GLOBAL BIOGEOCHEMICAL CYCLES, VOL. 25, GB3006, doi:10.1029/2010GB003945, 2011

Controls on soil organic carbon and nitrogen in Inner Mongolia, China: A cross-continental comparison of temperate grasslands

Sarah E. Evans,[1] Ingrid C. Burke,[2] and William K. Lauenroth[3]

Received 17 August 2010; revised 3 March 2011; accepted 22 March 2011; published 12 July 2011.

[1] Most global ecosystem models assume that controls over soil organic matter are alike in climatically similar regions. In this study, we tested the generality of controls over soil organic carbon (SOC) and soil organic nitrogen (SON) in temperate grasslands. We measured organic matter pools in Inner Mongolia, China, along the Northeast China Transect, and analyzed the relationship of SOC and SON to climate, soil texture, and land use variables. We then compared our data to values simulated by a regression model developed in the U.S. Great Plains and also to Century model simulations. We found that, as in the U.S. Great Plains, climate and soil texture variables could explain a large proportion of variation in observed SOC and SON, but a regression model developed in the Great Plains overestimated SOC and underestimated SON in Inner Mongolia. Using Century, we found that simulated SOC and SON values were sensitive to both inclusion of altered land use and changes in N deposition and that the model that best fit our data included higher-intensity grazing and N deposition values higher than that in the Great Plains. This model also produced aboveground net primary production (ANPP) values comparable with values observed in the literature for Inner Mongolian grasslands, but these values were higher than ANPP predicted by previously published regression models. These results suggest that different controls over SOC and SON cycling in Inner Mongolia may affect our ability to predict SOC and SON pool sizes using relationships in other regional models.

Citation: Evans, S. E., I. C. Burke, and W. K. Lauenroth (2011), Controls on soil organic carbon and nitrogen in Inner Mongolia, China: A cross-continental comparison of temperate grasslands, *Global Biogeochem. Cycles*, *25*, GB3006, doi:10.1029/2010GB003945.

1. Introduction

[2] A central challenge in global biogeochemical modeling is developing a generalizable structure that accurately captures variation among ecosystems. Capturing variation in controls on carbon cycling is especially important as it is coupled to and often drives other biogeochemical cycles. Soil organic carbon (SOC) storage is determined by the long-term net balance of photosynthesis and total respiration in terrestrial ecosystems. Therefore, in all systems, factors that influence these processes such as climate, topography, soil texture, and land use management, exert strong control over SOC and soil organic nitrogen (SON) dynamics. However, the relative importance of these parameters, and their relationships to soil organic matter, may vary depending on many different ecosystem properties. In an attempt overcome this variation, most global ecosystem models assume that in climatically similar regions, such as grasslands, relationships between SOC and its environmental controls are the same, despite regions evolving independently. The extrapolation of these relationships in ecosystem models allows us to predict ecosystem dynamics in the future and over large regions for which we have little data, and improve our understanding of these systems. However, to do this we must be certain whether controls that have been established in one temperate grassland can indeed be generalized to other climatically similar regions.

[3] Grassland ecosystems play a significant role in the global carbon cycle, covering nearly one fifth of global land area [*Leith*, 1978] and storing between 200 and 300 Pg of soil carbon [*Anderson*, 1991; *Eswaran et al.*, 1993; *Scurlock and Hall*, 1998]. Climate and soil texture are considered major controls of total soil carbon and the relative proportions of carbon pools in grasslands [*Miller et al.*, 2004; *Plante et al.*, 2006; *Wang et al.*, 2005]; SOC, SON, and C:N generally increase with increasing precipitation and clay content and decreasing temperature [*Burke et al.*, 1989; *Paruelo et al.*, 1998]. However, even within grasslands, the relative importance of each of these factors may shift under altered climate, plant species composition, nutrient input, or land use management [*Miller et al.*, 2004], due to imperfect

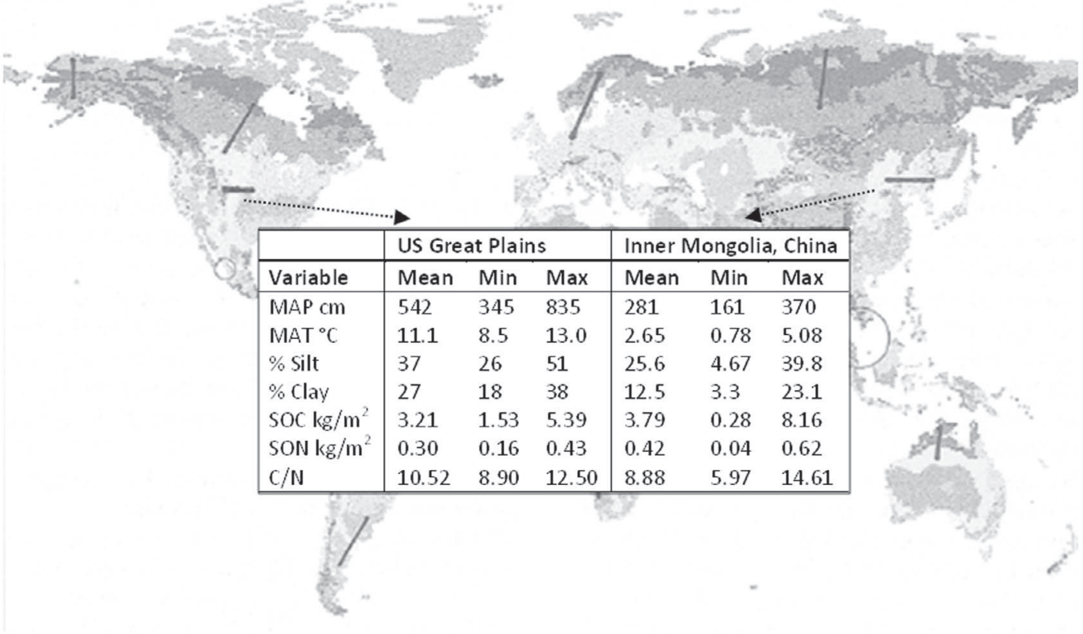
---

[1] Graduate Degree Program in Ecology, Colorado State University, Fort Collins, Colorado, USA.

[2] Environment and Natural Resources Program, Department of Botany, Department of Renewable Resources, and Program in Ecology, University of Wyoming, Laramie, Wyoming, USA.

[3] Department of Botany and Program in Ecology, University of Wyoming, Laramie, Wyoming, USA.

0886-6236/11/2010GB003945



	US Great Plains			Inner Mongolia, China		
Variable	Mean	Min	Max	Mean	Min	Max
MAP cm	542	345	835	281	161	370
MAT °C	11.1	8.5	13.0	2.65	0.78	5.08
% Silt	37	26	51	25.6	4.67	39.8
% Clay	27	18	38	12.5	3.3	23.1
SOC kg/m^2	3.21	1.53	5.39	3.79	0.28	8.16
SON kg/m^2	0.30	0.16	0.43	0.42	0.04	0.62
C/N	10.52	8.90	12.50	8.88	5.97	14.61

Figure 1. International Geosphere-Biosphere Program (IGBP) Global Change and Terrestrial Ecosystems (GCTE) global terrestrial transects, modified from *Koch et al.* [1995]. Lines represent transects of climatic gradients. Data from Inner Mongolia, China, were collected in this study and compared with data from other studies to assess generality of environmental controls over soil organic matter.

ecological convergence and recent global change. Considering these potential influences, it is important to continue to test the generality of grassland models developed in one region for their application for all regions of similar climate.

[4] In this study, we aim to test the generality of the relationships of SOC and SON with environmental factors in temperate grasslands by 1) identifying the most important drivers of soil SOC, SON, and organic matter fractions across a major environmental gradient in China, and 2) assessing the extent to which predictive relationships from North American grasslands are accurate for Inner Mongolia. We do this by examining the relationships among SOC and SON data collected in Inner Mongolia with environmental controls, and also by testing the ability of other grassland models to accurately predict our observed values. We use a grassland regression model developed in the Great Plains [*Burke et al.*, 1989] to test its applicability to Chinese grasslands, and the more highly parameterized Century model [*Parton et al.*, 1987] to investigate which parameters are most important for simulating predictions comparable to our data.

[5] Much of China's temperate grassland lies in the northern province of Inner Mongolia. This arid and semiarid region is predicted to see some of the strongest and earliest effects of climate change [*Intergovernmental Panel on Climate Change*, 2007; *Office for Interdisciplinary Earth Studies*, 1991]. In addition, increasing population in Inner Mongolia has led to increased nitrogen (N) deposition [*Lu and Tian*, 2007] and intensification of land use, which, in addition to altering ecosystem carbon dynamics, has altered soil fertility and threatened personal livelihoods [*Chuluun and Ojima*, 2002; *Jiang et al.*, 2006]. Therefore, in addition to possible differences due to spatially independent evolutionary paths, changes unique to this grassland region could alter fundamental relationships developed in grassland SOC models. In particular, other studies have found that historical land use may alter the relationship between SOC and its environmental controls in this region [*Wang et al.*, 2005; *Zhou et al.*, 2007]. Further, *Chuluun and Ojima* [2002] suggest that, although both are currently changing, land use may be more important than climate parameters in predicting SOC values in the future.

[6] We hypothesize that land use management and nitrogen deposition may be more important controlling factors of SOC and SON in Inner Mongolia than in other temperate grasslands, and that this interaction could alter carbon turnover and fractional pools in the short term, and in the long term, challenge the predictive relationships previously proposed for SOC in temperate grasslands. Land management in Inner Mongolia has a longer history compared to other grassland regions, and has recently intensified [*Xiong et al.*, 2008]. Carbon balance in this area is also sensitive to additional N inputs [*Zeng et al.*, 2010], and has been used to explain observations of higher plant production in this region for a given climate [*Xiao et al.*, 1996]. Therefore, SOC and SON in Inner Mongolia may be even more affected by N deposition [*Lu and Tian*, 2007] occurring as a result of increased population density in this region [*Jiang et al.*, 2006].

