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Dynamic Assessment of Current Management in an Intensively Managed Agroecosystem

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Abstract: To assess management impacts on the functionality of intensively managed agroecosystems, a modeling framework was developed with a bottom-up approach and spatially distributed, process-based models. The framework is equipped with dynamic, data-informed indicators and indices to illuminate the factors influencing sustainability. The proposed dynamic indices consider natural and human aspects of an agroecosystem such as erosion, biogeochemistry, and economics. Most current indicators are static, or slow-changing, soil characterization parameters that reflect better long-term interactions between landscape features, climate, and biology. However, the ever-changing land management and climate necessitates the use of dynamic parameters that reflect agroecosystem responses to different land management on similar timescales (e.g., seasonally). Our framework examines the performance of different ecosystem services including crop productivity, carbon storage, and net income under three different strategies with varying degrees of tillage intensity. The strategy with the highest intensity produced the highest yields, but also had the highest production costs. The second most intense strategy also had high yields, as well as the highest net income. However, these two strategies produced high erosion rates, which depleted the recalcitrant soil carbon, a critical component of system productivity and health. The index that provided the clearest picture of improvement within an agroecosystem was the Carbon Management Index (*CMI*), which incorporates carbon lability and the implicit accounting of soil carbon redistribution from erosion. The *CMI* for the more intense strategies decreased in recent years, showing that they are not sustainable despite their high short-term productivity or profitability. The least intense strategy had the lowest Soil Organic Carbon (SOC) depletion through erosion and the highest *CMI* with a trend that is still-increasing through the present. Our study shows that to augment SOC storage, it important not only to increase the overall organic matter input, but also increase the amount of recalcitrant carbon in the soil and the longevity of all soil carbon through aggregate formation.

Keywords: *bottom-up modeling framework, management practices, erosion, soil organic carbon, Carbon Management Index, Clear Creek, IA, ecosystem services*

As the Earth's population approaches nine billion people in the upcoming decades, food production must increase by nearly 70% to satisfy this global need (Reytar et al. 2014). To quell global food demands, productivity has increased in recent years through agrotechnological advances (e.g., Fisher et al. 2014) and crop breeding (Klumper and Qaim 2014), but the ramp-up of production and associated intensive management are degrading agroecosystems (Lal 2004), thereby threatening their sustainability (Karlen et al. 2014).

Ecosystem degradation due to intensive agriculture is seen vividly in Iowa, which has become a world-leading, commodity-based agroecosystem due to its overall fertile soils. However, the enhanced erosion triggered by accelerating production levels, in conjunction with more frequent shifts between high intensity rainfall and deep droughts, is so extreme that two-thirds of the farmland in Iowa is losing soil at a rate over 11 Mt/ha/yr, which is equivalent to a little over a dump truck load per hectare per year (Tevis

2015). During a week of intense storms in 2014, Iowa farmers lost an estimated $1 billion in yield due to erosion (Eller 2014).

Additionally, intensive management has depleted the soils of essential nutrients (Pimentel et al. 1995) making Iowa a "nutrient-hungry system." To maintain high production, artificial fertilizers are broadly applied in mass quantities, but nearly half of these applied fertilizers are lost through leaching, runoff, and erosion (Burkart et al. 2005). As a result, recent record nitrate levels in the Raccoon and Des Moines Rivers in Iowa, predominantly from agricultural runoff, are taxing the aging infrastructure of the Des Moines Water Works and have even prompted lawsuits against three upstream, agricultural counties (Eller 2015). Further downstream, the problems persist as nutrient fluxes from the Upper Mississippi River Basin have been linked to hypoxia in the Gulf of Mexico (Porter et al. 2015).

The widespread ramifications of this ecosystem degradation have pushed the development of adaptive, watershed-scale management plans for improving the sustainability of agroecosystems like those in Iowa. But, do we have the necessary tools to assess accurately at the watershed scale the performance of the different practices currently in use or possible alternatives (e.g., Robertson et al. 2014)? As a substitute to extensive field monitoring studies, computer models can work as an indirect form of adaptive management (Prato 2003) to evaluate watershed-scale responses to different combinations of current and projected practices.

One factor to consider in developing these assessment modeling tools is that intensive land management and the resulting soil redistribution can augment the natural heterogeneity observed in agroecosystems along a hillslope for properties like Soil Organic Carbon (SOC) and other nutrients by dictating flowpath development and soil erodibility (e.g., Wilson et al. 2009; Dlugoβ et al. 2010; Stavi and Lal 2011; Papanicolaou et al. 2015b). Hence, the selective entrainment and resulting fluxes of these constituents may differ significantly in areas where erosion dominates relative to areas where deposition occurs (Van Oost et al. 2006; Wang et al. 2015), resulting in the increased variability across the landscape.

Since most biogeochemical models (e.g., ROTH-C, DNDC, CENTURY) used to quantify changes in SOC or other nutrients focus within a soil column, it is unlikely that they will capture this augmented, field-level heterogeneity. As a result, existing models will either overestimate or underestimate stock predictions depending on the location within a field, since they do not consider selective mobilization (e.g., Parton et al. 1987; Paustian et al. 1992; Harden et al. 1999; Manies et al. 2001).

The majority of studies that have attempted field-scale assessments of SOC linked these existing biogeochemical models with lumped erosion models, such as those based on the Universal Soil Loss Equation, USLE (e.g., Monreal et al. 1997; Zhang et al. 2014), to account for redistribution along the downslope. These erosion models provide gross, annual average estimates but do not capture the seasonal variability in concentrations and fluxes (Harden et al. 1999) or account for deposition and hence in-field storage (Gregorich et al. 1998; Van Oost et al. 2006).

A handful of recent, but significant modeling efforts have accounted for the collective dynamics of erosion and deposition on SOC redistribution and storage (e.g., Billings et al. 2010; Dlugoβ et al. 2012). Fewer studies have incorporated the effects of selective entrainment and deposition of the finer size fraction of organic rich soils by the flow (Papanicolaou et al. 2015b). However, these modeling advances have not yet transferred from the field-scale to the watershed-scale.

In general, integrative assessments at the watershed scale typically use Earth system models that follow a top-down approach, working to identify emergent patterns of SOC. These integrative assessments still use lumped erosion models, along with coarse field erosion estimates and decomposition rates from regression equations with climate variables to determine Net Ecosystem Exchange (e.g., Brenner et al. 2001; Ahmadov et al. 2007; Yadav and Malanson 2008). These estimates can potentially lead to inaccurate estimates for ecosystem services related to SOC, with direct implications on the design of efficient economic policies for providing incentive payments or credits to potential producers (Antle and Mooney 2002; Babcock 2010). Thus, these assessments are not truly dynamic in that they provide only gross averaged estimates of erosion, SOC formation, and decomposition on an annual basis (e.g., Polasky et al. 2008).

To reflect better the impacts of agricultural management, dynamic indices are needed that consider the spatial and temporal variability of key biogeochemical properties and processes in agricultural fields and pertain to ecosystem services, like food security, nutrient cycling, and climate regulation. Dynamic indices are better suited to assess functionality and sustainability of an agroecosystem than the static, or slow-changing, soil characterization parameters that more accurately reflect long-term interactions between landscape features, climate, and biology (Doran and Parkin 1996; Lehman et al. 2015), as they reflect the response of the agroecosystem to different land management on relevant time scales. Dynamic indices, which combine different parameters or different aspects of similar parameters, can capture this complexity, meaning they are able to change on similar time scales as the dominant drivers of climate and management. Thus, they provide a more complete accounting of sustainability (Hester and Little 2013).

An extensive literature review was performed to identify different indices currently used to assess sustainability (e.g., Costanza et al. 1997; de Groot et al. 2002; Millennium Ecosystem Assessment 2005). The selection was based on certain criteria (Dale et al. 2013). The indices had to be (1) practical, easy to measure, and inexpensive; (2) responsive to different drivers from both nature (e.g., climate) and humans (e.g., land management); and (3) responsive in a known and predictable manner, to facilitate better understanding of the associated changes in the measurements. In the end, most of these indices related to SOC.

SOC is closely linked to different services of an agroecosystem like food security, greenhouse gas regulation, and climate regulation, because it is inherently tied to several soil functions, such as nutrient cycling and habitat productivity. Soil carbon is the main source of energy for soil microorganisms, which drive decomposition and transformation of soil nutrients into a form that is readily available for plant uptake. More available nutrients leads to higher crop yields for food security and reduced fertilizer needs, which lowers production costs. Additionally, SOC can regulate the ability of the soil to hold water, especially in times of drought (e.g., Vereecken et al. 1990; Righetti and Lucarelli 2007; Papanicolaou et al. 2015a). The organic matter also acts as a "glue" holding the soil particles together and preventing erosion (e.g., Tisdall and Oades 1982; Papanicolaou et al. 2015b). Lastly, soils can be a storage reservoir for atmospheric CO_2 (e.g., West et al. 2010), helping in climate regulation.

