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Soil health changes from grassland to row crops conversion on *Natric Aridisols* in South Dakota, USA

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ABSTRACT

In the northern Great Plains vast amounts of native grassland have given way to crops, mostly small grains and corn, over the past half century. It is well understood that over the long-term, grassland conversion accelerates erosion and generally decreases many soil functions. It is less clear, however, what short-term effects occur to the soil from grassland conversion; after the first or second year of conversion. The objectives of this study were to assess the short-term (first year) effects of converting land that is considered long-term grassland to small grain production through either conventional tillage (CT) or no-till (NT) practices using various indicators of soil health and to demonstrate how tools such as the Comprehensive Assessment for Soil Health (CASH), can be used to document soil health indicator decline immediately upon conversion from grassland to small grains. The CASH offers a suite of chemical, physical and biological soil tests to broadly assess soil health. In general, these indicators showed a more rapid decline in soil health under CT than NT. After the first year of grassland conversion, aggregate stability declined by 7% and 19% in the NT and CT plots, respectively when compared to the grassland control. Likewise, CT produced significantly greater declines in permanganate oxidizable carbon (POX-C) and soil protein (ACE-Protein), particularly under reduced precipitation. This study highlights how the CASH can provide an intuitive framework for monitoring the effects of land use change and can be used by land managers to identify potential soil constraints and formulate potential interventions.

1. Introduction

The northern Great Plains (NGP) landscape is a mosaic of land uses. The NGP comprises 24% of the farmland and nearly 30% of range and pastureland in the United States (USDA, 2018). As a land use priority, grasslands and row crop acreage are often at odds. Indeed, temperate grasslands are commonly thought of as one of the most threatened biomes globally – risking the loss of an extremely biodiverse ecosystem and habitat to numerous threatened and endangered species (Hoekstra et al., 2004). Grassland displacement for crop production has occurred rapidly over the last half-century, where an estimated 60% of all native mixed grass prairie in South Dakota, North Dakota, and Montana have been converted to cropland (Higgins et al., 2002). Between 2008 and 2012, nearly 3 million hectares of previously uncultivated land was transitioned to cropland nationwide (Lark et al., 2015). Of this 3 million hectares, 77% was converted from grasslands, located largely in the NGP. Recent research found that as much as 5% of the entire NGP grassland was being converted to cropland each year during this same time period (Wright and Wimberly, 2013). While this trend has slowed somewhat since the period of this study, it still remains a critical issue (Gage et al., 2016).

Land use change is a dynamic process across the NGP – driven by economic forces often underwritten by governmental policies. Programs such as the Sodbuster Provision and the Conservation Reserve Program (CRP), authorized within Farm Bills, significantly influence the extent of land that is brought into or out of production in any given time period. These programs disincentivize converting grassland to row crops by providing monetary benefits for maintaining perennial cover and protecting sensitive ecological areas. However, funding for CRP has steadily decreased, which has reduced land enrollment. Cotton and Acosta-

Abbreviations: AW, Available Water; WAS, Water Aggregate Stability; POX-C, Permanganate Oxidizable Carbon; ACE-Protein, Acid Citrate Extractable Protein; CT, Conventional Tillage; NT, No Tillage.

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Martínez (2018) noted that CRP enrollment has dropped from a high of 14.2 million ha in 1995 to 9.5 million ha in 2017 with further decreases expected through 2022. Moreover, a significant portion of this land is returning into production agriculture. For example, Lark et al. (2015) found that over a four-year period (2008–2011), 42% of all land converted to row crops was reverted from expired CRP contracts.

It is likely that this land use change has significant environmental impacts. The eastern NGP is thought to house the breeding and nesting grounds of nearly half of the nation's migratory birds. Fragmentation of grasslands can have severe, negative impacts on nest survival, which can alter the population dynamics of many different wildlife species. As this rapid change across the landscape proceeds, however, it is unclear what the impacts are to either soil health or ecosystem functioning as a whole.

The vast majority of grassland conversion across the NGP is for the purposes of growing wheat, corn and soybeans (Gage et al., 2016; Johnston, 2014; Lark et al., 2015). The method in which this land is converted is likely to have significant impacts on various soil functions. Over the long-term, it is well established that tillage negatively affects soil functioning. Tilled land was shown to decrease total soil organic matter at nearly twice the rate of no-till over twenty years of corn production following conversion from native grassland (Ismail et al., 1994). Other long-term trends comparing tillage to no-till practices include increased runoff and erosion as well as decreased nutrient retention, microbial activity and carbon sequestration (Dick, 1984; Dick et al., 1991; Halvorson and Havlin, 1992; Phillips et al., 1980; Sainju et al., 2006; Triplett and Dick, 2008).

The land manager is the ultimate determinant of soil quality and health (Doran, 2002). However, very little data exists highlighting the short-term effects of land conversion or tools that can be used by land managers to document this change. For land managers to understand the impacts resulting from land conversion, simple, comprehensive tools are necessary to provide context to management decisions. One potential tool is the Comprehensive Soil Assessment Tool (CASH), which offers a suite of standard nutrient analyses along with physical and biological tests that represent critical soil functions (Moebius-Clune et al., 2016). The framework can be used to identify physical, chemical, or biological constraints due to management decisions. However, for it to be useful to land managers, it must be responsive on short time scales.

The objectives of this study were to use the CASH to assess the short-term effects of converting land that is considered long-term grassland – similar to that in a CRP contract – to small grain production. Specifically, this experiment was designed to investigate if, and how, soil health declines are measurable immediately following grassland conversion to row crops using either tillage or no-till practices and if the CASH is a viable tool to document changes in critical soil function.

2. Methods and materials

2.1. Site description and experimental design

A field experiment was conducted at the Cottonwood Field Station (43.95° N., 101.86° W.) in western South Dakota. The site was established in 1907 by the Agriculture Experiment Station at South Dakota State University. Initially started as an agricultural station, it was converted to rangeland research in the 1940s where the study site has remained in perennial grasses. The research was conducted on an *Absted silty clay loam* soil series (Fine, smectitic, mesic Haplic Ustic Natrargids) with a sand, silt and clay content of 18%, 49% and 33%, respectively (Kettler et al., 2001), and a slope of 0–2%.

