Assessing the effects of embedding prairie strips within row crop fields on soil hydraulic properties and soil health

by

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DEDICATION

This thesis is dedicated to my great-great-grandfather George Henning and great-grandfather Elmer Henning, who never plowed three acres of native prairie at the homestead near Kramer, Nebraska, and inspired future generations to be good stewards of the land.

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ABSTRACT

Prairie strips (PS) are an increasingly popular conservation strategy being implemented around Iowa and the Midwest, with over 14,000 acres of strips having been planted as of 2022 (nrem.iastate.edu/research/STRIPS). Foundational PS research (Phase I) occurred within the Neal Smith National Wildlife Refuge in Jasper County, Iowa. In recent years, efforts have expanded to tens of on-farm research sites around the Midwest (Phase II). By integrating native perennial vegetation within row crop (RC) fields that produce corn (*Zea mays* L.) and soybean [*Glycine max.* (L.) Merr.], prairie strips have increased pollinator and bird abundance, reduced field sediment and nutrient export, and have a favorable perception in farming and nonfarming populations. However, a thorough analysis of prairie strips' impact on many soil properties has not yet occurred.

The first study in this thesis aimed to quantify and compare soil hydraulic properties between PS and RC across locations and establishment stages. We took measurements of unsaturated and saturated hydraulic conductivity and pore size distribution with a tension infiltrometer at two Phase I sites in 2010, 2011, and 2021. Field-saturated infiltration rate and sorptivity data were acquired from six Phase II sites containing six- to seven-year-old PS with the Cornell Sprinkle Infiltrometer system. Overall, between Phase I and II sites, we found few differences in saturated hydraulic conductivity and field-saturated infiltration rate between PS and RC. The most notable and decisive difference between treatments was observed at one Phase II site, where PS field-saturated infiltration rates were 3.6 times greater than RC across three sampling periods. This site's soil type and history of topsoil degradation likely contributed to its relatively quick response to PS in saturated infiltration capacity. Comparisons of sorptivity between PS and RC treatments were more distinct than saturated infiltration capacity, as PS

sorptivity was 26 and 38% greater than RC in fall sampling periods at three Phase II sites. Since sorptivity relates to early infiltration when capillarity controls water flow, this result implies that PS can limit runoff and protect regional soil and water quality. Greater sorptivity in PS is likely due to greater evapotranspiration compared to RC during the spring and fall.

The second study investigated soil health differences between PS and RC by employing soil physical, chemical, and biological analyses. We selected three Phase II sites for a full suite of soil health testing. Additionally, increased emphasis was placed on estimating wet-aggregate stability differences between PS and RC by analyzing multiple soil depth increments, utilizing two methodologies, and including three additional sites. Across twelve soil properties, several displayed clear treatment differences or lack thereof at each site, while it became apparent that others had differing responses depending on soil type and other site characteristics. Out of four physical properties tested, the most pronounced difference between PS and RC was found in measurements of wet-aggregate stability, as PS was consistently greater than RC across all sites, depths, and testing methods. For chemical properties, extractable potassium was significantly greater in PS than RC across all sites, as the mean values were 255 mg kg^{-1} and 192 mg kg^{-1}, respectively. Soil pH was also significantly greater in PS and RC, but only at sites located within the Southern Iowa Drift Plain landform region. Treatment differences in biological properties were also limited to Southern Iowa Drift Plain sites, as measurements of soil organic matter and carbon to nitrogen ratio were greater in PS than in RC. Overall, this study showed that soil health was not definitively greater in PS than in RC across sites at the current establishment stage; however, the treatment difference observed in wet-aggregate stability may signal changes to come, given its ability to enable other soil processes.

The final study utilized two soil health indices – Cornell's Comprehensive Assessment of Soil Health (CASH) and the Soil Management Assessment Framework (SMAF) – to assess differences in PS and RC treatments. Additionally, we evaluated the utility of these scoring indices in the context of PS and RC treatment comparison. Both CASH and SMAF use scoring functions to transform observed values of soil health indicators into unitless scores ranging between 0 and 100. Scores generated for each indicator can then be integrated to produce an overall soil health score. CASH and SMAF agreed that PS had marginally greater overall soil health than RC across Southern Iowa Drift Plain sites. This difference was statistically significant for CASH, while lack of replication limited statistical analysis of SMAF scores. It was apparent that greater wet-aggregate stability in PS than in RC drove the treatment difference observed in CASH overall scores, and it is likely that wet-aggregate stability is a leading indicator of overall soil health improvement due to PS. Soil health scoring indices provided value to PS and RC soil health comparison by supplying a framework to integrate multiple soil properties into a single assessment and easing the interpretation of observed values. However, it was clear that overall soil health scores, individual indicator scores, and observed values should be used to supplement each other to perform the most accurate and complete assessment. Additionally, inherent soil quality must supplement soil health scores if productivity assessments are desired.

CHAPTER 1. GENERAL INTRODUCTION

The state of Iowa has lost at least 99% of its historical 12.5 million hectares of tallgrass prairie over the past two centuries (Samson & Knopf, 1994). This dramatic land use transformation and its associated effects have introduced widespread environmental concerns such as biodiversity loss and soil and water quality deterioration. Recent efforts to mitigate these environmental issues have taken many forms, including the targeted re-establishment of perennial native prairie vegetation. In 2007, the Science-based Trials of Rowcrops Integrate with Prairie Strips (STRIPS) project was implemented at the Neal Smith National Wildlife Refuge in Jasper County, Iowa (referred to as Phase I) to investigate the potential multifunctional benefits of strategically embedding prairie strips within corn (*Zea mays* L.) and soybean [*Glycine max.* (L.) Merr.] row crop fields. This initial research effort yielded promising results spanning biological, hydrological, financial, and social outcomes (Schulte et al., 2017). Since then, STRIPS research has expanded to over 60 privately-owned on-farm research sites (referred to as Phase II) across Iowa and the Midwest. Additionally, with the amendment of Conservation Practice 43 – Prairie Strip to the 2018 U.S. Farm Bill's Conservation Reserve Program, thousands of new prairie strip acres have been established nationwide.

As the implementation of prairie strips has grown, new research opportunities have followed. The study of soil responses to prairie strip establishment across varying landscapes garners investigation as soil management effects often vary by the factors of soil formation described by Jenny (1941): climate, biota, relief, parent material, and time. Although numerous studies have analyzed soils under prairie restoration (Jastrow et al., 1998; Baer et al., 2002; Matamala et al., 2008; Chandrasoma et al., 2016), the positioning of prairie strips directly within the row crop landscape distinguishes them from traditional prairie reconstructions. To date, only two peer-reviewed studies have explicitly focused on soil properties' response to prairie strips (Gutierrez-Lopez et al., 2014; Pérez-Suárez et al., 2014), and both utilize data solely from the initial Phase I experiment.

In naturally drained landscapes, the soil's ability to infiltrate water can play a central role in sustaining soil and water quality by mitigating sediment transport, surface runoff quantity, and nutrient export. While inherent soil properties influence infiltration dynamics (Brady & Weil, 2008; Thompson et al., 2010), land cover and management significantly contribute as well. Native prairies – characterized by their expansive root systems, high soil organic content, and minimal soil disturbance – possess a greater ability to infiltrate water than most of their agroecosystem counterparts (Fuentes et al., 2004; Stone & Schlegel, 2010). Several studies have analyzed the extent to which prairie restoration can recover pre-agricultural infiltration rates, and a broad spectrum of results has been reported. Bharati et al. (2002), Udawatta et al. (2008), and Alagele et al. (2019) all found that after 10-12 years, land seeded to native prairie species had significantly greater infiltration than comparable row crop land. Conversely, Anderson et al. (2020) and Pey and Dolliver (2020) reported that infiltration in retired land seeded to a native grassland mix did not differ from row crop infiltration after more than ten years.

Although infiltration dynamics are critical to consider when analyzing soil responses to prairie strips, they are just one component of a broader picture. Soil health is defined as "the capacity of a soil to function within ecosystem and land-use boundaries to sustain biological productivity, maintain environmental quality, and promote plant and animal health" (Doran & Parkin, 1994; Bünemann et al., 2018). In the Midwest, long-term cultivation has led to a decline relative to native prairie in many soil health indicators, including compaction (Murphy et al., 2004), nutrient retention (Burke et al., 1995; Brye et al., 2002), and soil organic carbon

(Schlesinger, 1986; Knops & Tilman, 2000; Thaler et al., 2021). While prairie restorations have been reported to increase several soil health indicators (Allison et al., 2005; De et al., 2020; Li et al., 2021), the demonstrated ability of prairie strips to act as sediment and nutrient filters could alter their soil health response (Helmers et al. 2012; Hernandez-Santana et al., 2013; Zhou et al., 2014). Recently, efforts have been made to quantify soil health assessment by translating observed values of soil health indicators into unitless scores (Andrews et al., 2004; Moebius-Clune et al., 2016). These developments provide an easily-interpretable framework to integrate and assess multiple soil health indicators.

The objectives of this thesis were to:

1. Quantify changes in water infiltration properties induced by prairie strips at varying locations and establishment stages.
2. Analyze soil health under prairie strips using a suite of soil physical, chemical, and biological properties.
3. Use soil health scoring indices to compare prairie strip and row crop soil health and assess the functionality of index scoring in this context.

Thesis Organization

Chapter 2 fulfills Objective 1 by comparing prairie strip and row crop field-saturated infiltration rate and sorptivity at six Phase II sites over three sampling periods. Additionally, measurements of saturated and unsaturated hydraulic conductivity and macroporosity in prairie strips and row crops at Phase I sites are included. Chapter 3 summarizes physical, chemical, and biological soil parameters measured at three STRIPS2 sites in 2021. Increased emphasis was placed on wet-aggregate stability as additional sites, methods, and sampling periods were included, along with measurements covered by the 2021 suite. Chapter 4 assesses differences between prairie strips and row crops in soil health indicators and overall soil health as characterized by the Soil Management Assessment Framework (SMAF) and Cornell's Comprehensive Assessment of Soil Health (CASH). Further, the values and limitations of using SMAF and CASH in this context are explored. Finally, Chapter 5 summarizes general conclusions made from the entirety of the thesis and provides suggestions for future research.

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CHAPTER 2. INFILTRATION DYNAMICS IN PRAIRIE STRIPS ACROSS MULTIPLE ESTABLISHMENT STAGES IN IOWA

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Abstract

The integration of native prairie vegetative strips into row crop agriculture is a promising conservation strategy that has gained momentum in adoption rates throughout the US Midwest. Previous studies have shown that prairie strip (PS) establishment can lead to several positive soil and water quality outcomes, such as reductions in surface runoff and nutrient and sediment exports. However, the impacts of PS on soil infiltration dynamics are not well known. This study utilized the Cornell Sprinkle Infiltrometer system to measure differences between PS and row crop (RC) treatments in field-saturated infiltration rate and sorptivity at six sites across Iowa over a two-year span. Additionally, hydraulic conductivity and pore size distribution data were generated with tension infiltrometers at two sites in 2010, 2011, and 2021. Aside from a few exceptions, differences between PS and RC were mostly undetected in measurements of saturated infiltration capacity like field-saturated infiltration rate and saturated hydraulic conductivity. However, PS sorptivity was 26 and 38% greater than RC sorptivity across three sites during fall sampling periods. Soil moisture dynamics related to evapotranspiration likely contributed more than soil structural changes to the observed differences in sorptivity. While apparent changes in saturated infiltration capacity seem to be related to site-specific

characteristics and likely occur over decadal timescales in most cases, PS sorptivity improvements should not be undervalued. Enhanced sorptivity within PS can increase early infiltration and limit runoff generation, mitigating sediment transport and improving regional soil and water quality.

Introduction

Since expansive European settlement of the Midwest, USA began in the mid-1800s, row crop agriculture has largely replaced native prairie ecosystems of the region. In Iowa, less than 1% of 12.5 million historical tallgrass prairie hectares remain, and corn (*Zea mays* L.) and soybean [*Glycine max.* (L.) Merr.] croplands account for approximately 68% of the state's land cover (Sampson & Knopf, 1994; USDA, 2022). In recent decades, interest in efforts to re-establish portions of the native ecosystem has grown. In 2007, the Science-based Trials of Row-crops Integrated with Prairie Strips (STRIPS; https://www.nrem.iastate.edu/research/STRIPS/) project was established at the Neal Smith National Wildlife Refuge in Jasper County, Iowa, and STRIPS research has since expanded across the Midwest. Studies from this project have investigated an array of implications associated with the strategic conversion of 10-20% of crop field land area to native prairie vegetation in the form of contour and foot slope strips.

Although multiple findings of soil and water quality improvements in response to prairie strip establishment have been a principal aspect of this research effort (Helmers et al. 2012; Hernandez-Santana et al., 2013; Pérez-Suárez et al., 2014; Zhou et al., 2014), questions remain surrounding the impacts of prairie strip establishment on water infiltration dynamics (Lockett, 2012; Brittenham, 2017). In naturally drained landscapes, increased infiltration has the potential to mitigate multiple soil and water concerns such as surface runoff quantity, nutrient loss, and sediment transport. Several studies have shown that remnant native prairies possess a significantly greater ability to infiltrate water than cropped systems (Fuentes et al., 2004; Stone & Schlegel, 2010). Greater infiltration within prairies is tied to enhanced soil macroporosity, as the abundance and distribution of macropores within the soil profile fundamentally controls saturated hydraulic conductivity (Brady & Weil, 2008). Both intrinsic and extrinsic factors influence soil macroporosity, including soil texture, organic matter (OM), disturbance, and biological activity.

Several factors suggest that prairie restoration efforts such as prairie strip establishment should improve water infiltration compared to row crop agriculture. First, common agricultural occurrences like tillage and wheel traffic disrupt soil structure and can impede hydraulic conductivity (Ankeny et al., 1990), whereas, in prairie strips, these disturbances do not occur. Additionally, several chronosequence studies have shown that prairie restorations can replenish OM to pre-cultivated levels on a decadal timescale (McLauchlan et al., 2006; Matamala et al., 2008) and that increases in soil organic carbon, a component of OM, occur most rapidly in the beginning stages of re-establishment (Bugeja & Castellano, 2018). Organic matter enhances soil structure and contributes to pore distributions favorable to infiltration (Boyle & Frankenberger, 1989; Franzluebbers 2002). Plant root and fungal hyphae growth increases during prairie restoration and can create water transport channels (Jastrow et al., 1998; Wu et al., 2017). However, it should also be noted that root growth's influence on infiltration is temporally variable as it has the potential to clog channels at different physiological stages (Gish & Jury, 1983; Liu et al., 2019).

While abundant evidence exists to support expected improvements in infiltration as a result of prairie strip establishment, previous research has not provided a consistent explanation of how these changes might occur over time and space. Both Bharati et al. (2002) and Alagele et al. (2019) found that infiltration under switchgrass (*Panicum Virgatum* L.), a native tallgrass prairie species, was significantly greater than under row crop less than ten years after establishment. Additionally, Udawatta et al. (2008) reported greater saturated hydraulic conductivity and macropores per unit area in a 12-year-old restored prairie compared to a row cropping system in Missouri. However, Anderson et al. (2020) found no differences in water infiltration between at least 10-year-old Conservation Reserve Program (CRP) grassland and row crop fields in Arkansas, and Pey and Dolliver (2020) predicted that retired land in Minnesota seeded to a native grass mix would take 128 years to fully recover its pre-cultivation infiltration rate.

The goal of this study was to characterize infiltration responses to prairie strips embedded in row crops across multiple establishment stages and locations in Iowa. Two different systems – the Cornell Sprinkle Infiltrometer and the Tension Infiltrometer – were employed to measure infiltration parameters such as field-saturated infiltration rate, sorptivity, hydraulic conductivity, and macroporosity. Study locations varied by soil type, agricultural management practices, and time since prairie strip establishment. Additionally, testing occurred over multiple years and seasons to account for temporal variation in infiltration dynamics and analyze any potential trends occurring over time. We hypothesized that the establishment of prairie strips would increase soil macropores over time and enhance the soil's ability to infiltrate water.

Materials and Methods

Site Descriptions

Infiltration experiments were carried out at eight sites located across the state of Iowa. Six sites – ARM, HOE, MCN, RHO, WHI, and WOR – were 100% row crop (RC) fields until prairie strip (PS) establishment between 2014-2015. These sites are referred to as Phase II sites.

The remaining two sites – IN1 and WE2 – are located within the Neal Smith National Wildlife Refuge (NSNWR) in Jasper County, Iowa, and are considered Phase I sites. Phase I sites were under bromegrass for at least ten years prior to 2007, when they were converted to RC fields containing PS. All eight sites are located within either the Des Moines Lobe or Southern Iowa Drift Plain landform region (Figure 2.1). The Des Moines Lobe is characterized by a nearly level to gently rolling landscape with deep, loamy soils and highly productive cropland. Dominant soil orders are Mollisols and, to a lesser extent, Alfisols and Inceptisols (NRCS, 2006). The Southern Iowa Drift Plain is a mostly rolling to hilly region covering a large swath of Iowa's southern half. Mollisols and Alfisols, along with some Entisols, make up the dominant soil orders of the region (NRCS, 2006). While each site has been in corn (*Zea mays* L.) and soybean (*Glycine mas.* (L.) Merr.) production in recent years, several notable contrasts in management exist (Table 2.1). ARM, WHI, IN1, and WE2 are all managed using no-till farming practices, while HOE, MCN, RHO, and WOR are conventionally tilled. Additionally, beef cattle graze at MCN and RHO following the corn phase.

Sampling Locations

At Phase II sites, sampling locations were determined using USDA-NRCS Web Soil Survey data. We randomly selected three replications of paired PS and RC sampling points within each soil series and phase present at each site. For PS points, sampling was performed in the center of the strip. For RC points, sampling was performed directly upslope of PS points, 3 m upslope from the PS edge.

At Phase I sites, a similar procedure was used to determine paired sampling points; however, landscape position was the basis for selection rather than soil series and phase. Two landscape positions were identified within each site – summit and footslope – and two repetitions of paired PS and RC points were randomly placed at each landscape position. PS points were placed 3 m into the strip from its upslope edge, and RC points were placed 3 m upslope of that edge.

Paired treatment points were sampled on the same day so that environmental conditions remained consistent, ensuring an accurate comparison between PS and RC treatments. Additionally, to avoid the influence of mechanical compaction, testing never occurred within obvious wheel tracks.

Cornell Sprinkle Infiltrometer

We measured field-saturated infiltration rate (FSIR) and sorptivity with the Cornell Sprinkle Infiltrometer system (Ithaca, NY) at Phase II sites. We collected data at ARM, RHO, and WOR in fall 2020, summer 2021, and fall 2021, and at HOE, MCN, and WHI in summer 2021 only. Briefly, the Cornell Sprinkle Infiltrometer procedure involved simulating rainfall over a metal ring inserted into the soil and measuring the subsequently observed rainfall and runoff rates to calculate the infiltration rate (Figure 2.2). Specific information regarding the equipment and its operation is outlined in van Es and Schindelbeck (2015).

Per van Es and Schindelbeck (2015) recommendations, simulated rainfall rates were maintained near 0.5 cm min^{-1} for the duration of the wetting period at each sampling point. Rainfall rates were calculated by measuring the height of water in the infiltrometer every three minutes (Equation 2.1).

$$r = \frac{(h_1 - h_2)}{t_f} \qquad (2.1)$$

where r is the simulated rainfall rate (cm min^{-1}), h_1 is the water height (cm) at the beginning of the time interval, h_2 is the water height (cm) at the end of the time interval, and t_f is the time elapsed between height measurements. Simultaneous to measurements taken for rainfall rates, the volume of generated runoff was also measured at three-minute intervals to calculate runoff rates (Equation 2.2).

$$ro_t = \frac{V_t}{A * t} \tag{2.2}$$

where ro_t is the runoff rate (cm min^{-1}), V_t is the runoff volume (cm^3), A is the soil surface area within the metal ring (457.30 cm^2), and t is the time elapsed between runoff volume measurements. We concluded Cornell Sprinkle Infiltrometer operation at each sampling point once measured runoff volumes were within 10 mL of each other for three consecutive time intervals, indicating that steady-state conditions had been achieved. The infiltration rate was calculated as the difference between rainfall and runoff rates (Equation 2.3).

$$i_t = r - ro_t \tag{2.3}$$

Rainfall and runoff rates for each time interval were smoothed by averaging each rate measurement with its preceding and subsequent measurement to resolve inconsistencies in simulated rainfall rates under field conditions. Initial and final rates were not smoothed. The average of the last three measured infiltration rates was used to calculate the field-saturated infiltration rate at each sampling point. It was necessary to multiply this value by a conversion factor of 0.80 to correct for three-dimensional flow at the base of the metal ring (Equation 2.4).

$$i_{fs} = i_t * 0.80 \tag{2.4}$$

where i_{fs} is the field-saturated infiltration rate (cm min^{-1}) and i_t is the infiltration rate (cm min^{-1}). The conversion factor of 0.80 represents a ring insertion depth of 7.5 cm and a loam soil texture which best represents our soils (Reynolds & Elrick, 1990).

