DISSERTATION

SOIL HEALTH FOR THE SEMI-ARID WEST: A NEXUS OF AGRICULTURAL SOIL MANAGEMENT AND ECOSYSTEM SERVICES

Submitted by

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In partial fulfillment of the requirements

For the Degree of Doctor of Philosophy

Colorado State University

Fort Collins, Colorado

Spring 2025

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ABSTRACT

SOIL HEALTH FOR THE SEMI-ARID WEST: A NEXUS OF AGRICULTURAL SOIL HEALTH AND ECOSYSTEM SERVICES SUSTAINABILITY

Our agricultural systems face increasing pressure to simultaneously intensify their operations while reducing the unintended environmental consequences of production. In the semi-arid western US, the most pressing concerns for the continued sustainability of agriculture include: 1) maintaining a healthy topsoil for reliable productivity, 2) mitigation of water pollution from agricultural runoff, and 3) ensuring profitability of farms to maintain or improve quality of life for producers. These concerns have been studied individually but have rarely been connected empirically. Even fewer studies have attempted a holistic, systems-wide approach to soil health management in semi-arid regions, where irrigation is critical to maintaining robust production. To address this gap in the current research, I performed a series of soil health assessments across five sites in Colorado, USA using the Soil Management Assessment Framework and connected these soil health measurements to indicators of runoff water quality, water conservation, or economic welfare.

These sites were a combination of small and medium-sized farms operated by either research staff at research farms or by small farmers on private land. The objective of the research was to monitor soil health on sites implementing Best Management Practices (BMPs) for soil or water conservation and evaluate the ecosystem service impact of these practices. Evaluated BMPs included: 1) conservation tillage under furrow irrigation, 2) transition from furrow

irrigation to sprinkler irrigation, 3) installation of a vegetated filter strip, and 4) use of management-intensive grazing. The soil health impact of these BMP's was mixed; at some sites the long-term reduction in intensity of tillage had positive effects on soil health, whereas on others, the management of the field under deficit irrigation resulted in significant salinization of the soil subsurface.

At the long-term conservation tillage site, empirical connections were established between edge of field water quality measurements and soil health indicators to identify that in these furrow-irrigated systems, improvement of both soil health and water quality can be achieved through improvement of infiltration and protection of soil aggregates. In irrigated systems, improvements of soil functioning to retain and store water is a critical ecosystem service. Furthermore, at this site, the economic impact and greenhouse gas mitigation potential of adopting conservation tillage with cost-sharing or carbon (C) offsetting and selling was assessed using a 12-year enterprise budget analysis.

Results indicated that over the long-term, conservation tillage may be more profitable than conventional tillage, particularly when funding incentives are used to offset the early costs of adoption. Taken together, the results of these multiple studies indicate that management for water conservation may indeed improve soil health and economic outcomes, but continued monitoring of the soil system is necessary. This approach provides a blueprint for future systems-wide studies of conservation agriculture, which should consider hydrologic, agronomic, and socioeconomic impacts.

ACKNOWLEDGEMENTS

I would like to graciously thank the dedicated work of the many people who made this dissertation possible. The many hours of revising manuscripts, mentorship, long lab days, and exhausting field work have all contributed to the effort of this dissertation, and I am incredibly grateful. I would like to particularly thank my committee: Jim Ippolito, Erik Wardle, Daren Harmel, and Andy Seidl. Your tireless efforts have been immensely important and will always be appreciated. I would also like to thank the Agricultural Water Quality Program at Colorado State University. Troy Bauder, Emmanuel Deleon, AJ Brown, and Christina Welch all supported this work through their efforts and the many undergraduates who spent their summers in a corn field with me are appreciated. Finally, I would like to acknowledge the farmers who offered their land, their time, and their expertise, and their lemon cakes for this project. In particular, Karl Whitman and the staff at CSU's ARDEC Facility were critical in the success of this work. Thank you.

Funding for this work came in part from the Colorado Department of Public Health and Environment's Grant: Water Quality, Soil Health and Regenerative Agriculture: A Nexus for Sustainability

DEDICATION

I would like to dedicate this work to my family and loved ones who supported me over the years.

To the ones who could not celebrate with me, I miss and love you.

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CHAPTER 1 – INTRODUCTION

Environmental protection in the context of production agriculture is increasingly becoming a topic of political, social, and economic importance. Effective and well-informed communication between legislators, consumers, and producers is essential to developing practices and legislation that supports a shared mission to reduce the negative impacts of production agriculture while maintaining yield to feed a growing population. However, these lofty goals are not without their challenges, such as maintaining or supplying essential crop nutrients as well as supporting soil health practices that create resilient, sustainable agroecosystems. Moreover, sustainable intensification of our agricultural systems is tantamount to balancing the need for highly productive agricultural land with our environmental goals. We must find a way to obtain high yields on increasingly limited land while limiting the negative externalities of agricultural production.

For example, the same nutrients that are essential for plant growth may accumulate in water bodies - due to runoff or leaching - and pose a risk to human and environmental health. Water quality damage from agriculture presents a threat to global biodiversity, oceanic systems, and human health. Excess nutrients from fertilizer may be carried off fields by precipitation, irrigation, or wind movement, and eventually reach natural water bodies. These nutrients (e.g., N, P) are readily consumed by algae and cyanobacteria, contributing to large algal blooms that cause eutrophication and fish kills globally (Guo et al., 2019). The Food and Agriculture Organization of the United Nations released an executive summary that described “a global water quality crisis” and outlined the significant contribution of agriculture to declining water quality (Mateo-Sagasta et al., 2017). Agricultural intensification, through increased use of inputs such as pesticides, fertilizers, and irrigation water, has spurred a rapid growth in crop production, but also puts water bodies at risk of significant pollution. Pesticides and nitrates in drinking water also pose human health risks; some countries have banned persistent pesticides. Water pollution from agriculture is a serious and persistent concern for agricultural development going forward and must be incorporated into sustainable agricultural policies and practices.

The most pressing concerns for agricultural sustainability include: 1) maintaining a healthy topsoil for agricultural production, 2) mitigating water pollution from agricultural runoff, 3) reducing the greenhouse gas intensity of agricultural operations, and 4) ensuring profitability of farms to maintain or improve quality of life for producers. The challenge for future agricultural researchers and producers lies in balancing these four needs, and if possible, identifying a set of management practices that improves all of them. A variety of soil management practices have been developed to address these challenges, many of them focusing on soil health management. The lens of water quality improvement may provide some insight. Best Management Practices, or BMPs, are a selection of tools and management strategies designed to minimize environmental degradation from agricultural practices (Tamini et al., 2012; Deleon et al., 2020). These BMPs are often designed to improve runoff water quality or minimize negative impacts of agriculture (Tamini et al., 2012), but it has been hypothesized that these practices also have the potential to improve soil health.

While the connection between water quality and soil health has been linked conceptually, there is little research that empirically connects them. Soil health is highly contextual and site-specific (Andrews et al., 2004). As a result, most research that does exist focuses on regions with relatively high precipitation, while little research exists that connects these concepts in the semi-arid west, where most producers rely heavily on irrigation to meet crop water needs. Given the ecological importance of rivers in the semi-arid west and, maintaining high quality surface water is increasingly important. Estimations indicate that 320-350 billion gallons of water are withdrawn daily in the US, with 37% of this going to irrigation. This water use is not shared evenly across the country; over 50% of withdrawals came from just 12 states, of which Colorado is one (Dieter et al., 2017). Improving the quality of water that leaves fields after irrigation is critically important for future use. Several emerging technologies focus on treating wastewater or effluent and making it clean enough for human consumption or non-consumptive use. However, in-field practices offer an opportunity to both produce cleaner agricultural runoff, reducing treatment costs, and to retain nutrients and topsoil on the field, leading to potential maintenance or improvements in soil health.

Healthy soils confer numerous benefits to the landowners who manage them and society as a whole (Turmel et al., 2015; Drobnik et al., 2018), yet these benefits are difficult to capture in traditional markets. Natural resource economists recognize the fundamental value of fertile, healthy soils for agricultural production. Concomitantly, global loss of soil fertility is well-documented and is a growing concern for both environmental and economic reasons (Barry, 2015). Thus, maintaining or improving soil health is generally viewed as a way to both improve farm profitability and ensure environmental quality standards. Thus, the premise of my study aims to investigate the soil health impacts of BMPs within irrigated agroecosystems, the intimate connection between BMPs, water quality, and soil health. I further investigate the economic implications of conservation tillage techniques in semi-arid systems and compare multiple compensation schemes for adopting conservation tillage.

1 BEST MANAGEMENT PRACTICES

Soil and environmental scientists have long recognized the potential connection between soil health and water quality. However, few studies have comprehensively connected dynamics of soil health and water quality to better understand total ecosystem impact (Zimnicki et al., 2020). Furthermore, while the efficacy of BMPs in improving water quality is frequently researched, the impacts of these practices on soil health is less well-understood, and there is little research mechanistically connecting soil properties to effluent water quality. Advanced evaluation of BMPs should incorporate sediment and pollutant loading reductions, while also including soil health changes as an important factor. Research exists that evaluates these BMPs, but evaluation rarely considers ecosystem impacts beyond water quality mitigation.

Even fewer studies have examined the relationship between soil health and water quality in semi-arid environments like Colorado. In these systems, evapotranspiration exceeds precipitation considerably, particularly in the growth of crops like corn and alfalfa which are frequently grown in Colorado and the western US. Consequently, much of the state relies on irrigation. Best Management Practices are a wide variety of structural or cultural practices designed to reduce water pollution from agricultural processes or otherwise minimize the negative impact of agriculture on the environment.

Farmers may choose to implement BMPs for financial, social, or personal reasons. This study aims to not only evaluate the impact of these BMPs on soil health, but on how multiple ecosystem services provided by healthy soils – water filtration, sediment stabilization, greenhouse gas emissions reductions, and provisioning of crop growth – interact with each other. Understanding the global potential for BMPs to reduce water quality impairment requires an understanding of the economic incentives and impacts of BMP management. For this reason, a cultural and holistic analysis may best be attempted with a series of observational studies that aim to evaluate BMPs as they would be managed by producers. While controlled research studies, such as those at Kerbel Tillage (located at Colorado State University's ARDEC, 40°40'40"N 104°59'51"W, elevation = 1567 m), have the potential to more robustly document the direct impacts of BMPs on environmental quality, observational studies on farms potentially reflect "actual" farm practices. The beliefs and perspectives of land managers can be somewhat occluded on research farms, where existing grant infrastructure and reliable funding results in financial outcomes playing a smaller role in management decisions.

Throughout this study, I evaluated the soil health and water quality on several farms using a varying selection of BMPs to attempt to answer the question: how do agricultural BMPs affect soil health? Furthermore, if the soil health, water quality, and/or ecosystem service capacity improves, are there ways for producers to capture this value on the market? Ultimately, this study aims to investigate what additional benefits are conferred from BMPs beyond the intended water quality improvements. This investigation is valuable not only because it aims to quantify soil health impacts of BMPs, but because it may help lead to better *informed* adoption of BMPs.

To answer these questions, I worked with local producers across Northern Colorado and Colorado State University's ARDEC and Arkansas Valley Research Center's Research Farms to establish study sites where I monitored soil health and water quality changes with the implementation of BMPs. When working with private producers, these BMPs were selected and implemented ahead of time by the producers, solely on a voluntary basis. The BMPs studied during this project include conservation tillage and irrigation delivery method. At times, farmers include other BMPs as additional measures on the same field, including, but not limited to conservation tillage and nutrient management.

1.1 Soil Health as a Best Management Practice Outcome

Increasing attention is being paid to soil health as a secondary farm management goal. Stakeholders, farmers, and public officials are becoming more aware of the role healthy soils play across landscapes and farms. Soil health, oftentimes associated with or used interchangeably with the term “soil quality”, can largely be described as the capacity of a soil to fulfill a series of ecosystem or structural services. The specific target ecosystem services are site-specific and contextual to the intended or natural function of the soil (Andrews et al., 2004). Generally, a healthy soil fulfills those services that are essential for human, animal, and plant health. In the context of agriculturally productive soils, these services include, but are not limited to: water capture, infiltration, and storage; nutrient retention and cycling; support of a diverse and healthy microbial community; heavy metal or otherwise toxic pollutant retention and buffering; and maintenance of a biodiverse agroecosystem (Drobnik et al., 2018). The capacity of our soils to perform these key functions is threatened by soil erosion, organic matter and C losses, and pollutant contamination (Andrews et al., 2004), all of which are both direct and indirect impacts of common agricultural practices. Yet, as land managers of vast swathes of ecosystems with high primary productivity and active nutrient cycling, farmers have the potential to generate a huge positive impact on environmental quality. Recognizing this potential requires that we also acknowledge that soil functions are highly tied to broader ecosystem services (Drobnik et al., 2018). Intentional management of soils then offers an avenue to manage agroecosystems for ecosystem service benefits; several management practices that may improve soil health and environmental quality are outlined below.

1.2 Conservation Tillage

A number of BMPs, including conservation tillage, exist to mitigate water quality and erosion concerns on agricultural land. Conservation tillage describes the practice of leaving as much residue on the field as possible and minimizing practices or operations that disturb or "turn over" the soil. The USDA-NRCS has released several documents describing conservation tillage practices, estimating remaining residue cover from various tillage practices, and giving recommendations for improved residue retention (USDA NRCS, 2016). Historical practices of tillage heavily disturb the soil and many "invert" the top soil layers, exposing the nutrient-rich, moist subsoil to intense weathering processes. Long-term and repetitive tillage results in the destruction of soil structure, loss of organic matter, evaporation of stored soil water, increased sediment concentration in runoff, and overall loss of soil fertility. Generally, conservation tillage aims to minimize soil disturbance and destruction, leaving at least 30% of crop residue on the field (Wardle et al., 2015). Residue on the soil surface intercepts moving water and wind that would otherwise dislodge soil particles and lead to erosion. Conservation tillage has been proven to reduce sediment and nutrient loads and concentrations exiting fields, particularly in systems with high runoff potential (Uribe et al., 2018; Liu et al., 2017; Dinnes et al., 2004). In addition, leaving residue on the soil surface helps increase soil organic matter; increased organic matter on/near the soil surface and in the subsurface has been shown to increase infiltration rates and thus reduce runoff (Busari et al., 2015). As a result, more water is captured and stored for subsequent plant use and less is lost as sediment-rich and nutrient-rich runoff. Increased organic matter also increases soil water holding capacity and improves aggregate stability, which in turn increases water infiltration rate and hydraulic conductivity in soils (Busari et al., 2015). These many processes together support the belief that conservation tillage has the potential to reduce erosion and water waste on irrigated land (Uribe et al., 2018).

Conservation tillage may also provide additional benefits to soil health (Deleon et al., 2020). At the Kerbel Tillage site at Colorado State University's Agricultural Research Development and Education Center (ARDEC), past experiments have shown an improvement in key soil quality parameters. This data supports other research on similar practices (e.g., Busari et al., 2015), indicating an improvement in macrofauna abundance, organic C, microbial community diversity, nutrient retention and fertility, infiltration rate and capacity, bulk density, and aggregate stability (Deleon et al., 2020). We used this site as part of a multi-year study on soil health, water quality, and soil microbial community structure. Our project benefits from the fact that inputs, harvest, and management practices are thoroughly documented, allowing us to perform a series of enterprise budget analyses on various tillage practices. The first of these budget analyses is an "actual" budget, incorporating the realized costs and benefits of each management style. I then consider the availability of C credits and government incentives to evaluate the potential additional benefits that could be realized if producers are able to access these funds. Together, these scenarios aim to document the potential for improved profitability on farms that decrease tillage practices. This improved profitability is often derived from decreased input costs with competitive yield, a well-documented occurrence. The potential for improved profitability increases when we introduce C credit markets and government incentives to help producers realize economic benefits from the environmental benefits conferred from best management practices. As we revisit this site over time, we hope to elucidate more information about the impact of conservation tillage on water quality and soil health.

