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SOM Loss and Soil Quality in the Clear Creek, IA

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The Clear Creek, IA Experimental Watershed (CCEW), which drains to the Iowa River, experiences severe surface erosion due to a combination of high slopes, erodible soils, and extensive agriculture. Concurrent with soil loss is the removal of Soil Organic Matter (SOM). High values of SOM have been related to soil quality; therefore, excessive SOM loss corresponds to degrading soil health. Soil quality assessments are important tools for evaluating management practices in agricultural systems; however, it is difficult to measure soil quality directly at the watershed scale because it varies with a number of site-specific soil characteristics. The coupling of soil surveys with GIS and Non-Point Source computer simulation models will effectively forecast the impacts of ever-changing management practices on soil quality at the watershed scale in less time. NPS models can be extended to evaluate the movement of additional particle-bound constituents like SOM, by incorporating erosion rates and enrichment ratios. The ANNualized AGricultural Non-Point Source pollution modeling system (AnnAGNPS) was used to evaluate upland erosion, enrichment ratios, and SOM loss at the watershed scale in the headwaters of the CCEW using current crop rotations. Gross erosion rates averaged 7.73 MT/ha/yr for individual cells within the watershed. In addition, enrichment ratios, which were determined using gross and net erosion values from AnnAGNPS, were coupled with an organic matter coverage map of the watershed to determine an SOM loss of 0.41 MT/ha/yr, which was similar to the loss rates determined by AnnAGNPS (0.29 MT/ha/yr). To understand the state of soil health in this watershed, the NRCS Soil Conditioning Index (SCI) was determined for the watershed. The average SCI for the watershed was 0.38, which suggests improving soil health conditions. This improvement is most probably due to conservation practices like reduced tillage.

INDEX DESCRIPTORS: Soil Quality, SOM, Erosion.

Soil aggregates are comprised of individual soil grains conjoined by organic matter. These aggregates can be disassociated by the erosional forces of both raindrop impact and runoff. As overland flow progresses on a hillslope, it will preferentially entrain and transport smaller, lighter particles of the disassociated aggregates, like organic matter (Lal et al. 2004; Polyakov and Lal 2004). The fraction of the Soil Organic Matter (SOM) displaced by the flow, i.e., eroded, is described through an Enrichment Ratio (ER), which is expressed as the proportion of SOM in transported sediment, SOM_{ts}, to that of SOM in uneroded soil, SOM_{ur} (Teixeira and Misra 2005).

$$ER = \frac{SOM_{ts}}{SOM_{ur}} \tag{1}$$

An ER > 1 indicates that the eroded sediment is enriched in SOM relative to the uneroded soil, whereas an ER <1 denotes the opposite. Enrichment ratios largely range between 1.2 and 5.6 for most soils under varying agricultural land uses (Jacinthe et al. 2004), which suggests the likelihood of high rates of SOM loss during runoff events.

ERs facilitate estimation of cumulative or single event SOM loss associated with erosion in agricultural fields, when they are applied to soil loss rates (MT/ha/yr) and near surface (< 5 cm) SOM concentrations (Starr et al. 2000). The following equation expresses the relationship between these variables:

$$SOM\ loss = soil\ loss \times SOM\ concentration \times ER \tag{2}$$

Erosion, which has been enhanced in agricultural systems due to tillage (Williams 1981; Lal 1984), can remove large quantities of stable SOM due to its relatively low density and the bonds between soil and SOM (Lal 2003; Moorman et al. 2004; Conant et al. 2007; US Department of Agriculture – Natural Resources Conservation Service, USDA-NRCS 2007). In fact, tillage-induced erosion has removed approximately one-half of the topsoil (i.e., the highly organic O and A horizons) in Iowa since settlement (Pimental et al. 1995). This substantial loss of SOM has resulted in an overall decrease in soil quality (Williams 1981; USDA-NRCS 1998; Lal et al. 2004; Karlen et al. 2008).

SOM content is integral to soil quality (Andrews et al. 2002; Duiker and Myers 2005) because it is related to several other soil parameters (Tisdale and Oades 1982; USDA-NRCS 2003; Cambardella et al. 2004; Chivenge et al. 2007). For example, SOM stabilizes soil structure by acting as a binding agent for soil aggregates (Tisdale and Oades 1982; Oades and Waters 1991). In addition, SOM can hold orders of magnitude more water and nutrients than inorganic soil minerals, as well as promote soil biota activity (USDA-NRCS 2003; Righetti and Lucarelli 2007).

Despite the importance of SOM to soil quality, multiple soil properties are used to determine overall soil quality. Soil quality

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combines multiple, site specific, biogeochemical properties of the soil into a comprehensive evaluation tool (Karlen et al. 1997). A possible minimum data set identified in the literature to assess soil quality includes SOM, pH, electrical conductivity, and phosphorus (Doran and Parkin. 1996; Andrews et al. 2002). In addition, soil structure, infiltration, water holding capacity, and soil biota are important soil quality parameters (USDA-NRCS 2001). The relevant parameters at a site are then incorporated into indices to assess soil quality (Andrews et al. 2002, 2003).

Soil quality assessments have become important tools for evaluating management practices in agricultural systems (e.g., Larson and Pierce 1991; Doran and Parkin 1994, 1996; Karlen and Stott 1994; Karlen et al. 1996, 1998, 2008; Andrews et al. 2002, 2003, 2004; Cambardella et al. 2004). However, it is difficult to measure soil quality directly at the watershed scale (>10 km^2) because it is a function of several site-specific soil characteristics (USDA-NRCS 2001) and reflects a dynamic response of the soil to applied anthropogenic alterations (Seybold et al. 1998; USDA-NRCS 2001).

Conducting a soil quality assessment at the watershed scale generally follows two approaches: extensive soil surveys (Cambardella et al. 1994; Brejda et al. 2000; Karlen et al. 2008) or paired studies between systems of contrasting practices (Cambardella et al. 2004; Moorman et al. 2004). These studies provide valuable information but involve extensive field and laboratory work, as well as take considerable time because important parameters (like SOM) respond slowly to management changes (Starr et al. 2000; Kravchenko et al. 2006).

The coupling of soil surveys with Geographic Information Systems (GIS) and currently available computer simulation Non-Point Source (NPS) models would greatly facilitate soil quality assessments by effectively forecasting the impacts of ever-changing management practices on soil quality at the watershed scale in less time. With increasing technological development to facilitate the use of large-scale, spatially variable data (i.e., GIS), erosion and NPS pollution models are becoming popular tools for landscape management (Kirnak 2002). NPS models, which describe in-field hydrological and biogeochemical processes, including flow and sediment/ contaminant mobilization, as well as in-stream transport, can be extended to evaluate the movement of additional particle-bound constituents like SOM, by incorporating erosion rates and enrichment ratios.

