ojima D.S., Cole C.V., Schimel D.S. (1994). A general model for soil organic matter dynamics - sensitivity to litter chemistry, texture and management, in: R. B. Bryant and R. W. Arnold (Eds.), Quantitative Modeling of Soil Forming Processes. pp. 147-167.

Parton W.J., Hartman M., Ojima D., Schimel D. (1998). DAYCENT and its land surface submodel: description and testing. Global and Planetary Change 19:35-48.

Paustian K., Parton W.J., Persson J. (1992). Modeling soil organic-matter in organic-amended and nitrogen-fertilized long-term plots. Soil Science Society of America Journal 56:476-488.

Perlack R.D., Wright L.L., A.Turhollow, Graham R.L., Stocks B., Erbach D.C. (2005). Biomass as a feedstock for a bioenergy and bioproducts industry: The technical feasibility of a billion ton annual supply., USDOE and USDA.

Peterson G.A., Westfall D.G. (2004). Managing precipitation use in sustainable dryland agroecosystems. Annals of Applied Biology 144:127-138.

Peterson G.A., Westfall D.G., Cole C.V. (1993). Agroecosystem approach to soil and crop management research. Soil Science Society of America Journal 57:1354-1360.

Renard K.G., G.R. Foster, G.A. Weesies, D.K. McCool, and D.C. Yoder. (1997). Predicting soil erosion by water: A guide to conservation planning with the Revised Universal Soil Loss Equation (RUSLE). Agricultural Handbook No. 703, Agricultural Research Service, USDA, Washintong, DC.

Sarath G., Mitchell R.B., Sattler S.E., Funnell D., Pedersen J.F., Graybosch R.A., Vogel K.P. (2008). Opportunities and roadblocks in utilizing forages and small grains for liquid fuels. Journal of Industrial Microbiology & Biotechnology 35:343-354. DOI: 10.1007/s10295-007-0296-3.

Schaap M.G., Leij F.J., van Genuchten M.T. (2001). ROSETTA: a computer program for estimating soil hydraulic parameters with hierarchical pedotransfer functions. Journal of Hydrology 251:163-176.

Searchinger T., Heimlich R., Houghton R.A., Dong F.X., Elobeid A., Fabiosa J., Tokgoz S., Hayes D., Yu T.H. (2008). Use of US croplands for biofuels increases greenhouse gases through emissions from land-use change. Science 319:1238-1240. DOI: 10.1126/science.1151861.

Shaver T.M., Peterson G.A., Sherrod L.A. (2003). Cropping intensification in dryland systems improves soil physical properties: regression relations. Geoderma 116:149-164. DOI: 10.1016/s0016-7061(03)00099-5.

Shaver T.M., Peterson G.A., Ahuja L.R., Westfall D.G., Sherrod L.A., Dunn G. (2002). Surface soil physical properties after twelve years of dryland no-till management. Soil Science Society of America Journal 66:1296-1303.

Sheehan J., Aden A., Paustian K., Killian K., Brenner J., Walsh M., Nelson R. (2003). Energy and Environmental Aspects of Using Corn Stover for Fuel Ethanol. Journal of Industrial Ecology 7:117-146.

Sherrod L.A., Peterson G.A., Westfall D.G., Ahuja L.R. (2003). Cropping intensity enhances soil organic carbon and nitrogen in a no-till agroecosystem. Soil Science Society of America Journal 67:1533-1543.

Sherrod L.A., Peterson G.A., Westfall D.G., Ahuja L.R. (2005). Soil organic carbon pools after 12 years in no-till dryland agroecosystems. Soil Science Society of America Journal 69:1600-1608. DOI: 10.2136/sssaj2003.0266.

Shukla M.K., Lal R., Ebinger M. (2006). Determining soil quality indicators by factor analysis. Soil & Tillage Research 87:194-204. DOI: 10.1016/j.still.2005.03.011.

Sissine F. (2007). Energy Independence and Security Act of 2007: A Summary of Major Provisions., in: C. R. Service (Ed.), Washington, DC.

Somerville C., Youngs H., Taylor C., Davis S.C., Long S.P. (2010). Feedstocks for Lignocellulosic Biofuels. Science 329:790-792. DOI: 10.1126/science.1189268.

Stehfest E., Heistermann M., Priess J.A., Ojima D.S., Alcamo J. (2007). Simulation of global crop production with the ecosystem model DayCent. Ecological Modelling 209:203-219. DOI: 10.1016/j.ecolmodel.2007.06.028.

Stone K.C., Hunt P.G., Cantrell K.B., Ro K.S. (2010). The potential impacts of biomass feedstock production on water resource availability. Bioresource Technology 101:2014-2025. DOI: 10.1016/j.biortech.2009.10.037.

Thornton P.E., Running S.W., White M.A. (1997). Generating surfaces of daily meteorological variables over large regions of complex terrain. Journal of Hydrology 190:214-251.

Unger P.W. (1978). Straw-mulch rate effect on soil-water storage and sorghum yield. Soil Science Society of America Journal 42:486-491.

Wilhelm W.W., Johnson J.M.F., Karlen D.L., Lightle D.T. (2007). Corn stover to sustain soil organic carbon further constrains Biomass supply. Agronomy Journal 99:1665-1667. DOI: 10.2134/agronj2OO7.0150.

Wilhelm W.W., Johnson J.M.F., Hatfield J.L., Voorhees W.B., Linden D.R. (2004). Crop and soil productivity response to corn residue removal: A literature review. Agronomy Journal 96:1-17.

Wilson G.V., Dabney S.M., McGregor K.C., Barkoll B.D. (2004). Tillage and residue effects on runoff and erosion dynamics. Transactions of the Asae 47:119-128.

TABLES AND FIGURES

Table 1. Elevation and observed average climate properties for long-term, dryland cropping systems research locations at Sterling, Stratton, and Walsh, Colorado. Adapted from Sherrod et al., (2005).

Site	*Elevation (m)*	*Mean annual temperature (C°)*	*MAP (mm) **	*Days above 32 °C*	*OPE (mm)†*	*Deficit water (mm)‡*
Sterling	1341	9.3	425	42	1600	-1160
Stratton	1335	10.8	405	54	1725	-1310
Walsh	1134	12.2	400	64	1975	-1580

*MAP = mean annual precipitation from 1985-2007

†OPE = open pan evaporation during the growing season

‡Deficit water = precipitation – open pan evaporation

Table 2. Soil physical and hydraulic properties for the top two soil horizons by landscape position for the Sterling, Stratton, and Walsh, Colorado long-term dryland cropping system sites. Adapted from Ascough et al., 2010.

Soil horizon name	Horizon depth (cm)	Bulk density (33kPa) (g cm^{-3})	Sand (%)	Clay (%)	Total organic carbon (kg ha^{-1})	Total nitrogen (kg ha $^{-1}$)	Field capacity water content (33 kPa) (%)	Wilting point water content (1500 kPa)	Ksat * (cm h^{-1})
Sterling summit (Weld loam - fine, mixed, mesic Aridic Argiustoll)									
Ap1	0-8	1.37	45	21	11,674	900	0.21	0.11	1.78
Bt1	8-20	1.35	33	31	14.592	1670	0.28	0.15	0.85
Sterling sideslope (Satanta loam – fine-loamy, mixed, mesic Pachic Argiustoll)									
Ap1	0-11	1.47	54	21	15,642	1321	0.22	0.11	1.60
Ap2	11-20	1.34	44	26	11,144	1060	0.29	0.13	1.09
Sterling toeslope (Albinas loam– fine-loamy, mixed, mesic Pachic Argiustoll)									
Ap1	0-7	1.31	42	18	15,009	1223	0.22	0.10	2.55
Ap2	7-18	1.49	47	20	12,210	1139	0.20	0.11	1.94
Stratton summit (Norka clay loam – fine-silty, mixed, mesic Aridic Argiustoll)									
Ap	0-13	1.35	25	34	20,553	1874	0.27	0.16	.80
Bt	13-39	1.26	20	36	33,987	3517	0.31	0.17	.78
Stratton sideslope (Richfield loam - fine, montmorillonitic, mesic Aridic Argiustoll)									
Ap1	0-10	1.36	41	20	14,994	1323	0.26	0.11	2.07
Ap2	10-18	1.33	35	28	8928	918	0.28	0.14	1.04
Stratton toeslope (Kuma loam – fine-silty, mixed, mesic Pachic Argiustoll)									
Ap	0-15	1.28	25	26	41,040	3866	0.28	0.13	1.45
Ab1	15-30	1.26	23	25	44,838	4738	0.30	0.13	1.64
Walsh summit (undefinted loams sand – fine-loamy, mixed mesic Aridic Ustochrept									
Ap	0-18	1.48	65	14	9954	1024	0.17	0.05	1.62
Bk1	18-40	1.49	66	18	7944	967	0.14	0.07	1.41
Walsh sideslope (undefined sandy loam – fine, montmorillonitic, mesic Ustollic Haplargid									
Ap	0-10	1.55	72	10	6552	588	0.15	0.05	1.98
BAk	10-20	1.50	57	20	6519	731	0.15	0.08	0.80
Walsh toeslope (Nunn sandy clay loam – fine, montmorillonitic, mesic Aridic Argiustoll									
Ap	0-13	1.32	38	24	17,391	1569	0.28	0.11	0.70
Ab	13-24	1.34	32	26	27,090	2752	0.27	0.12	0.60

*Ksat, saturated hydraulic conductivity (estimated with Rosetta)

Table 3. Climate and soil files used in RUSLE2 water erosion modeling at Sterling, Stratton, and Walsh.

Site	Location climate file	Landscape position	Soil
Sterling	Logan Co.	Summit	Weld Loam
	(R 16-18)	Side	Rago Clay Loam
		Toe	Norka Loam
Stratton	Kit Carson Co.	Summit	Norka Silt Loam
	(R 16-18)	Side	Richfield Silt Loam
		Toe	Kuma-Keith Silt Loam
Walsh	Baca Co.	Summit	Dalhart Sandy Loam
	(R 16-18)	Side	Dalhart Sandy Loam
		Toe	Nunn Clay Loam

Table 4. Date and description of management operations used in RUSLE2 water erosion simulations.

Date	Management operation
5/1/01	Plant corn (double disk opener)
10/5/01	Harvest; 50% standing stubble
4/15/02	Weed growth
5/20/02	Spray weeds
5/21/02	Weed growth
7/15/02	Spray weeds
7/16/02	Weed growth
9/1/02	Spray weeds
9/10/02	Drill wheat (single disk openers)
7/10/03	Harvest wheat (50% standing stubble)

Table 5. Mean and standard deviation of observed stover, grain, and total biomass production for corn and sorghum from 1986-2006 by soil position for the Sterling, Stratton, and Walsh Colorado long-term dryland cropping systems sites.

Slope	Stover (kg ha^{-1})	Grain (kg ha^{-1})	Total biomass (kg ha^{-1})
STERLING			
Summit	2480 ± 1580	2140 ± 1310	4620 ± 2480
Side	2550 ± 1140	2670 ± 1350	5220 ± 2350
Toe	3170 ± 1360	3500 ± 1730	6660 ± 2930
Site Average	2730 ± 1390	2770 ± 1560	5500 ± 2700
STRATTON			
Summit	2490 ± 1830	2460 ± 1750	4950 ± 3190
Side	2550 ± 1290	2490 ± 1330	5040 ± 2280
Toe	4210 ± 1690	4300 ± 1600	8500 ± 2830
Site Average	3080 ± 1780	3080 ± 1770	6170 ± 3210
WALSH			
Summit	2200 ± 1500	2510 ± 1810	4700 ± 2760
Side	2390 ± 1370	2430 ± 950	4820 ± 2090
Toe	2710 ± 1530	2710 ± 1110	5420 ± 2290
Site Average	2430 ± 1460	2550 ± 1330	4980 ± 2380
Average all Sites			
Summit	2390 ± 1620	2370 ± 1620	4760 ± 2780
Side	2500 ± 1250	2530 ± 1210	5030 ± 2200
Toe	3360 ± 1650	3500 ± 1620	6860 ± 2950
Site Average	2750 ± 1570	2800 ± 1570	5550 ± 2810

Analysis of Variance	Stover	Grain	Total Biomass
	P > F	P > F	P > F
Site	0.0006	0.0152	0.0007
Slope	<.0001	<.0001	<.0001
Year	<.0001	<.0001	<.0001

Table 6. Mean and standard deviation of observed stover, grain, and total biomass production for wheat from 1986-2006 by soil position for the Sterling, Stratton, and Walsh Colorado long-term dryland cropping systems sites.

Site and Slope	Stover (kg ha^{-1})	Grain (kg ha^{-1})	Total biomass (kg ha^{-1})
STERLING			
Summit	3260 ± 1370	1690 ± 750	4920 ± 1810
Side	3590 ± 1290	1750 ± 680	5340 ± 1690
Toe	4550 ± 1460	2160 ± 770	6710 ± 1750
Site Average	3800 ± 1470	1860 ± 760	5660 ± 1890
STRATTON			
Summit	3850 ± 1490	1890 ± 660	5740 ± 1880
Side	3270 ± 1710	1670 ± 720	4940 ± 2180
Toe	6360 ± 2440	2570 ± 1100	8650 ± 3040
Site Average	4470 ± 2320	2080 ± 940	6550 ± 2970
WALSH			
Summit	3030 ± 1350	1690 ± 580	4720 ± 1840
Side	3390 ± 1650	1810 ± 670	5200 ± 2190
Toe	4010 ± 1720	2260 ± 800	6270 ± 2320
Site Average	3480 ± 1610	1920 ± 720	5400 ± 2180
Average all Sites			
Summit	3400 ± 1430	1750 ± 670	5150 ± 1870
Side	3420 ± 1540	1740 ± 680	5160 ± 1990
Toe	5030 ± 2150	2370 ± 920	7400 ± 2680
Site Average	3950 ± 1880	1950 ± 820	5840 ± 2440

Analysis of Variance	Stover	Grain	Total Biomass
	P > F	P > F	P > F
Site	<.0001	0.0172	0.0001
Slope	<.0001	<.0001	<.0001
Year	<.0001	<.0001	<.0001

Table 7. RUSLE2 predicted annual soil loss levels with 0% and 50% of wheat and corn stover removed at harvest at the Sterling, Stratton, and Walsh, Colorado long-term dryland cropping systems sites.

Site	No removal (Mg ha^{-1} yr^{-1})				50% removal (Mg ha^{-1} yr^{-1})			
	Summit	Side	Toe	Average	Summit	Side	Toe	Average
Sterling	0.13	0.29	0.09	0.17	0.27	0.70	0.23	0.40
Stratton	0.15	0.53	0.08	0.25	0.36	1.40	0.23	0.66
Walsh	0.18	0.41	0.10	0.23	0.39	0.96	0.21	0.52
Soil average	0.15	0.41	0.09		0.34	1.02	0.22	

Table 8. Observed and simulated mean carbon in wheat and corn grain and stover at Sterling and Stratton, Colorado. Model fit statistics are relative error, and normalized objective function (NOF).

Location	Observed Mean	Simulated mean	Root mean squared error	Relative Error	NOF
	(g C m^{-2})	(g C m^{-2})	(g C m^{-2})	(%)	(unitless)
Sterling					
Wheat grain yield	78	74	14	-5.4	0.18
Winter stover	139	122	52	-12.6	0.37
Corn grain yield	134	138	41	2.4	0.30
Corn stover	117	116	42	-0.7	0.36
Stratton					
Wheat grain yield	93	101	34	8.6	0.36
Wheat stover	164	163	56	-0.6	0.34
Corn grain yield	172	165	38	-4.1	0.22
Corn stover	156	156	42	0.0	0.27

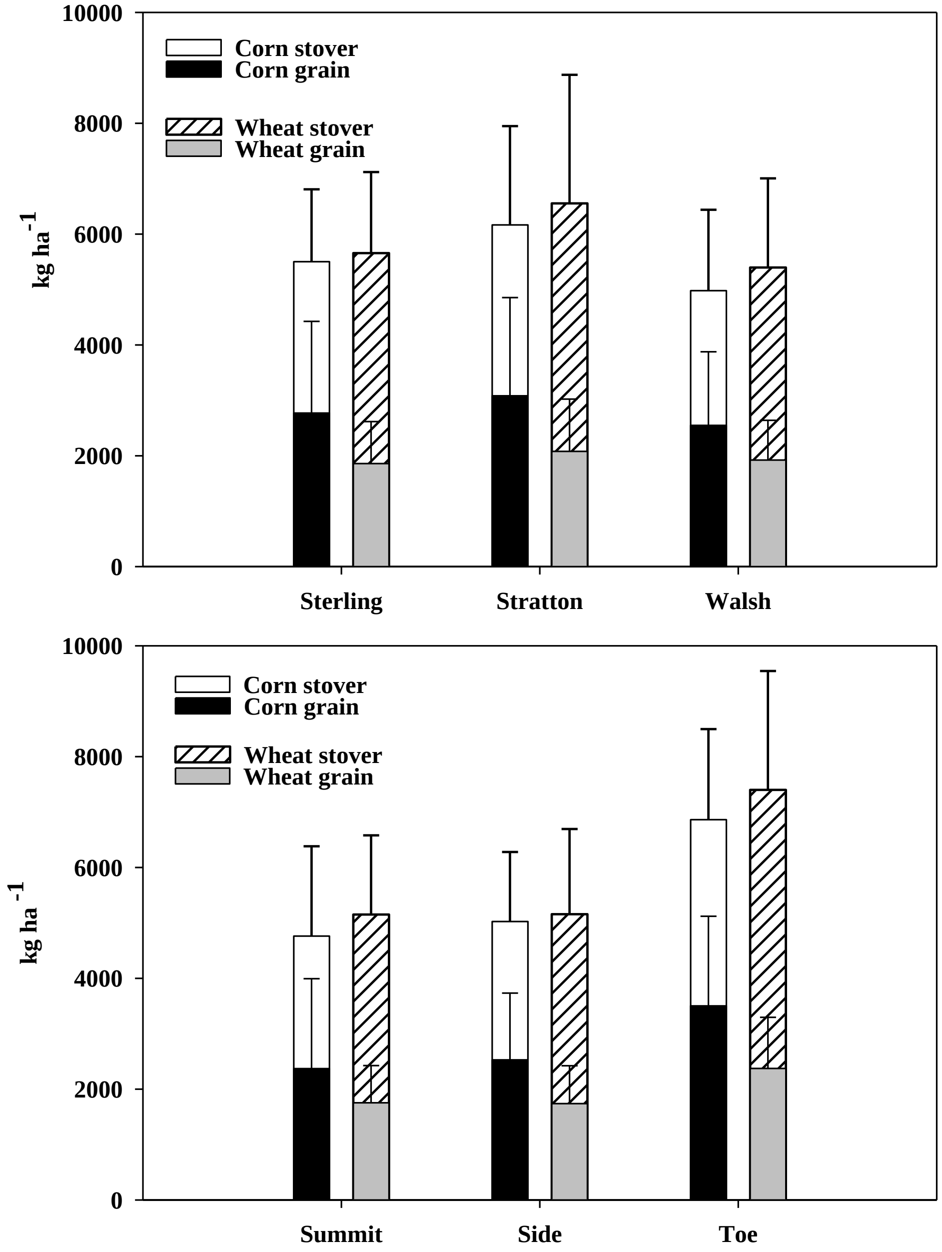

Figure 1. Observed grain and stover production averaged from 1986-2008 at Sterling, Stratton, and Walsh Colorado in long-term dryland cropping systems for corn (grain sorghum at Walsh) and wheat (upper panel) and average grain and stover by soil position, averaged over all sites (lower panel). Bars represent one standard deviation, given separately for grain and stover components.