In this study, the central objective was to develop a bottom-up, yet practical modeling framework to assess the performance of current management strategies (e.g., different combinations of crops and tillage intensities) in an intensively managed agroecosystem of the U.S. Midwest for sustainability. Herein, we define sustainability as the agroecosystem's ability to lessen the negative effects of erosion and SOC loss on the environment while remaining productive and profitable in the long-term. In developing this framework, we sought to overcome the limitations of Earth system models used for watershed assessments. Our modeling framework integrates spatially distributed, process-based erosion and biogeochemical models with an economic assessment. The framework was used to provide indices that reflect the dynamic response of the landscape to the various management strategies and distinguish the differences between short-term productivity/profitability and long-term sustainability resulting from the different strategies.

With this framework, we can help address the following questions: (1) are current management strategies geared towards high production ultimately degrading our agroecosystems beyond the point of functioning effectively, even with added supplements?; and (2) how can we better govern these systems so that they will still be productive in 50 years? Therefore, in order to ensure long-term, high-yield production and establish a sense of food security in light of volatile market demands and climate, we must develop more effective, data-informed, watershed management and governance plans for our agroecosystems that are also sustainable. A major contribution of this study is the development of a watershed-scale, modeling framework that captures the cause-and-effect relationship between land management and soil health to cultivate better management/governance plans.

Study Site

We tested the developed framework in the Clear Creek, IA watershed (Figure 1), which is part of the National Science Foundation Intensively Managed Landscapes - Critical Zone Observatory (IML-CZO; http://criticalzone.org/iml). The IML-CZO has available hydrologic, erosion, biogeochemical, and economic databases for educating the modeling framework. Additional data from remote sensing sources were also used to look at information from the watershed scale and develop carbon budgets for Clear Creek that incorporate the erosion effects of different land management practices, namely tillage.

The Clear Creek watershed is a 270-km^2, mixed agroecosystem in east-central Iowa (Figure 1). Clear Creek is representative of Iowa and most of the Midwest in terms of land use (predominantly agricultural with developing small pockets of urban areas), soil (Alfisols and Mollisols), and climate (humid-continental).

Land Use

Over 80% of the watershed has been converted from prairie and forest to row-crop agriculture and pasture, a pattern that typifies the American Midwest (Rayburn and Schulte 2009). Since 1991, the dominant management strategies in the watershed are 2-year, corn-soybean rotations that use in parts conservation tillage practices (reduced and no-till). These strategies are identified as the following: no till corn – fall till bean (NTC-FTB); spring till corn – no till bean (STC-NTB); and fall till corn – no till bean (FTC-NTB). A detailed description of the associated practices and equipment is in Abaci and Papanicolaou (2009).

For the purposes of this study, it is important to note that fall tillage is typically in October with a chisel plow and is deeper than the reduced tillage of spring, performed with a field cultivator in April (Reicosky 2015). Table 1 lists the areal extents for each strategy, as well as their relative intensities, based on the Soil Disturbance Rating (SDR). The SDR considers the impact of the agricultural equipment used in a particular management strategy through inversion, mixing, lifting, shattering, aeration, and compaction of the soil. Higher values reflect a greater disturbance to the soil. The order of the different strategies in terms of decreasing intensity is as follows: NTC-FTB (53) > FTC-NTB (41) > STC-NTB (32).

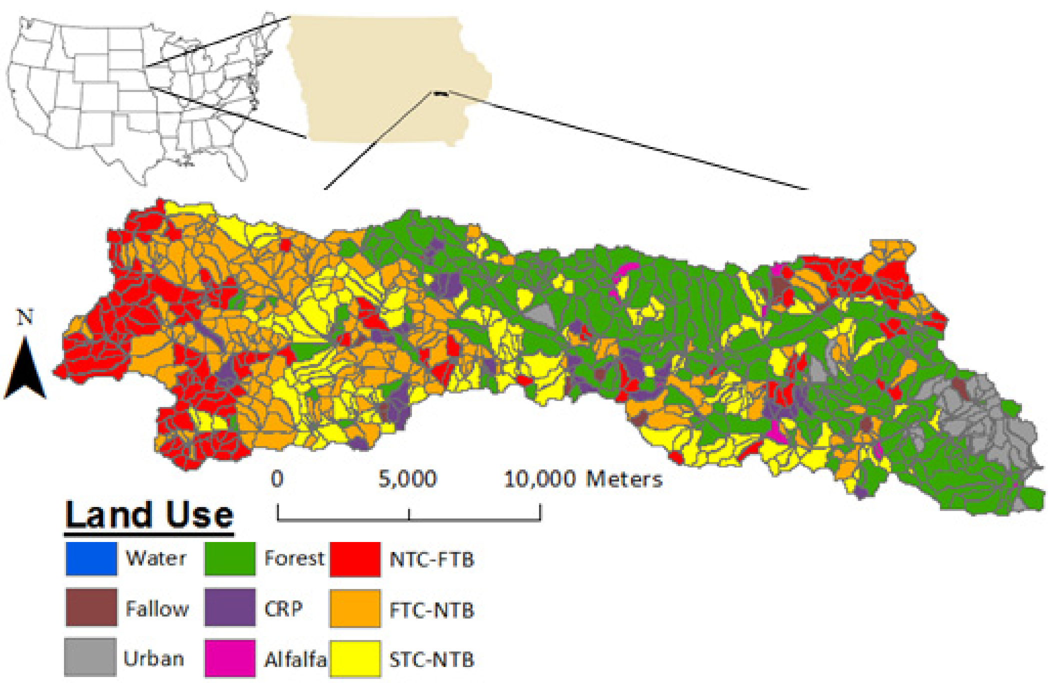

Figure 1. The Clear Creek, IA watershed and a reclassified land management map for Clear Creek. NTC-FTB = no till corn – fall till bean; STC-NTB = spring till corn – no till bean; and FTC-NTB = fall till corn – no till bean.

Table 1. Dominant corn-bean rotations in Clear Creek, IA.

Rotation	Areal Coverage (%)	Soil Disturbance Rating	Tillage Intensity
NTC-FTB	26	53	High
FTC-NTB	40	41	Med
STC-NTB	34	32	Low

See Abaci and Papanicolaou 2009 for areal coverage and details on management strategies.
NTC-FTB = no till corn – fall till bean; STC-NTB = spring till corn – no till bean; and FTC-NTB = fall till corn – no till bean.

Soils

The watershed is situated in the Southern Iowa Drift Plain, which is comprised of glacial deposits broken up by rolling hills (slopes in Clear Creek range up to 12%) with many small creeks and streams. Windblown Peorian loess deposits cover the area to depths ranging from 1.5 to 9 m (Ruhe 1969). The loess-derived soils are very productive agricultural lands due to their ability to hold water and nutrients (Jones et al. 1967), but being loess-derived, the soils are relatively homogeneous, consisting predominantly of silt-sized particles, and are highly erodible under intensive land management, namely tillage (Bettis et al. 2003).

Mollisols and Alfisols are the dominant soil orders found in the watershed. The most common upland soil associations are the Tama-Downs and Fayette-Downs associations (Dideriksen et al. 2007), which are both well drained. The dominant lowland association, which forms on either stream terraces or flood plains, is the Colo-Nevin-Nodaway association with drainage classes ranging from poorly drained to moderately well drained (Highland and Dideriksen 1967). Moving downstream, the dominant soil texture changes from silty clay loam in the headwaters to a silty loam near the mouth of Clear Creek.

Climate

Due to the mid-continental location of Iowa, the climate in Clear Creek is characterized by hot summers, cold winters, and wet springs (Highland and Dideriksen 1967). Average annual precipitation is approximately 889 mm/yr with convective thunderstorms prominent in the summer and snowfall in the winter. Monthly precipitation averages range from approximately 20 mm/month between December and February to almost 120 mm/month in June. The majority of stream flow occurs during spring and summer, with peak stream flows in May and June.

Erosion

Fields in Clear Creek have some of the highest erosion rates in Iowa. These elevated rates are mainly due to a combination of swelling (i.e., smectite-rich) and highly erodible soils with steep slopes (up to 12%) and intensive agriculture (Piest et al. 1975; Cruse et al. 2006; Gilroy 2006). Reported erosion rates in Clear Creek averaged 20 Mt/ha/yr, with individual fields reporting erosion rates up to 150 Mt/ha/yr (e.g., Abaci and Papanicolaou 2009; Papanicolaou et al. 2009; Wilson et al. 2009).