Additionally, we calculated sorptivity to describe the early stages of infiltration where capillarity controls flow. Estimation of this soil hydraulic property is expressed in Equation 2.5 (Kutilek, 1980)

$$S = (2T_{RO})^{0.5} * r \tag{2.5}$$

where S is sorptivity (cm min$^{-0.5}$), T_{RO} is time to runoff (min), and r is the initial rainfall rate (cm min^{-1}).

On rare occasions, runoff generation did not occur in response to the simulated rainfall rate of 0.5 cm min^{-1}. In these instances, the infiltration rate was conservatively estimated as 0.5 cm min^{-1} to reflect the complete infiltration of the simulated rainfall. In the absence of runoff generation, sorptivity was not calculated for these sampling points.

Tension Infiltrometer

Concurrent to summer 2021 Cornell Sprinkle Infiltrometer testing at Phase II sites, we estimated soil hydraulic properties of RC and PS soils with the Tension Infiltrometer at Phase I sites. Summer 2021 testing replicated the methods and locations used for the collection of 2010 and 2011 field data described in Lockett (2012). Tension infiltrometers equipped with a 20 cm diameter tension disc (Figure 2.3) were used to determine porosity and unsaturated hydraulic conductivity (K(ψ)) at tensions of -11, -5, -2, -1, and 0 cm H_2O. Tension infiltrometer testing was conducted in triplicate at each sampling point, with three infiltrometers running simultaneously. Operating procedures closely followed those detailed in Soilmoisture Equipment Corporation (2008).

Prior to the placement of the tension disc, several steps were taken at each sampling point. First, we placed a 20 cm diameter metal ring, removed any residues, and clipped vegetation within the ring area. Next, to avoid soil slaking during operation, a piece of cheesecloth was placed on top of the prepared area. Then, a thin layer of slightly moistened, fine sand was applied to the soil surface and leveled to ensure proper hydraulic contact between the soil surface and the tension disc. In some instances of extreme soil surface roughness or slope, we removed approximately 2-3 cm of soil for leveling purposes.

Once the area was adequately prepared, the tension disc was placed, and the operation of the tension infiltrometer began at the -11 cm H_2O tension. Measurements of the water level within the reservoir occurred at four-minute intervals until water level changes were within 0.2 cm for four consecutive time intervals, indicating steady-state conditions. Once measurements at the -11 cm H_2O cm tension concluded, the tension was sequentially set to -5, -2, -1, and 0 cm H_2O, following the same methodology for determining steady-state conditions at each tension. Time intervals for the -5, -2, -1, and 0 cm H_2O tensions were 2 minutes, 1 minute, 1 minute, and 30 seconds, respectively.

Steady-state infiltration rates measured with tension infiltrometers were then used to determine unsaturated hydraulic conductivity at each tension (Ankeny et al. 1991). First, the infiltration rate was converted to an infiltration flux, Q (cm^3 hr^{-1}), and applied to the Wooding (1968) equation for steady-state infiltration from a circular source (Equation 2.6)

$$Q(\psi_i) = \pi r^2 K_{sat} e^{\alpha\psi}[1 + \frac{4}{\pi r \alpha}] \tag{2.6}$$

where $Q(\psi)$ is the steady infiltrating flux (cm^3 hr^{-1}), ψ is the pressure potential at the infiltrometer disc (cm), r is the radius of the infiltrometer disc (cm), K_{sat} is the field-saturated hydraulic conductivity (cm hr^{-1}), and α is an empirical fitting parameter (Equation 2.7). K_{sat} is calculated using the Gardner (1958) equation describing an exponential relationship between hydraulic conductivity and pressure potential (Equation 2.8).

$$\alpha = \frac{\ln[\frac{Q(\psi_i)}{Q(\psi_{i+1})}]}{\psi_i - \psi_{i+1}} \tag{2.7}$$

$$K(\psi) = K_{sat}e^{\alpha\psi} \tag{2.8}$$

Pore size distribution was also determined using tension infiltrometer data. Macropores are defined as pores that drain at greater than -3 cm H_2O tension, and mesopores are defined as pores that drain between -3 and -300 cm H_2O tension (Luxmoore, 1981). Calculation of the number of pores per unit area for this study utilized the method described by Watson and Luxmoore (1986) (Equations 2.9 and 2.10)

$$r = \frac{-2\sigma cos\beta}{\rho gh} \tag{2.9}$$

where r is the pore radius (cm), σ is the surface tension of water (72.8 g s^{-2}), $\beta(^o)$ is the contact angle (assumed to be zero), ρ is the density of water (1 g cm^{-3}), g is the acceleration due to gravity (980.6 cm s^{-2}), and h is the applied tension (cm)

$$N(r) = \frac{8\mu K_m}{\pi\rho g r^4} \tag{2.10}$$

where $N(r)$ is the number of macropores per unit area, μ is the dynamic viscosity of water (0.01 g cm^{-1} s^{-1}), and K_m is the difference in conductivities between tensions.

Statistical Analysis

All statistical analysis was run using R software (R Core Team, 2020), and plots were generated with the ggplot2 package (Wickham, 2016). Data from Phase II sites was log-transformed to normalize the dataset and facilitate between-site comparison. We used a linear model testing paired differences to determine treatment effects at each site and sampling period combination, and contrasts and comparisons were determined with least-squares means (Lenth, 2020). For the ARM, RHO, and WOR sites, analysis was also performed for all sampling periods combined. Statistical significance was categorized as marginal ($p < 0.1$), significant ($p < 0.05$), and strongly significant ($p < 0.01$). Analysis of Phase I data also utilized the linear model to test paired treatment differences with least-squares means.

Results

Field-Saturated Infiltration Rate

Across all sampling periods, field-saturated infiltration rates varied widely within each site and treatment group (Table 2.2). Intrinsic soil differences led to variation among sites but were not statistically analyzed, as the differences between treatments were of primary concern.

In fall 2020, the average field-saturated infiltration rate of PS soils was greater than that of RC soils at ARM, RHO, and WOR (Table 2.2). At RHO, this difference was strongly significant ($p < 0.01$), and at WOR, it was marginally significant ($p < 0.1$). The direction of the difference between PS and RC rates was inconsistent between sites during the summer 2021 and fall 2021 sampling periods. The only statistically significant difference observed during these two sampling periods occurred at RHO in summer 2021 ($p < 0.05$), where the PS field-saturated infiltration rate was 2.63 times greater than RC. Treatment differences varied by site across all sampling periods combined for ARM, RHO, and WOR (Figure 2.4). We did not find a significant difference in field-saturated infiltration rate between PS and RC treatments at ARM and WOR. However, the field-saturated infiltration rate was 3.6 times greater in PS than RC at RHO.

Sorptivity

As the estimation of soil sorptivity using the Cornell Sprinkle Infiltrometer system varies with antecedent moisture content, the comparison of descriptive statistics between sites and sampling periods in the absence of soil moisture measurements is null. However, paired design and same-day testing of treatment pairs permit the analysis of treatment differences.

For the majority of the site and sampling period combinations, the PS treatment had greater sorptivity than RC (Table 2.3). In the fall of 2020, this difference was significant at ARM ($p < 0.05$), RHO ($p < 0.01$), and across all three fall 2020 sites combined ($p < 0.01$). During the summer 2021 sampling period, RC sorptivity was significantly greater than PS sorptivity at MCN ($p < 0.01$). Lastly, in the fall of 2021, PS sorptivity was greater than RC at WOR ($p < 0.1$) and across the three sites combined ($p < 0.05$). Across three sampling periods combined at ARM, RHO, and WOR, greater average sorptivity was observed in PS than RC at all three sites (Figure 2.5). This difference was strongly significant at ARM ($p < 0.01$).

Hydraulic Conductivity and Pore Size Distribution

Although paired design permitted the analysis of treatment differences for each tension, comparisons of raw hydraulic conductivity values between years were limited to K(0), since it represents saturated hydraulic conductivity and circumvents most effects of antecedent soil moisture. However, notable differences in hydraulic conductivity between PS and RC were relatively sparse across all site years (Table 2.4). Additionally, the magnitude of measured saturated hydraulic conductivities varied considerably between sites and years (Table 2.5 and Figure 2.5), making it difficult to detect any trends.

Only two statistically relevant treatment differences occurred in the 2021 sampling period. At IN1, K(-11) was 0.21 cm hr^{-1} less in PS than in RC ($p < 0.01$) and at WE2, K(0) was 16.18

cm hr^{-1} greater in PS than in RC ($p < 0.1$). While not always statistically significant, unsaturated hydraulic conductivities were lower in PS than RC at the smallest tensions (-5 and -11 cm H_2O) for both sites in 2021. At the higher tensions (-2, -1, and 0 cm H_2O), the general direction of treatment differences (PS - RC) varied by site as WE2 maintained positive differences and IN1 was slightly negative.

The 2021 results differed from those collected ten and eleven years prior. At IN1, the direction of treatment differences in hydraulic conductivity was inconsistent at most tensions, and no tension values had consecutive statistically significant differences between 2010 and 2011 (Table 2.4). Concerning each year individually, the only noteworthy result at WE2 in 2010 and 2011 occurred at the -5 cm H_2O tension in 2011, when PS had 2.53 cm hr^{-1} greater conductivity than RC ($p<0.01$).

Only two trends in treatment differences between 2010 and 2021 were moderately evident between the two sites. At IN1, the difference in K(-11) appears to have decreased over time, as PS had 0.11 cm hr^{-1} greater hydraulic conductivity than RC in 2010 ($p<0.1$) and 0.21 cm hr^{-1} less conductivity than RC in 2021 ($p<0.01$) (Table 2.4). On the other end of the spectrum, the treatment difference in K(0) was negative (PS<RC) at WE2 in 2010 and 16.18 cm hr^{-1} in 2021 (PS>RC) ($p<0.1$). An analysis of variance test determined that the treatment difference in hydraulic conductivity was not affected by landscape position, while interactions between site, year, and site:year occasionally occurred (Table 2.6).

Estimations of pores per unit area based on tension infiltrometer data mirrored the results of unsaturated hydraulic conductivity (Table 2.7). A greater abundance of larger pores (>0.05 cm) was correlated with higher hydraulic conductivities at the 0, -1, and -2 cm tensions. In contrast, the number of smaller pores (0.025-0.05 and 0.01-0.025 cm) was related to hydraulic conductivity at the -5 and -11 cm tensions. In general, pores of all sizes were estimated to be most numerous in 2011.

Discussion

Field-Saturated Infiltration Rate

At six to seven years post-establishment of prairie strips, differences in field-saturated infiltration rate were only evident at one of the six Phase II sites. While this study's primary objective was to determine the difference in saturated infiltration capacity between RC and PS land covers, other factors likely played a substantial role in the observed results.

Along with their location on the Des Moines Lobe landform region, the relatively higher sand content at HOE and WOR, 35% and 42% in the top 15 cm, respectively, set them apart from the other sites (Table 2.1). Under saturated flow conditions, sandier soils generally have greater hydraulic conductivities than finer-textured soils (Rawls et al., 1982). Since field-saturated infiltration rates were relatively high for both treatments at HOE and WOR compared to other sites, the effect of relatively coarse soils likely outweighed any potential effects of vegetative cover.

No-till (NT) farming practices likely contributed to the absence of differences observed between PS and RC soils at ARM and WHI. A recent review showed that NT increases water infiltration between 17 and 86% compared to conventional tillage (Blanco-Canqui & Ruis, 2018). Much like the establishment of perennial vegetation, NT farming leaves residue on the soil surface and reduces soil disturbance. These factors enhance macropore development by protecting the soil surface from raindrop impacts and increasing soil OM and biological activity (Kumar et al., 2012). While PS could provide additional mechanisms for infiltration enhancement, the impact may not be strong enough to differentiate from those shared with NT.

The remaining two sites, MCN and RHO, share several similarities, including eroded silt loam to silty clay loam soils, conventionally tilled (CT) RC fields, and fall grazing. However, only RHO displayed significant differences in saturated infiltration capacity between treatments. A possible explanation for this disparity between sites might be attributed to the different soil orders present at each site: Mollisols at MCN and Alfisols at RHO. This difference is influential with respect to saturated infiltration capacity because Mollisols possess a deeper organic-rich A horizon (topsoil) than Alfisols (Brady & Weil, 2008). Higher OM is associated with greater soil aggregation and pore size distributions favorable to infiltration (Boyle et al., 1989; Franzluebbers 2002). Since both sites have a history of erosion (Web Soil Survey), it is likely that a greater proportion of RHO's topsoil has been depleted over time, and therefore, it possesses less OM and soil macroporosity.

A combination of factors limiting macroporosity like conventional tillage, lower OM, and fine-textured soils were all present at RHO. Together, these circumstances positioned RHO to have the greatest potential for the fast and marked improvement of saturated infiltration capacity within PS relative to other sites. At the five other sites – ARM, HOE, MCN, WHI, and WOR, several components may have dampened any differences in saturated infiltration capacity between RC and PS at the current stage of PS establishment.

Sorptivity

We found that sorptivity was consistently greater in PS than RC during the fall but not summer. Since time-to-runoff was used to calculate sorptivity in this study, and it has a negative relationship with soil moisture content, it is likely that our observation of a treatment difference only being evident in the fall stems from disparities in initial soil moisture content rather than soil structural changes. Previous studies have indicated that the increased evapotranspiration

(ET) associated with perennial vegetative cover can lead to lower soil moisture content than soil in agricultural fields and that this difference is most pronounced in the spring and fall (Zhang & Schilling, 2005; Gutierrez-Lopez et al., 2014; Remigio, 2015). We could not correct sorptivity values for initial soil moisture content because we did not measure soil moisture at the time of testing. However, given the ample repetition over time and space, we can assume a wide range of initial soil moisture conditions.

While we cannot conclusively say that the PS treatment has greater sorptivity than the RC treatment when adjusted for soil moisture, our observation of greater sorptivity in PS than RC during the fall is meaningful nonetheless. A postponement in runoff generation has favorable soil and water quality conservation outcomes regardless of the mechanism causing it. Since it is likely that sorptivity differences between PS and RC arise from soil moisture and ET differences, we can deduce that PS have greater sorptivity than RC at the beginning and end of the annual growing season. Therefore, we can say that a field would generate less runoff if it contains PS than if it is 100% RC during a given rainfall event in the spring or fall.

Hydraulic Conductivity and Pore Size Distribution

Hydraulic conductivity and pore size distribution measurements were highly variable across sites and sampling years. These results corroborate literature descriptions of challenges associated with measuring field hydraulic conductivity due to spatial and temporal variability (Nielsen et al., 1971; Deb & Shukla, 2012). The inconsistency of measured hydraulic conductivity values caused difficulties in the assessment of trends.

The increase in saturated hydraulic conductivity differences between PS and RC treatments observed at WE2 over time can be attributed to a combination of previously mentioned factors like soil OM, biological activity, and disturbance. Slightly greater sand content at IN1 than WE2 may play a role in IN1 not displaying any treatment contrasts in saturated hydraulic conductivity. The decrease in differences observed for the -11 cm H_2O tension at IN1 likely relates to a greater abundance of smaller pores within the RC treatment, possibly caused by compaction. Disparities in antecedent soil moisture could factor into the hydraulic conductivity observations at the -11 cm H_2O tension. However, literature and the concurrent sorptivity analysis would suggest drier conditions within the PS treatment. Despite this, the RC treatment had greater conductivity than PS, reinforcing the notion of a greater abundance of smaller pores within the RC treatment.

Although differences in soil hydraulic properties between landscape positions have been reported in prairie and agricultural systems (Guzman & Al-Kaisi, 2011), our results indicate that landscape position did not significantly affect differences in unsaturated hydraulic conductivity between PS and RC treatments. Overall, the lack of distinction between PS and RC treatments suggests that the effects of PS on hydraulic conductivity are limited at 15 years since PS establishment.

Conclusions

This study analyzed differences in soil infiltration dynamics between prairie strips (PS) and row crops (RC) at two PS establishment stages (Phase I: 15 years, Phase II: 6-7 years). We did not find universal improvements in soil hydraulic properties due to PS; however, supporting evidence was found in specific circumstances. While saturated infiltration capacity was predominantly unchanged as a result of PS establishment, there were exceptions. Prairie strips increased saturated infiltration relative to RC at two sites (one Phase I and one Phase II). The Phase II site possessed several RC soil structure-limiting factors like tillage and low organic matter, which likely contributed to greater contrast between the two treatments at its relatively early stage of PS establishment. These results suggest that any management-induced changes in saturated infiltration capacity occur slowly, probably on at least a decadal timescale, unless certain site-specific factors are present. Contrary to saturated infiltration capacity observations, soil sorptivity improvements after PS implementation were more widespread. Differences were most pronounced in the fall, so soil moisture dynamics likely contributed more to sorptivity disparities than soil structural changes. Regardless, higher rates of early infiltration support the utility of PS as a surface runoff inhibitor and soil and water quality conservation tool. Future research should revisit infiltration dynamics of PS at times further since establishment. Additionally, since disparities between locations were evident, prairie reconstruction infiltration studies should be expanded to more locations to assess site-specific trends.

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Tables and Figures

Table 2.1. Site characteristics

Phase	Site	Dominant soil order	Sand (%)	Clay (%)	Silt (%)	Tillage[a]	2020 crop	2021 crop
II	**ARM**	Mollisol	5	30	66	NT	Soybean	Corn
II	**HOE**	Mollisol	35	28	37	CT	Corn	Soybean
II	**MCN**	Mollisol	8	32	60	CT	Soybean	Corn
II	**RHO**	Alfisol	3	23	74	CT	Corn	Corn
II	**WHI**	Mollisol	11	32	57	NT	Corn	Soybean
II	**WOR**	Mollisol	42	22	36	CT	Soybean	Corn
I	**IN1**	Mollisol	21	32	47	NT	Corn	Soybean
I	**WE2**	Mollisol	11	33	56	NT	Corn	Soybean

Note: Soil texture for 0-15 cm depth acquired from Web Soil Survey for Phase II sites and Lockett (2012) for Phase I sites.
[a]NT, no-tillage; CT, conventional tillage.

Table 2.2. Field-saturated infiltration rates in prairie strip (PS) and row crop (RC) treatments over three sampling periods at Phase II sites

Site	**Treatment**	**Fall 2020**				**Summer 2021**				**Fall 2021**			
		n	Mean (cm min^{-1})	CV (%)	Median ratio (PS/RC)	n	Mean (cm min^{-1})	CV (%)	Median ratio (PS/RC)	n	Mean (cm min^{-1})	CV (%)	Median ratio (PS/RC)
ARM	Prairie Strip	12	0.27	41	1.35	12	0.19	77	0.75	12	0.18	55	0.92
	Row Crop	12	0.22	50		12	0.21	51		12	0.23	56	
HOE	Prairie Strip	-	-	-	-	15	0.15	78	0.65	-	-	-	-
	Row Crop	-	-	-		15	0.22	48		-	-	-	
MCN	Prairie Strip	-	-	-	-	11	0.03	77	0.59	-	-	-	-
	Row Crop	-	-	-		11	0.09	89		-	-	-	
RHO	Prairie Strip	9	0.20	40	11.3***	9	0.11	77	2.63**	9	0.07	86	1.58
	Row Crop	9	0.02	84		9	0.07	89		9	0.04	87	
WHI	Prairie Strip	-	-	-	-	12	0.17	72	1.24	-	-	-	-
	Row Crop	-	-	-		12	0.10	82		-	-	-	
WOR	Prairie Strip	9	0.19	48	2.05*	9	0.08	72	0.65	9	0.15	67	0.85
	Row Crop	9	0.14	64		9	0.20	67		9	0.19	77	
All	Prairie Strip	30	0.24	42	2.90***	68	0.09	88	0.89	30	0.11	67	1.06
	Row Crop	30	0.14	86		68	0.10	72		30	0.10	77	

Note: Asterisks indicate significance of treatment difference (* $p < 0.1$, ** $p < 0.05$, *** $p < 0.01$).