2. Methods

2.1. Experimental Approach and Sites

[7] To assess the generality of controls over SOC and SON in semiarid temperate grasslands in this study, we first collected new data from a precipitation gradient in Inner Mongolia, China and analyzed it for significant trends and predictor variables. We then compared this data to output from a model developed from data in the U.S. Great Plains [*Burke et al.*, 1989] and the

Table 1. Description of Sites Sampled Across the Northeast China Transect

Site	Grassland Type	MAP (mm)	MAT (°C)	%Clay Range	%Silt Range	Land Use	Subplots	Mean C (kg m^{-2})
1	Meadow steppe	380	4.77	14–18	31–41	Not grazed	2	0.851
2	Meadow steppe	350	5.08	2–4	5–9	50% grazed	3	2.010
3	Typical steppe	331	3.16	12–17	24–27	Fenced 20 years	2	1.595
4	Typical steppe	331	2.02	10–16	29–37	80% grazed	2	1.899
5	Typical steppe	331	2.02	11–13	20–32	Fenced 9 years	2	2.009
6	Typical steppe	300	0.78	9–23	24–31	Fenced 7 years	2	2.582
7	Typical steppe	300	0.78	8–12	17–24	70% grazed	2	1.853
8	Typical steppe	277	2.67	15–19	33–40	Fenced 10 years	2	2.336
9	Typical steppe	277	2.67	12–15	34–39	70% grazed	2	2.936
10	Desert steppe	178	2.1	8–12	12–29	Fenced 1 year	2	0.611
11	Desert steppe	171	2.1	8–11	16–21	60% grazed	2	0.771
12	Desert steppe	160	2.41	9–13	15–21	60% grazed	2	1.167

Century model [*Parton et al.*, 1987]. These transects in the Great Plains and Inner Mongolia span midlatitude, semiarid temperate grasslands, and were identified by the Global Change and Terrestrial Ecology (GCTE) International Geosphere-Biosphere Program (IGBP) as key gradients that incorporate trends over large spatial scales with regional and global implications [*Koch et al.*, 1995] (Figure 1). We used both a correlative, regional model [*Burke et al.*, 1989] and a simulation model that has been widely validated, the Century model [*Parton et al.*, 1987], to test the generality of the control variables. In contrast to other modeling approaches that focus on parameter optimization and testing of mechanisms, by testing the generality of a simple model and then a highly parameterized model, we could better explain discrepancies that arise between predicted values and values we measured across the Inner Mongolia transect, and improve our understanding about how ecosystem dynamics may differ between the two regions.

[8] The Northeast China Transect (NECT) is located between 112° and 130°E and 42° and 46°N in Inner Mongolia, China. We selected 12 sites on the western 1000 km of the transect. This area spans three types of grasslands: meadow steppe, typical steppe, and desert steppe (Table 1 and Figure 2). Mean annual precipitation (MAP) at the sites ranged from 170 to 450 mm, mean annual temperature (MAT) from 0.78 to 5.6°C, and altitude from 478 to 1550 m (Figure 2). We acquired land use history information from a variety of sources, and although we had information for every site, uncertainty about land use history varied among sites. Most sites were previously established as research sites (Sites 3, 4, 5, 6, 7, 8, 9, 10, 11) and therefore we could accurately and confidently describe the number of years these site had been fenced, or the current grazing intensity (quantified by percent biomass removed per year), and when possible, the land use before the site became a research site. Other sites were private farms (site 2), or had been recently abandoned (site 1) and land use was estimated based on information from the land managers. All information, when possible, was verified with other studies that have previously used this gradient to examine climate and land use effects on environmental factors [*Ni and Zhang*, 2000; *Wang et al.*, 2005; *Zhang et al.*, 1997]. We also used accounts from land managers and several accounts from the literature to obtain information on longer history more general to the region. Much of the land in this region experienced drastic land intensification as a result of population increases and settlement of local farmers in the 1950s [*Jiang et al.*, 2006; *Sneath*, 1998; *Xiong et al.*, 2008], and this was confirmed by many site managers and farmers. In sum, we collected the best possible information about land use but given the very long settlement history of the region, there is substantial uncertainty.

2.2. Sampling

[9] We collected soils in 12 sites across the Northeast China Transect in July of 2008. Within each site, we established two (or in site 2, three) 100 m transects in two areas at least 500 m apart. We estimated soil texture in the field (and later quantified texture in the lab), aiming to maximize variation in soil texture between these transects within a site. We randomly located and collected three 5 × 20 cm cores along each transect, separating soil into a 0–10 cm depth and 10–20 cm depth. Soils were returned to the Chinese Academy of Sciences Institute of Botany laboratory in Beijing within one week. They were dried at 60°C and sieved to remove the soil fraction >2 mm. In all regression analyses, we averaged independent variables over the three cores along each transect, but did not average between transects within a site as soil textures, which were quantified more exactly in the lab, provided additional variation we did not want to ignore. Therefore, we had 12 sites total, but 25 points in our regression because all sites had two transects and site 2 had three transects (Table 1).

2.3. Particulate Organic Matter Fractionations

[10] We used size and density fractionations to estimate coarse and fine particulate organic matter pools (*Cambardella and Elliott* [1992], modified by *Kelly et al.* [1996]). These fractions are also called POM 500 and POM 53 fractions, respectively, referring to the particle size in μm. We shook 30 g soil samples in 0.5 mol L^{-1} sodium hexametaphosphate solution for 18 h and separated the coarse and fine fraction using 0.5 mm and 53 μm sieves, respectively. Carbonates were present in some typical and desert steppe soil samples, as we observed effervescence when soils came in contact with 1M HCl. In these samples, carbonates were removed using an acid pretreatment method [*Nelson and Sommers*, 1982] after fractionation, so removal treatment would not interfere with particle dispersion. We measured C and N on dried, ground soils using a LECO CHN-1000 analyzer. We calculated the mineral associated organic matter (MAOM, *Cambardella and Elliott* [1992]) fraction by subtracting the two POM fractions from the total C. The presence of a significant fraction of labile carbon in the total C would cause an overestimation of MAOM as determined by subtraction, but respiration measurements and previous work in grasslands characterizing

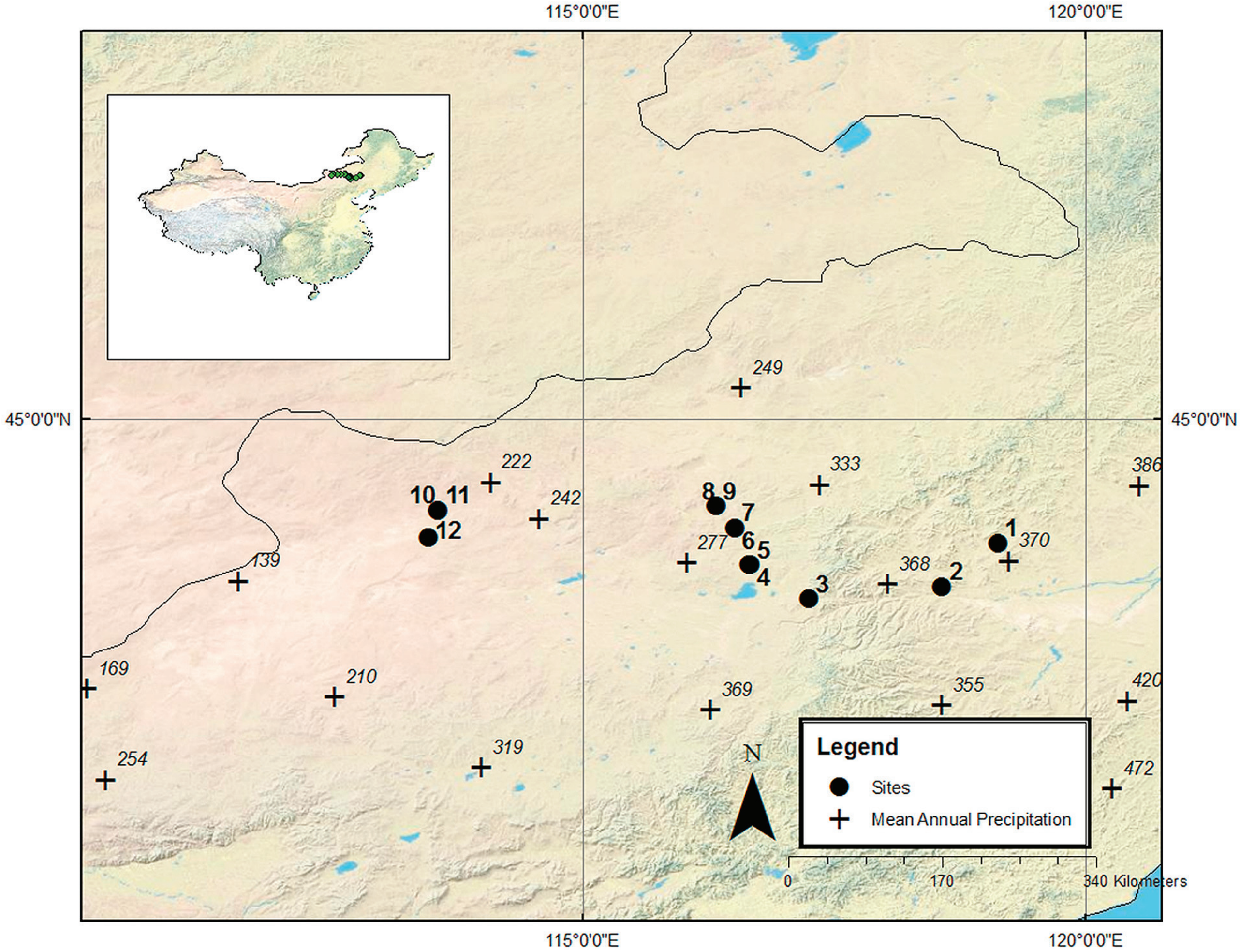


Figure 2. Map of study region in the Inner Mongolia province of China (inset), displaying study sites (solid circles) and average mean annual precipitation (plus signs, italic font) for the area.

highly labile SOC pools [e.g., *Gill et al.*, 1999; *Kelly et al.*, 1996] show that this pool is very small relative to other fractions. Therefore, in this analysis, we felt justified labeling the fraction remaining after subtracting coarse and fine POM from total C and N as MAOM.

2.4. Statistical Analysis

2.4.1. Response of C and N Fractions to Environmental Factors

[11] To describe the major controls over SOC and SON dynamics in Inner Mongolia and their relationship to environmental factors, we first determined the linear relationship of total C and N in soil organic matter and its fractions to *each* independent variable using Pearson's correlation coefficients in proc corr, SAS 9.2 (SAS Institute, Cary, NC). Independent variables included mean annual precipitation (MAP, in cm), mean annual temperature (MAP, °C), silt (%), clay (%), and land use variables that included an estimate of the percent biomass removed per year due to grazing, and the number of years (if any) the area had been fenced. Coarse (500) and fine (53) POM values were better predicted when combined into one POM pool, representing a carbon fraction more labile than MAOM. We added an additional type of dependent variable by calculating the relative proportion of C or N in the POM or MAOM pool as a percentage of total C.