1.3 Conversions to Sprinkler Irrigation

In semi-arid, water-limited Colorado agroecosystems, irrigation is usually needed to meet water needs for successive crops. Past studies have shown that irrigation practices that optimize water use efficiency also improve effluent water quality by minimizing runoff. This is unsurprising, reducing the quantity and intensity of applied irrigation water should logically reduce the intensity and quantity of runoff. In Colorado, 53% of irrigated land is under flood-furrow irrigation (Dieter et al., 2017), which presents a challenge for water quality. Flood-furrow irrigation relies on excess of applied water to ensure consistent delivery to the root zone on the bottom end of the field. The dynamic between proximity to the top of the field and water infiltration depth is well-documented and presents a challenge to producers aiming to maximize homogeneity while minimizing water over-application. Furthermore, these practices pose significant runoff risk at the end of the field and may exacerbate soluble nutrient leaching at the top of the field where water infiltrates most deeply. It is important to note that relatively high organic matter content in subsoil has the potential to improve water retention in some soils and thus reduces the rate of deep leaching. Concomitantly, many of the BMPs examined in this study should lead to increase soil organic matter content, pointing to the possibility that a selection of multiple BMPs may be best to help meet environmental challenges on furrow-irrigated farmland.

Irrigation is widespread in Colorado, with 87% of recent water withdrawals going towards irrigation (Dieter et al., 2017). Irrigation water is often applied in methods that optimize simplicity in management or require little infrastructure. These practices are often poorly optimized for water application efficiency and some produce excess runoff as a planned part of management-allowed return flows. A 2015 US Geological Survey found that of the 3.04 million irrigated acres in Colorado, roughly 53% were flood irrigated and 47% were sprinkler irrigated, with a very small number of acres utilizing drip irrigation systems. Compared to national totals of 37 and 55% under flood and sprinkler irrigation, respectively, Colorado has more flood irrigation and less sprinkler irrigation (Dieter et al., 2017). Implementation of sprinkler systems or high-efficiency irrigation systems has been identified as a BMP to improve water quality and potentially reduce the detrimental effect of flood irrigation on soil health (Perry, 2012). In addition to using less water than furrow irrigation to achieve the same plant available water in the root zone, sprinklers and drip systems produce more homogeneity in applied irrigation and the method of application reduces the dislodging and subsequent exportation of sediment. The ability to control the flow rate and kinetic energy of irrigation using a sprinkler system allows for farmers to irrigate in ways that prevent soil erosion, potentially leading to long-term soil health maintenance or improvements. Due to a complicated set of water laws in the American west (which are too extensive to detail in this paper), there is little direct financial incentive to improve water application efficiency, though social pressure to improve water efficiency is likely to climb in the future, as the southwestern United States continues to experience a long drought and a boom in population growth. Given the potential impact of population growth and development on soil health, there is increasing pressure to find agricultural practices that simultaneously improve soil health and improve water management outcomes.

Sprinkler-delivered irrigation water interacts with the soil surface in a manner more consistent with rainfall. Sprinkler irrigation uses less water than flood-furrow irrigation, with water use efficiencies ranging 75-90% and 20-50%, respectively (Bauder et al., 2014). Furthermore, the aerial application of water that covers the soil surface in a more homogenous way results in water delivered more directly to the crop and less runoff. What little runoff does occur tends to carry less sediment and nutrients than flood-furrow irrigation, in part due to field hydrology (Perry, 2012). In flood-furrow irrigation, the surface soil is susceptible to the concentrated, high velocity flow of most furrow water. Conversely, sprinkler-delivered water does not rely on a rapid lateral velocity to deliver water to the bottom of the field, and much of the water delivered is infiltrated very close to where it falls, with little water running off the field. The result of this dynamic is that furrow irrigation produces erosion and deposition wherein fertile topsoil is eroded from the top of the field and deposited at the bottom (Ippolito et al., 2017). Well-managed sprinkler irrigation, on the other hand, does not cause overland flow and topsoil is not continually transported downfield. Transition to sprinkler irrigation has the potential to improve soil health, particularly at the top of the field, ending the erosional effect of furrow irrigation. This dynamic has been investigated by Ippolito et al. (2017), citing soil health improvements in semi-arid fields that were transitioned from furrow to sprinkler irrigation within five to eight years. Taken together, the water quality and soil health impacts of converting from furrow to sprinkler irrigation indicate the co-benefits of BMPs, but this impact has not been extensively studied together on the same field at the same time. We studied this conversion at Lempka Farms from 2021 (year 1 of sprinkler irrigation) to 2023, comparing it to a similar site in southern Colorado in year 5 of a furrow-to-sprinkler transition.

2 SOIL CARBON, SOIL HEALTH, AND ECONOMICS

Soil C is one of the most important indicators of soil health (Paustian et al., 2019). Soil C is associated with increased aggregation and infiltration, increased cation exchange capacity, and supporting beneficial microbial communities, among other soil health benefits (Blanco-Canqui et al., 2013). Benefits of soil C to soil properties are broad and well-documented, and multiple studies have shown that C sequestration and storage in soils contributes positively to soil health.

The role of soils in C sequestration is becoming an increasingly important subject in global land management (Paustian et al., 2019; Six et al., 2000). Generally, agricultural intensification has been associated with the net loss of soil C. Heavy tillage, nutrient excess resulting in C mineralization, residue removal, and the conversion of wetlands and forests to agricultural land are direct consequences of agriculture that result in a loss of C from global soils (Lal et al., 2004). Historical agricultural land use has resulted in an estimated loss of 116 Pg of C from the global soil C pool (Sanderman, Hengl, and Fiske, 2017). However, the ability of soils to store C is postulated as one potential way to partially mitigate increases in atmospheric C. The technical potential of global soils to sequester C is different from the economic potential, with technical potential exceeding economic potentials (Smith et al., 2020).

A number of BMPs have been shown to increase soil C, particularly the conversion from heavy tillage to conservation tillage or no-till (Lal et al., 2004). No-till C sequestration may be due to several factors. Six et al. (1999) showed that the longer life cycle of macroaggregates and formation of intra-aggregate microaggregates in no-till systems leads to sequestration of soil C, particularly noting that particulate organic matter C (POM-C) within aggregates drives this sequestration potential.

2.1 Carbon Offsetting and Carbon Credits in Agricultural Contexts

Soil C sequestration has emerged as a way to fight global climate change, and consumers are increasingly using their pocketbooks to voice their agenda for reduced C emissions. Subsequently, a market has arisen for C emissions to be traded as if they were a commodity. Public pressure and marketing campaigns have resulted in the creation of C credit markets where emitters of C (typically industrial users) can “purchase” a certain amount of C to be emitted. The credits are traded to C sinks, which is usually a land manager looking to adopt a practice that sequesters more C than their current practice. This C market trading allows farmers and land managers to recoup some losses from changing practices, and C emitters are able to market their goods as being less C-intensive, or even C neutral. As C markets become more accessible and globally relevant, they represent a potentially significant source of income for land managers.

Carbon markets often require minimal or no soil C fractionation work and use existing greenhouse gas accounting models to inform sequestration potential and C flux (Downey, 2023; Sellers et al., 2021). However, most global regulations on greenhouse gas abatement require sequestered C to remain sequestered for at least 100 years (Sanderman et al., 2013). Given the variable turnover rate and life cycle of soil C, C credits potentially poorly capture actual *long-term* C sequestration dynamics. Most C crediting programs attempt to address this issue by verifying increases in soil C stocks and land management practices. However, this level of soil C analysis does not provide information on soil C pools and turnover dynamics.

To better understand the relationship between BMP adoption and actual soil C sequestration and stabilization potential, we will perform a size fractionation of soil organic C pools. A review of many fractionation processes indicated that separating organic matter based on size allows us to appropriately estimate turnover time in organic matter pools as they respond to management practice (Leuthold et al., 2022; Popelau et al., 2013; Six et al., 2001). Since our primary research goal is to understand how management practices impact soil health properties, with soil C fractions as an indicator of soil health, a particle size fractionation seems most appropriate to ascertain these pools. Through this analysis, we hope to highlight the potential gap between compensated C sequestration credits and actual realized C sequestration. Furthermore, we hope to investigate the relationship between different organic C pools and measured soil health metrics, as well as potentially correlating C pool fractions with water quality metrics, informing our understanding of the soil-water relationship on lands under various BMPs.

2.2 The Economics of Soil Conservation

Improvements to soil health and water quality have intrinsic environmental values that are poorly captured in the open market. However, it is understood by both economists and ecologists that these non-market values are important to producers, localities, and global environmental quality. The improvement of soil health and water quality has the potential to improve an ecosystem's overall service value. The value of these improvements is recognized internationally: conservation agriculture, as defined by the Food and Agricultural Organization of the U.N., aims to minimize mechanical and chemical disturbance of soil, maintain soil fertility, and utilize crop rotations (Barry, 2015) with the goal of promoting healthier soils. Healthy soils help to filter pollutants before they enter water bodies, provide habitats for diverse flora and fauna, sequester carbon, provide flood control, cycle nutrients, stabilize ecosystems, and provide other key ecosystem services (Daily et al., 1997). However, translating ecosystem services into economic or financial benefits is challenging.

Scant literature exists that focus on the economics of soil health and water quality improvements. To translate ecosystem services into economic/financial benefits, a number of institutes and consortiums are developing models to compensate producers for C sequestration, water quality improvements, and the general improvement of ecosystem services, hence internalizing the positive externalities of conservation agriculture. Groups like the Soil Health Institute and the Ecosystem Service Market Consortium (ESMC) have evaluated and developed models to help producers understand how to translate management practices into economic impact. The ESMC has even developed a voluntary market for carbon credits, water quality credits, and biodiversity credits. This non-profit organization launched its market system in 2022, where municipalities, industrial companies, and food and beverage companies can purchase carbon credits and water quality credits to meet conservation goals (Informa Agribusiness Consulting, 2018). Likewise, Indigo Ag (www.indigoag.com) offers project management and carbon sequestration verification to compensate farmers for adopting practices that sequester carbon with relatively low data requirements. The certification of these carbon offsets on farmer land can be sold to these companies to meet their carbon footprint goals.

Participants in these markets on the producer side include farmers, ranchers, and land-owners who wish to capitalize on financial incentives for BMPs and changes to management that improve ecosystem service functioning or sequester carbon. As these voluntary markets grow and become a potential means for producers to supplement their income, stakeholders who wish to become involved in them should have a thorough understanding of the economic incentives (positive and negative) of management practices. Without the implementation of these voluntary markets, the economic benefits of BMPS to improve soil health and water quality are often marginal or poorly captured by the market, in the case that they are quantified at all. Thorough enterprise budget analyses of changes to management practice should then also include the potential increased income from participation in voluntary carbon credit markets. If BMPs are successful, they should generate measurable and significant changes to soil health and sequester soil carbon, providing real economic value. Given the inherent upfront additional management load of adopting BMPs, compensating farmers for adopting BMPs provides a needed pathway for ecosystem services to be captured economically. However, the current structure of compensation in carbon offset markets may fail to adequately address the barriers to adoption of conservation agriculture. Most carbon offset contracts are structured such that payments are received annually for an estimated amount of carbon that would be sequestered over a decade or more. Additionally, these contracts typically “lock” farmers into a set of practices for a length of

10+ years with the goal of ensuring that carbon is sequestered. While this framework ensures that farmers do not receive payments and fail to deliver on management goals, they offer little flexibility with regards to tillage management. In systems where no-till is easily adopted and irrigation requirements are minimal, this may be feasible. In general, this is also most feasible in systems where high levels of C sequestration are attainable: temperate climates with high rainfall and natural C rich soils that readily accept C due to their degraded status. In short, the market for agricultural C credits is a much better fit for the temperate US Midwest or global subtropics than it is for semi-arid irrigated regions. There is realistically little C to be sequestered in semi-arid soils and cover cropping can be severely limited by water limitation.

However, the potential of soil health and water quality benefits of conservation agriculture warrants government funding (through programs like the Environmental Quality Incentives Program [EQIP] or Conservation Reserve Program [CSP]) to convert to conservation tillage, limited irrigation, improved nutrient use, cover cropping, and many other such practices. The contracts provided by these programs offer cost-sharing and upfront investments that allow farmers to install and maintain conservation practices. Alternative front-loaded payment schedules may improve profitability and reduce the barriers to adoption. The funds provided may have also provided financial flexibility for farmers to test different equipment combinations on their field. A study by Bilal et al. (2024) indicated that government funds to build community structure and knowledge-sharing have had a profound impact on the adoption of conservation agriculture. Payment plans that focus on barriers to adoption could be more effective at improving adoption rates and result in landscape-scale carbon sequestration. Differing discount rates of private and public entities should continue to be taken advantage of to inform incentive policies in conservation agriculture. We plan to model economic analyses of conservation tillage, including hypothetical carbon markets and cost-sharing provided by government funding. Carbon offset credits will be modeled primarily by measuring increases in sequestered C and using average prices of carbon offsets, while cost-sharing will be modeled based on existing EQIP contract structures.

3 ASSESSING SOIL HEALTH IN AGROECOSYSTEMS

Soil is increasingly being viewed as a natural resource of significant economic and ecological value. Soils act as a medium for plant growth, support macro- and microfauna, filter water, house biogeochemical transformations of pollutants, sequester carbon, and store much of the world's freshwater. Due to our reliance on healthy soils for sustainable agricultural productivity, managing for soil health is important and proper management of our agroecosystems requires a thorough framework to measure the state of their health. Hence, scientists use a number of frameworks to define or grade soil health as it relates to a change in land use or land management. In our attempts to do so, we must acknowledge that soil health is often abstract or difficult to define. Articulating what soil health "is" to farmers and land managers is challenging, but several models and scoring systems have been developed to evaluate soil health. Among these are the Haney Soil Test, the Solvita test, Cornell Soil Health Assessment, and the Soil Management Assessment Framework (SMAF).

Each test aims to quantify soil health in a slightly different way. The Haney Soil Health Test uses water and organic acid as extractants to measure organic C and N plant available nutrients (Haney et al., 2018), while the Solvita test uses carbon dioxide respiration to estimate microbial activity and mineralizable nutrients (Bavougian et al., 2019). While these tests provide a solid basis for understanding biological response to soil management change, they may not detect broader changes to overall soil physical and chemical function as a result of management practice. The Cornell Framework introduces the Comprehensive Assessment of Soil Health (CASH) protocol, which measures soil texture, “available water capacity, field penetrometer resistance, wet aggregate stability, organic matter content, soil proteins, respiration, active carbon, and macro- and micro-nutrient content assessment”. Additional indicators may be included, if the study site or objective requires them. Scoring curves are generated for each soil health indicator and measurements are rated 0-100, with high scores indicating “healthier” soil. Depending on the indicator, curves are generated on a “more is more”, a “less is more”, or an “optimal range” scoring function (Moebius-Clune et al., 2016). The CASH program’s extensive measurement of biological, physical, and chemical indicators makes it an attractive tool for understanding overall soil health and ecosystem function. However, the CASH program’s scoring curves are calibrated for soils in the Northeastern US, where precipitation and irrigation requirements, climate, and vegetation differ significantly as compared to the western US. Furthermore, the use of soil texture as the primary variable in scoring soil health makes the CASH framework susceptible to poor calibration in soils where carbonates accumulate and form concretions that impact estimates of soil texture.

The Soil Management Assessment Framework (SMAF) is a soil health model based on soils from Georgia, Iowa, California, and the Pacific Northwest (Andrews et al., 2004). Since it includes soils from the western US, SMAF is likely a tool better calibrated for Colorado’s semi-arid climate. In semi-arid climates, inorganic carbonates may play a key role in soil health metrics and estimates of soil texture. If the goal of our soil health frameworks is to develop a suite of tools to understand agroecosystem function, we need to utilize frameworks that take climate into context. The SMAF adequately relies on climatic variables to drive the scoring functions and incorporates inorganic C to inform soil texture. Similar to CASH, SMAF uses a series of measured soil chemical, nutrient, physical, and biological indicators (or characteristics) to evaluate soil health in a given field. Each soil indicator is scored via algorithms that describe the indicator as “more is better”, “less is better” or “somewhere in the middle is better”. Single numeric scores are provided for each indicator on a scale from 0 to 1, with 0 being “worst” and 1 being “best” in terms of that indicator. The indicator scores are pooled, weighted, and scored (from 0 to 1) into chemical, nutrient, physical, and biological soil health; an overall, weighted soil health score is also created by combining all four soil health scores together.