Table 2.3. Sorptivity in prairie strip (PS) and row crop (RC) treatments over three sampling periods at Phase II sites

Site	Treatment	**Fall 2020**				**Summer 2021**				**Fall 2021**			
		n	Mean (cm min$^{-0.5}$)	CV (%)	Median ratio (PS/RC)	n	Mean (cm min$^{-0.5}$)	CV (%)	Median ratio (PS/RC)	n	Mean (cm min$^{-0.5}$)	CV (%)	Median ratio (PS/RC)
ARM	Prairie Strip	10	1.62	50	1.54**	12	1.64	58	1.26	12	1.35	30	1.27
	Row Crop	10	0.97	26		12	1.28	52		12	1.03	58	
HOE	Prairie Strip	-	-	-	-	14	1.47	61	1.26	-	-	-	-
	Row Crop	-	-	-		14	1.16	72		-	-	-	
MCN	Prairie Strip	-	-	-	-	11	0.77	39	0.55***	-	-	-	-
	Row Crop	-	-	-		11	1.43	35		-	-	-	
RHO	Prairie Strip	8	1.91	33	1.76***	9	0.95	27	0.84	9	1.27	45	1.15
	Row Crop	8	1.04	17		9	1.18	37		9	1.13	45	
WHI	Prairie Strip	-	-	-	-	12	1.26	72	1.28	-	-	-	-
	Row Crop	-	-	-		12	0.99	53		-	-	-	
WOR	Prairie Strip	8	1.71	53	0.95	9	1.48	51	0.99	9	1.46	34	1.39*
	Row Crop	8	1.67	34		9	1.46	46		9	1.06	47	
All	Prairie Strip	26	1.57	44	1.38***	67	1.17	60	1.01	30	1.29	34	1.26**
	Row Crop	26	1.13	39		67	1.16	54		30	1.02	47	

Note: Asterisks indicate significance of treatment difference (* p <0.1, ** p <0.05, *** p <0.01).

Table 2.4. Hydraulic conductivity mean paired treatment differences (prairie strip – row crop) at Phase I sites

Site	Year	K(0) (cm hr^{-1})	K(-1) (cm hr^{-1})	K(-2) (cm hr^{-1})	K(-5) (cm hr^{-1})	K(-11) (cm hr^{-1})
IN1	2010	-7.12	9.27	11.1**	1.05	0.11*
	2011	-18.2**	-7.77	-0.19	-1.21*	0.01
	2021	-1.92	-2.87	-1.41	-0.44	-0.21***
WE2	2010	-11.5	-7.16	-2.45	-0.06	-0.03
	2011	15.3	3.08	3.32	2.53***	0.05
	2021	16.2*	5.35	1.46	-0.17	-0.09

Note: Asterisks indicate significance of treatment difference (* p <0.1, ** p <0.05, *** p <0.01).

Table 2.5. Summary of hydraulic conductivities for prairie strip and row crop treatments combined at Phase I sites

Site	Tension (cm)	2010		2011		2021		All Years	
		Mean (cm hr^{-1})	CV (%)	Mean (cm hr^{-1})	CV (%)	Mean (cm hr^{-1})	CV (%)	Mean (cm hr^{-1})	CV (%)
IN1	-11	0.18	98	0.15	61	0.20	75	0.18	79
	-5	0.85	127	1.94	105	0.58	64	1.06	130
	-2	9.62	138	15.0	81	2.71	107	9.08	125
	-1	22.2	86	34.4	50	7.76	94	21.4	87
	0	40.1	62	62.5	33	29.2	74	44.0	60
WE2	-11	0.20	56	0.18	50	0.15	48	0.18	52
	-5	0.57	52	1.71	146	0.51	55	0.82	161
	-2	3.08	145	6.37	110	2.28	104	3.68	134
	-1	7.55	144	20.2	86	6.87	104	10.9	121
	0	14.0	104	44.1	74	20.5	79	25.4	97
Both Sites	-11	0.19	77	0.17	56	0.18	68	0.18	67
	-5	0.71	112	1.84	120	0.54	60	0.94	143
	-2	6.35	162	11.3	98	2.48	105	6.43	143
	-1	14.9	114	28.3	65	7.28	98	16.2	105
	0	27.0	89	54.5	51	24.7	78	34.9	78

Table 2.6. Analysis of variance table of effects on paired treatment differences (prairie strip – row crop) of hydraulic conductivity at Phase I sites across three sampling periods

Effect	**Ψ = 0**		**Ψ = -1**		**Ψ = -2**		**Ψ = -5**		**Ψ = -11**	
	F	**p**	**F**	**p**	**F**	**p**	**F**	**p**	**F**	**p**
Position	0.60	0.45	0.19	0.67	0.97	0.34	0.00	0.95	0.57	0.46
Site	1.90	0.18	0.25	0.78	0.39	0.68	0.75	0.49	6.43	0.01***
Year	4.11	0.06*	0.01	0.93	0.37	0.55	1.97	0.18	0.00	0.98
Site:Year	2.21	0.14	2.68	0.10*	1.97	0.17	5.68	0.01**	2.31	0.13

Note: Asterisks indicate significance level (* p <0.1, ** p <0.05, *** p <0.01).

Table 2.7. Estimated average pores per square meter at Phase I sites

Site	**Pore radius (cm)**	**2010**		**2011**		**2021**	
		Prairie Strip	Row Crop	Prairie Strip	Row Crop	Prairie Strip	Row Crop
IN1	> 0.05	31	33	41	58	13	18
	0.025-0.05	1282	339	1380	1169	171	247
	0.01-0.025	2509	435	1848	4825	563	972
WE2	> 0.05	3	18	28	41	18	7
	0.025-0.05	74	365	341	463	235	89
	0.01-0.025	742	823	4893	803	570	639

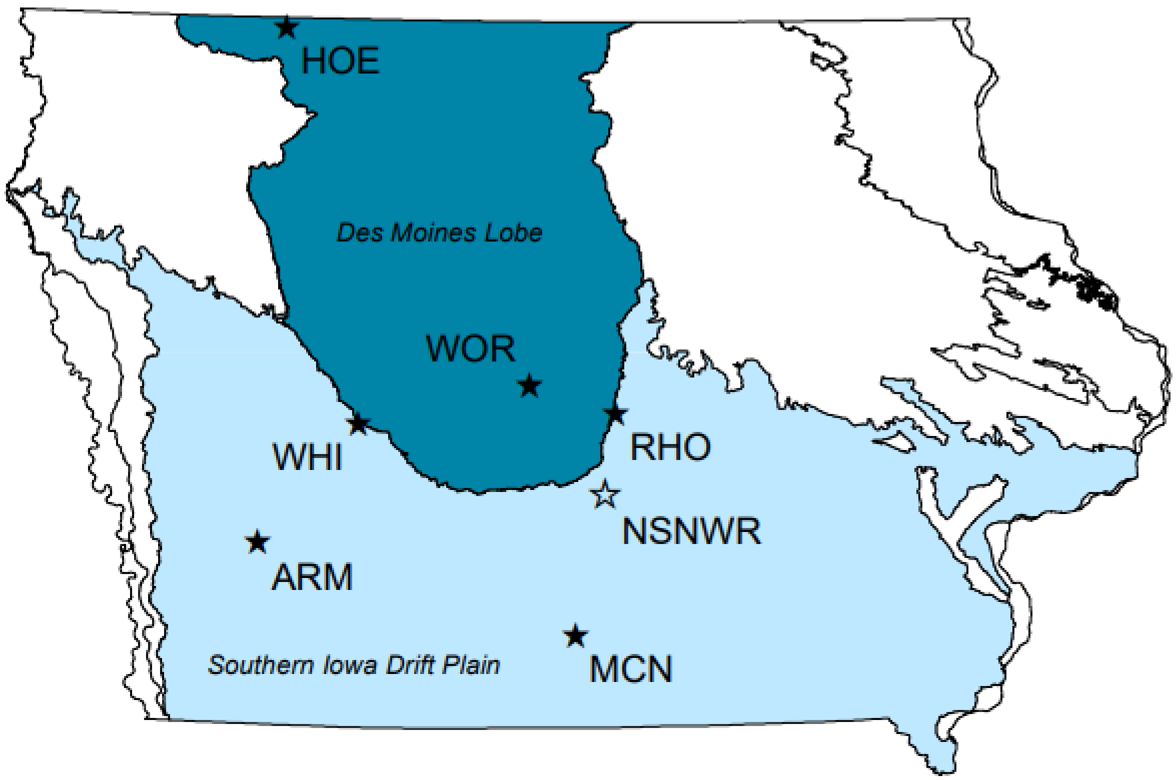

Figure 2.1. Locations of Phase I (NSNWR, open star) and Phase II sites (filled stars) in relation to Iowa landform regions

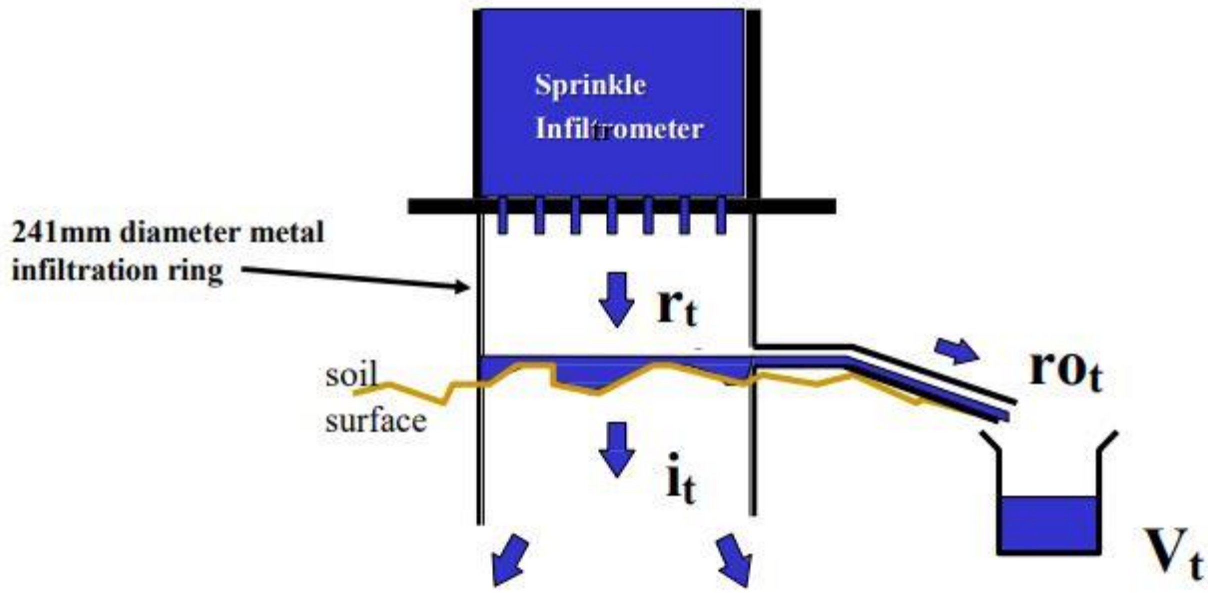

Figure 2.2. Cornell Sprinkle Infiltrometer system schematic from van Es and Schindelbeck (2015)

r_t = simulated rainfall rate

i_t = infiltration rainfall rate

ro_t = runoff rate

V_t = runoff volume

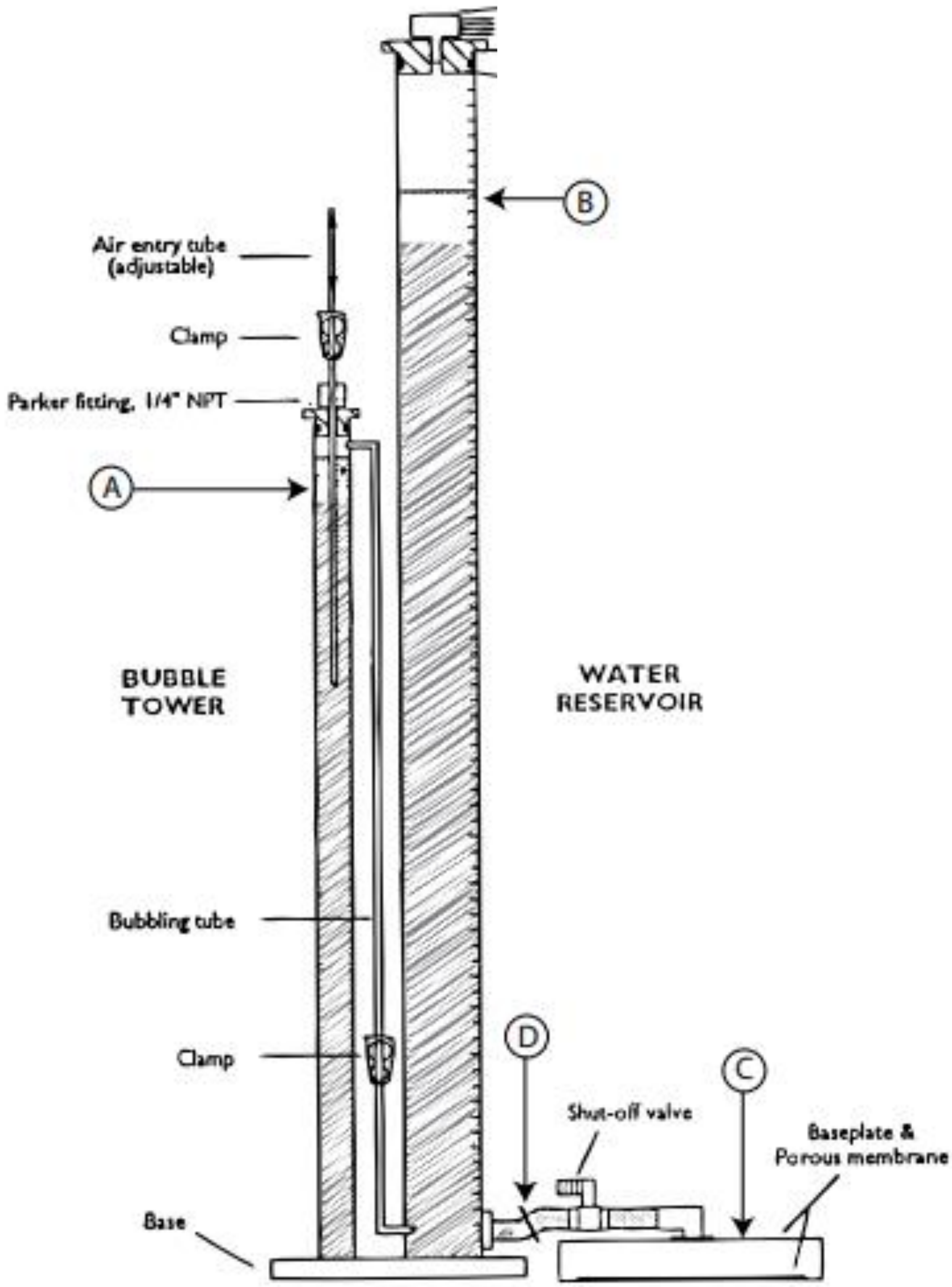

Figure 2.3. Tension Infiltrometer schematic from Soilmoisture Equipment Corporation (2008)

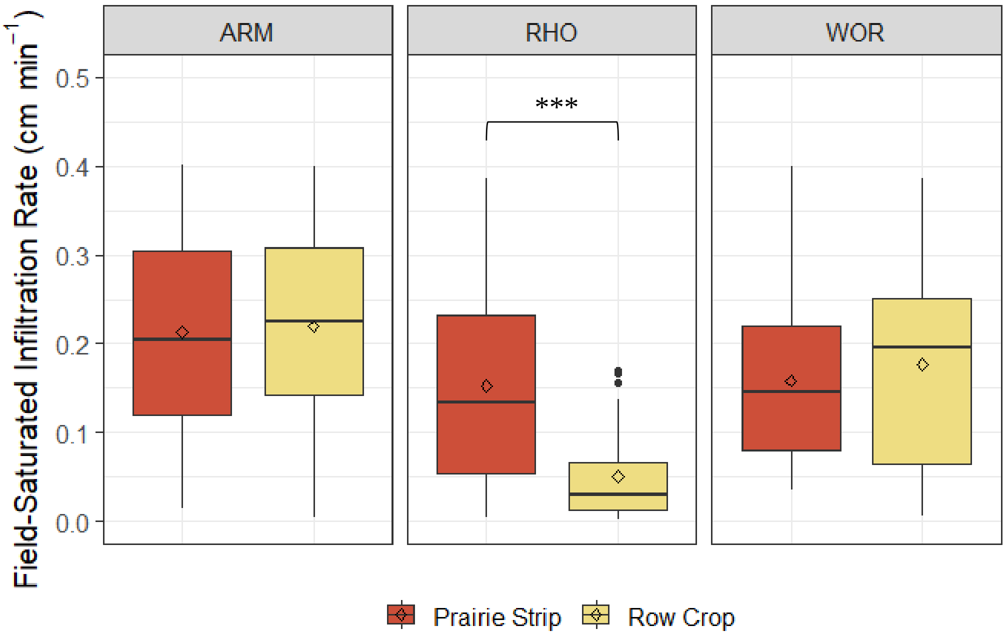

Figure 2.4. Field-saturated infiltration rates (cm min^{-1}) for ARM, RHO, and WOR sites from Fall 2020, Summer 2021, and Fall 2021 sampling periods combined

Note: Asterisks indicate significance of treatment difference (* p <0.1, ** p <0.05, *** p <0.01).

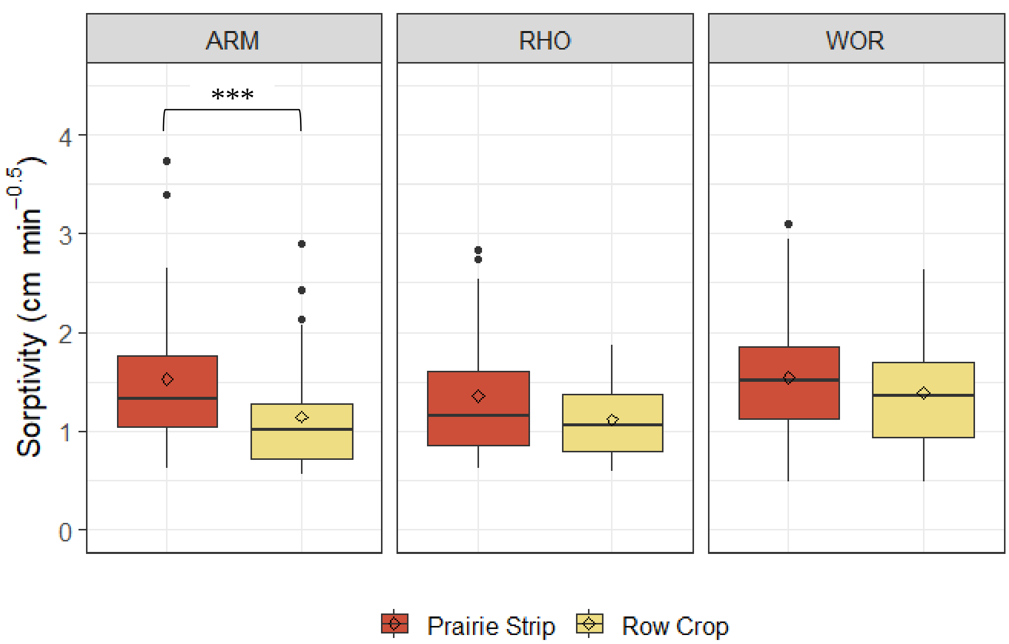

Figure 2.5. Sorptivity (cm min$^{-0.5}$) for ARM, RHO, and WOR sites from Fall 2020, Summer 2021, and Fall 2021 sampling periods combined

Note: Asterisks indicate significance of treatment difference (* p <0.1, ** p <0.05, *** p <0.01).