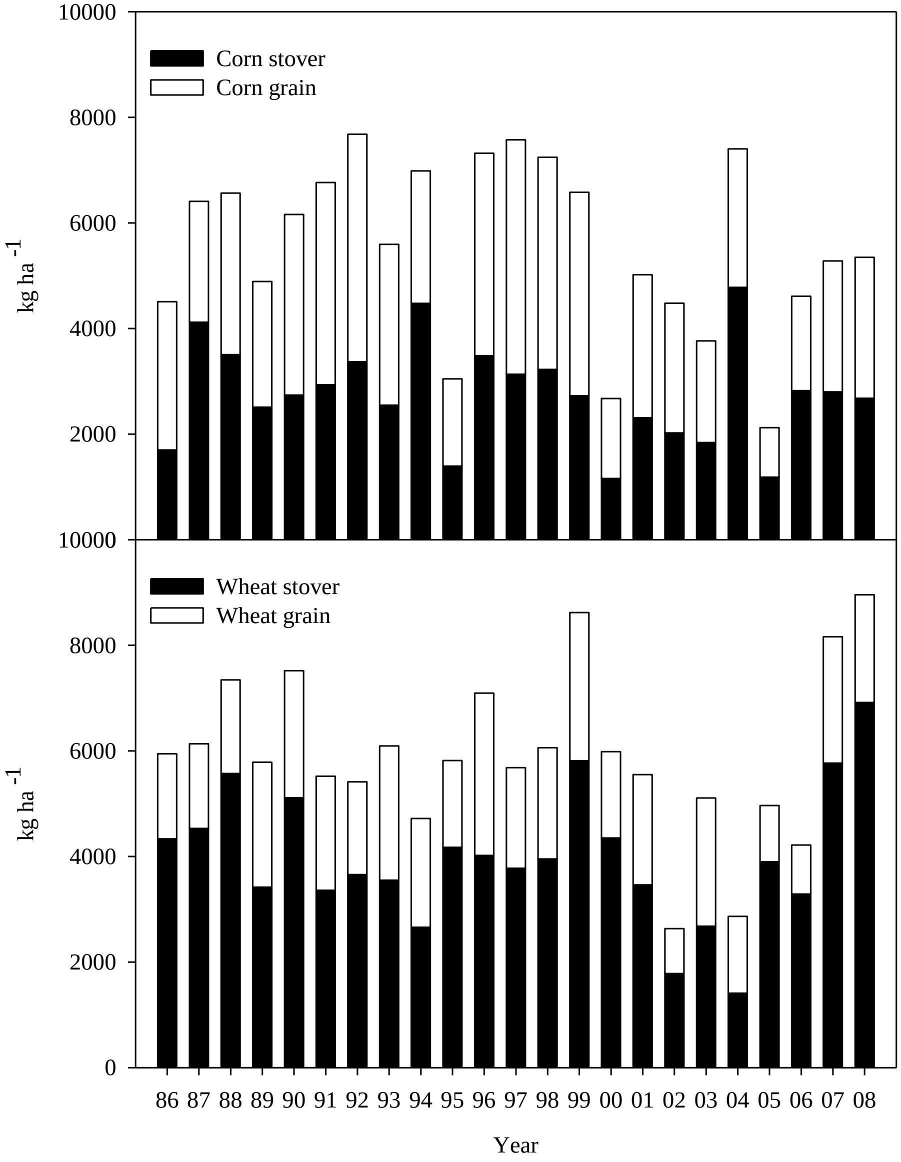

Figure 2. Variation in annual grain, stover, and total biomass production from 1986-2006 averaged over the Sterling, Stratton, and Walsh, Colorado long-term dryland cropping system sites for corn (upper) and wheat (lower).

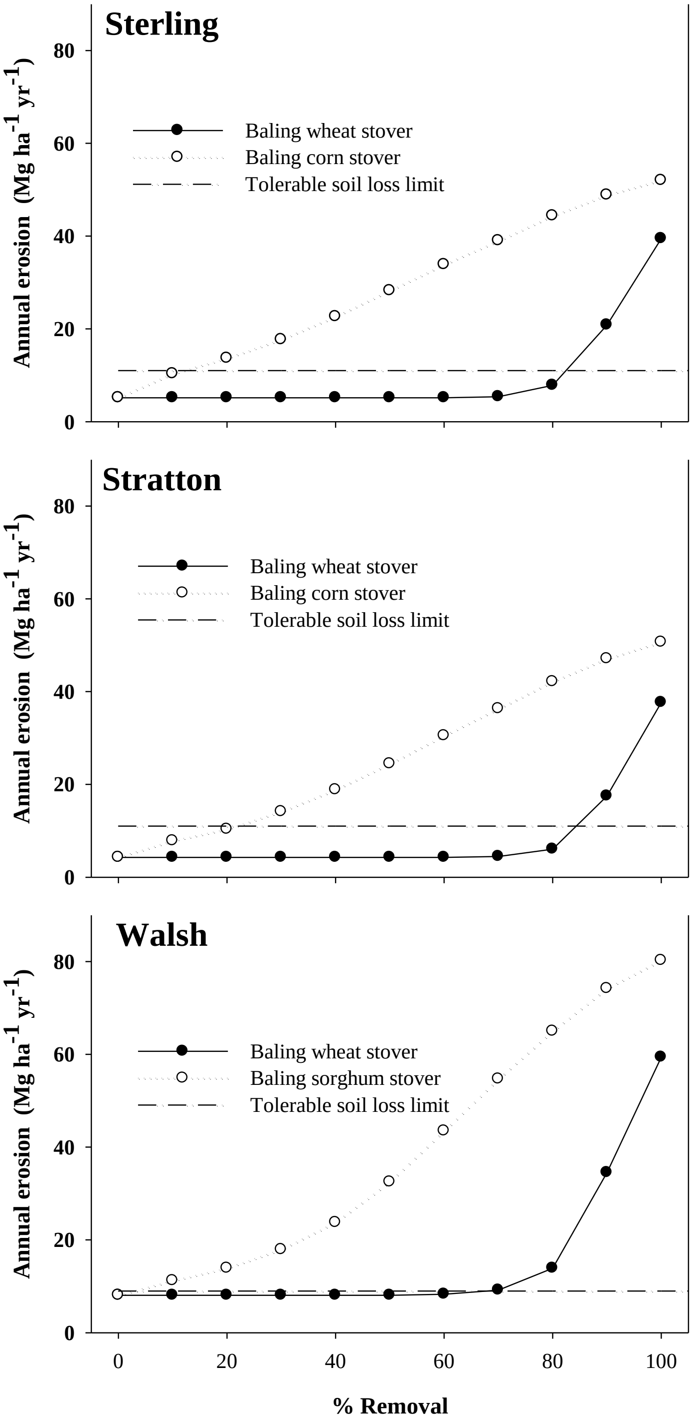

Figure 3. Predicted annual wind erosion rates under different levels of wheat, corn, and sorghum stover removal in long-term dryland cropping systems sites at (A) Sterling, (B) Stratton, and (C) Walsh, Colorado

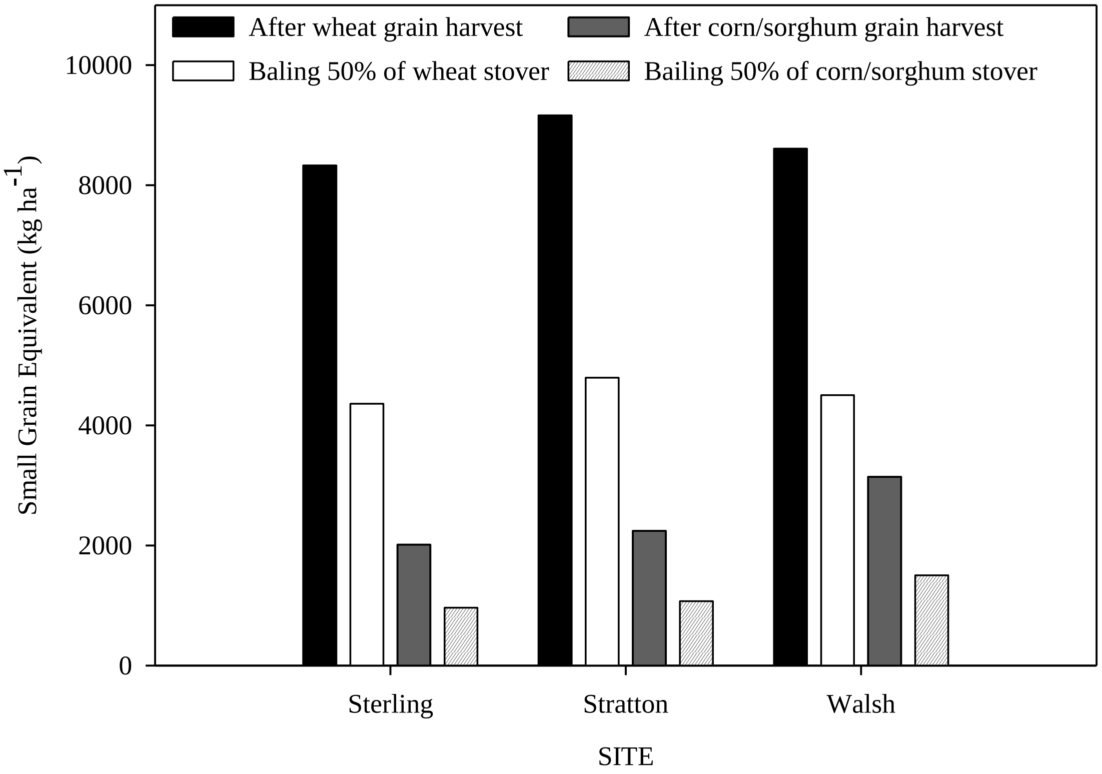

Figure 4. Small grain equivalent (SGe) for wheat, corn, and sorghum (Walsh) at harvest and after baling 50% of stover at Sterling, Stratton, and Walsh.

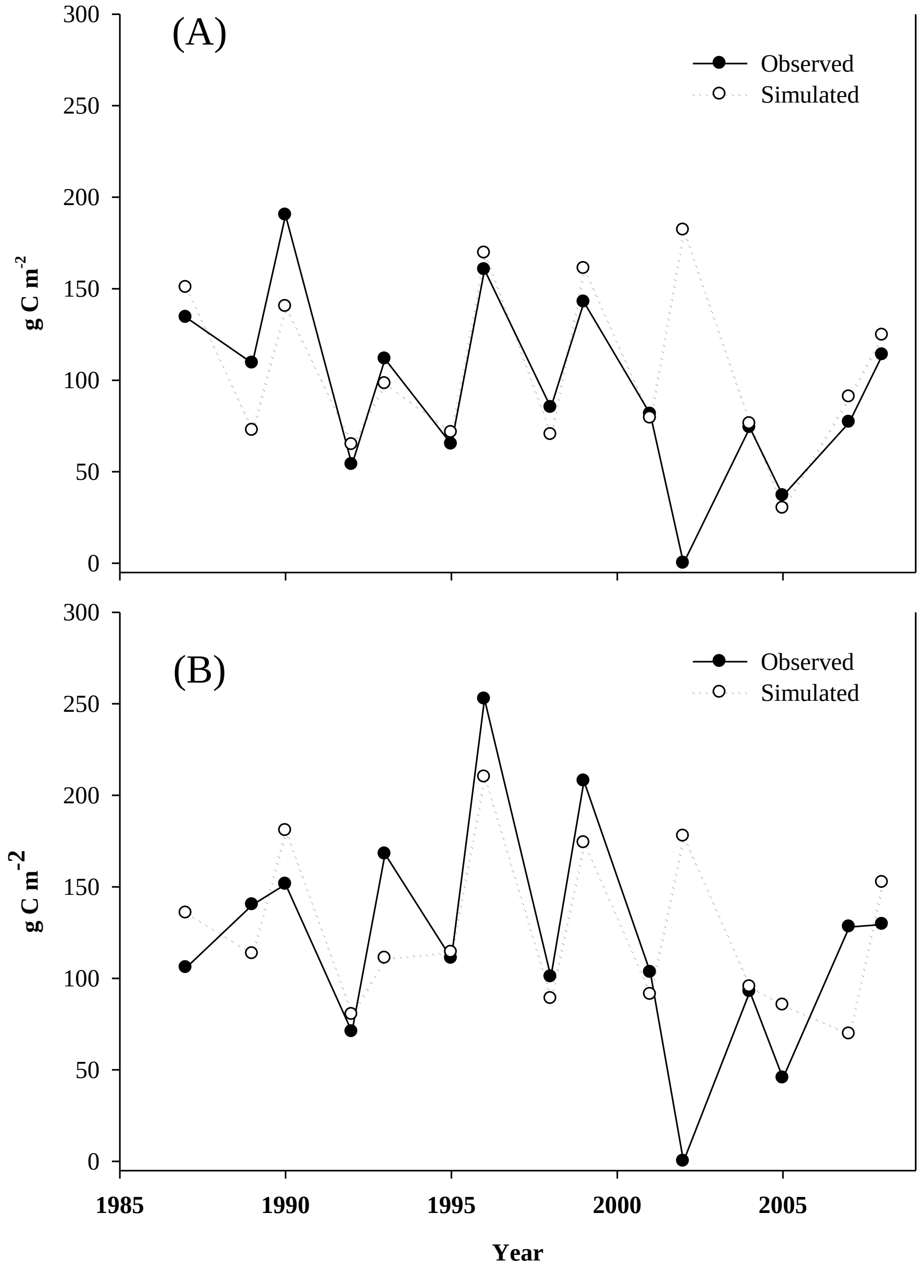

Figure 5. Observed and simulated carbon in harvested grain (wheat and corn) from 1986-2006 at (A) Sterling and (B) Stratton, Colorado long-term dryland cropping system sites.

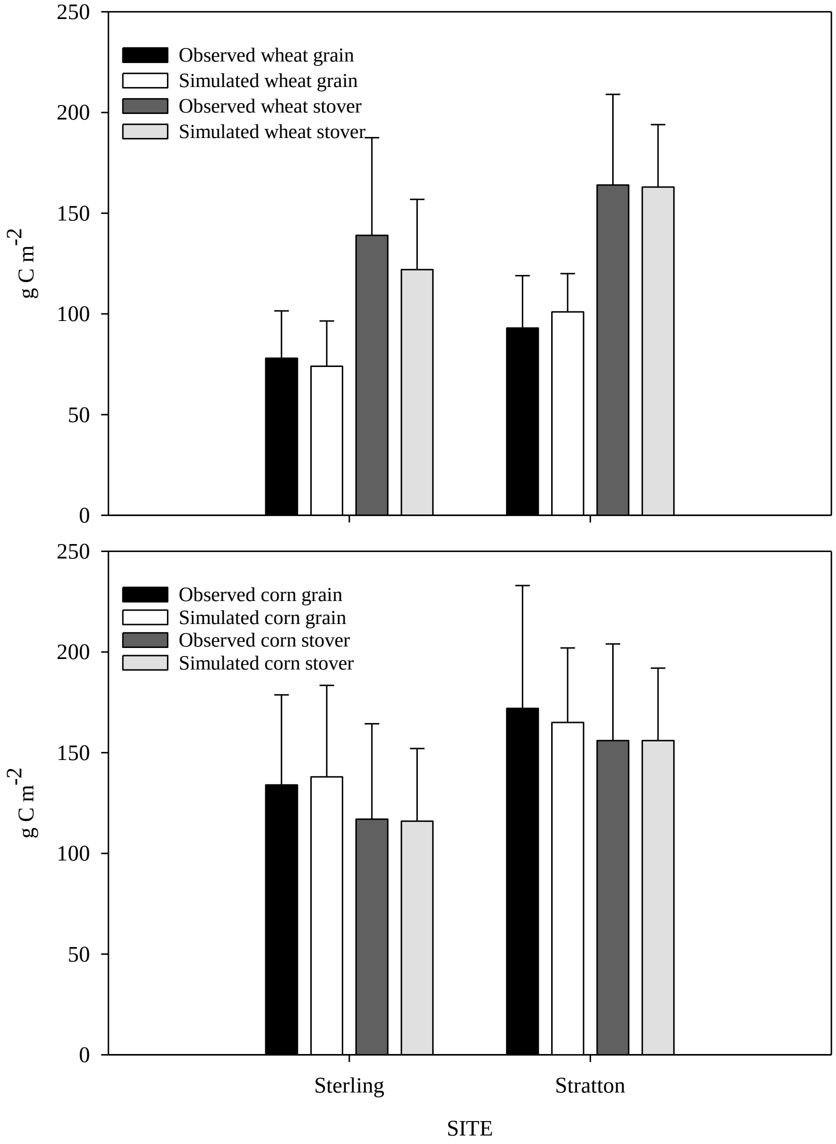

Figure 6. Mean observed and simulated carbon in grain and stover for wheat (upper) and corn (lower). Error bars show one standard deviation for winter wheat (n=14) and corn (n=13).

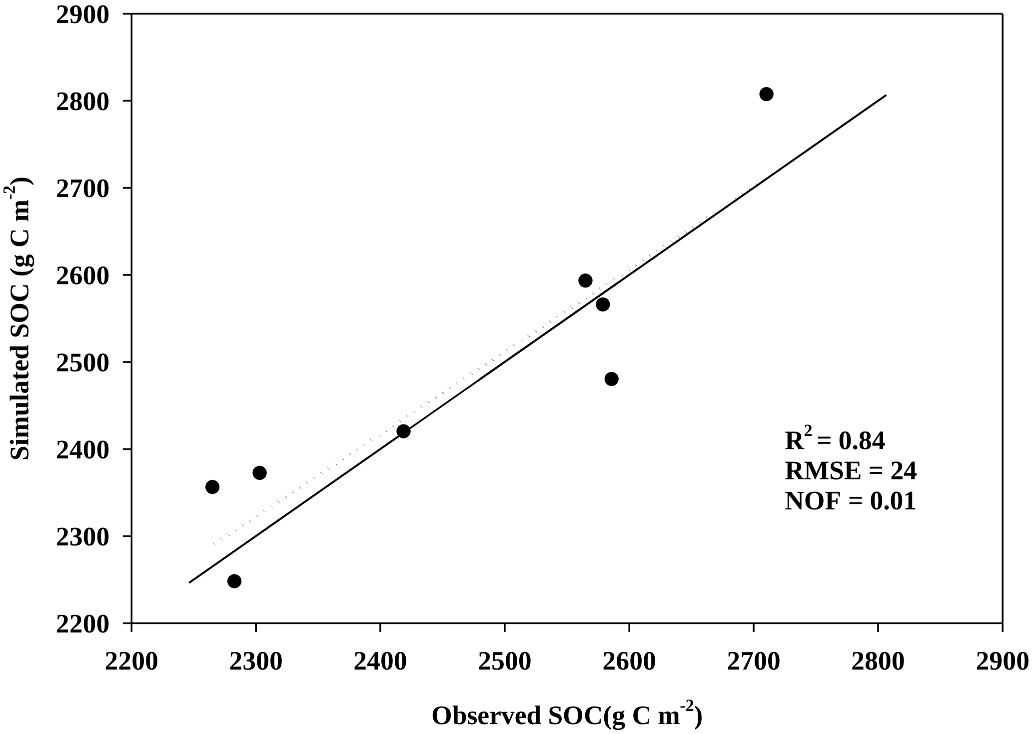

Figure 7. Observed versus simulated soil organic carbon (SOC) at Sterling and Stratton, Colorado long-term dryland cropping systems sites for the years 1986, 1997, 2001, and 2006. Each point represents the mean value from three landscape positions. Solid line is a 1:1 line.

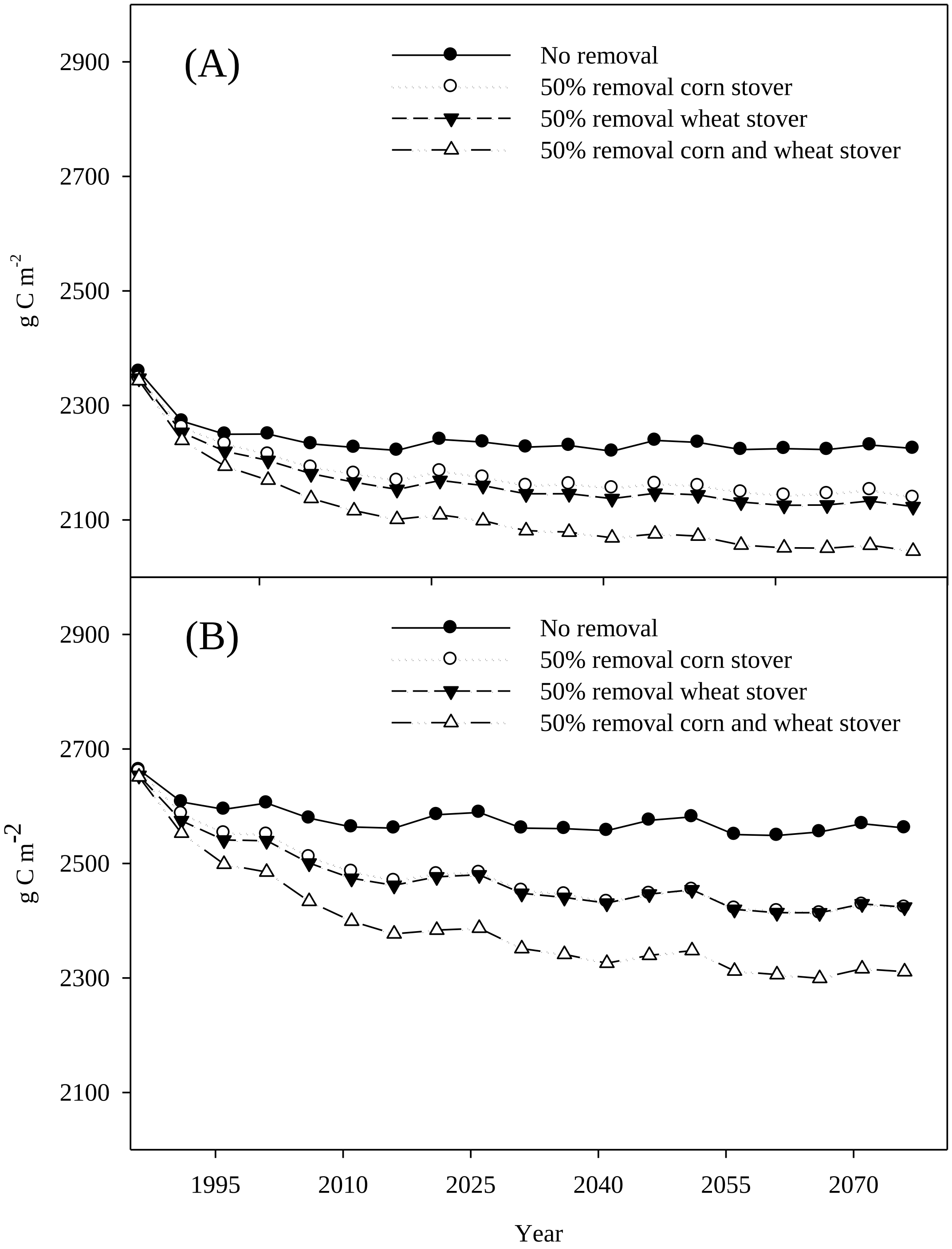

Figure 8. Simulated site level changes in SOC levels from 1986 to 2080 with 0% and 50% of corn and/or wheat stover at (A) Sterling and (B) Stratton. Data is presented as a 5-year moving average.