Modeling Simulations

The simulation time period used for this study was from 1991 to 2012, which captures the time frame that conservation practices (namely, reduced tillage) have been practiced in Clear Creek. Daily values were determined during the simulation and the results were aggregated annually. The first 10 years were used as a spin-up period to allow the system to adjust to the new rotation (Wilson et al. 2009), with the results from the latter 10 years being averaged and used as the study results.

Modeling Framework

The modeling framework (Figure 2) used herein incorporates a bottom-up approach that integrates spatially distributed, process-based models to address erosion and biogeochemistry in an agroecosystem, coupled with an economic component. This bottom-up approach allows for capturing the cause-and-effect relationships between different cropping rotations and the processes of infiltration, runoff, and water-driven/tillage-enhanced erosion, which shape biogeochemical stabilization and degradation of SOC. Additionally, this framework allows us to incorporate the influence of heterogeneity in the vertical, downslope, and planar directions for different landscape properties on these processes. These benefits translate to more accurate assessments of ecosystem services, as coarser estimates from lumped approaches fail to capture the combined effects of management and local conditions on nutrient cycling or the effects of processes like deposition within a field.

Erosion

This modeling framework (Figure 2) uses the Water Erosion Prediction Project, or WEPP *model (version* 2012.8) for simulating runoff and erosion (Papanicolaou et al. 2015b). WEPP is well established as a process-based model that works at both the hillslope and small watershed scales (e.g., Papanicolaou and Abaci 2008; Dermisis et al. 2010). Further details for WEPP can be found in Flanagan and Livingston (1995), Ascough et al. (1997), and Abaci and Papanicolaou (2009).

WEPP has a strong management component for agroecosystems with input parameters to describe the different crops, tillage regimes/schedules, and other practices. However, due to the importance of land management, and especially tillage, on soil erosion, soil respiration, and hence carbon budgets, a more concerted effort was made to capture the different degrees of tillage intensity across the watershed. Many past studies have shown that the degree of tillage intensity directly relates to the level at which it affects soil, biogeochemical, and hydrologic processes (e.g., Papanicolaou et al. 2015b). Therefore, to assess accurately the role of management on the overall functions of agroecosystems, it is important to distinguish between conservation and conventional tillage.

For our framework, a more-detailed land use classification map (Figure 1) was developed using satellite and aerial imagery with a correlation between soil reflectance and tillage intensity, namely the Normalized Difference Tillage Index, NDTI (e.g., Daughtry et al. 2006; Serbin et al. 2009). More disturbed soils appear darker due to less surface litter following tillage (Van Deventer et al. 1997). In this study, an implicit land management detection algorithm was developed using the NDTI to reclassify the row crop category in the National Land Cover Database (NLCD) maps into more detailed land management rotations (Figure 3).

Landsat images for the month of November were collected for areas in Clear Creek classified as row crop. November images were used to ensure that

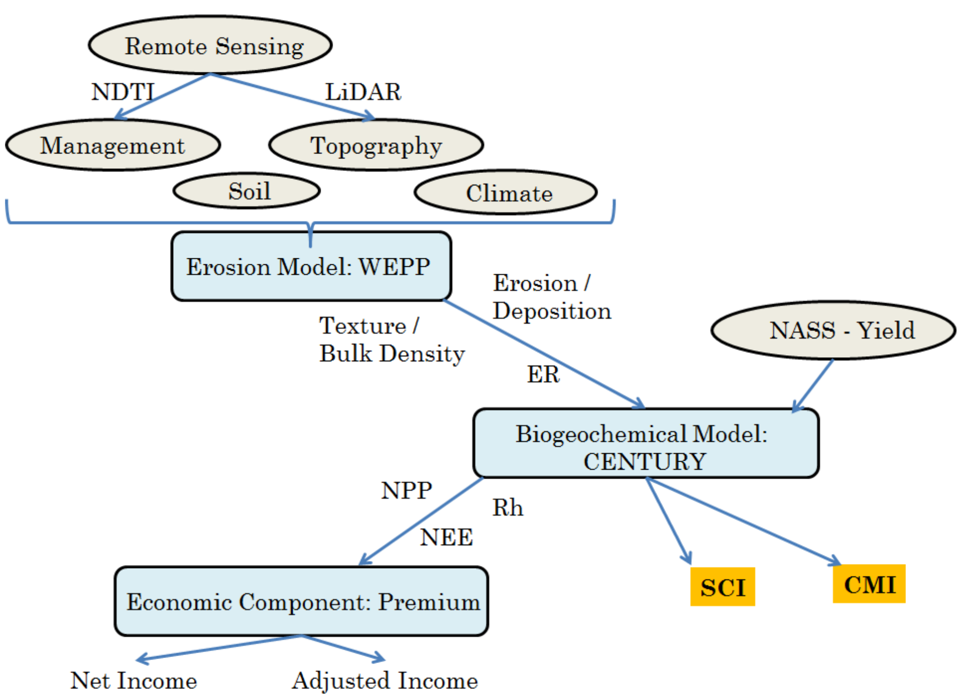

Figure 2. The linked WEPP-CENTURY modeling framework. From Papanicolaou et al. (2015b).

harvest and fall tillage practices were complete (Abaci and Papanicolaou 2009). The NDTI (Figure 3) was determined through a manipulation of reflectance wavebands (band 5, 1550–1750 nm; band 7, 2080–2350 nm) from the Landsat images (Daughtry et al. 2006). The average NDTI values for these fields ranged from 0.245 to 0.489. The NDTI values were then distributed throughout the corn and soybean fields of Clear Creek using the areal distribution of the selected management strategies found in Clear Creek (Table 1), with the lower NDTI values relating to higher tillage disturbance as shown in Figure 3a.

Along with the detailed management component, another benefit of WEPP is the ability to simulate multiple non-uniformities along a hillslope in terms of soils, slopes, and management by dividing the hillslope into smaller Overland Flow Elements (OFEs). This sub-division into OFEs by WEPP makes it a spatially distributed model that can effectively capture the overall effects of spatial heterogeneity in terms of hydropedological properties, land use, and hillslope curvature on runoff and erosion for different event, seasonal, and inter-annual periods (e.g., Abaci and Papanicolaou 2009; Dermisis et al. 2010). Capturing this heterogeneity in landscape properties is important due to its impact on observed flowpaths.

To define accurately the flowpaths, LiDAR data for the watershed, collected as part of the IML-CZO, were used to produce a digital elevation model (DEM), reclassified to a 5-m resolution. The DEM was then used to delineate the channel and hillslopes.

The hillslopes and channels in Clear Creek were connected using the geospatial version of WEPP, which incorporates subroutines like TOPAZ (Garbrecht et al. 1996) to pass water (and constituents) from one OFE to the next along topographic changes. Other WEPP subroutines are then applied individually in each OFE along the identified flowpaths for calculating runoff, infiltration, and net rill and interrill erosion. The "flowpath" approach takes our framework from the soil column to the landscape by allowing the physical mechanisms of runoff and erosion to redistribute SOC along a hillslope, in light of different management schedules for tillage and fertilizer, as well the presence of tiles, which influence carbon stocks at depth (Liu et al. 2011).

In addition to runoff and erosion rates, enrichment ratios (*ER*) were determined through WEPP, which

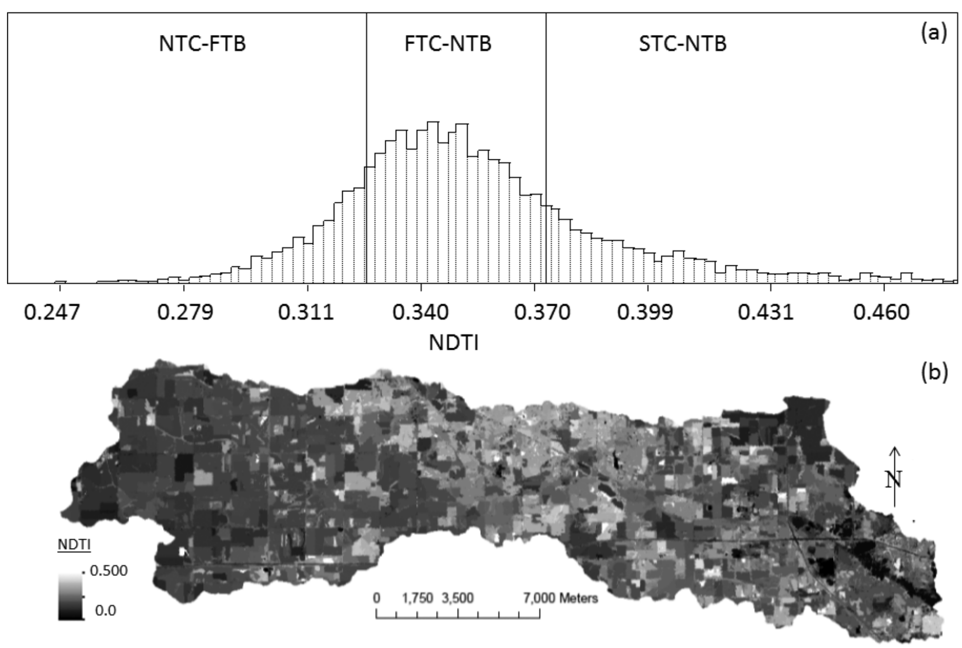

Figure 3. (a) Distribution of 10-year average Normalized Difference Tillage Index (NDTI) values for Clear Creek. (b) A map of the 10-year average NDTI values. NTC-FTB = no till corn – fall till bean; STC-NTB = spring till corn – no till bean; and FTC-NTB = fall till corn – no till bean.