Soils relatively high in aggregate stability, for example, receive a higher score in aggregate stability, as this is scored via a “more is better” algorithm. Soils relatively low in bulk density are scored via algorithms that follow the “less is better” approach. Other indicators, such as pH or plant-available P, are graded on a “somewhere in the middle is better” approach. While the exact meaning and interpretation of a single score is difficult to understand, it does simplify the complexities of dynamic soil health. Scores can somewhat be easily communicated with producers, land-owners, and stakeholders in a way that does not require an in-depth scientific background in soil science.

Due to the better climatic representation of the SMAF tool compared to the CASH tool and the comprehensive set of indicators assessed in the SMAF tool, SMAF was chosen as the best framework for soil health assessment for our project in Colorado. The indicators used in this study are split into four categories: biological, physical, chemical, and nutrient indicators of soil health. Biological indicators include ß-Glucosidase activity, microbial biomass carbon, potentially mineralizable nitrogen, and soil organic carbon. Physical indicators include bulk density (and as a result, porosity), water-stable aggregates, and soil texture (as a calibrating measure for other indicators). Chemical indicators include pH and EC. Nutrient indicators include plant-available P and K. Furthermore, I performed a soil C fractionation to estimate the portion of soil organic C found as particulate organic matter and mineral-associated organic matter. Together, this suite of tests aims to measure soil health and fertility in response to a series of management practices.

A similar model has been developed to simplify and score water quality parameters and generate a single score: the Freshwater Water Quality Index (WQI; US EPA, 2013). Like the SMAF, the WQI involves the selection of scoring parameters, fitting those parameters to a scoring function that best represents a high-quality water sample, and pooling of these scores into one aggregated score, scored on a scale from 1 to 100 (Akter et al., 2016). Like the SMAF score, this score is designed to be easily interpreted by stakeholders and the public, and the ambiguity of such an aggregated score tends to leave out important context and details pertaining to how that score is generated. As with the SMAF, high quality reporting and scoring should highlight the key contributors to the score, focusing on the context of the system in question. For example, a SMAF score may indicate increased soil health in conservation tillage sites over conventional tillage sites, but the parameters that guided this score are relevant to interpretation. A decrease in the soil-water EC in a minimally-tilled site may result in a higher score than in its conventionally-tilled counterpart, but does not hold the same contextual relevance as an increase in soil organic matter in the same site. In short, while these pooled scores are helpful for science communication and imparting relevance to interested parties, they may fail to impart the mechanistic differences across treatments. Thus, proper selection of indicators and interpretation of indicator scores is vital to properly reporting and communicating the impact of BMPs on improving soil health and water quality.

Part of the motivation of our project is centered around water quality impairment across Northern Colorado, so throughout the studies water quality is often reported in the context of interim guidelines for water quality impairment. While the WQI provides some context for regional water quality, this project is motivated by documented impairments to water quality by specific pollutants in the selected watersheds. As a result, water quality is more often assessed in reduction of load and concentration of these pollutants, when possible. Furthermore, the use of a scoring index, such as the SMAF or WQI, is to attach a simplified quantitative measurement to a system where no such measurement exists. In this author's view, water quality is often described in quantitative terms, indicating risk of exposure to a toxic pollutant above some minimal threshold. However, soil health is often a more qualitative discussion and tends to require site-specific context. For all of these reasons, a soil health index is considered appropriate for the scope of this project, while a water quality index will not be used.

4 PROJECT SCOPE AND DESIGN

Northern Colorado is rapidly growing, bringing the interface of agriculture and residential areas to public attention. Water quality impairments from upstream agriculture use can impact downstream users, with downstream farmers facing increased pressures to reduce the effect of pollution on or from their farms. This is particularly true along the Little Thompson River near Berthoud, CO. This river is part of a large watershed that is symbolic of Colorado: developing rapidly but still largely agricultural. A significant amount of water is used in agriculture, municipalities, and suburban development within this watershed. As with most of the state, this region relies heavily on irrigation to meet crop needs. The combined effect of these activities has resulted in significant water quality impairments in the region. At multiple stretches along each river, 303(d) impairments have been generated by the state for nitrogen, phosphorus, *E. Coli*, selenium, and total suspended solids. All of these pollutants are associated with agriculture activity. For this reason, the state has taken particular interest in understanding the efficacy of agricultural BMPs in reducing pollutant loads. The Agricultural Water Quality Program (AWQP) of the Colorado Department of Agriculture, hosted at Colorado State University, has a successful history of working with producers in the region, evaluating BMPs on their land and serving in an extension role to promote adoption of BMPs across the state. This research project was a collaborative effort between the AWQP and the Ippolito research team at Colorado State University, a team that focuses primarily on soil health in agricultural contexts.

This team was brought together to address a growing question among farmers and policy-makers involved in conservation agriculture: how are BMPs affecting both water quality and concomitantly soil health in the region? Furthermore, how do they interact with other key ecosystem services, namely carbon sequestration and provisioning of profitable crop production? These questions are brought on by the growing recognition that healthy soils improve crop productivity and lower input costs can improve farm profitability. The fundamental principles of most BMPs align with the NRCS's goals of soil health management: "maximize living roots", "minimize [soil] disturbance", "maximize biodiversity", and to "maximize soil cover" (*Soil Health | NRCS*).

This project aims to measure soil health on sites adopting BMPs and, when possible, report on the efficacy of these BMPs on mitigating water quality concerns. Furthermore, I evaluated the costs and benefits of BMP adoption, utilizing budget analyses and hypothetical carbon credit and cost-sharing opportunities to estimate farm profitability changes from BMP adoption. I then used these findings to inform policy-making at the state level and to communicate to farmers, land-managers, and stakeholders the potential benefits of BMPs.

The BMPs investigated were voluntarily adopted by producers and used to improve water quality on their farms in accordance with their own management plan. As a result, field layout and design are often incongruent with more traditional field research experiments. Many similar studies have examined soil health and water on controlled research farms with multiple randomized blocks. While this research approach has relatively higher statistical power than the approach used here, observational studies on farms using BMPs offer the opportunity to study environmental quality changes. However, I also studied the effects of conservation tillage on a long-term tillage research field at Colorado State University's Agricultural Research, Development, and Education Center and a transition from furrow to sprinkler irrigation at Colorado State University's Arkansas Valley Research Center in Rocky Ford, CO (38.0386, -103.6932). The Kerbel Tillage Research Project is a long-term study on reduced tillage that began in 2011 and is designed with block replicates and multiple levels of control. To take advantage of both types of research, the following research discovery structure is used:

1) Identify highly correlated water quality and soil health parameters on the Kerbel Tillage Project at CSU's research farm;
2) Take key findings from Kerbel as a template for examining patterns at voluntary producer BMP sites;
3) Evaluate farm profitability on the Kerbel Tillage study and apply to voluntary producer BMP sites;

4) Investigate the change in soil health parameters on the voluntary producer BMP sites where BMP efficacy is well-documented; and
5) Determine the role of fractionated carbon pools in driving other soil health indices.

This research approach is used to address the following hypotheses:

Hypotheses

Hypothesis 1: Farms where water quality BMPs are being implemented will have improved soil health.

Sub-hypothesis: Soil characteristics monitored under the SMAF and those that characterize soil carbon pools will directly correlate to water quality improvements through improvement of soil function in conservation tillage.

Sub-hypothesis: Soil management in semi-arid systems can effectively manage for both soil health and productivity simultaneously while decreasing water quality concerns.

Hypothesis 2: Farm profitability will increase on fields adopting BMPs due to decreased input costs and improved soil productivity.

Sub-hypothesis: This difference in farm profitability will increase when carbon credit and federal grant opportunities are accounted for.

Sub-hypothesis: Frontloading of payments for carbon sequestration or ecosystem services will improve cost-effectiveness, efficiency, and equity of distributional effects of these transactions.

The farm sites used to investigate the above hypotheses include the following:

- The Kerbel Conservation Tillage site to monitor soil health, water quality, greenhouse gas mitigation, and economic potential of conservation tillage systems.
- The AVRC linear sprinkler field to study soil health impacts of transitioning from furrow to sprinkler irrigation in a research setting
- A private farmer's sprinkler pivot field to study soil health impacts of transitioning from furrow to sprinkler irrigation in a private producer setting

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CHAPTER 2 - CONVERSION FROM FLOOD TO SPRINKLER IRRIGATION HAS VARYING EFFECTS ON SOIL HEALTH

SUMMARY

Despite increased funding for conversion from furrow to sprinkler irrigation to conserve water in semi-arid agricultural watersheds, little is known about the effects of this conversion on soil health. To address this gap, soil health changes were monitored under two fields that underwent a furrow-to-sprinkler transition: one field at a university research station and the other a producer-managed field. Soil samples were collected at the top and bottom of each field in the first year and one to four years after the conversion. Soil health was assessed using the Soil Management Assessment Framework (SMAF), a scoring tool for ten soil health characteristics that indicate physical, biological, chemical, and nutrient soil health. Results showed that conversion to sprinkler irrigation generally improved soil health, though salinity concerns emerged at the research field site, highlighting the need for a holistic approach to soil health monitoring in these systems. Overall, the largest change in soil characteristics was a soil health homogenization of the field as erosion-deposition dynamics no longer drove spatial variability. Spatial homogenization should be views as a soil health improvement as it simplifies decisions relating

Submitted to Agrosystems, Geosciences & Environment

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to nutrient and irrigation management and helps farmers to predict yields across the field. Consequently, converting from furrow to sprinkler irrigation may help producers more easily manage homogenized fields due to on-site soil health improvements.

1 INTRODUCTION

Soil health has increasingly become a management goal for agricultural land managers who desire "the continued capacity of soil to function as a vital living ecosystem…[that] gives us clean air and water, bountiful crops and forests, productive grazing lands, diverse wildlife, and beautiful landscapes" (i.e., the soil health definition; USDA-NRCS, 2024). One of the most significant controls on global soil health is land management change, particularly in agricultural landscapes. Increasingly, management change has been focused on regenerative agriculture and research has consequently focused on the soil health impacts of these management changes. However, the majority of research in soil health has prioritized tillage and residue management, nutrient management, or organic amendments (such as biochar or compost). To date, there has been very little research focusing on irrigation management impacts on soil health.

This lack of connected irrigation-soil health research is surprising: climate change has increased both the need for irrigation in agriculture and the pressure on fragile water resources (EPA, 2024). For example, the semi-arid US west has encountered increasing drought pressure over the last decade, pushing producers and water resource managers to implement water conservation practices on agricultural lands. One of the most prominent and impactful recent changes to land management in these regions is the transition of irrigation systems from furrow irrigation to sprinkler irrigation. This transition has been shown to improve water application efficiency from ≤60% to ≥80% and reduce or eliminate surface runoff (Stein, 2011). From 1984 to 2018, acreage in the western US under pressure irrigation (sprinkler or drip) doubled, while those under gravity irrigation (furrow or flood) fell by half (USDA-ERS, 2023).

While the water quality and quantity implications have been studied (e.g., Bjorneberg et al., 2015), there is relatively little research on the soil health outcomes of these changes. Yet, it is important to understand how these transitions impact agricultural soils. There are significant funding initiatives designed to promote conservation practices across the United States such as the Conservation Effects Assessment Program (CEAP; Bjorneberg et al., 2008), the Conservancy District Program, the Environmental Quality Incentives Program, and the Colorado Department of Public Health and Environments' Non-Point Source Funding. These initiatives, alongside a growing cultural pressure to conserve water, are likely to push producers to further adopt conservation irrigation practices.

Studies that have investigated these systems have noted the importance of erosional and depositional dynamics in driving spatial variability of soil characteristics in furrow-irrigated systems (Trout, 1996), whereas sprinkler-irrigated systems tend to be more spatially homogenous (Koehn et al., 2014; Ippolito et al., 2017). While spatial heterogeneity is not a direct measure of soil health, a more spatially heterogenous field may result in decreased functionality of the entire soil system (Nyengere, 2023). Therefore, a furrow-irrigated field that has been converted to a sprinkler-irrigated field may now have approximately the same overall soil health across its entirety, and thus could be considered to have gained an agronomic benefit from sprinkler irrigation beyond increased water efficiency. There are other important agronomic and soil considerations for sprinkler irrigation. For example, in comparison to furrow irrigation, sprinkler irrigation can result in decreased leaching, potentially resulting in increased salinity, particularly when poor quality irrigation water is used (Karimzedah et al., 2024).

A series of publications (i.e., Bjorneberg et al., 2008, 2015, 2020) reported the water quality and watershed balance effects of a watershed-scale effort to pay producers through CEAP to convert from furrow to sprinkler irrigation in the Upper Snake River watershed in southern Idaho. Nearly two decades of research in this watershed indicated that the transition to sprinkler irrigation contributed to decreased sediment loss (Bjorneberg et al., 2008), P loss (Bjorneberg et al., 2015), and increased irrigation efficiency (Bjorneberg et al., 2020). Ippolito et al. (2017) followed up on these studies, investigating the soil health of sprinkler as compared to furrow irrigation on a series of paired producer fields in the same watershed, utilizing the SMAF to measure soil health on both continuously furrow-irrigated fields and fields that had undergone transition to sprinkler irrigation 5 to 8 years prior. The authors noted overall soil health improvements across entire fields under sprinkler irrigation, where soil health degradation existed in the top portion of furrow irrigated fields, similar to results presented by Trout (1996) and Koehn et al. (2014).

While studies above provide important insight into the differences between furrow-irrigated fields and sprinkler-irrigated fields, few study this transition explicitly. Consequently, it's still somewhat unclear how soil health changes throughout the transition to sprinkler-irrigated fields – and how quickly this transition can be measured in the soil. Moreover, Ippolito et al. (2017) called for a need to study this conversion within a single field or management system, as variability in land management and edaphic conditions had the tendency to mask soil health trends. This style of proposed study may limit the ability to extrapolate magnitude of difference to other sites but could provide key mechanistic insight to inform expectations around these changes.

To address this gap in research, we studied two fields undergoing this furrow-to-sprinkler transition, utilizing the SMAF to evaluate soil physical, biological, chemical, nutrient health, and overall soil health. One site was a research field measured in 2018 (immediately prior to conversion) and 2022 (four years after conversion), and the other site was a producer field measured from 2021 (at time of conversion) to 2023 (one and two years after conversion). Based off current literature, we formed the following hypotheses:

a) Soil health will generally improve after conversion to sprinkler irrigation; and
b) Soil characteristics will become more homogenously distributed across fields under sprinkler irrigation as compared to furrow irrigation.

2 MATERIALS AND METHODS

2.1 Study Sites

In 2018, a research field at Colorado State University's Arkansas Valley Research Center (AVRC) in southeastern Colorado, USA was converted from furrow irrigation via siphon tubes to linear sprinkler irrigation, providing researchers an opportunity to track soil health changes throughout this transition. Prior to this change, the field had primarily been used to grow alfalfa (*Medicago sativa*), grain corn (*Zea mays spp.*), and grain sorghum (*Sorghum bicolor*). In 2017, the field was plowed, mulched, disked, levelled, and bedded, then flood irrigated for grain sorghum. In 2018, the field was disked to plant corn and sprinkler irrigated. From 2019 to 2021, the field was strip tilled and sprinkler irrigated to grow pinto beans (*Phaseolus vulgaris spp.*), corn, and pinto beans in each year, respectively. In 2022, the field was plowed and sprinkler irrigated for onions. Soil samples were taken on April 11th, 2018 and again on June 21st 2022, representing years zero and four of the transition.