2.4.2. Development of Predictive Models for Total C and N and C and N Fractions

[12] We used a multiple linear regression approach to identify and evaluate the contributions of the strongest predictive variables for all dependent variables. To identify the best predictive models for each variable, we used an all possible subsets regression analysis (SAS proc reg) to select the five models that best fit the data, then a likelihood approach to determine the best predictive model. This approach ranks competing models relative to one another, instead of assuming a true model. Specifically, we used the corrected Akaike information criterion (AICc) to rank models because it includes a correction term for potential bias produced by sample size [*Hurvich and Tsai*, 1989]. AICc judges a model by how closely the fitted values tend to be to the "true values," but also penalizes the model with each added parameter [*Burnham and Anderson*, 2002]. With the best model, we estimated the parameter for each independent variable, tested it for significance ($p < 0.01$), and calculated the standardized coefficient to allow comparison among independent variables that have different units by placing them on the same scale. When independent variables are correlated in

multiple linear regression models, estimations of regression coefficients are not accurate. Therefore, we tested all variables for the occurrence of collinearity, and accounted for this by testing collinear variables individually for significance in case the presence of both caused both to be insignificant parameters.

2.4.3. Comparison of Predictive Models to Observed Values in Inner Mongolia

2.4.3.1. Statistical Comparison of Predicted and Observed Values for All Models

[13] Our goal was to assess the generality of the Great Plains model, first by using a simple regression model, then by varying parameters in the Century model, and comparing the resulting predictions from both to observed values in Inner Mongolia. To statistically evaluate how these values compared, we performed linear regressions between observed values (y) and predicted values (x) for each of the models (regression and Century models), as suggested by *Piñeiro et al.* [2008]. We calculated the r-squared of this relationship and tested the null hypotheses that the estimated slope ($\beta 1$) = 1 and intercept ($\beta 0$) = 0.

[14] To evaluate overall goodness-of-fit, we calculated the root-mean-squared deviation (RMSD) as

$$\mathrm{RMSD} = \sqrt{[1/(n-1)]} \sum_{i=1}^{n} (\mathrm{pre_i} - \mathrm{obs_i})^2 \tag{1}$$

where pre_i represents the predicted values, obs_i the observed values, and n the number of observations. This value represents the mean deviation of the predicted values from the observed. Like the sum of squares, RMSD evaluates the overall goodness-of-fit of the model to the data, but has the advantage of calculating values in the same units as the model variables it describes.

[15] To further partition model error we also calculated Theil's partial inequality coefficients [*Paruelo et al.*, 1998; *Smith and Rose*, 1995; *Theil*, 1958], which separate error into three parts: U_{bias}, which compares the differences in means of observed and predicted values; $U_{\beta=1}$, which quantifies the proportional difference of the slope of the predicted versus observed regression from a 1:1 line; and U_e, which describes the variance that is unexplained after a model is fit to the predicted and observed values. This analysis allowed us to evaluate whether the model residuals are systematic in some way that have functional significance, or the result of unexplainable variability. We calculated these errors terms as follows:

$$U_{bias} = \left[n(\mathrm{OBS} - \mathrm{PRE})^2\right]/\mathrm{SSPE} \tag{2}$$

$$U_{\beta=1} = \left[(\beta - 1)^2 \sum\nolimits_n (\mathrm{pre_i} - \mathrm{PRE})^2\right]/\mathrm{SSPE} \tag{3}$$

$$U_e = \sum\nolimits_n (\mathrm{est_i} - \mathrm{obs_i})^2/\mathrm{SSPE} \tag{4}$$

Where obs and pre represent the observed and predicted value (i subscript) or mean (capital letters), β represents the slope of the regression between observed and predicted, est_i are the values estimated from a fitted linear model developed from the relationship between observed and predicted values, and n is the number of observations. SSPE is the squared sum of the predicted error, calculated as:

$$\mathrm{SSPE} = \sum\nolimits_n (\mathrm{obs_i} - \mathrm{pre_i})^2 \tag{5}$$

We calculated these error terms, in addition to the slope and intercept, to describe how the observed data compared to each prediction using the models described below.

2.4.3.2. Predictive Ability of Great Plains Regression Model

[16] To compare relationships in Inner Mongolia with a previous model developed in the U.S. Great Plains [*Burke et al.*, 1989] for both rangeland and cultivated land, we entered climate and texture data obtained and collected from Inner Mongolia, used the Great Plains model to predict SOC and SON, and regressed the model output against the observed data. Although none of our sites were ever cultivated to our knowledge (only grazed), we compared predictions from the model developed in cultivated sites to our data as an exploratory exercise to see if simulated carbon losses due to cultivation would better predict our data, and to provide insight into a somewhat uncertain historical past of these soils.

2.4.3.3. Incorporating Additional Parameters and Predictive Power Using the Century Model

[17] We used the Century model (v4.5) to ask whether additional parameters acting within a dynamic model could predict soil C and N values observed in this region better than the simple regression model. Because we knew that this region may have undergone significant land use intensification in the 1950s [*Sneath*, 1998; *Xiong et al.*, 2008], and that recent estimates report N deposition levels higher than that in the Great Plains [*Lu and Tian*, 2007], we focused on N deposition and land use as possible influences of simulated SOC and SON. Century is a model that simulates biogeochemical fluxes on a monthly time step [*Parton et al.*, 1987]. It was originally designed in the U.S. Great Plains but has been used extensively all over the world [*Parton et al.*, 1993]. In the model, soil fluxes are controlled by temperature, water, and soil texture, in addition to lignin/N and C/N ratios. Land use history is implemented in Century by designating certain land use types in repeating blocks for specific periods of time. Climate input data for our simulations were obtained from Zhou Guangsheng (personal communication, 2008) from weather stations nearest to the experimental sites. Maximum and minimum monthly temperature and precipitation were averaged over the 50 year climate record; monthly values were stochastically generated by Century based on these means. Soil texture parameters were obtained from texture analyses on soil samples. We tested both observed bulk density values and values calculated using soil texture and SOC [*Rawls*, 1983] to see if this affected the model output, but differences were negligible.

2.4.3.4. Sensitivity of SOC, SON, and ANPP to Elevated Nitrogen Deposition

[18] We were first interested in how elevated N deposition, simulated by Century, affected SOC, SON and aboveground net primary production (ANPP) in Inner Mongolia. We examined the sensitivity of these parameters to N deposition 1) at equilibrium (light grazing for 5000 years), and also 2) when including probable land use histories of the sites and region, in order to better compare simulated values to observed values.

[19] N deposition has increased in this area in the last 60 years, as population levels have increased [*Jiang et al.*, 2006]. Current estimates in western Inner Mongolia are within the range of 0.32–1.15 g N m^{-2} yr^{-1} [*Lu and Tian*, 2007]. The default values for Century, stemming from estimates in the Great Plains, are about 0.3 g N m^{-2} yr^{-1}. We tested three N deposition values at equilibrium: 0.05, 0.9, and 1.5 g N m^{-2} yr^{-1}. We treated these parameters as fixed (not as a function of precipitation) in order to simplify our sensitivity analysis, and did not include any additional land use changes after the time when the model reached equilibrium. By doing this, we were first able to see how SOC, SON, and ANPP responded to changes in N deposition in this area *at equilibrium*.

[20] However, studies reporting land use changes in the last 60 years in this region suggest the assumption that these sites are currently at SOC and SON equilibrium is not valid, and that these values would not be comparable to observed data. To simulate SOC and SON values that were more comparable to the values we observed in Inner Mongolia, we did a second analysis of N deposition, which included a 60 year period of intensive grazing [*Xiong et al.*, 2008], and the current known land use type. Under these conditions we simulated two N deposition levels: 1) parameter values for N deposition used for Great Plains Century simulations and 2) 0.9 g N m^{-2} yr^{-1}. Century4.5 simulations for the Great Plains model N deposition as a function of precipitation, using two parameters, epnfa(1) and epnfa(2) as slope and intercept. Parameterization for the Great Plains (epnfa(1) = 0.21 and epnfa(2) = 0.0028) result in an average N deposition of 0.3 g N m^{-2} yr^{-1} over all Inner Mongolia sites (such that N deposition = 0.21+ precip*0.0028), as sites have an average mean annual precipitation of about 35 cm.

2.4.3.5. Sensitivity of SOC and SON to Inclusion of Periods of Intensive Land Use

[21] Given that this region experienced significant land use intensification in the 1950s [*Sneath*, 1998; *Xiong et al.*, 2008], and that current land use practices on each site varied, we were also interested in whether the inclusion of specific periods of changes in land use would result in SOC and SON predictions closer to observed values than predictions by the Great Plains regression model. Century simulates land use changes over time by separating land use into periods within which specified events repeat. We separated the history of these sites into 3 periods: equilibrium, which consisted of light grazing and lasted 5000 years; 60 years of intensive grazing, as a result of population growth, settlement, and land use intensification in the area beginning in the 1950s [*Jiang et al.*, 2006; *Sneath*, 1998; *Xiong et al.*, 2008]; and current (20 years or less) land use based on our knowledge from each site (described in Table 1). Although we could not confirm that all of our sites experienced increases in grazing intensities in the 1950s, studies suggest that this trend occurred in the region as a whole, and we were interested to know whether this was an important factor. Grazing effects were determined based on relationships described by *Ojima et al.* [1990] and *Holland et al.* [1992], in which the relationship of grazing to biomass production changes as grazing intensity increases. This approach has been used to simulate grazing variation in other studies in this region [*Wang et al.*, 2007].

[22] We wanted to see how the inclusion of these periods in the model, individually and in combination, affected SOC and SON output for sites in Inner Mongolia. Therefore, we tested three different "histories": 1) a 5000 year equilibrium period, and a period of current known land use 2) a 5000 year equilibrium period, and 60 year period of more intense grazing and 3) a 5000 year equilibrium period, a 60 year period of more intense grazing, and a period of current known land use. We used a scaled N deposition of 0.9 g N m^{-2} yr^{-1} for each of these runs.

2.4.3.6. Comparison of Model Predictions of ANPP to Observed ANPP in Inner Mongolia

[23] Because most ecosystem carbon enters the system through photosynthesis, ANPP represents a major control over organic matter storage. In this way, simulated ANPP values can provide additional insight into variation in SOC and SON under different modeling scenarios. Adjustments in N deposition and land use affect ANPP, and we were interested in whether simulated ANPP fit with observed ANPP in Inner Mongolia under the same scenarios that simulated SOC and SON fit with observed SOC and SON in Inner Mongolia. Because we did not measure ANPP in Inner Mongolia in 2008, we used ANPP values described in the literature across the Northeast China Transect [*Hu et al.*, 2007; *Yu et al.*, 2004; *Zhou et al.*, 2006]. We compared these "observed" values to ANPP simulated by Century under parameters that produced the best fit to SOC and SON (elevated N deposition and inclusion of intensive land use). We also compared observed values to ANPP predicted for these sites by two regression models relating ANPP to mean annual precipitation: one developed in the Great Plains [*Sala et al.*, 1988], and one developed along the Northeast China Transect [*Zhou et al.*, 2002a].