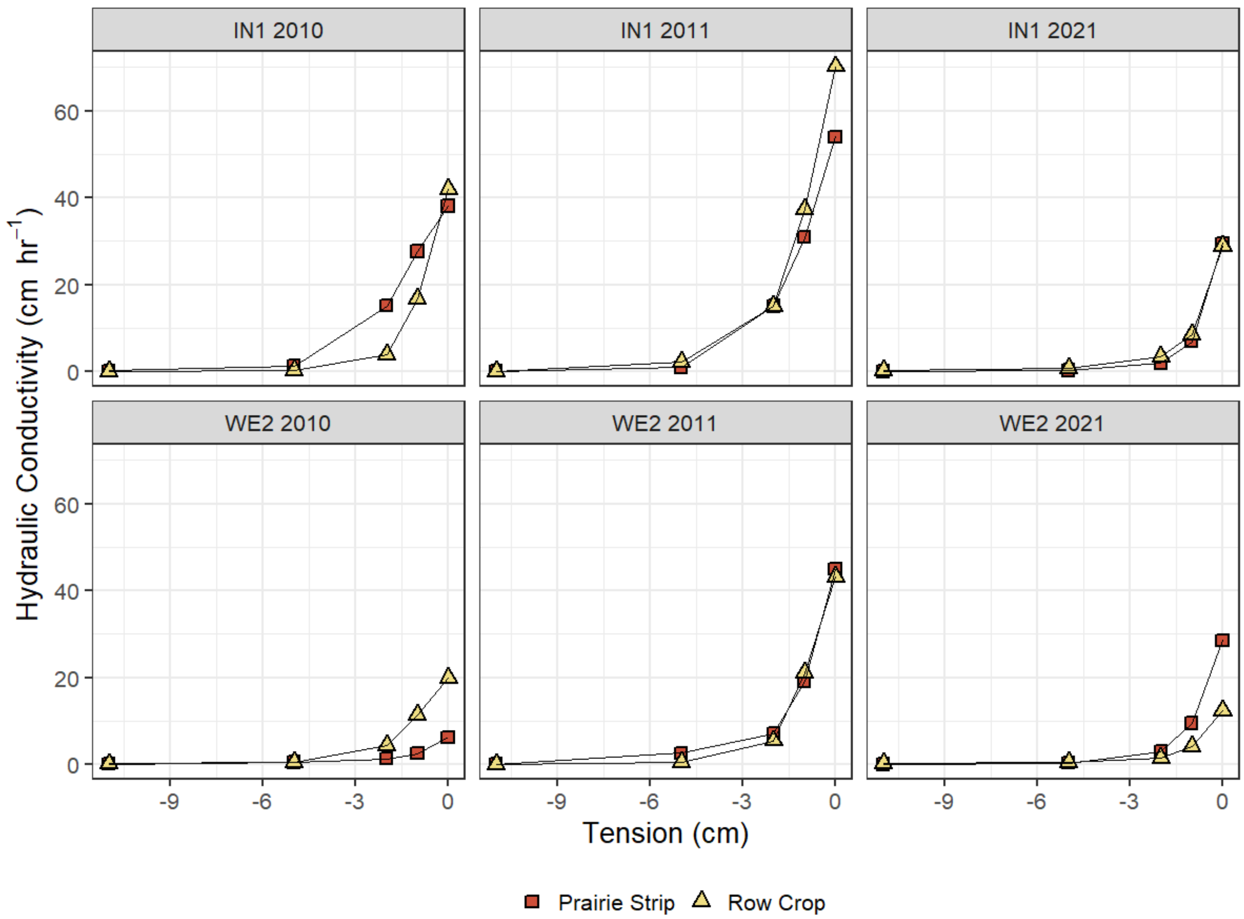

Figure 2.6. Average hydraulic conductivities (cm hr-1) at each tension at Phase I sites

CHAPTER 3. SOIL HEALTH RESPONSES TO PRAIRIE STRIPS AFTER SIX TO SEVEN YEARS SINCE ESTABLISHMENT

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Abstract

The widescale transformation of native tallgrass prairie to annual row crop agriculture in the U.S. Corn Belt has introduced a multitude of environmental concerns, including a general decline in soil health. The integration of prairie vegetative strips into row crop fields is an increasingly popular conservation strategy that has been shown to improve biodiversity and mitigate sediment and nutrient export. However, the impact of narrow (5-10 m) prairie strips (PS) on soil health is unclear. This study investigated the effects of PS on soil health by performing a suite of soil physical, chemical, and biological analyses at six- to seven-year-old PS sites around Iowa. PS consistently improved wet-aggregate stability across six sites, as the mean weight diameter of water-stable aggregates in 0-5 cm depth samples was 25% greater in PS than in row crop (RC). Conversely, we found inconsistent evidence of a PS effect for other soil parameters. However, the extent of PS effects on soil properties varied considerably by site. While an overall improvement of soil physical, chemical, and biological properties due to PS establishment was not abundantly clear, enhanced wet-aggregate stability in PS signifies soil structure optimization and greater resistance to erosion. Since wet-aggregate stability plays an

integral role in processes that contribute to soil health, it may serve as a leading indicator for PS-related soil health improvements that are slower to detect.

Introduction

The U.S. Corn Belt looks much different than it did prior to the beginning of extensive European settlement in the 1800s. Today, less than 1% of Iowa's 12.5 million historical tallgrass prairie hectares remain (Samson & Knopf, 1994), and the majority of that land is dedicated to corn (*Zea mays* L.) and soybean [*Glycine max.* (L.) Merr.] production (USDA, 2022). Iowa's dramatic land use transformation has profoundly impacted soil health. Soil health is defined as "the continued capacity of soil to function as a vital living ecosystem that sustains plants, animals, and humans" (USDA-NRCS). While traditional soil assessments focused chiefly on physical or chemical soil properties (Schoenholtz et al., 2000; Weil et al., 2003), the soil health concept emphasizes the integration and optimization of physical, chemical, and biological soil processes that contribute to productivity and environmental quality (Karlen et al., 1997; Bünemann et al., 2018). In many parts of the Corn Belt, soil health has diminished over time as long-term cultivation has led to increased soil bulk density (Udawatta et al., 2008), diminished nutrient retention (Kemp & Dodds, 2001), and a decline in soil organic carbon (SOC) between 30 and 60% (Davidson & Ackerman, 1993; Kucharik et al., 2001; Salemme et al., 2018) compared to native prairie.

In recent decades, efforts to restore portions of the native tallgrass prairie ecosystem have grown, with expected soil health improvement serving as one of the numerous motives. Reestablishment of prairie vegetation effectively satisfies the four principles of soil health outlined by the USDA-NRCS: (i) maximize living roots, (ii) minimize disturbance, (iii) maximize soil cover, and (iv) maximize biodiversity, and multiple studies have analyzed the effects of prairie restoration on an array of soil health indicators (Baer et al., 2002; Allison et al., 2005; De et al., 2020; Li et al., 2021). Most prairie restoration studies report positive effects on soil health compared to cropland, but the rate and extent to which soil health improvements occur have been inconsistent.

Since 2007, the Science-based Trials of Row-crops Integrated with Prairie Strips (STRIPS; https://www.nrem.iastate.edu/research/STRIPS/) project has investigated the strategic conversion of 10-20% of row crop field area to native prairie vegetative contour and foot slope strips. STRIPS research efforts have shown that by establishing prairie strips (PS) within cultivated fields, the delivery of multiple ecosystem services can accrue at levels disproportionately greater than the land area that PS occupy (Schulte et al., 2017). However, there is still a lot we do not understand about how narrow strips of prairie affect soils. The placement of PS directly within row crop (RC) fields and their function as a sediment and nutrient buffer introduces unique circumstances that could potentially affect soil health differently than traditional prairie reconstructions, which usually occupy large swaths of land. At a central Iowa location, Pérez-Suárez et al. (2014) found SOC and nitrogen accumulation in PS three years after establishment and a decline in soil C:N ratio – a result uncommon in most prairie reconstruction studies. In two- to four-year-old PS in central and southwest Iowa, Flater (2020) did not find significant differences in microbial diversity between PS and adjacent RC soils. Given these results, there is a need for a more comprehensive analysis of overall soil health responses to PS across different locations and ages. Improved soil health under PS could signify a reduced risk of soil degradation, enhanced water and nutrient cycling, and atmospheric C sequestration (Lal, 2015; Paustian et al., 2016).

The goal of this study was to thoroughly assess soil health under PS compared to row RC by collecting a suite of soil physical, chemical, and biological data at multiple PS sites in Iowa. We hypothesized that soil health improvements in PS would become apparent relatively quickly at only six to seven years post-establishment. In particular, we expected wet-aggregate stability and active carbon to respond most strongly to PS establishment as previous studies have suggested these properties as early indicators of change in prairie restorations (Anderson et al., 2019; De et al., 2020; Li et al., 2021). We also anticipated elevated levels of P and TN within PS as a consequence of their demonstrated capability to act as nutrient and sediment filters (Helmers et al. 2012; Hernandez-Santana et al., 2013; Pérez-Suárez et al., 2014; Zhou et al., 2014). Certain parameters like bulk density, soil pH, and soil organic matter were predicted to have relatively weaker responses to PS since other studies have reported significant changes on decadal timescales (Baer et al., 2002; McLauchlan et al., 2006; De et al., 2020; Glass et al., 2021; Libbey & Hernández, 2021).

Materials and Methods

Site Descriptions

Six sites across the state of Iowa were selected to compare PS and RC soil parameters. These sites – ARM, HOE, MCN, RHO, WHI, and WOR – were 100% RC fields growing corn (*Zea mays* L.) and soybean (*Glycine max.* (L.) Merr.) prior to PS establishment between 2014-2015. In recent years, no-till management has been practiced at ARM and WHI, while conventional tillage takes place at HOE, MCN, RHO, and WOR. Additionally, beef cattle graze MCN and RHO after each corn phase. All six sites are located within either the Des Moines Lobe or Southern Iowa Drift Plain landform region (Figure 3.1). Relatively recent glacial activity (12,000 to 14,000 years ago) accounts for the deposits of deep, loamy till that predominate the

Des Moines Lobe's flat to gently rolling landscape (Prior & Lohmann, 1991). An extensive network of artificial subsurface drainage has helped transform this naturally poorly drained region into highly productive cropland. The Southern Iowa Drift Plain's most recent glacial episode occurred hundreds of thousands of years ago. As a result, this large swath of Iowa's southern half is now characterized by a loess-mantled, rolling to hilly landscape (Prior & Lohmann, 1991). Silt loam and silty clay loam are typical soil textural classes present in this region. Mollisols and – to a greater extent in the Southern Iowa Drift Plain – alfisols make up the dominant soil orders of both the Des Moines Lobe and Southern Iowa Drift Plain (NRCS, 2006).

Soil Sampling Techniques

Sampling locations were determined using soil series and phase data from the USDA-NRCS Web Soil Survey. Three replications of paired PS and RC sampling points were randomly selected within each soil series and phase present at each site (Figure 3.1). PS samples were collected from the center of the strip, while RC samples were collected 3 m directly upslope from the PS edge. Obvious wheel tracks were avoided to reduce the influence of mechanical compaction.

In June-August of 2021, soil cores were collected from 0-5 cm and 5-15 cm depths using a 2.54-cm diameter hand probe for analysis of wet-aggregate stability via the wet sieving method. Six to ten cores were taken from within an approximate 5 m radius around each sampling point and stored in sealed zip-top bags. Care was taken to keep soil cores intact and undisturbed. This protocol was also followed at select sites between October and December in 2016, 2018, and 2020 (Brittenham, 2017).

Soil collection for chemical and biological property analyses and water-stable aggregates via the rainfall simulator method occurred between October and November 2021, shortly after harvest and before any fall tillage. Collection techniques and storage procedures followed Moebius-Clune et al. (2016). Within an approximate 5 m radius around each sampling point, three 5 cm by 15 cm slices of soil were collected using a spade and mixed to obtain a composite sample. Soil was contained in sealed zip-top bags and stored in a 4 ºC cooler prior to shipment for analysis at a commercial laboratory (Cornell Soil Health Lab, Ithaca, NY). Simultaneous to soil collection, penetrometer resistance measurements were taken using a FieldScout SC 900 Soil Compaction Meter (Aurora, IL). Additionally, bulk density cores were collected using an AMS 2" by 6" core sampler equipped with a slide hammer (American Falls, ID) and stored in capped plastic liners.

Soil Physical Properties

Surface and subsurface hardness values represent the highest reading (psi) recorded as a field penetrometer was slowly pressed through 0-15 cm and 15-45 cm soil depths, respectively. Three measurements were taken around each sampling point, corresponding to where soil samples were collected and averaged to create a composite value.

Bulk density was calculated for 0-5 cm, 5-15 cm, and 0-15 cm depths as the mass of solids (m_s) divided by the volume of the core (V_c) (Equation 3.1).

$$\rho_b = \frac{m_s}{V_c} \tag{3.8}$$

To obtain m_s, the contents of each core were oven-dried at 105 ºC for 24 hours. For 0-5 cm and 5-15 cm bulk density values, field-moist 15 cm cores were cut at 5 cm in the laboratory. In some instances, soil compaction occurring during core extraction resulted in void space at the top of a core sample. Evenly distributed compaction was assumed, and one-third of the void space length was subtracted from the 5 cm cut length to avoid inflation of 0-5 cm bulk density values.

Two methods – rainfall simulator and wet sieving – were utilized to estimate wet-aggregate stability. The rainfall simulator method was performed on 0-15 cm samples by a commercial lab (Cornell Soil Health Lab, Ithaca, NY) and followed an adapted protocol from Moebius et al. (2007). In this procedure, a single layer of 0.25-2.0 mm air-dried soil aggregates (~30g) were spread evenly on a 0.25 mm sieve. A rainfall simulator was placed 500 mm above the aggregates, and 12.5 mm of rainfall was simulated over a five-minute period, mimicking the rainfall energy delivered by a heavy thunderstorm. Upon completion of the rainfall simulation, soil that had not passed through the sieves was collected, dried, and weighed. This weight was corrected for gravel and considered the weight of stable soil aggregates (W_{stable}). The percentage of water-stable aggregates (WSA) was calculated as W_{stable} divided by the total weight of aggregates prior to rainfall simulation (W_{total}) (Equation 3.2).

$$WSA\ (\%) = \frac{W_{stable}}{W_{total}} * 100 \tag{3.9}$$

The wet sieving procedure was performed on 0-5 cm and 5-15 cm soil samples using an adaptation of the method described in Yoder (1936). Initial pre-processing began by breaking field-moist soil cores along natural fissures and passing the soil through an 8 mm sieve (Ontl et al., 2015). Aggregates were then air-dried for at least 48 hours with occasional mixing to ensure an even drying distribution. Roots longer than 1 cm and gravel were removed from the air-dried soil. A 10 g subsample was extracted from each air-dried soil sample and dried at 105 ºC for 24 hours to determine the air-dried gravimetric moisture content. Next, approximately 100 g of air-dried soil was spread evenly on a petri dish lined with filter paper. The day before sieving, the air-dried soil was capillary wetted to field capacity with deionized water, taped shut, and stored overnight at 4 ºC (Márquez et al., 2004). The next day, moist aggregates were spread evenly on a stack of sieves with 2.00, 1.00, and 0.21 mm openings and submerged in water for five minutes prior to wet sieving. Subsequent steps closely followed the wet-aggregate size distribution protocol detailed in Ninmo and Perkins (2002), where submerged sieves oscillated up and down for 10 minutes with a 4 cm stroke length and 30 stroke min^{-1} frequency. The contents remaining on each sieve were then backwashed onto pre-weighed tins, dried at 60 ºC for 48 hours, and weighed. Van Bavel (1950) 's mean weight diameter (Equation 3.3) and an approximation of WSA (Equation 3.2) were used to represent the aggregate size distribution.

$$MWD = \sum_{i=1}^{n} x_i m_i \tag{3.10}$$

where MWD is the mean weight diameter (mm), n is the number of aggregate size ranges, x_i is the mean diameter of the aggregate size range (mm), and m_i is the fraction of the total sample weight remaining in the corresponding aggregate size range. A sand correction was performed for HOE and WOR due to their considerable sand content by subtracting the weight of sand from the weight of aggregates in each size range and the total weight (Márquez et al., 2004). To measure sand content in each aggregate fraction, 10 g of dried soil and 30 mL of 5 g L^{-1} sodium hexametaphosphate solution were added to a 125 mL bottle and oscillated on a reciprocal shaker for 15 hours (Cambardella & Elliott, 1992). The dispersed solution was passed through a 0.053 mm sieve and rinsed with water. Contents remaining on the sieve were considered sand and were oven-dried and weighed.

Since 0-5 cm and 5-15 cm samples were processed separately, a bulk density-weighted average was used to calculate 0-15 cm values. It should also be noted that the wet sieving procedure likely overestimates WSA slightly since its 0.21 mm bottom sieve is smaller than the standard 0.25 mm.

Soil Chemical Properties

Phosphorus (P) and potassium (K) were extracted using Modified Morgan's solution, an ammonium acetate plus acetic acid solution buffered at pH 4.8. After shaking a mixture of soil solution, the extraction slurry was filtered, and the filtrate was analyzed on an inductively coupled plasma emission spectrometer (ICP, Spectro Arcos) to determine nutrient values. A pH electrode probe measured pH in a 1:1 soil to water suspension. Lastly, total nitrogen (TN) (organic and inorganic forms) was determined with the Dumas combustion methodology (Dumas, 1831).

Soil Biological Properties

Loss on ignition with a 500 ºC furnace was used to determine the percentage of soil organic matter (SOM) in oven-dried soil. Similarly, total carbon (TC) was calculated via complete oxidation of carbon through high-temperature combustion (1,100 ºC). Total carbon measurements included both organic and carbonate components. Permanganate-oxidizable carbon (POXC) was determined through potassium permanganate oxidation (Weil et al., 2003).

Statistical Analysis

All statistical analysis was run using R software (R Core Team, 2020), and plots were generated with the ggplot2 package (Wickham, 2016). Soil property data were log-transformed to normalize the dataset and facilitate between-site comparison. We analyzed paired differences between PS and RC treatments using a linear model, and contrasts and comparisons were determined with least-squares means (Lenth, 2020). For all soil physical, chemical, and biological properties, treatment differences were tested at each site (ARM, RHO, and WOR). Additionally, given the shared land formation history of ARM and RHO (Southern Iowa Drift Plain) and their similar soil textures (silt loam to silty clay loam), analyses were performed at the landform region level. Statistical significance was categorized as marginal ($p < 0.1$), significant ($p < 0.05$), and strongly significant ($p < 0.01$).

Results

Soil Physical Properties

Differences between PS and RC treatments were minimal across all levels for both surface and subsurface hardness (Table 3.2). Between these two measurements, only one significant difference occurred at the WOR site, where the average RC surface hardness was 215 psi, and the average PS surface hardness was 247 psi. In general, the magnitude of penetrometer resistance was very similar in PS at both surface and subsurface depths. Similar to surface and subsurface hardness, substantial differences in bulk density between PS and RC were absent. Although the differences were statistically insignificant, greater average bulk density was observed in PS at ARM, while RC had greater average bulk density at RHO.

Treatment differences in measurements of wet-aggregate stability were evident across nearly all sites, depths, and methods in 2021. At the 0-5 cm depth, PS had a greater average MWD than RC at each site, and PS MWD was between 19 and 29% greater than RC across all six sites (Table 3.3 and Figure 3.2). At ARM and RHO, the significant treatment differences observed in 2021 were contrary to results from previous years in which no conclusions could be made at the 90% confidence level (Figure 3.3). At ARM, the mean difference between PS and RC MWD increased every year, while at RHO, no trend was apparent between 2016 and 2021. For estimations of percent WSA in 0-15 cm samples taken from ARM and RHO, PS was higher than RC regardless of the testing procedure (Table 3.4). Across three sites, the average percentage of water-stable aggregates, as measured via rainfall simulation, was between 10.4 and

14.4% greater in PS than in RC. Wet sieving of MCN samples yielded the only observation of no difference in wet-aggregate stability between PS and RC.

Soil Chemical Properties

Both measurements of TN and P did not vary significantly between PS and RC (Table 3.5). Phosphorus was highly variable between and within each site, especially for RC. In contrast to TN and P, we found differences between PS and RC in measurements of K and pH. Potassium was consistently higher in PS than in RC, and the difference was strongly significant at each of the three sites. Between the two Southern Iowa Drift Plain (SIDP) sites combined (ARM and RHO), the average K values were 255 mg kg^{-1} and 192 mg kg^{-1} for PS and RC, respectively. For pH, no differences were found at the individual site level; however, a marginally significant positive difference was shown between PS and RC pH when the two SIDP sites were combined.

Soil Biological Properties

The distinction between PS and RC concerning soil biological properties was relatively weak (Table 3.6). At WOR, average SOM, POXC, and soil C:N values were nearly identical between treatments. At ARM, average SOM, TC, and soil C:N were slightly higher in PS than RC, but the difference was statistically insignificant. Contrarily, POXC was significantly greater in RC than PS, with average values of 565 and 497 mg kg^{-1}, respectively. Lastly, at RHO, PS had consistently greater values of biological properties than RC. Most outstanding, a strongly significant treatment difference was observed in SOM, with the average percentage of SOM being 2.77 in PS and 2.49 in RC. Additionally, PS soil had a significantly greater C:N ratio than RC. When treatment differences were analyzed together for the Southern Iowa Drift Plain sites (ARM and RHO), SOM and C:N were significantly greater in PS than RC.

Discussion

Soil Physical Properties

Out of four soil physical parameters tested between PS and RC, wet-aggregate stability showed the most salient response to PS establishment. The effect of PS on bulk density, surface hardness, and subsurface hardness after six to seven years since establishment was minimal.

The confounding effects of tillage likely played a prominent role in surface and subsurface hardness observations at tilled sites – RHO and WOR. Recent tillage loosens and aerates surface soil, while repeated use of some tillage implements can form plow pans beneath the plowing layer (Gaultney et al., 1982). The lower penetrometer resistance measured in the 0-15 cm layer of RC soil at WOR can undoubtedly be attributed to the effects of recent tillage. If a significant plow pan was present, "biodrilling" performed by robust root systems such as those possessed by certain native prairie species could have functionally reduced the associated subsurface hardness (Williams & Weil, 2004). However, the relatively minuscule surface area that can be assessed through measurements of penetrometer resistance in combination with the sporadic presence of possible "biodrilling" sites severely inhibited their detection in PS. At the only no-till site, ARM, average surface and subsurface hardness values were nearly identical between PS and RC, suggesting no PS effect in no-till farming environments.