Meanwhile, in 2021 a producer in northern Colorado (near the town of Berthoud, Colorado, USA) was identified who was undergoing a similar transition: from gated pipe furrow irrigation to pivot sprinkler irrigation. In the past, this field had been used to grow a grain corn – winter wheat (*Triticum aestivum*) rotation. On July 20th 2021, the field was converted from furrow irrigation to sprinkler irrigation under grain corn. In 2022, triticale was grown, but water was not applied until after the first cutting of triticale, after which the field received only 4 irrigations. The field received a single disc tillage operation in the spring of 2022 and 2023, while in October of 2022 the field received two disc tillage operations in October; winter wheat was seeded in November 2022. Due to heavy spring rains, no irrigation was applied until September of 2023, after wheat had been harvested. Soils were sampled July 21st of 2021 (one day post-conversion), July 20th of 2022 (one year post-conversion), and September 13th of 2023 (two years post-conversion). Ideally, the 2023 sampling would have been closer to those in 2021 and 2022, but due to heavy rains through the spring and summer, the field could not be accessed until later in the growing season. Further details of each site are presented in Table 2.1.

Table 2.1. Site Characteristics

Characteristic	AVRC Site	Berthoud Site
Management	Research Plot	Producer-Managed
Latitude and Longitude	38° 02'15"N, 103° 41'23"W	~40° 18'18"N, ~105° 04'43"W
Elevation	1269 m	1516 m
Max / Min Temperature	22 °C / 2 °C	19 °C / 0 °C
Average Annual Precipitation	314 mm	400 mm
Soil Series	Rocky Ford Series	Table Mountain Loam/Caruso Clay Loam
Percent (%) Clay	39%	35%
Dominant Parent Material	Alluvium	Alluvium
Crop Rotation	(Corn - beans) x2 – sorghum	Wheat – Forage - Wheat

2.2 Soil Sampling and Analysis

2.2.1 Soil Sampling

All soils were sampled to a depth of 15 cm using a 2.5 cm diameter soil probe. At three GPS-located points along the top and bottom of each field (~30 m inside the field from the top and bottom borders, respectively), a composite soil sample was obtained. Each composite soil sample was composed of approximately 30 cores within a 3 m radius of the GPS-located point, transferred to a plastic Ziploc bag, sealed, and placed in a dry cooler with ice packs. At the center of each sampling radius, a Madera probe (Evett et al., 2022; Precision Machine Co., Lincoln, Nebraska, USA) was used to take an intact bulk density (sample of ~61 cm^3, at approximately the same depth. The samples were transferred to pre-weighed tin cans and sealed to prevent evaporation. Soils were promptly returned to the laboratory and stored at 4 °C until processing. The composite soil samples were passed through an 8-mm sieve to remove rocks and large plant debris, with a further ~150 g of soil passed through a 2-mm sieve and air-dried. Of the 8-mm sieved soil, ~150 g was stored at field moist conditions at 4 °C for MBC determination, and the remainder was air-dried.

2.2.2 Soil Management Assessment Framework (SMAF)

The SMAF (Andrews et al., 2004) is a Microsoft Excel-based tool used to score and interpret soil health measurements. Within the Excel workbook, raw measurements of soil properties are translated to unitless scores ranging from 0 to 1 (0 being “worst” and 1 being “best”). Scores are generated within the context of climate, intended crop, and base soil taxonomy and texture. The SMAF uses several different sets of soil health indicators, including: physical indicators (bulk density [, and water-stable aggregates [WSA]); biological indicators (soil organic C [SOC], microbial biomass C [MBC], potentially mineralizable N [PMN], and β-glucosidase activity [BG]); chemical indicators (pH [$pH_{1:1}$] and electrical conductivity [$EC_{1:1}$]); and nutrient indicators (plant-available P and K). Additionally, inorganic C (IC) was included as a soil characteristic, yet this indicator does not exist in SMAF and thus it was not scored. Indicators were selected to highlight soil ecosystem services, soil-water relations, agronomic capacity, and sensitivity to management change. Interested readers should consult Andrews et al. (2004) for further information regarding use of the SMAF. Readers interested in the soil analyses methods used should refer to Trimarco et al. (2023).

2.2.3 Statistical Analysis

Each composite soil sample was considered an individual replicate for statistical analysis; the two sites were analyzed separately. The Aligned Rank Transformation in ANOVA test was used to perform non-parametric rank analysis with interacting terms (Wobbrock et al., 2011), an analysis tool that has been used in similar soil systems (Trimarco et al., 2023). *Post hoc* pairwise comparisons were performed as necessary (Elkin et al., 2021). Models were created with sampling year and within field location (i.e., top vs. bottom) as interacting factorial predictors and the raw measurements of soil health indicators as outcome variables. The same process was repeated with SMAF scores as outcome variables. The decision to include field location as a predictor was based on prior research by Ippolito et al. (2017) that found the conversion from furrow to sprinkler irrigation has a significant impact on the spatial distribution of soil health parameters. Analysis was performed using R (Version 4.4.0) in RStudio (Build 748) using the stats package (Version 4.4.0), the sjPlot package (Version 2.8.16) and the ARTool package (Version 0.11.1). Significance was evaluated at $\alpha \leq 0.05$.

3 RESULTS AND DISCUSSION

3.1 Results

Table 2.2. Raw Measurements (Mean ± SE) of Soil Health Indicators and Aligned Rank Transform Test Results – AVRC Site

BD = bulk density; WSA = water stable aggregates; BG = β-glucosidase activity; MBC = microbial biomass carbon; SOC = soil organic carbon; PMN = potentially mineralizable nitrogen; EC = electrical conductivity; P = plant-available P; K = plant-available potassium; IC = inorganic carbon.

*Inorganic carbon is not scored under the SMAF framework. Both EC and K SMAF scores were not evaluated in the Aligned Rank Transform in ANOVA due to the extremely high number of ties.

Year	**Field Location**	**BD** (g cm^{-3})	**WSA** (%)	**BG** (mg pnp kg^{-1} soil hr^{-1})	**MBC** (mg kg^{-1})	**SOC** (%)	**PMN** (mg kg^{-1})	**$pH_{1:1}$**	**$EC_{1:1}$** (dS m^{-1})	**P** (mg kg^{-1})	**K** (mg kg^{-1})	**IC** (%)
2018	Top	1.53 ± 0.02	61.3 ± 10.2	166 ± 27	106 ± 6	1.54 ± 0.10	21.3 ± 1.8	7.85 ± 0.04	0.49 ± 0.01	16.7 ± 9.3	195 ± 16	0.43 ± 0.02
	Bottom	1.54 ± 0.06	39.7 ± 7.2	137 ± 4	137 ± 20	1.08 ± 0.04	17.5 ± 1.9	7.86 ± 0.02	0.46 ± 0.03	37.5 ± 4.2	210 ± 10	0.38 ± 0.05
2022	Top	1.60 ± 0.13	57.4 ± 10.6	105 ± 11	177 ± 19	1.39 ± 0.10	24.7 ± 1.3	8.26 ± 0.04	1.57 ± 0.17	19.1 ± 3.3	224 ± 48	0.39 ± 0.05
	Bottom	1.53 ± 0.06	65.6 ± 12.4	145 ± 32	180 ± 24	1.68 ± 0.10	20.6 ± 2.6	8.08 ± 0.05	1.85 ± 0.03	14.7 ± 1.0	203 ± 9	0.45 ± 0.02
Raw Measurements Model P-Values												
Year		0.337	0.338	0.197	**0.0148**	0.077	0.208	**0.00103**	**< 0.001**	0.155	0.888	0.499
Field Location		0.587	0.689	0.890	0.270	0.338	0.0520	0.0513	0.0764	0.153	0.587	0.893
Year-Field Location		0.338	0.208	0.340	0.414	**0.00106**	0.790	**0.0316**	0.0716	**0.0284**	0.774	0.415
SMAF Scores (Mean ± SE)												
Year	**Field Location**	**BD**	**WSA**	**BG**	**MBC**	**SOC**	**PMN**	**pH**	**EC***	**P**	**K***	IC*
2018	Top	0.29 ± 0.01	0.97 ± 0.03	0.38 ± 0.18	0.09 ± 0.00	0.26 ± 0.03	0.99 ± 0.01	0.74 ± 0.01	1.00 ± 0.00	0.67 ± 0.33	0.98 ± 0.01	-

	Bottom	0.29 ± 0.04	0.77 ± 0.13	0.17 ± 0.08	0.13 ± 0.03	0.12 ± 0.01	0.95 ± 0.03	0.74 ± 0.01	1.00 ± 0.00	1.00 ± 0.00	1.00 ± 0.00	-
2022	Top	0.30 ± 0.07	0.94 ± 0.06	0.16 ± 0.06	0.20 ± 0.04	0.21 ± 0.02	1.00 ± 0.00	0.73 ± 0.01	1.00 ± 0.00	1.00 ± 0.00	0.98 ± 0.01	-
	Bottom	0.30 ± 0.04	0.95 ± 0.05	0.14 ± 0.01	0.19 ± 0.03	0.29 ± 0.03	0.99 ± 0.01	0.77 ± 0.01	1.00 ± 0.00	0.96 ± 0.04	1.00 ± 0.00	-
SMAF Score Model P-Values												
Year		0.778	0.497	0.244	**0.00721**	**0.0450**	0.115	0.267	-	0.324	-	-
Field Location		1.000	0.334	0.376	0.592	0.270	0.0877	0.0513	-	0.324	-	-
Year-Field Location		0.676	0.264	0.683	0.589	**0.00106**	0.163	**0.0316**	-	0.291	-	-

SMAF Soil Health Indices (Mean ± SE)						
		Physical	Biological	Chemical	Nutrient	Overall
2018	Top	0.63 ± 0.02	0.43 ± 0.05	0.87 ± 0.01	0.83 ± 0.16	0.64 ± 0.04
	Bottom	0.53 ± 0.05	0.34 ± 0.03	0.87 ± 0.00	1.00 ± 0.00	0.62 ± 0.00
2022	Top	0.62 ± 0.06	0.39 ± 0.03	0.86 ± 0.00	0.99 ± 0.00	0.65 ± 0.02
	Bottom	0.62 ± 0.04	0.40 ± 0.02	0.89 ± 0.01	0.98 ± 0.00	0.66 ± 0.02
Model P-Values						
	Year	0.414	0.587	0.267	0.336	0.496
	Field Location	0.416	0.491	0.0513	0.336	0.496
	Year-Field Location	0.338	0.0736	**0.0316**	0.339	0.493

Table 2.3. Raw Measurements (Mean ± SE) of Soil Health Indicators and Aligned Rank Transform Test Results – Berthoud Site

BD = bulk density; WSA = water stable aggregates; BG = β-glucosidase activity; MBC = microbial biomass carbon; SOC = soil organic carbon; PMN = potentially mineralizable nitrogen; EC = electrical conductivity; P = plant-available P; K = plant-available potassium; IC = inorganic carbon.

*Inorganic carbon is not scored under the SMAF framework. Both EC and K SMAF scores were not evaluated in the Aligned Rank Transform in ANOVA due to the extremely high number of ties

Year	**Field Location**	**BD** (g cm^{-3})	**WSA** (%)	**BG** (mg pnp kg^{-1} soil hr^{-1})	**MBC** (mg kg^{-1})	**SOC** (%)	**PMN** (mg kg^{-1})	**pH$_{1:1}$**	**EC$_{1:1}$** (dS m^{-1})	**P** (mg kg^{-1})	**K** (mg kg^{-1})	**IC** (%)
2021	Top	1.55 ± 0.03	76.2 ± 4.2	213 ± 27	138 ± 3	1.51 ± 0.08	21.6 ± 1.4	7.77 ± 0.03	0.29 ± 0.02	24.2 ± 0.9	307 ± 14	0.00 ± 0.00
	Bottom	1.60 ± 0.02	26.7 ± 4.5	114 ± 85	113 ± 3	1.39 ± 0.06	24.3 ± 1.5	8.00 ± 0.13	0.62 ± 0.09	40.9 ± 8.3	463 ± 66	0.08 ± 0.03
2022	Top	1.36 ± 0.16	76.1 ± 3.4	422 ± 17	72 ± 9	1.54 ± 0.03	17.3 ± 1.1	7.95 ± 0.06	0.67 ± 0.03	16.5 ± 1.4	277 ± 18	0.02 ± 0.01
	Bottom	1.21 ± 0.08	44.9 ± 14.3	288 ± 18	83 ± 10	1.44 ± 0.02	16.3 ± 4.1	8.37 ± 0.08	0.62 ± 0.03	23.4 ± 6.6	378 ± 49	0.09 ± 0.02
2023	Top	1.34 ± 0.08	74.8 ± 0.5	211 ± 30	185 ± 19	1.44 ± 0.03	27.2 ± 5.3	7.61 ± 0.05	0.39 ± 0.04	28.7 ± 3.0	312 ± 19	0.05 ± 0.01
	Bottom	1.41 ± 0.04	30.2 ± 3.4	107 ± 18	138 ± 8	1.45 ± 0.08	28.3 ± 1.1	7.96 ± 0.05	0.44 ± 0.05	28.4 ± 5.5	460 ± 71	0.13 ± 0.04
Model P-Values												
Year		**0.0011**	0.54	**0.0021**	**< 0.001**	0.723	**0.0071**	**< 0.001**	**0.0036**	0.061	0.31	0.084
Field Location		0.80	**< 0.001**	**0.0076**	**0.028**	0.157	0.74	**< 0.001**	**0.014**	0.22	**0.0014**	**0.0075**
Year-Field Location		0.35	0.68	0.81	**0.032**	0.624	0.68	0.50	**0.0015**	0.22	0.65	0.86
SMAF Scores (Mean ± SE)												
Year	**Field Location**	**BD**	**WSA**	**BG**	**MBC**	**SOC**	**PMN**	**pH**	**EC***	**P**	**K***	**IC***
2021	Top	0.28 ± 0.01	1.00 ± 0.00	0.21 ± 0.06	0.11 ± 0.00	0.22 ± 0.02	0.99 ± 0.00	0.76 ± 0.01	1.00 ± 0.00	1.00 ± 0.00	1.00 ± 0.00	-
	Bottom	0.37 ± 0.07	0.63 ± 0.08	0.35 ± 0.31	0.14 ± 0.02	0.28 ± 0.05	1.00 ± 0.00	0.69 ± 0.04	1.00 ± 0.00	1.00 ± 0.00	1.00 ± 0.00	-
2022	Top	0.53 ± 0.23	1.00 ± 0.00	0.74 ± 0.04	0.06 ± 0.01	0.24 ± 0.01	0.95 ± 0.02	0.08 ± 0.01	1.00 ± 0.00	0.94 ± 0.03	1.00 ± 0.00	-
	Bottom	0.82 ± 0.17	0.80 ± 0.10	0.97 ± 0.02	0.09 ± 0.02	0.3 ± 0.04	0.81 ± 0.18	0.02 ± 0.01	1.00 ± 0.00	0.92 ± 0.08	1.00 ± 0.00	-

2023	Top	0.55 ± 0.13	1.00 ± 0.00	0.21 ± 0.05	0.19 ± 0.04	0.21 ± 0.01	0.98 ± 0.02	0.19 ± 0.03	1.00 ± 0.00	1.00 ± 0.00	1.00 ± 0.00	-
	Bottom	0.67 ± 0.18	0.71 ± 0.06	0.27 ± 0.08	0.21 ± 0.05	0.31 ± 0.06	1.00 ± 0.00	0.07 ± 0.01	1.00 ± 0.00	1.00 ± 0.00	1.00 ± 0.00	-
Model P-Values												
Year		0.087	0.41	**0.0098**	**< 0.001**	0.83	0.48	**< 0.001**	-	0.34	-	-
Field Location		0.14	**< 0.001**	0.14	0.35	0.14	0.17	**< 0.001**	-	0.34	-	-
Year-Field Location		0.80	0.41	0.46	0.93	0.78	0.23	0.22	-	0.98	-	-

SMAF Soil Health Indices (Mean ± SE)						
		Physical	Biological	Chemical	Nutrient	Overall
2021	Top	0.64 ± 0.01	0.39 ± 0.01	0.88 ± 0.00	1.00 ± 0.00	0.66 ± 0.01
	Bottom	0.50 ± 0.05	0.44 ± 0.06	0.85 ± 0.02	1.00 ± 0.00	0.65 ± 0.02
2022	Top	0.76 ± 0.12	0.50 ± 0.02	0.54 ± 0.01	0.97 ± 0.02	0.65 ± 0.02
	Bottom	0.81 ± 0.12	0.54 ± 0.05	0.51 ± 0.00	0.96 ± 0.04	0.67 ± 0.04
2023	Top	0.77 ± 0.07	0.40 ± 0.02	0.60 ± 0.01	1.00 ± 0.00	0.63 ± 0.02
	Bottom	0.69 ± 0.09	0.45 ± 0.05	0.54 ± 0.01	1.00 ± 0.00	0.62 ± 0.04
Model P-Values						
Year		0.051	**0.022**	**< 0.001**	0.34	0.77
Field Location		0.74	0.35	**< 0.001**	0.34	0.97
Year-Field Location		0.65	0.98	0.22	0.98	0.92

3.1.1 AVRC Research Site

Means of raw soil health indicators, SMAF scores, and soil health indices (SHI), alongside the outcome of ANOVA tests can be found for the AVRC site are presented in Table 2.2. Physical soil health did not change as the system transitioned from 2018 to 2022. Bulk density and WSA were neither affected by year nor field location with regards to both raw measurements and SMAF score. However, WSA did appear to increase in the bottom of the field from 2018 to 2022, bringing the top and bottom of the field closer to each other. While not significant, this trend was in line with our hypothesis that fields would generally homogenize. There was no significant effect of year, field location, or the interaction of year x field location on physical SHI, a composite of SMAF scores for BD and WSA.