3. Results

3.1. Response of C and N Fractions to Environmental Factors

[24] Total C and N increased as precipitation and fine-textured soil increased, and decreased as mean annual temperature (MAT) increased (Table 2). Total C contained in the intermediate POM fraction increased with MAT and decreased with percent silt and clay, whereas percent C in MAOM fraction, representing passive C associated with silt and clay particles, had the opposite response to these factors. Total C:N significantly correlated with total SOC, but surprisingly, decreased as total SOC increased. Both "Biomass Removed per Year" and "Time Fenced" were tested against all response variables in linear regressions, but alone did not significantly explain any of the variation (and therefore are not listed in Table 2).

3.2. Multiple Regression Models for Observed Independent Variables

[25] Best-fitting models revealed that with climate, texture, and interactions terms alone, we could explain 76% of the variability in total SOC and 71% of total SON in this region (Table 3). Land use terms ('Biomass Removed per Year' and 'Time Fenced') were not significant explanatory variables for total SOC when included in the model, but contributed significantly to the POM-C, POM-N and MAOM-C models. Longer periods of time that sites were fenced resulted in increased MAOM passive C, but decreased POM-C.

Table 2. Correlations Between Parameters and Dependent Variables Measured in Inner Mongolia[a]

	MAP	MAT	% Silt	% Clay	TotC[b]
Total C	+	−	+	+	
POM-C	+	+	+		
%POM-C		+	−	−	
MAOM-C	+	−	+	+	
%MAOM-C		−	+	+	
Total N	+	−	+	+	+
POM-N		+	+	+	+
%POM-N		+	+	+	+
MAOM-N		−	−		
%MAOM-N		−	−	−	−
C:N					−
C:N POM		−	−	−	−
C:N MAOM				+	

[a] All relationships (positive or negative) reported were significantly related in a Pearson Correlation ($p < 0.05$). "Percent biomass removed per year" and "Time Fenced" were not reported because there were no significant relationships with dependent variables.

[b] Total Carbon is listed as an independent variable to show with which dependent variables it correlated.

3.3. Generality of Great Plains Regression Model

[26] To determine the generality of the SOC and SON models developed in the U.S. Great Plains compared to other temperate grassland regions, we compared the observed carbon values in Inner Mongolia to values predicted by a regression model for the U.S. Great Plains [*Burke et al.*, 1989] (Figure 3). The model from the Great Plains explained a significant proportion of the variation in the China soils ($r^2 = 0.58$, $p < 0.0001$; see Table 4). However, on average, observed values for China were 30% lower than predicted values from the U.S. Great Plains model, and regressions between predicted and observed values included an intercept term significantly different than 1 for both SOC and SON (Table 4). Predictions of soil N by the U.S. model also explained a large proportion of the variability ($r^2 = 0.85$, $p < 0.0001$), but in contrast, *under*estimated total N values along the Northeast China Transect (intercept significantly greater than 1; see Table 4). When error was partitioned, more error was found in U_{bias} and U_e terms, and $U_{\beta=1}$ error was low.

[27] *Burke et al.* [1989] also developed a model predicting C and N in cultivated sites, and we measured how predictions from this model fit our data as a general investigation of how the incorporation of land use might affect the goodness of fit (Figure 3). The cultivated model did not produce a higher r^2 value (SOC: $r^2 = 0.39$, SON: $r^2 = 0.77$), but predicted lower C and higher N than for range soils (closer to our observed values), a lower bias term, and a lower RMSD.

3.4. Century Simulations

3.4.1. Sensitivity of SOC, SON, and ANPP to Elevated Nitrogen Deposition

[28] Century simulations revealed that adjusting N deposition and land use history parameters produced simulated SOC and SON closer to observed values. At *equilibrium* (no land use scenarios included), SOC, SON, and ANPP were sensitive to changes in N deposition, but showed a greater response to deposition changes from 0.05 to 0.9 g N m^{-2} yr^{-1} than from 0.9 to 1.5 g N m^{-2} yr^{-1} (Figure 4). Comparisons among simulations of the three N deposition levels at equilibrium and observed values suggested that SOC and SON simulated at an N deposition level of 0.9 g N m^{-2} yr^{-1} produced the best fit with observed data (smallest RMSD; see Table 4). When decomposing model error, U_e and U_{bias} error terms were highest, suggesting that the predicted values have a consistent relationship to observed values at different N deposition levels at equilibrium ($U_{\beta=1}$ was low), but simulations at equilibrium left a large amount of variability unexplained (high U_e; see Table 4).

[29] We also tested Great Plains and elevated N deposition levels, while including land use conditions according to our knowledge for the region and each site (60 year period of intensive grazing and current known land use for each site) to better compare model output with observed data.

Table 3. Best Predictive Models for SOC, SON, and C and N Fractions Measured in Inner Mongolia[a]

Variable	Coefficient	Standardized Coefficient	p Value
Total Carbon: $R^2 = 0.75$, Adj $R^2 = 0.75$			
MAT^2	0.163	0.6825	0.0322
MAT	−1.085	−0.7529	0.0137
MAP^2	−0.016	−3.2491	<.0001
MAP	0.858	3.3207	<.0001
%Clay	0.164	0.3291	0.0005
%Silt	0.089	0.4396	<.0001
Intercept	−9.725		<.0001
% POM-C: $R^2 = 0.37$, Adj $R^2 = 0.34$			
MAT	0.092	0.4333	<.0001
%Silt	−0.008	−0.2655	0.0116
%BiomassRemvd/yr	−0.003	−0.4510	0.0048
Years fenced	−0.021	−0.5211	0.0012
Intercept	0.571	0	<.0001
% MAOM-C: $R^2 = 0.475$ Adj $R^2 = 0.442$			
MAP*MAT	−0.003	−0.5370	<.0001
%Silt	0.013	0.4248	<.0001
%BiomassRemvd/yr	0.003	0.4527	0.0021
Years fenced	0.023	0.5209	0.0005
Intercept	0.234	0	0.0155
Total Nitrogen: $R^2 = 0.71$ Adj $R^2 = 0.68$			
MAT^2	0.026	1.147	0.0110
MAP*MAT	−0.006	−1.839	0.0004
MAP	0.007	0.269	0.0531
MAP*%silt	0.001	0.965	<.0001
%Biomass Removed/yr	0.002	0.335	0.0043
Years fenced	0.011	0.448	0.0007
Intercept	0.031	0	0.6064
% POM-N: $R^2 = 0.851$ Adj $R^2 = 0.839$			
MAT^2	−0.333	−2.443	<.0001
MAP^2	−0.028	−11.048	<.0001
MAP	1.322	9.834	<.0001
MAP*MAT	0.077	3.834	<.0001
Years fenced	−0.051	−0.355	<.0001
Intercept	−14.977	0	<.0001
% MAOM-N: $R^2 = 0.474$ Adj $R^2 = 0.453$			
MAP*MAT	0.846	1.459	<.0001
MAT*%silt	−0.577	−1.254	<.0001
Intercept	1.608	0	0.0768

[a] Multiple linear regression models were determined using all possible subset selections of six independent variables: mean annual temperature (MAT in °C), mean annual precipitation (MAP in cm), silt and clay (%), and two metrics of land use: number of years fenced and percent biomass removed per year due to grazing.

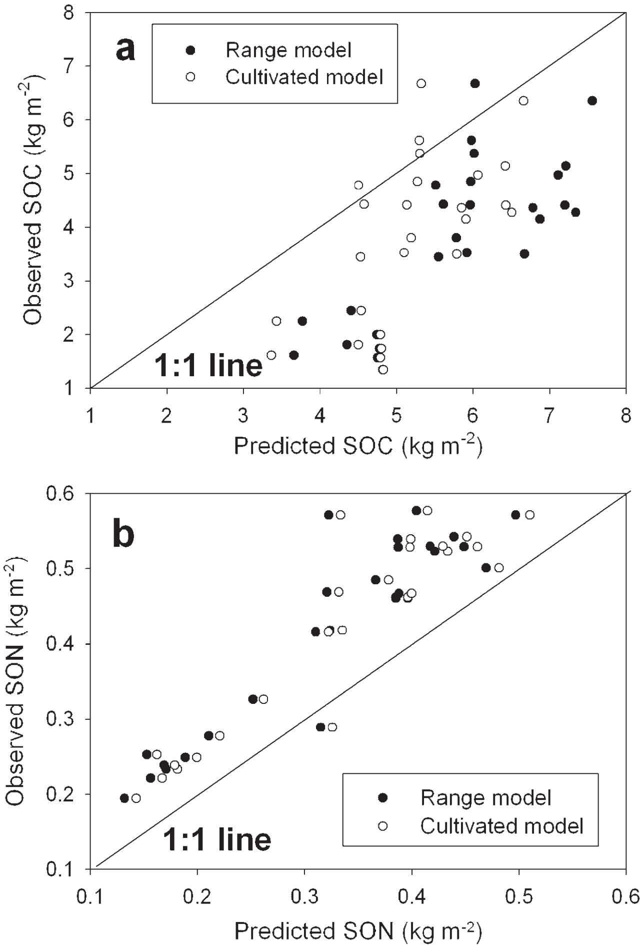


Figure 3. Predicted values from the Great Plains regression model [*Burke et al.*, 1989] compared to observed values from Inner Mongolia for (a) SOC and (b) SON. We simulated values for Inner Mongolia using both the model developed for U.S. rangeland (solid circles) and cultivated land (open circles).

After including these periods, SOC and SON under elevated N deposition (0.9 g N m^{-2} yr^{-1}) was significantly related to the observed values (SOC: $r^2 = 0.53$, $p < 0.0001$; SON: $r^2 = 0.67$ $p = 0.002$) and had a lower RMSD than values under N deposition parameters used for the Great Plains (SOC: $r^2 = 0.221$ $p = 0.78$; SON: $r^2 = 0.043$ $p = 0.062$) (Figure 5 and Table 4). The lack of fit in the Great Plains model was primarily related to unexplained variance (U_e), and bias (U_{bias}). Although overall error was low, any remaining error in the elevated N deposition model was most attributed to unexplained variance (U_e) for SOC, and lack of consistency ($U_{\beta=1}$) for SON.

3.4.2. Sensitivity of SOC and SON to Changes in Land Use History

[30] SOC and SON were closest to observed values under land use scenario 3, as determined by the lowest RMSD when compared with the data (Table 4). This scenario included a 5000 year period of equilibrium, a 60 year period of intensive grazing and the current known land use for each site (e.g., fenced 7 years, heavily grazed 3 years, etc. depending on the site). The inclusion of only the current land use period (scenario 1) produced the worst fit with the observed values, including a slope significantly lower than 1 and an intercept significantly higher than 0 (Figure 6 and Table 4). Inclusion of the 60 year period (scenario 2) produced predictions with a statistically significant relationship to observed values. The unexplained variance (U_e) made the largest contribution to the lack of fit for the competing models, and remained the largest source of error even in the best-fitting model. Thus, the superior fit of the best model was the result of reduced error in model consistency ($U_{\beta=1}$) and bias (U_{bias}).