In the present study, the ANNualized AGricultural Non-Point Source pollution model (AnnAGNPS) was used to evaluate gross and net erosion for estimating ERs in the headwaters of the Clear Creek, IA Experimental Watershed (CCEW). AnnAGNPS has been extensively used to simulate sediment/nutrient transport at the watershed scale (Bingner and Theurer 2007). The soil loss and ERs were coupled with a SSURGO-based, organic matter coverage layer for the watershed to determine SOM loss. The computed SOM loss using equation 2 was compared to the organic matter load determined by AnnAGNPS using spatial heterogeneities, which reflect the current state of soils, land use, and management practices in the watershed. The model was used to determine the expected annual average soil and SOM loss for the watershed should current rotations continue. The average annual SOM loss rates were then compared to the Soil Conditioning Index (SCI) to assess the state of soil health in this watershed as a proxy for evaluating soil quality. The SCI is a reflection of soil health with tespect to SOM (i.e., whether it is improving or degrading). This study will show the effectiveness of coupling GIS and NPS models with soil surveys to assess soil quality.

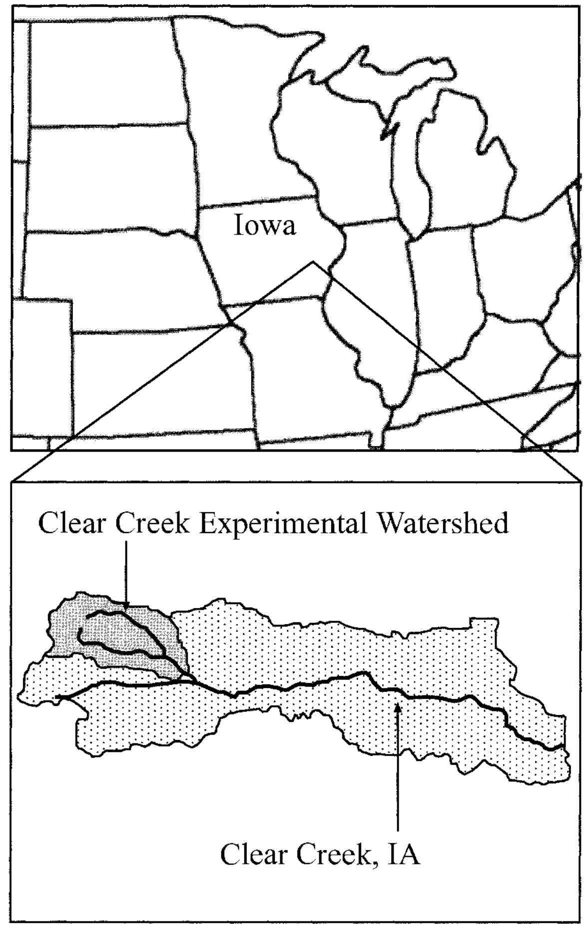

Fig. 1. The Clear Creek, IA watershed with the Clear Creek Experimental Watershed.

METHODS

Study Site

The Clear Creek, IA watershed (HUC-10: 0708020904) spans approximately 270 km^2 in east-central Iowa (Fig. 1). Anthropogenic activities, including both row-crop agriculture and urbanization, strongly influence flow and sediment processes in the watershed.

Since settlement, the majority of the watershed has been converted from a prairie and forested area to row-crop agriculture and pastures. Historically, corn was the dominant row crop, either planted continuously or as part of a corn-oat-meadow rotation. In the 1970s, soybeans were introduced to the watershed and their spatial extent has grown steadily. Currently, a corn-soybean rotation is dominant in the watershed (~80%) with the two crops in roughly equal proportions throughout the watershed. Pastures and hay fields comprise another 10% of the watershed with urban areas, roads, and waterways in the remaining area of the watershed. Although agriculture is still prominent in the watershed, the area is experiencing a marked increase in urbanization (U.S. Census Bureau 2007).

Clear Creek flows approximately 40 river km with an elevation drop of approximately 90 m from west to east into the Iowa River. The river has experienced widespread channelization and

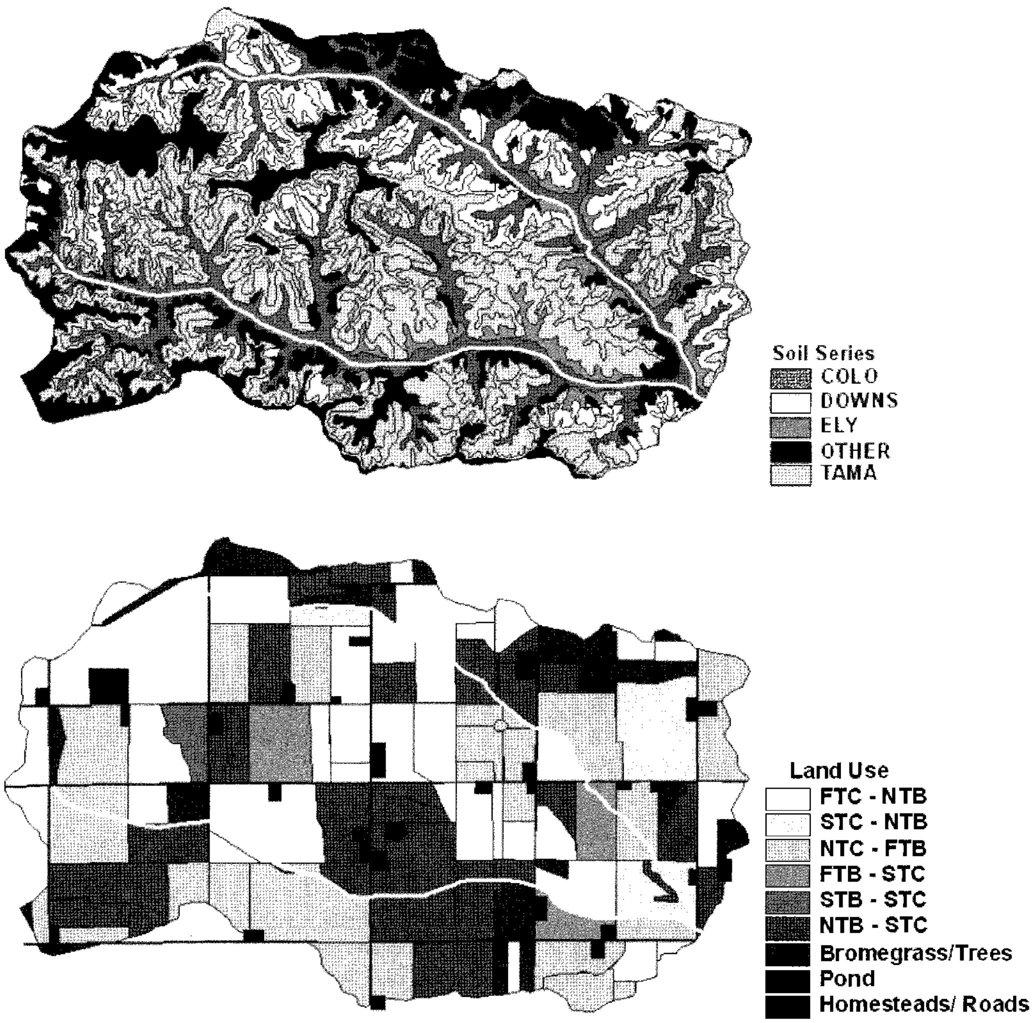

Fig. 2. (a) Soils in the CCEW. (b) Land use in the CCEW. FTB is Fall-Till Bean, FTC is Fall-Till Corn, STC is Spring-Till Corn, STB is Spring-Till Bean, NTC is No-Till Corn, NTB is No-Till Bean.