The lack of significant observations related to bulk density was not totally unexpected. Bulk density provides practical insight into the soil's physical condition and accommodates useful volumetric unit conversion for other measurements (Doran & Parkin, 1996). However, the effects of similar land cover changes on bulk density can be slow to occur and difficult to detect (Karlen et al., 1999; Pey & Dolliver, 2020).

Contrary to other physical properties, PS clearly had a positive impact on wet-aggregate stability. The greater wet-aggregate stability in PS compared to RC is credited to macroaggregate stabilization through root and fungal hyphae growth (Tisdall & Oades, 1982; Oades, 1984) and the destruction of larger soil aggregates by tillage and cultivation (Cambardella & Elliott, 1993; Bronick & Lal, 2005). Although we predicted treatment differences, the consistency of significant observations across depth increments, tillage practices, and time since PS establishment made these results especially notable. The PS effect observed on MWD at no-till sites at both 0-5 cm and 0-15 cm depths accentuates the aggregate stabilizing capabilities of root and fungal hyphae growth in PS. As significant differences in 0-5 cm MWD occurred at ARM and RHO starting in 2021 and three of the four remaining 2021 sites also had significantly greater PS MWD, our results suggest that wet-aggregate stability improvements in PS become evident six to seven years post-establishment. It is probable that these changes occur gradually over time, as evidenced by the positive trend of MWD treatment differences recorded at ARM. Our results closely match those found in a Missouri prairie reconstruction chronosequence study which reported peak wet-aggregate stability in 8-year restorations (Li et al., 2021).

Soil Chemical Properties

The effects of PS on soil chemical properties were generally inconsequential. Although we hypothesized that PS would retain and accumulate mobile nitrogen and phosphorus, the results of this study showed no difference between PS and RC in measurements of TN and P. A previous study reported increased TN retention in PS soil as a result of multiple simultaneous biogeochemical mechanisms such as increased plant N uptake and chemical fixation by microorganisms (Pérez-Suárez et al., 2014). Also, available P has been shown to increase in vegetative buffer strips relative to cropland (Stutter et al., 2009), and reduced P export in small watersheds containing PS has been reported (Zhou et al., 2014). However, our results show minimal differences between PS and RC concerning these parameters. This discrepancy may not reflect results truly contradictory to previous studies but rather stem from differences in experimental design. Our paired design was based on soil series and phase, and as a result, landscape positions were not represented equally in soil sampling. Both P and TN have been shown to accumulate most within perennial vegetative strips at the foot slope position (Tomer et al., 2007; Pérez-Suárez et al., 2014). Our lack of replication at the foot slope limits the analysis of TN and P strictly at that position, and therefore, possible changes that may be occurring at the foot slope are drowned out in our dataset.

Potassium differences between PS and RC likely result simply from crop harvest K losses. While erosion and leaching are possible loss pathways for K in RC (Goulding et al., 2021), the concurrent absence of TN and P differences suggests that erosion and leaching do not account for major K losses at the selected sites.

Our soil pH results indicated a greater soil acidification rate in PS than RC at Southern Iowa Drift Plain sites. Soil acidification in RC systems is attributed to synthetic nitrogen fertilizer application and annual crop harvest (Brady & Weil, 2008). In prairie systems, soil can undergo acidification due to year-round organic matter oxidation and root dynamics unique to perennial prairie plants (Brye et al., 2008). While Brye and Pirani (2005) reported more acidic pH under restored prairie than RC, our results at PS sites suggest the opposite. Since acidity dictates many soil biotic processes (Husson, 2013), maintaining soil pH within an optimal range is critical for plant growth and overall soil ecosystem function. If soil indeed acidifies more rapidly under RC, this could result in the need for more frequent pH amelioration compared to PS.

Soil Biological Properties

Overall, PS establishment affected soil biological properties in a limited capacity. However, differences in intrinsic soil properties and management likely contributed to variation between sites. The interactions of site characteristics such as topography, soil texture, and moisture can substantially affect C and N storage dynamics in restored grasslands (O'Brien et al., 2010; Whisler et al., 2016; Auerswald & Fiener, 2019). Our results reflected this understanding as sites located within the Southern Iowa Drift Plain (silt loam to silty clay loam soils) displayed greater average SOM, TC, and TC:TN in PS than RC while biological properties were indistinguishable between PS and RC at the Des Moines Lobe site, WOR (loam soils). Similarly, Brye and Kucharik (2003) and Auerswald and Fiener (2019) observed considerable C accrual in prairie restorations with fine-textured soils, while coarser textures and improved drainage stymied C storage.

Between the two Southern Iowa Drift Plain sites, RHO displayed much more robust responses to PS. The different tillage practices of the two sites likely explains this disparity. Cessation of tillage reduces aggregate turnover and effectively stabilizes and sequesters C in surface soil (Six et al., 2000). Since ARM has continuously practiced no-till management since PS establishment, it is probable that C accrual has occurred in both RC and PS principally as a result of minimal soil disturbance. In agreement with this hypothesis, a similar study investigating no-till to native prairie conversion found no difference in physically-protected organic matter pools between the two systems (Bugeja & Castellano, 2018). In contrast to ARM, conventional tillage practices at RHO opened the door for significant SOM increases in PS through aggregate stabilization. Supplementing the effects of tillage discontinuation, the relatively degraded state of RHO may have factored into its SOM improvement within PS as the rate of SOM increase is negatively related to SOM content (Knops & Tilman, 2000), and low SOC soils have greater potential for C sequestration (von Haden & Dornbush, 2017).

While PS were assumed to have greater belowground C inputs than RC (Guzman & Al-Kaisi, 2010), it appears that the anticipated effect of this factor was not fully met. In fact, POXC was significantly greater in RC than PS at ARM and indistinguishable between treatments at RHO and WOR - contradicting the idea that prairie's year-round root activity increases active carbon (Li et al., 2021). This trend was echoed in TC results as well, as PS and RC were not significantly different at any level. However, the greater C:N ratio observed in PS relative to RC at Southern Iowa Drift Plain sites indicated a change in soil C and N dynamics due to PS vegetation. High C:N ratios typical of prairie grasses presumably account for this shift, and given the expected tight coupling of C and N (Cotrufo et al., 2019), it could signify progress towards increased capacity for C and N storage in PS and conditions more like those found in remnant prairies (Glass et al., 2021). Increases in soil C:N within PS actually disagrees with prior PS findings (Pérez-Suárez et al., 2014), but previously mentioned landscape position effects likely contribute to the disparity. It is important to note that several studies suggest that plant species composition significantly contributes to C and N dynamics within prairie restorations (O'Brien et al., 2010; Pérez-Suárez et al., 2014; Whisler et al., 2016). However, in the absence of pertinent, quantitative data, analysis of this factor was withheld for this study. Overall, our results suggest that potential soil biological property improvements resulting from PS establishment do not become apparent after six to seven years. However, while significant changes were not strongly evident, the possibility that C accrual is currently occurring undetected exists as a new C equilibrium in restored grasslands can take over 20 years to reach (Smith, 2014; Bugeja & Castellano, 2018).

Conclusions

After six to seven years since establishment, prairie strips (PS) improved wet-aggregate stability consistently, while other soil health parameters did not display strong trends. Since wet-aggregate stability was the only parameter to show distinct responses, we cannot make a conclusive statement about PS improvement on overall soil health at this point. However, implications associated with the wet-aggregate stability enhancement due to PS include erosion resilience and favorable soil pore distributions for water movement and biological activity. Given these associations, wet-aggregate stability could potentially serve as a leading indicator of future improvements to other soil health metrics that may be slower to detect. While we hypothesized wet-aggregate stability would increase in PS, and this was supported, our study did not support hypothesized mobile nitrogen and phosphorus accumulation within PS. However, landscape position interactions may have limited its detection and should be accounted for in future studies. Future research should also investigate the possible effects of prairie plant species composition and its impact on soil health responses in PS. Site characteristics factored greatly into soil health responses to PS in this study, especially in regards to biological properties. Repeated analysis in the future could facilitate determining the rate of change associated with soil health properties and how they can vary by location.

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Tables and Figures

Table 3.1. Site characteristics

Site	Dominant soil order	Sand (%)	Clay (%)	Silt (%)	Tillage[a]	2020 crop	2021 crop
ARM	Mollisol	5	30	66	NT	Soybean	Corn
HOE	Mollisol	35	28	37	CT	Corn	Soybean
MCN	Mollisol	8	32	60	CT	Soybean	Corn
RHO	Alfisol	3	23	74	CT	Corn	Corn
WHI	Mollisol	11	32	57	NT	Corn	Soybean
WOR	Mollisol	42	22	36	CT	Soybean	Corn

Note: Soil texture for 0-15 cm depth acquired from Web Soil Survey.
[a]NT, no-tillage; CT, conventional tillage.

Table 3.2. Average soil physical properties for prairie strip and row crop treatments at ARM, RHO, WOR, and Southern Iowa Drift Plain (SIDP) sites combined

Site	Treatment	Water-stable aggregates (%)	Bulk density (g cm^{-3})	Surface hardness (psi)	Subsurface hardness (psi)
ARM[a]	Prairie Strip	38.3	1.02	210	224
	Row Crop	23.9	0.99	212	229
		***	-	-	-
RHO[a]	Prairie Strip	24.3	1.05	294	295
	Row Crop	13.9	1.08	306	277
		***	-	-	-
WOR	Prairie Strip	26.9	-	247	243
	Row Crop	14.8	-	215	256
		***	-	**	-
SIDP	Prairie Strip	30.7	1.04	246	254
	Row Crop	18.1	1.03	252	249
		***	-	-	-

Note: Asterisks indicate significance of treatment difference (* p <0.1, ** p <0.05, *** p <0.01).
[a]SIDP sites.

Table 3.3. Average mean weight diameter for prairie strip (PS) and row crop (RC) treatments and corresponding paired treatment ratio at each site

Site	Treatment	Mean (mm)	CV (%)	Median ratio (PS/RC)	90% CI (PS/RC)
ARM	Prairie Strip	2.26	14.4	1.19	[1.05 1.34]
	Row Crop	2.05	20.8		
HOE	Prairie Strip	3.30	13.1	1.09	[0.98 1.22]
	Row Crop	2.99	14.0		
MCN	Prairie Strip	3.11	9.38	1.30	[1.14 1.47]
	Row Crop	2.42	17.8		
RHO	Prairie Strip	2.41	23.6	1.36	[1.18 1.57]
	Row Crop	1.80	31.2		
WHI	Prairie Strip	2.79	9.59	1.19	[1.05 1.34]
	Row Crop	2.39	21.8		
WOR	Prairie Strip	3.18	9.20	1.62	[1.40 1.86]
	Row Crop	2.00	22.0		
All Sites	Prairie Strip	2.86	18.83	1.25	[1.19 1.29]
	Row Crop	2.32	26.81		

Table 3.4. Wet-aggregate stability method comparison for prairie strip and row crop treatments

Site	Treatment	Rainfall simulator WSA[a] (%)		Wet sieving WSA[a] (%)	
		Mean	CV (%)	Mean	CV (%)
ARM	Prairie Strip	38.3	28.51	85.0	7.61
	Row Crop	23.9	26.78	80.6	8.12
		***		**	
MCN	Prairie Strip	-	-	83.7	4.68
	Row Crop	-	-	83.0	3.96
				-	
RHO	Prairie Strip	24.3	50.34	81.4	8.7
	Row Crop	11.5	48.58	75.6	10.46
		***		***	
WOR	Prairie Strip	25.3	20.87	-	-
	Row Crop	14.9	14.26	-	-
		***			

Note: Asterisks indicate significance of treatment difference (* p <0.1, ** p <0.05, *** p <0.01).
[a]Water-stable aggregates.

Table 3.5. Average soil chemical properties for prairie strip and row crop treatments at ARM, RHO, WOR, and Southern Iowa Drift Plain (SIDP) sites combined

Site	Treatment	Total nitrogen (g m^{-2})	Phosphorus (mg kg^{-1})	Potassium (mg kg^{-1})	pH
ARM[a]	Prairie Strip	275	3.98	264	7.06
	Row Crop	273	3.53	202	6.85
		-	-	***	-
RHO[a]	Prairie Strip	234	19.0	243	7.02
	Row Crop	229	19.2	162	6.75
		-	-	***	-
WOR	Prairie Strip	-	2.37	184	5.51
	Row Crop	-	2.40	138	5.55
		-	-	***	-
SIDP	Prairie Strip	255	7.25	255	7.04
	Row Crop	251	6.04	192	6.81
		-	-	***	*

Note: Asterisks indicate significance of treatment difference (* $p < 0.1$, ** $p < 0.05$, *** $p < 0.01$).
[a]SIDP sites.

Table 3.6. Average soil biological properties for prairie strip and row crop treatments at ARM, RHO, WOR, and Southern Iowa Drift Plain (SIDP) sites combined

Site	Treatment	Soil organic matter (%)	Active carbon[a] (mg kg^{-1})	Total carbon (g m^{-2})	C:N
ARM[b]	Prairie Strip	3.53	497	3005	10.91
	Row Crop	3.48	565	2919	10.66
		-	**	-	-
RHO[b]	Prairie Strip	2.77	494	2397	10.23
	Row Crop	2.49	458	2251	9.80
		***	-	-	**
WOR	Prairie Strip	2.86	382	-	11.98
	Row Crop	2.86	369	-	12.03
		-	-	-	-
SIDP	Prairie Strip	3.21	496	2701	10.62
	Row Crop	3.06	519	2585	10.29
		**	-	-	**

Note: Asterisks indicate significance of treatment difference (* $p < 0.1$, ** $p < 0.05$, *** $p < 0.01$).
[a]Active carbon describes permanganate oxidizable carbon.
[b]SIDP sites.

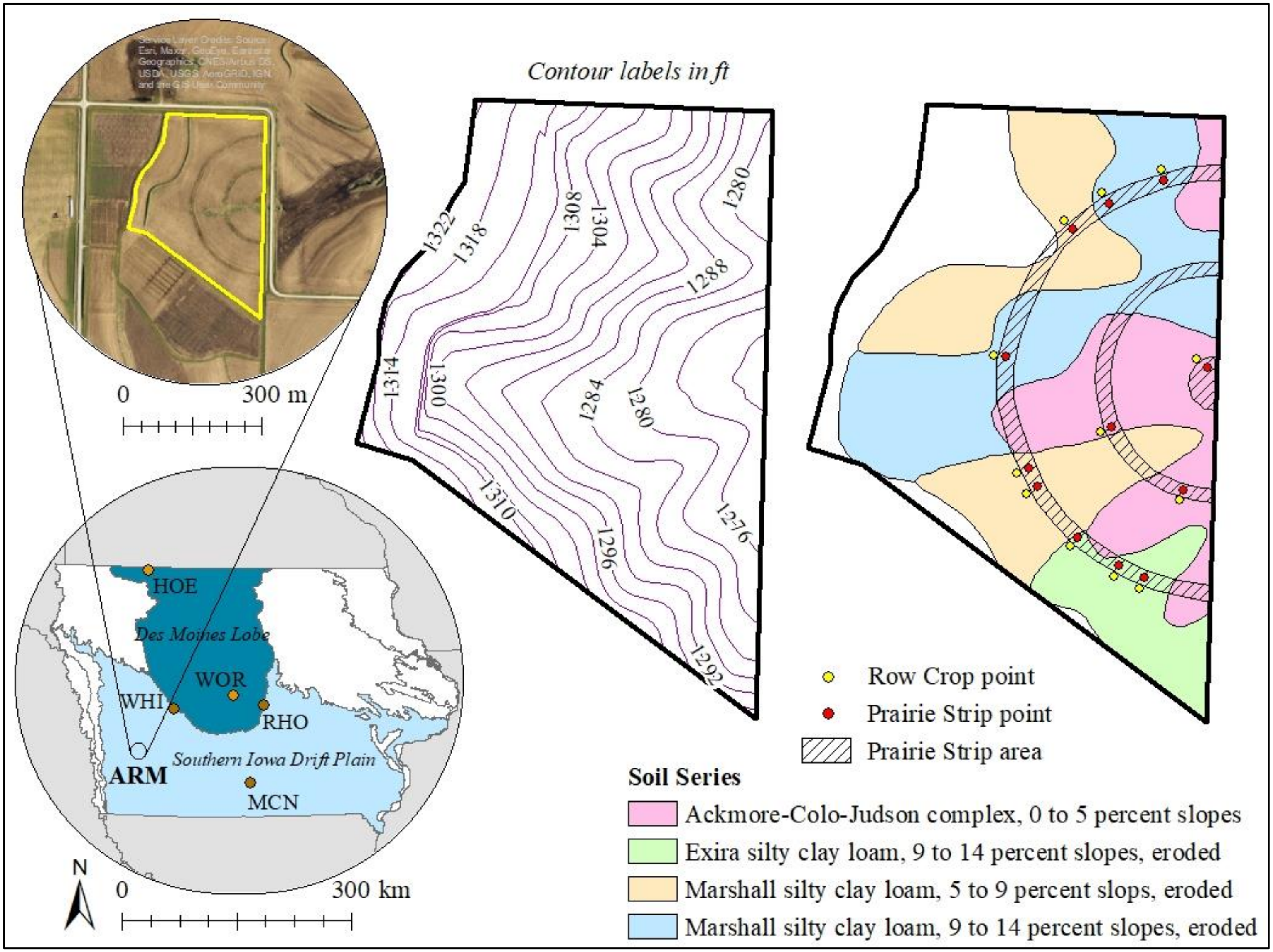

Figure 3.1. Site locations in relation to Iowa landform regions with example aerial imagery, elevation, and soil sampling maps for the ARM site

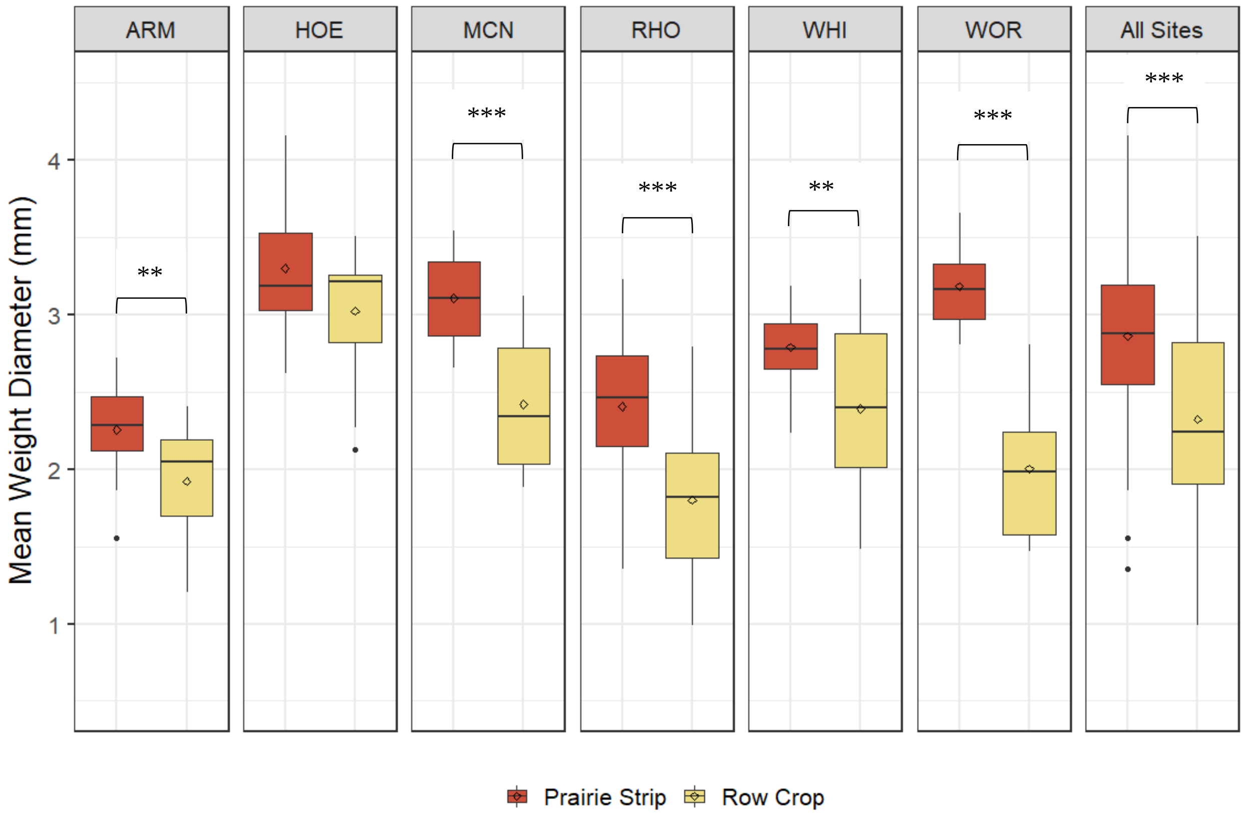

Figure 3.2. Comparison of prairie strip and row crop mean weight diameter (mm) values determined with the wet-sieving method at each site in 2021
Note: Asterisks indicate significance of treatment difference (* p <0.1, ** p <0.05, *** p <0.01).