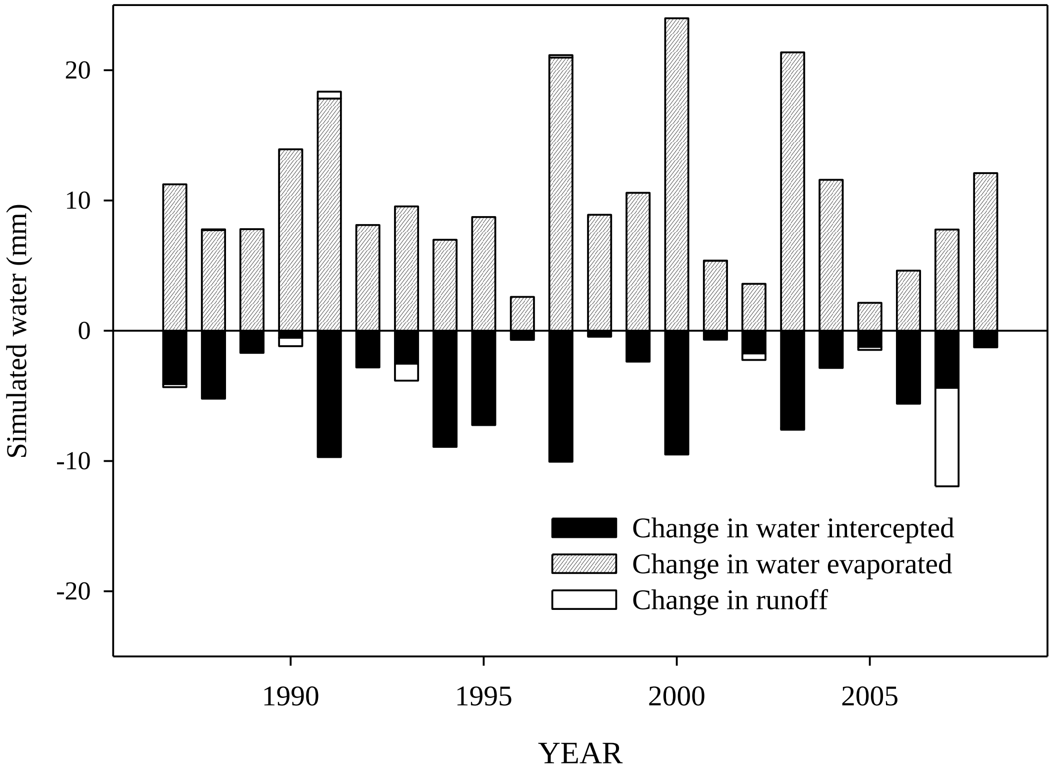

Figure 9. Simulated change in the amount of runoff, intercepted, and evaporated water with 50% stover removal for the Stratton, Colorado long-term dryland cropping systems site.

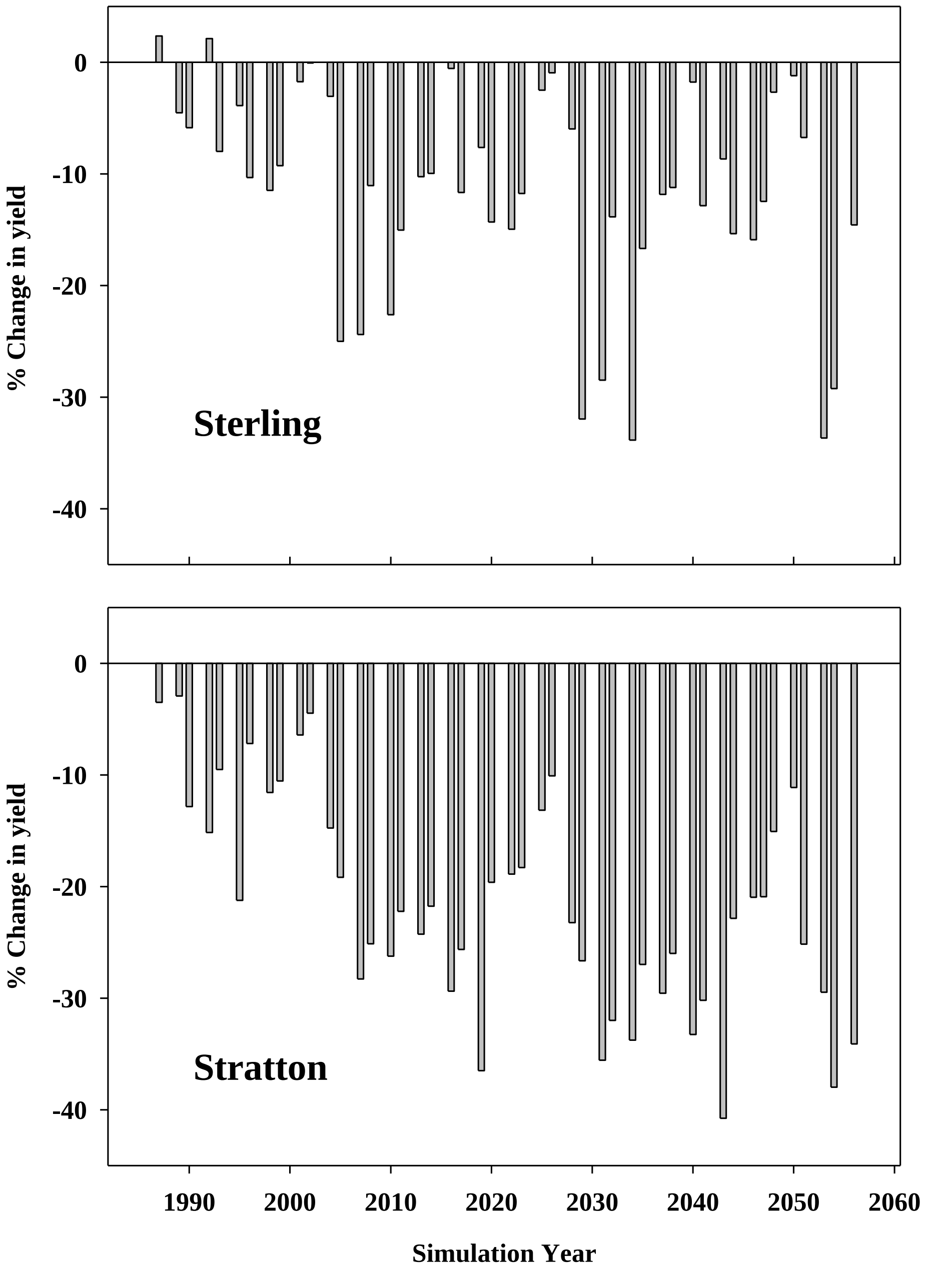

Figure 10. Change in grain yield with 50% removal of corn and wheat stover at Sterling (upper) and Stratton (lower). Percent change is expressed relative to simulated yields with no stover removal.

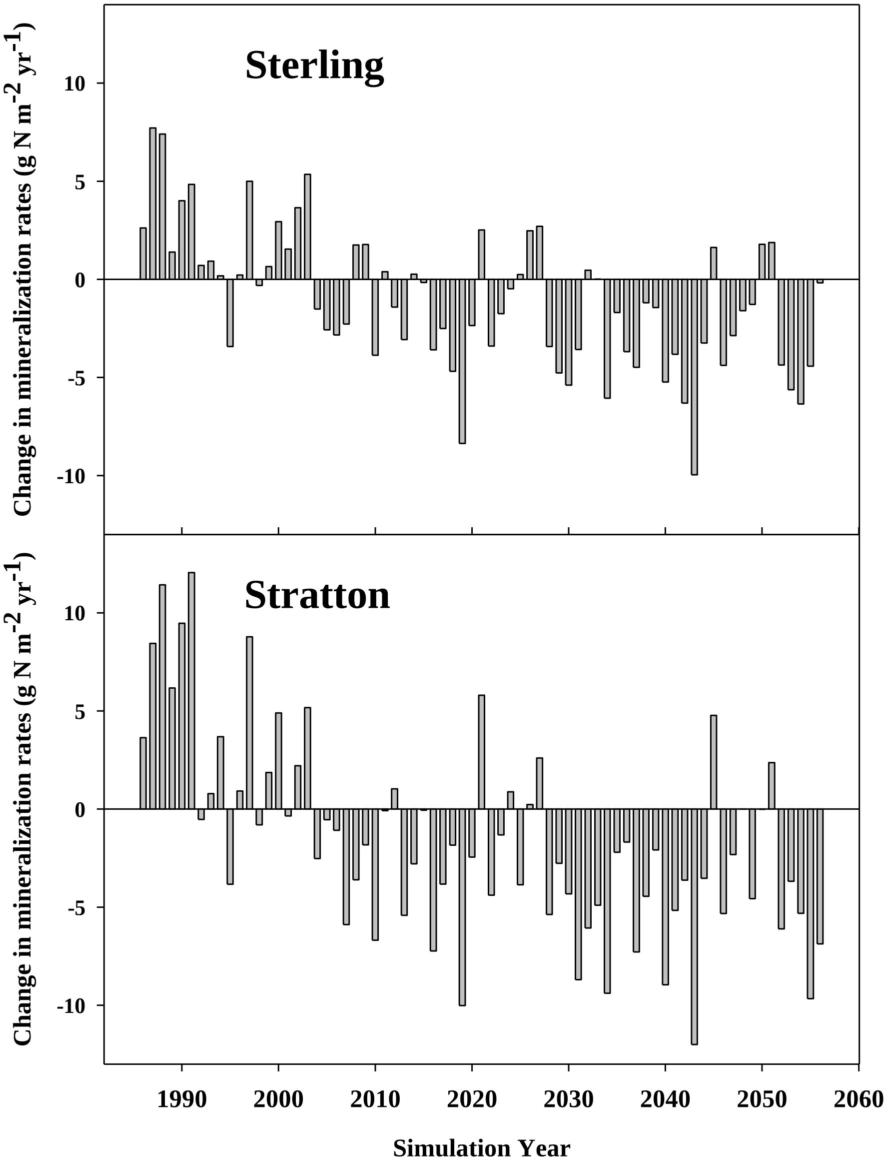

Figure 11. Change in annual rate of N mineralization at (A) Sterling and (B) Stratton with 50% removal of corn and wheat stover. Percent change is expressed relative to mineralization rates with no stover removal.

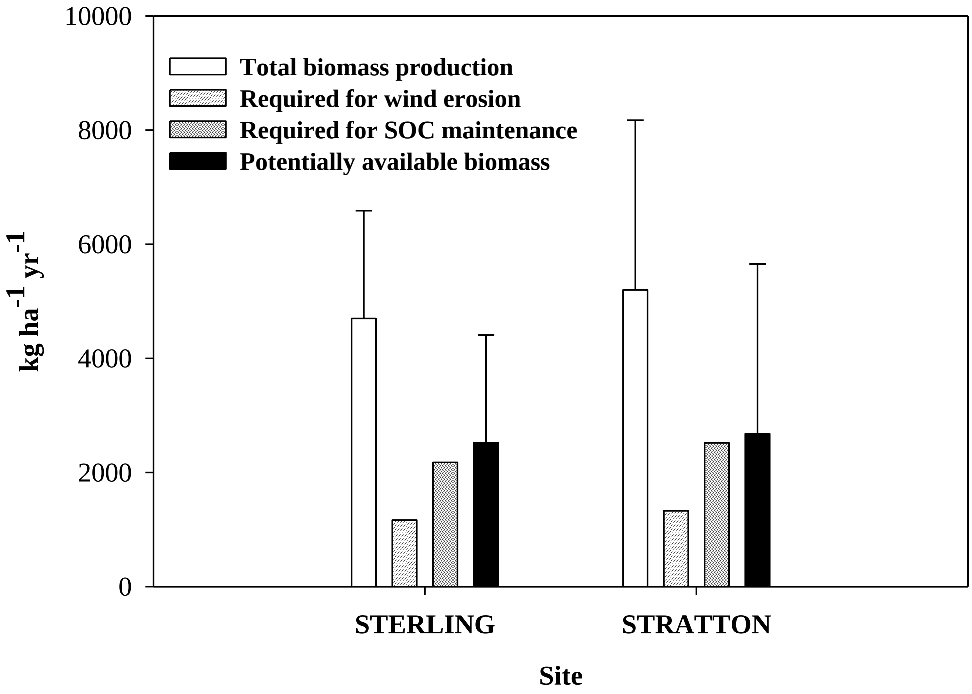

Figure 12. Potential production of dedicated biomass crop, residue required to control wind erosion, residue required for SOC maintenance, and potentially available biomass. Potentially available biomass is estimated by annualizing total wheat and corn biomass, adding 1000 kg ha^{-1} for increased yield of non-grain crop, and subtracting residue that must remain to maintain SOC in a WCF system. Error bars represent annualized standard deviation of total biomass production for wheat and corn.

CHAPTER 2: TEMPORALLY AND SPATIALLY VARIABLE RESPONSES TO STOVER HARVEST

INTRODUCTION

Crop residues are receiving attention as potential feedstocks for lignocellulosic biofuels (Farrell et al., 2006; Lynd et al., 2009; Perlack et al., 2005). Removal of crop residues for biofuel could result in removal of nutrients, increased erosion, reduced fallow efficiency, and losses of soil carbon (Graham et al., 2007; Johnson, 2007; Lal, 2009; Mann et al., 2002; Wilhelm et al., 2007; Wilhelm et al., 2004). Surface crop residues are a direct source of crop nutrients; continuous removal can result in depletion of soil fertility (Blanco-Canqui and Lal, 2009) and increase levels of sediment and nutrient runoff (Blanco-Canqui et al., 2009; Wilson et al., 2004). In areas where soil erosion is problematic, residues protect the soil surface from wind and raindrop impact. Water-limited regions need residues to help capture and store water, moderate soil temperature, and maintain infiltration rates (Peterson and Westfall, 2004; Unger, 1978).

Ensuring that the development of the biofuels infrastructure and industry does not impede the future productivity of our soils and reduce ecosystem services is a great challenge. Predicting the impact of removing agricultural residues is a critical task, as non-sustainable residue removal could result in soil degradation and decreased food security (Huggins et al., 2011; Lal, 2009). Crop production and SOC are clearly influenced by residue removal, yet the pattern and magnitude of impact is not consistent - some studies show negative yield and SOC declines with stover harvest while others show no change or slight increases (Wilhelm et al., 2004). In a short term study of stover removal's impact on corn yield, Blanco-Canqui et al., (2006) found that removing stover resulted in yield declines at one of the three sites examined, while yields at the other two sites were not impacted. Tarkalson et al., (2011) found that wheat and barley straw could be removed from irrigated production systems without a decline in SOC, but rain-fed systems required approximately 4 Mg ha^{-1} of straw be returned to maintain SOC. The mixed results of stover removal on SOC and crop yield are influenced by differences in climate, soils, and management. Studies on the regional impacts of stover harvest are very important to indicate the capabilities and constraints of specific geographic areas, as well as studies comparing average field responses. While it very important to understand the magnitude and direction of yield, fertility, or soil carbon change on regional, county, and field scales, it is also important to examine responses within one field or landscape. A sustainable bioenergy landscape will require spatially and temporally specific management.

Numerous agroecosystem models have been developed to predict changes in SOC and nutrient dynamics under different management practices and land use change (Powlson et al., 1996). Models are used to create national inventories on potential soil C sequestration or greenhouse gas emission by coupling them with geospatial databases. Agroecosystem models are used to inform local and national policy, as well as to predict the impacts of shifts in climate or management. Models have the ability to expand field studies that are performed on single combinations of soil, climate, and cropping systems, making it possible to expand observed results to different soils and climactic conditions (Bondeau et al., 2007; Ogle et al., 2005). While models are very useful in scaling up local information, if residue harvest is to be implemented sustainably, tools are also needed to predict how SOC, water, and nutrient availability will be impacted at finer spatial detail across a heterogeneous farm-scale landscape.

Agricultural fields often contain an amalgam of different soil types and topography with variable fertility and hydrology. Variation in soil types, topography, and prior and current land use history create landscape level differences in soil carbon, water, and nitrogen dynamics that can be quite large (Dharmakeerthi et al., 2005). Differences in cropping systems and management methods will also affect C:N ratios of remaining soil residue and soil mineralization rates. Using averaged field data will not be sufficient to represent the spatial and temporal variability of these dynamic processes. For example, harvesting stover at a single static rate (determined 'sustainable' on a site average basis for that production system) may result in site specific soil degradation in some areas that are more vulnerable than others to residue removal. A better understanding of field and landscape level differences in soil carbon, hydrology, and fertility could help farmers to assess whether biofuel production or stover harvest is sustainable on their land as well as guide land management decisions to ensure that biomass harvest does not have unacceptable impacts on soil quality and productivity.

Little attention has been given to the impacts of residue harvest in the semi-arid Great Plains, largely due to low production levels, and the strong erosive forces of wind. However, expanding interest in growing dedicated biofuel crops on marginal lands makes examination of the capabilities and constraints of cropping systems in the semi-arid Great Plains a relevant task. In addition, the Great Plains is a major production region for wheat and other forages being considered as potential feedstocks (Sarath et al., 2008).

The objective of this study was to simulate the variable impacts of stover removal on soil carbon, soil fertility, and grain yields on a field landscape-level basis, using data from a long-term field study of dryland cropping systems in the semi-arid Great Plains to drive model simulations. Results were used to infer the temporally and spatially explicit impacts of stover removal on yield, fertility, and soil organic carbon levels in semi-arid agricultural systems.

MATERIALS AND METHODS

Overview of the Long-Term Study

The data used in this study comes from 24 years of research conducted at two sites in the West Central Great Plains of eastern Colorado, for a total of 72 site years. This study was initiated in 1985 to identify the response of selected soil chemical and physical properties, water-use efficiency, and crop production to various no-till dryland cropping systems (Peterson et al., 1993). The experiment includes three variables: (1) climatic gradient, represented by site differences in precipitation and PET, (2) soil properties (e.g., plant available water) across a landscape position, and (3) cropping intensity. The two sites used in this study are located near Sterling (40.37°N, 103.13° W) and Stratton (39.18°N, 102.26°W) in eastern Colorado. An automated weather station at each site measured daily maximum and minimum air temperature, precipitation, mean relative humidity, total solar radiation, wind direction, and wind speed throughout the duration of the experiment.

This experiment was a split-split-block design. Site location was the first block, and landscape position the sub-block. The experiment included multiple cropping systems, all managed with no-till techniques. Each experimental unit was an individual soil within a site and within a cropping system. Each cropping system phase was assigned randomly in strips within each block and extended over a sequence of soils. All phases of each rotation were present in two replications each year. For example, in a wheat-corn-fallow rotation, every phase of the rotation (wheat, corn, and fallow phases) was replicated twice at each site, making a total of 4 plots for this system at Sterling and Stratton. Each experimental unit was an individual soil within a site and within a cropping system. All experimental units were 6.1 m wide, but varied in length between particular sites (185 to 305 m). Although a variety of cropping systems were evaluated in the long-term study (cite), only the wheat-corn-fallow (WCF) cropping system was examined in this study. Full details of the experimental design can be found in Peterson et al., (1993).