3.4.3. Simulated ANPP for Inner Mongolia Compared to Values Observed in the Literature

[31] We examined the ANPP output from Century under the model scenario that produced SOC and SON closest to observed values (0.9 g N m^{-2} yr^{-1} N deposition and intensive and current land use periods; see Table 4). Simulated ANPP by this Century model were comparable to those estimated in the literature for these sites ($r^2 = 0.72$ $p < 0.001$) (Figure 7 and Table 4). Values predicted by the Great Plains ANPP regression model [*Sala et al.*, 1988] were also significantly related to estimated ANPP from the literature ($r^2 = 0.54$ $p < 0.01$), but values from the regression model were generally *lower* than observed ANPP (intercept = 32.6, $p < 0.05$). Much of the observed lack of fit was associated with mean differences (U_{bias}) and lack of consistency ($U_{\beta=1}$). A regression model developed in this area by *Zhou et al.* [2002b] provided the best fit to observed values ($r^2 = 0.76$ $p < 0.001$) and the lowest RMSD of the three models.

4. Discussion

[32] We found that although SOC and SON in Inner Mongolia are controlled by the same climate and texture variables used to predict SOC and SON in the Great Plains, in this region, values of SOC were lower, SON higher, and ANPP higher than those predicted by regression models developed in the Great Plains. The incorporation of both elevated N deposition and an intensive land use history in Century was necessary to obtain simulated values of SOC, SON, and ANPP near the values observed across the Northeast China transect.

4.1. Response of SOC to Land Use and Texture in Inner Mongolia

[33] We used two approaches to compare soil organic matter response to climate and land use in grasslands across continents: (1) we determined the relationships – and importance of the relationships – of climate, texture, and land use to total SOC and SON and fractions, and compared them to relationships described for other grasslands in previous studies and (2) we compared the values predicted by previous grassland regression and simulation models to those observed in Inner Mongolia (section 4.3).

[34] Our results show that climate and texture variables exert dominant controls on all fractions of soil organic matter in Inner Mongolian grasslands. SOC increased with increasing precipitation and decreased with increasing temperature. Other studies have found similar trends in temperate semiarid regions [*Alvarez and Lavado*, 1998; *Burke et al.*, 1989; *Paruelo et al.*, 1998], and across this transect [*Zhou et al.*, 2002b]. This trend can be explained by production responding more than decomposition to increased precipitation across the spatial gradient in water-limited areas, and decomposition rates respond-

Table 4. Summary of Regression and Goodness of Fit Statistics for Model Simulations Compared to Observed Data[a]

Model Information		r^2		Slope		Intercept		RMSD[b]		U_{bias}[c]		$U_{\beta=1}$[d]		U_e[e]	
N Deposition	Land Use	SOC	SON	SOC	SON	SOC	SON	SOC	SON	SOC	SON	SOC	SON	SOC	SON
	CENTURY: Sensitivity to Elevated N Deposition at Equilibrium														
0.05	Equilibrium	**0.17**	0.077	−1.524	0.447	**4.332**	0.381	3.428	1.34	0.850	0.068	0.012	1.4E-03	0.998	0.073
0.9	Equilibrium	**0.849**	**0.472**	1.09	**2.51**	−0.253	**−0.791**	0.579	1.2	8.2E-03	2.9E-03	0.037	2.8E-03	0.320	0.012
1.5	Equilibrium	**0.78**	**0.227**	**0.684**	0.738	0.637	−0.087	1.153	1.18	0.465	0.052	0.267	4.3E-04	0.419	0.057
	CENTURY: Sensitivity to Inclusion of Land Use Periods														
0.9	Current land use only	0.134	0.231	**0.211**	**0.216**	**2.434**	**0.297**	2.579	1.21	0.388	0.003	0.053	1.1E-03	0.934	0.022
0.9	60-year intensive grazing	**0.456**	0.232	0.536	0.482	1.103	0.192	1.873	1.18	0.534	0.010	0.145	0.013	0.774	0.035
0.9	60-year grazing & current land use	**0.531**	**0.672**	0.993	0.725	0.024	0.257	1.43	0.24	4.9E-05	0.074	3.7E-05	0.004	0.036	0.059
	CENTURY: Elevated Versus Great Plains N Deposition With Best-Fitting Land Use Scenarios														
0.9	60-year grazing & current land use	**0.531**	**0.672**	0.993	0.725	0.024	0.257	1.43	0.24	4.9E-05	0.074	3.7E-05	0.004	0.036	0.059
0.3[f]	60-year grazing & current land use	0.221	0.043	0.888	**0.648**	**1.788**	0.095	2.006	0.19	0.653	0.192	2.0E-03	0.025	1.002	0.535
		r^2		Slope		Intercept		RMSD[b]		U_{bias}[c]		$U_{\beta=1}$[d]		U_e[e]	
Citation	Developed in	SOC	SON	SOC	SON	SOC	SON	SOC	SON	SOC	SON	SOC	SON	SOC	SON
	Great Plains Regression														
Burke et al. [1989]	US Great Plains rangeland sites	**0.577**	**0.853**	1.045	1.094	**−2.243**	**0.064**	2.267	0.351	0.795	0.076	0.001	0.001	0.892	0.083
Burke et al. [1989]	US Great Plains cultivated sites	**0.389**	**0.772**	1.124	1.087	−2.029	0.054	1.878	0.346	0.568	0.060	0.003	0.001	0.868	0.067
Citation	Developed in	ANPP[h]		ANPP		ANPP		ANPP		ANPP		ANPP		ANPP	
	ANPP Predictions From Century and Regressions Models Compared to ANPP in Inner Mongolia														
Sala et al. [1988]	US Great Plains for ANPP	**0.538**		0.887		**32.569**		33.646		0.359		0.517		0.340	
CENTURY[g]	Best-fitting SOC and SON model	**0.717**		1.168		−2.937		31.862		0.295		0.288		0.659	
Zhou et al. [2002a]	Inner Mongolia grasslands for ANPP	**0.763**		0.923		18.346		27.705		0.014		0.208		0.097	

[a] Regressions represent observed (y) versus predicted (x). Bold indicates regression model (r-squared) is significant, slope is significantly different than 1, and intercept is significantly different than 0 ($p < 0.01$).

[b] Root-mean-squared deviation, represents overall goodness-of-fit of model (see equation (1)).

[c] Error term that compares the differences in means of observed and predicted values (see equation (2)).

[d] Error term which quantifies the proportional difference of the slope of the predicted versus observed regression from a 1:1 line (see equation (3)).

[e] Error term that describes the variance unexplained by a model fit to the observed values (see equation (4)).

[f] Value represents average nitrogen deposition simulated for the Great Plains; nitrogen deposition varied with precipitation.

[g] This model was developed in this paper using Century and included an elevated N deposition level (0.9) and a period of 60 year grazing and current land use.

[h] Regression of Century model simulations of ANPP (y) compared to observed ANPP from literature (x).

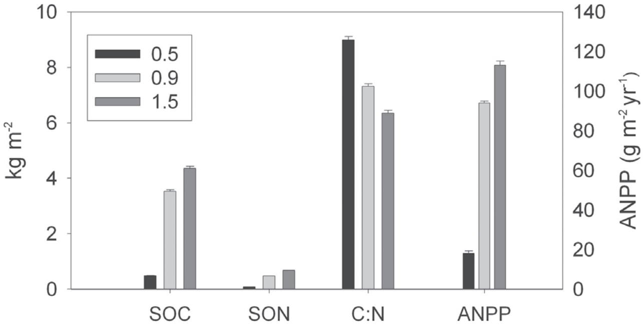


Figure 4. Average SOC (kg C m^{-2}), SON (kg N m^{-2}), C:N, and ANPP (g m^{-2} yr^{-1}, second axis) across all sites simulated by Century at three different nitrogen deposition levels (g N m^{-2} yr^{-1}) for Inner Mongolian sites. Output was recorded after 5000 years of equilibrium conditions at each site, with no current land use periods.

ing more strongly than production to higher temperatures across the gradient [*Epstein et al.*, 2002; *Guo et al.*, 2006]. However, our total SOC explanatory power ($R^2 = 0.76$; see Table 3) with only these variables surprised us; we sampled across a land use gradient, and studies quantifying the losses due to degradation in this area suggest that land use is a strong determinant of SOC – and perhaps an even stronger control than climate – of SOC in this area [*Chuluun and Ojima*, 2002]. For this reason, we originally hypothesized the land use in this area would alter the predictive power that climate variables have over SOC and SON values. One possible reason for the lack of a significant role of land use parameters in the multiple regression we developed is that our variables describing land use were only describing present or recent (decadal scale) practices, and could not describe longer-term effects that would have a larger impact on current SOC and SON levels. However, this as an explanation alone would have resulted in a much lower R^2 value than we produced with climate and texture data. Our comparison of these results with data and models developed in other continents allowed us to gain more insight into the importance of historical land use on SOC values in this area.

4.2. Relationships of SOC to Environmental Variables

[35] Overall, other models relating environmental variables to observed SOC and SON found similar relationships (positive or negative; see Table 2) as we did among variables. For instance, *Burke et al.* [1989] found that similar factors as we included in our model should be included in a predictive model in the Great Plains, but found a change in a standardized MAT unit caused the largest change in SOC. In our study, MAP was the most important, as *Guo et al.* [2006] found for areas receiving less than 1000 mm MAP in the United States. In U.S. forests, *Homann et al.* [1995] found climate and texture variables explained a large proportion of the variation in SOC, but MAT had a stronger, and positive, effect on SOC. *Percival et al.* [2000] sampled sites in New Zealand and found that soil chemical characteristics, rather than climate or soil texture, explained much more of the variation in SOC in grasslands there.