modification through the construction of drainage systems (Rayburn and Schulte 2006). The United States Geological Survey (USGS) operates stream gages near Oxford, IA (#05454220) and Coralville, IA (#05454300), which is near the outlet. Discharges at these sites are highest between February and June (Averages - Oxford: 1.50 ± 0.20 m^3/s; Coralville: 2.72 ± 0.17 m^3/s).

Due to the mid-continental location of Iowa, its climate is characterized by hot summers, cold winters, and wet springs (Ruhe 1956). Summer months are influenced by warm, humid air masses from the Gulf of Mexico. Dry Canadian air masses dominate the winter months. Average daily temperature is about 10° C ranging from an average July maximum of 29° C to an average January minimum of −13° C. The growing season lasts about 180 days in southeast Iowa. Average annual precipitation is approximately 889 mm/yr with convective thunderstorms prominent in the summer and snowfall in the winter, which averages 762 mm annually. This data was compiled from National Weather Service (http://www.crh.noaa.gov/dvn/) and Iowa Mesonet websites (http://mesonet.agron.iastate.edu/).

This study focused in the predominantly rural headwaters of Clear Creek. This area is the focal point of the 26-km^2 Clear Creek Experimental Watershed (CCEW; Fig. 1) with an infrastructure maintained by IIHR – Hydroscience & Engineering at the University of Iowa to monitor rainfall, streamflow, suspended sediment concentration, and other water quality parameters.

In the CCEW, there are two main sub-basins, both of which contain first order tributaries. Each tributary is approximately 6 river km long during the wet season (March–June). The outlet of the CCEW is approximately 30-river km above the Iowa River confluence.

The dominant surface soil texture within the CCEW is silty clay loam. There are four main soil series in the watershed comprising approximately 80% of the total area (Fig. 2a). The uplands in the southern sub-basin are mostly comprised of Tama series soils, while the Downs series soils are found predominantly in the northern sub-basin uplands. Both soil series consist of well-drained soils derived from loess (Highland and Dideriksen 1967). Floodplains in the CCEW are alluvium-derived soils comprised of mostly the Ely and Colo series (Highland and

Dideriksen 1967). Soils in the Ely series are somewhat poorly drained, while soils in the Colo series are poorly drained (Highland and Dideriksen 1967).

An extensive database regarding current and historical management practices exists for the CCEW. These data sets were obtained from local USDA-NRCS representatives (James Martin of the Johnson County, IA Soil and Water Conservation District and Steven Johnston of the Iowa County, IA Soil and Water Conservation District), in the form of a spatiotemporally explicit land use algorithm that allowed the production of data about crop-rotations and associated land management activities within the CCEW. The captured data included the time, location, amount and type of tillage practices performed, seeds planted, and crops harvested. The current land uses in the watershed include row crop agriculture (namely corn and soybean), Conservation Reserve Program (CRP) lands, pastures, and hay farming (Fig 2b). The corn-soybean rotations in the watershed follow six unique sets of management strategies. The dominant tillage practices for both crops (Corn, C and Soybean, B) are no-till (NT), reduced spring tillage (ST), deep fall tillage with secondary tillage in the spring (FT). To further elaborate, FTC-NTB, NTB-STC and NTC-FTB stand for Fall Till Corn-No Till Bean, No Till Bean-Spring Till Corn and No Till Corn-Fall Till Bean crop rotations, respectively. These three rotations together with prairie bromegrass (*Bromus wildenowii*) for Conservation Reserve Program (CRP), hay, and pasture fields comprised approximately 90% of the total acreage.

Fields in the CCEW have some of the highest erosion rates in Iowa. The elevated rates within the CCEW are mainly due a combination of swelling (i.e., smectite-rich) and highly erodible soils with steep slopes (up to 10%) and intensive agriculture (Highland and Dideriksen 1967; Piest et al. 1975; Cruse et al. 2006, Steckly 2007).

Model Description

AnnAGNPS is the annualized non-point source pollutant loading component of the suite of computer simulation models, known as AGNPS, which was jointly developed by the USDA - Agricultural Research Service (ARS) and - NRCS (Cronshey and Theurer 1998; Bingner and Theurer 2001, 2007). The deterministic, distributed-parameter model predicts average annual fluxes of water, sediment, and nutrients in watersheds up to 1650 km^2.

AnnAGNPS discretizes the watershed into individual grid cells (0.16–1 km^2; Young et al. 1989). Each cell can contain spatially distributed characteristics, but these cells are linked by reaches to form sub-watersheds. Inputs to AnnAGNPS include different categories of data with topography, land use, crop characteristics, management practices, soil properties, channel descriptions, and climate being most important.

Fluxes of water, sediment, and nutrients are evaluated on daily time steps in AnnAGNPS. Runoff is calculated via the Soil Conservation Service (SCS) Runoff Curve Number (RCN) method (USDA-SCS 1986). Gross upland erosion, which is considered as the total in-field mobilization of sediment, is determined using the Revised Universal Soil Loss Equation (RUSLE). RUSLE incorporates the erosive strength of the rainfall, soil erodibility, slope length and steepness, crop cover, and management practices (Renard et al. 1996). Deposition within the field is determined using HUSLE, or the Hydro-geomorphic Universal Soil Loss Equation (Theurer and Clark 1991). The difference between the gross erosion and deposition within a field, or net erosion, determines the amount of sediment exported from the field. Enrichment ratios can be calculated using the relationship of net erosion to gross erosion.

Water, sediment, and nutrients are routed through cells and reaches to a singular system outlet, where total exported loads are determined for the watershed. Routing through the system is controlled by transport capacity, which is evaluated using the equation in Bagnold (1966). The relationship between the total load exported from the watershed and the gross erosion within the watershed provides Sediment Delivery Ratios (SDRs) for the system.

Model Data

As mentioned above, important inputs into AnnAGNPS include topography, soil properties, land use and weather. Topography of the CCEW was established using a GIS interface and 30-m resolution Digital Elevation Models (DEMs) from the USGS National Elevation Dataset (NED). The elevation layer was overlain with a soil layer that contained data from both the Iowa Soils Properties And Interpretations Database (ISPAID) and Soil SURvey Geographic (SSURGO) Database (Fig. 2a). A detailed land use coverage map was developed in conjunction with the Iowa County Natural Resources Conservation Service (NRCS) office (Fig. 2b), which incorporates the various management scenarios utilized within the watershed.