The study sites run along a north to south climatic gradient of increasing potential evapotranspiration (PET). Selected climatic properties, location, and elevation for the two sites are shown in Table 9. Both sites have approximately the same long-term annual precipitation, but increase in PET from north to south. The difference between annual precipitation and open pan evaporation represents the water deficit.

Along with climatic differences, each site has unique topography. Each site was laid out on a soil catena consisting of a summit, side, and toe soil position (Fig. 13). Each landscape position was comprised of a unique soil mapping unit with a distinct soil textural class. Soil classifications, as well as physical and hydraulic properties for the top two soil horizons by site and landscape position are given in Table 10. Reported saturated hydraulic conductivity values were calculated using a pedotransfer function tool (Schaap et al., 2001).

The experimental study sites were intentionally laid out across soil catenas with variable soil properties, fertility, and hydrology, and are hence highly suitable for examining landscape effects of stover harvest. The catenary sequence at Sterling has a maximum elevation difference of 5.9 m with transition zones between the summit/sideslope and the sideslope/toeslope. The transition zone between the sideslope and toeslope has a slight ridge that can act as a water catchment, resulting in lateral surface flow during some rainfall events. The most complex topography of the two locations is found at Stratton, where the sideslope is a steep slope (approximately 4%) that bottoms into a toeslope that is a three-way drainage area from the east, south, and west slopes. The toeslope at Stratton has produced the highest crop yields across the two experimental sites, due to the funnel effect that delivers run-on water, nutrients, and depositional material. Toeslope landscape positions generally have higher levels of carbon and nitrogen than summit and sideslope positions, with the Stratton toeslope containing approximately two times the total soil carbon and nitrogen than the toeslope position at Sterling. This is largely due to sediment deposition and run-on water. Full details of the unique topography of each site can be found in Ascough et al., (2010).

Biomass production

Grain and stover yields from a wheat-corn-fallow rotation were evaluated for their long-term production and compared to model simulations. Even though other annual non-grain species may be promising as biomass crops, the current evaluation of the grain crops from the long-term study illustrates the potential and limitations of biomass production in semi-arid dryland environments.

All systems in the long-term dryland cropping system study were managed with no-till techniques. Herbicides were chosen to permit crop rotations with no carryover problems. Nitrogen fertilizer was applied annually at planting in accordance with soil tests obtained from each soil and the crop present in a given year. Phosphorus was band-applied at planting of all crops at a rate of approximately 9.5 kg ha $^{-1}$. Grain yields were measured with a plot combine by harvesting an area approximately 30 m x 1.2 m in the middle of each experimental unit. Total aboveground biomass samples were collected by hand, oven dried, weighed, and separated into grain and stover (all harvested non-grain biomass). Harvest index (dry mass grain yield/total aboveground biomass) was determined from the hand samples. In addition, grain yields were obtained on entire plots using a plot combine. Grain samples were collected from the combine for each experimental unit to determine moisture content and test weight. The total biomass values were obtained by dividing grain yields from the combine by the ratio of grain mass to stover mass found in the hand-collected biomass sample.

For some site-years, there were no measured biomass yields. For example, in site-years where extreme drought resulted in minimal grain yield, harvest was not performed and consequently, no biomass data was collected. No wheat biomass yields were measured at Stratton in 1987 and 2004. The reported biomass statistics for corn do not include data from 2002. In addition, one year of documented herbicidal damage was not used in model comparisons.

DAYCENT Modeling

The DAYCENT model was used to estimate changes in grain yield, soil fertility, and soil organic carbon (SOC) with 50% residue harvest on a soil specific basis (three soil types at each experimental site). DAYCENT is the daily-time-step version of the CENTURY model. DAYCENT and CENTURY are generalized ecosystem models that simulate carbon, nitrogen, and phosphorus dynamics in grassland, forest, savanna, and agricultural ecosystems. DAYCENT models decomposition, nutrient flows, soil water, and soil temperature on a finer time scale than CENTURY, and has increased spatial resolution for soil layers. Both CENTURY and DAYCENT have been extensively tested in agricultural, grassland, and forest systems using observed data on soil organic matter, production levels, nutrient cycling, and trace gas fluxes (Del Grosso et al., 2002a; Del Grosso et al., 2002b; Kelly et al., 1997; Parton et al., 1994; Parton et al., 1998; Paustian et al., 1992; Stehfest et al., 2007).

Model Description

DAYCENT operates on a daily time step for nutrient cycling, water flow, and soil organic matter turnover, but uses a weekly time step submodel for plant production. Soil organic carbon (SOC) and nitrogen (N), and N mineralization rates are calculated for the 0-20 cm soil depth. The soil organic matter (SOM) submodel computes the cycling of above and belowground dead plant material into the active, slow, and passive soil organic matter pools. The SOM submodel also estimates nutrient mineralization rates, which are dependent on dead plant material decomposition rates and turnover rates of soil organic matter pools. First order decomposition rate constants are modified by soil temperature and moisture, tillage management practices, soil texture, and litter quality. Potential plant production and nutrient uptake are calculated as a function of soil water stress, leaf area index, soil temperature, and incoming solar radiation. Actual plant production is limited according to soil nutrient availability. The land surface submodel simulates water flow and evapotranspiration for the plant canopy, litter, and soil profile, along with soil temperature (Parton et al., 1998). Full model and submodel descriptions can be found in Parton et al., (1987, 1994).

DAYCENT uses historical schedule files, generated by the user, along with soil and weather data to predict crop yields. Daily precipitation and daily maximum and minimum temperatures are used as weather driver inputs. Detailed soil input variables include texture, bulk density, field capacity, wilting point, pH, and saturated hydraulic conductivity on a soil layer basis.

Model Setup

Soil files for each landscape position (summit, side, and toe) at each site were created from NRCS primary characterization data, and can be found in Tables 1-6 of Appendix A. To initialize the SOM and nutrient pools, two spin-up simulation periods were run prior to the start of the long-term experiment in 1985. From model year zero through model year 1934, a grazed native grassland system of 50% warm season grasses and 50% cool season grasses was simulated. For the 50-year period from 1935-1985, a winter wheat-fallow system was modeled using varieties with low to medium harvest indices and low levels of nitrogen inputs, simulating conventional practices in the region for that time period. In 1985, the DAYCENT crop and vegetation files were changed to reflect the shift to no-till management. Long-term site-specific records of annual planting and harvest dates were used to make schedule files as precise as possible, along with historical site-specific annual fertilizer rates. Planting dates ranged from September 14th to October 2nd for wheat, with a median planting date of September 23rd. Harvest dates for wheat ranged from July 5th to July 15th, with a median harvest date of July 12th. Corn planting dates ranged from May 4th to June 1st (exceptionally late year of planting), with a median planting date of May 16th. Corn harvest dates ranged from October 7th to October 29th, with a median harvest date of October 20th. Nitrogen fertilizer rates ranged from 2.2 - 11.3 g N m^{-2}, and were based on measured soil NO^{-}_{3} concentration and expected crop yields.

For the spin-up and base periods, one hundred years of repeating daily climate data from the Vegetation/Ecosystem Modeling and Analysis Project (VEMAP; resolution 0.5 degrees latitude x 0.5 degrees longitude) and DAYMET (resolution 1 km^2) was used from the centroid of the counties of the experimental sites (Kittel et al., 2004; Thornton et al., 1997). For the period from 1985-2008, climatic data collected from the on-site weather stations at each experimental location was used.

No model equations were adjusted. One crop parameter that controls harvest index (HIMAX), which was adjusted slightly to better represent long-term average observed harvest index values (HIMAX = 0.55 for corn and 0.4 for wheat). Otherwise, parameter constants controlling crop growth were left unchanged, as the model has been previously tested in dryland conditions with wheat and corn (Del Grosso et al., 2002a). Crop files with parameter values can be found in supplementary Tables 7-10 in Appendix A.

In the long-term wheat-corn-fallow rotations, all phases of the rotation are present in any given cropping year. Hence, we created accurate schedule files for the wheat-corn-fallow and corn-fallow-wheat rotations in order to have more years of simulated grain yields to compare with observed grain yields. Each schedule file was created for the 24 year period of 1985-2008. First, simulations were run with a schedule file reflecting actual management (no residue removal). Separate simulations were run for the three landscape positions (summit, sideslope, toeslope) at each site. Model output from each soil position was compared with observed yields. To compare observed and simulated stover yields, we assumed a carbon concentration of 0.40 for both wheat and corn stover. Grain yields from the first simulation year (1986) were excluded in the comparison with observed yield, based on the fact some tillage occurred during site establishment and soil water conditions were impacted.

A neutron attenuation probe was used to measure soil water content at planting and harvest at each site and cropping system in 30 cm increments throughout the soil profile. To validate the DAYCENT water submodel, ten years of soil water data were compared with model simulated soil water content (1988-1998). For some field positions (i.e., the toeslope), there were more observed neutron probe data points (Sterling toeslope, n = 37; Stratton toeslope, n = 38; Stratton summit, n= 43) for comparison, while at other soil positions, there were few readings (Stratton side, n = 25; Sterling summit, n=36; Sterling side, n = 22).

Field soil sampling for total carbon occurred numerous times during the 24-year experiment, but not always to the depth of 20 cm, the increment simulated by DAYCENT. Four years of observed total soil carbon (0-20 cm) were compared to simulated total soil carbon for 1986, 1997, 2001, and 2006. Methods of carbon sample preparation and analysis can be found in Sherrod et al., (2003). Observed carbon is reported as the mean of two replicates taken from the same treatment. In some years, samples were obtained from more than one phase of the same treatment (i.e., from W-C-F, and C-F-W). In these cases, the mean of both phases was used for comparison with simulated. In addition, DAYCENT reports simulated total soil carbon on a monthly basis. Because field sampling for observed carbon always occurred between May and September, values for annual simulated soil carbon is reported as the mean of values from this 5-month period. The combination of sites, soil type, crop rotation phase, and year of carbon determination gave a total of 24 individual comparisons of modeled and observed SOC data points.

After validating simulations with observed data, new schedule files were created for three different scenarios: (1) 50% removal of corn stover at harvest, (2) 50% removal of wheat stover at harvest, and (3) 50% removal of both corn and wheat stover. The first two scenarios result in stover harvest once every three years, while the third scenario has a stover harvest two out of every three years. The new schedule files were identical to the base schedule files, save for the addition of a stover harvest event. To project the long-term impacts of stover harvest, the 24-year schedule files were repeated as blocks for a total of 96 years. Observed weather data from 1985-2008 was repeated in the long-term projections so that changes in yield, carbon, or fertility due to management shifts (i.e., stover harvest) would be clearly distinct from weather-driven changes. Hence, model simulations do not simulate shifts in weather or atmospheric CO_2 concentration.

The long-term impacts of stover removal on soil fertility, soil carbon, and grain yields were inferred by examining grain yields (g C m^{-2}), net annual mineralization rates (g N m^{-2} yr^{-2}), and SOC (g C m^{-2}) with and without stover removal for three different soil types on each site.

Evaluation Statistics

Two model evaluation statistics were calculated to quantify the DAYCENT model performance: (1) Relative error (RE), which shows the bias of the predicted mean relative to the observed mean, and (2) a normalized objective function (NOF), based on the root mean square error (RMSE), which shows the average deviation between predicted and observed values, regardless of sign. Relative error is expressed:

$$RE = \frac{\left(\overline{\mathrm{P}} - \overline{\mathrm{O}}\right)}{\overline{\mathrm{O}}} * 100$$

Where $\overline{P}$ is the predicted mean and $\overline{O}$ is the observed mean. If the model under-predicts, the RE will have a positive bias, and if the model over-predicts, a negative RE indicates a negative model bias. Because RE is an arithmetic average over the duration of data, it is an indication of long-term bias. While simple in interpretation, bias alone is not sufficient to explain model errors. For example, a RE of near zero could be the result of very small model errors in all situations or of large errors in some years that cancel out years of over and under prediction. The root mean square error (RMSE) eliminates the problem of compensation between over and under prediction by averaging the squared differences, hence weighting large differences more heavily. The RMSE is often more convenient to work with than the mean square error, because the RMSE has the same units as the response variable. The RMSE is calculated as:

$$RMSE = \sqrt{\frac{\sum_{\mathrm{i}=1}^{\mathrm{n}}(\mathrm{P} - \mathrm{O})^2}{\mathrm{n}}}$$

Where P_i is the *i*th predicted value, O_i is the *i*th observed value, and *n* is the number of data pairs. The NOF (unitless) is calculated as:

$$NOF = \frac{\mathrm{RMSE}}{\overline{\mathrm{O}}}$$

where RMSE and $\overline{O}$ are as previously defined. An NOF of zero indicates a perfect fit between experimental data and simulated results, while an NOF of less than 1 can be interpreted as a simulation error of less than 1 standard deviation around the experimental mean.

RESULTS AND DISCUSSION

Crop grain and stover yield

Long-term observed average grain and stover yield for each site by soil position can be found in table 3. Yields were significantly influenced by soil position (see Chapter 1), with the highest yields observed at the toeslope. Average yield by soil position at Sterling ranged from 2140 – 3500 kg ha^{-1} for corn stover and from 2480 – 3170 kg ha^{-1} for corn grain. Average wheat grain production at Sterling ranged from 1690 – 2160 kg ha^{-1}; average wheat stover yields ranged from 3260 – 4550 kg ha^{-1}. At the Stratton location, average corn stover production was 2490 kg ha^{-1} at the summit, 2550 kg ha^{-1} at the sideslope, and 4200 kg ha^{-1} at the toeslope. Average corn grain by soil position yields ranged from 2460-4300 kg ha^{-1}. Average wheat stover yields by soil position were between 3850 – 6360 kg ha^{-1}, and average wheat grain ranged from 1890 – 2570 kg ha^{-1}.

Observed and simulated yields followed similar patterns through the 24 year period at both the Sterling and Stratton locations (Figs. 14 and 15). The first 16 years of the experiment (1986 – 2000) were years of good or above average precipitation, while 2001-2008 were years of drought and below average precipitation. In wetter years, predictions are reasonable, but in years of low precipitation or drought stress, model predictions are poorer.

On average, yields were over predicted for wheat grain (RE = 18.5%) and strongly over predicted for corn grain (RE = 43%) at the Sterling summit. At the Sterling sideslope, wheat grain was under predicted by -13.5%, and corn grain was slightly under predicted (RE = -4.3%). At the Sterling toe, yields were generally underestimated, with a relative error of -9.2 and -17.3 % for wheat and corn grain, respectively. At Stratton, a similar trend in simulated yields was observed (Fig. 3). Wheat and corn grain yields were overestimated at the summit position for wheat (RE = 13.9%, and underestimated for corn grain (RE = -11.0%). Yields were over predicted at the sideslope for both wheat grain (RE = 22.6%) and corn (RE = 8.0%). At the toeslope, wheat and corn grain yields were underestimated (RE = -3.6 % and -6.4%).

Examination of Figures 14 and 15 shows that the model was unable to accurately simulate a crop failure that occurred in 2002, a year of severe drought with approximately 250 mm of total precipitation. Reduced yields due to drought in 2004-2005 at Stratton were also not well simulated. The failure of the model to simulate low grain yields in years with exceptionally low amounts of precipitation (<250 mm) was also noted by Alvaro-Fuentes et al., (2009). Dryland environments in eastern Colorado are characterized by periods of extreme water and temperature stresses. In some years, water stress may advance to the point that the crop dies and is unable to respond to a subsequent rain event. This type of water stress situation is very difficult to model. In most years, DAYCENT simulations matched observed grain yields better than expected given the high stress situation. DAYCENT is not a crop growth model, but if it is adopted for use as a predictive tool in dryland environments, the sensitivity of the model to drought stress should be improved.

Soil Water

Simulated soil profile water at the Sterling summit position was quite good, with a relative error of 0.7% (Table 12). Soil profile water was underestimated at both the side and toe positions at Sterling, with the model underestimating water in the 0-120 cm depth by 14.8 and 27.7% at the side and toe, respectively. This trend of underestimation was consistent in the 0-60 and 0-90 cm depths for both soil positions. This is likely due to the fact that the DAYCENT model is not spatially linked with processes that account for slopes, gradients, or transfer of water from one landscape position to the next via run-on. Underestimation of simulated soil water partly explains the underestimation of yield at the side and toe positions.

The overestimation of crop yields at the Sterling summit, despite good model simulation of total soil profile water content, could be due to the models handling of the unique soil profile at the Sterling summit. Soil texture at the summit position changes from a clay content of 21% at the surface, to 31% in the 8-20 cm depth, to 38% in the 20-30 cm depth. This fine texture clay layer greatly reduced water infiltration into the soil. At around the 85 cm depth, there is a marked increase in lime content, which mixes with the clay to create a partially cemented zone. This zone is slowly permeable to water and impermeable to plant roots. DAYCENT did simulate higher levels of runoff at this soil position than the other two, but could not account for the reduction in yields due to low plant available water, or the inability of plant roots to penetrate the partially cemented lime layer. Overestimation of wheat yield at the summit could be attributed to the fact that the simulated soil water content relative error was 1.3% in the 0-120 cm depth, but was 14.5% for the 0-60 cm depth. As the majority of crop roots would be found in these shallower depths, overestimation of soil water in the top of the profile may explain overestimation of yields at the summit and side.

At Stratton, soil water was again quite good at the summit (RE = 1.3%) and sideslope (RE = 0.6%), but strongly underestimated at the toeslope (RE = -28.9%). No relative error is given on a site average basis, due to the fact that field readings of soil moisture by soil position were not always available on the same day and total number of readings varied by soil position.

Soil carbon

A similar pattern of over and under estimation by soil position was observed for total soil carbon. At both Sterling and Stratton, water and wind forces erode carbon rich soil and plant residues from the summit position and redeposit much of it at the toeslope soil position. DAYCENT cannot simulate landscape level processes of redistribution of water and carbon to other soil positions.

Simulated and observed carbon levels correlated quite well at Sterling (Fig. 16). After running the spin-up and base files, initial simulated SOC conditions at each soil position correlated quite well with observed data. Simulated changes in carbon levels also tracked closely in direction and magnitude with observed data. There was a small overestimation of simulated carbon level at the summit, and an underestimation of simulated carbon at the toeslope.

The complex topography and water dynamics at Stratton described earlier created difficulty in simulating initial SOC to match the carbon levels observed at the start of the experiment in 1985. After running the spin-up and base simulations, initial simulated carbon levels at the summit were nearly 20% higher than observed SOC, and initial carbon at the toe was 24% lower than observed. Throughout the simulation runs from 1985-2008, the model did simulate the proper direction of change. However, the inability of the model to simulate initial conditions led to poor correlation of observed and predicted SOC. Model simulations of soil carbon by soil position virtually flipped the observed carbon values, suggesting erosion and deposition processes have occurred. Summit SOC values were overestimated (Fig. 17; filled circles) and toeslope values were underestimated by DAYCENT (Fig. 17; filled triangles). The complex topography at this site results in run-on water and sediment coming into the toeslope from a larger contributing area than the experimental summit position. Field observations at this site show sediment being deposited at this position during heavy rainstorm events. Deposition results in increased carbon and nutrients, as well as higher productivity from the run-on water and nutrients. Higher productivity, and hence higher rates of carbon return, also increases carbon at the toeslope.

Spatial patterns of model fit

Simulation runs on a soil position basis over or under estimate grain yield, depending on landscape position (Fig. 18). Model agreement varied with site and slope position for grain and stover yields of wheat and corn (Table 12). At both Sterling and Stratton locations, model predictions had a positive bias when predicting corn and wheat yields in the summit position and had a negative bias for toeslope positions. The best model fit for grain and stover yields were those for the site average (Table 12). The site-averages of mean and observed simulated crop yields show that overall, the model does very well in simulating grain and stover yield on a long-term site average basis (Fig. 18). On a site-average basis, relative error ranged from an under prediction of 5.4% to an over prediction of 9.0% for wheat grain at Sterling and Stratton, respectively. Relative error was nearly zero for corn grain at Stratton, with a slight over prediction of 2.4% at Stratton. NOF values showed no clear pattern by crop or location. While acceptable, the relatively large NOF values are mainly due to overestimation of the model in years of drought, and underestimation in years of good precipitation, where simulated yields appeared to be limited by nitrogen. In addition, DAYCENT cannot account for reductions in yields via insect, hail, or herbicide damage.

DAYCENT performs very well at simulating changes in SOC at the site-level. When SOC is averaged over soil position, site-level relative error was -1.3% at Sterling (NOF = 0.01), and 0.0% at Stratton (NOF =0.03). When the Sterling and Stratton sites were combined to evaluate model fit for SOC predictions on a site average basis, the fit was nearly perfect (NOF=0.01; data not shown). The model cannot account for movement of carbon-bound sediment, or higher productivity at the toeslope due to run-on water, but as a majority of the carbon is being redistributed within slope positions at each site and not lost, observed and simulated carbon correlate well on a site-level basis. In addition, the overestimation of yield at the summit position and underestimation of yield at the toe position appear to cancel out when a site average is taken.