Biological soil health was generally improved as the AVRC site matured, with a significant effect of year observed for MBC and a near-significant effect for SOC with regards to raw measurements. Beta-glucosidase appeared to (insignificantly) decrease from 2018 to 2022, particularly in the top of the field, a pattern that was also observed for SOC. Microbial biomass C increased from 2018 to 2022 across the entire field, and while there wasn't a significant effect of year on PMN, PMN did increase slightly from 2018 to 2022. There was also a near significant effect of field location for PMN, with field tops exceeding field bottoms. Soil organic C did have a unique trend; field top SOC fell slightly, but field bottom SOC increased dramatically, reversing the 2018 trend where field top SOC exceeded field bottom SOC and resulting in a significant effect of year x field location for SOC with regards to SMAF score and raw measurements With regards to SMAF biological soil health scores, year played a significant role in MBC and SOC, with a near-significant effect of field location on PMN. There was no effect of year, field location, or year x field location on biological SHI, a composite of all biological soil health indicator SMAF scores.

In terms of chemical soil health at the AVRC site, both pH and EC showed significant effects of year for raw measurements and pH was also significantly affected by the year x field location interaction and nearly by field location. A near-significant effect was observed for field location and year x field location with regards to raw measurements of EC, largely driven by the bottom of the field becoming slightly more saline than the top of the field, where in 2018 both tops and bottoms of fields were roughly equivalent. Although not a part of SMAF, inorganic carbon may be considered a chemical indicator. Inorganic carbon was not significantly affected by year, field location, or the year x field location interaction. The SMAF pH score was significantly affected by field location and nearly by year x field location. Generally, pH increased from 2018 to 2022, particularly in the top of the field. While EC more than tripled from 2018 to 2022, every sample had a SMAF score of 1.00 (Table 2.2), largely because EC is only problematic for plant growth above 2 dS m^{-1}, so SMAF score could not be evaluated statistically. The chemical SHI, a composite of pH and EC scores, had a significant interaction of year x field location and a near-significant effect of field location. It seemed as if from 2018 to 2022, the field bottoms slightly improved with regards to chemical health, while the fields tops slightly worsened, but due to the small changes to pH and EC SMAF scores, this change was also very small.

Nutrient soil health did not change year-to-year because of irrigation management change. For neither P nor K was the field location nor the year a significant factor, but there was a significant impact of the year x field location interaction for P. While in 2018, the bottom of the field had roughly doubled the P as the top of the field, the field locations were essentially the same in 2022 (and in fact increased slightly at the top of the field). While this did not result in a statistically significant result, the trend towards homogenization of field nutrient characteristics is important. A more spatially homogenous field, particularly with regards to nutrient status, simplifies management decisions and improves field workability. There was no significant effect of year, field location, or year x field location on P SMAF score and K SMAF score could not be evaluated statistically due to the very high number of ties at 1.00. Consequently, there was no significant effect of year, field location or year x field location on nutrient SHI, a composite of P and K SMAF scores.

The overall SHI, a composite of all SMAF scores, was not significantly affected by year, field location, or year x field location. It does appear that the overall SMAF score increased slightly across the field from 2018 to 2022, but the magnitude of this change was relatively small.

3.1.2 Berthoud Producer Site

Means of raw soil health indicators, SMAF scores, and soil health indices, alongside the outcome of ANOVA tests can be found for the Berthoud site are presented in Table 2.3. In contrast to the AVRC site, irrigation transition did appear to improve physical soil health at the Berthoud site. There was a significant effect of sampling year on and of field location on WSA. 2021 had a significantly greater than 2022 ($p = 0.0017$) and 2023 ($p = 0.0039$), though 2022 and 2023 were not different from each other ($p = 0.88$). The trend seemed to be an initial sharp decrease in that levelled out by 2022. The field tops had significantly greater WSA than the field bottoms, though it did appear as though this started to become a less intense dynamic in 2022 and 2023 (though year x field location was not a significant interaction). With respect to SMAF scores, only the field location effect on WSA was observed, though the sampling year effect trended towards significance. The physical SHI score (an average of and WSA scores) was not significantly affected by year, field location, or year x field location, though there was a trend towards significance for year.

Soil biological health demonstrated mixed trends. Sampling year played a significant role in BG and PMN, field location played a significant role on BG, and year, field location, and year x field location had a significant effect on MBC. For BG, 2022 was significantly greater than either 2021 ($p = 0.0036$) or 2023 ($p = 0.0054$) and the field tops were significantly greater than the field bottom – often by a factor of ~2. For MBC, 2023 was greater than 2022 ($p < 0.001$) and 2021 ($p < 0.001$). Year played a significant role in PMN with 2023 PMN levels exceeding 2022 PMN levels ($p = 0.0056$). With regards to biological soil health SMAF scores, a significant effect of year was observed for both BG and MBC, with similar trends as was observed for raw measurements. The biological SHI, a composite of all biological soil health indicators, was significantly affected by year, but not by field location or year x field location.

Soil chemical health was strongly impacted by the transition to sprinkler irrigation. Both pH and EC were significantly affected by sampling year and field location, and EC was also significantly affected by the year x field location interaction. With regards to pH, 2022 was significantly greater than 2021 ($p = 0.011$) and 2023 ($p < 0.001$) and field bottoms were consistently greater than field tops. Electrical conductivity was greater in 2022 than in 2021 ($p = 0.013$) and 2023 ($p = 0.0046$). Although not a part of SMAF, inorganic C was significantly affected by field location (with bottoms exceeding tops), and there was a trend towards significance with respect to year, as it appeared that IC slightly increased each year. In 2021, there was a significant difference between the top and bottom of the field with respect to EC ($p = 0.0025$), but this difference was not observed for 2022 or 2023. Only pH was evaluated for SMAF scores, as every sample scored 1.0 for EC. As with raw scores, year and field location had a significant impact on pH SMAF score, with 2021 having by far the greatest scores with 2022 and 2023 having lower scores. The SMAF scores for EC could not be evaluated statistically as every soil had an EC of 1.0, and consequently the same pH score pattern was observed for chemical SHI, a composite of pH and EC SMAF scores.

Nutrient soil health did not appear to be changed by irrigation management, though there was a near-significant effect of year on extractable P with regards to raw measurements. It did seem as though soil P levels fell from 2021 to 2023 and homogenized. Extractable K was significantly affected by field location with respect to raw measurements, with field bottoms generally exceeding field tops. With regards to SMAF scores, there was no significant effect of any predictor on soil P, and soil K could not be evaluated due to exact ties in all SMAF scores. Nutrient SHI, a composite of P and K SMAF scores, was not significantly affected by year, field location, or year x field location.

The overall SHI, a composite of all SMAF scores, was not significantly affected by year, field location, or year x field location. It did seem as though SHI fell slightly over the two years, but this difference was small in magnitude.

3.2 Discussion

The most notable positive soil health change across both fields was the homogenization and improvement of WSA, a soil physical health component. At both sites, WSA were much greater at the field tops than at the field bottoms, with this difference dissipating over time, though it seems as if the longer study period at the AVRC site resulted in more WSA homogenization. A study by Trimarco et al. (In review) observed decreased WSA in furrow-irrigated field bottoms but did not investigate the role of transitioning irrigation. Ippolito et al. (2017) did not find a difference in WSA between field tops and bottoms but did note that long-term erosional and depositional dynamics drive spatial distribution in furrow-irrigated fields, but not sprinkler-irrigated fields. In furrow-irrigated fields, heavy erosion carries sediment from the field top and deposits it at the field bottom (Trout, 1996), likely transporting the portion of soil that is not associated with stable aggregates. Consequently, the bottom of furrow-irrigated fields could logically have decreased WSA (Trimarco et al., in review), with this difference disappearing with the adoption of sprinkler irrigation. However, Koehn et al. (2014) observed that tensile strength of aggregates was greater at the bottom of fields in furrow-irrigated systems, though this disappeared soon after irrigation. This lack of consistency between multiple studies with respect to aggregate stability and distribution in furrow-irrigated fields warrants further research. At the AVRC site, the field was essentially homogenized by the fourth year after conversion to sprinkler irrigation, whereas the Berthoud site was trending towards homogenization but still displayed clear spatial variability two years after sprinkler adoption. It was also worth noting that decreased from 2021 to 2022 at the Berthoud site and remained at this lower level into 2023. Liu et al. (2016) observed decreased compaction in shallow-tilled deficit-irrigated wheat-corn systems as opposed to fully-irrigated systems, noting that high moisture conditions in fully-irrigated systems result in collapsing of soil structure and long-term compaction. Given that sprinkler irrigation manages for lower irrigation inputs, this may be why the Berthoud site observed decreased BD after conversion and seemed to become slightly more compacted in 2023 when heavy rains resulted in consistently high soil moisture. However, we did not observe a change in BD at the AVRC site, indicating that BD effects from transitioning to sprinkler irrigation may be relatively small or site-specific.

Measures of soil biological health indicated mixed trends. Soil organic C enrichment at the field bottoms and homogenization of SOC across the field at both sites is of particular note, largely because it follows a pattern similar to WSA. The relationship between SOC and WSA has been well documented (i.e., Six et al., 1999, 2000; Bhattacharyya et al., 2009; Zheng et al., 2018) and in this study the soils with the highest WSA tended to contain the greatest SOC. Following decades of furrow irrigation, weakly-aggregated, SOC-poor soils may have been eroded from the bottom of the field. Consequently, when furrow-irrigated fields are converted to sprinkler irrigation, heavy erosion and deposition decreases and both WSA and SOC cease to be greatly impacted by erosional dynamics. In short time (i.e., two to four years following conversion), homogenization of both WSA and SOC across the field begins.

At the Berthoud site, PMN seemed to be lowest in 2022, whereas BG was greatest in that same year. It would be expected that soils with high PMN would also have high BG as both are proxies of organic matter availability and biological activity (Andrews et al., 2004). However, as BG proxies microbial activity and PMN proxies organic matter stock, in a single field over short time periods, they may be inversely related. Severe drought in 2021 was compounded by sprinkler mechanical failure; soil conditions throughout the growing season were consistently extremely dry. The return of water in 2022 may have temporarily spiked BG (Criquet et al., 2002) and resulted in temporary PMN depletion, but this returned to stability in 2023. Regardless, at both the AVRC and Berthoud sites, MBC and PMN were greatest in the last sample year (2022 for AVRC and 2023 for Berthoud), indicating that the transition to sprinkler irrigation has resulted in broadly improved soil biological health. Decreased intensity of water application may help to stabilize aggregates, particularly at the bottom of fields, protecting organic matter and improving biological function (Gupta and Germida, 1988). The relationship between aggregate stability and biological soil health has been documented in similar furrow-irrigated systems in northern Colorado (Trimarco et al., in review). Martín-Lammerding et al., (2013) documented increased PMN in semi-arid Spanish soils under no-till as compared to conventional till, but did not explore the role of irrigation in soil aggregate stability. However, resilience of soil aggregates should logically result in protection of organic matter (including PMN), regardless of the method of disturbance.

The most dramatic negative soil health change in this study was the observed reduction in chemical soil health with transition from furrow to sprinkler irrigation. Furrow irrigation is inefficient, but the excess water applied pushes salts down through the root zone and reduces salt accumulation at shallow depths. Conversely, sprinkler irrigation aims to reduce excess leaching and improve water use efficiency, but this can result in the accumulation of salts in the root zone (Karimzadeh et al., 2024). The reduction in drainage in sprinkler systems may be compounded in fine-textured soils with low hydraulic conductivity, such as the heavy clay soils at both sites (Table 2.1). Additionally, the irrigation water itself may be steadily salinizing. Surface water from the nearby Rocky Ford canal is used for irrigation at the AVRC site, while surface water from the Little Thompson River is used for irrigation at the Berthoud site, and salinization of surface and ground water is a growing risk in semi-arid regions (Bauder et al., 2011) similar, if not identical, to both these sites. Salinity at the AVRC site is likely of greater concern due to the intense salinization of surface water in the Arkansas River, a consequence of irrigation return flows (Lin and Garcia, 2012), whereas the Berthoud site is much closer to the headwaters of the Little Thompson River, reducing the salinizing effect of upstream irrigation.

As pH varied little within a single field over the course of the study, it is unlikely that it strongly affected soil fertility characteristics. However, pH likely had an impact on inorganic C (Batool et al., 2024), prompting further investigation into a model identical to the other models used for soil health indicators but with inorganic C as an outcome (Tables 2.2 and 2.3). At Berthoud, 2021 had the lowest level of carbonates, whereas 2023 had the highest, perhaps indicating that decreased irrigation inputs resulted in more carbonate precipitation and thus accumulation over time. In furrow-irrigated systems, field tops also receive deeper leaching, which may be why carbonates were consistently lower at the top of the field (Batool et al., 2024). This effect did not seem to be changed in the short life of this study: the bottom of the Berthoud field consistently had greater inorganic C and both the top and bottom increased each year. We did not observe this same trend at the AVRC site, but the observations at the Berthoud site warrant further research on carbonate accumulation deeper in the soil profile in future irrigation research.

The consistent pattern of greater EC in the bottom of the fields may be due to the same dynamic driving increased inorganic C in the bottom of the field: decreased infiltration from irrigation. As we switch to irrigation systems that aim to maximize water use efficiency, we also decrease leaching in the root zone. This dynamic also has important implications on soil characteristics that remain underappreciated in most soil health studies. Carbonates in soil can increase aggregate stability, bind P, and buffer changes to pH (Batool et al., 2024). Even in alkaline western US soils, soil acidification has been documented as a growing problem as a consequence of ammonium fertilizer inputs (Blaylock, 2024). Soil alkalinization due to conversion from furrow to sprinkler irrigation may help reverse this acidification process, but given that sprinkler irrigation also results in salinization and most semi-arid soils are not in danger of acidification, the chemical shift towards a more saline, alkaline soil should be a concern in terms potential decline of soil health in these and similar systems.