Two different sources were considered for preparing the model weather input data. For the calibration simulation, 25 years of weather input data were developed by using the stochastic weather generator, CLIGEN (Nicks 1985). CLIGEN incorporated monthly averaged values for Williamsburg, IA, which is approximately 5 miles from the watershed. The 25-year simulation period was chosen because erosion rates and sediment delivery ratios in the CCEW were shown to approach equilibrium within this period (e.g., Papanicolaou and Abaci 2008). In addition, management practices within a field are often adjusted on a frequency less than a 25-year period. Most numerical NPS models, when performing continuous simulations, require an initialization period to stabilize the model inputs (De Jong Van Lier et al. 2005; Cruse et al. 2006; Papanicolaou and Abaci 2008). For this study the initialization period was 10 years.

For the validation simulation, observed precipitation data was used. NCEP (National Centers for Environmental Prediction) Stage IV precipitation data were available for a period of 8 years (2000–2007) through the Iowa MESONET. The NCEP Stage IV data comprise hourly precipitation measurements that incorporate both rain gage data and WSR-88D (Weather Surveillance Radar 1988 Doppler Version) data.

Finally, in order to groundtruth the simulation results, a series of detailed field measurements were conducted from May to November 2007 and June to October 2008. Water level, flow velocity, and sediment concentration were measured at the outlet of the CCEW. By using the collected data, relationships between water stage, water discharge and sediment load were developed for the outlet of the study site. Further, the measured stage values at the CCEW outlet (during May–November 2007) were correlated with long-term data from a downstream USGS streamgage (05454220 Clear Creek near Oxford) by following an approach that is similar to traditional regional analysis techniques (Riggs, 1973; Adamowski, 2000; Gordon et al., 2004; Arbeláez and Castro, 2007). The developed correlation allowed the transfer of long-term daily stage data for the period 2000–2007 from the Oxford streamgage to the CCEW outlet. By using the long-term stage data and the flow/sediment relationships, yearly outlet fluxes were calculated. The details of the field work at the CCEW outlet can be found in Abaci and Papanicolaou (In Press).

Calibration

In order to calibrate the model for the study site, continuous simulations were performed for a period of 25 years. Calibration of AnnAGNPS for the CCEW was conducted by adjusting specific sensitive parameters within physical ranges, which were determined either via implicit/explicit measurements or based on values reported in the literature (e.g., Santhi et al. 2001). With spatially distributed, physical models, the calibration of too many variables can result in overparameterization. Calibration of the most sensitive parameters helped limit this overparameterization. A set of governing factors, which represent the physical forcings, pedologic characteristics, and management practices within the watershed, was selected here to provide indirect accounting of the entire range of processes (Buol et al. 1997). According to the literature (e.g., Fox and Papanicolaou 2007; Bingner and Theurer 2001, 2007), the runoff curve number (RCN) was the only sensitive parameter regarding runoff. Along the same lines, erosion rates were most sensitive to the erodibility coefficient that reflects aggregate stability as affected by tillage practices and also the rate of erosion. In AnnAGNPS, which uses RUSLE to calculate upland erosion, the K term dictates soil erodibility (Renard et al. 1996). The effects of the management practices (especially those related to tillage and residue cover) in RUSLE are expressed through the C and P coefficients (Renard et al. 1996) also significantly affected erosion (Baginska and Milne-Home 2003). Sediment transport, as seen through the total sediment load at the outlet, was influenced by the reach Manning's n term. The predetermined ranges of these factors were evaluated to determine the variables, which were most influential at controlling runoff, erosion/deposition, and sediment transport (Table 1).

Because of the limited number of influential parameters determined during the sensitivity analysis, the model was manually calibrated by adjusting one parameter and keeping the remaining parameters constant (Renard et al. 1996). Calibration continued until the annual average outlet discharge, sediment load, and SDR for a 25-year period matched the long-term annual averages for the CCEW (Abaci and Papanicolaou In Press). The calibration procedure began with flow (i.e., the driving mechanism for upland erosion and sediment transport), and continued with the soil and management practices, ending with the reach characteristics (Santhi et al. 2001).

Runoff Curve Number (RCN)

AnnAGNPS has the capability to adjust the RCN based on daily soil moisture and crop cycle (Licciardello et al. 2007). Therefore, the starting RCN values were calibrated (Table 1).

Statistically defendable estimates of the RCN were provided for calibration of the AnnAGNPS simulations from documented literature values (USDA-SCS 1986), as well as in-situ measurements within Iowan agricultural fields using automated rainfall simulators (Elhakeem and Papanicolaou In Press). The measurements were designed to account for rainfall intensity, soil type, antecedent moisture condition, management practice, crop rotation, and residue cover. The RCN equations for the different hydrologic soil groups in Iowa developed from Elhakeem and Papanicolaou (In Press) were re-evaluated in the CCEW at different experimental plots representing the three dominant crop rotations.

Table 1. Parameters used for calibration of AnnAGNPS.

Sensitive Parameters for AnnAGNPS Calibration

Runoff Curve Number

Practice	Hydrologic Soil Group A	B	C	D
Fallow Bare	77	86	91	94
Fallow Crop Residue	76	85	90	93
Legume Straight Row	72	81	88	91
Row Crop Straight Row	66	77	85	89
Pasture w/ Grazing	68	79	86	89
Small Grain Straight Row	65	76	84	88
Meadow	30	58	71	78

RUSLE K-parameter

Soil	K
Tama	0.30
Downs	0.30
Colo/Ely	0.16
Colo	0.15
Colo Overwash	0.30
Ely	0.20

Manning's n

Reach	n
Grassed waterways	0.20
CCEW channel	0.12
Outlet	0.04

Elhakeem and Papanicolaou (In Press) confirmed the findings of previous studies, which showed the sensitivity of the RCN to residue cover and antecedent soil moisture conditions. The range of RCNs from the rainfall simulator experiments in summer months agreed well (deviation less than 6%) with ranges reported in the literature. However, the measured RCNs in the fall were generally less than the literature reported values (deviation of about 40%) and considered in the calibration of the model to reflect the seasonal effects on our continuous simulations.

Erosion – RUSLE K, C, P

The initial K values selected for the calibration included data sets from ISPAID and SSURGO, as well as values calculated using the nomograph in Wischmeier and Smith (1978). The AnnAGNPS option to adjust K over the year, based on climate conditions, was used for this study. The optimal K values selected at the end of the calibration process are presented in Table 1. The cover – management factor was developed using detailed management scenarios for each field in the CCEW with help from local NRCS agents. Due to the level of detail for the management scenarios, the parameters affecting the C factor were not altered during calibration. The C values in AnnAGNPS also adjust over the course of the year based on the scheduled management practices. The sub-P factor that deals with the reduction of erosion due to subsurface drainage was adjusted in the AnnAGNPS simulations during calibration. In the literature

(Renard et al. 1996), there is little scientific evidence to determine this value, but a value of 0.6 was suggested. In this study, a value of 0.5 was used.