This examination of observed and simulated yields on a landscape basis demonstrates that there are important landscape processes that influence yield and SOC dynamics, but that the model did not account for. This is likely due to the inability of the model to account for explicit spatial processes such as water transfer between landscape positions, as demonstrated in the over and underestimation of soil water by soil position.

Yield impacts with 50% stover removal

Yields decreased at all sites and soil positions with simulated removal of 50% of corn and wheat stover (Fig. 19; Table 13). Model output shows the effect of residue removal superimposed on patterns of climate. Yield declines as great as 1615 and 1382 kg ha^{-1} were simulated on a soil type basis for corn and wheat, respectively. Simulated yield declines varied by site and soil position. The summit and side positions at Sterling show similar levels of average yield declines. Surprisingly, simulated stover harvest had a positive impact on yield in some years at the Sterling toeslope. Maximum simulated yield declines were higher at Stratton than at Sterling. The first 24 years of the simulation show a less severe decline than yields during the second 24 year block. After 48 simulation years, yield declines stabilized to a new lower yield level. Differences in patterns and magnitudes of simulated yield changes between sites and soils demonstrate the difficult nature of predicting the impacts of stover harvest.

Nitrogen Mineralization and Removal Rates

Decline in simulated yields with stover can be attributed mainly to changes in nitrogen mineralization rates. Nitrogen mineralization is the conversion of organic nitrogen to NH4+ through aminzation and ammonification, and occurs through the activity of heterotrophic microorganisms that require organic carbon for energy (Havlin et al., 2005). In any given year, if we simulated a situation of excess available nitrogen, only 1-2% yield declines were observed with residue removal. These small simulated declines are likely due to small changes in soil temperature (approx. 1-2° C) or miniscule changes in water loss via evaporation. Declines in annual mineralization rates as high as 13 g N m^{-2} yr^{-1} were observed, but simulated changes in mineralization rates appear to be highly variable between soil types, with net mineralization rates increasing with stover removal in some years (Fig. 20). N mineralization rates are a function of temperature, precipitation, C/N ratio of soil organic matter, pH, and soil texture, and thus show variability by soil and year.

At both Sterling and Stratton, an overall initial increase in mineralization rates lasting approximately 18-20 years can be observed (Fig. 20). Increased rates of mineralization are likely due to decreased N immobilization. Previous in situ nitrogen mineralization experiments at Sterling and Stratton found that total net N mineralization in W-C-F cropping system was half that of the W-F system (Kolberg et al., 1999). The authors attributed this to greater accumulation of crop residue carbon on the soil surface and surface layers in the more intensive W-C-F system, and hence greater N immobilization. Nitrogen immobilization rates are notoriously high for nitrogen-poor material such as cereal straw. Hence, it is likely that removal of stover initially reduces the C:N ratio, and promotes mineralization over immobilization.

After approximately 20 simulation years, a strong decline in mineralization rates can be observed. Because simulations were run in 24-year 'blocks', we were able to compare changes in mineralization between blocks. In block one, there was an increase in mineralization rates; mineralization rates strongly declined in block two, and by block three, mineralization rates stabilized at a new lower level. Simulated yield declines stabilized at virtually the same point as N mineralization (approx 40-50 years). These results are not surprising, as continually removing stover depletes both available N and C.

A notable point from this simulation is that one field unit typically possesses several soil types, and each of these soil types can be expected to respond differently to residue harvest. Net mineralization rates and plant available nitrogen are influenced by soil texture and structure, organic matter, pH, water content, temperature, and the C/N ratio of plant inputs. Differences in these properties between soils results in different rates of N mineralization, and different responses to stover harvest. Accurately predicting the rate at which nitrogen is made available via N mineralization is an extremely complicated task, dependent on management, climate, and variable soil properties. However, predicting soil specific responses to stover removal is crucial to the long-term sustainability of a biomass industry, which will rely on consistent and reliable biomass yields.

Predicting accurate spatial responses to stover harvest is also very important from a nitrogen management perspective. For example, if additional N fertilizer were applied at Sterling to compensate for N removed in the stover and reductions in N mineralization, the summit position would receive too little nitrogen to make up for reductions, while the toe position would receive excess N. Predictions of spatial variations in N mineralization rates and plant available nitrogen are only possible if there is temporary stability of the spatial patterns and factors influencing them. Dharmakeerthi et al. (2005) found that spatial patterns of plant available nitrogen were relatively stable over time, suggesting a temporal consistency. This suggests that if a field is managed with precision agriculture techniques, where spatial variability in soil is known and variable rate fertilization and harvest take place, yield losses due to reduced mineralization could be minimized or eliminated. However, anticipating dynamic N mineralization responses to stover harvest will be a difficult task, and care must be taken to prevent overcompensation with excess fertilizer.

Nutrients removed in stover have implications for soil fertility, and nutrients removed with stover harvest will have to be replaced to sustain yields. Examination of simulated amounts of N removed with stover shows a range in percent nitrogen of 0.9 - 2.1%, similar to the range of observed percent nitrogen in wheat and corn stover. The simulated amount of total nitrogen removed with 50% harvest of wheat and corn stover were dependent on crop yield and ranged from 0.1 – 1.0 g m^{-2} (1.0 – 10.0 kg ha^{-1}). Johnson et al. (2010) found that the average collective cost to replace nitrogen, phosphorus, and potassium removed in the stover was \$17.59 Mg^{-1} for above-ear stover, and \$18.11 Mg^{-1} for below ear stover. Sustainable stover harvest will require accurate accounting of the nutrients removed in the harvested stover, as well as the impact of reduced carbon inputs on rates of N mineralization.

Soil Carbon impacts with 50% stover removal

Simulated carbon trends at Sterling with no stover removal varied by soil position (Fig. 21). Initial declines observed in early simulation years represent a continuation of the decline under previous management (wheat-fallow system with tillage). A switch to no-till management and corresponding reduction in fallow frequency in 1986 results in stabilization of that decline within approximately five years (1991). All comparisons of the impacts of various stover harvest scenarios are based on comparison with carbon values five years after the start of the no-till management. With no stover harvest, simulations show stable levels of SOC at the summit soil position after 96 years of no-till management, with a less than a 1% decline in SOC from 1991 levels. Even with no stover harvest, simulations show a decline of 7% at the side position. It should be noted that carbon levels in these semi-arid soils are very low, around 0.8 % carbon in the 0-30 cm depth. Hence, even small declines in soil carbon can represent a large fraction of overall soil carbon. Simulation of a no-till WCF system for 96 years resulted in a 2% increase in soil carbon at the Sterling toe position. Simulated stover yields were underestimated at the sideslope position and overestimated at the summit position (Table 4), so we cannot confirm that C levels would in fact decline at the sideslope position. However, the varied soil responses emphasize the difficulty of predicting soil carbon dynamics with no stover harvest.

Simulated harvest of 50% of both wheat and corn stover decreased SOC at all soil positions, but the magnitude of change varied based on soil position. Compared to soil carbon levels with no stover harvest, over the course of the 96 year simulation 2.1 g C m^{-2} yr^{-1} were lost from the summit position, 2.5 g C m^{-2} yr^{-1} from the toe position, and 1.3 g C m^{-2} yr^{-1} from the side. Clearly, these lower annual rates at the side are due to the fact that the baseline comparison (soil C without stover harvest) was declining as well.

Yield declines also contribute to simulated losses of soil carbon with stover harvest, although yield reduction is not the primary driver of simulated SOC declines. Assuming a 1:1 ratio for grain and stover, a 10% reduction in grain yield means 10% less stover is produced (and a farmer would then be harvesting 50% of a smaller value). However, stover declines represent a very small carbon loss when compared with harvesting 50% of total stover. We simulated a 20% increase in fertilizer rates, enough to bring yields back to, or above, levels without stover harvest. Preventing yield declines mitigated SOC declines with stover harvest by approximately 2%, but did not halt the decline.

Variable responses to residue removal emphasize the need for landscape specific management. If a farmer were to harvest at one average rate based on the need to maintain SOC on a site average basis, it could result in higher levels of removal on some soils or field areas than can be sustained without SOC declines. The goal of any sustainable bioenergy economy fueled by agricultural feedstocks must be to minimize or eliminate carbon losses and degradation across all portions of the field landscape.

CONCLUSIONS

This modeling exercise highlights several of the agronomic issues facing bioenergy feedstock production. Spatial variations in soil organic carbon, nitrogen mineralization, and erosion rates must be determined before the economic and ecological sustainability of agricultural stover harvest can be accurately assessed at the field level. Landscape scale predictive models are needed to assess the impact of stover removal on soil fertility and SOM dynamics and transfer processes. However, simulations characterized by a single combination of soil and climate are not sufficient to account for the complexity of soils or the interactions and feedbacks of sediment, nutrient, and water transport. A fully integrated framework for modeling SOM in a three-dimensional, spatially explicit manner that is representative of the landscape has yet to be fully developed, but great steps are being made (Viaud et al., 2010). Such models could be important management and decision-making tools in a bioenergy production system.

In addition to predictive model support systems, sustainable field management of variability will likely require adoption of precision agriculture technologies and practices such as variable rate harvesting and fertilization. As noted by Cruse and Herndl (2009), sustainable biomass harvest will require the ability to identify acceptable (and spatially variable) removal rates across a field or landscape, implement harvest technology that removes residue at these variable rates, and return nutrients using variable rate technology. Developing spatially sensitive harvest techniques requires that we know both the spatial variability in a field, and how each variable unit will respond to stover harvest. Concern over maintenance of soil organic carbon should be coupled with predictions about how N mineralization and other soil fertility variables will be influenced by removal of stover. There is need for precision harvest equipment that can measure and map crop nutrient removal in addition to monitoring yields. This type of technology could be used in conjunction with variable-rate nutrient application equipment to insure that yields are not impacted by reduced soil fertility.

LITERATURE CITED

Alvaro-Fuentes J., Lopez M.V., Arrue J.L., Moret D., Paustian K. (2009). Tillage and cropping effects on soil organic carbon in Mediterranean semiarid agroecosystems: Testing the Century model. Agriculture Ecosystems & Environment 134:211-217. DOI: 10.1016/j.agee.2009.07.001.

Ascough J.C., Andales A.A., Sherrod L.A., McMaster G.S., Hansen N.C., DeJonge K.C., Fathelrahman E.M., Ahuja L.R., Peterson G.A., Hoag D.L. (2010). Simulating landscape catena effects in no-till dryland agroecosystems using GPFARM. Agricultural Systems 103:569-584. DOI: 10.1016/j.agsy.2010.06.005.

Blanco-Canqui H., Lal R. (2009). Corn Stover Removal for Expanded Uses Reduces Soil Fertility and Structural Stability. Soil Science Society of America Journal 73:418-426. DOI: 10.2136/sssaj2008.0141.

Blanco-Canqui H., Lal R., Post W.M., Owens L.B. (2006). Changes in long-term no-till corn growth and yield under different rates of stover mulch. Agronomy Journal 98:1128-1136. DOI: 10.2134/agronj2006.0005.

Blanco-Canqui H., Stephenson R.J., Nelson N.O., Presley D.R. (2009). Wheat and Sorghum Residue Removal for Expanded Uses Increases Sediment and Nutrient Loss in Runoff. Journal of Environmental Quality 38:2365-2372. DOI: 10.2134/jeq2009.0021.

Bondeau A., Smith P.C., Zaehle S., Schaphoff S., Lucht W., Cramer W., Gerten D. (2007). Modelling the role of agriculture for the 20th century global terrestrial carbon balance. Global Change Biology 13:679-706. DOI: 10.1111/j.1365-2486.2006.01305.x.

Cruse R.M., Herndl C.G. (2009). Balancing corn stover harvest for biofuels with soil and water conservation. Journal of Soil and Water Conservation 64:286-291. DOI: 10.2489/jswc.64.4.286.

Dominguez-Faus R., Powers S.E., Burken J.G., Alvarez P.J. (2009) The Water Footprint of Biofuels: A Drink or Drive Issue? Environmental Science & Technology 43:3005-3010. DOI: 10.1021/es802162x.

Del Grosso S., Ojima D., Parton W., Mosier A., Peterson G., Schimel D. (2002a). Simulated effects of dryland cropping intensification on soil organic matter and greenhouse gas exchanges using the DAYCENT ecosystem model. Environmental Pollution 116:S75-S83.

Del Grosso S.J., Ojima D.S., Parton W.J., Mosier A.R., Peterson G.A. (2002b). Regional assessment of net greenhouse gas fluxes from agricultural soils in the USA Great Plains under current and improved management Millpress Science Publishers, Rotterdam.

Dharmakeerthi R.S., Kay B.D., Beauchamp E.G. (2005). Factors contributing to changes in plant available nitrogen across a variable landscape. Soil Science Society of America Journal 69:453-462.

Farrell A.E., Plevin R.J., Turner B.T., Jones A.D., O'Hare M., Kammen D.M. (2006). Ethanol can contribute to energy and environmental goals. Science 311:506-508. DOI: 10.1126/science.1121416.

Graham R.L., Nelson R., Sheehan J., Perlack R.D., Wright L.L. (2007). Current and potential US corn stover supplies. Agronomy Journal 99:1-11. DOI: 10.2134/argonj2005.0222.

Havlin, J.L., Beaton, J.D., Tisdale, S.L., Nelson, W.L., 1999. Soil Fertility and Fertilizers. An Introduction to Nutrient Management. Sixth ed., Prentice Hall, Upper Saddle River, New Jersey.

Huggins D.R., Karow R.S., Collins H.P., Ransom J.K. (2011). Introduction: Evaluating Long-Term Impacts of Harvesting Crop Residues on Soil Quality. Agronomy Journal 103:230-233. DOI: 10.2134/agronj2010.0382s.

Johnson J.M.-F., Coleman Mark D., Gesch Russ, Jaradat Adbullah, Mitchell Rob, Reicosky Don, Wilhelm W.W. (2007). Biomass-bioenergy crops in the United States: a changing paradigm. The American Journal of Plant Science and Biotechnology Vol. 1(1): 1-28

Johnson J.M.F., Wilhelm W.W., Karlen D.L., Archer D.W., Wienhold B., Lightle D.T., Laird D., Baker J., Ochsner T.E., Novak J.M., Halvorson A.D., Arriaga F., Barbour N. (2010). Nutrient Removal as a Function of Corn Stover Cutting Height and Cob Harvest. Bioenergy Research 3:342-352. DOI: 10.1007/s12155-010-9093-3.

Kelly R.H., Parton W.J., Crocker G.J., Grace P.R., Klir J., Korschens M., Poulton P.R., Richter D.D. (1997). Simulating trends in soil organic carbon in long-term experiments using the century model. Geoderma 81:75-90.

Kittel T.G.F., Rosenbloom N.A., Royle J.A., Daly C., Gibson W.P., Fisher H.H., Thornton P., Yates D.N., Aulenbach S., Kaufman C., McKeown R., Bachelet D., Schimel D.S., Participants V. (2004). VEMAP phase 2 bioclimatic database. I. Gridded historical (20th century) climate for modeling ecosystem dynamics across the conterminous USA. Climate Research 27:151-170.

Kolberg R.L., Westfall D.G., Peterson G.A. (1999). Influence of cropping intensity and nitrogen fertilizer rates on in situ nitrogen mineralization. Soil Science Society of America Journal 63:129-134.

Lal R. (2009). Soil quality impacts of residue removal for bioethanol production. Soil & Tillage Research 102:233-241. DOI: 10.1016/j.still.2008.07.003.

Lynd L.R., Larson E., Greene N., Laser M., Sheehan J., Dale B.E., McLaughlin S., Wang M. (2009). The role of biomass in America's energy future: framing the analysis. Biofuels Bioproducts & Biorefining-Biofpr 3:113-123. DOI: 10.1002/bbb.134.

Mann L., Tolbert V., Cushman J. (2002). Potential environmental effects of corn (Zea mays L.) stover removal with emphasis on soil organic matter and erosion. Agriculture Ecosystems & Environment 89:149-166.

Ogle S.M., Breidt F.J., Paustian K. (2005). Agricultural management impacts on soil organic carbon storage under moist and dry climatic conditions of temperate and tropical regions. Biogeochemistry 72:87-121. DOI: 10.1007/s10533-004-0360-2.

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

Parton W.J., Ojima D.S., Cole C.V., Schimel D.S. (1994). A general model for soil organic matter dynamics - sensitivity to litter chemistry, texture and management, in: R. B. Bryant and R. W. Arnold (Eds.), Quantitative Modeling of Soil Forming Processes. pp. 147-167.

Parton W.J., Hartman M., Ojima D., Schimel D. (1998). DAYCENT and its land surface submodel: description and testing. Global and Planetary Change 19:35-48.

Paustian K., Parton W.J., Persson J. (1992). Modeling soil organic-matter in organic-amended and nitrogen-fertilized long-term plots. Soil Science Society of America Journal 56:476-488.

Perlack R.D., Wright L.L., A.Turhollow, Graham R.L., Stocks B., Erbach D.C. (2005). Biomass as a feedstock for a bioenergy and bioproducts industry: The technical feasibility of a billion ton annual supply., USDOE and USDA.

Peterson G.A., Westfall D.G. (2004). Managing precipitation use in sustainable dryland agroecosystems. Annals of Applied Biology 144:127-138.

Peterson G.A., Westfall D.G., Cole C.V. (1993). Agroecosystem approach to soil and crop management research. Soil Science Society of America Journal 57:1354-1360.

Powlson D., P.Smith, J.U.Smith. (1996). (Eds.) Evaluation of soil organic matter models using existing long-term datasets., Springer-Verlag, Berlin.

.

Sarath G., Mitchell R.B., Sattler S.E., Funnell D., Pedersen J.F., Graybosch R.A., Vogel K.P. (2008). Opportunities and roadblocks in utilizing forages and small grains for liquid fuels. Journal of Industrial Microbiology & Biotechnology 35:343-354. DOI: 10.1007/s10295-007-0296-3.

Schaap M.G., Leij F.J., van Genuchten M.T. (2001). ROSETTA: a computer program for estimating soil hydraulic parameters with hierarchical pedotransfer functions. Journal of Hydrology 251:163-176.

Sherrod L.A., Peterson G.A., Westfall D.G., Ahuja L.R. (2003). Cropping intensity enhances soil organic carbon and nitrogen in a no-till agroecosystem. Soil Science Society of America Journal 67:1533-1543.

Stehfest E., Heistermann M., Priess J.A., Ojima D.S., Alcamo J. (2007). Simulation of global crop production with the ecosystem model DayCent. Ecological Modelling 209:203-219. DOI: 10.1016/j.ecolmodel.2007.06.028.

Thornton P.E., Running S.W., White M.A. (1997). Generating surfaces of daily meteorological variables over large regions of complex terrain. Journal of Hydrology 190:214-251.

Unger P.W. (1978). Straw-mulch rate effect on soil-water storage and sorghum yield. Soil Science Society of America Journal 42:486-491.

Viaud V., Angers D.A., Walter C. (2010). Toward Landscape-Scale Modeling of Soil Organic Matter Dynamics in Agroecosystems. Soil Science Society of America Journal 74:1847-1860. DOI: 10.2136/sssaj2009.0412.

Wilhelm W.W., Johnson J.M.E., Karlen D.L., Lightle D.T. (2007). Corn stover to sustain soil organic carbon further constrains Biomass supply. Agronomy Journal 99:1665-1667. DOI: 10.2134/agronj2OO7.0150.

Wilhelm W.W., Johnson J.M.F., Hatfield J.L., Voorhees W.B., Linden D.R. (2004). Crop and soil productivity response to corn residue removal: A literature review. Agronomy Journal 96:1-17.

Wilson G.V., Dabney S.M., McGregor K.C., Barkoll B.D. (2004). Tillage and residue effects on runoff and erosion dynamics. Transactions of the Asae 47:119-128.

TABLES AND FIGURES

Table 9. Selected climatic properties for the long-term dryland cropping systems research sites in Sterling and Stratton, Colorado. Adapted from Sherrod et al. (2005).

Site	Elevation (m)	Mean annual temperature (C°)	MAP (mm) *	Days above 32 °C	OPE (mm)†	Deficit water (mm)‡
Sterling	1341	9.3	425	42	1600	-1160
Stratton	1335	10.8	405	54	1725	-1310

*MAP = mean annual precipitation from 1985-2007

†OPE = open pan evaporation during the growing season

‡Deficit water = precipitation – open pan evaporation

Table 10. Soil physical and hydraulic properties for the top two soil horizons by landscape position for the long-term dryland cropping systems research sites in Sterling and Stratton, CO. Adapted from Ascough et al., 2010.