are needed for quantifying the fraction of SOC that has been eroded (Papanicolaou et al. 2009; 2015b). *ER* is important for capturing the selective entrainment of lighter, but more organically rich soil. *ER* is expressed as the proportion of SOC in transported sediment compared to that of SOC in uneroded soil. The enrichment ratio was determined using the specific surface area of the soil (SSA_{SOIL}) within each hillslope and the specific surface area of the different size fractions of eroded sediment exiting a hillslope (SSA_{SED}) as follows:

$$ER = \frac{SSA_{Sed}}{SSA_{Soil}} \tag{1}$$

An $ER > 1$ indicates that the eroded sediment is enriched in SOM relative to the uneroded soil, whereas an $ER < 1$ denotes the opposite. In this study, it is important to note that due to long-term averaging, the functionality of the *ER* is limited since *ER* best capture the preferential redistribution of SOC at the event scale. Enrichment ratios within the Clear Creek watershed range between 0.8 and 3.0 for single events (Papanicolaou et al. 2009; 2015b), suggesting the likelihood of high rates of SOM loss during runoff events. Over the long-term, however, the *ER* approaches unity.

To show the benefits of using the process based WEPP model when determining erosion and SOC lost, the erosion rates from WEPP were compared against other lumped models, namely the E_{30} method and the USLE. The section below provides a short description of each method although the more detail-oriented reader is referred to the following publications: Hazarika and Honda (2001) for the E_{30} method, Wischmeier and Smith (1978) for the USLE, and Flanagan and Nearing (2000) for further information on WEPP.

The E_{30} method was developed for quantifying erosion in areas that are inaccessible for collecting physical measurements. The method relies on only the field slope, *S*, (normalized to a 30 degree slope, S_{30}) and remote sensing images for vegetative indices, such as the Normalized Difference Vegetation Index, NDVI (Hazarika and Honda 2001) to estimate annual erosion rates, *E*.

$$E = E_{30}\left(\frac{S}{S_{30}}\right)^{0.9} \tag{2}$$

where E_{30} is the rate of soil erosion for a 30 degree slope. It is quantified as below:

$$E_{30} = Exp\left[\left(\frac{ln(10)\text{-}ln(114)}{NDVI_{max}\text{-}NDVI_{min}}\right) \text{x } (NDVI\text{-}NDVI_{min}) + ln(114)\right] \tag{3}$$

The maximum and minimum erosion rates for a 30 degree slope in Clear Creek using a representative soil type and management were 10 and 114 mm/yr, respectively. These become the coefficients in Equation 3. The NDVI data were obtained from the Landsat-TM images for the region. Minimum NDVI values were obtained from April images spanning the 1991-2012 simulation period, while August images were used as the maximum NDVI images.

The USLE was also used to determine annual average erosion rates, *E*. The model develops individual indices for the dominant factors controlling erosion: rainfall and runoff erosivity (*R*), soil erodibility (*K*), slope length (*L*), slope steepness (*S*), vegetative cover/management practices (*C*), and conservation measures (*P*) (Equation 4) (Wischmeier and Smith 1978). The parameters of the USLE are empirically based on a large historical dataset. The values used in this study come from the USLE manual (Wischmeier and Smith 1978) and the available hydrologic, erosion, and biogeochemical databases of the IML-CZO (Table 2).

$$E = R \text{ x } K \text{ x } LS \text{ x } C \text{ x } P \tag{4}$$

The erosion rates from the E_{30} and USLE analyses were compared with the annual average rates provided by WEPP in the herein-developed modeling framework. WEPP determines erosion using an excess shear relationship:

$$\frac{\delta E(x)}{\delta x} = (\tau - \tau_c)(E_e - E(x)) \tag{5}$$

where τ is the shear stress of the runoff; τ_c is the critical shear stress of the soil; E_e is the equilibrium erosion rate; and *E(x)* is the local erosion rate for a unit of the hillslope. It is important to note that the WEPP rates are summed along discrete lengths of the hillslope, which include both the upland erosional areas and the downslope depositional areas.

Biogeochemistry

Our framework (Figure 2) loosely couples WEPP with the CENTURY soil biogeochemical model (version 4.6) to address nutrient cycling (Papanicolaou et al. 2015b). CENTURY simulates carbon, nitrogen, phosphorus, and sulfur dynamics through plant-soil interactions in different grassland, agricultural, and forest ecosystems. Multiple sub-models (e.g., soil organic matter/ decomposition sub-model, water budget sub-model, and grassland/crop sub-model) are coupled with a management/event scheduling function to compute nutrient fluxes. For CENTURY, more detail can be found in Parton et al. (1987), Paustian et al. (1992), Harden et al. (1999), Manies et al. (2001), Jarecki et al. (2008), Tornquist et al. (2009), and Wilson et al. (2009).

WEPP and CENTURY were coupled to address certain limitations of existing biogeochemical models. As mentioned above, most biogeochemical models like CENTURY are limited to the soil column and cannot adequately capture the effects that spatial heterogeneity across the landscape (e.g., soil properties, tillage-induced roughness, crop coverage, patchiness), organizational complexity (e.g., relief, curvature, drainage network), and soil erosion/deposition have on SOC stocks (Van Oost et al. 2007; Kuhn et al. 2009; Papanicolaou et al. 2015b).

The net erosion, enrichment ratios, bulk density, and texture for each hillslope were passed from WEPP to CENTURY, which further accounts for the effects of plant phenology, litter fall, and decomposition on SOC concentrations. Net erosion rates and ER were used to quantify the loss of SOC due to redistribution during a runoff event. The flux of eroded carbon (SOC_{EROD}) was determined using the following relationship:

$$SOC_{EROD} = (TSOC)\left(\frac{ER \text{ x } EROD_{Net}}{BD \text{ x } d}\right) \quad (6)$$

where $TSOC$ is soil organic carbon; $EROD_{Net}$ is net erosion provided by WEPP; BD is soil bulk density; and d is depth of the active layer, which is the top 20 cm.

The loss of SOC due to erosion was added as a correction to the Net Ecosystem Exchange (*NEE*) calculations, which were used to determine carbon budgets in Clear Creek. The effects of erosion on SOC are often neglected in watershed carbon budgets (e.g., Brenner et al. 2001; West et al. 2010); however, erosion depletes SOC stocks directly (Papanicolaou et al. 2015b), and indirectly through changes in production and respiration.

NEE represents the net exchange of CO_2 from the biosphere to the atmosphere (Kirschbaum et al. 2001). It is the difference between Net Primary Productivity (*NPP*) and the microbial respiration from the soil (R_h):

$$NEE = -NPP + R_h \quad (7)$$

NPP is the gross plant production minus the CO_2 respired by the plants (Kirschbaum et al. 2001). In our framework, *Aboveground Net Primary Productivity* (*ANPP*) is determined using annual crop yields, *Y*, from the U.S. Department of Agriculture (USDA) - National Agriculture Statistics Service (NASS) or calculated with NDVI. These spatially averaged values are adjusted for each hillslope using the Corn Suitability Rating (Secchi and Babcock 2007) provided by the Iowa Soils Properties And Information Databases (ISPAID). The *ANPP* is then adjusted using a moisture weight correction for the plant, *MW* (normally 0.845), the fraction of carbon in the plant, C_{plant} (assumed to be 0.43; Latshaw and

Table 2. Universal Soil Loss Equation (USLE) parameters.

Rotation	R	K	LS	C	P	E (Mt/ha/yr)
NTC-FTB	63 – 250	0.31 ± 0.03	0.83 ± 0.54	0.3498	0.55 ± 0.05	20.5 ± 12.9
STC-NTB	63 – 250	0.32 ± 0.06	1.30 ± 0.57	0.3449	0.52 ± 0.04	33.9 ± 16.2
FTC-NTB	63 – 250	0.31 ± 0.04	1.16 ± 0.63	0.3527	0.53 ± 0.04	28.6 ± 16.7

NTC-FTB = no till corn – fall till bean; STC-NTB = spring till corn – no till bean; and FTC-NTB = fall till corn – no till bean.