In general, the trends observed with regards to nutrient soil health indicators were consistent with the dynamic of decreased erosional dynamics in sprinkler irrigated fields. At the AVRC site, extractable P concentrations at the bottom of the field were ~2x those at the top of the field in 2018, but by 2022 this difference disappeared. Phosphorus is bound to sediment and in furrow irrigated systems, the erosion and deposition of P-rich sediment likely plays an important role in P distribution across the field as Olsen P is well-correlated to runoff P (Turner et al., 2004). The greater P concentrations at the bottom of the field in 2018 implied that P was transported and deposited at the bottom of the field, where it accumulated. It is therefore unsurprising that this dynamic disappeared in 2022 when erosion and deposition became a far less important force in the sprinkler-irrigated system. Differences in extractable K content were not as pronounced at the AVRC site, but the Berthoud site seemed to contain greater extractable K concentrations in the bottom as compared to the top of the field. Given that some portion of extractable K is associated with soil sediment (Goulding et al., 2020), it may not be surprising that extractable K spatial variation exists across fields formally under furrow irrigation.

Results indicated that the overarching effect of furrow-to-sprinkler conversion is a removal of erosion/deposition dynamics from driving spatial variability. This change is best reflected in soil physical, biological, and nutrient properties. In contrast, chemical soil health appears to have responded to the decreased irrigation inputs, perhaps causing degradation of soil health. Overall, these results seem to support our hypothesis conversion from furrow to sprinkler irrigation will lead to homogenization of soil health, but does provide some evidence to contrast our hypothesis of broadly improved soil health under sprinkler irrigation. The lack of overall improvement may be in part due to the short time scales of this study. Further use of sprinkler irrigation may improve soil health in the system over time periods of 10-15 years and beyond as decreased erosion supports aggregate stability formation and a nutrient-rich surface horizon. The sustained effects of reducing erosion may elevate physical and biological health to a new steady state past the timeframe of this study, though it was promising to observe homogeneity in these fields within just 3 years of the transition to sprinkler irrigation. Despite these potential benefits, the somewhat rapid increase in salinity at the AVRC site highlights a concern with sprinkler irrigation. Continued monitoring of salinity in these systems is necessary to facilitate long-term plant growth, particularly if salinization of the irrigation source is a concern. The contrasting changes in soil health over the length of this study indicate that, at least in the short term, the physical and biological soil health benefits are balanced by chemical soil health degradation. Further research should continue to study these systems over longer time scales to understand how these systems mature over time.

4 CONCLUSIONS

The relatively short-term impact of converting from furrow to sprinkler irrigation on soil health was mixed. At both a research site (AVRC) and a producer-managed site (Berthoud), homogenization of soil characteristics was observed. Under furrow irrigation, spatial variability of soil characteristics was driven by erosion and deposition dynamics, resulting in drastic differences between the top and bottom of the field. Under sprinkler irrigation, both fields showed decreased spatial variability from the top to bottom of the field, and moreover showed generally improved soil physical and biological health. Increases in and homogenization of soil health characteristics should generally be seen as a benefit, as consistency across a field simplifies farmer decision-making and decreases management requirements. These findings support our hypothesis that soil health would be generally improved and homogenized under sprinkler irrigation. However, negative soil health effects were also observed as a consequence of decreased irrigation inputs: namely the accumulation of salts in the root zone that may in the future lead to salinity concerns. Using occasional excessive irrigation to leach salts from the root zone may address this concern but would partially offset the water savings of sprinkler irrigation. Our findings suggest that switching from furrow to sprinkler irrigation should generate broad soil health improvements, but farmers who manage salinity concerns should continue to monitor soil conditions. Long-term research of these transitioning systems is necessary to understand the ecosystem effects of transitioning to higher efficiency irrigation systems.

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CHAPTER 3 – CONSERVATION TILLAGE UNDER FURROW IRRIGATION IMPROVES SOIL HEALTH

SUMMARY

Despite the growing body of evidence that conservation tillage provides benefits to soil health, water quality, farm profitability, and greenhouse gas emissions of agroecosystems, widespread conversion to conservation tillage in semi-arid, furrow-irrigated systems remains limited due to challenges in managing residue in irrigation furrows. A long-term conservation tillage experiment was established in 2011 in Northern Colorado to study residue and tillage management strategies in furrow-irrigated agriculture, comparing conventional till to minimum and strip till. In 2021 and 2022, soil health was assessed using the Soil Management Assessment Framework (SMAF), which utilizes edaphic, site, and climate characteristics to interpret soil health measurements. After 10-11 years following conversion from conventional tillage, soil physical health was improved in the minimum and strip till plots, largely driven by decreases in bulk density (${}_b$), as was soil biological health, largely driven by increased potentially mineralizable N and biological activity, proxied by β-glucosidase, with smaller improvements in soil organic C. Soil chemical and nutrient health were similar in all treatments, but the minimum and strip till plots did express decreased salinity, which may be an important factor in semi-arid

Submitted to Soil Systems for Review and Publication

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systems. Grain corn (*Zea mays* L.) yields were consistent across all treatments in 2021 and 2022 indicating that as this system matures, competitive yields and improved soil health are achievable in conservation tillage systems under furrow-irrigation. Understanding soil health implications of varying tillage intensities should improve our ability to increase adoption of conservation management practices in furrow-irrigated agroecosystems.

1 INTRODUCTION

Conservation tillage has increasingly become a popular management practice for farmers looking to improve soil health and mitigate negative environmental impacts of crop production. Conservation tillage describes the practice of reducing soil disturbance and leaving at least 30% of the soil surface covered with crop residue (Conservation Tillage Information Center, 2002). The long-term practice of conventional tillage, traditionally characterized by soil inversion via moldboard plowing, has been shown to cause soil degradation through compaction, erosion, destruction of biological communities, and loss of soil organic C (Afshar et al., 2022; Bhattacharyya et al., 2022; Deleon et al., 2020; Green et al., 2007; Martin-Lammerding et al., 2013; Tan et al., 2002). Conversely, conservation tillage has the potential to improve soil biological activity (Deleon et al., 2020; Green et al., 2007), surface hydraulic conductivity (Baumhardt et al., 1992; Benjamin, 1993; Deleon et al., 2020), long term carbon storage (Afshar et al., 2022; Six et al., 1999, 2000), water filtration and storage, resistance to erosion, and nutrient retention (Fermanich et al., 2023; Fortuna et al., 2023; Patni et al., 1998; Waring et al., 2024). A study by Martin-Lammerding et al. (2013) observed benefits to active soil organic C and N that further supported biological activity in no-till and minimum till systems (i.e., conservation tillage systems) compared to conventional tillage. Moreover, Hedayatipoor and Alamooti (2020) observed improvements in soil aggregate stability and water infiltration in conservation tillage systems as compared to conventionally tilled fields, improving soil-water relations. This study did however measure increased bulk density (ρ_b) in conservation tillage systems, highlighting the importance of agronomic considerations in soil health assessments.

The adoption of conservation tillage still varies across agroecosystems, and relatively few studies have researched soil responses to conservation tillage in semi-arid furrow-irrigated systems. The interaction of tillage intensity and furrow-irrigation can be characterized by challenges in managing in-field residue to ensure proper water movement (Afshar et al., 2022). This dynamic has driven some producers in semi-arid regions to adopt a conservation till, but not a no-till management scheme to manage crop residue while reducing tillage intensity. A variety of conservation tillage practices that significantly reduce soil disturbance have been adopted by producers (e.g., strip tillage, mulch tillage, ridge tillage, minimum tillage, and zone tillage; Nowatzki et al., 2017; Wardle et al., 2015), but few studies have evaluated the effect of these tillage practices in these systems on soil health, and even fewer still have focused on tillage practice alterations and soil health in furrow-irrigated systems. Noting the lack of research on semi-arid, furrow-irrigated systems, the objective of this study was to evaluate the soil health impacts from multiple tillage intensity levels in these agroecosystems. To understand the potential for conservation tillage to improve soil health in a semi-arid, furrow-irrigated agroecosystem, we monitored soil health using the Soil Management Assessment Framework (SMAF) in a series of conservation tillage plots after ten years of conversion from conventional tillage. Research was conducted on production-scale field plots at an agricultural research station in Northern Colorado, where conventional till (CT) was compared against two conservation tillage methods: minimum till (MT) and strip till (ST; see supplementary materials for full tillage regime details). This study built upon prior work performed on the same series of tillage plots with varying tillage intensity (Deleon et al., 2020) that measured residue coverage, macrofauna populations, permanganate oxidizable C, aggregate stability, and infiltration capacity, finding generally positive results for soil health in the conservation tillage systems. The current study expands past observations by including the SMAF, a tested methodology to specifically quantify soil health. Additionally, we quantified particulate organic matter C (POM-C) and mineral associated organic matter C (MAOM-C) in order to understand how tillage may impact both the biologically active soil C and sequestered C. In agricultural systems, C fractionation may provide insight into how soil C impacts soil health: POM-C is generally more biologically available and may impact organic nutrient availability, whereas MAOM-C has a longer turnover time and contributes to long-term C sequestration.

Based on current literature, we formed the following hypotheses for comparisons of conservation tillage to conventional tillage:

a) Minimum and strip till (i.e., conservation tillage) will result in improved soil physical health by increasing aggregate stability and reducing compaction;

b) Minimum and strip till will increase soil organic matter as quantified by SOC, POM-C, and MAOM-C and improve biological soil health;

c) Soil chemical and nutrient health will not change between tillage treatments as all plots received similar fertilizer and irrigation inputs.

2 MATERIALS AND METHODS

2.1 Site Description

In 2011, a field experiment was established at Colorado State University's Agricultural Research Development and Education Center (ARDEC, 40°40'40"N 104°59'51"W, elevation = 1567 m) to compare three tillage regimes: MT, ST, and CT. Soils are predominantly the Garrett series, a fine-loamy, mixed, mesic Pachic Argiustoll (Soil Survey Staff, 2023). Monthly mean maximum temperatures reach a maximum in July around 30° C and a minimum in December around 5 °C, and average precipitation is approximately 270 mm, with most occurring between April to October (Colorado Climate Center, 2023).

Each tillage treatment was replicated in two blocks with all three treatments in both blocks as a randomized complete block design. Each production-scale plot was approximately 320 m long and 28 m wide and consisted of 36 furrows approximately 77 cm from crest of bed to crest of bed. While shorter plots would have allowed for greater replication and improved inferential statistical capacity, consultation from local farmers guided the decision to create long plots that represent the water movement challenges faced by farmers attempting to adopt conservation tillage in furrow-irrigated agriculture and has yielded important insights that continue to inform farmers in the region. Every other row received irrigation water delivered by siphon tube. Prior to the start of the experiment (2005), the entire field was under the same management regime: a sunflower (*Helianthus annus)*- Corn (*Zea mays* L.) – dry beans (*Phaseolus vulgaris)* rotation under full moldboard-plow tillage comparable to the current CT treatment.

Field operations in 2021 and 2022 for each tillage treatment are summarized in the Supplementary Table S1. All plots received two vertical till operations per year, while CT plots received an additional moldboard plow operation and ST plots received and additional strip till operation. CT plots also received additional field operations consistent with common agricultural practices in the region, including a harrow operation in 2021, and mulching and cultipacking operations in 2022. While retention of residue has been shown to improve soil health characteristics including water retention (Schneekloth et al., 2020), removal of residue via chop and bale was necessary to manage furrow irrigation. Though residue removal was consistent across plots, the CT treatment mixed more of this residue into the subsoil through moldboard plowing. Thus, conservation tillage in this system should be viewed as a combination of reduced soil disturbance and increased surface crop residue. All plots received cultivation and row cleaning to manage water movement in furrows. While all plots received equivalent N inputs in 2021 (~180 kg ha^{-1} N), the CT and MT plots also received ~34 kg P ha^{-1} and the ST plots received 25 kg P ha^{-1}. In 2021, the CT plots received all N and P inputs in a broadcast operation of mixed DAP and urea in early April; the MT plots received 50% of the N and 100% of their P in an early April DAP-urea broadcasting, and 50% of the N in a mid-June banding UAN application; and the ST plots received 38% of the N and 100% of the P in an early April ammonium phosphate-urea banding application (with small additions of Black Label Zn and ReaX K), and 62% of the N in a mid-June UAN banding application. The improved application efficiency of banding applications over broadcasting is a justification for decreased P application levels in the ST plots. This split-application is included as part of the management scheme to reflect the addition of multiple improved management practices used by producers, and the pre-plant band application in the ST plots takes advantage of fertilizer lines on the back of most strip till implements. In 2022, increased fertilizer costs forced farm managers to apply a consistent fertilizer application across all fields: ~ 2 kg N ha^{-1} and ~ 6 kg P ha^{-1} as a banded Optistart Gold pre-plant application in late April, followed by an application of ~ 3 kg N ha^{-1} as NPACT in early June and a band application of UAN at ~71 kg N ha^{-1} a week later. In 2021 and 2022, a flexible dent corn variety (DeKalb DKC 47-54RIB or Pioneer P9489Q, respectively) was sown for grain in all plots. In both years, the seeding rate was 88,950 seeds ha^{-1}.

2.2 Soil Sampling and Analysis

2.2.1 Soil Management Assessment Framework (SMAF)

The Soil Management Assessment Framework (SMAF; Andrews et al., 2004) is a Microsoft Excel-based tool used to score and provide relative interpretations of soil health measurements within the context of climactic conditions, cropping system, soil taxonomy and texture. Selection of soil indicators may be based on an intended research goal, but are broadly split into four categories: soil physical indicators (ρ_b, and water-stable aggregates [WSA]), soil biological indicators (soil organic C [SOC], microbial biomass C [MBC], potentially mineralizable N [PMN], and β-glucosidase activity [BG]), soil chemical indicators (pH [$pH_{1:1}$] and electrical conductivity [$EC_{1:1}$]), and soil nutritional indicators (plant-available P and K). Indicators are selected to represent key soil ecosystem services, agronomic needs, and sensitivity to changes in management (Andrews et al., 2004). The SMAF translates the raw soil indicator measurements into unitless scores from 0 to 1 (0 being "worst" and 1 being "best"), based on algorithms accounting for soil texture, climate, and cropping system. The SMAF has been used extensively to monitor soil health in systems converting between tillage schemes, and the indicators listed above have been shown to be sensitive to changes in soil disturbance associated with conservation tillage (Afshar et al., 2022; Andrews et al., 2004; Ye et al., 2021; Yuan et al., 2022).

2.2.2 Soil Sampling

Soils were collected using a 2.5 cm diameter step probe to a depth of 15 cm. Approximately 30 cores were sampled within a 3-m radius centered around a GPS located point, composited, and mixed in a plastic bucket before being transferred to a plastic Ziploc bag, sealed, and placed in a dry cooler with ice packs. Within this sampling radius, an additional intact core was taken for $_b$ determination. Soils for ρ_b were placed into pre-weighed tin cans, and the lids immediately sealed with tape to reduce evaporative losses. All soil samples, including $_b$ samples, were obtained on the irrigated shoulder of furrows. In 2021, two composite samples were taken in each plot, one approximately 30 m up slope from the bottom of the plot, and one approximately 30 m down slope from the top of the plot, each centered on row number 15 of 36. In 2022, an additional composite sample was taken near the top and bottom in row 19 of each plot as a water quality sampling device was placed in this row at the bottom of the field. Samples were also taken in row 15 to maintain continuity between 2021 and 2022. Sampling occurred on July 16^{th}, 2021 and July 21^{st}, 2022 during the peak of corn growing season.

Soils were returned to the laboratory the same day and stored at 4 ºC until processing. Bulk density and moisture content was determined by immediately weighing moist cores stored in tin cans, removing the lid, drying at 105 ºC for 24 hr, then weighing. The remainder of the soil samples were passed through an 8-mm sieve to remove rocks and large plant debris.

Approximately 150 g of field-moist, 8-mm sieved soil was stored at 4 ºC prior to further analysis for MBC (approximately 2-3 weeks after sampling). An additional ~150 g of soil was passed through a 2-mm sieve and air-dried, and the remainder of the 8-mm sieved soil was air-dried, and then subsamples were returned to Ziploc bags and stored at room temperature.