Soil horizon name	Horizon depth (cm)	Bulk density) (g cm^{-3})	Sand (%)	Clay (%)	Total carbon (kg ha^{-1})	Total nitrogen (kg ha $^{-1}$)	Field capacity water content (33 kPa) (%)	Wilting point water content (1500 kPa)	Ksat * (cm h^{-1})
Sterling summit (Weld loam - fine, mixed, mesic Aridic Argiustoll)									
Ap1	0-8	1.37	45	21	11,674	900	.28	.11	1.78
Bt1	8-20	1.35	33	31	14.592	1670	.33	.15	0.85
Sterling sideslope (Satanta loam – fine-loamy, mixed, mesic Pachic Argiustoll)									
Ap1	0-11	1.47	54	21	15,642	1321	.26	.11	1.60
Ap2	11-20	1.34	44	26	11,144	1060	.29	.13	1.09
Sterling toeslope (Albinas loam– fine-loamy, mixed, mesic Pachic Argiustoll)									
Ap1	0-7	1.31	42	18	15,009	1223	.27	.10	2.55
Ap2	7-18	1.49	47	20	12,210	1139	.27	.11	1.94
Stratton summit (Norka clay loam – fine-silty, mixed, mesic Aridic Argiustoll)									
Ap	0-13	1.35	25	34	20,553	1874	.36	.16	.80
Bt	13-39	1.26	20	36	33,987	3517	.38	.17	.78
Stratton sideslope (Richfield loam - fine, montmorillonitic, mesic Aridic Argiustoll)									
Ap1	0-10	1.36	41	20	14,994	1323	.28	.11	2.07
Ap2	10-18	1.33	35	28	8928	918	.32	.14	1.04
Stratton toeslope (Kuma loam – fine-silty, mixed, mesic Pachic Argiustoll)									
Ap	0-15	1.28	25	26	41,040	3866	.34	.13	1.45
Ab1	15-30	1.26	23	25	44,838	4738	.33	.13	1.64

*Ksat, saturated hydraulic conductivity (estimated with Rosetta)

Table 11. Mean and standard deviation of average stover and grain production for corn and wheat by soil position for the long-term dryland cropping systems research sites in Sterling and Stratton, CO.

Site and Soil	Corn stover (kg ha^{-1})	Corn grain (kg ha^{-1})	Wheat stover (kg ha^{-1})	Wheat grain (kg ha^{-1})
STERLING				
Summit	2480 ± 1580	2140 ± 1310	3260 ± 1370	1690 ± 750
Side	2550 ± 1140	2670 ± 1350	3590 ± 1290	1750 ± 680
Toe	3170 ± 1360	3500 ± 1730	4550 ± 1460	2160 ± 770
STRATTON				
Summit	2490 ± 1830	2460 ± 1750	3850 ± 1490	1890 ± 660
Side	2550 ± 1290	2490 ± 1330	3270 ± 1710	1670 ± 720
Toe	4210 ± 1690	4300 ± 1600	6360 ± 2440	2570 ± 1100

Table 12. Model fit statistics relative error (RE) and normalized objective function (NOF) for grain and stover yield, total profile water content, and soil organic carbon by soil position for the long-term dryland cropping systems research sites in Sterling and Stratton, CO.

Location	*Summit position*		*Sideslope position*		*Toeslope position*		*Site Average*	
	RE (%)	NOF (unitless)	RE (%)	NOF (unitless)	RE (%)	NOF (unitless)	RE (%)	NOF (unitless)
Sterling								
Wheat grain yield	18.5	0.55	-13.5	0.29	-9.2	0.25	-5.4	0.18
Wheat stover yield	10.8	0.53	-23.4	0.52	-26.0	0.45	-12.6	0.37
Corn grain yield	43.1	0.58	-4.3	0.30	-17.3	0.40	2.4	0.30
Corn stover yield	30.6	0.49	-7.0	0.42	-20.8	0.40	-0.7	0.36
Total profile water	0.7	0.02	-14.8	0.05	-27.7	0.05	N/A	N/A
Soil carbon	-1.5	0.02	6.2	0.04	-6.8	0.04	-1.3	0.01
Stratton								
Wheat grain yield	13.9	0.41	22.6	0.46	-3.6	0.39	9.0	0.36
Wheat stover yield	13.3	0.47	14.4	0.47	-20.5	0.45	-0.8	0.34
Corn grain yield	-11.0	0.40	8.0	0.18	-6.4	0.23	-0.39	0.22
Corn stover yield	27.6	0.59	10.3	0.33	-10.0	0.27	0.4	0.27
Total profile water	1.3	0.14	0.6	0.18	-28.9	0.31	N/A	N/A
Soil carbon	13.9	0.26	3.1	0.3	-18.4	0.19	0.0	0.03

Table 13. Average, minimum, and maximum yield difference at Sterling and Stratton by soil position and crop after 56 years of simulated stover harvest of wheat and corn. Stover harvest was 50% of both corn and wheat stover in a wheat-corn-fallow rotation.

	Average yield change (kg ha^{-1} yr^{-1})	Minimum yield change (kg ha^{-1} yr^{-1})	Maximum yield change (kg ha^{-1} yr^{-1})
		--STERLING--	
Wheat			
Summit	-387	-149	-716
Side	-332	-84	-1010
Toe	+13	-432	+507
Corn			
Summit	-510	-184	-1090
Side	-360	-10	-788
Toe	-177	+185	-1013
		--STRATTON--	
Wheat			
Summit	-544	-129	-1150
Side	-369	-51	-954

Toe	-572	-30	-1382
Corn			
Summit	-740	-207	-1406
Side	-547	-20	-1292
Toe	-800	-66	-1615

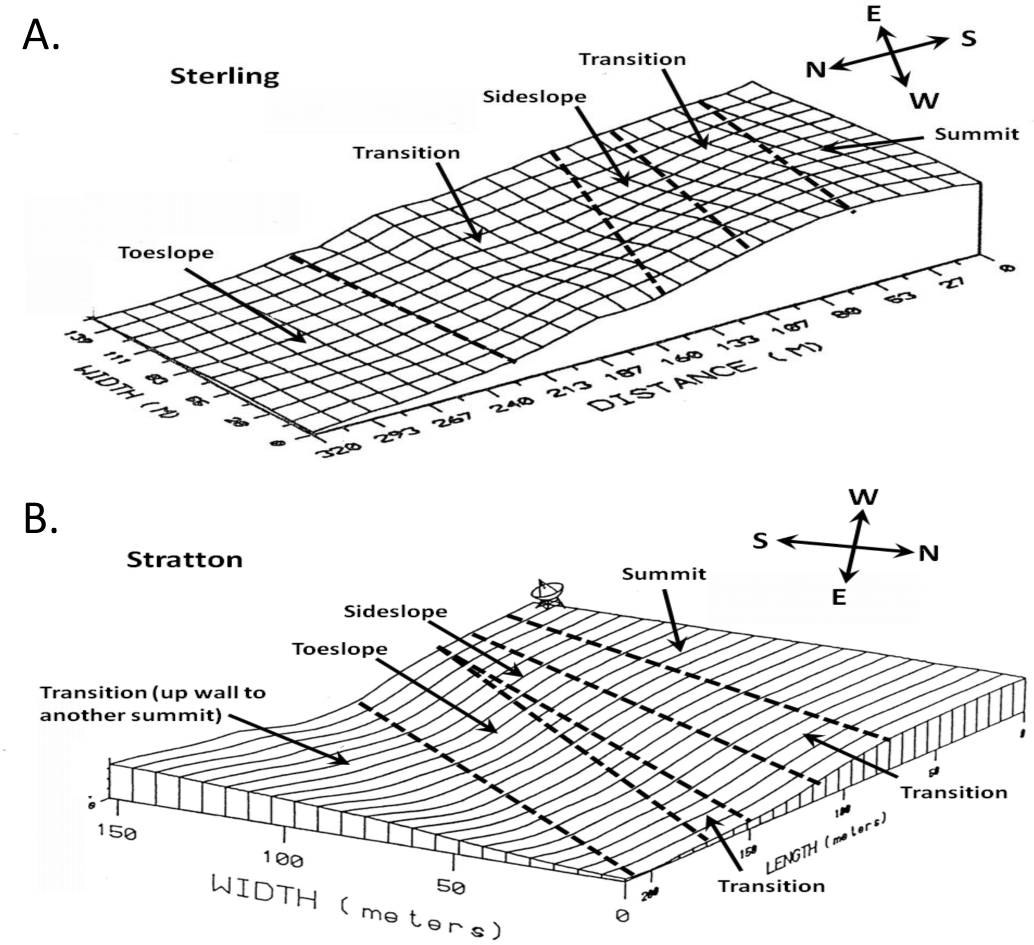

Figure 13. Catenary sequence of summit, sideslope, and toeslope landscape positions for (A) Sterling, and (B) Stratton. Adapted from Ashcough et. al, 2010.

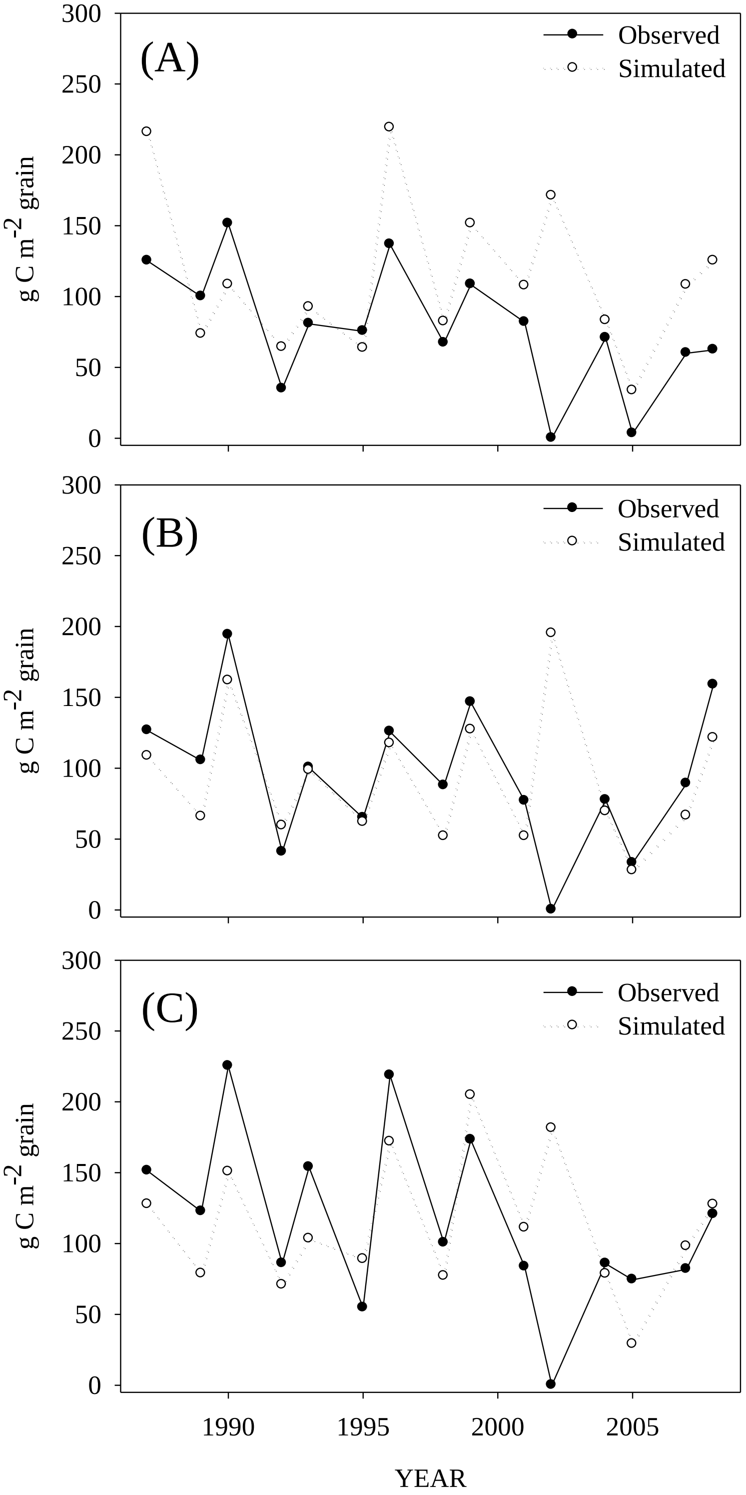

Figure 14. Observed and simulated carbon in harvested grain in the W-C-F rotation at Sterling (A) summit, (B) sideslope, and (C) toeslope.

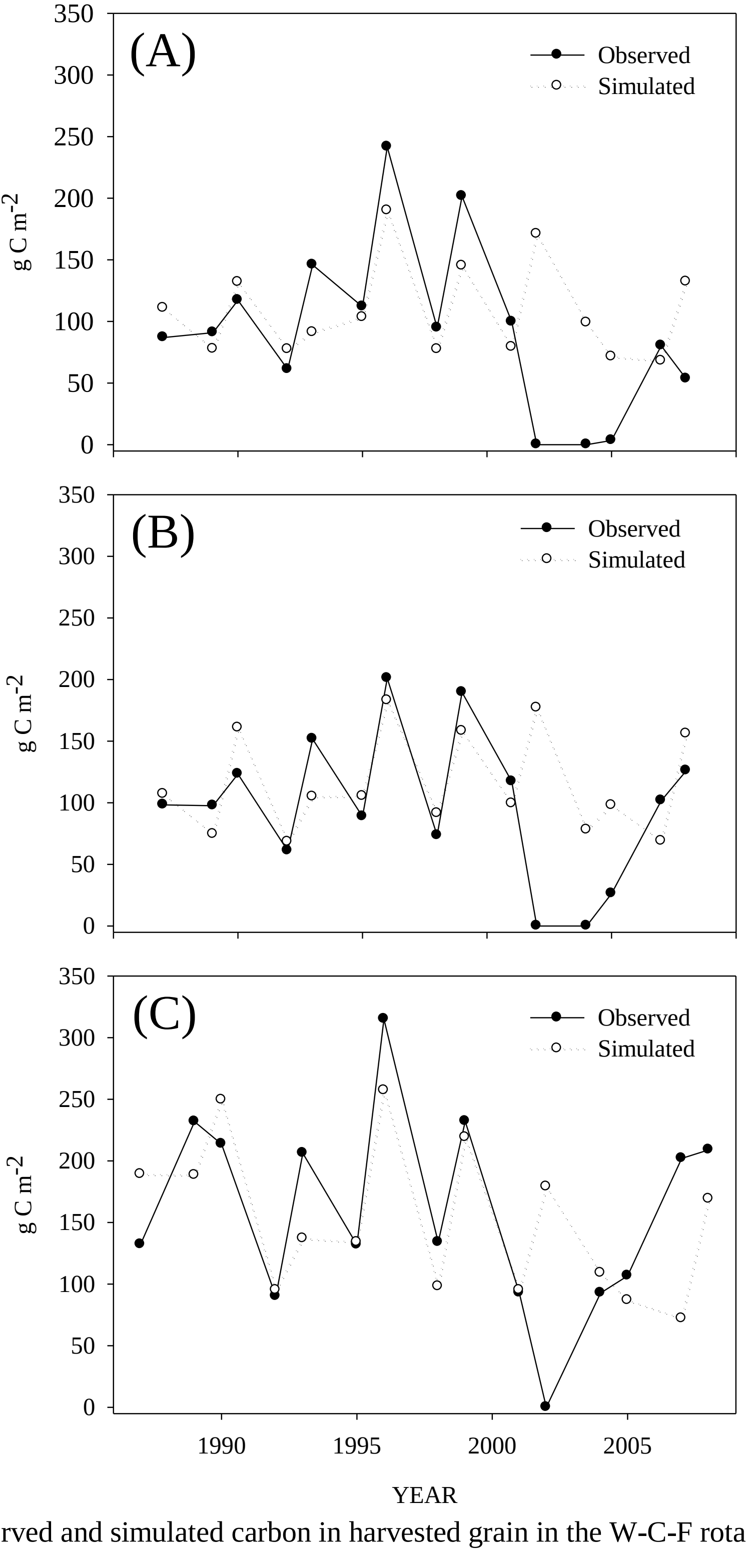

Figure 15. Observed and simulated carbon in harvested grain in the W-C-F rotation at Stratton (A) summit, (B) sideslope, and (C) toeslope.

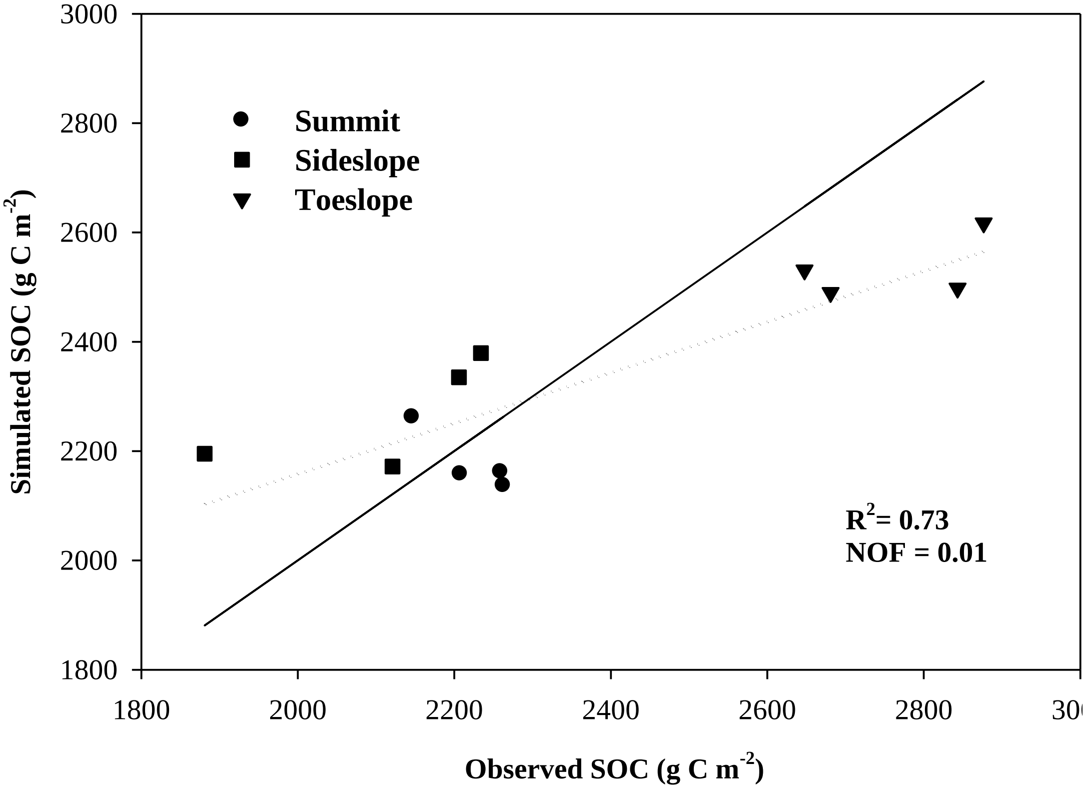

Figure 16. Observed versus simulated SOC in the WCF cropping system for all soil positions at Sterling, Colorado. Solid line indicates 1:1 line.

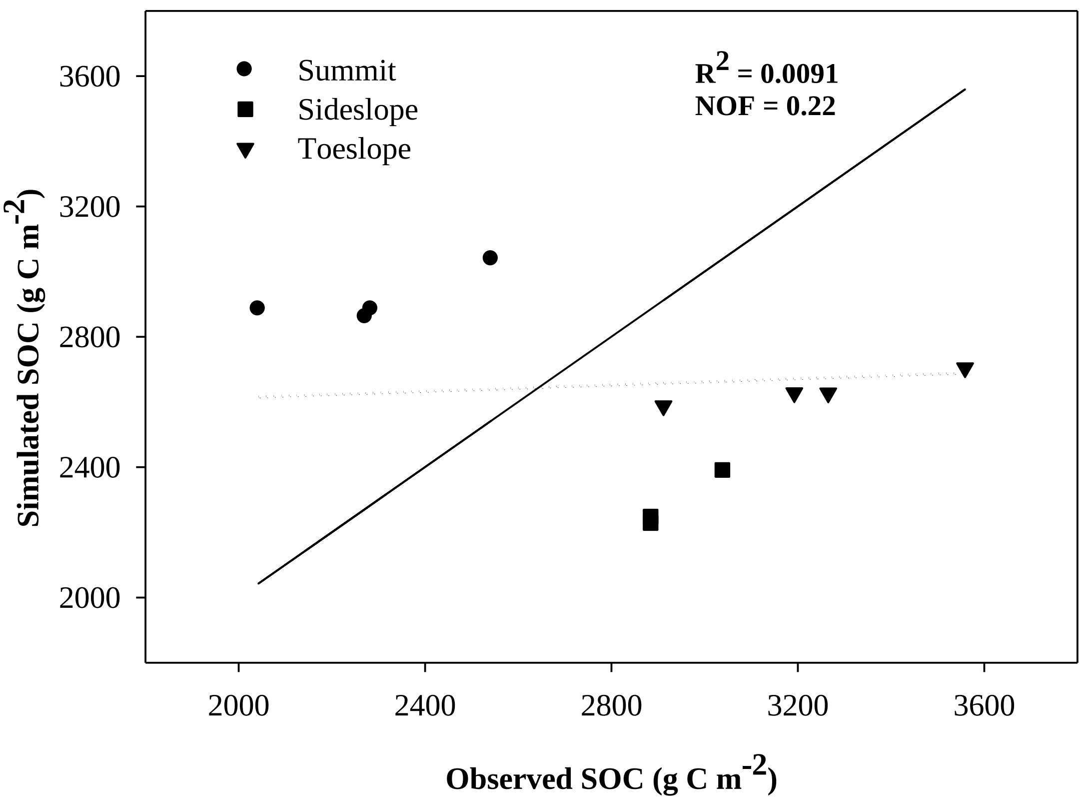

Figure 17. Observed versus simulated SOC in the WCF cropping system for all soil positions at Stratton, Colorado. Solid line indicates 1:1 line.