Miller 1924), and the average Harvest Index, *HI*, for corn-soybean rotations, reported as 0.50 (e.g., Prince et al. 2001; De bruin and Pederson 2009):

$$ANPP = \left(\frac{Y \text{ x } MW}{HI}\right) C_{plant} \tag{8}$$

The *Belowground Net Primary Production* (*BNPP*) is then estimated using the *ANPP* and the root to shoot ratio, *R:S,* in the following equation:

$$BNPP = ANPP \text{ x } R{:}S \tag{9}$$

For corn-soybean rotations, the *R:S* values are 0.18 (Prince et al. 2001).

The soil respiration component, R_h, in our framework, is determined by adjusting the maximum decomposition rate (K_i) for the effects of climate (*CDI*), soil texture (*TEX*), the prevalence of anaerobic conditions (*A*), and tillage ($Cult_i$), such that:

$$R_h = C_i \text{ x } K_i \text{ x } CDI \text{ x } TEX \text{ x } A \text{ x } Cult_i \tag{10}$$

where the *i* represents the various pools of carbon and *C* is the concentration of carbon. It is important to note that our framework accounts for the impact that tillage events have on respiration rates by increasing decomposition to reflect the breaking apart of soil aggregates and enhanced mineralization (Harden et al. 1999; Reicosky et al. 2005).

Economics

The economic component of this framework is represented by the net incomes of the different management strategies. The net incomes are determined as the difference between the crop yield returns and the production costs. The relative crop yield returns are quantified using average statewide prices and crop yields determined from above. The production costs (Table 3) attributed to machinery, planting, fertilizer, pesticides, and harvest, as well as costs for labor, land rental, and crop insurance (Duffy 2012) are included.

The approaches of Antle et al. (2007) and Feng et al. (2004) were adopted because they account for the production of ecosystem services based on the relative revenues of alternative production systems.

$$Net\ Income = Crop\ Yield\ Returns - Production\ Costs + Premium \tag{11}$$

The net incomes in this study are adjusted with a premium that is attributed to the additional ecosystem service benefits of lessening erosion as it relates to production costs and carbon storage.

For the soil loss reduction component of the premium term, the cost of fertilizer lost was considered. For each ton of eroded soil, which contains 1.05 kg of nitrogen and 0.45 kg of phosphorus, the cost of lost fertilizer alone is \$2.10 (Iowa Learning Farm 2013). High erosion not only depletes the soil of organic matter and other nutrients, but it also contributes to water quality concerns. For example, in the winter of 2014-2015, the Des Moines Water Works spent \$540,000 to run their nitrate removal equipment (Eller 2015). This cost for maintaining water quality was not considered in this study, but shows the extent of possible ramifications.

The part of the premium term related to SOC requires the carbon sequestered in a field determined from WEPP-CENTURY. The premium associated with SOC storage was set at a fixed price (\$100/metric ton of C stored). This price is used for only demonstration purposes and is not a recommendation. This premium was estimated using different published sources. The average rate for a ton of carbon currently used by the cap and trade, emission trade schemes, and carbon taxes in place around the world is \$115 (World Bank Group 2014). Past studies have proposed rates ranging for \$10 to \$125 per ton of carbon (e.g., Lewandrowski et al. 2004). Finally, enrolling a field in the Conservation Reserve Program (CRP), on average stores carbon at a rate of 0.92 Mt/ha/yr (Follett 2001; Post et al. 2004). With the current CRP rental rate in Iowa averaging \$437/hectare, this translates to \$66/metric ton of carbon. Further analysis would be needed to propose suggested target prices.

Indicators and Indices

The distinction between short-term productivity/profitability and long-term sustainability for each management strategy is established by assessing the response of different indicators and indices that reflect the associated management intensity.

Table 3. Production costs for different practices in Clear Creek ($/ha).

	FTC	STC	NTC	FTB	NTB	Grass
Planting	$47.20	$26.93	$38.30	$49.42	$38.30	$23.15
Seed	$238.20	$238.20	$238.20	$135.90	$135.90	n/a
Nitrogen	$152.14	$152.14	$152.14	$0.00	$0.00	$4.45
Herbicide	$87.72	$103.11	$103.11	$65.48	$65.48	n/a
Insurance	$30.15	$33.61	$33.61	$19.52	$19.52	n/a
Harvest	$103.90	$109.19	$109.19	$58.49	$58.49	$160.19
Labor	$83.52	$72.15	$73.88	$72.28	$72.28	$128.49
Land	$560.91	$674.57	$674.57	$560.91	$560.91	$328.64
Total	**$1,303.73**	**$1,409.91**	**$1,423.00**	**$962.00**	**$950.88**	**$644.92**

See Duffy (2012).

The specific indicators for productivity include Net Primary Productivity, which reflects crop production, and soil respiration, which measures biological decomposition activity and reflects the capacity of the soil to support life. In addition, the Net Ecosystem Exchange is also determined to provide a carbon budget for each rotation. In terms of profitability, the net income is quantified, as well as an adjusted income. The adjusted income includes a premium for lessening erosion and storing SOC in the soil.

To assess the sustainability of the different crop rotations, two indices are used that consider both erosion and carbon storage, which are dynamic parameters of an agroecosystem, namely the Soil Conditioning Index (*SCI*) and the Carbon Management Index (*CMI*). These dynamic parameters are better suited to assess a watershed's functionality and sustainability than the more basic soil health parameters (Doran and Parkin 1996; Lehman et al. 2015), as they reflect the response of the agroecosystem to different land management on similar timescales (e.g., seasonally). Traditional soil health parameters are more static and reflect long-term interactions between landscape features, climate, and biology (Stott et al. 2013).

The *SCI* (USDA - Natural Resource Conservation Service, NRCS 2002) is essentially a simple weighted sum of three, empirically derived factors: Organic Matter; Field Operations; and Erosion. However, the factors are complex in that they contain multiple input parameters and researchers must use care to select the correct terms to calculate the factors. The *Organic Matter* factor reflects the effect of biological inputs in terms of residue, roots, cover crops, and manure returned to the soil. The *Field Operations* factor reflects the effect of management on stimulating organic matter breakdown using the Soil Disturbance Rating. The *Erosion* factor reflects the effect of sheet and rill erosion at removing organic matter. It is based on the soil erosion rate in relation to the Soil Loss Tolerance value, or T-value, which defines the maximum rate of erosion at which the quality of a soil as a medium for plant growth can be maintained (USDA - Natural Resource Conservation Service, NRCS 2002).

The second index used, the Carbon Management Index, *CMI* (Blair et al. 1995), accounts for both the level of disturbance to the SOC stock relative to a reference value (i.e., Carbon Pool Index, *CPI*), and the change in the type of carbon due to a disturbance relative to a reference value (Lability Index, *LI*).

$$CMI = CPI \times LI \times 100 \tag{12}$$

The *CPI* is simply the ratio of the total SOC in the soil of an agricultural field relative to a reference value, such as an undisturbed prairie. *CMI* values closer to 100 mean you are closer to the reference condition. Larger differences between the total SOC in a field relative to the reference values reflects the intensity of the disturbance to the field. Additionally, the *LI* is the ratio of labile carbon to non-labile carbon in an agricultural soil relative to a reference value. In this study, labile SOC is considered the material of the active pool of carbon determined from the CENTURY model, while non-labile SOC is considered from the more recalcitrant slow and passive pools derived in CENTURY (e.g., Cambardella and Elliot 1992; Vieira et al. 2007).

Results and Discussion

For the discussion below, we focus on only those hillslopes in Clear Creek under row-crop agriculture (Figure 1) to determine which management strategy is most beneficial for the long-term sustainability of the agroecosystem. As identified above, the three main row-crop strategies in Clear Creek are the following: no till corn – fall till bean (NTC-FTB); spring till corn – no till bean (STC-NTB); fall till corn – no till bean (FTC-NTB). Additionally, we define a sustainable practice as one that minimizes erosion and SOC loss while remaining productive and profitable in the long-term.

Productivity

The corn yields for all fields in Clear Creek averaged 393 ± 59 bu/ha, which fit in the range of yields (336 – 427 bu/ha) for east-central Iowa. Soybean yields averaged 121 ± 17 bu/ha, which also fit in the east-central Iowa range of values (106 – 128 bu/ha). Combined, these average corn-soybean yields for all rotations convert to an average annual total *NPP* of 599 ± 89 g C/m^2/yr. The NTC-FTB strategy, which had the highest tillage intensity, also had the highest average production 620 ± 100 g C/m^2/yr. In comparison, the least intense STC-NTB strategy had the lowest production at 567 ± 78 g C/m^2/yr. Although the difference is only about 9%, it is statistically significant (ANOVA; df = 2; F = 17.8; $p < 0.001$).