Sediment Loads and Sediment Delivery Ratio – Manning's n

The sediment load at the outlet of the watershed was adjusted without changing the upland gross and net erosion by altering the reach Manning's n term (Table 1). This value affects stream velocities thereby controlling the transport capacity. By altering the load and not the upland erosion, the SDRs were calibrated to match other values.

Calibration Values

The AnnAGNPS-predicted average annual gross erosion rate for the CCEW was 7.73 MT/ha/yr with erosion rates for the individual cells ranging from 0.01 to 27.0 MT/ha/yr. The NRCS - National Resource Inventory 2003 report ranked Iowa second in the country for water erosion on cropland with 11.2 ± 0.4 MT/ha/yr (USDA-NRCS 2007). Other reported statewide erosion rates in Iowa were between 11 and 23 MT/ha/yr (Manies et al. 2000; Wilson et al. 2003; Fornes et al. 2005; Gilroy 2006).

The measured annual discharge and sediment load at the CCEW outlet were 294 mm/yr (s.d. = 48 mm/yr) and 2.03 MT/ha/yr (s.d. = 0.77 MT/ha/yr), respectively (Abaci and Papanicolaou In Press), while the range of SDRs for a watershed of this size as determined by Roehl (1962) was approximately 0.09 to 0.41. The AnnAGNPS results after calibration were 260 mm/yr, 2.25 MT/ha/yr, 0.29 for the discharge, sediment load, and SDR, respectively. The relative errors between the simulated and measured values for average annual discharge and sediment loads were 11% and 10%, respectively. The simulated discharge and sediment loads, as well as the SDR, fell within the errors of the measured values. Further the average annual runoff to rainfall ratio for the study site was found to be 0.33, which agrees well with other studies conducted in Iowa watersheds (e.g., Piest et al. 1975; Cruse et al. 2006; Papanicolaou and Abaci 2008; Abaci and Papanicolaou In Press).

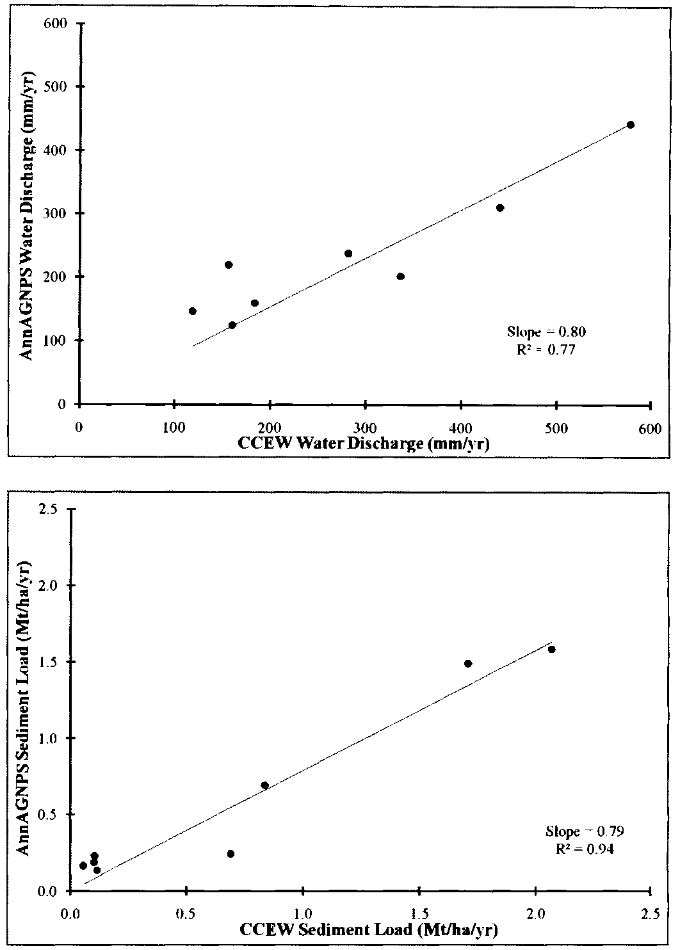

Fig. 3. Regression analysis of measured flow discharge (a) and sediment load (b) at the outlet of the CCEW and results simulated using AnnAGNPS.

Validation

After calibration, another period (2000–2007) was simulated for validation purposes without further adjusting the sensitive parameters (e.g., Santhi et al. 2001; Suir 2002; Saleh and Du 2004; Benaman et al. 2005; White and Chaubey 2005).

Individual annual discharges and sediment loads for the CCEW outlet during the 2000 to 2007 period were compared to simulated discharges and loads using observed climate data for this period. A series of statistical tests were conducted to evaluate the relationships between the measured discharges and loads and the simulated values from AnnAGNPS. The chosen tests (Krause et al. 2005) included the following: (1) Coefficient of Determination for the fit between the measured and simulated results, (2) Nash-Sutcliffe Efficiency, (3) Index of Agreement, (4) Root Mean Squared Error, and (5) t-tests between the mean of the measured values and the simulated values.

Values for the Coefficient of Determination (R^2), which approach 1, suggest good agreement between the measured and simulated values. For Nash Sutcliffe Efficiency (NSE) terms, values range between one and negative infinity with positive values suggesting good agreement and values less than zero suggesting that the mean of the measured data could predict the results better than the model (Krause et al. 2005). The Index of Agreement (IoA) has an acceptable range of values between 0 and 1 with 1 suggesting a perfect fit to the model (Krause et al. 2005). Regarding the Root Mean Squared Error (RMSE), lower values suggest better fits to the model. A p-value less than 0.05 for the Student's t-tests suggests that the means of the measured and simulated values were significantly different.

There was a strong correlation between the measured and simulated values for both flow and sediment loads (Fig. 3), which suggested the model was adequately calibrated. The additional goodness-of-fit tests provided further support for the successful calibration of the model (Table 2).

Additional verification of the model was conducted using the SDRs for subwatersheds of increasing area within the CCEW. These SDRs for the CCEW were compared to (1) SDRs from WEPP (Water Erosion Prediction Project) model simulations in the same watershed (Abaci and Papanicolaou In Press) and (2) SDRs from Roehl (1962). Previous studies have used SDRs as a form of validation (Perrone and Madramootoo 1999; Papanicolaou and Abaci 2008).

The SDRs from this study fall in line with these other values suggesting that the modeling of the relationship between in-field erosion and export of sediment was in agreement (Fig. 4; Roehl 1962). Previous studies have shown that SDRs decreased with

Table 2. Goodness of fit results for measured flow discharges / sediment loads and simulated values.