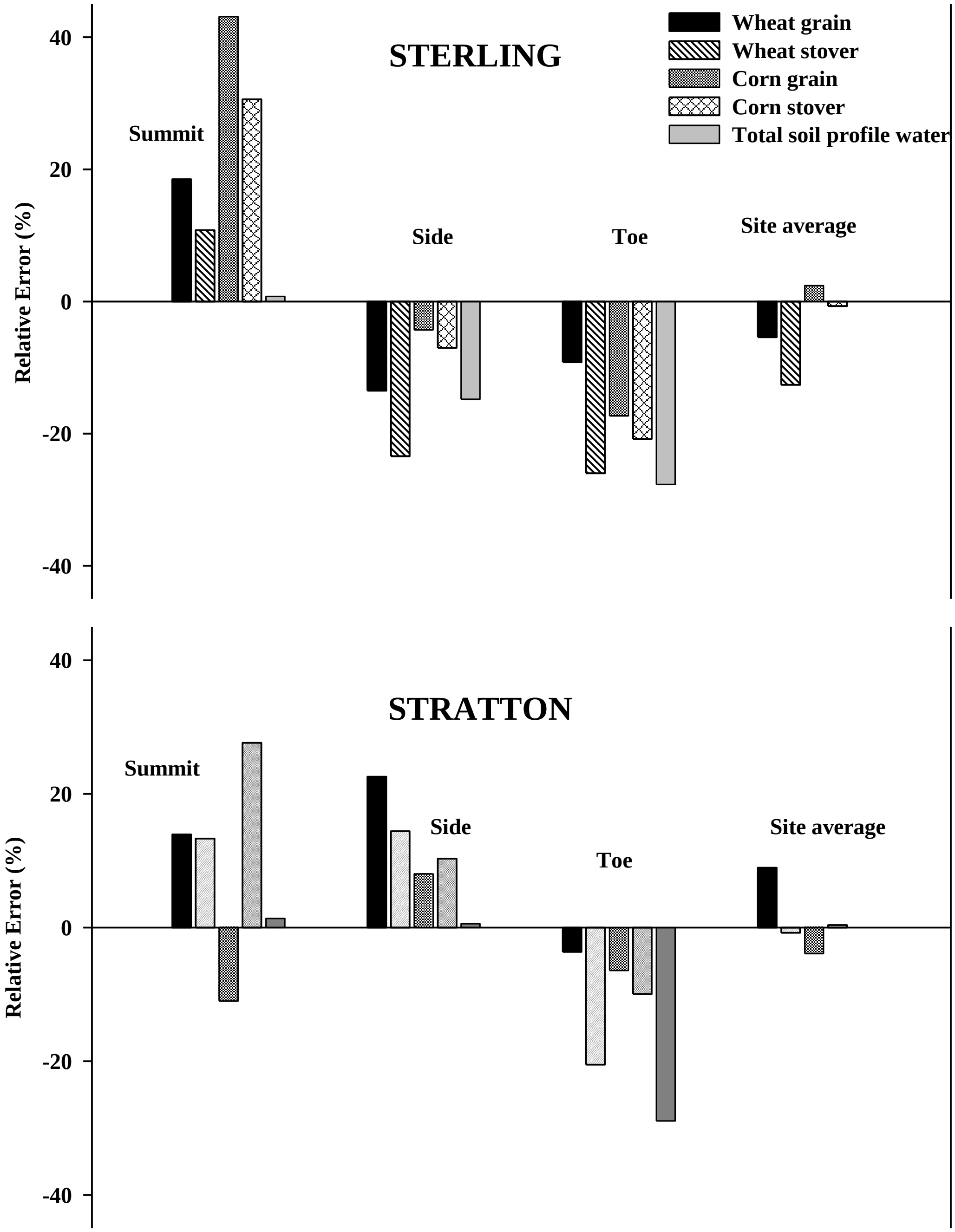

Figure 18. Relative error for simulated grain, stover, and total soil profile water by soil position at Sterling (upper) and Stratton (lower).

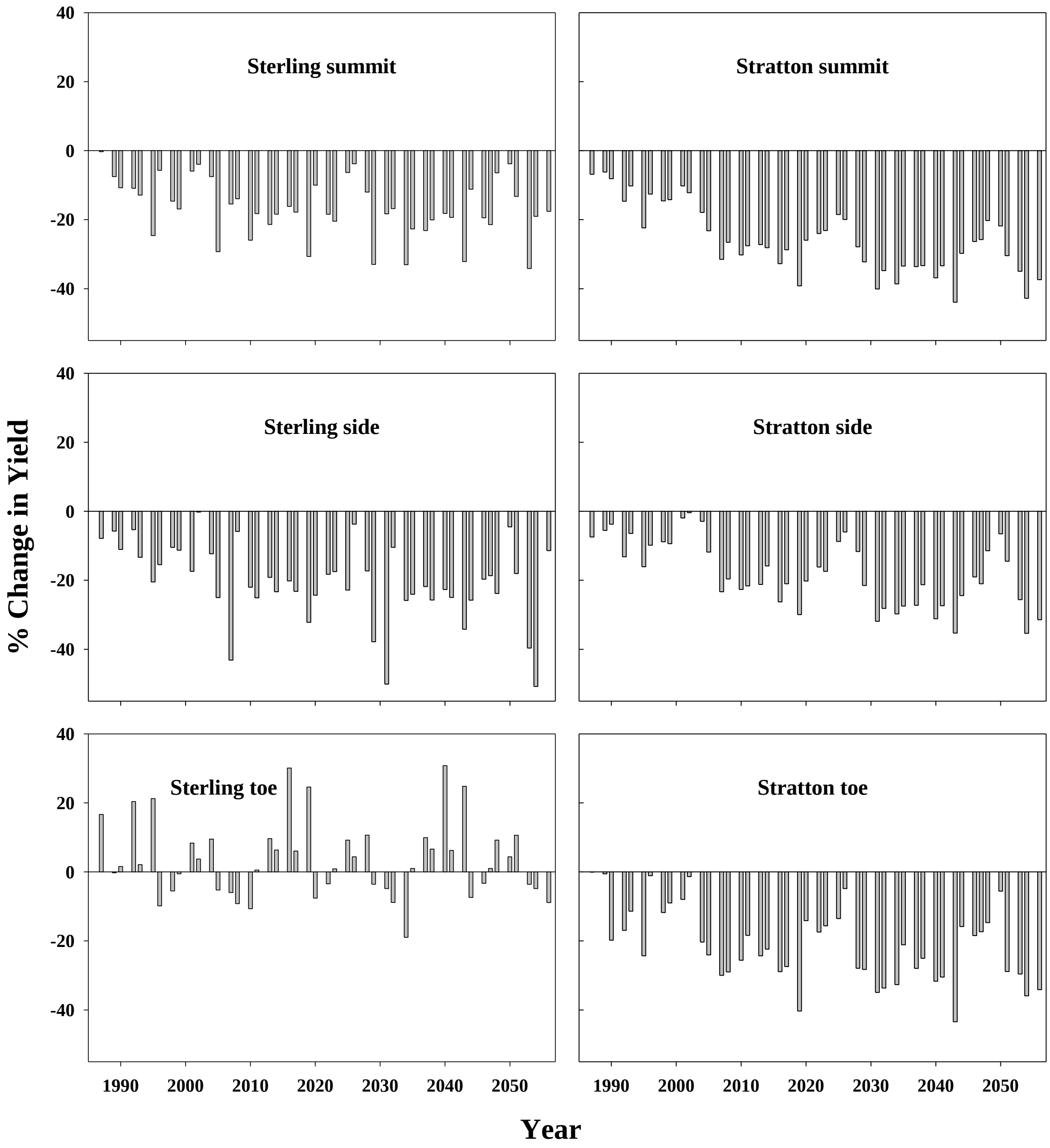

Figure 19. Percent change in average yield of corn and wheat in a wheat-corn-fallow rotation with harvest of 50% of wheat and corn stover for the summit, sideslope, and toeslope soil positions at Sterling and Stratton, Colorado.

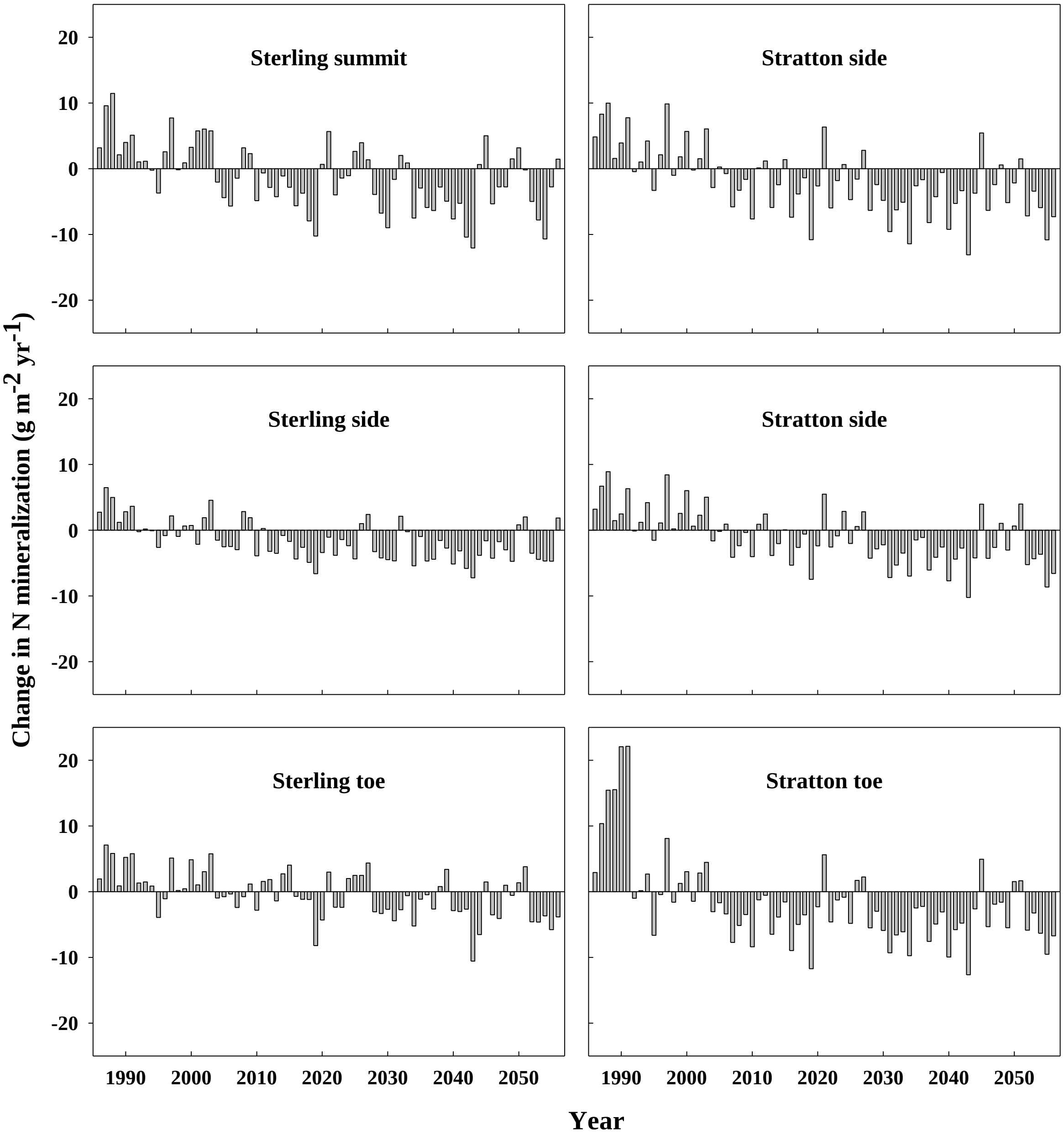

Figure 20. Changes in nitrogen mineralization with harvest of 50% of wheat and corn stover at the summit, sideslope, and toeslope soil positions at Sterling and Stratton, Colorado.

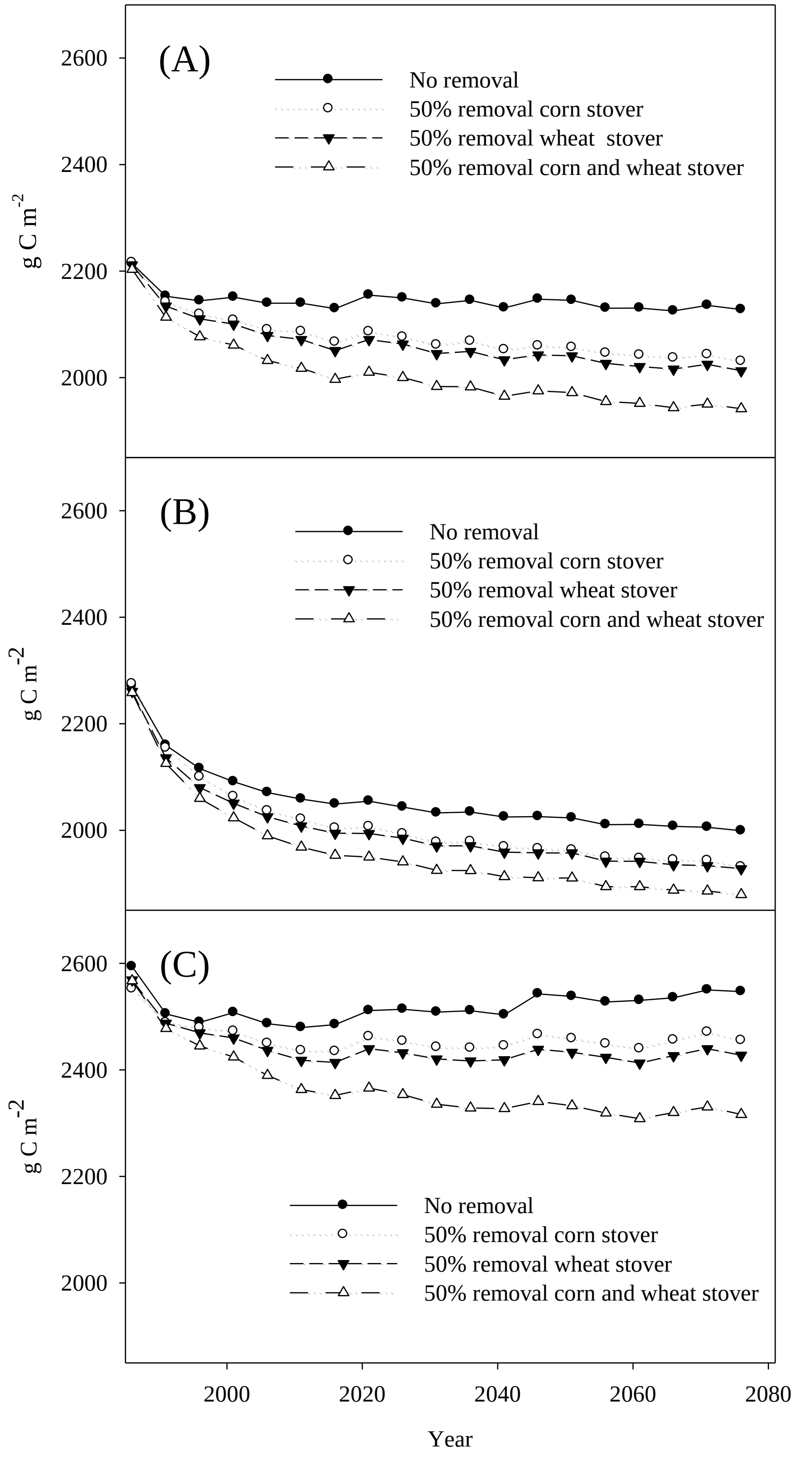

Figure 21. 5 year moving average of simulated changes in soil carbon with stover harvest at Sterling summit (A), sideslope (B), and toeslope (C).

APPENDIX A-1

Column definitions for Soils.in files

Column 1 – Minimum depth of soil layer (cm)

Column 2 – Maximum depth of soil layer (cm)

Column 3 – Bulk density of soil layer (g/cm3)

Column 4 – Field capacity of soil layer, volumetric

Column 5 – Wilting point of soil layer, volumetric

Column 6 – Evaporation coefficient for soil layer (volumetric)

Column 7 – Percentage of Roots in Soil layer, 0.0 – 1.0

Column 8 – Fraction of sand in soil layer, 0.0 -1.0

Column 9 – Fraction of clay in soil layer, 0.0 – 1.0

Column 10 – Organic matter in soil layer, fraction 0.0 – 1.0

Column 11 – Minimum volumetric soil water content below wilting point for soil layer

Column 12 – Saturated hydraulic conductivity of soil layer in centimeters per second

Column 13 – pH of soil layer

Table A-1. Soil file for Sterling summit soil.

0	2	1.37	0.21	0.10	0.80	0.02	0.45	0.21	0.01	0.03	0.0002	7.1
2	5	1.37	0.21	0.10	0.20	0.04	0.45	0.21	0.01	0.02	0.0002	7.1
5	10	1.37	0.21	0.10	0.00	0.25	0.45	0.21	0.01	0.01	0.0002	7.1
10	20	1.35	0.28	0.13	0.00	0.30	0.37	0.28	0.01	0.00	0.0002	7.1
20	30	1.23	0.34	0.18	0.00	0.10	0.25	0.38	0.01	0.00	0.0004	7.4
30	45	1.21	0.30	0.15	0.00	0.05	0.27	0.27	0.01	0.00	0.0004	8.3
45	60	1.31	0.31	0.13	0.00	0.04	0.31	0.22	0.01	0.00	0.0002	8.7
60	75	1.31	0.31	0.13	0.00	0.03	0.43	0.21	0.01	0.00	0.0002	8.7
75	90	1.34	0.30	0.10	0.00	0.02	0.37	0.23	0.01	0.00	0.0002	8.5
90	105	1.35	0.25	0.10	0.00	0.02	0.37	0.23	0.01	0.00	0.0002	8.5
105	120	1.35	0.25	0.10	0.00	0.00	0.31	0.25	0.01	0.00	0.0002	8.3
120	150	1.35	0.25	0.10	0.00	0.00	0.31	0.25	0.01	0.00	0.0002	8.3

Table A-2. Soil file for Sterling sideslope soil.

0	2	1.47	0.22	0.09	0.80	0.02	0.54	0.21	0.01	0.03	0.0002	7.7
2	5	1.47	0.22	0.09	0.80	0.04	0.54	0.21	0.01	0.02	0.0002	7.7
5	10	1.47	0.22	0.09	0.20	0.25	0.54	0.21	0.01	0.01	0.0002	7.7
10	20	1.34	0.25	0.12	0.00	0.30	0.44	0.26	0.01	0.00	0.0002	7.8
20	30	1.18	0.28	0.15	0.00	0.10	0.32	0.31	0.01	0.00	0.0004	8.0
30	45	1.18	0.28	0.15	0.00	0.05	0.32	0.31	0.01	0.00	0.0004	8.0
45	60	1.24	0.31	0.16	0.00	0.04	0.27	0.28	0.01	0.00	0.0003	8.3
60	75	1.27	0.20	0.12	0.00	0.03	0.36	0.22	0.01	0.00	0.0003	8.3
75	90	1.58	0.13	0.06	0.00	0.02	0.67	0.09	0.01	0.00	0.0004	8.5
90	105	1.47	0.10	0.04	0.00	0.02	0.79	0.03	0.01	0.00	0.0017	8.7
105	120	1.47	0.10	0.04	0.00	0.00	0.79	0.03	0.00	0.00	0.0017	8.7
120	150	1.47	0.10	0.04	0.00	0.00	0.79	0.03	0.00	0.00	0.0017	8.7

Table A-3. Soil file for Sterling toeslope.