Grassland conversion took place using either conventional tillage (CT) or no-till (NT) practices. Plots with dimensions of 7.6 m. x 30.5 m. were established in a sequential manner across three growing seasons between 2016 and 2018 following a randomized complete block design with four replications. In each growing season, a new set of conversion plots was established. Precipitation and temperature for the study period were measured on-site (Fig. 1).

Following site preparation, hard red spring wheat (HRS, *Triticum aestivum* L., cv. Surpass) was planted on 25 cm row spacing with a small

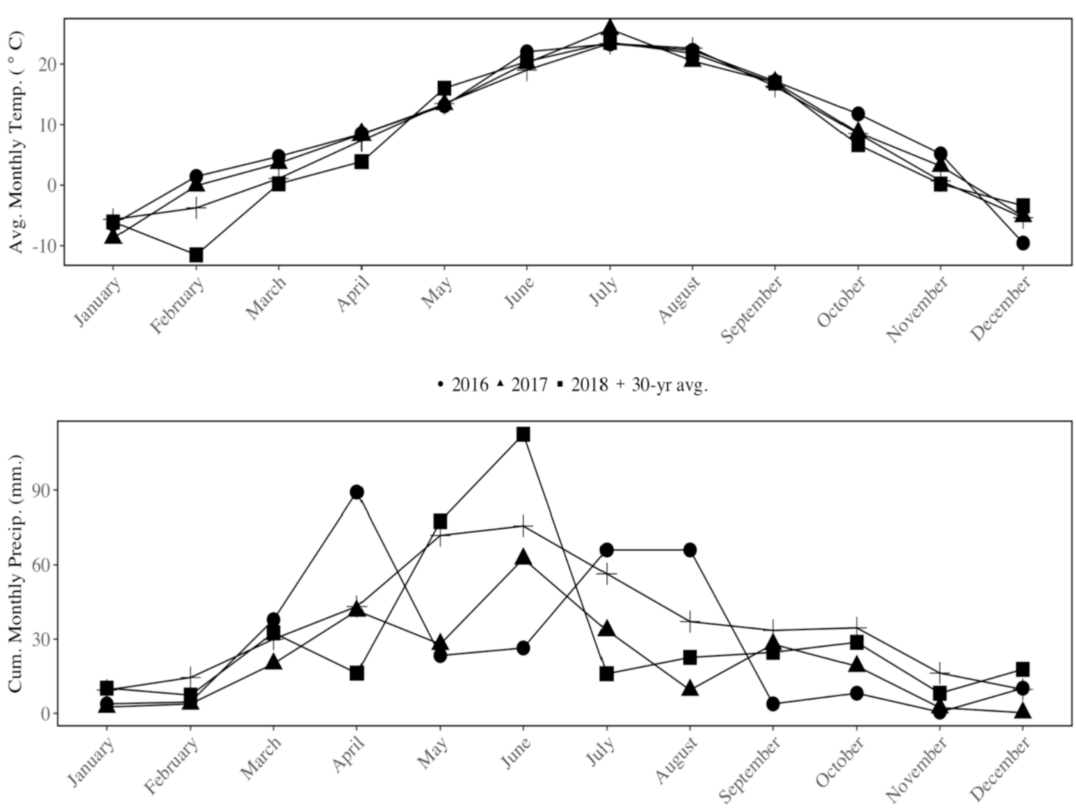

Fig. 1. Temperature and precipitation trends for the three-year study period (2016–2018) and 30-year climatological average for the study site.

grain drill (Model 750, John Deere Co., Moline, IL) at a population of 297 pure live seed m^{-2}. Planting dates were 6th April 2017, 24th April 2017, and 3rd May 2018.

For the CT treatment, sod was broken with a moldboard plow during the previous Fall to a depth of approximately 15 cm. for Spring planting. Within approximately one week prior to planting in the Spring, a second tillage pass was made using a tandem disc harrow to prepare the seedbed. The NT plots were established with a pre-plant burndown application of Roundup PowerMax [Glyphosate: N-(phosphonomethyl) glycine] (Bayer, Research Triangle, NC) and Banvel [Dicamba: (3,6-dichloro-o-anisic acid)] (Arysta, Cary, NC). A secondary application of Widematch [Clopyralid MEA salt: (3,6-dichloro-2-pyridinecarboxylic acid, monoethanolamine salt) + Fluroxypyr 1-methylheptyl ester: ((4-amino-3,5- dichloro-6-fluoro-2-pyridinyl)oxy)acetic acid, 1-methylheptyl ester] (Dow, Indianapolis, IN) was applied in HRS plots.

Fertilizers were applied at levels assumed to exceed sufficiency levels for the crop. Nitrogen was applied at planting as a mid-row band to both CT and NT as 28% urea-ammonium-nitrate at a rate of 135 kg N ha^{-1} and phosphorus, sulfur and zinc were applied with the seed as a starter application of 10–25–0-5-0.5 at a rate of 25.4 l ha^{-1}.

The Comprehensive Assessment of Soil Health (CASH) approach was selected as the standard set of metrics for which to assess soil health during grassland conversion. The focus of the current study is based on seven soil tests that can be broken down into physical, biological and chemical indicators. Soil samples were collected in the Fall following wheat harvest on 12 October, 2016, 25 October, 2017, and 23 October, 2018, respectively. All soil replicates were based on a composite of 8–12 samples taken from a depth of 0–15 cm according to Moebius-Clune et al. (2016). Following sampling, all samples were maintained at 4 °C until samples could be sent overnight to the Cornell University Soil Health Laboratory. Subsequent analysis was conducted using procedures described in Schindelbeck et al. (2016).

2.2. Physical indicators

Wet aggregate stability (WAS) is a measure of a soil aggregate's ability to maintain its integrity under simulated rainfall. This method is adapted from (Moebius, 2006; Moebius et al., 2007) in which 30 g of sieved (0.25 mm. – 2 mm) air dry soil is placed under a rainfall simulator and allowed a simulated rainfall of 12.5 mm. After wetting, the slaked soil that fell through the sieve is collected, dried and weighed. The proportion remaining on the sieve is considered the fraction of stable soil aggregates.