[36] In contrast to the most dominant controls over total C, POM-C and MAOM-C fractions were most strongly related to temperature, texture, and land use (Table 3). Percent MAOM-C decreased with MAT, suggesting that the proportion of soil carbon that is recalcitrant decreases with increasing temperature. Previous studies across climatic gradients have suggested that SOC recalcitrance *increases* with increasing temperature (and decreasing total SOC) [*Trumbore et al.*, 1996], but several studies have challenged this idea and its implications for possible decomposition feedbacks to predicted temperature increases [*Giardina and Ryan*, 2000]. Percent POM-C and MAOM-C were not significantly correlated to land use variables in simple regressions (Table 2), but these variables were significant predictors in multiple regressions (Table 3). Percent POM-C declined with increased grazing intensity, possibly because this fraction is more reduced by decreases in plant inputs than the total C pool [*Kelly et al.*, 1996]. Current grazing intensity was not a significant predictor of MAOM-C, but MAOM-C was higher in sites that were fenced. Studies examining recovery after cessation from cultivation [*Burke et al.*, 1995] and grazing

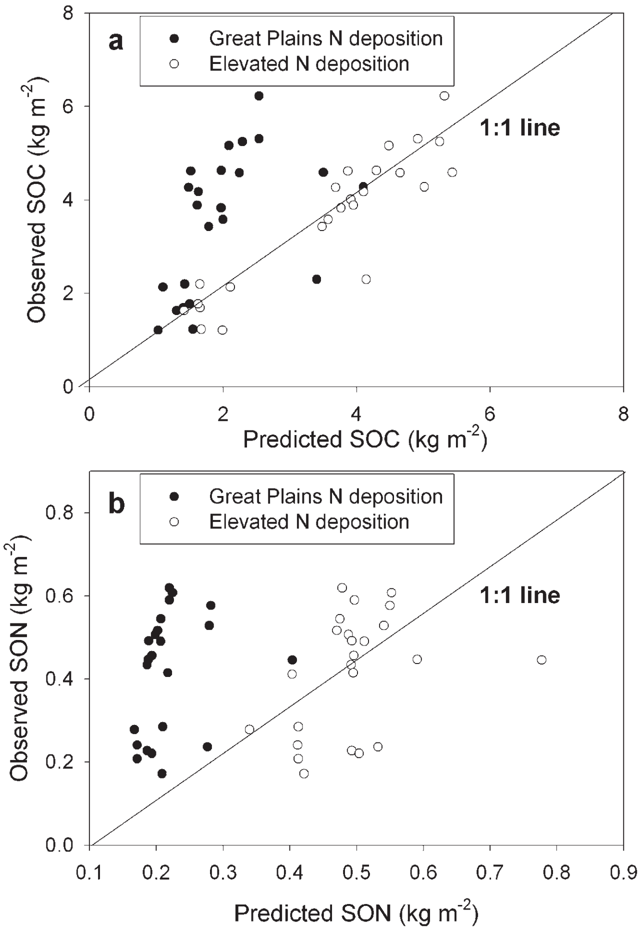


Figure 5. (a) SOC and (b) SON predicted by Century using nitrogen deposition as parameterized in the Great Plains (solid circles), which varies with precipitation but results in an average of 0.3 g N m^{-2} yr^{-1}, and fixed at 0.9 g N m^{-2} yr^{-1} (open circles), compared to observed values in Inner Mongolia at all sites. All runs included a 60 year intensive grazing period followed by a unique current land use period depending on the site. This combination proved to be the land use scenario that best predicted observed values (see Table 4).

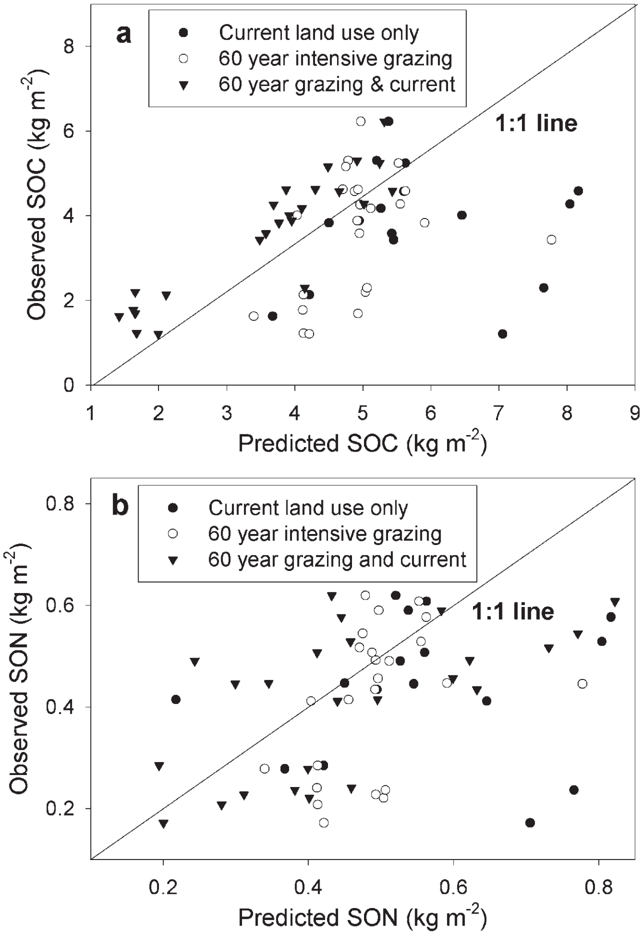


Figure 6. (a) SOC and (b) SON predicted by Century when we varied the inclusion of certain land use periods, compared to observed values in Inner Mongolia at all sites. Current land use refers to the information we acquired at sites from the current manager and is a 1 to 20 year history depending on the site. Sixty year intensive grazing is a period before the "current" land use that was implemented in all sites to represent the increase in grazing intensity as population density increased in Inner Mongolia around the 1950s. All runs used nitrogen deposition of 0.9 g N m^{-2} yr^{-1}.

[*Steffens et al.*, 2011] have similarly detected changes in C fractions and not total C in decadal recovery, but it is surprising that the return of C in our study occurs in the pool with the slowest turnover time. Studies examining recovery after cultivation, and a few after grazing, have found that recovery of this pool is not detectable on a decadal timescale [*Burke et al.*, 1999; *Robles and Burke*, 1998]. However, several recent studies on grazing have reported an increase in the passive pool with grazing exclosures, with no change in the intermediate (POM) pool [*Altesor et al.*, 2006; *Piñeiro et al.*, 2009].

4.3. Comparison of Model Predictions to Observed Values in Inner Mongolia

4.3.1. Regression Model

[37] Soil C and N in the U.S. Great Plains were also best predicted by precipitation, temperature, and soil texture, and had the same relationships (positive or negative) as our response variables did to each predictor variable [*Burke et al.*, 1989]. The model produced a good fit to our observed values, capturing a large amount of variability in our Inner Mongolia data set, which was surprising given the simplicity of this model and considering what our remaining analyses reveal (i.e., a strong sensitivity of soil organic matter to unknown past land use and N deposition). The main discrepancy was not consistency (slope deviating from 1), but that the U.S. Great Plains model overestimated C values and underestimated N values observed in Inner Mongolia. *Paruelo et al.* [1998] used this Great Plains model to assess the SOC drivers and predictability of another GCTE temperate grassland transect in Argentina, and found SOC observations at this site fit well with the Great Plains model (r^2 = 0.63), and, unlike our data from Inner Mongolia, fell evenly above and below the model predictions (y intercept of best-fitting line did not differ significantly from 0). Our model-data comparisons suggest that the relationship of SOC and SON to climate variables predicted by the Great Plains model is *consistent* (slope near 1), but that the Great Plains regression models produce a poorer fit to observations because of bias (differences of means, intercept greater or less than 0) and additional variance in the data that could not be explained by the model (Figure 3 and Table 4).

[38] In Inner Mongolia, observed carbon values that were *closest* to predicted values for rangeland models were from those sites along the Inner Mongolia transect that had been fenced for 12 and 20 years. In addition, the U.S. model developed from cultivated sites in the U.S. produced less bias than the rangeland model. Our sites were *not* cultivated, but the improved fit to our data suggested the gaps between observed and predicted values from the rangeland model may be due to reduced carbon storage as a result of long-term intensive land use in the Inner Mongolia region that could not be quantified by current land use metrics. Although this is impossible to know due to lack of precise

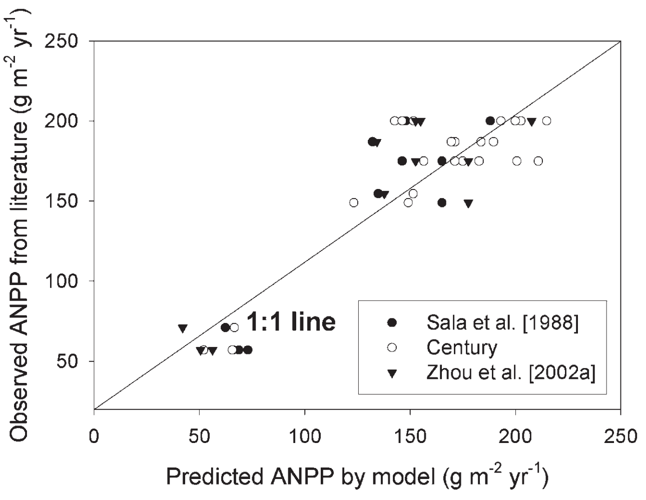


Figure 7. Simulated ANPP (g m^{-2} yr^{-1}) from *Sala et al.* [1988] (ANPP = −34 + 0.6 × MAP), *Zhou et al.* [2002a] (ANPP = −84.8 + 0.7905 × MAP) and as simulated by Century (all land use periods, N deposition of 0.9 g N m^{-2} yr^{-1}) versus ANPP estimates from the literature across the Northeast China Transect in Inner Mongolian grasslands.

knowledge of historical land use at these sites, several empirical and modeling studies have demonstrated the extent of the effect of historical land use on current SOC observations and processes. *Sandor et al.* [1986] found SOC was still 40% lower in areas that had been intensively cultivated but abandoned 1000 years ago in New Mexico. Although grazing may have varying short-term effects on SOC, studies suggest that when it does greatly reduce SOC, either due to its high intensity or the absence of evolutionary adaptation to grazing, effects can be similar to that of cultivated areas. *Piñeiro et al.* [2006] used coupled grazed and fenced sites and found that long-term grazing reduces SOC by 15–30%, with largest reductions occurring in the slow and passive pools. Century simulations suggest that 400 years of low intensity grazing would produce a 10% reduction in SOC [*Alvarez*, 2001]. Few long-term studies testing the effect of long-term grazing and subsequent exclosures exist due to the absence of a reliable control, but observed reductions in slow and passive carbon pools suggest that recovery would be similar to recovery from these effects due to cultivation (i.e., on the order of centuries), and be longest in areas receiving the least precipitation [*Paustian et al.*, 1997].

4.3.2. Century Model

[39] Using the U.S. regression model [*Burke et al.*, 1989], we found observed SOC in Inner Mongolia was lower than predicted by the U.S. model, and observed SON was higher than predicted by the model (Figure 3 and Table 4). Therefore, we used the Century model to ask what other factors – that we did not measure – might explain these results, and what changes were needed to simulate values closer to observed data. SOC and SON were both sensitive to changes in N deposition, especially changes from very low (0.05 g N m^{-2} yr^{-1}) to values of deposition that might be occurring today (0.9 g N m^{-2} yr^{-1}) (Figure 4 and Table 4). SOC and SON were not as sensitive to changes in N deposition from 0.9 g N m^{-2} yr^{-1} to 1.5 g N m^{-2} yr^{-1}, but sites in the meadow steppe (wetter end of the gradient) were more responsive than sites in the drier end, suggesting that the diminished response could be related to water limitation.