0	2	1.31	0.22	0.10	0.80	0.02	0.42	0.18	0.01	0.03	0.0021	7.60
2	5	1.31	0.22	0.10	0.80	0.22	0.42	0.18	0.01	0.02	0.0021	7.60
5	10	1.31	0.22	0.10	0.20	0.22	0.42	0.18	0.01	0.01	0.0021	7.60
10	20	1.49	0.20	0.09	0.00	0.22	0.47	0.20	0.01	0.00	0.0021	7.00
20	30	1.32	0.24	0.11	0.00	0.18	0.46	0.24	0.01	0.00	0.0002	7.00
30	45	1.32	0.24	0.11	0.00	0.12	0.46	0.24	0.01	0.00	0.0002	7.00
45	60	1.33	0.27	0.12	0.00	0.04	0.40	0.25	0.01	0.00	0.0002	8.00
60	75	1.30	0.30	0.13	0.00	0.03	0.30	0.27	0.01	0.00	0.0002	8.30
75	90	1.54	0.19	0.11	0.00	0.02	0.38	0.20	0.01	0.00	0.0001	8.50
90	105	1.54	0.19	0.11	0.00	0.01	0.38	0.20	0.01	0.00	0.0001	8.50
105	120	1.50	0.17	0.07	0.00	0.00	0.59	0.15	0.00	0.00	0.0003	8.70
120	150	1.50	0.17	0.07	0.00	0.00	0.59	0.15	0.00	0.00	0.0003	8.70

Table A-4. Soil file for Stratton summit

0	2	1.34	0.27	0.17	0.80	0.02	0.25	0.34	0.01	0.03	0.0002	7.50
2	5	1.34	0.27	0.17	0.20	0.04	0.25	0.34	0.01	0.02	0.0002	7.50
5	10	1.34	0.27	0.17	0.00	0.25	0.25	0.34	0.01	0.01	0.0002	7.50
10	20	1.26	0.31	0.22	0.00	0.30	0.20	0.36	0.01	0.00	0.0003	7.80
20	30	1.26	0.31	0.22	0.00	0.10	0.20	0.36	0.01	0.00	0.0003	7.80
30	45	1.26	0.31	0.22	0.00	0.05	0.20	0.36	0.01	0.00	0.0003	7.80
45	60	1.28	0.23	0.12	0.00	0.04	0.27	0.21	0.01	0.00	0.0003	8.40
60	75	1.35	0.17	0.10	0.00	0.03	0.35	0.18	0.01	0.00	0.0002	8.60
75	90	1.35	0.17	0.10	0.00	0.02	0.35	0.18	0.01	0.00	0.0002	8.60
90	105	1.35	0.17	0.10	0.00	0.02	0.35	0.18	0.01	0.00	0.0002	8.60
105	120	1.33	0.16	0.09	0.00	0.00	0.34	0.14	0.00	0.00	0.0003	8.60
120	150	1.33	0.16	0.09	0.00	0.00	0.34	0.14	0.00	0.00	0.0003	8.60

Table A-5. Soil file for Stratton sideslope.

0	2	1.36	0.26	0.10	0.80	0.02	0.41	0.20	0.01	0.03	0.0009	8.00
2	5	1.36	0.26	0.10	0.80	0.04	0.41	0.20	0.01	0.02	0.0009	8.00
5	10	1.36	0.26	0.10	0.20	0.25	0.41	0.20	0.01	0.01	0.0009	8.00
10	20	1.33	0.28	0.14	0.00	0.30	0.35	0.28	0.01	0.00	0.0005	8.10
20	30	1.37	0.26	0.17	0.00	0.10	0.29	0.32	0.01	0.00	0.0001	8.00
30	45	1.34	0.29	0.18	0.00	0.05	0.21	0.35	0.01	0.00	0.0002	8.10
45	60	1.32	0.26	0.13	0.00	0.04	0.33	0.26	0.01	0.00	0.0002	8.30
60	75	1.32	0.26	0.13	0.00	0.03	0.33	0.26	0.01	0.00	0.0002	8.30
75	90	1.51	0.19	0.08	0.00	0.02	0.43	0.18	0.01	0.00	0.0001	8.40
90	105	1.51	0.19	0.08	0.00	0.02	0.43	0.18	0.01	0.00	0.0001	8.40
105	120	1.51	0.19	0.08	0.00	0.00	0.43	0.18	0.00	0.00	0.0001	8.40
120	150	1.64	0.17	0.09	0.00	0.00	0.72	0.20	0.00	0.00	0.0002	8.40

Table A-6. Soil file for Stratton toeslope.

0	2	1.28	0.28	0.14	0.80	0.02	0.25	0.26	0.02	0.03	0.0002	7.90
2	5	1.28	0.28	0.14	0.80	0.04	0.25	0.26	0.02	0.02	0.0002	7.90
5	10	1.28	0.28	0.14	0.20	0.25	0.25	0.26	0.02	0.01	0.0002	7.90
10	20	1.26	0.30	0.13	0.00	0.30	0.23	0.25	0.01	0.00	0.0003	8.00
20	30	1.26	0.30	0.13	0.00	0.10	0.23	0.25	0.01	0.00	0.0003	8.00
30	45	1.18	0.30	0.10	0.00	0.05	0.33	0.22	0.01	0.00	0.0005	7.80
45	60	1.31	0.24	0.09	0.00	0.04	0.42	0.19	0.01	0.00	0.0003	7.60
60	75	1.35	0.23	0.11	0.00	0.03	0.34	0.23	0.01	0.00	0.0002	7.70
75	90	1.35	0.23	0.11	0.00	0.02	0.34	0.23	0.01	0.00	0.0002	7.70
90	105	1.35	0.30	0.17	0.00	0.02	0.24	0.36	0.01	0.00	0.0001	7.40
105	120	1.35	0.30	0.17	0.00	0.00	0.24	0.36	0.00	0.00	0.0001	7.40
120	150	1.37	0.29	0.13	0.00	0.00	0.28	0.28	0.00	0.00	0.0001	7.40

Table A-7. Parameters used for high harvest index (modern) corn (C6).

Parameter	*Value*	*Parameter*	*Value*	*Parameter*	*Value*
'PRDX(1)'	1.25	'PRBMN(2,1)'	390	'RTDTMP'	2
'PPDF(1)'	30	'PRBMN(3,1)'	340	'CRPRTF(1)'	0
'PPDF(2)'	45	'PRBMN(1,2)'	0	'CRPRTF(2)'	0
'PPDF(3)'	1	'PRBMN(2,2)'	0	'CRPRTF(3)'	0
'PPDF(4)'	2.5	'PRBMN(3,2)'	0	'MRTFRAC'	0.05
'BIOFLG'	0	'PRBMX(1,1)'	60	'SNFXMX(1)'	0
'BIOK5'	1800	'PRBMX(2,1)'	420	'DEL13C'	-15
'PLTMRF'	0.5	'PRBMX(3,1)'	420	'CO2IPR(1)'	1
'FULCAN'	150	'PRBMX(1,2)'	0	'CO2ITR(1)'	0.77
'FRTCINDX'	2	'PRBMX(2,2)'	0	'CO2ICE(1,1,1)'	1
'FRTC(1)'	0.5	'PRBMX(3,2)'	0	'CO2ICE(1,1,2)'	1
'FRTC(2)'	0.1	'FLIGNI(1,1)'	0.12	'CO2ICE(1,1,3)'	1
'FRTC(3)'	60	'FLIGNI(2,1)'	0	'CO2ICE(1,2,1)'	1
'FRTC(4)'	0.2	'FLIGNI(1,2)'	0.06	'CO2ICE(1,2,2)'	1
'FRTC(5)'	0.2	'FLIGNI(2,2)'	0	'CO2ICE(1,2,3)'	1
'CFRTCN(1)'	0.3	'FLIGNI(1,3)'	0.06	'CO2IRS(1)'	1
'CFRTCN(2)'	0.25	'FLIGNI(2,3)'	0	'KMRSP(1)'	1
'CFRTCW(1)'	0.5	'HIMAX'	0.55	'CKMRSPMX(1)'	0.01525
'CFRTCW(2)'	0.1	'HIWSF'	0.5	'CKMRSPMX(2)'	0.38
'BIOMAX'	700	'HIMON(1)'	1	'CKMRSPMX(3)'	0.26
'PRAMN(1,1)'	12	'HIMON(2)'	0	'CGRESP(1)'	0.25
'PRAMN(2,1)'	150	'EFRGRN(1)'	0.75	'CGRESP(2)'	0.25
'PRAMN(3,1)'	190	'EFRGRN(2)'	0.6	'CGRESP(3)'	0.29
'PRAMN(1,2)'	62.5	'EFRGRN(3)'	0.6	'NO3PREF(1)'	0.25
'PRAMN(2,2)'	150	'VLOSSP'	0.04	'CLAYPG'	4

'PRAMN(3,2)'	150	'FSDETH(1)'	0	'CMIX'	0.5
'PRAMX(1,1)'	20	'FSDETH(2)'	0	'TMPGERM'	15
'PRAMX(2,1)'	230	'FSDETH(3)'	0	'DDBASE'	1250
'PRAMX(3,1)'	230	'FSDETH(4)'	500	'TMPKILL'	-3.5
'PRAMX(1,2)'	125	'FALLRT'	0.1	'BASETEMP'	10
'PRAMX(2,2)'	230	'RDRJ'	0.05	'BASETEMP(2)'	30
'PRAMX(3,2)'	230	'RDRM'	0.1	'MNDDHRV'	400
'PRBMN(1,1)'	45	'RDSRFC'	0.14	'MXDDHRV'	600
				'CMXTURN'	0.12

Table A-8. Parameters used for high harvest index (modern) wheat (W3).

Parameter	*Value*	*Parameter*	*Value*	*Parameter*	*Value*
'PRDX(1)'	0.9	'PRBMN(2,1)'	390	'RTDTMP'	2
'PPDF(1)'	18	'PRBMN(3,1)'	340	'CRPRTF(1)'	0
'PPDF(2)'	35	'PRBMN(1,2)'	0	'CRPRTF(2)'	0
'PPDF(3)'	0.7	'PRBMN(2,2)'	0	'CRPRTF(3)'	0
'PPDF(4)'	5	'PRBMN(3,2)'	0	'MRTFRAC'	0.05
'BIOFLG'	0	'PRBMX(1,1)'	60	'SNFXMX(1)'	0
'BIOK5'	1800	'PRBMX(2,1)'	420	'DEL13C'	-27
'PLTMRF'	0.4	'PRBMX(3,1)'	420	'CO2IPR(1)'	1.3
'FULCAN'	150	'PRBMX(1,2)'	0	'CO2ITR(1)'	0.77
'FRTCINDX'	2	'PRBMX(2,2)'	0	'CO2ICE(1,1,1)'	1
'FRTC(1)'	0.6	'PRBMX(3,2)'	0	'CO2ICE(1,1,2)'	1
'FRTC(2)'	0.1	'FLIGNI(1,1)'	0.15	'CO2ICE(1,1,3)'	1
'FRTC(3)'	90	'FLIGNI(2,1)'	0	'CO2ICE(1,2,1)'	1.3
'FRTC(4)'	0.2	'FLIGNI(1,2)'	0.06	'CO2ICE(1,2,2)'	1
'FRTC(5)'	0.1	'FLIGNI(2,2)'	0	'CO2ICE(1,2,3)'	1
'CFRTCN(1)'	0.4	'FLIGNI(1,3)'	0.06	'CO2IRS(1)'	1
'CFRTCN(2)'	0.25	'FLIGNI(2,3)'	0	'KMRSP(1)'	1
'CFRTCW(1)'	0.6	'HIMAX'	0.4	'CKMRSPMX(1)'	0.01525
'CFRTCW(2)'	0.1	'HIWSF'	0.5	'CKMRSPMX(2)'	0.38
'BIOMAX'	400	'HIMON(1)'	1	'CKMRSPMX(3)'	0.26
'PRAMN(1,1)'	20	'HIMON(2)'	0	'CGRESP(1)'	0.25
'PRAMN(2,1)'	100	'EFRGRN(1)'	0.75	'CGRESP(2)'	0.25
'PRAMN(3,1)'	100	'EFRGRN(2)'	0.6	'CGRESP(3)'	0.29
'PRAMN(1,2)'	50	'EFRGRN(3)'	0.6	'NO3PREF(1)'	0.25
'PRAMN(2,2)'	160	'VLOSSP'	0.04	'CLAYPG'	5

'PRAMN(3,2)'	200	'FSDETH(1)'	0	'CMIX'	0.5
'PRAMX(1,1)'	40	'FSDETH(2)'	0	'TMPGERM'	10
'PRAMX(2,1)'	200	'FSDETH(3)'	0	'DDBASE'	1000
'PRAMX(3,1)'	230	'FSDETH(4)'	200	'TMPKILL'	-5
'PRAMX(1,2)'	120	'FALLRT'	0.12	'BASETEMP'	5
'PRAMX(2,2)'	260	'RDRJ'	0.05	'BASETEMP(2)'	30
'PRAMX(3,2)'	270	'RDRM'	0.1	'MNDDHRV'	100
'PRBMN(1,1)'	45	'RDSRFC'	0.14	'MXDDHRV'	300
				'CMXTURN'	0.12

Table A-9. Parameters used for low harvest index wheat (W2).

Parameter	*Value*	*Parameter*	*Value*	*Parameter*	*Value*
'PRDX(1)'	0.75	'PRBMN(2,1)'	390	'RTDTMP'	2
'PPDF(1)'	18	'PRBMN(3,1)'	340	'CRPRTF(1)'	0
'PPDF(2)'	35	'PRBMN(1,2)'	0	'CRPRTF(2)'	0
'PPDF(3)'	0.7	'PRBMN(2,2)'	0	'CRPRTF(3)'	0
'PPDF(4)'	5	'PRBMN(3,2)'	0	'MRTFRAC'	0.05
'BIOFLG'	0	'PRBMX(1,1)'	60	'SNFXMX(1)'	0
'BIOK5'	1800	'PRBMX(2,1)'	420	'DEL13C'	-27
'PLTMRF'	0.4	'PRBMX(3,1)'	420	'CO2IPR(1)'	1.3
'FULCAN'	150	'PRBMX(1,2)'	0	'CO2ITR(1)'	0.77
'FRTCINDX'	6	'PRBMX(2,2)'	0	'CO2ICE(1,1,1)'	1
'FRTC(1)'	0.6	'PRBMX(3,2)'	0	'CO2ICE(1,1,2)'	1
'FRTC(2)'	0.1	'FLIGNI(1,1)'	0.15	'CO2ICE(1,1,3)'	1
'FRTC(3)'	90	'FLIGNI(2,1)'	0	'CO2ICE(1,2,1)'	1.3
'FRTC(4)'	0.2	'FLIGNI(1,2)'	0.06	'CO2ICE(1,2,2)'	1
'FRTC(5)'	0.1	'FLIGNI(2,2)'	0	'CO2ICE(1,2,3)'	1
'CFRTCN(1)'	0.4	'FLIGNI(1,3)'	0.06	'CO2IRS(1)'	1
'CFRTCN(2)'	0.25	'FLIGNI(2,3)'	0	'KMRSP(1)'	1
'CFRTCW(1)'	0.6	'HIMAX'	0.4	'CKMRSPMX(1)'	0.01525
'CFRTCW(2)'	0.1	'HIWSF'	0.42	'CKMRSPMX(2)'	0.38
'BIOMAX'	400	'HIMON(1)'	1	'CKMRSPMX(3)'	0.26
'PRAMN(1,1)'	20	'HIMON(2)'	0	'CGRESP(1)'	0.25
'PRAMN(2,1)'	100	'EFRGRN(1)'	0.65	'CGRESP(2)'	0.25
'PRAMN(3,1)'	100	'EFRGRN(2)'	0.6	'CGRESP(3)'	0.29
'PRAMN(1,2)'	50	'EFRGRN(3)'	0.6	'NO3PREF(1)'	0.25
'PRAMN(2,2)'	160	'VLOSSP'	0.04	'CLAYPG'	4
'PRAMN(3,2)'	200	'FSDETH(1)'	0	'CMIX'	0.5
'PRAMX(1,1)'	40	'FSDETH(2)'	0	'TMPGERM'	10
'PRAMX(2,1)'	200	'FSDETH(3)'	0	'DDBASE'	1000
'PRAMX(3,1)'	230	'FSDETH(4)'	200	'TMPKILL'	-5
'PRAMX(1,2)'	120	'FALLRT'	0.12	'BASETEMP'	5
'PRAMX(2,2)'	260	'RDRJ'	0.05	'BASETEMP(2)'	30
'PRAMX(3,2)'	270	'RDRM'	0.1	'MNDDHRV'	100
'PRBMN(1,1)'	45	'RDSRFC'	0.14	'MXDDHRV'	300

		'CMXTURN'	0.12

Table A-10. Parameters used for G3 grass (50% cool season, 50% warm season).

Parameter	*Value*	*Parameter*	*Value*	*Parameter*	*Value*
'PRDX(1)'	0.35	'PRBMN(2,1)'	390	'RTDTMP'	2
'PPDF(1)'	22	'PRBMN(3,1)'	340	'CRPRTF(1)'	0
'PPDF(2)'	38	'PRBMN(1,2)'	0	'CRPRTF(2)'	0
'PPDF(3)'	0.3	'PRBMN(2,2)'	0	'CRPRTF(3)'	0
'PPDF(4)'	5	'PRBMN(3,2)'	0	'MRTFRAC'	0.05
'BIOFLG'	1	'PRBMX(1,1)'	55	'SNFXMX(1)'	0
'BIOK5'	60	'PRBMX(2,1)'	420	'DEL13C'	-21
'PLTMRF'	1	'PRBMX(3,1)'	420	'CO2IPR(1)'	1.15
'FULCAN'	100	'PRBMX(1,2)'	0	'CO2ITR(1)'	0.77
'FRTCINDX'	1	'PRBMX(2,2)'	0	'CO2ICE(1,1,1)'	1
'FRTC(1)'	0.75	'PRBMX(3,2)'	0	'CO2ICE(1,1,2)'	1
'FRTC(2)'	0.35	'FLIGNI(1,1)'	0.02	'CO2ICE(1,1,3)'	1
'FRTC(3)'	30	'FLIGNI(2,1)'	0.0012	'CO2ICE(1,2,1)'	1.15
'FRTC(4)'	0.2	'FLIGNI(1,2)'	0.26	'CO2ICE(1,2,2)'	1
'FRTC(5)'	0.1	'FLIGNI(2,2)'	-0.0015	'CO2ICE(1,2,3)'	1
'CFRTCN(1)'	0.6	'FLIGNI(1,3)'	0.26	'CO2IRS(1)'	1
'CFRTCN(2)'	0.4	'FLIGNI(2,3)'	-0.0015	'KMRSP(1)'	1
'CFRTCW(1)'	0.75	'HIMAX'	0.02	'CKMRSPMX(1)'	0.01525
'CFRTCW(2)'	0.35	'HIWSF'	0	'CKMRSPMX(2)'	0.38
'BIOMAX'	400	'HIMON(1)'	2	'CKMRSPMX(3)'	0.26
'PRAMN(1,1)'	30	'HIMON(2)'	1	'CGRESP(1)'	0.25
'PRAMN(2,1)'	390	'EFRGRN(1)'	0	'CGRESP(2)'	0.25
'PRAMN(3,1)'	340	'EFRGRN(2)'	0	'CGRESP(3)'	0.29
'PRAMN(1,2)'	70	'EFRGRN(3)'	0	'NO3PREF(1)'	0.25
'PRAMN(2,2)'	390	'VLOSSP'	0.15	'CLAYPG'	4
'PRAMN(3,2)'	340	'FSDETH(1)'	0.2	'CMIX'	0.5
'PRAMX(1,1)'	50	'FSDETH(2)'	0.95	'TMPGERM'	10
'PRAMX(2,1)'	440	'FSDETH(3)'	0.2	'DDBASE'	1500
'PRAMX(3,1)'	440	'FSDETH(4)'	150	'TMPKILL'	7

'PRAMX(1,2)'	120	'FALLRT'	0.15	'BASETEMP'	10
'PRAMX(2,2)'	440	'RDRJ'	0.05	'BASETEMP(2)'	30
'PRAMX(3,2)'	440	'RDRM'	0.1	'MNDDHRV'	100
'PRBMN(1,1)'	50	'RDSRFC'	0.14	'MXDDHRV'	200
				'CMXTURN'	0.12