Available water holding capacity (AWC) was assessed as a measure of potential plant available water. In practice, AWC is defined as the difference between field capacity and the permanent wilting point. Air-dry soil was placed on ceramic plates, saturated, then subjected to a pressure of 10 kPa to determine field capacity and 1500 kPa to determine the permanent wilting point. After equilibration at the desired pressure, the soil is weighed and then dried at 105 °C and re-weighed. The AWC is then calculated as the gravimetric difference in water loss measured at the two pressures (Reynolds and Topp, 2008).

2.3. Biological indicators

Soil organic matter (SOM) was determined by calculating the loss-on-ignition after dried soil is exposed for 2 h in a muffle furnace at 500 °C. Organic matter content was determined based on the equation by Nelson and Sommers (1996).

Permanganate oxidizable carbon (POX-C) is a measure of the labile portion of organic matter (Weil et al., 2003). A 2.5 g (<2 mm) sample of air-dried soil is reacted with 20 mL 0.02 M potassium permanganate ($KMnO_4$) solution (pH 7.2). Extracts were shaken (120 rpm, 2 min), then allowed to settle for 8 min. An aliquot of solution was diluted 100 times before measurement for absorbance at 550 nm using a handheld spectrophotometer (Hach, Loveland, CO). POX-C is then calculated based on a standard curve and equations developed by Weil et al. (2003).

Autoclaved citrate extractable protein (ACE-Protein) was used as a measure of the organically-bound nitrogen in soil organic matter (Hurisso et al., 2018; Wright and Upadhyaya, 1996). Briefly, 3 g of soil was mixed with 24 ml of sodium citrate buffer (20 mM, pH 7.0) in pressure- and heat- stable glass screw-top tubes. The solution was shaken to disperse aggregates and mixed well (5 min at 180 rpm). The tubes were then autoclaved for 30 min (121 °C, 15 psi). After cooling to room temperature, an aliquot of each extract was centrifuged (10,000 xg, 3 min) and the quantity of extracted protein in solution was measured using the colorimetric bicinchoninic acid (BCA; Thermo Pierce, Waltham, MA) assay with a 96-well spectrophotometric plate reader (Bio-Tek Inc., Winooski, VT; Walker, 2002). Sample absorbance readings were calibrated using standard concentration curves of Bovine Serum Albumin.

Lastly, soil respiration was measured to assess the change in the metabolic activity of the soil microbial community. The protocol was adapted from Zibilske (1994). Briefly, 20 g of air-dried, sieved soil was re-wetted and placed in an airtight jar for four days. Carbon dioxide was captured using a beaker filled with 9 ml of 0.5 M KOH. Total respiration was determined by measuring the change in electrical conductivity (EC) of the solution with a calibrated electron probe (ThermoFisher Scientific, Inc., Waltham, MA) (Wolf et al., 1952; Wollum and Gomez, 1970).

2.4. Chemical indicators

Soil pH was measured from a suspension of 1:1 water to soil (v/v) ratio using a six-channel robotic pH tester equipped with refillable, double junction glass bulb pH electrodes (LIGNIN, LLC, Albuquerque, NM).

2.5. Statistical analysis

From east to west within the study site, soil clay content steadily decreased. Soil type ranged from silty clay loam to silt loam which created a strong east-west gradient for most variables across the study site, which makes inter-year analysis difficult due to a shifting control baseline. For example, in the final year of study (2018), which accounts for all three years of conversion, SOM in the control plots ranged from 3.96% in the 2016 conversion, 3.31% in the 2017 conversion and 2.94% in the 2018 conversion. A similar trend was observed for POX-C and ACE-Protein (Table 1).

As a result, data are presented as analysis of the full dataset along with analysis of each individual year. Data were analyzed statistically as a linear mixed-effects ANOVA model using the Kenward-Roger approximation for denominator degrees of freedom using the *lme4()* and *lmerTest()* modules (Bates et al., 2015; Kuznetsova et al., 2017) in

Table 1
Means for biologically active soil health variables from grassland control samples only. These comparisons illustrate the east to west gradient experienced as the study expanded over multiple years. This comparison is drawn from the 2018 samples taken from all three grassland control blocks to eliminate seasonal effects. The shifting control baseline makes inter-year comparison difficult, which is the basis for further analysis based on relative change from the control by tillage treatments within the same conversion block i.e., year.

Conversion Year (Field position)	SOM[§]	ACE-Protein	POX-C	Respiration
	%	*mg protein-N g^{-1} soil*	*mg C kg^{-1} soil*	*mg CO_2 kg^{-1} soil*
2016 Control (east)	3.76a[δ]	4.06a	482.00a	0.43
2017 Control (central)	3.31ab	3.55ab	451.10ab	0.43
2018 Control (west)	2.94b	3.23b	406.75b	0.39
p-value	0.09	0.06	0.05	0.27

[§] SOM, soil organic matter; ACE-Protein, autoclaved citrate extractable protein; POX-C, Permanganate oxidizable carbon. Different letters following means denote statistically different values based on Tukey's HSD test.

the R statistical package (R Core Development Team, 2014). Short-term tillage effects were the primary treatment factor of interest with the goal of generalizing the results across a range of weather outcomes. Therefore, conversion method was analyzed as a fixed effect with Year, a continuous factor with 3 levels, analyzed as a random effect. Replications were also analyzed as a random effect. For individual year analysis, conversion method was analyzed as a fixed effect with replication as a random effect.

Boxplots were used to assess the data structure and potential outliers. Residual Q-Q plots, plots of residual error versus fitted values and the Shapiro-Wilk test were applied to examine normality of residuals and any departure from homogenous variance. Significance was determined at $P \leq 0.05$ (unless otherwise stated) with means separation determined using the Tukey method in the *lsmeans* module (Lenth, 2016). Linear correlations were determined using Pearson's Correlation Coefficient.

3. Results and discussion

3.1. Weather trends

Temperatures during the study period did not vary significantly from long-term trends. However, all three years of the study were below the long-term average for precipitation - particularly in 2017. The yearly, cumulative growing-season precipitation was 77%, 61% and 83% of the long-term average for 2016, 2017, and 2018, respectively (Fig. 1). In 2016 and 2018, early season rainfall dominated and then tended toward drier weather in the fall, which was favorable for crop water status. Dry conditions persisted through most of the 2017 growing season, which received roughly 75% of the growing season precipitation of the other study years.