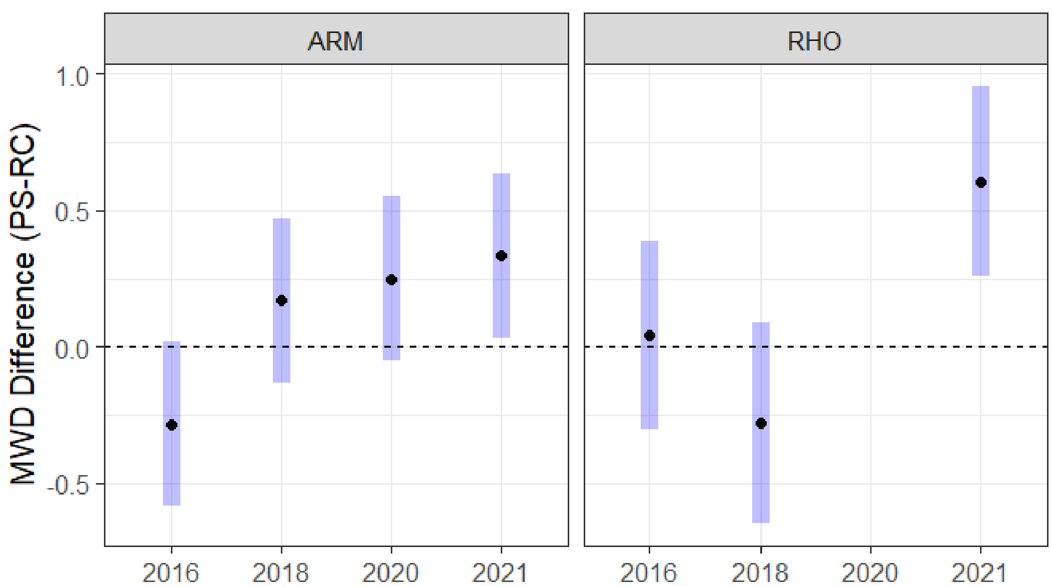

Figure 3.3. Average mean weight diameter (mm) paired differences between prairie strip (PS) and row crop (RC) treatments over time
Note: Blue bars represent 90% confidence intervals.

CHAPTER 4. EXPLORATION OF SOIL HEALTH INDEX SCORING FOR COMPARING PRAIRIE STRIP AND ROW CROP SOILS

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Abstract

Soil health and quantitative methods for assessing soil health have become exponentially popular in the past decade as the need for more sustainable soil management has become increasingly apparent around the globe. In the US Midwest, a recent conservation strategy called prairie strips (PS) involves integrating strips of native prairie vegetation into existing row crop (RC) production fields and provides multiple ecosystem services. This practice also has the potential to improve regional soil health. This study compared soil health between PS and RC treatments at multiple sites in the state of Iowa using Cornell's Comprehensive Assessment of Soil Health (CASH) and the Soil Management Assessment Framework (SMAF). Both CASH and SMAF use scoring functions to translate the observed values of soil health indicator measurements into unitless scores. While we used these scores to assess differences in PS and RC treatments, we also wanted to explore the uses and limitations of soil health index scoring in the context of comparative studies like this one. We found that the PS treatment had slightly greater overall soil health than the RC treatment according to both CASH and SMAF. Wet-aggregate stability differences were the driving force behind the greater soil health scores recorded for PS compared to RC, and our results suggest that wet-aggregate stability is a leading

indicator for overall soil health improvements. We found that CASH and SMAF both provided meaningful value in our analysis of PS and RC soil health by affording easy interpretation of soil health indicator values and a framework to integrate them into comprehensive scores. However, certain nuances of the true soil condition can be missed if overall scores, individual indicator scores, and observed values are not all analyzed together thoroughly. Soil organic matter values had the strongest correlation with overall CASH scores and can likely be an adequate proxy for soil health in similar studies if additional analyses are not feasible.

Introduction

The term "soil health" has become increasingly common in the United States and around the globe in the past decade. The concepts behind what is broadly defined as soil health are not necessarily new (Lehmann et al., 2020; Karlen et al., 2021); however, growing pressure on the world's soil resources to produce food, fiber, and fuel has heightened interest in more sustainable soil management and the prevention of continued soil degradation (Karlen & Rice, 2015; Lal, 2015). While conventional thought may consider soil simply a medium for crop production, the soil health framework favors a more holistic view. The USDA-NRCS defines soil health as "the continued capacity of soil to function as a vital living ecosystem that sustains plants, animals, and humans." Thus, in contrast to traditional soil assessment, which primarily focuses on the soil physical and chemical properties relevant to crop production (Schoenholtz et al., 2000; Weil et al., 2003), soil health assessment integrates physical, chemical, and biological analyses and accounts for environmental quality in addition to productivity (Karlen et al., 1997; Bünemann et al., 2018). Continued research on avenues for improving soil health has influenced the USDA-NRCS's establishment of four fundamental soil health principles: (i) maximize living roots, (ii) minimize disturbance, (iii) maximize soil cover, and (iv) maximize biodiversity.

In the United States Midwest, a prevailing conservation strategy that meets all four soil health principles is the restoration of native tallgrass prairie vegetation. Several studies throughout the region have proven that prairie reconstructions on historically farmed land can improve an array of soil health indicators (Baer et al., 2002; Allison et al., 2005; De et al., 2020; Li et al., 2021). However, restoring prairie can take multiple forms, and the rate and extent of related soil health improvements have not always been consistent.

One recently developed iteration of prairie restoration called prairie strips (PS) involves embedding native prairie vegetation within existing row crop fields. Prairie strips are strategically planted within row crop (RC) fields along the contour and at the foot slope and take up 10-20% of a field's area. Research conducted as part of the Science-based Trials of Row-crops Integrated with Prairie Strips (STRIPS; https://www.nrem.iastate.edu/research/STRIPS/) project has shown that PS can deliver multiple ecosystem services, including nutrient and sediment export reduction and increased pollinator abundance, at levels disproportionately greater than the land area that they occupy (Schulte et al., 2017). However, the soil health impacts of PS establishment have yet to be fully described. Given several unique characteristics of PS, such as their proximity to intensely managed RC fields and their role as a sediment and nutrient filter, the soil health response to PS may vary from traditional reconstructions, which convert larger tracts of land to prairie vegetation. In fact, of the limited studies that have investigated the soil impacts of PS, results such as soil C:N ratio decline (Pérez-Suárez et al., 2014) and the absence of a significant microbial diversity response (Flater et al., 2020) have been in contrast to reports common in prairie restoration literature.

Soil health indicator aggregated scoring systems, like Cornell's Comprehensive Assessment of Soil Health (CASH) and the Soil Management Assessment Framework (SMAF) were developed as a response to growing interest in defining and quantifying soil health assessment (Karlen et al., 2019). Both scoring systems have evolved from the framework established by Andrews et al. (2004) to analyze management's impact on soil health with easily-interpreted scores (Moebius-Clune et al., 2016). For each scoring system, the first step in the assessment process is selecting multiple soil health indicators that are relevant to the given situation and which cover physical, chemical, and biological properties. Once indicator values are collected, unitless scores that reflect each indicator's status relative to the optimal condition are generated with functions that account for environmental factors and inherent soil properties. How the functions score each observed indicator value depends on whether it falls into one of three simple categories relating to soil function: more is better (e.g., wet-aggregate stability), less is better (e.g., bulk density or surface hardness), or an optimum range is better (e.g., pH). The higher the generated score, the closer it is to the optimal condition for sustaining productivity, maintaining environmental quality, and promoting biological health. Both CASH and SMAF allow for the scoring of each soil health indicator and combined overall soil health scores. Efforts to refine quantitative soil health assessment have continued with recent developments such as the soil health assessment protocol and evaluation (SHAPE) tool (Nunes et al., 2021) and a proposed pipeline for soil health assessments (Wade et al., 2022).

We used CASH and SMAF to comprehensively compare soil health between PS and RC treatments at four sites in Iowa, USA. Overall, this study involved two main objectives: (i) assess differences in PS and RC soil health, and (ii) compare and explore the effectiveness of the soil health scoring indices, CASH and SMAF, in the context of PS and RC treatment comparison. We hypothesized that the PS treatment would have greater overall soil health scores than the RC treatment. We also anticipated that specific indicators like wet-aggregate stability, active carbon, and extractable phosphorus would be most strongly affected by PS due to either their demonstrated responsiveness to similar land use changes (wet aggregate stability and active carbon – Anderson et al., 2019; De et al., 2020; Li et al., 2021) or the role of PS as a nutrient and sediment filter (extractable phosphorus – Stutter et al., 2009; Helmers et al. 2012; Hernandez-Santana et al., 2013; Zhou et al., 2014). Given CASH and SMAF's shared foundational framework, some level of correlation between the two indices was expected; however, reported inconsistencies in both treatment comparisons and the scaled magnitude of scores (Ye et al., 2021; Crookston et al., 2022) warranted further examination of the indices' performance.

Materials and Methods

Site Descriptions

We compared PS and RC treatments at four sites between 2020 and 2021. At each site, PS establishment occurred between 2014 and 2015 within corn (*Zea mays* L.) and soybean (*Glycine mas.* (L.) Merr.) RC fields. Three sites – ARM, MCN, and RHO – are located within the Southern Iowa Drift Plain landform region, while the remaining site, WOR, sits within the Des Moines Lobe (Figure 4.1). The Southern Iowa Drift Plain (SIDP) has rolling to hilly topography, a loess mantle, and dominant soil orders consisting primarily of Mollisols and Alfisols (NRCS, 2006). At the three SIDP sites described in this study, soil texture ranges from silt loam to silty clay loam. The Des Moines Lobe possesses a relatively young landscape compared to the SIDP. Its most recent glacial episode occurred 12,000 to 14,000 years ago and carved out a nearly level to gently rolling landscape with deep, loamy soils (Prior & Lohmann, 1991). While naturally poorly drained, an extensive network of artificial subsurface drainage within the region enables highly productive RC agriculture. In addition to varying land formation histories, the four sites also differ in RC tillage and grazing management (Table 4.1).

Conventional tillage is practiced at MCN, RHO, and WOR, while ARM employs no-till management. Beef cattle graze at MCN and RHO following the corn phase, while ARM and WOR have no recent grazing history.

Soil Health Indicator Selection

To fully integrate the status of soil structure maintenance, nutrient cycling, and C storage and microbial activity in PS and RC treatments, we selected at least one parameter from each of the physical, chemical, and biological indicator groups available for CASH and SMAF scoring (Andrews et al., 2004; Moebius-Clune et al., 2016). Cost and feasibility of assessment, relevance to the expected changes resulting from PS establishment, and data availability were all considered when selecting indicators. In total, CASH scoring involved the evaluation of nine soil health indicators, and SMAF scoring involved seven (Table 4.2).

Soil Sampling and Processing – CASH

Soil samples collected for CASH were taken in the fall of 2021 at three sites – ARM, RHO, and WOR. Sampling locations were determined in the same fashion as in Chapter 3, with three replications of paired PS and RC sampling points randomly placed within each soil series and phase present at each site (Figure 4.1).

Per Moebius-Clune et al. (2016) recommendations, we collected soil samples by extracting 5 cm by 15 cm soil slices with a spade. At each sampling point, three slices were taken from an approximate 5 m radius around the point and mixed to create a composite 0-15 cm depth sample. The samples were stored in sealed zip-top bags and kept in a 4 ºC cooler before shipment to a commercial laboratory (Cornell Soil Health Lab, Ithaca, NY) for analysis. Simultaneous to soil sample collection, we took three penetrometer resistance measurements using a FieldScout SC 900 Soil Compaction Meter (Aurora, IL) at 0-15 cm and 15-45 cm depths.

The three penetrometer resistance measurements for each depth were then averaged to calculate composite surface and subsurface hardness values for each sampling point. Soil sampling occurred after plant senescence and before any fall tillage. Obvious wheel tracks were avoided to limit the influence of mechanical compaction.

At a commercial laboratory (Cornell Soil Health Lab, Ithaca, NY), soils were analyzed for wet-aggregate stability (AS), pH, extractable phosphorus (P), extractable potassium (K), minor elements (magnesium, iron, manganese, and zinc), soil organic matter (SOM), and permanganate oxidizable carbon or active carbon (AC) (Table 4.2). The rainfall simulator method described in Chapter 3 was utilized to estimate AS, and the percentage of water-stable aggregates (WSA) was calculated. The determination of pH, P, K, SOM, and AC also followed methods outlined in Chapter 3. Modified Morgan's solution was used to extract minor elements, and an inductively coupled plasma emission spectrometer (ICP, Spectro Arcos) analyzed extraction filtrate to determine nutrient values.

Soil Sampling and Processing – SMAF

SMAF scoring was used on soil data from three sites – ARM, MCN, and RHO. Soil health indicators selected for SMAF included AS, bulk density, pH, P, K, soil organic carbon (SOC), and microbial biomass carbon (MBC) (Table 4.2). Procedures used to collect and analyze soil for AS (wet sieving method) and bulk density were identical to those described in Chapter 3. For input into the SMAF program, AS data was represented by percentage WSA.

Soil sampling for pH, P, K, SOC, and MBC occurred in the fall of 2020 after plant senescence and before fall tillage. At each site, soil was collected from the center of the prairie strip and directly upslope from that point 3 m into the RC field from the strip edge. Multiple 0-15 cm depth soil cores were collected with a hand probe and mixed to create a composite sample at each sampling point. Contrary to previous descriptions of sampling location selection, three sampling points were randomly placed at each site rather than within each soil series and phase at each site. Therefore, a total of three PS and three RC paired composite soil samples were taken from each site. Before laboratory analysis, soil was passed through 4 mm and 2 mm sieves and air-dried.

Soil pH was determined using a 1:1 soil to water mixture and a pH electrode probe. A Mehlich III extraction was performed and analyzed with an ICP-OES 7300 Machine (Perkin Elmer, Waltham, MA, USA) to determine extractable P and K. SOC analysis was executed using a ThermoFinnigan Delta Plus XL mass spectrometer attached to a GasBench II with a CombiPal autosampler. Lastly, to determine MBC, twin ~5 g replicates were taken from each soil sample, and one was fumigated with $CHCl_3$. Afterward, both replicates were extracted with 25 mL of 0.5 M K_2SO_4, and non-purgeable organic carbon was measured with a Shimadzu TOC-L analyzer (Shimadzu Corporation, Kyoto, Japan) and compared between replicates (Brookes et al., 1985: Vance et al., 1987). Data from all chemical and biological analyses were acquired from Dutter (2022).

Index Score Calculation

CASH uses an expansive soil database and cumulative normal distributions to assign indicator values a score between 0 and 100. The numerical scores can be grouped into five qualitative rating classes: very low, low, medium, high, and very high, which reflect scores ranging between 0-20, 20-40, 40-60, 60-80, and 80-100, respectively. For specific indicators, the scoring function varies by soil texture, which is categorized into three classes: fine, medium, and coarse. Additionally, regional differences are considered by adapting scoring functions for Major Land Resource Areas defined by the USDA-NRCS (NRCS, 2006). The overall CASH index is calculated by averaging the scores of all individual indicators. Specifics on scoring function development and application can be found in Fine et al. (2017) and Moebius-Clune et al. (2016).

While CASH assigns scores based on empirical data distributions, SMAF uses expert opinion and values taken from published literature to set scoring thresholds (Andrews et al., 2004). When an indicator value is inputted into SMAF, an algorithm or logic statement with alternative algorithms translates the value into a unitless score between 0 and 1. The score adapts to user input that describes the region, climate, mineralogy, soil weathering class, soil texture, organic matter class, sampling time, crop, and analytical methods. The sum of all individual indicator scores is divided by the number of indicators analyzed to calculate an additive overall soil quality index (SQI). For this study, scores were multiplied by 100 to facilitate the comparison between SMAF and CASH. Andrews et al. (2004), Cherubin et al. (2016), and Wienhold et al. (2009) provide more detailed descriptions of the scoring of soil health indicators using SMAF.

Corn Suitability Rating

We acquired corn suitability ratings for three sites – ARM, RHO, and WOR – to provide an alternative perspective on soil assessment. Iowa State University scientists developed the corn suitability rating (CSR) system over several decades to estimate potential soil productivity based on soil survey information and climate in Iowa (Fenton, 1971). CSR values specifically reflect the potential for row crop production and are linked to inherent soil properties in order to remain consistent over time (Craft et al., 1992). In 2013, the original CSR system was revised as CSR2 for better pairing with modern soil mapping services (Burras, 2013). While CSR2 does not assess soil health, we included it in this study because many stakeholders use it to make productivity assessments, similar to how they might use CASH or SMAF. At ARM, RHO, and WOR, CSR2 values were acquired from Web Soil Survey for each soil series and phase to correlate with CASH sampling. An average CSR2 value for each site was calculated. We withheld comparisons of CSR2 and SMAF since the sampling procedure for SMAF was not based on soil series and phase.

Statistical Analysis

All statistical analysis was run using R software (R Core Team, 2020), and plots were generated with the ggplot2 package (Wickham, 2016). Observed soil property values and soil health scores were log-transformed to normalize the dataset and facilitate between-site comparison. We analyzed paired differences between PS and RC treatments using a linear model, and contrasts and comparisons were determined with least-squares means (Lenth, 2020). For CASH, each soil sample was considered a replication and assigned a score for every tested indicator. Treatment differences were tested statistically at both the site and landform region level. Conversely, due to inconsistent sampling procedures, sites were considered replications for SMAF. Therefore, a PS and an RC score were generated for all indicators at each site, reflecting each treatment's average indicator value. We justified using the sites as replications because of the soil type similarity (silt loam to silty clay loam) and the shared land formation history (Southern Iowa Drift Plain) of the three sites. However, this decision meant that each treatment had only three replications, limiting statistical power. Comparisons of the two indices were made at the SIDP landform region level. Statistical significance was categorized as marginal ($p < 0.1$), significant ($p < 0.05$), and strongly significant ($p < 0.01$).

Results

CASH

The PS treatment had a marginally greater overall CASH score than the RC treatment across both SIDP sites – ARM and RHO (Table 4.3). The only Des Moines Lobe site, WOR, was analyzed separately and did not display any significant treatment effect for the overall CASH score (Table 4.4). For individual indicator scores, AS consistently displayed a strongly significant difference between PS and RC across all levels, and the average PS score was 107% higher than RC (Table 4.4). Conversely, treatment differences were scant for all other indicators. Only three other treatment differences were observed: surface hardness scores were greater in RC than PS at WOR, AC was greater in RC than PS at ARM, and SOM was greater in PS than RC at RHO and SIDP sites combined. The average chemical and biological scores for each treatment did not vary qualitatively, as both received very high chemical and low biological ratings. In contrast, the PS treatment scored medium for physical indicators while the RC treatment scored low.

When comparing the observed values of indicators, more significant differences between treatments were evident (Table 4.5). PS strongly affected K as PS was significantly greater than RC at all sites. Additionally, soil pH was marginally greater in PS than RC across all SIDP sites. Significant differences in indicator scores (WOR surface hardness, ARM AC, and RHO SOM) were echoed in comparisons of observed values.

SMAF

SMAF did not yield striking differences in PS and RC treatments. For both treatments, all physical and chemical indicators scored at least 95, suggesting near-complete optimization of soil physical and chemical processes (Table 4.6). Biological scores for both treatments were relatively lower, and the most notable treatment difference was observed in MBC, where PS scored 82 and RC scored 70. The overall SQI for PS and RC treatments was 90 and 88, respectively, so both treatments displayed high overall soil functioning according to SMAF. Although SMAF scored both treatments 100 for AS, analysis of observed values at each site showed a significantly greater percentage of WSA in the PS treatment at ARM and RHO (Table 4.7). No significant differences between PS and RC were detected in observed bulk density values at any site.

Index Comparison

Both indices favored the PS treatment slightly over RC when comparing overall soil health scores. Additionally, both indices generated very high scores for chemical indicators across PS and RC treatments. However, the two indices differed in their assessment of physical and biological indicators. SMAF indicated optimal soil physical functioning regardless of treatment, as both treatments averaged 100 and 99 for AS and bulk density scores, respectively (Table 4.6). Physical indicator scores were relatively lower for CASH, as the average individual physical indicator scores ranged from 12 to 67 (Table 4.4). Also, CASH yielded a significant treatment difference in AS, while SMAF did not. Similar to physical indicator scoring, SMAF generated higher biological scores than CASH for both treatments. Overall, while CASH and SMAF agreed on treatment differences, SMAF estimated much healthier soils than CASH, regardless of treatment.