Past literature provides mixed results when relating tillage intensity and production (e.g., Phillips et al. 1980; Steiner 2002; Reicosky 2015), so it is difficult to attribute the yield differences entirely as a result of the specific management. A part of this difference may be due to where in the watershed the farmers are using the specific strategies. The inherent soil "fertility" also influences the productivity. Approximately 35% of the fields using FTC-NTB and 45% of the fields using NTC-FTB were in the 6 sub-watersheds with the highest average corn suitability ratings. This is in contrast to only 22% of the fields using STC-NTB. Therefore due to complicating factors including soil characteristics and the mixed results when relating tillage intensity and production, yield should not be the only indicator used to assess management sustainability.

Regarding soil respiration, the more intense strategies of NTC-FTB (385 ± 47 g C/m^2) and FTC-NTB (386 ± 50 g C/m^2) had respiration rates similar to the STC-NTB rotation (391 ± 51 g C/m^2; ANOVA; df = 2; F = 0.85; p = 0.4). From a modeling perspective, CENTURY accounts for tillage intensity, along with texture and the soil microclimate when determining soil respiration. However, the net effect of management on the R_h term can be similar despite the intensity.

Soil respiration is a direct reflection of microbial activity in the soil (Schlesinger and Andrews 2000). However, higher microbial activity may result from multiple mechanisms. Tillage will spark activity by bringing residue at the surface in contact with the microbes in the soil and by aerating the soil (e.g., Reicosky and Lindstrom 1993). Yet, the associated higher erosion rates will remove a high number of the microbes residing there, which will also tend to reduce respiration (Pimentel and Kounang 1998). Practices that leave more crop residue at the soil surface, like reduced and no-till practices (i.e., STC-NTB), add more organic matter to the soil thereby promoting soil respiration. These practices will also improve soil aggregation and porosity, meaning the soils are better aerated and hold more moisture (Papanicolaou et al. 2015a; Wacha 2016). Hence soil respiration is also not a good sole indicator to assess management.

For the NEE budgets, the more intense NTC-FTB (-194 ± 133 g C/m^2) and FTC-NTB (-196 ± 108 g C/m^2) strategies created larger sinks of carbon than the STC-NTB strategy (-155 ± 107 g C/m^2; ANOVA; df = 2; F = 7.63; $p < 0.001$). In looking at these carbon budget values for the three strategies, one may get the impression that the more intense strategies are suitable because they increase carbon storage. However, what these numbers fail to show is that the majority of this stored carbon is in the above ground plant material. As used in Equation 9, the root-to-shoot ratio for corn-soybean rotations is 0.18 (Prince et al. 2001), which indicates very little of the biomass is belowground. Moreover, the harvest index for these crops used in Equation 8 is 0.5, so half of the aboveground C is removed entirely from the system and never goes into storage in the soil.

Enhanced crop production can mask the negative effects of intense tillage practices (Gliessman 2004). Advances in agrotechnology have pushed yields higher; however, it is important to look not only *where* the carbon is stored (i.e., aboveground), but also at the residence *time* of the SOC that is being stored (Burdock and Crawford 2015). For simplicity sake, SOC can be separated into two pools - one that decomposes quickly (or labile carbon), which reflects better the role of nutrient cycling, and one that decomposes slowly (or recalcitrant carbon), which reflects more the storage of carbon in the system. So are we storing fast, labile carbon which drives nutrient cycling? Or, is it slow, non-labile carbon that promotes long-term storage and sustainability of SOC, as well as physical stability of the soil? Land managers can enhance SOC storage by increasing overall organic matter input, the partition of recalcitrant carbon, and the formation of soil aggregates (Post et al. 2004).

Figure 4 shows the simulated trends of the recalcitrant carbon pool for the three strategies since their introduction around 1991. The amount of recalcitrant carbon for STC-NTB is highest and is steadily increasing throughout the simulation period ranging from 137 to 158 g C/m^2. The current stocks of the recalcitrant carbon for FTC-NTB and NTC-FTB (128 and 84 g C/m^2, respectively) are actually less than the initial values of 137 g C/m^2. In fact, the NTC-FTB stocks are continuing to decrease (Figure 4). Although the stocks for FTC-NTB are increasing, they are increasing at a rate of only 0.42 g C/m^2/yr, while the rate of increase for the STC-NTB is 3.5 times greater (1.47 g C/m^2/yr).

One central influence on the recalcitrant carbon stocks is the amount lost due to erosion, as the slow decomposition rates limit the carbon lost due to respiration. The effect of erosion is most apparent in the trend line for the NTC-FTB strategy with the punctuated drops in 1993, 1996, and 2008 (Figure 4). During those years, heavy rains resulting in large floods occurred in Clear Creek and the Midwest. As a result, the corresponding high amounts of erosion removed large quantities of SOC. These large drops are not seen in the STC-NTB strategy, where the decreased tillage keeps a high abundance of residue on the ground to protect against rainsplash mobilization and promotes aggregate formation (Abaci and Papanicolaou 2009; Wacha 2016). In summary, the STC-NTB strategy is losing less SOC, thereby allowing it to build up in the soil column.

Profitability

To assess the profitability of the different strategies, the net income was determined as the yield profit less the production costs. The NTC-FTB strategy had the highest production, and hence the highest yield profit. However, it also had the highest average production cost of \$1191/ha/yr (Table 3). The highest net income was seen with the FTC-NTB strategy resulting from the combination of high yields and lower production costs. For comparison, FTC-NTB fields on average had 47% more profit than the STC-NTB fields.

In addition to the net income, a second indicator was determined to assess the profitability of the different strategies. The adjusted income accounts for the possibility of enhancing other ecosystem services, like lessening erosion and corresponding nutrient losses.

The NTC-FTB strategy had the highest tillage intensity, which resulted in the highest erosion and consequently the highest loss of SOC due to erosion. NTC-FTB lost, on average, more than double the soil (30 ± 20 Mt/ha/yr) and SOC (41.7 ± 48.6 g C/m^2/yr) compared to STC-NTB (12 ± 7 Mt/ha/yr and 20.5 ± 21.2 g C/m^2/yr, respectively). Comparatively, the second most intense strategy of FTC-NTB lost 54% more soil (19 ± 15 Mt/ha/yr) and 15% (23.6 ± 30.2 g C/m^2/yr) more SOC than STC-NTB (ANOVA; df = 2; F = 59.7 for erosion and 19.9 for SOC; $p < 0.001$).

To put this in different terms, in Iowa, nearly all soils have a Soil Loss Tolerance value of 11 Mt/ha/yr and the T-value provides a qualitative assessment at best over the long-term. On average, all three strategies had erosion rates greater than the T-value (Figure 5). Yet for STC-NTB almost half of the hillslopes (48%) had erosion rates less than the T-value and only 10% of the hillslopes had values more than twice the T-value. For the other two strategies, less than a quarter of the hillslopes had erosion rates below the T-value with more than a third of the hillslopes having values more than double the T-value.

High erosion depletes the soil of organic matter and other nutrients. Over time, it will require more work (which translates to higher production costs) to sustain the productivity levels in light of these higher erosion rates depleting the soil. Again, for each ton of eroded soil, the cost of lost fertilizer is $2.10 (Iowa Learning Farm 2013). On average, the NTC-FTB and FTC-NTB strategies

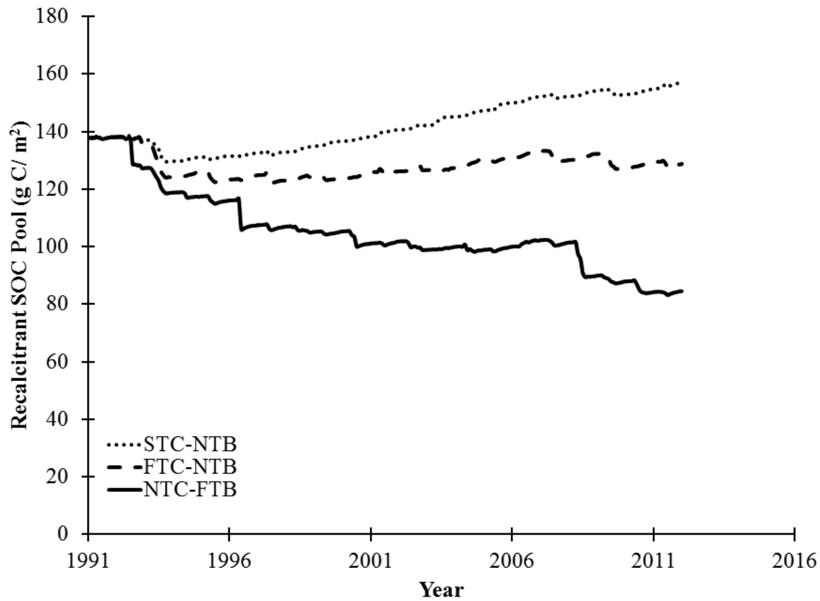

Figure 4. Changes in the recalcitrant pool of soil organic carbon. NTC-FTB = no till corn – fall till bean; STC-NTB = spring till corn – no till bean; and FTC-NTB = fall till corn – no till bean.