[40] SOC and SON were, on average, closest to observed values when deposition was 0.9 g N m^{-2} yr^{-1}, after incorporating the land use history that is likely for this region (addressed below). This value for N deposition is approximately 0.4–0.5 g N m^{-2} yr^{-1} higher than parameterized simulations in the Great Plains. When we used the same parameters used in the Great Plains, in addition to all land use scenarios, we found carbon was underestimated by an average of 1.45 kg C m^{-2} yr^{-1} and N was underestimated by 0.18 kg N m^{-2} yr^{-1} (Figure 5). These results are in line with our findings from the U.S. regression model, which did not include N deposition and underestimated N (and arguably underestimated C as well, when no land use was included) compared to observed values. We were able to explain this discrepancy by increasing N input, and increasing ANPP, using the Century model. This suggests that N deposition, or some N input, is important in elevating ANPP in Inner Mongolia and producing SOC and SON values that are higher than expected based on Great Plains data.

[41] SOC and SON were also sensitive to changes in land use history. The inclusion of 60 years of intensive grazing before the current land use period had a stronger effect on SOC and SON than the inclusion of the current known period of grazing (1–20 years) (Table 4). These current land use periods varied depending on site, but included fencing treatments or various grazing intensities for different (known) lengths of time. The simulations, however, suggest that the potential degradation and reduction in SOC and SON caused by the period of land intensification as a result of settlement and increased population was more important in determining SOC levels. However, SOC and SON responses to changes in simulated land use history were not consistent among site (Figure 6). This may be because differences in site climate and soil type cause sites to respond differently to intensive land use in this area, but it is more likely due to inaccuracy of land use history before the current period (i.e., some sites may have experienced more or less intensive land use prior to the current land use). This is supported by the fact that unexplained error (U_e) still contributed to any remaining lack of fit in the model with the best-fitting land use scenarios (Table 4).

[42] We did not measure ANPP at our sites in the summer of 2008, but previous studies recorded ANPP at these and other sites along the Northeast China Transect [*Hu et al.*, 2007; *Yu et al.*, 2004], and *Zhou et al.* [2002a] developed a regression model to predict ANPP in this region using annual precipitation. Perhaps not surprisingly, ANPP for the Northeast China Transect sites fit well with those predicted by this model developed in Inner Mongolia [*Zhou et al.*, 2002a] (Figure 7 and Table 4). However, values reported for the desert steppe, typical steppe, and meadow steppe in Inner Mongolia in the literature were slightly *higher* than values predicted by the widely used ANPP regression model presented by [*Sala et al.*, 1988], developed in the Great Plains. Similarly, ANPP values in Inner Mongolia were also higher than ANPP simulated by Century when N deposition parameters from the Great Plains were used (data not shown). This follows results from early Century model validations [*Parton et al.*, 1993], which reported that peak live biomass was underestimated by Century for Asian sites, in contrast to the other 9 sites [*Gilmanov et al.*, 1997]. Previous studies have suggested that this discrepancy may be due to a higher prevalence of C3 plants in this area compared with regions in North American with a similar climate [*Tieszen et al.*, 1999]. However, a hypothesis that fits with the rest of our data is that this higher ANPP is related to higher nitrogen inputs in China. Century simulations *did* produce ANPP values within the range reported by the literature under *elevated* N deposition (0.9 g N m^{-2} yr^{-1}). This is also the Century scenario that, with the inclusion of intensive land use, estimated SOC and SON values nearest to those observed (Table 2). This suggests that in Inner Mongolia a higher ANPP (C input), enabled by increased N deposition, is necessary to produce the greater equilibrium-stage SOC and SON values, which fit observed values when losses due to intensive land use are accounted for as well.

5. Conclusions

[43] The results of this study challenge the generality of relationships between environmental factors and C and N pools in temperate grasslands. SOC and SON data we collected in Inner Mongolia were strongly related to texture and climate, as they are in other similar regions of the world, and data had consistent relationships with values predicted from Great Plains models across this range of sites. However, these models showed strong bias (overestimation of C and underestimation of N) in predicting SOC and SON values in Inner

Mongolia. Elevated N deposition levels were required to simulate accurate predictions for biogeochemical pools in Inner Mongolia, suggesting either that there is unaccounted for nitrogen input in this region, or differences in fundamental nitrogen cycling properties, such as nitrogen use efficiency, compared to other grasslands. In addition, model simulations were only accurate for organic carbon when we included periods of overgrazing to reduce simulated carbon stocks for this area. The relationships of environmental controls with SOC and SON in grasslands are perhaps not as generalizable as many widely used models, and modelers, assume. The possible divergence of these relationships in Inner Mongolia from those used in models for grassland ecosystems could affect our ability to predict regional ecosystem dynamics, and also add uncertainty to global predictions of carbon flux.

[44] **Acknowledgments.** This work was funded by National Science Foundation's East Asia and Pacific Summer Institute (EAPSI) (Award OISE-0812825) and Shortgrass Steppe Long-term Ecological Research (NSF DEB 0823405 and NSF DEB 0217631). We thank Zhou Guangsheng for his support from the Chinese Academy of Sciences Institute of Botany in Beijing, and assistance from members of his laboratory, especially Wang Yuhui, Wang Fengyu, and Bao Fang, and Matthew Wallenstein for suggestions on the manuscript. We would also like to acknowledge two anonymous reviewers for helpful comments on the manuscript.

References

Altesor, A., G. Piñeiro, F. Lezama, R. B. Jackson, M. Sarasola, and J. M. Paruelo (2006), Ecosystem changes associated with grazing in subhumid South American grasslands, *J. Veg. Sci.*, *17*, 323–332, doi:10.1111/j.1654-1103.2006.tb02452.x.

Alvarez, R. (2001), Estimation of carbon losses by cultivation from soils of the Argentine Pampa using the Century Model, *Soil Use Manage.*, *17*, 62–66, doi:10.1111/j.1475-2743.2001.tb00010.x.

Alvarez, R., and R. S. Lavado (1998), Climate, organic matter and clay content relationships in the Pampa and Chaco soils, Argentina, *Geoderma*, *83*, 127–141, doi:10.1016/S0016-7061(97)00141-9.

Anderson, J. M. (1991), The effects of climate change on decomposition processes in grassland and coniferous forests, *Ecol. Appl.*, *1*, 326–347, doi:10.2307/1941761.

Burke, I. C., C. M. Yonker, W. J. Parton, C. V. Cole, K. Flach, and D. S. Schimel (1989), Texture, climate, and cultivation effects on soil organic matter content in U.S. grassland soils, *Soil Sci. Soc. Am. J.*, *53*, 800–805, doi:10.2136/sssaj1989.03615995005300030029x.

Burke, I. C., W. K. Lauenroth, and D. P. Coffin (1995), Soil organic matter recovery in semiarid grasslands: Implications for the conservation reserve program, *Ecol. Appl.*, *5*, 793–801, doi:10.2307/1941987.

Burke, I. C., W. K. Lauenroth, R. Riggle, P. Brannen, B. Madigan, and S. Beard (1999), Spatial variability of soil properties in the shortgrass steppe: The relative importance of topography, grazing, microsite, and plant species in controlling spatial patterns, *Ecosystems*, *2*, 422–438, doi:10.1007/s100219900091.

Burnham, K., and D. Anderson (2002), *Model Selection and Multimodel Inference: A Practical Information-Theoretic Approach*, 2nd ed., Springer, New York.

Cambardella, C. A., and E. T. Elliott (1992), Particulate soil organic-matter changes across a grassland cultivation sequence, *Soil Sci. Soc. Am. J.*, *56*, 777–783, doi:10.2136/sssaj1992.03615995005600030017x.

Chuluun, T., and D. Ojima (2002), Land use change and carbon cycle in and semi-arid lands of east and central Asia, *Sci. China C Life Sci.*, *45*, 48–54.

Epstein, H. E., I. C. Burke, and W. K. Lauenroth (2002), Regional patterns of decomposition and primary production rates in the U.S. Great Plains, *Ecology*, *83*, 320–327.

Eswaran, H., E. Vandenberg, and P. Reich (1993), Organic carbon in soils of the world, *Soil Sci. Soc. Am. J.*, *57*, 192–194, doi:10.2136/sssaj1993.03615995005700010034x.

Giardina, C. P., and M. G. Ryan (2000), Evidence that decomposition rates of organic carbon in mineral soil do not vary with temperature, *Nature*, *404*, 858–861, doi:10.1038/35009076.

Gill, R., I. C. Burke, D. G. Milchunas, and W. K. Lauenroth (1999), Relationship between root biomass and soil organic matter pools in the shortgrass steppe of eastern Colorado, *Ecosystems*, *2*, 226–236, doi:10.1007/s100219900070.

Gilmanov, T. G., W. J. Parton, and D. S. Ojima (1997), Testing the 'Century' ecosystem level model on data sets from eight grassland sites in the former USSR representing a wide climatic/soil gradient, *Ecol. Modell.*, *96*, 191–210, doi:10.1016/S0304-3800(96)00067-1.

Guo, Y. Y., P. Gong, R. Amundson, and Q. Yu (2006), Analysis of factors controlling soil carbon in the conterminous United States, *Soil Sci. Soc. Am. J.*, *70*, 601–612, doi:10.2136/sssaj2005.0163.

Holland, E. A., W. J. Parton, J. K. Detling, and D. L. Coppock (1992), Physiological responses of plant populations to herbivory and their consequences for ecosystem nutrient flow, *Am. Nat.*, *140*, 685–706, doi:10.1086/285435.

Homann, P. S., P. Sollins, H. N. Chappell, and A. G. Stangenberger (1995), Soil organic carbon in a mountainous, forested region: Relation to site characteristics, *Soil Sci. Soc. Am. J.*, *59*, 1468–1475, doi:10.2136/sssaj1995.03615995005900050037x.

Hu, Z. M., J. W. Fan, H. P. Zhong, and G. R. Yu (2007), Spatiotemporal dynamics of aboveground primary productivity along a precipitation gradient in Chinese temperate grassland, *Sci. China Ser. D*, *50*, 754–764.

Hurvich, C. M., and C. L. Tsai (1989), Regression and time-series model selection in small samples, *Biometrika*, *76*, 297–307, doi:10.1093/biomet/76.2.297.

Intergovernmental Panel on Climate Change (2007), *Climate Change 2007. Synthesis Report*, edited by S. Solomon et al., IPCC, Geneva.