3.2. Physical indicators

AWC was not statistically different between the grassland control and either conversion method (Table 2). In the driest year for the study (2017), there were virtually no differences in AWC between the tillage practices. As precipitation increased, however, tillage had a greater influence and larger differences were observed (Table 3). Nunes et al. (2018) and van Es and Karlen (2019) also failed to measure significant effects of tillage management, which is mostly influenced by inherent soil differences (texture and mineralogy) and less by management effects.

AWC is a gravimetric measurement and is therefore confounded with concomitant changes in bulk density. Tillage is generally thought to increase compaction in the long-term, and it is well established that compaction decreases porosity and hydraulic conductivity (Douglas and McKyes, 1978). Conversely, a decrease in bulk density generally increases AWC (Jamison and Kroth, 1958). It is likely that in this study CT (moldboard + disc) initially did decrease the bulk density of the surface soil coming out of long-term sod, which may have been associated with a small improvement of AWC in the short-term (Nunes et al., 2019). However, further tillage generally decreases soil aggregation, surface residue and soil organic matter, all of which combine to improve soil hydraulic attributes (Franzluebbers, 2002).

In contrast, WAS, a measure of the soil's ability to resist erosion and aggregate degradation, was strongly influenced by tillage. WAS decreased through both conversion methods, but to a greater extent in the CT plots, decreasing by 7% and 19% for NT and CT, respectively (Table 2). In a study of various management practices at long-term research stations across three different regions in North Carolina and New York, van Es and Karlen (2019) and Nunes et al. (2018) also found strong WAS effects among management practices and negative relations with tillage intensity. WAS differences are often more sensitive in larger aggregate sizes (>250 μm) because small aggregates are inherently less susceptible to degradation (Beare et al., 1994). Coming out of long-term grassland, it is likely that aggregation was more evenly distributed in the current study and thus susceptible to more invasive tillage practices (Rillig et al., 2002; Rezaei et al., 2006).

3.3. Biological indicators

SOM was on average always lower in the CT than the grassland control, however no statistical differences were observed between tillage treatments for any year of the study (Table 3). Similarly, Grandy and Robertson (2006) found that total carbon changes were not immediately detectable following tillage of uncultivated land. The authors observed that soil aggregates decreased in concurrence with the current study, noting that aggregate size had decreased to a level indistinguishable to aggregate sizes measured in adjacent fields under continuous tillage for more than fifty years.

Soil organic matter (SOM) plays a stabilizing role in the soil by binding soil particles, which increases aggregate stability (Onweremadu et al., 2007; Watts and Dexter, 1997). Without proper time to rebuild OM, land that is brought into production through intensive tillage is likely to continue to leave the soil with an increased risk of long-term erosion.

POX-C is considered to be a more sensitive indicator to track such changes over various time periods. Similar to SOM, POX-C showed a downward trend with increasing tillage activity, however overall, no statistical differences were found overall (Table 2). However, the decrease was most drastic during the driest year, 2017 (Table 3). In a long-term tillage study, van Es and Karlen (2019) measured substantial tillage effects on POX-C and found the variable most strongly associated with year-averaged soybean yield (*Glycine* max L.; $R^2 = 0.93$) and corn yield (*Zea mays* L.; $R^2 = 0.85$). Culman et al. (2012) found POX-C to be strongly correlated to other measures of soil carbon, but also found that the sensitivity of POX-C to detect changes due to tillage was less robust than sensitivity to timing (Year effect) of sampling. Despite a general trend toward decreased POX-C with tillage, the current study found similar statistical results (Table 2).

In a study of ten long-term experiment stations across Europe, Bongiorno et al. (2019) concluded that POX-C was more sensitive to tillage

Table 2
Least Squared Means (LS Means) of measured soil health variables after the first growing season following conversion from grassland with analysis of variance (ANOVA) results for the main effect of 'tillage', sampled at a depth of 0–15 cm. ANOVA statistics and means represent analysis across all site-years ($n = 36$).

Tillage	AWC	WAS	SOM[§,‡]	ACE-Protein	POX-C	Respiration	pH
	$g\ H_2O\ g^{-1}$	%	%	mg protein-N g^{-1} soil	mg C kg^{-1} soil	mg CO_2 kg^{-1} soil	
Control	0.29	51.7a	3.7	4.78ab	541.8	0.59a	6.55
NT	0.29	48.1ab	3.7	4.84a	519.2	0.53ab	6.36
CT	0.30	41.8b	3.5	4.31b	500.6	0.49b	6.44
SE	0.01	5.2	0.4	0.75	69.2	0.06	0.26
p-value	0.55	0.02	0.14	0.03	0.28	0.02	0.07

[§] AWC, water holding capacity; WAS, wet aggregate stability; SOM, soil organic matter; ACE-Protein, autoclaved citrate extractable protein; POX-C, Permanganate oxidizable carbon. Different letters following means denote statistically different values based on Tukey's HSD test.

[‡] SE – standard error of the mean.

Table 3
Least Squared Means (LS Means) of measured soil health variables after the first growing season following conversion from grassland with analysis of variance (ANOVA) results for the main effect of 'tillage', sampled at a depth of 0–15 cm. ANOVA statistics and means represent analysis from each individual conversion year of the study.

Conversion Year	Tillage	AWC	WAS	SOM[§,‡]	ACE-Protein	POX-C	Respiration	pH
		$g\ H_2O\ g^{-1}$	%	%	mg protein-N g^{-1} soil	mg C kg^{-1} soil	mg CO_2 kg^{-1} soil	
2016	Control	0.3	59.4	3.9	4.75	571.3	0.65a	6.37
	NT	0.3	50.6	3.9	5.22	569.5	0.51ab	6.03
	CT	0.30	42.5	3.7	4.46	589.8	0.45b	6.23
	SE	0.01	4.9	0.1	0.33	40.3	0.04	0.15
	p-value	0.79	0.10	0.52	0.24	0.88	0.008	0.24
2017	Control	0.28	52.9	4.4	6.38a	647.3a	0.73	6.28a
	NT	0.28	55.6	4.2	5.72ab	603.8ab	0.65	6.12b
	CT	0.28	51.3	4.0	5.38b	543.3b	0.70	6.20ab
	SE	0.01	3.6	0.2	0.20	29.2	0.06	0.05
	p-value	0.89	0.44	0.21	0.02	0.04	0.42	0.05
2018	Control	0.30a	42.8	2.9	3.23	406.8	0.39	7.0
	NT	0.32ab	38.2	3.1	3.57	384.4	0.44	6.93
	CT	0.33b	31.5	2.9	3.09	368.7	0.32	6.89
	SE	0.01	3.2	0.1	0.18	23.5	0.04	0.11
	p-value	0.02	0.09	0.62	0.21	0.36	0.14	0.58