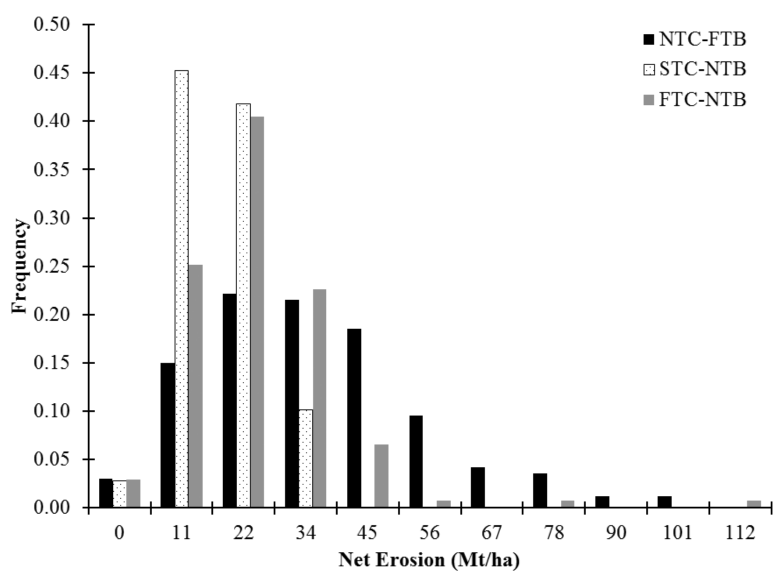

Figure 5. Net soil erosion histogram for each hillslope in Clear Creek that is practicing one of the three major crop rotations. NTC-FTB = no till corn – fall till bean; STC-NTB = spring till corn – no till bean; and FTC-NTB = fall till corn – no till bean.

lost 18 Mt/ha/yr and 6.5 Mt/ha/yr, respectively more soil than the STC-NTB strategy. This translates to a respective savings of $41.29 and $15.25 per hectare per year in added fertilizer costs, if the less intense STC-NTB is used. As fertilizer costs continue to rise (Huang 2009), this could dramatically alter the net incomes from the various practices. Moreover, it is unclear how the recent lawsuits brought by the Des Moines Water Works will translate to production costs (Eller 2015) through repercussions to the land-owners.

A second component of the adjusted income was an additional incentive payment for storing SOC. Although there is no active carbon trading currently in the U.S., the process is firmly established around the globe (World Bank Group 2014). With dwindling government assistance for BMPs in this country, a market-based approach for carbon storage could be revisited as a means of promoting conservation efforts (Eller 2014). One positive example that could be followed is the Ohio River Basin Trading Project that focuses on water quality (i.e., nutrients such as phosphorus and nitrogen) and allows permitted emitters to purchase nutrient reductions from another source (http://wqt.epri.com/).

This premium for storing SOC is set initially, for demonstration purposes, as a fixed price of $100 per ton of carbon stored in the soil per year. On average the STC-NTB strategy had the highest premium at $17,017 ± $14,240/ha/yr. For the other two more intense strategies, the average premiums were less, but only slightly. Their higher production rates helped balance the carbon losses due to erosion. However, when accounting for the savings due to less erosion and the added benefit of the premium, the differences between the income from the STC-NTB strategy and the other two strategies drop to less than 16%, and they are no longer significant (ANOVA; df = 2; F = 1.33; p = 0.26).

Sustainability

The Soil Conditioning Index (Figure 6) is an ecological/biogeochemical index that reflects the ability of a soil to respond to the external influences of erosion and land management using SOC (Papanicolaou et al. 2009). It provides a sense of how well your system is handling these external stresses, either positively or negatively. The *SCI* in the fields practicing STC-NTB had the highest average value of 0.29 ± 0.16, relative to the other two strategies (ANOVA; df = 2; F = 65.9; p < 0.001). FTC-NTB still had a positive *SCI* of 0.19 ± 0.23, but NTC-FTB had an average value of 0.00 ± 0.31. The more positive *SCI* suggests the STC-NTB fields have an improving soil condition, and are more capable of handling potential shocks to the system, whether they come from the climate or from humans (Lehman et al. 2015). The *SCI* time series in Figure 6 shows considerable variability as the crops switch between corn and soybean. This variability prevents any clear trends from being discerned in the soil condition that may suggest advancement towards sustainability, thereby limiting its utility.

The Carbon Management Index or *CMI* (Blair et al. 1995), considers two aspects of SOC. One is the level of disturbance to the SOC stock relative to a reference value (i.e., Carbon Pool Index, *CPI*). The second relates to residence time and looks at the lability of the SOC due to a disturbance (i.e., Lability Index, *LI*). Changes in SOC lability can be a measure of sustainability. Short-term productivity is driven by nutrient cycling of labile carbon; however, long-term, sustainable productivity is driven by the recalcitrant SOC levels (Post et al. 2004). Both are reflections of management and microbial activity, despite the different residence times. A balance should be maintained to keep current productivity levels high, but having an eye on future sustainable production.

The *CMI* for each strategy increased initially once reduced tillage was initiated in the watershed (Figure 6). However, the *CMI* for the two more intense strategies of FTC-NTB and NTC-FTB began to decrease in recent years. On average, the *CMI* for NTC-FTB rose initially from a value of 50 to 66 after 10 years, then it began to decrease to an average value around 60. For FTC-NTB, it took about 18 years for the *CMI* to peak at a value of 75 and then decrease. The decreasing trend is a reflection that these strategies are not sustainable despite that they may be more productive or have higher net income in the short-term. The least intense tillage schedule in the STC-NTB strategy

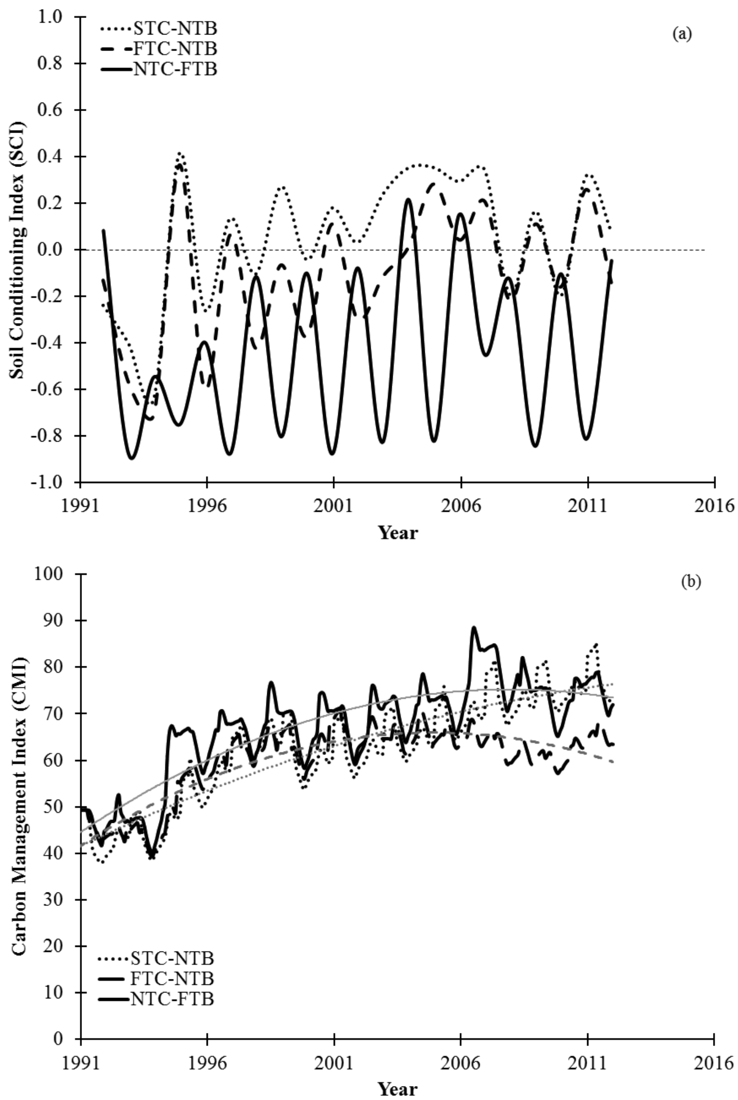

Figure 6. Sustainability figures of Soil Conditioning Index (a) and Carbon Management Index (b). The *SCI* and *CMI* values for the three main crop rotations in Clear Creek. *CMI* values closer to 100 represent a practice that is closest to the reference site of a prairie. NTC-FTB = no till corn – fall till bean; STC-NTB = spring till corn – no till bean; and FTC-NTB = fall till corn – no till bean.

helped minimize the depletion of SOC through erosion. As a result, it has the highest *CMI* at the end of the period with an average value of 77 and the trend is still increasing. In addition, despite the higher productivity from the FTC-NTB and NTC-FTB strategies, they had lesser amounts of non-labile carbon, which is needed to maintain SOC stocks over the long-term, relative to the STC-NTB strategy (Figure 4). This is most likely related to the higher erosion losses removing non-labile carbon, as well as the breaking of soil aggregates (Wacha 2016).