Test	Abbreviation	Water Discharge (mm/yr)	Sediment Load (MT/ha/yr)
Regression	R^2	0.84	0.94
Nash-Sutcliffe Efficiency	NSE	0.67	0.88
Index of Agreement	IoA	0.88	0.96
Root Mean Square Error	RMSE	87.1	0.58
t-test	tt	0.23	0.41

increasing drainage basin areas (Roehl 1962); however, for smaller systems (<0.1 km^2), there appeared to be no trend (Roehl 1962; Lu et al. 2004). This non-linearity in smaller watersheds exists because storm durations are long relative to residence times of sediment moving through the system; therefore, the majority of the sediment moves through the stream channel (Lu et al. 2004). In larger systems, only a fraction of eroded sediment moved through the outlets because the storm duration was usually less than the residence time of sediment in the system.

Due to this limited lag time between erosion within the field and export of sediment from this system, channel effects are minor in the CCEW. Thus, the simulated water and sediment loads at the outlet of the CCEW can be compared to measured values to help verify the model.

Organic Matter – Field

The soils in the CCEW are highly organic (Fig. 5), with the organic matter content in the different soils ranging from 2 to 6% in the surface layer (Steckly et al. 2007). The areas along the stream channel have the highest concentrations of organic matter. These areas contain predominantly the Colo and Ely soil series (3 to 6% SOM; Steckly et al. 2007). The upland soils, which contain mostly the Tama and Downs series, are less organic than the floodplain soils (2 to 3% SOM; Steckly et al. 2007). SOM measurements from soil samples collected in a test field within the CCEW were in agreement with these documented patterns in organic matter content between the upland and the floodplain soils (Highland and Dideriksen 1967). Upland concentrations of organic matter (average = 2196 ± 215 g/m^2; 3.4 ± 0.3%) were less than concentrations on the floodplain (average = 2329 ± 180 g/m^2; 3.7 ± 0.3%).

Higher concentrations of organic matter in the floodplain soils can be explained by both deposition and drainage. Eroded organic matter from the uplands has been deposited on the floodplain before leaving the field. Stallard (1998) estimated that more than 70% of eroded topsoil is stored in adjacent depositional zones. Moreover, decomposition rates of organic matter have been reduced on the poorly drained floodplains due to increased wetness and decreased aeration inhibiting microbial activity (Stallard 1998; Smith et al. 2001; McCarty and Ritchie 2002; Berhe et al. 2007). Ely and Colo soils are considered somewhat poorly and poorly drained, respectively (Highland and Dideriksen 1967).

The upland soil series in the CCEW, Tama and Downs, are loess-derived and considered well drained (Highland and Dideriksen 1967). Thus, available water content was not a

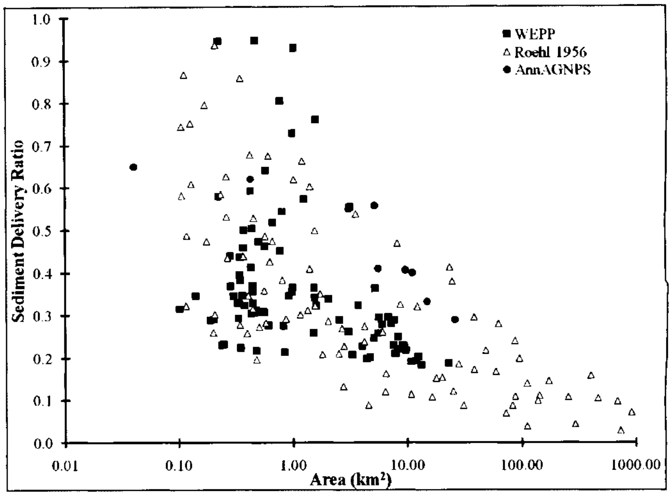

Fig. 4. Comparison of SDR for subwatersheds of increasing areas in the CCEW and other Iowa Sites. The black circles represent the AnnAGNPS results used in this study. The green squares are values from WEPP simulations in the CCEW (Abaci and Papanicolaou unpubl.). Yellow triangles are from Roehl (1956).

controlling factor on organic matter decomposition. However, small-scale variations in erosion more likely affected distributions of organic matter in these areas (Papanicolaou et al. 2008).

Soil Conditioning Index (SCI)

The SCI reflects the ability of a soil, through the proxy of SOM, to respond to external influences, namely erosion and management impacts (USDA-NRCS 2002). SOM strongly reflects the soil condition because it relates to several soil characteristics, including aggregate stability and water holding capacity (USDA-NRCS 2002). A positive SCI suggests fields have an improving soil condition, while conversely a negative SCI suggests a degrading condition.

The SCI accounts for organic matter as residue quantity, management practices, and erosion with the following equation:

$$0.4 * OM + 0.4 * MP + 0.2 * E = SCI \qquad (3)$$

where OM represents the amount of organic matter returned to the soil, MP represents the combined effects of the management practices that reduce SOM through enhancing decomposition, and E is the average annual erosion rate.

The organic matter returned to the soil is the sum of the crop residue, dead root material, and external applications such as manure. Residue quantity is calculated in the SCI using the crop yield, the weight of the yield unit, and the residue to harvest ratio. The additions of the root mass are determined using a multiplicative factor based on the crop. The total organic matter returned to the soil is compared to a "suggested maintenance amount", which is the amount of residue needed for the field to sustain its current amount of SOM under a specified set of management practices and erosion rates determined during the original SCI experiments (USDA-NRCS 2002). These values are adjusted based on climate and normalized to account for changes in residue-specific decomposition rates.

The effects of individual management practices are determined in the SCI using a Soil Disturbance Rating, which is quantified based on the extent of the following five actions: inversion,

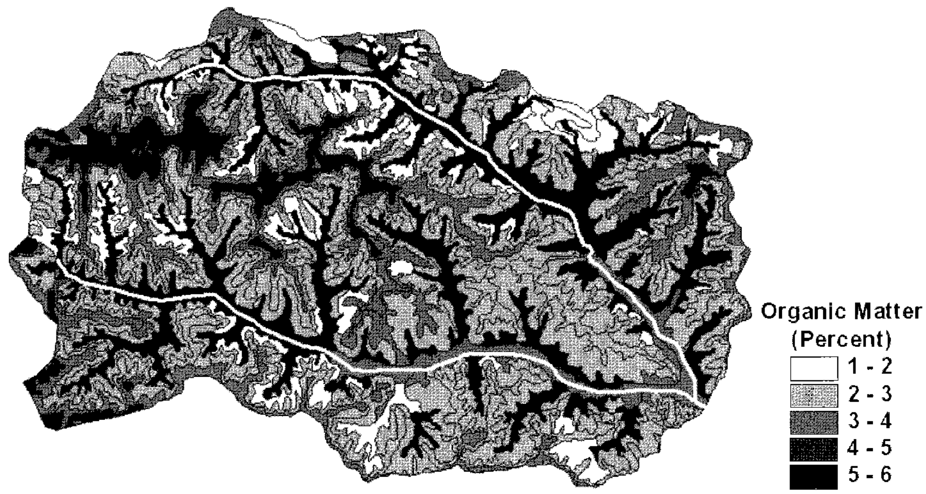

Fig. 5. Percent organic matter in the soil for the CCEW.

shattering, mixing, aeration, lifting, and compaction. For example, low impact practices, like most planting and harvesting, have low Soil Disturbance Rating (<5), while moldboard plows and rototillers have high impacts (>25).