2.2.3 Soil Physical Health Indicators

Bulk density and soil moisture content were determined by weighing intact, field-moist soil cores of known volume, then drying at 105 °C for 24 h followed by weighing. During sampling, ρ_b samples were taken using a Madera sampling probe, a tool that preserves the known shape and volume of a soil core and minimizes compaction (~ 60 cm^{-3}).

Water-stable aggregates were determined according to Kemper and Rosenau (1986). One hundred g of 8 mm-sieved, air-dried soil was placed on four stacked sieves 23 cm in diameter. A 2 mm sieve was on top followed by a 1.0, 0.5, and 0.25 mm sieve so the finest screen size was on the bottom. The sieves were submerged in a 19 L bucket of water on a Yoder sieving machine at 30 strokes min^{-1} for 5 min. Soil remaining on each sieve was rinsed into a single pre-weighed aluminum pan. The percent of water stable aggregates was determined as the weight of soil in the pan after drying at 105 °C until complete dryness, divided by the original soil mass, accounting for the mass of rocks and plant material > 2mm.

2.2.4 Soil Biological Health Indicators

β-glucosidase activity was determined according Green et al. (2007). In triplicate, 1.0 g of air-dried 2 mm sieved soil was weighed into 50 mL Erlenmeyer flasks to make a control, sample, and duplicate set. An additional blank Erlenmeyer flask was added to the set of control flasks. Then, 4 mL of modified universal buffer (pH 6.0) and 0.25 mL of toluene were added to all three sets of flasks, and 1 mL of 0.05 M ρ-nitrophenyl-β-glucopyranoside (PNG) was only added to the sample flasks and duplicate flasks. All samples were then swirled and incubated for 1 h at 37 °C, at which point 1 mL of 0.5 M $CaCl_2$ and 4 mL of 0.1 M TRIS (hydroxymethyl) aminomethane (THAM) buffer (pH 12) was added to all flasks to halt the reaction, and 1 mL of 0.05 M PNG is added to the control flasks. Following filtration through Whatman #2 filter paper, the filtrate was diluted by adding 4 mL of 0.1 M THAM to 1 mL of sample. B-glucosidase activity was measured on a Genesys 10S UV-VIS spectrophotometer at 410 nm. A standard curve was developed using *p*-nitrophenol at 0, 10, 20, 30, 40, and 50 μg L^{-1} in the sample matrix (1 mL of 0.5 M $CaCl_2$ and 4 mL of 0.1 M THAM).

Microbial biomass C was determined via chloroform fumigation (Beck et al., 1997; Hobbie, 1998). Duplicate, 8-mm fresh, field-moist soil samples (no more than three weeks old) were weighed into either a glass beaker or 125 mL Nalgene bottle at an equivalent of to 10 g oven-dried soil. The samples placed in glass beakers were incubated in the dark for 4 days following chloroform fumigation, while the duplicate samples were left unfumigated. Next, all samples were shaken in 0.5 M K_2SO_4 for 1 hr and filtered through Whatman #2 filter paper. Filtrate dissolved C was measured on a TIC/TOC analyzer (Shimadzu TOC-L; Shimadzu Scientific Instruments, Inc., Kyoto, Japan). Microbial biomass C was determined by subtracting unfumigated dissolved C from fumigated dissolved C.

Total C was measured by dry combustion on a VELP Dumas Elemental Analyzer (VELP Scientifica, Usmate Velate, Italy; Nelson and Sommers, 1996) using 0.100 g of powder-ground air-dried soil, while inorganic C was determined via the pressure transducer method (Sherrod et al., 2002) using 1.00 g of powder-ground air-dried soil. Soil organic C was determined by subtracting inorganic C from total C.

Potentially mineralizable N was determined through a 28-day incubation (Curtin and McCallum, 2004). Approximately 30 g of air-dried, 2 mm sieved soil was weighed into a 50 mL beaker and brought to a $_b$ of approximately 1.0 g cm^{-3}. Deionized water was added to bring the soil to 60% water-filled pore space, and the flask was placed in a Mason jar with ~1 cm of water in the bottom of the jar to ensure a humid incubation environment. Each jar was placed in a dark cabinet at room temperature, and every 7 days the jars were opened briefly to allow air exchange. Following the incubation, a 10 g subsample of soil was removed and placed into a 125 mL plastic bottle. Concurrently, a 10 g sample of the non-incubated air-dried 2 mm sieved soil was weighed into a 125 mL plastic bottle as a control. All bottles of soil received 50 mL of 2M KCl and shaken for 30 min prior to filtration through Whatman #1 filter paper. Filtrate NO_3-N was determined by dilution 15 µL of sample filtrate in 250 µL of Vanadium (III) Chloride reagent and 35 µL 2M KCl (Doane and Horwath, 2003). This process forces a Griess reaction, and the resulting NO_2-N concentration was measured colorimetrically on a Genesys 10S UV-VIS spectrophotometer at 540 nm. Filtrate NH_4-N was determined by dilution 15 µL of sample filtrate with 25 µL citrate reagent, 50 µL salicylate-nitroprusside reagent, 25 µL hypochlorite reagent, and 160 µL 2M KCl (Mulvaney, 1996). NH_4-N was measured colorimetrically on a Genesys 10S UV-VIS spectrophotometer at 610 nm. Separate standard curves of both NO_3-N and NH_4-N created at 0, 0.1, 0.5, 1, 2, 5, 10, 20, and 40 mg L^{-1} of the respective analyte.

2.2.5 Soil Chemical Health Indicators

Soil pH and EC were determined in a 1:1 soil solution, shaking 20 g of 2-mm sieved, air-dried soil in 20 mL DI water for 2 hr. Soil pH was read directly in the slurry using a pH electrode, and $EC_{1:1}$ was measured in the liquid phase using a conductivity meter immediately after centrifuging the samples for 5 min to separate the liquid from solid phase.

2.2.6 Soil Nutrient Health Indicators

Due to the high pH of all soil samples, plant-available P and K concentrations were determined via an Olsen extraction (Olsen et al., 1954) where 2.00 g of 2-mm sieved, air-dried soil was shaken in 40 mL of 0.5 M sodium bicarbonate and filtered through Whatman #2 filter paper. Filtrates were loosely sealed in parafilm and left to sit overnight to allow CO_2 gas release. Filtrate was diluted at a 10:1 ratio in DI water. Extractable P and K were measured using inductively coupled plasma-optical emission spectrophotometry (ICP-OES) at 213.617 nm and 766.490 nm, respectively.

2.2.7 Soil C Fractionation

Bulk soils were fractionated according to Cotrufo et al. (2019) to separate the POM and MAOM on the 2021 soil samples only as fractionation is relatively time intensive. In short, 5.25 – 5.75 g of 2-mm sieved, oven-dried soil was placed in a 50 mL centrifuge tube with 12 glass beads and 30 mL of 0.5% sodium hexametaphosphate solution before being shaken at a 120 rpm for 18 hrs. The dispersed soil was then poured over a 2 mm sieve stacked on top of a 53 µm sieve to easily remove the glass beads. The soil on the 53 μm sieve was then rinsed until the water ran through clear. All soil that passed through the 53 μm sieve was collected and classified as MAOM while the soil on top of the 53-μm sieve was retained and classified as POM. Both fractions were oven-dried at 60 °C, weighed, finely ground, and analyzed by dry combustion on a VELP Dumas Elemental Analyzer (VELP Scientifica, Usmate Velate, Italy) for total POM-C and total MAOM-C. Bulk soil C recovery after fractionation was determined using total C data from prior SMAF SOC and inorganic C measurements. Samples with total C recovery falling outside of 90-110 % were re-analyzed. If after reanalysis, samples fell outside of 80 – 120% total C recovery, those samples (of which there were 2) were removed from statistical analysis.

2.2.8 Grain Yield and Surface Residue

Grain yield was determined as the weight of harvested corn grain in a 12-row section in the center of each plot. The grain was immediately transferred to a grain truck, and the truck weighed on an industrial scale. A subsample of the grain was processed in a GAC 2700-AGRI Corn Grain instrument to determine corn grain moisture at the time of harvest, and yield was then adjusted to represent 15.5% moisture.

Surface crop residue was determined near the top, middle, and bottom in each plot. A 1.5 m^2 quadrat was randomly placed in the approximate lateral center of each plot. All crop residue that could be gathered in the square without picking up soil or removing buried residue was collected. The residue was placed in a paper bag at 60 °C for at least 72 hr and weighed. This process was performed immediately before planting, immediately after harvest, and immediately after chopping and baling crop residue.

2.2.9 Statistical Analysis

Analysis of variance (ANOVA) was performed on all raw measurements of soil indicators as a linear mixed effects model with year, treatment, and sampling section (top vs. bottom of the field) as fully interacting factorial predictor variables and block and plot as non-interacting random effect variables. Logarithmic transformation of the outcome variable was performed to fulfill the assumptions of the ANOVA model, and statistical significance was evaluated at the $\alpha \leq 0.05$ level. Pairwise comparisons were performed for significant year, treatment, or sampling location effects with a Tukey adjustment of p-values and are presented in the results and discussion to provide a more holistic view of field dynamics. The same analysis was repeated for all SMAF scores of soil indicators, including those that are averages of multiple scores (noted as physical, biological, chemical, nutrient, and overall soil health indices (SHIs)). All statistical analyses were performed in RStudio (Build 369) using R (v 4.2.2; RStudio Team, 2020) and the stats package (v4.2.2), lme4 package (v1.1-35.1) and lmerTest package (v3.1-3). This process was repeated for crop residue at each sampling time ($n = 18$ per year), and POM-C / MAOM-C ($n = 12$, but two samplers were removed due to percent recovery for $n = 10$).

3 RESULTS AND DISCUSSION

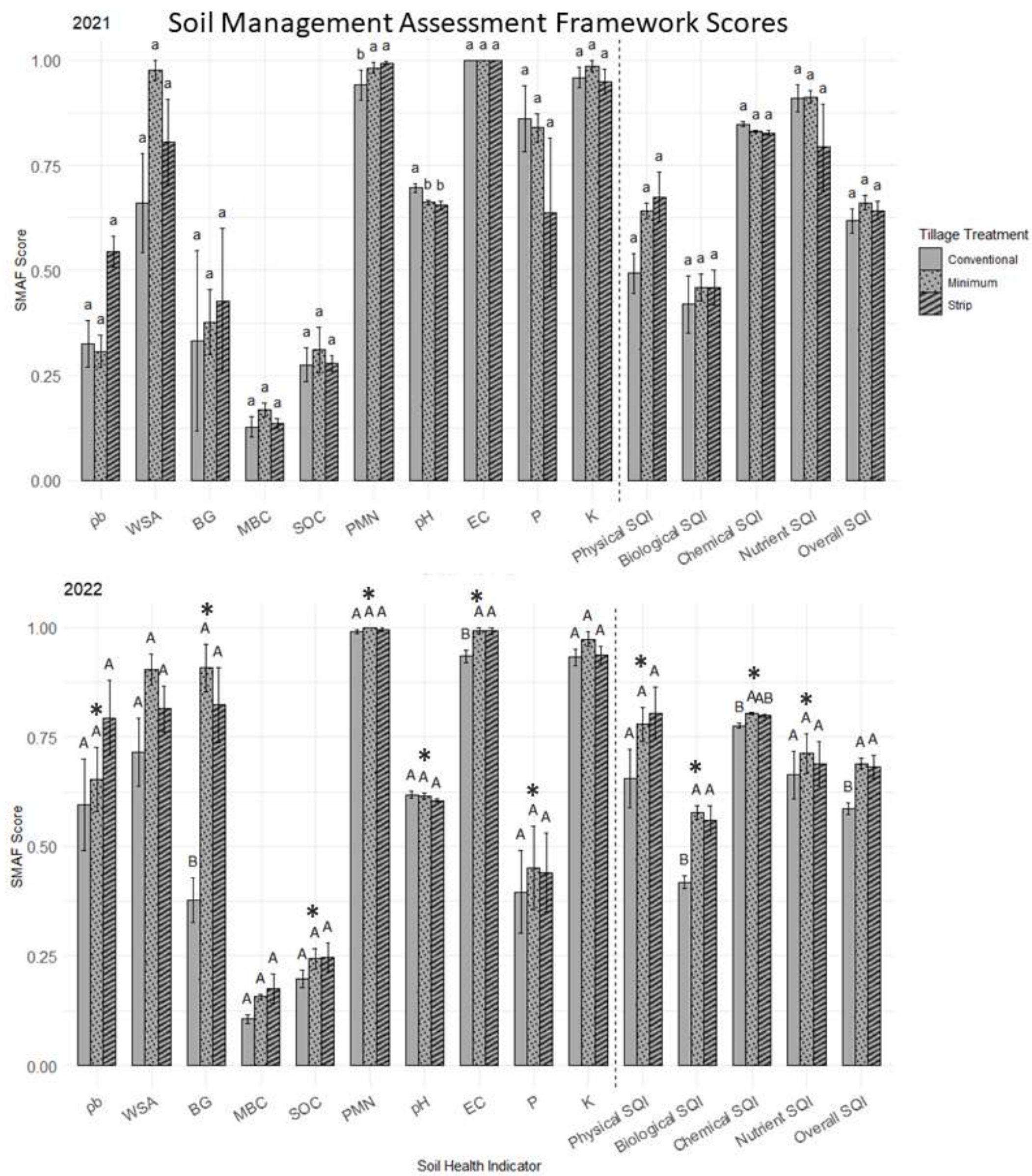
3.1 Soil Health Outcomes

All SMAF indicators are presented in Table 3.1, and SMAF indicator scores are presented in Figure 3.1. Alongside these measures is the p-value associated with ANOVA factors, interactions, and pairwise comparisons.

Table 3.1. Conventional (CT), minimum (MT) and strip tillage (ST) means (± SE) of each soil health indicator for 2021 and 2022 and ANOVA test for all parameters and two-way interactions.

Soil Indicator†	Year	CT	MT	ST	ANOVA p-Value: Year	Tillage	Section	Year x Tillage	Year x Section	Section x Tillage
Physical										
ρ_b (g cm^{-3})	2021	1.63 ± 0.03	1.58 ± 0.07	1.37 ± 0.05	0.0021, 2021 > 2022	0.0012	0.33	0.56	0.28	0.75
	2022	1.43 ± 0.07	1.32 ± 0.03	1.23 ± 0.04						
WSA (%)	2021	35.6 ± 13.3	55.4 ± 5.9	45.7 ± 11.0	0.52	0.60	0.0057, Top > Bottom	0.80	0.064	0.26
	2022	36.5 ± 7.0	49.1 ± 5.1	42.5 ± 4.1						
Biological										
BG (mg pnp kg^{-1} soil hr^{-1})	2021	122 ± 52	158 ± 11	137 ± 35	< 0.001, 2022 > 2021	< 0.001	0.17	0.013	0.37	0.11
	2022	131 ± 11	290 ± 12	253 ± 13						
MBC (mg g^{-1})	2021	111 ± 10	151 ± 14	132 ± 11	0.26	0.12	0.24	0.39	0.011	0.74
	2022	95 ± 3	144 ± 8	135 ± 8						
SOC (%)	2021	1.44 ± 0.16	1.58 ± 0.07	1.54 ± 0.11	0.0034, 2021 > 2022	0.46	0.24	0.76	0.12	0.43
	2022	1.19 ± 0.05	1.43 ± 0.08	1.37 ± 0.05						
PMN (mg kg^{-1}) *	2021	16.5 ± 1.8	19.1 ± 1.0	26.0 ± 7.3	0.029, 2022 > 2021	0.28	0.77	0.12	0.61	0.012
	2022	19.6 ± 1.4	25.6 ± 1.1	23.3 ± 1.6						
Chemical										
pH, 1:1	2021	8.00 ± 0.03	8.12 ± 0.02	8.14 ± 0.04	< 0.001, 2022 > 2021	0.0044	0.16	0.094	0.41	0.77
	2022	8.26 ± 0.03	8.27 ± 0.02	8.31 ± 0.01						
EC, 1:1 (dS m^{-1})	2021	0.84 ± 0.09	0.65 ± 0.05	0.70 ± 0.02	< 0.001, 2022 > 2021	0.059	0.15	0.35	0.22	0.056
	2022	1.29 ± 0.04	1.11 ± 0.04	1.04 ± 0.04						
Nutrient										
P (mg kg^{-1})	2021	14.3 ± 2.5	12.1 ± 0.8	10.1 ± 2.8	< 0.001, 2021 > 2022	0.55	0.0075, Bottom > Top	0.095	0.43	0.045
	2022	5.7 ± 1.0	6.7 ± 0.9	6.5 ± 0.9						
K (mg kg^{-1})	2021	221 ± 17	223 ± 13	200 ± 25	0.18	0.087	0.011, Bottom > Top	0.69	0.099	0.60
	2022	196 ± 11	221 ± 12	185 ± 8						

Figure 3.1. Conventional (CT), minimum (MT), and strip tillage (ST) means (± SE, n = 6) of SMAF indicator scores for 2021 and 2022 and pairwise comparisons of the ANOVA test for treatment and year.