[§] AWC, water holding capacity; WAS, wet aggregate stability; SOM, soil organic matter; ACE-Protein, autoclaved citrate extractable protein; POX-C, Permanganate oxidizable carbon. Different letters following means denote statistically different values based on Tukey's HSD test.
[‡] SE – standard error of the group mean.

effects than other indicators of labile soil C, but there was strong stratification as a result of a lack of soil disturbance. Moreover, under no-till soils, macroaggregates degrade at a slower rate, which in turn allows for greater sequestration of new carbon (Six et al., 2002). Some of this retained carbon is likely to be in the form of POX-C, which is considered to be a more labile fraction of SOM but maintains a strong correlation with SOM (Table 4). POX-C was also found to be more sensitive to describing clay dispersibility than SOM (Jensen et al., 2019). Thus, it appears likely that the differences in POX-C between tillage treatments will continue to separate as further tillage takes place and may also be the most apparent in the surface layers.

Comparatively, total ACE-Protein trended slightly lower in this study when compared to larger regional studies. Fine et al. (2017) found an average ACE-Protein of 5.5 mg protein-N g^{-1} soil across a range of soils from the Midwestern United States, while our study had an average of 4.78, 4.84 and 4.31 mg protein-N g^{-1} soil for the grassland, NT and CT treatments, respectively (Table 2). With a Pearson correlation of 0.89 and 0.84, ACE-Protein correlated strongly with SOM and POX-C, respectively (Table 4). Statistically, NT was greater than the CT, while neither conversion treatment differed from the grassland (Table 2).

Soil proteins constitute the largest pool of organic N in SOM (Nannipieri and Eldor, 2009; Weintraub and Schimel, 2005). Moreover, ACE-Protein serves as an accurate indicator for this pool (Hurisso et al., 2018). Hence, these results suggest that tillage served to decrease the overall organic N pool. van Es and Karlen (2019) also measured strong tillage effects on ACE-Protein and found it the most strongly associated with year-averaged corn yield of all CASH indicators ($R^2 = 0.88$), but less correlated with soybean yield ($R^2 = 0.55$).

During the driest year (2017), ACE-Protein was decreased in both tillage treatments relative to the grassland control, but to a much greater extent in the CT plots, roughly 15% (Table 3). Despite the addition of N-fertilizer, this effect was much larger when compared to the NT plots, which received the same level of fertility.

Finally, the general decreasing trend observed in many variables with CT was similar for respiration. Respiration was significantly lower in the CT plots versus the grassland, while NT was not statistically different (Table 2). In general, there were still fairly strong correlations between soil respiration and other biological indicators (Table 4). Franzluebbers et al. (2018) found a strong correlation between the flush of CO_2 and net N mineralization. In combination with the decrease of ACE-Protein and POX-C through CT, these results indicate that N availability may be limited to a greater extent in grasslands converted to small grains through intensive tillage.

Table 4
Pearson correlation coefficient matrix for the biologically active portions of the CASH. All correlations have significance level of $p < 0.001$.[§]

	SOM	POX-C	ACE-Soil Protein	Respiration
SOM	1.00			
POX-C	0.83	1.00		
ACE-Soil Protein	0.89	0.84	1.00	
Respiration	0.89	0.68	0.79	1.00

[§] SOM, soil organic matter; ACE-Protein, autoclaved citrate extractable protein; POX-C, Permanganate oxidizable.

3.4. Chemical indicators

In contrast to other indicators, pH showed less variability across the field (Table 3). NT plots had lower pH overall (average of 6.36 versus 6.55 and 6.44 for the grassland and CT plots, respectively) and was exacerbated during the driest year of the study (Table 3). This pattern is well documented and typically ascribed to the ammonium-based fertilizers being left on the soil surface with NT, hence a strong pH stratification (Godsey et al., 2007; Reeves and Liebig, 2016). While this effect is not surprising, what was unexpected was the rate at which pH declined in the NT soils. In long-term no-till studies in Montana, (Aase and Pikul, 1995) found that pH decreased by 0.06 units yr^{-1} whereas in our study the rate of pH decline was three times as rapid. Little attention has been given to this in the context of grassland conversion. Land conversion could have a significant effect on soil pH at a large scale. Where converted land is reverting back to grassland, a reduced pH is likely to affect nutrient availability and potentially adverse effects on plant species diversity (Janssens et al., 1998).

3.5. Utility and sensitivity of soil health indicators

The different CASH indicators varied in their response to the treatment effects. However, when taken as a suite of indicators, these protocols can be used to measure soil health in relation to management effects. As Table 4 indicates, strong correlation between indicators, particularly POX-C, ACE-Protein and SOM, suggests that these measures are sensitive to short-term fluctuations in both carbon and nitrogen cycling processes to a varying extent. Despite a strong correlation between indicators, sufficient variability still exists within treatments both among and between years that statistical differences (e.g., SOM and POX-C) among tillage treatments in short-term were not detected, which is well-reported (Cooper et al., 2020; Idowu et al., 2009; Sotomayor-Ramírez et al., 2006).

Traditional chemical approaches are effective in increasing agricultural production but fail to identify soil degradation (Karlen et al., 1997). When combined, the CASH indicators serve as proxies for defining critical physical, biological and chemical properties important to agricultural production and correlated to ecosystem processes (Culman et al., 2012; Lal, 2009; Moebius-Clune et al., 2016). Assessment over multiple timepoints provides a sensitive reference for monitoring ecosystem function in response to land management decisions. Moreover, this comprehensive set of indicators can be used to identify potential soil constraints and provide insight for future interventions by land managers (Idowu et al., 2009). For example, following tillage we documented significant declines in both WAS and soil protein, which suggest rapidly declining availability of organic carbon and nitrogen (Beare et al., 1994; Hurisso et al., 2018). These indicators can serve as benchmarks for targeted studies to determine mechanistic effects of ecological functioning following invasive interventions such as grassland conversion.