Jiang, G. M., X. G. Han, and J. G. Wu (2006), Restoration and management of the inner Mongolia grassland require a sustainable strategy, *Ambio*, *35*, 269–270, doi:10.1579/06-S-158.1.

Kelly, R. H., I. C. Burke, and W. K. Lauenroth (1996), Soil organic matter and nutrient availability responses to reduced plant inputs in shortgrass steppe, *Ecology*, *77*, 2516–2527, doi:10.2307/2265750.

Koch, G. W., P. M. Vitousek, W. L. Steffen, and B. H. Walker (1995), Terrestrial transects for global change research, *Vegetatio*, *121*, 53–65, doi:10.1007/BF00044672.

Leith, H. (Ed.) (1978), *Patterns of Primary Productivity in the Biosphere*, 342 pp., Hutchinson Ross, Stroudsberg, Pa.

Lu, C. Q., and H. Q. Tian (2007), Spatial and temporal patterns of nitrogen deposition in China: Synthesis of observational data, *J. Geophys. Res.*, *112*, D22S05, doi:10.1029/2006JD007990.

Miller, A. J., R. Amundson, I. C. Burke, and C. Yonker (2004), The effect of climate and cultivation on soil organic C and N, *Biogeochemistry*, *67*, 57–72, doi:10.1023/B:BIOG.0000015302.16640.a5.

Nelson, D. W., and L. E. Sommers (1982), Total carbon, organic carbon and organic matter, in *Methods of Soil Analysis*, edited by A. L. Page, pp. 539–579, ASA and SSSA, Madison, Wisc.

Ni, J., and X. S. Zhang (2000), Climate variability, ecological gradient and the Northeast China Transect (NECT), *J. Arid Environ.*, *46*, 313–325, doi:10.1006/jare.2000.0667.

Office for Interdisciplinary Earth Studies (1991), *Arid Ecosystems Interactions: Recommendations for Drylands Research in the Global Change Research Program*, *Rep. 6*, OIES, Boulder, Colo.

Ojima, D. S., W. J. Parton, D. S. Schimel, and C. E. Owensby (1990), Simulated impacts of annual burning on prairie ecosystems, in *Fire in North American Prairies*, edited by S. L. Collins and L. Wallace, pp. 118–132, Oklahoma Press, Norman, Okla.

Parton, W. J., D. S. Schimel, C. V. Cole, and D. S. Ojima (1987), Analysis of factors controlling soil organic matter levels in great plains grasslands, *Soil Sci. Soc. Am. J.*, *51*, 1173–1179, doi:10.2136/sssaj1987.03615995005100050015x.

Parton, W. J., et al. (1993), Observations and modeling of biomass and soil organic-matter dynamics for the grassland biome worldwide, *Global Biogeochem. Cycles*, *7*, 785–809, doi:10.1029/93GB02042.

Paruelo, J. M., E. G. Jobbagy, O. E. Sala, W. K. Lauenroth, and I. C. Burke (1998), Functional and structural convergence of temperate grassland and shrubland ecosystems, *Ecol. Appl.*, *8*, 194–206, doi:10.1890/1051-0761(1998)008[0194:FASCOT]2.0.CO;2.

Paustian, K., et al. (1997), Agricultural soils as a sink to mitigate CO_2 emissions, *Soil Use Manage.*, *13*, 230–244, doi:10.1111/j.1475-2743.1997.tb00594.x.

Percival, H. J., R. L. Parfitt, and N. A. Scott (2000), Factors controlling soil carbon levels in New Zealand grasslands: Is clay content important?, *Soil Sci. Soc. Am. J.*, *64*, 1623–1630, doi:10.2136/sssaj2000.6451623x.

Piñeiro, G., J. M. Paruelo, and M. Oesterheld (2006), Potential long-term impacts of livestock introduction on carbon and nitrogen cycling in grasslands of southern South America, *Global Change Biol.*, *12*, 1267–1284, doi:10.1111/j.1365-2486.2006.01173.x.

Piñeiro, G., S. Perelman, J. P. Guerschman, and J. M. Paruelo (2008), How to evaluate models: Observed vs. predicted or predicted vs. observed?, *Ecol. Modell.*, *216*, 316–322, doi:10.1016/j.ecolmodel.2008.05.006.

Piñeiro, G., J. M. Paruelo, E. G. Jobbagy, R. B. Jackson, and M. Oesterheld (2009), Grazing effects on belowground C and N stocks along a network of cattle exclosures in temperate and subtropical grasslands of South America, *Global Biogeochem. Cycles*, *23*, GB2003, doi:10.1029/2007GB003168.

Plante, A., R. Conant, E. Paul, K. Paustian, and J. Six (2006), Acid hydrolysis of easily dispersed and microaggregate-derived silt- and clay-sized fractions to isolate resistant soil organic matter, *Eur. J. Soil Sci.*, *57*, 456–467, doi:10.1111/j.1365-2389.2006.00792.x.

Rawls, W. J. (1983), Estimating soil bulk density from particle size analysis and organic matter content, *Soil Sci.*, *135*, 123–125, doi:10.1097/00010694-198302000-00007.

Robles, M. D., and I. C. Burke (1998), Soil organic matter recovery on conservation reserve program fields in southeastern Wyoming, *Soil Sci. Soc. Am. J.*, *62*, 725–730, doi:10.2136/sssaj1998.03615995006200030026x.

Sala, O. E., W. J. Parton, L. A. Joyce, and W. K. Lauenroth (1988), Primary production of the central grassland region of the United-States, *Ecology*, *69*, 40–45, doi:10.2307/1943158.

Sandor, J. A., P. L. Gersper, and J. W. Hawley (1986), Soils at prehistoric agricultural terracing sites in New Mexico. 2. Organic matter and bulk density changes, *Soil Sci. Soc. Am. J.*, *50*, 173–177, doi:10.2136/sssaj1986.03615995005000010032x.

Scurlock, J. M. O., and D. O. Hall (1998), The global carbon sink: A grassland perspective, *Global Change Biol.*, *4*, 229–233, doi:10.1046/j.1365-2486.1998.00151.x.

Smith, E. P., and K. A. Rose (1995), Model goodness-of-fit analysis using regression and related techniques, *Ecol. Modell.*, *77*, 49–64, doi:10.1016/0304-3800(93)E0074-D.

Sneath, D. (1998), Ecology: State policy and pasture degradation in inner Asia, *Science*, *281*, 1147–1148, doi:10.1126/science.281.5380.1147.

Steffens, M., A. Kolbl, E. Schork, B. Gschrey, and I. Kogel-Knabner (2011), Distribution of soil organic matter between fractions and aggregate size classes in grazed semiarid steppe soil profiles, *Plant Soil*, *338*, 63–81, doi:10.1007/s11104-010-0594-9.

Theil, H. (1958), *Economic Forecasts and Policy*, pp. 32–35, North Holland, Amsterdam.

Tieszen, L., et al. (1999), Grasslands of Asia and North America: Distribution of soil organic carbon and stable isotopes in grasslands of the Mongolian steppe, in *People and Rangelands Building the Future*, vol. 1, edited by D. Eldridge and D. Freudenberger, pp. 138–139, VI Int. Rangeland Congr., Townsville, Qsld., Australia.

Trumbore, S. E., O. A. Chadwick, and R. Amundson (1996), Rapid exchange between soil carbon and atmospheric carbon dioxide driven by temperature change, *Science*, *272*, 393–396, doi:10.1126/science.272.5260.393.

Wang, S. P., G. S. Zhou, S. H. Gao, and J. P. Guo (2005), Soil organic carbon and labile carbon along a precipitation gradient and their responses to some environmental changes, *Pedosphere*, *15*, 676–680.

Wang, Y. H., G. S. Zhou, and Y. H. Wang (2007), Modeling responses of the meadow steppe dominated by Leymus chinensis to climate change, *Clim. Change*, *82*, 437–452, doi:10.1007/s10584-006-9145-z.

Xiao, X. M., D. Chen, Y. M. Peng, X. Y. Cui, and D. S. Ojima (1996), Observation and modeling of plant biomass of meadow steppe in Tumugi, Xingan league, Inner Mongolia, China, *Vegetatio*, *127*, 191–201, doi:10.1007/BF00044640.

Xiong, Z. Q., J. R. Freney, A. R. Mosier, Z. L. Zhu, Y. Lee, and K. Yagi (2008), Impacts of population growth, changing food preferences and agricultural practices on the nitrogen cycle in East Asia, *Nutr. Cycl. Agroecosyst.*, *80*, 189–198, doi:10.1007/s10705-007-9132-4.

Yu, M., J. E. Ellis, and H. E. Epstein (2004), Regional analysis of climate, primary production, and livestock density in inner Mongolia, *J. Environ. Qual.*, *33*, 1675–1681, doi:10.2134/jeq2004.1675.

Zeng, D. H., L. J. Li, T. J. Fahey, Z. Y. Yu, Z. P. Fan, and F. S. Chen (2010), Effects of nitrogen addition on vegetation and ecosystem carbon in a semi-arid grassland, *Biogeochemistry*, *98*, 185–193, doi:10.1007/s10533-009-9385-x.

Zhang, X., G. Quiong, and Y. Dianan (1997), A gradient analysis and prediction on the Northeast China Transect (NECT) for global change study, *Acta Bot. Sin.*, *39*, 785–799.

Zhou, G., Y. Wang, and S. Wang (2002a), Responses of grassland ecosystems to precipitation and land use along the Northeast China Transect, *J. Veg. Sci.*, *13*, 361–368, doi:10.1111/j.1654-1103.2002.tb02060.x.

Zhou, G. S., Y. H. Wang, Y. L. Jiang, and Z. Z. Xu (2002b), Carbon balance along the Northeast China Transect (NECT-IGBP), *Sci. China C Life Sci.*, *45*, 18–29.

Zhou, Z., O. J. Sun, J. Huang, Y. Gao, and X. Han (2006), Land use affects the relationship between species diversity and productivity at the local scale in a semi-arid steppe ecosystem, *Funct. Ecol.*, *20*, 753–762, doi:10.1111/j.1365-2435.2006.01175.x.

Zhou, Z. Y., O. J. Sun, J. H. Huang, L. H. Li, P. Liu, and X. G. Han (2007), Soil carbon and nitrogen stores and storage potential as affected by land-use in an agro-pastoral ecotone of northern China, *Biogeochemistry*, *82*, 127–138, doi:10.1007/s10533-006-9058-y.

I. C. Burke, Environment and Natural Resources Program, Department of Botany, Department of Renewable Resources, and Program in Ecology, University of Wyoming, Laramie, WY 82071, USA.

S. E. Evans, Graduate Degree Program in Ecology, Colorado State University, Fort Collins, CO 80521, USA. (evanssar@gmail.com)

W. K. Lauenroth, Department of Botany and Program in Ecology, University of Wyoming, Laramie, WY 82071, USA.