In summary, the *CMI* is a valuable addition to the list of indices for system functionality as it provides a clear picture of improvement within an agroecosystem. The *CMI* uncovers the underlying soil degradation occurring with some intensive management strategies, despite the fact that they are producing high yields with substantial inputs of labile SOC to the system. The added labile SOC cannot replace the more recalcitrant SOC lost with erosion. In this case it appears that the least intense strategy of STC-NTB is best for sustaining the productivity of the field.

The Role of Heterogeneity

One of the benefits of the bottom-up approach used in this study is its capability to capture heterogeneity related to different soil properties, which dictates the flowpaths of runoff, sediment, carbon, and other nutrients. As the soil and attached SOC are redistributed across the landscape, the natural heterogeneity is augmented. The importance of capturing this heterogeneity is shown using a three level analysis, which included the use of the E_{30} method, USLE, and WEPP. The E_{30} method was used as a reference condition for soil mobilization, as it only really incorporates the role of hillslope gradient. The USLE method considers not only the hillslope gradient, which is pseudo-normalized using the slope length, but also considers soil characteristics, rainfall/runoff, and management effects in a lumped sense to determine annual average erosion rates. These erosion rates are for gross erosion though, since deposition is not considered. This lumped nature of the USLE is compared against the process-based, spatially distributed WEPP model. WEPP can break the hillslope into more discrete OFEs in order to capture the depositional areas at the bottom of the downslope.

The comparison results of the three methods are shown in Figure 7 and clearly heterogeneity matters. The annual rates of net erosion determined by WEPP (average = 20 ± 16 Mt/ha/yr) are less when compared to the E_{30} (average = 31 ± 13 Mt/ha/yr) and USLE (average = 27 ± 16 Mt/ha/yr) methods (ANOVA; df = 2; F = 91.5; $p < 0.001$). We attribute this reduction in net erosion due to the deposition that occurs in the hillslopes. The E_{30} and USLE methods do not capture the beneficial effects that conservation practices have on reducing soil erosion and over-predict annual erosion rates relative to the spatially-distributed WEPP values.

For the NTC-FTB, the average net erosion rates predicted by WEPP are higher than the ones estimated by the E_{30} and USLE. Clearly the latter two methods cannot account for the dynamic within-season changes due to the absence of cover and residue during the high storm-intensity period of May – June in Clear Creek (Abaci and Papanicolaou 2009). The 3-method simulations highlight the need for using a physically based, spatially oriented model like WEPP for addressing issues of soil quality, estimation of dynamic indices, and sustainability in intensively managed landscapes.

Summary and Conclusions

Our central objective for this study was to develop a framework that can assess how well

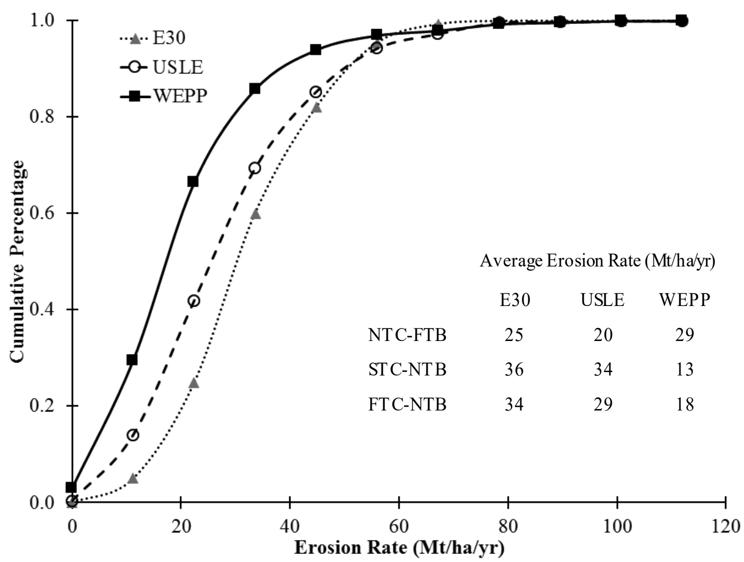

Figure 7. Cumulative frequency distribution of annual erosion rates using different methods that incorporate higher degrees of spatial heterogeneity: E30, USLE, and WEPP. NTC-FTB = no till corn – fall till bean; STC-NTB = spring till corn – no till bean; and FTC-NTB = fall till corn – no till bean.

land management strategies balance short-term productivity and profitability with the long-term sustainability of the agroecosystem. Our presented framework uses a bottom-up approach that integrates two established, process-based models, namely WEPP and CENTURY, with an economic assessment that considers the production of ecosystem services related to food security and soil organic carbon. The bottom-up approach allows us to capture better the overall effects of spatial heterogeneity, in terms of hydropedological properties, land use/cover, and hillslope curvature, on erosion for different event, seasonal, and inter-annual periods.

Most biogeochemical models (including CENTURY) are limited to the soil column and cannot adequately reproduce the effects that spatial heterogeneity across the landscape, organizational complexity, soil erosion/deposition, and land-use management practices have on SOC stocks. These drawbacks have direct implications on the design of efficient economic policies for providing incentive payments or credits to potential producers. WEPP applies a landscape-oriented approach to CENTURY, and allows us to consider physical mechanisms affecting SOC storage along a hillslope, including erosion and deposition, management schedules for tillage and fertilizer, and tiles which affect carbon stocks at depth. Hence, our framework allows us to improve our carbon budget predictions by adding a correction to the NEE equation that accounts for erosion directly and indirectly through changes in respiration and production.

However, it is not enough to just account for erosion. Herein we show the potential errors in accounting for erosion through the use of lumped models that neglect the redistribution of soil (and associated SOC). The E_{30} methods and USLE methods do not capture the beneficial effects that conservation practices have on lessening soil erosion and over predict annual erosion rates compared to the WEPP derived erosion rates.

Finally, our framework used the Carbon Management Index, *CMI*, which provided the clearest picture of improvement within an agroecosystem. The *CMI* captures both the level of the management disturbance through the loss of SOC compared to a reference value, and a reflection of the type of carbon, in terms of residence time, being stored in the system. In Clear Creek, it appears that the least intense strategy of STC-NTB is best for sustaining the productivity of a field. This is seen through the combination of the Carbon Pool Index, *CPI*, which captures the level of the management disturbance through the loss of SOC compared to a reference value, with the Lability Index, *LI*, which reflects the type of carbon being stored in the system.

The "take-home" message from this study is that when developing future land management strategies for agroecosystems, it is important to consider those practices that promote the production of organic matter in the soil. SOC is inherently tied to nearly all key soil functions in some fashion. Current practices are geared for short-term high production. However, there is a high level of soil degradation associated with these practices that threaten the long-term sustainability of our agroecosystems.

In this study, it was shown that no-till or reduced tillage practices allow SOC to build back in the soil after years of high erosion. Additionally, cover crops can provide seasonal protection from raindrop impact and runoff, as well as increase soil organic matter as a form of green manure. Multiple crop rotations have long been seen as a way to avoid depleting the soil of nutrients and improving soil resiliency against pest propagation. The mixing of legumes with a high nitrogen-demanding crop, followed by a low nitrogen-demanding crop, can help reduce fertilizer applications and production costs. The application of manure may be another way of reducing fertilizer costs, as well as providing additional soil health benefits. Finally, the use of precision farming can target only those areas needing extra work and can help minimize production costs, while potentially limiting water quality problems.

These practices can be readily implemented by the farmers as they rely more on the farmer rather than on institutional help. BMPs, such as grassed waterways and riparian buffer strips, can be beneficial but they are more focused to minimize water quality concerns, as opposed to soil health concerns. In addition, funding for these practices is dwindling away. Therefore, when developing an outreach message for farmers and all other stakeholders in the decision-making tree of the agroecosystem, it should be stressed that implementing SOC-enhancing practices are the best bet to improve system functionality, and hence system health.

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