For the erosion term, the SCI only considers removal and sorting due to sheet and rill erosion within the study site. The erosion rate of the sampled field is compared to the erosion rates of the original SCI test fields (~9 MT/ha/yr).

RESULTS AND DISCUSSION

Erosion

Having a detailed model (i.e., including all soil types and land uses in Fig. 2) allowed more accurate replication of the study site conditions viz. outlet fluxes and the spatial distribution of soil loss within the study site (Fig. 6). However, for explanation of the results, only the major soil series and land uses were considered.

Selective grouping of the soil series based on prevalence and location in the watershed helped identify significant trends (Table 3). Colo and Ely series soils were grouped because they were dominant at the toe slope/ floodplain. Tama and Downs series soils were grouped because they were dominant in the uplands.

Upland soils, which consist of Tama and Downs series soils, had significantly higher erosion rates (for all land uses) when compared to floodplain soils, which contain the Colo-Ely soil series. The finding of uplands having higher erosion rates than floodplains was expected due to higher slopes and higher K values. Floodplains tend to have slopes less than 5%, while upland slopes tend to range up to 10%. Soils of the Tama and Downs soil series had an average K value of 0.30, while soils in the Colo-Ely series had average K values around 0.20 (Table 1).

Characterization of the management and conservation practices in agricultural fields have been shown to be more influential to overall long-term erosion rates than rainfall erosivity, soil erodibility, and the length-slope parameters (Baginska and Milne-Home 2003; Abaci and Papanicolaou In Press). Evaluating erosion rates in the CCEW based on land uses and management practices (Table 3) suggested that the type and intensity of tillage practices during a rotation were influential. The relationship between tillage and erosion has been well established (e.g., Lal 1984; Montgomery 2007). Out of the major crop rotations, the

Fig. 6. Gross upland erosion for the CCEW.

Table 3. Soil and SOM loss predictions from AnnAGNPS for the dominant soil series and land uses in the CCEW.

Soil	Land Use	Erosion (Mt/ha/yr)	SOM Loss (Mt/ha/yr)	Area (ha)
Colo-Ely	FTC-NTB	6.20	0.39	78.08
	NTB-STC	3.48	0.22	94.15
	NTC-FTB	3.15	0.20	92.4
Downs-Tama	FTC-NTB	12.5	0.64	590.3
	NTB-STC	7.27	0.37	459.6
	NTC-FTB	5.98	0.29	488.8

Table 4. Goodness of fit results for SOM load rates calculated using AnnAGNPS and Equation 2, which uses ERs.

Test	Abbreviation	SOM Load (MT/ha/yr)
Regression	R^2	0.91
Nash-Sutcliffe Efficiency	NSE	0.65
Index of Agreement	IoA	0.88
Root Mean Square Error	RMSE	0.12
t-test	tt	<0.05

FTC-NTB rotation had the highest erosion rate for both soil groupings (Table 3). The FTC-NTB rotation utilized multiple tillage events in fall and spring of a single rotation year, which contributed to the higher erosion rates.

Enrichment Ratios (ERs)

Outputs from AnnAGNPS provide for the determination of ERs for SOM in each cell, which were determined as the ratio of gross to net erosion values. In this study, ERs ranged between 1.00 and 2.44 (Fig. 7). The ERs in other studies ranged from 1.2 to 5.6 (Jacinthe et al. 2004; Quinton et al. 2006). An ER > 1 indicated that the eroded sediment was enriched in SOM relative to uneroded soil suggesting the preferential removal of finer clay particles with sorbed SOM.

SOM Loss

The calculated rates of SOM loss using Eq. 2 within the watershed ranged from 0.00 to 1.19 MT/ha/yr (Fig. 8). The mean value for the watershed was 0.41 MT/ha/yr, which was slightly higher than other reported values of 0.02 to 0.34 (Jacinthe et al. 2004; Quinton et al. 2006; Berhe et al. 2007). These studies were reported from sites that had, for the most part, equivalent erosion rates ($<$10 MT/ha/yr). The one site in these studies with higher erosion rates had $<$1% SOM in the soil (Berhe et al. 2007).

AnnAGNPS also simulated the export of Total Organic Carbon (TOC) from the watershed, which was compared to the estimated load using the SOM loss within the CCEW determined from Eq. 2 and the SDR for each cell. Total organic carbon (TOC) from the AnnAGNPS simulations was first adjusted to relate the value to SOM using a simple multiplier (1.72, as suggested by Neitsch et al. 2002). The annual average SOM load exported from the watershed that was evaluated using the AnnAGNPS total carbon load and the conversion factor of 1.72 was 0.21 MT/ha/yr, which was less than the 0.41 MT/ha/yr determined using Eq. 2. Despite the difference, some of the statistical tests used to compare the values were favorable (Table 5).

The SOM loss was related to both erosion rate and organic matter content to understand their influences on SOM removal (Fig. 9). A linear relationship was clearly seen between erosion and SOM loss (Fig. 9a), which had been observed in other studies (Jacinthe et al. 2004; Quinton et al. 2006). Quinton et al. (2006) also observed a strong linear relationship between organic matter content and SOM loss. In this study, the relationship between SOM content and SOM loss contained considerable scatter (Fig. 9b). The stronger linear relationship between erosion and SOM loss may be attributed to the equation (Eq. 2) used to calculate SOM loss. Erosion was considered twice because it was used to calculate the ER. Because of this relationship, SOM loss followed similar trends as soil loss when grouping by soil series and management practices. Therefore, the highest SOM loss was seen from the upland soils, which contained Tama and Downs series soils, as well as from the lands under the FTC-NTB land use (Table 3).