Discussion

Do Prairie Strips Have Greater Soil Health Than Row Crops?

The CASH and SMAF indices agreed that the PS treatment had slightly greater overall soil health than the RC treatment at SIDP sites. While a lack of replication limited statistical backing for this difference in SMAF, we observed significant results for CASH. When analyzing the individual indicators contributing to the overall CASH score, AS clearly had the strongest response to PS. Macroaggregate stabilization through root and fungal hyphae growth (Tisdall & Oades, 1982; Oades, 1984) and the destruction of larger soil aggregates by tillage and cultivation (Cambardella & Elliott, 1993; Bronick & Lal, 2005) explain this finding. While tillage likely decreased AS in RC at RHO, the treatment difference observed at ARM, a no-till site, highlights the aggregate-stabilizing ability of root and fungal hyphae growth in PS. Although SMAF generated equally optimal AS scores for the two treatments, analysis of the observed values demonstrated a positive PS effect (Figure 4.2a). Since AS can play a pivotal role in mitigating erosion, regulating soil water movement, and facilitating biological activity (Amézketa, 1999), the greater overall scores in PS compared to RC may be indirectly attributed to AS differences. In support of this hypothesis, we found a positive linear relationship ($R^2 = 0.41$) between AS in the PS treatment and the overall CASH score (calculated excluding AS) at SIDP sites (Figure 4.3). At the same time, no correlation was evident between these two variables for the RC treatment. This finding emphasizes the positive influence of PS aggregate stabilization on multiple soil health indicators and suggests that it may be a leading indicator for PS-derived soil health improvements when differences in other indicators remain undetected.

Although PS improved the overall CASH score at SIDP sites, the Des Moines Lobe site, WOR, did not display any treatment differences. While AS was greater in PS than RC at WOR, its contribution to the overall CASH score was offset by a higher surface hardness score in RC than PS. Although tillage can increase soil bulk density and compaction over the long term through SOM depletion and soil structure disturbance (Eynard et al., 2004), the temporary aeration and loosening of surface soil through recent tillage likely caused lower penetrometer resistance for RC at WOR. While RHO also experiences RC tillage, different effects were observed, likely due to RHO's finer-textured soils being more susceptible to compaction (Brady & Weil, 2008).

A crucial aspect of the WOR site that complicated the assessment of its overall CASH scores was the relatively acidic soil pH values observed across treatments (5.51 for PS and 5.55 for RC) (Table 4.5). Soil acidification occurs in RC fields due to sustained N fertilizer application without periodic liming (Mallarino et al., 2011). Since soil pH is often considered a "master variable" due to its control over nutrient availability and microbial activity (Clay & Reitsma, 2009; Rousk et al., 2009; Husson, 2013), many soil health-building processes were inhibited in both PS and RC treatments at WOR. If the WOR site were to be limed and soil pH was assumed to be raised within the optimal range, the average overall CASH scores for both treatments would increase by between seven and nine solely as a result of the new pH scores. Soil pH amelioration would also likely enable other soil processes that could further boost CASH scores at WOR.

Value of Soil Health Scoring

Using soil health indices successfully facilitated the comparison of PS and RC treatments in the context of overall soil functioning and provided helpful context for observed results. In the case of CASH, which analyzed nine soil health indicators, only two indicators, AS and SOM, displayed a statistically significant difference between treatments across SIDP sites. Therefore, by assessing each soil health indicator individually, it would be challenging to conclude overall soil health differences between PS and RC. However, when all individual indicator scores are integrated into the overall CASH score, a treatment difference is apparent at a marginally significant level—pairing this result with the trends observed in Figure 4.3 signals that other indicators may have responded to PS establishment but not at magnitudes large enough to detect individually.

Another benefit of using soil health scoring for this study was that observed values could be more easily interpreted in the context of optimal soil functioning. While a regional expert may easily decipher what the observed value of each indicator means, other interested parties may not possess the background knowledge to comprehend every value fully. An example of CASH scoring facilitating the meaningful interpretation of observed values can be found in comparing extractable potassium between treatments. Although significantly greater K was reported in PS than in RC, both treatments had a CASH score of 100 (Figure 4.2b). Without the CASH score, one might erroneously infer that the RC treatment has depleted K levels to the point of inhibiting soil health.

Limitations of Soil Health Scoring

While CASH and SMAF provided valuable insights into the comparison of PS and RC soil health, several limitations of using soil health scoring in this context became apparent. Although flaws in scoring may exist and warrant correction, these issues were by and large outside the scope of this study. However, a possible overestimation of the effects of AS observed within SMAF should be noted. SMAF generated perfect scores for both treatments, yet a significant difference between observed values was found (Figure 4.2a). A slight methodological modification (0.21 mm sieve rather than 0.25 mm sieve) may have contributed to inflated observed WSA values for this study; however, Nunes et al. (2020) also noted SMAF's probable overestimation of AS effects. In contrast to the very high AS scores that SMAF generated, average CASH AS scores ranged from 16 to 64 (Table 4.5), providing additional evidence that SMAF scores may be overestimated. This disparity in the magnitude of scores observed between

CASH and SMAF was not limited to just AS, as SMAF scored all physical and biological indicators much more favorably than CASH across treatments, resulting in notably higher overall scores. While the two indices agreed when comparing treatments, the differing magnitudes of overall scores complicated assessing the soil health status relative to the optimal condition. This discrepancy was also reported by Ye et al. (2021) and Crookston et al. (2022).

Another issue with soil health scoring in this study was the disagreement between treatment differences in scored and observed values. This pattern, where observed and scored differences are incompatible, was also identified in Andrews et al. (2004). It is imperative to recognize this pattern for specific indicators like soil pH. Even though PS and RC treatments had indistinguishably high scores at SIDP sites, the significantly greater pH observed in PS versus RC has meaningful implications (Figure 4.2c). This result potentially signifies a faster rate of soil acidification in RC than in PS, providing a valuable consideration for future soil pH management and insight on treatment effects. While translating observed values to unitless scores can ease practical interpretation, caution should be taken not to disregard the observed values. Further complications regarding CASH's scoring of soil pH also appeared in a different manner at the WOR site.

At WOR, relatively low soil pH was observed across treatments. Since CASH uses optimum curve scoring for soil pH (Moebius-Clune et al., 2016; Fine et al., 2017) and many observed values landed on the steep section of the curve (Figure 4.4), the range of paired treatment differences for pH scores was substantially greater than any other indicator. Thus, soil pH effectively held the greatest influence in comparing the overall CASH score between treatments and may have nullified differences in other indicators. This observation does not necessarily suggest that CASH's pH scoring method is flawed. However, it emphasizes the importance of analyzing individual indicators and their contribution to overall scores along with the overall scores.

Finally, comparing overall CASH scores and CSR2 ratings emphasized the importance of distinguishing soil health and inherent soil quality. CASH and CSR2 generated completely opposite results when comparing scores between sites (Figure 4.5). Although CASH and CSR2 differ in the breadth of soil functions that they aim to assess, both include a productivity component. As a result, one may be tempted to make productivity assessments based on either of these scores alone. However, these scores mean two very different things. The CASH score reflects how a site has maximized its soil health potential, while the CSR2 rating indicates the inherent potential for row crop production. To make any predictions on productivity metrics like yield based on either of these scores alone would likely be a misguided exercise.

CASH Score Correlations

Overall CASH scores were plotted against individual indicator values at SIDP sites to test which indicators could potentially best predict overall soil health. Data from the WOR site were withheld due to its low soil pH driving overall scores. The strongest relationship was found with SOM ($R^2 = 0.67$) (Figure 4.6). This finding is not unsurprising, as SOM directly impacts many soil ecosystem services and plays a central role in soil health (Karlen et al., 1999; Post & Kwon, 2000). While active carbon has been shown to respond to management more quickly than SOM (Weil et al., 2003) and was predicted to also have a strong correlation with the overall CASH score, its relationship ($R^2 = 0.19$) was relatively weaker than SOM's (Figure 4.7). This finding closely matches a study that utilized CASH to compare tillage treatments in South Carolina (Ye et al., 2021). While it may not be comprehensive, using SOM alone is a moderately strong proxy for overall soil health and can be useful when a parsimonious assessment is required.

Conclusions

This study showed that soil health, as characterized by CASH and SMAF scoring indices, was slightly greater in prairie strips (PS) than in row crops (RC). The driving force behind enhancements to soil health in PS was improved wet-aggregate stability (AS). The positive correlation between AS and the average of all other PS CASH indicator scores suggests that AS is a leading indicator for overall soil health improvements induced by PS. Overall, the scoring framework provided by CASH and SMAF aided the interpretation of soil health assessment in PS and RC treatments with mixed success. While CASH and SMAF agreed regarding treatment comparison, the magnitude of scored values was quite different. Translating observed values to scores was helpful in providing context about optimal soil health conditions, especially for certain indicators such as extractable potassium. Additionally, by integrating multiple parameters, the overall index scores offered a valuable overview of soil health in PS and RC treatments. However, this study emphasized the necessity of analyzing observed values, individual indicator scores, and overall scores altogether, as several important considerations have the potential to be lost at each level. If productivity evaluations are desired, soil health assessments must also be accompanied by inherent soil quality assessments, as the two may suggest varying outcomes. Out of all the soil health indicators analyzed, soil organic matter (SOM) had the strongest association with the overall CASH score and could likely serve as an adequate proxy for overall soil health in similar assessments if feasibility issues limit the examination of other indicators.

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Tables and Figures

Table 4.1. Site characteristics

Site	Dominant soil order	Sand (%)	Clay (%)	Silt (%)	Tillage[a]	2020 crop	2021 crop
ARM	Mollisol	5	30	66	NT	Soybean	Corn
MCN	Mollisol	8	32	60	CT	Soybean	Corn
RHO	Alfisol	3	23	74	CT	Corn	Corn
WOR	Mollisol	42	22	36	CT	Soybean	Corn

Note: Soil texture for 0-15 cm depth acquired from Web Soil Survey.
[a]NT, no-tillage; CT, conventional tillage.

Table 4.2. Soil health indicators selected for Soil Management Assessment Framework (SMAF) and Cornell's Comprehensive Assessment of Soil Health (CASH)

Soil health indicator group	SMAF	CASH
Physical	Macroaggregate stability (AS)	Wet-aggregate stability (AS)
	Bulk density (BD)	Surface hardness (SH)
		Subsurface hardness (SSH)
Chemical	pH	pH
	Extractable phosphorus (P)	Extractable phosphorus (P)
	Extractable potassium (K)	Extractable potassium (K)
		Minor elements (ME)
Biological	Soil organic carbon (SOC)	Soil organic matter (SOM)
	Microbial biomass carbon (MBC)	Active carbon (AC)

Table 4.3. Cornell's Comprehensive Assessment of Soil Health (CASH) average quantitative and qualitative scores in prairie strip and row crop treatments for selected soil health indicators at Southern Iowa Drift Plain (SIDP) sites – ARM and RHO

Soil health indicator[a]	**Prairie strip**		**Row Crop**		**Treatment difference**
AS	53	Medium	28	Low	***
SH	21	Low	12	Very low	-
SSH	65	High	67	High	-
Physical score	46	Medium	36	Low	***
pH	98	Very high	96	Very high	-
P	79	High	77	High	-
K	100	Very high	100	Very high	-
ME	97	Very high	100	Very high	-
Chemical score	96	Very high	94	Very high	-
SOM	17	Very low	13	Very low	**
AC	36	Low	41	Medium	-
Biological score	29	Low	30	Low	-
Overall score	64	High	61	High	*

Note: Asterisks indicate significance of treatment difference (* p <0.1, ** p <0.05, *** p <0.01).
[a]AS, wet-aggregate stability; SH, surface hardness; SSH, subsurface hardness; P, extractable phosphorus; K, extractable potassium; ME, minor elements; SOM, soil organic matter; AC, active carbon.

Table 4.4. Average Cornell's Comprehensive Assessment of Soil Health (CASH) scores for each soil health indicator at each site

		Soil health indicator[a]									
Site	**Treatment**	**AS**	**SH**	**SSH**	**pH**	**P**	**K**	**ME**	**SOM**	**AC**	**Overall Score**
ARM	Prairie Strip	64	30	75	97	82	100	96	32	36	68
	Row Crop	35	31	73	95	67	100	100	29	48	64
		***	-	-	-	-	-	-	-	**	-
RHO	Prairie Strip	38	6	52	100	90	100	100	9	36	59
	Row Crop	16	6	58	99	96	100	100	5	31	57
		***	-	-	-	-	-	-	***	-	-
WOR	Prairie Strip	41	19	70	24	68	100	85	9	17	49
	Row Crop	18	30	65	32	68	100	100	10	16	49
		***	**	-	-	-	-	-	-	-	-

Note: Asterisks indicate significance of treatment difference (* p <0.1, ** p <0.05, *** p <0.01).
[a]AS, wet-aggregate stability; SH, surface hardness; SSH, subsurface hardness; P, extractable phosphorus; K, extractable potassium; ME, minor elements; SOM, soil organic matter; AC, active carbon.

Table 4.5. Average observed physical, chemical, and biological soil properties used for input into Cornell's Comprehensive Assessment of Soil Health (CASH)

		Soil health indicator[a]							
Site	**Treatment**	**WSA (%)**	**SH (psi)**	**SSH (psi)**	**pH**	**P (mg kg^{-1})**	**K (mg kg^{-1})**	**SOM (%)**	**AC (mg kg^{-1})**
ARM[b]	Prairie Strip	38.3	210	224	7.06	3.98	264	3.53	497
	Row Crop	23.9	212	229	6.85	3.53	202	3.48	565
		***	-	-	-	-	***	-	**
RHO[b]	Prairie Strip	24.3	294	295	7.02	19.0	243	2.77	494
	Row Crop	13.9	306	277	6.75	19.2	162	2.49	458
		***	-	-	-	-	***	***	-
WOR	Prairie Strip	26.9	247	243	5.51	2.37	184	2.86	382
	Row Crop	14.8	215	256	5.55	2.40	138	2.86	369
		***	-	-	-	-	***	-	-
SIDP	Prairie Strip	30.7	246	254	7.04	7.25	255	3.21	496
	Row Crop	18.1	252	249	6.81	6.04	192	3.06	519
		***	-	-	*	-	***	**	-

Note: Asterisks indicate significance of treatment difference (* p <0.1, ** p <0.05, *** p <0.01).
[a]WSA, water-stable aggregates; SH, surface hardness; SSH, subsurface hardness; P, extractable phosphorus; K, extractable potassium; SOM, soil organic matter; AC, active carbon.
[b]Southern Iowa Drift Plain (SIDP) sites.

Table 4.6. Soil Management Assessment Framework (SMAF) average scores in prairie strip and row crop treatments for selected soil health indicators at Southern Iowa Drift Plain (SIDP) sites – ARM, MCN, and RHO

Soil health indicator[a]	Prairie strip	Row crop
AS	100	100
BD	99	99
Physical Soil Quality Index	100	99
pH	96	99
P	99	95
K	98	96
Chemical Soil Quality Index	98	97
SOC	56	54
MBC	82	70
Biological Soil Quality Index	69	62
Overall Soil Quality Index	90	88

Note: [a]AS, wet-aggregate stability; BD, bulk density; P, extractable phosphorus; K, extractable potassium; SOC, soil organic carbon; MBC, microbial biomass carbon.

Table 4.7. Average observed physical, chemical, and biological soil properties used for input into the Soil Management Assessment Framework (SMAF)

		Soil health indicator[a]						
Site	**Treatment**	**WSA (%)**	**BD (g/cm^3)**	**pH**	**P ($mg\ kg^{-1}$)**	**K ($mg\ kg^{-1}$)**	**SOC (%)**	**MBC (mg/kg)**
ARM	Prairie Strip	85.0a	1.02a	6.7	16.7	222	1.86	277
	Row Crop	80.6b	0.99a	6.2	13.0	192	1.80	256
MCN	Prairie Strip	83.7a	0.93a	6.6	17.7	151	2.36	439
	Row Crop	83.0a	0.95a	6.2	11.7	129	2.23	325
RHO	Prairie Strip	81.4a	1.05a	7.0	87.7	244	1.68	362
	Row Crop	75.6b	1.08a	6.7	103	195	1.73	338

Note: Only WSA and BD were statistically analyzed. Different letters indicate a treatment difference at the $p < 0.10$ level in a given soil property at the corresponding site.
[a]WSA, water-stable aggregates; BD, bulk density; P, extractable phosphorus; K, extractable potassium; SOC, soil organic carbon; MBC, microbial biomass carbon.

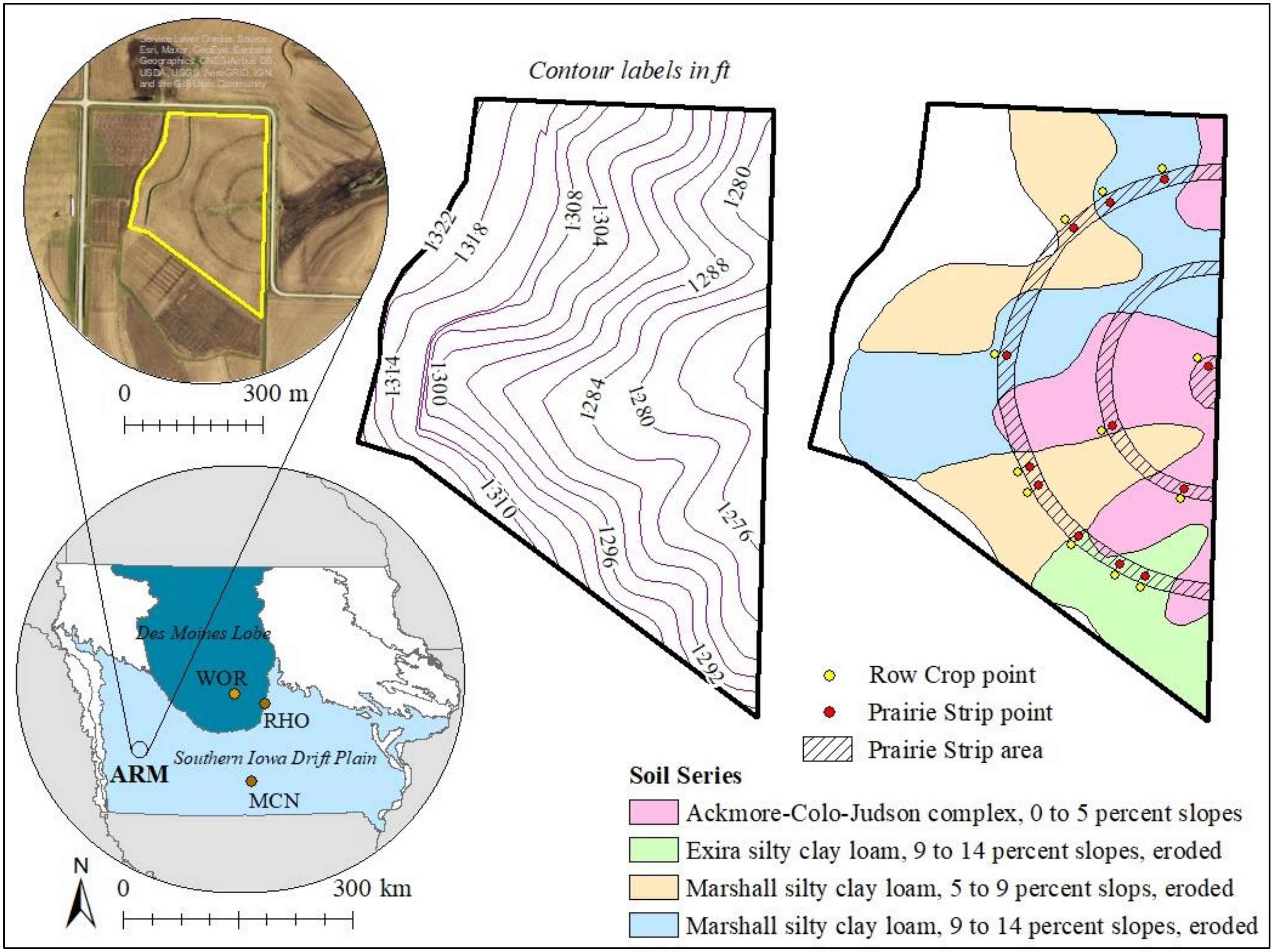

Figure 4.1. Site locations in relation to Iowa landform regions with example aerial imagery, elevation, and soil sampling maps for the ARM site

a)

b)

c)

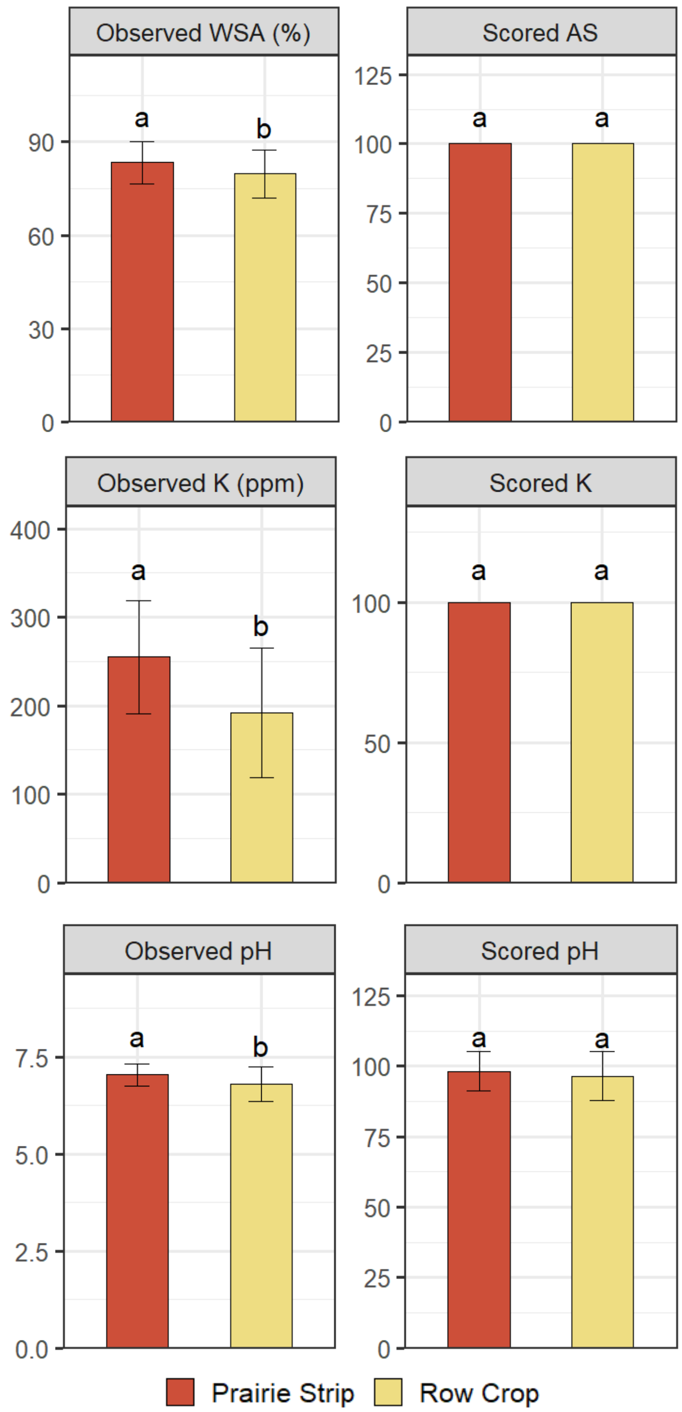

Figure 4.2. Comparisons of average observed and scored values for selected soil health indicators

(a) Soil Management Assessment Framework (SMAF) wet-aggregate stability; (b) Cornell's Comprehensive Assessment of Soil Health (CASH) extractable potassium; (c) CASH pH.