In the context of the semi-arid climate of this study, this study suggests that certain indicators are more responsive during drought years. Using ACE-Protein and POX-C as examples, both CT and NT treatments remained stable relative the grassland control during seasons with greater precipitation (Table 3). Following a significant decrease in precipitation, however, both indicators were significantly lower in CT relative to the grassland. NT, in general was more resilient during this period, which is important in a semi-arid climate with erratic yearly precipitation. As both indicators are microbially mediated, it is likely that microbial dynamics play a significant role and the magnitude of change will be affected by current precipitation trends (Acosta-Martínez and Cotton, 2017; Calderón et al., 2000). Hence, these results suggest that the potential for soil health degradation during grassland conversion to row crops is likely to be more severe in dry years. However, these results only address the immediate consequences following conversion and do not address the potential ramifications over a longer time period.

4. Conclusion

Overall, it is evident that soil health potentially declines rapidly upon conversion from grassland to small grains. In general, the decline was greater under conventional tillage than no-tillage, and these effects occurred within the first year of conversion. This is an intuitive result given the well documented effects of tillage on a number of different soil health indicators; however, this study provides new data documenting how quickly these changes begin to occur.

Moreover, the CASH provides an intuitive framework for monitoring the effects of land use change and can be used by land managers to identify potential soil constraints and formulate potential interventions. These results suggest that several important soil health indicators, notably, WAS, ACE-Protein and Soil Respiration are sensitive in the short-term to conversion to row crops from grassland based on tillage method. There were, however, annual effects of treatments on some indicators such as POX-C, which may have a strong dependence on prevailing weather and/or sampling conditions.

The current study demonstrates how these measures serve as a set of interconnected and reinforcing indicators providing a basis for documenting soil degradation through tillage. The CASH can quantify the potential loss of carbon through declining soil organic matter and POX-C, declining mineralization potential through ACE-Protein and increased potential of erosion through decreased wet aggregate stability, but precipitation plays a significant role in the variability of CASH results. Further work is necessary to determine whether these indicators maintain stability in direction and magnitude under diverging climates.

Declaration of Competing Interest

None.

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References

Aase, J.K., Pikul, J.L., 1995. Crop and soil response to long-term tillage practices in the northern Great Plains. Agron. J. 87, 652–656. https://doi.org/10.2134/agronj1995.00021962008700040008x.

Acosta-Martínez, V., Cotton, J., 2017. Lasting effects of soil health improvements with management changes in cotton-based cropping systems in a sandy soil. Biol. Fertil. Soils 53, 533–546. https://doi.org/10.1007/s00374-017-1192-2.

Bates, D., Maechler, M., Bolker, B., Walker, S., 2015. _lme4: Linear mixed-effects models using Eigen and S4_. R package version 1.1-8. <URL: Http://CRAN.R-project.org/package=lme4> (accessed 04/29/2019).

Beare, M.H., Hendrix, P.F., Coleman, D.C., 1994. Water-stable aggregates and organic matter fractions in conventional- and no-tillage soils. Soil Sci. Soc. Am. J. 58, 777–786. https://doi.org/10.2136/sssaj1994.03615995005800030020x.

Bongiorno, G., Bünemann, E.K., Oguejiofor, C.U., Meier, J., Gort, G., Comans, R., Mäder, P., Brussaard, L., de Goede, R., 2019. Sensitivity of labile carbon fractions to tillage and organic matter management and their potential as comprehensive soil quality indicators across pedoclimatic conditions in Europe. Ecol. Indic. 99, 38–50. https://doi.org/10.1016/j.ecolind.2018.12.008.

Calderón, F.J., Jackson, L.E., Scow, K.M., Rolston, D.E., 2000. Microbial responses to simulated tillage in cultivated and uncultivated soils. Soil Biol. Biochem. 32, 1547–1559. https://doi.org/10.1016/S0038-0717(00)00067-5.

Cooper, R.J., Hama-Aziz, Z.Q., Hiscock, K.M., Lovett, A.A., Vrain, E., Dugdale, S.J., Sünnenberg, G., Dockerty, T., Hovesen, P., Noble, L., 2020. Conservation tillage and soil health: lessons from a 5-year UK farm trial (2013–2018). Soil Tillage Res. 202, 104648.

Cotton, J., Acosta-Martínez, V., 2018. Intensive tillage converting grassland to cropland immediately reduces soil microbial community size and organic carbon. Agric. Environ. Lett. 3, 180047. https://doi.org/10.2134/ael2018.09.0047.

Culman, S.W., Snapp, S.S., Freeman, M.A., Schipanski, M.E., Beniston, J., Lal, R., Drinkwater, L.E., Franzluebbers, A.J., Glover, J.D., Grandy, A.S., Lee, J., Six, J., Maul, J.E., Mirksy, S.B., Spargo, J.T., Wander, M.M., 2012. Permanganate oxidizable carbon reflects a processed soil fraction that is sensitive to management. Soil Sci. Soc. Am. J. 76, 494–504. https://doi.org/10.2136/sssaj2011.0286.

Dick, W.A., 1984. Influence of long-term tillage and crop rotation combinations on soil enzyme activities. Soil Sci. Soc. Am. J. 48, 569–574. https://doi.org/10.2136/sssaj1984.03615995004800030020x.

Dick, W.A., McCoy, E.L., Edwards, W.M., Lal, R., 1991. Continuous application of no-tillage to Ohio soils. Agron. J. 83, 65–73.

Doran, J.W., 2002. Soil health and global sustainability: translating science into practice. Agric. Ecosyst. Environ. 88, 119–127.

Douglas, E., McKyes, E., 1978. Compaction effects on the hydraulic conductivity of a clay soil. Soil Sci. 125, 278–282.

Fine, A.K., van Es, H.M., Schindelbeck, R.R., 2017. Statistics, scoring functions, and regional analysis of a comprehensive soil health database. Soil Sci. Soc. Am. J. 81, 589–601. https://doi.org/10.2136/sssaj2016.09.0286.

Franzluebbers, A., 2002. Water infiltration and soil structure related to organic matter and its stratification with depth. Soil Tillage Res. 66, 197–205. https://doi.org/10.1016/S0167-1987(02)00027-2.