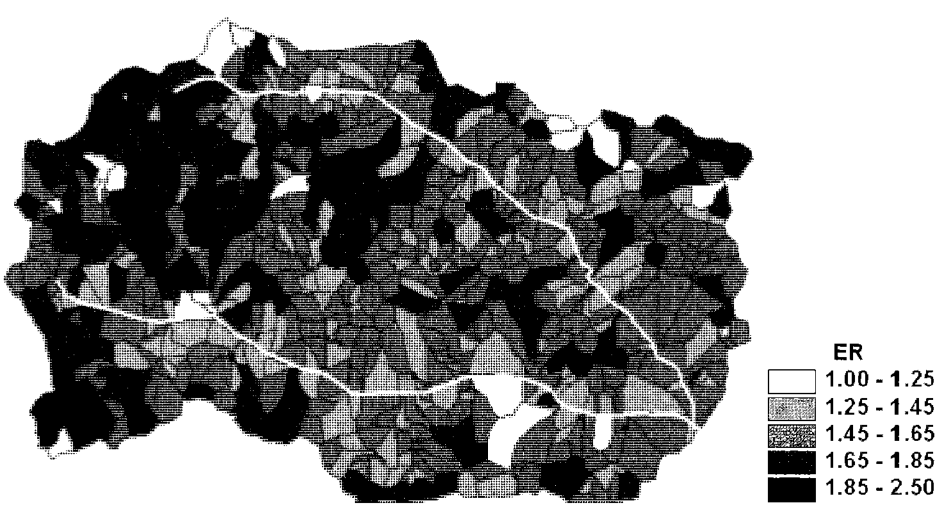

Fig. 7. Organic matter enrichment ratio (ER) for the CCEW.

Fig. 8. Gross SOM loss for the CCEW.

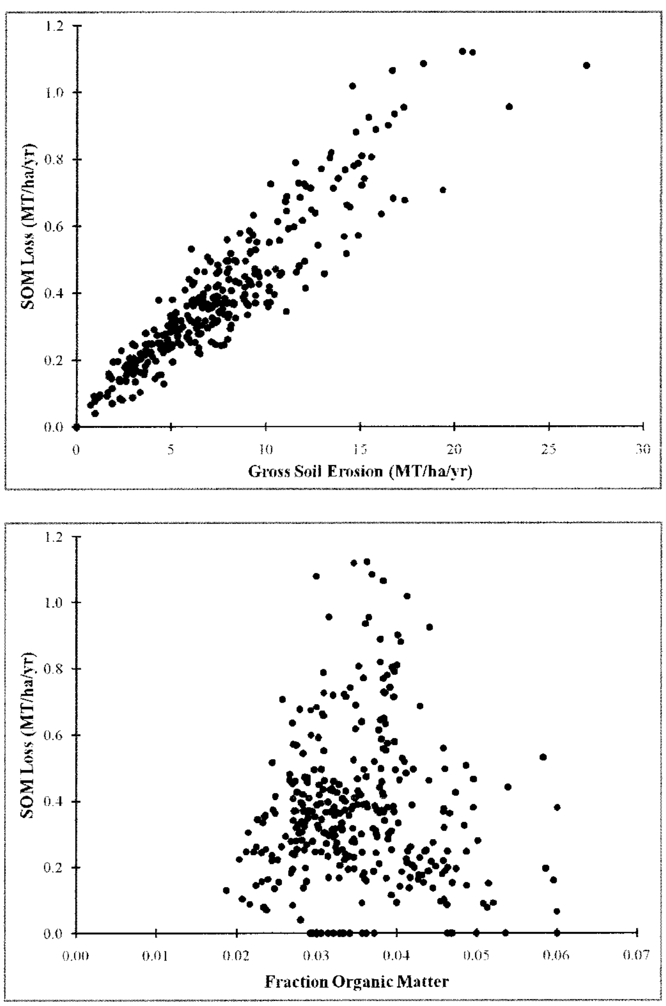

Fig. 9. Relationships between SOM loss and (a) gross upland erosion and (b) percent organic matter for the CCEW.

Soil Conditioning Index (SCI)

High erosion rates and associated SOM loss can be detrimental to soil quality, especially since estimates of soil renewal rate are on the order of 1.1 MT/ha/yr (McCormack et al. 1979; Montgomery 2007). To understand the state of soil health in this watershed, the NRCS Soil Conditioning Index (SCI) was determined for the individual AnnAGNPS cell (Fig. 10).

The SCI determines the effects of management practices in a system on SOM content (USDA-NRCS 2002). SOM strongly reflects the soil condition because it relates to several soil characteristics, including aggregate stability and water holding capacity (USDA-NRCS 2002). Again, a positive SCI suggests fields have improving SOM, while conversely a negative SCI suggests degrading SOM. For this study, the average SCI for the watershed was 0.38 ± 0.15. Only 1.3% of the cells reported SCIs less than 0.1. Thus, the state of SOM in the CCEW is improving due to ongoing conservation practices, namely no-till and reduced tillage farming.

CONCLUSIONS

In the present study, gross upland erosion was determined using AnnAGNPS for a small sub-watershed in the headwaters of the Clear Creek, IA watershed. The sediment delivery ratio for each cell was used to calculate an ER for SOM. Using the gross cell erosion, the ER, and an organic matter content GIS map, the SOM loss due to erosion was calculated.

Although organic matter content is not the sole characteristic used to determine soil health, it has been shown that more organic matter relates to higher soil quality. The SCI is not a predictor of soil quality; however, it determines if systems have improving or degrading SOM conditions. The sub-watershed farms had a positive SCI; thus, the state of the CCEW is improving due to ongoing conservation practices (namely conservation tillage, which is used during at least one year of all six corn-soybean rotations). This study can be duplicated using additional soil quality parameters to determine the overall health for a watershed. The use of GIS is useful in establishing the current state of health in a watershed and the coupling with

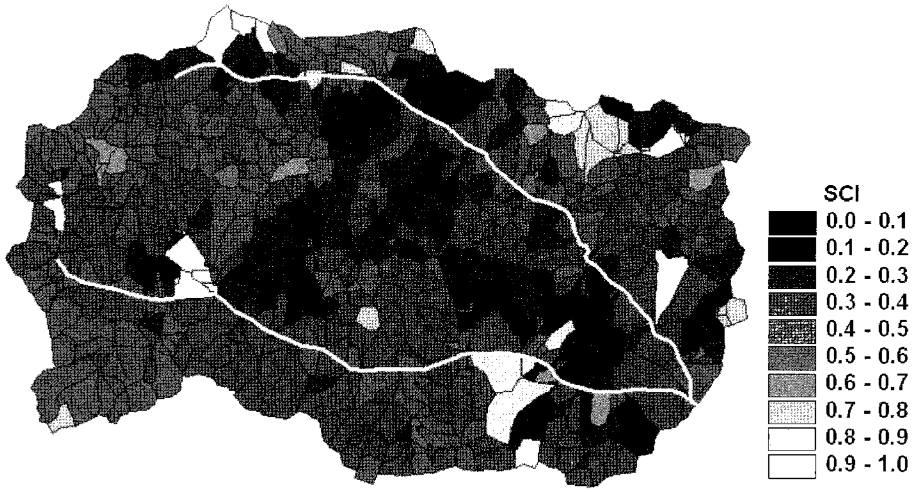

Fig. 10. SCI for the CCEW.

NPS models can provide the effects of implementing certain management practices within a system.

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