Note: Different letters within each plot indicate a treatment difference at the $p < 0.10$ level. Error bars represent one standard deviation in either direction of the mean.

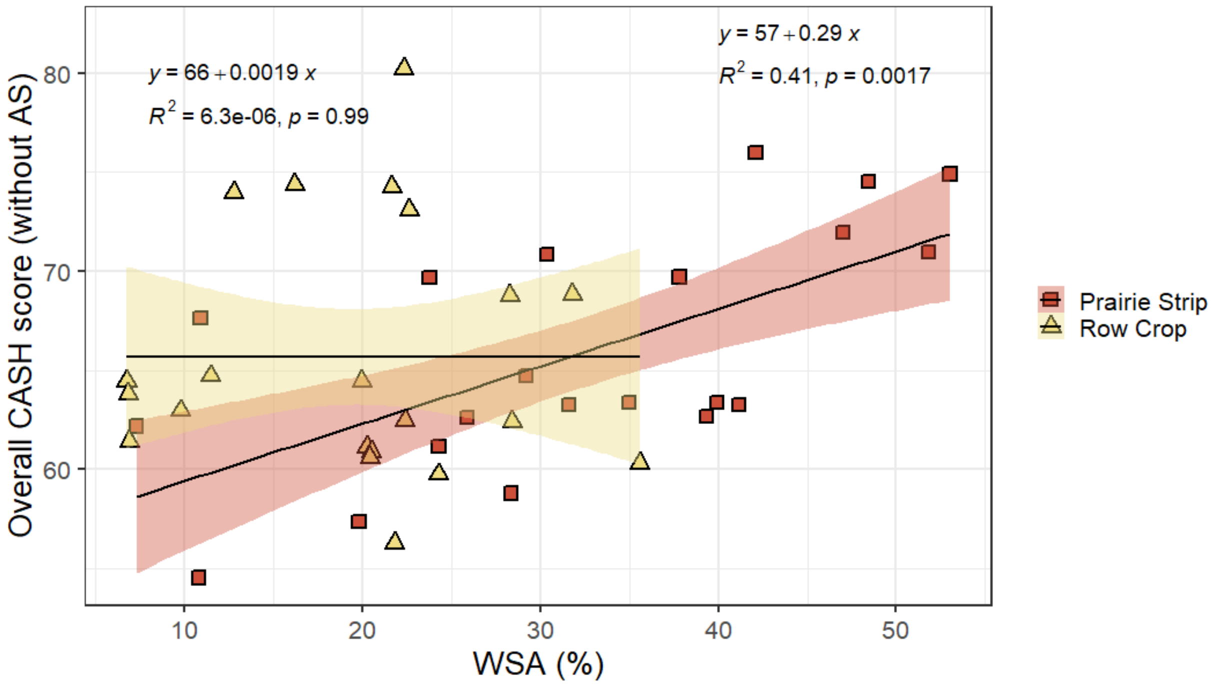

Figure 4.3. Linear relationships between Cornell's Comprehensive Assessment of Soil Health (CASH) observed water-stable aggregates (WSA) values and overall score (calculated without wet-aggregate stability (AS) score) for each treatment across Southern Iowa Drift Plain (SIDP) sites – ARM and RHO

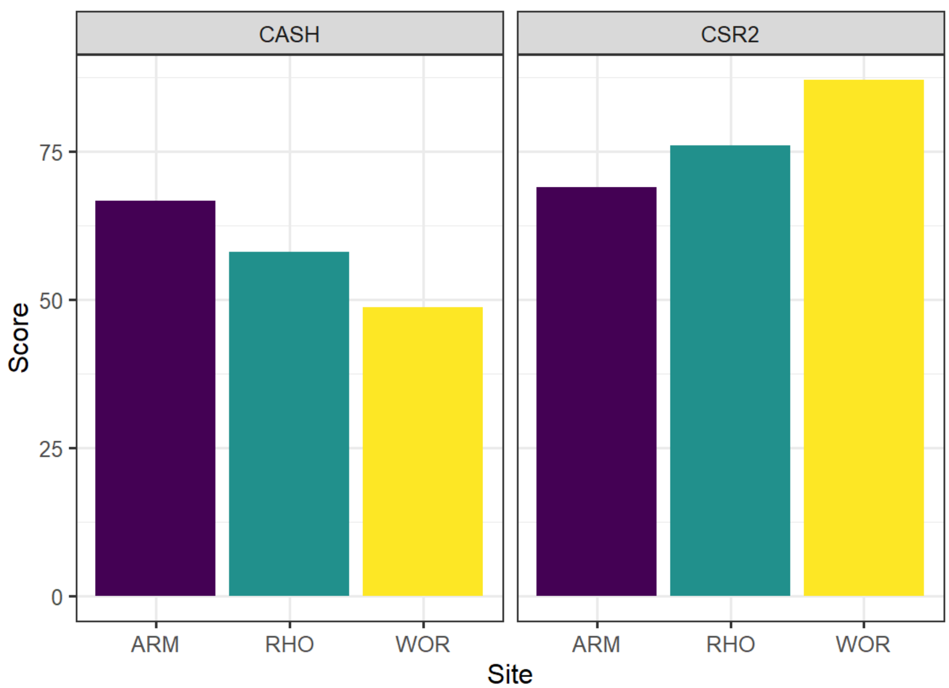

Figure 4.4. Average Cornell's Comprehensive Assessment of Soil Health (CASH) scores and Iowa Corn Suitability Rating 2 (CSR2) scores for three sites

Note: CASH scores are averaged over prairie strip and row crop treatments.

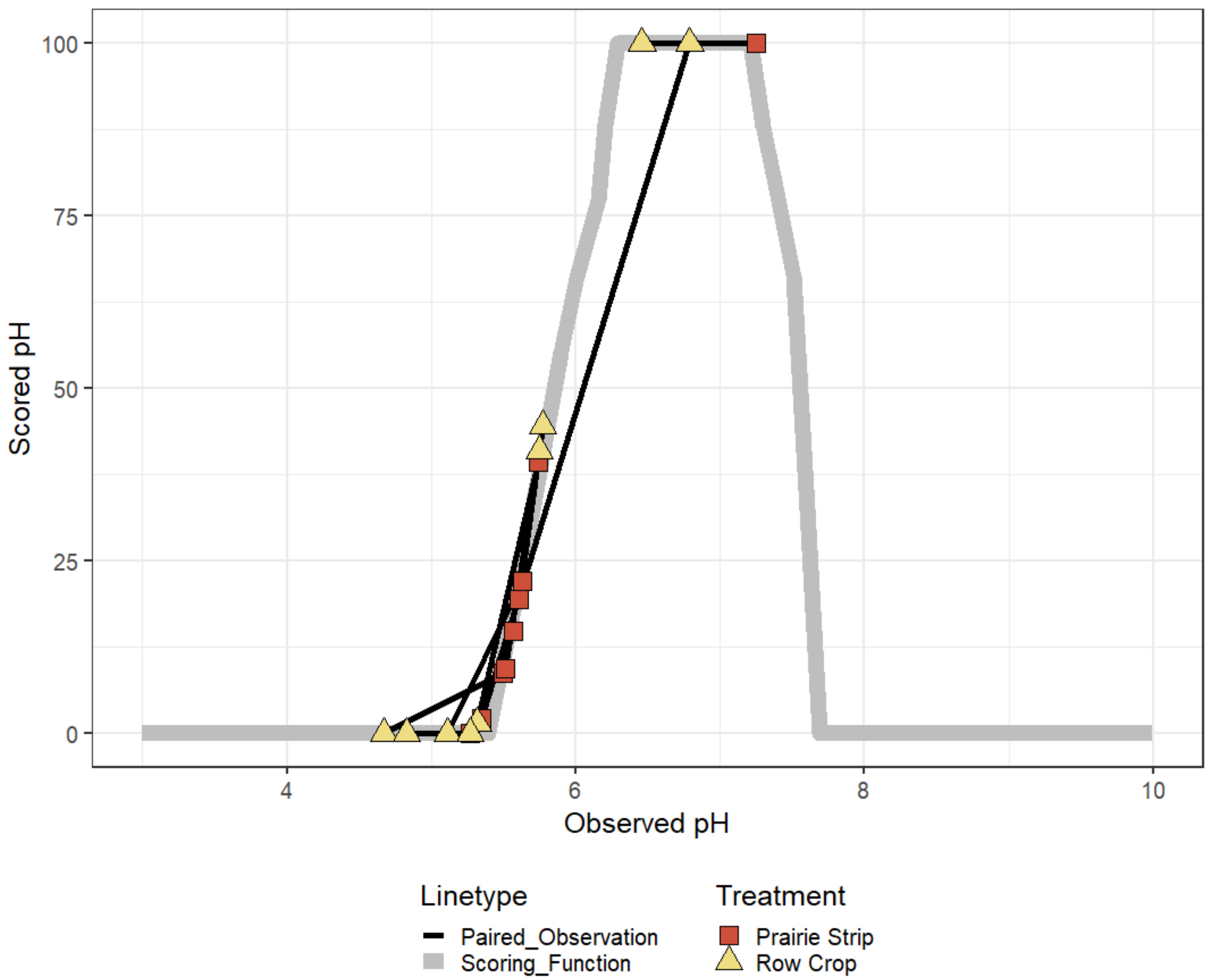

Figure 4.5. Cornell's Comprehensive Assessment of Soil Health (CASH) pH scores and observed pH values for each treatment at the WOR site

Note: The grey line represents the scoring function used in CASH. Black lines connect paired sampling points.

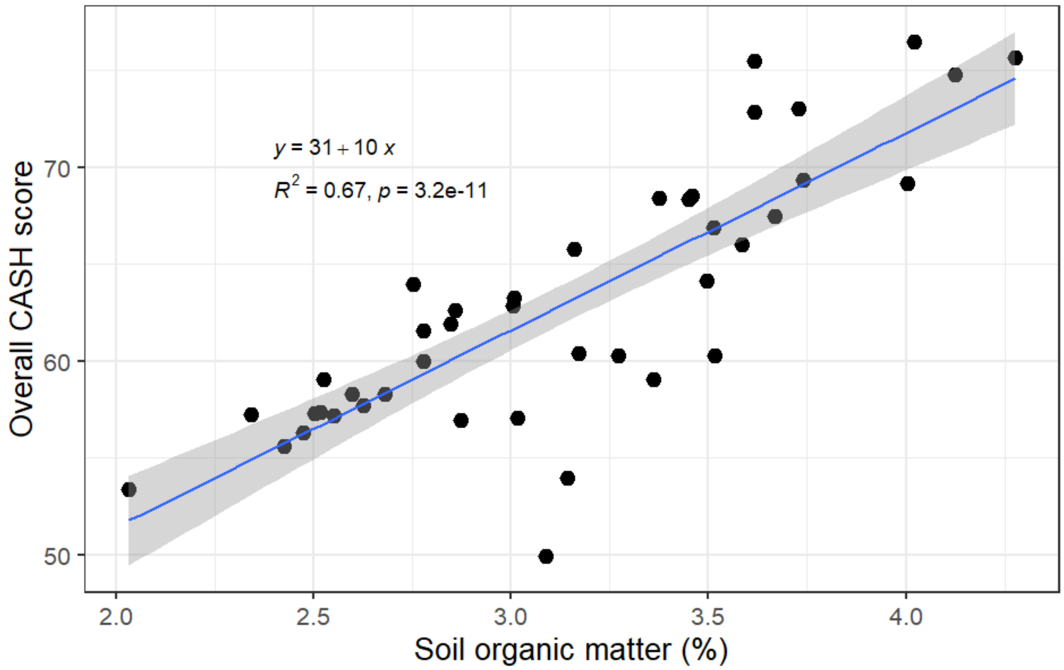

Figure 4.6. Linear relationships between Cornell's Comprehensive Assessment of Soil Health (CASH) observed soil organic matter (%) values and corresponding overall CASH score for each treatment across Southern Iowa Drift Plain (SIDP) sites – ARM and RHO

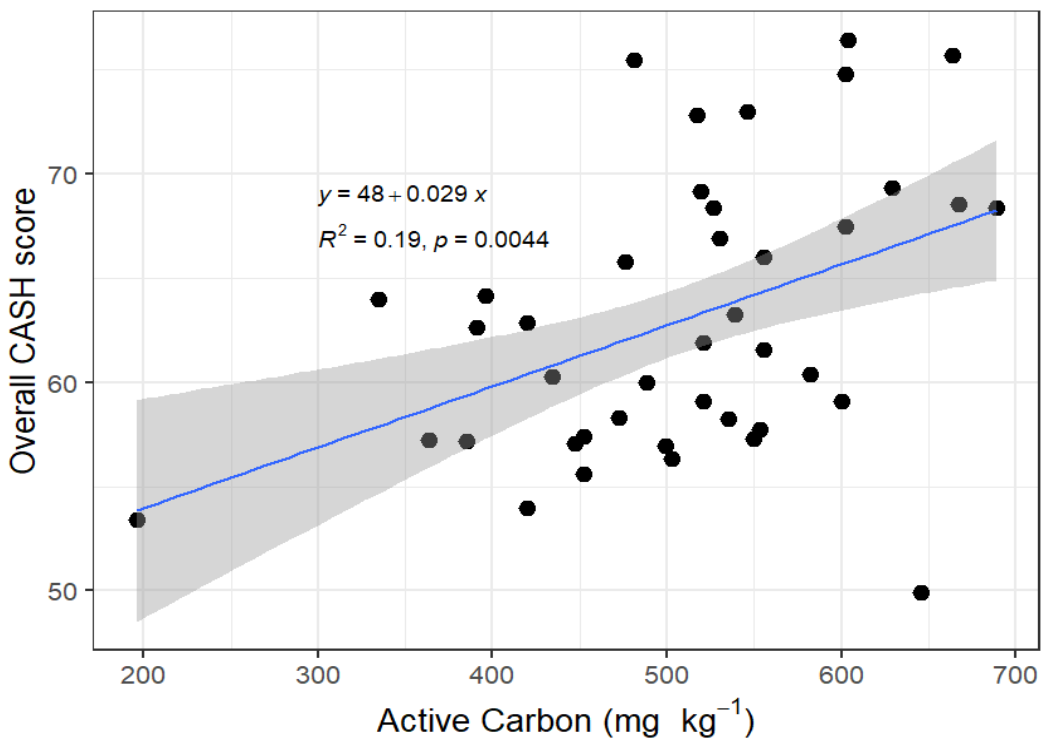

Figure 4.7. Linear relationships between Cornell's Comprehensive Assessment of Soil Health (CASH) observed active carbon (mg kg^{-1}) values and corresponding overall CASH score for each treatment across Southern Iowa Drift Plain (SIDP) sites – ARM and RHO

CHAPTER 5. GENERAL CONCLUSION

While prairie strips (PS) were hypothesized to increase saturated infiltration capacity relative to row crops (RC), this study showed that PS effects after six to seven years since establishment are limited. The strongest effects of PS on saturated infiltration capacity were observed at one seven-year-old PS site, where field-saturated infiltration rates in PS were significantly greater than in RC across three sampling periods. This site was distinguishable from others in that it possessed fine-textured soil, low RC soil organic matter, and conventional RC tillage. Together, these factors likely accelerated the rate of saturated infiltration capacity improvements due to PS establishment. At fifteen-year-old PS sites, hydraulic conductivity results were not especially definitive, but one site displayed greater saturated infiltration capacity in PS than in RC. Although saturated infiltration capacity was unaffected by PS at most sites, sorptivity differences between PS and RC were evident across all sites during fall sampling. Since sorptivity describes early infiltration and has a positive relationship with time-to-runoff, greater sorptivity within PS has positive implications for limiting runoff generation and protecting soil and water quality.

After analyzing a suite of soil health indicators, a connection with infiltration results emerged as the site that displayed treatment differences in saturated infiltration capacity also had significantly greater soil organic matter in PS than RC. However, across all sites and most soil health indicators, significant treatment differences were minimal in general. The most notable and robust soil health response to PS was found in measurements of wet-aggregate stability. Wet-aggregate stability differences between PS and RC increased over time at sites with multiple years of data, and PS wet-aggregate stability was consistently greater than RC wet-aggregate stability at all sites assessed in 2021. Macroaggregate stabilization caused by root and fungal hyphae growth and limited soil disturbance within PS is associated with favorable soil pore distributions for biological activity and water transport. Therefore, even though differences between PS and RC were not as distinguishable for other soil health indicators, it is probable that wet-aggregate stability may serve as a leading indicator for additional changes.

When all soil health indicators were integrated into overall soil health scores using Cornell's Comprehensive Assessment of Soil Health (CASH), PS had significantly greater scores than RC across Southern Iowa Drift Plain landform region sites. The Soil Management Assessment Framework (SMAF) also generated greater soil health scores in PS than in RC. Both scoring indices facilitated the interpretation of the soil health condition in PS and RC. However, the overall scores produced with CASH and SMAF should be accompanied by observed and scored values of individual indicators for more comprehensive and accurate assessments of soil health. Additionally, soil health assessments should be paired with inherent soil quality analysis if productivity predictions are desired. The soil health indicator with the strongest relationship with the overall soil health score was soil organic matter.

Future research should revisit the effects of PS on various soil properties in later stages of PS establishment. While some promising evidence of improvements to soil hydraulic properties and soil health as a result of PS establishment was discovered in this study, it is probable that a new equilibrium state within PS soil at the selected sites has not been reached. Therefore, the full potential of PS to enhance soil within the RC landscape may not yet be fully illustrated. An additional consideration to investigate should be PS plant community composition and its relation to soil. Different seed mixes, environmental factors, and varying management all contribute to a wide spectrum of plant communities at PS sites and may lead to different soil responses. Finally, as it was clear that inherent soil properties and RC management history greatly factored into PS soil effects, analysis at additional sites could illuminate and strengthen potential patterns between site characteristics.