Franzluebbers, A.J., Pershing, M.R., Crozier, C., Osmond, D., Schroeder-Moreno, M., 2018. Soil-test biological activity with the flush of CO_2: I. C and N characteristics of soils in corn production. Soil Sci. Soc. Am. J. 82, 685–695. https://doi.org/10.2136/sssaj2017.12.0433.

Gage, A.M., Olimb, S.K., Nelson, J., 2016. Plowprint: tracking cumulative cropland expansion to target grassland conservation. Gt. Plains Res. 26, 107–117.

Godsey, C.B., Pierzynski, G.M., Mengel, D.B., Lamond, R.E., 2007. Management of soil acidity in no-till production systems through surface application of lime. Agron. J. 99, 764–772. https://doi.org/10.2134/agronj2006.0078.

Grandy, A.S., Robertson, G.P., 2006. Aggregation and organic matter protection following tillage of a previously uncultivated soil. Soil Sci. Soc. Am. J. 70, 1398–1406.

Halvorson, A.D., Havlin, J.L., 1992. No-till winter wheat response to phosphorus placement and rate. Soil Sci. Soc. Am. J. 56, 1635–1639. https://doi.org/10.2136/sssaj1992.03615995005600050050x.

Higgins, J.J., Larson, G.E., Higgins, K.F., 2002. Managing tallgrass prairie remnants: the effects of different types of land stewardship on grassland bird habitat. Ecol. Restor. 18–22.

Hoekstra, J.M., Boucher, T.M., Ricketts, T.H., Roberts, C., 2004. Confronting a biome crisis: global disparities of habitat loss and protection. Ecol. Lett. 8, 23–29. https://doi.org/10.1111/j.1461-0248.2004.00686.x.

Hurisso, T.T., Moebius-Clune, D.J., Culman, S.W., Moebius-Clune, B.N., Thies, J.E., van Es, H.M., 2018. Soil protein as a rapid soil health indicator of potentially available organic nitrogen. Agric. Environ. Lett. 3, 180006. https://doi.org/10.2134/ael2018.02.0006.

Idowu, O.J., van Es, H.M., Abawi, G.S., Wolfe, D.W., Schindelbeck, R.R., Moebius-Clune, B.N., Gugino, B.K., 2009. Use of an integrative soil health test for evaluation of soil management impacts. Renew. Agric. Food Syst. 24, 214–224. https://doi.org/10.1017/S1742170509990068.

Ismail, I., Blevins, R.L., Frye, W.W., 1994. Long-term no-tillage effects on soil properties and continuous corn yields. Soil Sci. Soc. Am. J. 58, 193–198. https://doi.org/10.2136/sssaj1994.03615995005800010028x.

Jamison, V.C., Kroth, E.M., 1958. Available moisture storage capacity in relation to textural composition and organic matter content of several Missouri soils. Soil Sci. Soc. Am. J. 22, 189–192.

Janssens, F., Peeters, A., Tallowin, J.R.B., Bakker, J.P., Bekker, R.M., Fillat, F., 1998. Relationship between soil chemical factors and grassland diversity. Plant Soil 202, 69–78.

Jensen, J.L., Schjønning, P., Watts, C.W., Christensen, B.T., Peltre, C., Munkholm, L.J., 2019. Relating soil C and organic matter fractions to soil structural stability. Geoderma 337, 834–843. https://doi.org/10.1016/j.geoderma.2018.10.034.

Johnston, C.A., 2014. Agricultural expansion: land use shell game in the U.S. Northern Plains. Landsc. Ecol. 29, 81–95. https://doi.org/10.1007/s10980-013-9947-0.

Karlen, D.L., Mausbach, M.J., Doran, J.W., Cline, R.G., Harris, R.F., Schuman, G.E., 1997. Soil quality: a concept, definition, and framework for evaluation (a guest editorial). Soil Sci. Soc. Am. J. 61, 4. https://doi.org/10.2136/sssaj1997.03615995006100010001x.

Kettler, T.A., Doran, J.W., Gilbert, T.L., 2001. Simplified method for soil particle-size determination to accompany soil-quality analyses. Soil Sci. Soc. Am. J. 65, 849–852. https://doi.org/10.2136/sssaj2001.653849x.

Kuznetsova, A., Brockhoff, P.B., Christensen, R.H.B., 2017. lmerTest package: tests in linear mixed effects models. J. Stat. Softw. 82, 1–26. https://doi.org/10.18637/jss.v082.i13.

Lal, R., 2009. Challenges and opportunities in soil organic matter research. Eur. J. Soil Sci. 60, 158–169. https://doi.org/10.1111/j.1365-2389.2008.01114.x.

Lark, T.J., Meghan Salmon, J., Gibbs, H.K., 2015. Cropland expansion outpaces agricultural and biofuel policies in the United States. Environ. Res. Lett. 10, 044003 https://doi.org/10.1088/1748-9326/10/4/044003.

Lenth, R.V., 2016. Least-squares means: the R package lsmeans. J. Stat. Softw. 69, 1–33. https://doi.org/10.18637/jss.v069.i01.

Moebius, B.N., 2006. Evaluation of Laboratory-Measured Soil Physical Properties as Indicators of Soil Health.

Moebius, B.N., van Es, H.N., Schindelbeck, R.R., Idowu, O.J., Thiers, D., Clune, D.J., 2007. Evaluation of laboratory-measured soil physical properties as indicators of soil quality. Soil Sci. 172, 895–912.

Moebius-Clune, B.N., Moebius-Clune, D.J., Gugino, B.K., Idowu, O.J., Schindelbeck, R.R., Ristow, A.J., van Es, H.M., Thies, J.E., Shayler, H.A., McBride, M.B., Wolfe, D.W., Abawi, G.S., 2016. Comprehensive Assessment of Soil Health–Cornell Framework Manual, Edition 3.1. Cornell University, Geneva, NY.

Nannipieri, P., Eldor, P., 2009. The chemical and functional characterization of soil N and its biotic components. Soil Biol. Biochem. 41, 2357–2369. https://doi.org/10.1016/j.soilbio.2009.07.013.

Nelson, D.W., Sommers, L.E., 1996. Total carbon, organic carbon, and organic matter. In: Sparks, D.L. (Ed.), Methods of Soil Analysis. Part 3. Chemical Methods. SSSA Book Ser. 5. SSSA, Madison, WI, pp. 961–1010.

Nunes, M.R., van Es, H.M., Schindelbeck, R., Ristow, A.J., Ryan, M., 2018. No-till and cropping system diversification improve soil health and crop yield. Geoderma 328, 30–43. https://doi.org/10.1016/j.geoderma.2018.04.031.

Nunes, M.R., Karlen, D.L., Denardin, J.E., Cambardella, C.A., 2019. Corn root and soil health indicator response to no-till production practices. Agric. Ecosyst. Environ. 285, 106607. https://doi.org/10.1016/j.agee.2019.106607.

Onweremadu, E.U., Onyia, V.N., Anikwe, M.A.N., 2007. Carbon and nitrogen distribution in water-stable aggregates under two tillage techniques in Fluvisols of Owerri area, southeastern Nigeria. Soil Tillage Res. 97, 195–206. https://doi.org/10.1016/j.still.2007.09.011.

Phillips, R.E., Thomas, G.W., Blevins, R.L., Frye, W.W., Phillips, S.H., 1980. No-tillage agriculture. Science 208 (80), 1108–1113.

R Core Development Team, 2014. R: A Language and Environment for Statistical Computing. R Found. Stat. Comput., Vienna, Austria.

Reeves, J.L., Liebig, M.A., 2016. Depth matters: soil pH and dilution effects in the northern Great Plains. Soil Sci. Soc. Am. J. 80, 1424–1427. https://doi.org/10.2136/sssaj2016.02.0036n.

Reynolds, W.D., Topp, G.C., 2008. Soil water desorption and imbibition: Tension and pressure techniques. Soil Sampling and Methods of Analysis, 2nd ed. CRC Press, Boca Raton, FL, pp. 981–997.

Rezaei, S.A., Gilkes, R.J., Andrews, S.S., 2006. A minimum data set for assessing soil quality in rangelands. Geoderma. 136, 229–234.

Rillig, M.C., Wright, S.F., Eviner, V.T., 2002. The role of arbuscular mycorrhizal fungi and glomalin in soil aggregation: comparing effects of five plant species. Plant Soil 238, 325–333.

Sainju, U.M., Singh, B.P., Whitehead, W.F., Wang, S., 2006. Carbon supply and storage in tilled and nontilled soils as influenced by cover crops and nitrogen fertilization. J. Environ. Qual. 35, 1507–1517. https://doi.org/10.2134/jeq2005.0189.

Schindelbeck, R.R., Moebius-Clune, B.N., Moebius-Clune, D.J., Kurtz, K.S., van Es, H.M., 2016. Cornell University Comprehensive Assessment of Soil Health Laboratory Standard Operating Procedures. Available at https://blogs.cornell.edu/healthysoil/files/2015/03/CASH-Standard-Operating-Procedures-030217final-u8hmwf.pdf.

Six, J., Conant, R.T., Paul, E.A., Paustian, K., 2002. 2002 Six Stabilizatio nmechanisms of SOM implications for C saturation of soils.pdf. Plant Soil 241, 155–176.

Sotomayor-Ramírez, D., Espinoza, Y., Rámos-Santana, R., 2006. Short-term tillage practices on soil organic matter pools in a tropical Ultisol. Soil Res. 44 (7), 687–693.

Triplett, G.B., Dick, W.A., 2008. No-tillage crop production: a revolution in agriculture! Agron. J. 100, 153–165. https://doi.org/10.2134/agronj2007.0005c.

United States Department of Agriculture (USDA), 2018. Census of Agriculture, 2012 Census Volume 1, Chapter 2: State Level Data. U.S. Department of Agriculture, National Agricultural Statistics Service, Washington, DC.

van Es, H.M., Karlen, D.L., 2019. Reanalysis validates soil health indicator sensitivity and correlation with long-term crop yields. Soil Sci. Soc. Am. J. 83, 721–732. https://doi.org/10.2136/sssaj2018.09.0338.

Walker, J.M., 2002. In: Walker, J.M. (Ed.), The Bicinchoninic Acid (BCA) Assay for Protein Quantitation BT - The Protein Protocols Handbook. Humana Press, Totowa, NJ, pp. 11–14. https://doi.org/10.1385/1-59259-169-8:11.

Watts, C.W., Dexter, A.R., 1997. The influence of organic matter in reducing the destabilization of soil by simulated tillage. Soil Tillage Res. 42, 253–275. https://doi.org/10.1016/S0167-1987(97)00009-3.

Weil, R.R., Islam, K.R., Stine, M.A., Gruver, J.B., Samson-Liebig, S.E., 2003. Estimating active carbon for soil quality assessment: a simplified method for laboratory and field use. Am. J. Altern. Agric. 18, 3–17. https://doi.org/10.1079/AJAA2003003.

Weintraub, M.N., Schimel, J.P., 2005. Seasonal protein dynamics in Alaskan arctic tundra soils. Soil Biol. Biochem. 37, 1469–1475. https://doi.org/10.1016/j.soilbio.2005.01.005.

Wolf, J.M., Brown, A.H., Goddard, D.R., 1952. An improved electrical conductivity method for accurately following changes in the respiratory quotient of a single biological sample. Plant Physiol. 27, 70–80. https://doi.org/10.1104/pp.27.1.70.

Wollum, A.G., Gomez, J.E., 1970. A conductivity method for measuring microbially evolved carbon dioxide. Ecology 51, 155–156.

Wright, S.F., Upadhyaya, A., 1996. Extraction of an abundant and unusual protein from soil and comparison with hyphal protein of arbuscular mycorrhizal fungi. Soil Sci. 161, 575–586.

Wright, C.K., Wimberly, M.C., 2013. Recent land use change in the Western Corn Belt threatens grasslands and wetlands. Proc. Natl. Acad. Sci. 110, 4134–4139. https://doi.org/10.1073/pnas.1215404110.

Zibilske, L., 1994. Carbon mineralization. In: Bottomley, P.S., Angle, J.S., Weaver, R.W. (Eds.), Methods of Soil Analysis: Part 2—Microbiological and Biochemical Properties. SSSA, Madison, WI, pp. 835–863.