† ρ_b = bulk density; WSA = water stable aggregates; BG = β-glucosidase activity; MBC = microbial biomass carbon; SOC = soil organic carbon; PMN = potentially mineralizable nitrogen; EC = electrical conductivity; P = plant-available phosphorus; K = plant-available potassium. Asterisks (*) indicate a significant effect of year for the respective SMAF score.

3.1.1 Soil Physical Health Indicators

Bulk density was significantly affected by treatment, driven by a lower $_b$ in the ST treatment than in the CT treatment in both years (Table 3.1). Interestingly, treatment was not a significant predictor of the $_b$ score (Figure 3.1), and most of the variance in the ANOVA model appeared to be explained by variation in year, with 2022 scoring significantly better due to decreased $_b$ in all plots. However, in both years the general trend appeared to be a decrease in $_b$ in the ST plots compared to the CT plots, with MT $_b$ falling between the two. Indeed, when comparing within 2021 or 2022, ST had significantly decreased $_b$ ($p = 0.031$) or nearly significantly decreased $_b$ ($p = 0.065$) as compared to CT, respectively. The small difference between ST and MT treatments was likely due to the strip tillage operation that was performed along the seedbed prior to planting. The purpose of a strip tillage operation is to prepare the seedbed to ensure proper emergence, and soil samples were taken in the shoulder of the seedbed. The moldboard plow operation performed in the CT treatment may loosen the soil surface in the short term but causes long-term compaction, especially at the plow pan depth (31.4 – 39.3 cm). Given that this moldboard plow operation takes place well before planting (late November 2021 and early March 2022), the soil surface may re-compact through the growing season, and decreased SOC in the conventionally tilled systems has been associated with increased long-term compaction (Nunes et al., 2018). In general, studies have reported mixed results on the effects of tillage and $_b$ or compaction (Afshar et al., 2022; Benjamin, 1993; Deleon et al., 2020; Nunes et al., 2018), though this study seems to indicate that as this system reaches full maturity after 10 years of conservation tillage practices, $_b$ is generally improved in these conservation systems, and particularly in those employing strip tillage.

While WSA was roughly 13-20% greater in the MT treatment and 6-10% greater in the ST treatment compared to the CT treatment over both years (Table 3.1), treatment was not a statistically significant predictor of WSA. Given the extensive research linking soil aggregation to management and other important soil health indicators and water quality (Afshar et al., 2022; Fermanich et al., 2023; Hedayatipoor & Alamooti, 2020; Martin-Lammerding et al., 2013) and the large quantity of intense runoff on these plots, it is notable that WSA did not seem to be improved by treatment, and no predictor had a significant effect on the WSA score (Figure 3.1). The only significant predictor of the raw WSA measurement was field section, with the top of the field having significantly greater WSA than the bottom of the field. In fact, in 2021 the top of the field had significantly greater WSA ($p = 0.0057$), but this was not observed in 2022 ($p = 0.34$). Ippolito et al. (2017) also found no difference in WSA between field tops and bottoms yet did not discuss reasons behind this observation. The dynamic observed in 2021 may be driven by the erosion of unstable aggregates from the top of the field, which are then deposited at the bottom of the field, leaving behind more stable aggregates at the top. It is worth noting that there was large variability in WSA within treatments, with standard errors in raw measurements ranging from ~10% to ~37%. A more sensitive and less variable method for evaluating soil aggregate stability may be appropriate for future studies.

Overall, soil physical health (i.e., an average of the SMAF scores for $_b$ and WSA) was slightly improved in conservation till plots, largely driven by improvements in $_b$ in the ST plots, but this did not result in a significant difference between treatment with respect to the physical SHI score (Figure 3.1). As with all physical indicators, the field section was not a significant predictor of physical SHI score. In both 2021 and 2022, the physical SHI was greater in the MT and ST plots than in the CT plots. This is in contrast with our hypothesis that overall physical soil health would be improved by conservation tillage, though it seems that there is still a benefit to $_{b}$ in the ST system even if not reflected in SMAF scores, which may be a result of the additional conservation tillage operation along the seedbed and may still have a meaningful agronomic – if not statistical – implications. Bulk densities exceeding 1.47 g cm^{-3} have been shown to limit and even halt root growth in clayey soils (Correa et al., 2019), so the finding that these conservation tillage operations reduced $_{b}$ by ~0.2 g cm^{-3} indicates that conservation tillage has likely reduced or eliminated the risk of over-compaction to yield.

3.1.2 Soil Biological Health Indicators

Sampling year had a significant effect on raw measurements of BG, SOC, and PMN; BG and PMN increased in 2022 and SOC decreased in 2022 (Table 3.1), and this pattern was identical with respect to SMAF scores (Figure 3.1). In general, there was little effect of treatment on biological indicators of soil health, but this appears to be highly stratified by year, as indicated by the low p-values for Year x Treatment for both BG and PMN.

β-glucosidase is a common soil health indicator used to proxy the potential of soil microbial communities to assimilate organic matter, generally indicating healthier soil microbial communities (Andrews et al., 2004; Stott et al., 2010). Notably, the significant interaction of Year x Treatment prompted further investigation, finding that differences in 2022 were much larger than in 2021. In 2022, β-glucosidase activity, as indicated by both raw measurements and SMAF scores, were significantly greater in the MT and ST treatments than in the CT treatment ($p = 0.044$ and $p = 0.013$, respectively for raw measurements (Table 3.1), and $p = 0.012$ and $p = 0.037$ respectively, for SMAF scores [Figure 3.1]), though no significant difference was detected in 2021 or between MT and ST plots in 2022. In fact, the 2022 raw measurements of β-glucosidase activity in either of the conservation till treatments were approximately double that of the CT treatment, indicating that increased crop residue and decreased soil disturbance had a positive impact on exoenzyme activity (Bandick & Dick, 1999; Paudel et al., 2011).

As a proxy of biological activity, improvements to β-glucosidase activity should also be associated with improvements in microbial biomass C, a measure of microbial community size. Indeed, raw measurements of MBC in 2022 were far greater in the MT and ST treatments compared to the CT treatment, though treatment did not have significant effect on either raw measurement of MBC or SMAF scores (Table 3.1 and Figure 3.1). This may be in part due to the extremely low scores observed for all samples: almost all fell below 0.2, perhaps as a function of the drier climate and annual cropping system limiting MBC population. Still, the raw measurements of microbial biomass C and β-glucosidase activity are both greater in the conservation till treatments than in the CT treatments, and thus suggests that conservation tillage is having a positive impact on microbiological health even if this is not statistically significant. Interestingly, the interaction of year and section was significant for both raw measurements and SMAF score ($p = 0.015$ for SMAF score), with the top of the field having greater raw measurements of MBC in 2021 ($p = 0.023$) and the bottom having non-significantly greater MBC in 2022 ($p = 0.18$), though SMAF scores only detected a difference in 2022 ($p = 0.24$ for 2021, $p = 0.0092$ for 2022). This resulted in a significant effect of section on MBC SMAF score ($p = 0.017$). This is confusing, as we expect to find statistical differences between field sections to be detected in both SMAF and raw measurements, or in neither. However, the SMAF scoring curve is extremely steep at the lowest end of the curve, and small changes in MBC result in large changes in score, so statistical findings should be interpreted with caution, and raw measurements may be more important.

Microbial communities depend on soil organic matter as a substrate for growth and reproduction, so we generally expect to see increased SOC in systems with increased β-glucosidase activity and MBC (Akhtar et al., 2018; Paudel et al., 2011; Zhang et al., 2019). Unsurprisingly, SOC follows a very similar pattern as MBC: relatively large differences in magnitude in raw measurements (Table 3.1), but smaller differences in magnitude in the SMAF scores (Figure 3.1). Although treatment did not have a significant effect in either ANOVA model, there was a significant effect of sampling year. As with microbial biomass C, while the magnitudes of change between treatments was somewhat large, this is not reflected in scores for SOC, as SOC is limited in semi-arid annual cropping systems, and current SMAF scoring curves may be poorly fit for these systems. Soil organic C was much greater in the MT treatment than in the CT treatment in both years, with ST falling between these two treatments. While average SOC scores were below 0.3, SOC values of ~1.4-1.5% in the conservation till treatments are actually reasonably high for row crop systems in the region, as very few soils exceed 2%, and many do not exceed 1% SOC. Regardless, the increases of SOC in the conservation tillage plots indicate that these plots are sequestering more C via crop residue, a dynamic that is well-documented in tillage studies (Butkevičienė et al., 2023; Afshar et al., 2022; Luo et al., 2010; Six et al., 1999, 2000). However, the lack of significant difference in SOC with respect to treatment over a decade into the study contrasts with the notion that conservation tillage has a significant effect on SOC accumulation, an important ecosystem service provided by soils (Fan et al., 2018; Liptzin et al., 2022; Luo et al., 2010; Paustian et al., 2019; Six et al., 1999, 2000; Smith et al., 2020). This lack of statistical significance may be somewhat driven by low sample size however, as the magnitude of difference between treatments may still be relevant for soil health and agronomic purposes.

Organic N may serve as an important plant available nutrient source, as well as a source of N for protein synthesis in soil microorganisms (Bridgham & Ye, 2015). Potentially mineralizable N serves as a measure of the potential of a system to convert organic N to plant-available N. PMN was greatest in the ST plots in 2021 and in the MT plots in 2022, with CT plots consistently having the lowest PMN (Table 3.1). Although PMN raw indicators were not affected by treatment, PMN SMAF scores were significantly affected by treatment, with MT and ST both exceeding CT (Figure 3.1). Averaging across sections, CT had a lower PMN SMAF score than MT or ST ($p = 0.011$ and $p = 0.0020$, respectively), but the remarkably high PMN SMAF scores limit the capacity for scoring curves to detect effect size, so all statistical interpretation of these scores should be interpreted with caution. Notably, the interaction of section and treatment was significant for both raw measurements and SMAF scores ($p < 0.001$ for SMAF scores), and closer investigation indicates that treatment played a stronger role in the top of the field than in the bottom, with raw PMN measurements in the MT and ST plots exceeding CT plots in the top of the fields (though insignificantly: $p = 0.33$ and $p = 0.086$, respectively), regardless of year, while the effect was much smaller in the bottom of the field (and again insignificant: $p = 0.66$ and $p = 0.98$, respectively). When we consider the dynamic observed in WSA, it seems as though water flow plays a large role in the spatial distribution of soil health parameters. Increased treatment effect at the top of the field may be an indicator of resistance to erosion, which would carry organic-borne N from the top of the field and deposit it in the field bottom or carry it off the field. Ippolito et al. (2017) indicated that in furrow-irrigated systems, erosion and deposition dynamics play a significant role in spatial distribution of soil health parameters, but they did not observe similar dynamics with respect to PMN. Field bottoms trended towards greater PMN as compared to field tops, resulting a significant effect of section on PMN SMAF score ($p < 0.001$). Regardless of spatial distribution of PMN, the general improvement in PMN across treatments agrees with a review by Mahal et al. (2018), finding that conservation tillage resulted in increased PMN in cropping systems, and indicated the potential for these systems to reduce reliance on chemical fertilizer, which may be more easily mobilized and transported off field (Dinnes, 2004). Therefore, while erosional dynamics of PMN require further research, current findings are supportive of our hypothesis that biological soil parameters would improve under conservation tillage – at least at the top of the fields.

Overall, biological soil health, an average of the SMAF scores for β-glucosidase activity, microbial biomass C, SOC, and PMN, was only mildly improved by conservation tillage, with both MT and ST plots outperforming CT plots and with MT plots slightly outperforming ST plots (Figure 3.1). In 2021, there was no observable difference between treatments, but in 2022, MT and ST plots had greater biological soil health indices than the CT plots, though there was no significant effect of treatment across years. While MBC score and PMN score were significantly affected by field section, field section was not a significant predictor of biological SHI score. As noted previously, it seems as though differences between treatments vary somewhat by year, and 2022 provided much larger differences than 2021, as indicated by both raw measurements and the biological SHI score. It generally seems that after 10 years, conservation tillage has resulted in increased SOC for microbial communities, and consequently, increased microbial size, activity, and function. This agrees with our hypothesis that the reduced soil disturbance and increased organic matter inputs of the conservation tillage systems would improve soil biological health, though small sampling size limits our statistical capacity to identify treatment effect and current SMAF scoring curves seem poorly fit for semi-arid systems.

3.1.3 Soil Chemical Health Indicators

Chemical indicators of soil health include pH and EC. In agroecosystems, pH and EC are used as chemical indicators of plant growth limitations. Soil pH controls bioavailability of key plant nutrients and has often been referred to as the "master variable" in soils (Neina, 2019). Soil pH was slightly higher in the ST and MT plots than in the CT plots in both years, resulting in a treatment effect on raw measurements (Table 3.1), but not on SMAF scores (Figure 3.1). In 2021 specifically, ST and MT had significantly greater pH than in CT ($p = 0.013$ and $p = 0.037$), resulting in a significantly lower SMAF score ($p = 0.014$ and $p = 0.039$, respectively), but this was not observed in 2022, and no other statistical differences were detected when averaging across field section. It is important to note that the difference in mean pH between the ST and MT plots and the CT plots was less than 0.2 units, and likely did not impact nutrient availability or plant growth.

In semi-arid regions, salinity is a significant concern for farmers, particularly in irrigated systems (Bauder et al., 2011; Waskom et al., 2012). High salinity may impact plant growth by limiting root growth, causing water stress, and potentially leaf burn (Waskom et al., 2012). Electrical conductivity was ~ 0.2 ds m^{-1} lower in the MT and ST plots than in the CT plots in both 2021 and 2022 (Table 3.1), resulting in a significant effect of treatment on SMAF score (Figure 3.1). As a note to emphasize the limitations SMAF scoring curves near extreme values, while in 2022 CT plots had significantly worse EC SMAF scores than MT or ST ($p = 0.044$ for both comparisons), every sample in 2021 had a SMAF EC score of 1.0, and every score was above 0.91 in 2022. This was due to the relatively low EC in all samples, with EC only presenting concerns above 2 ds m^{-1} (using the 1:1 method in this study). Still, it is worth noting that EC increased from 2021 to 2022 in all plots but much less so in the conservation tillage plots. Dalal (1989) found decreased EC and exchangeable sodium percentage in no-till systems compared to conventional till systems, citing improved infiltration resulting in leaching of salts as a potential mechanism for this difference. When considering just the bottom of the field, CT had a greater EC than either MT or ST ($p = 0.034$ and $p = 0.016$, respectively), and MT and ST were not significantly different from each other ($p = 0.77$). This may support the theory that improved infiltration at the bottom of the field, where water flow may be more limited to leach salts, is an important mechanism in reducing salinity in conservation tillage plots. Conservation tillage has been associated with improved infiltration (Benjamin, 1993; Deleon et al., 2020; Hedayatipoor & Alamooti, 2020; Nunes et al., 2018), particularly through mechanisms of $_b$ and WSA. This may also be the case in the current system, as conservation till plots with decreased $_b$ also had decreased EC.

The chemical SHI is an average of SMAF pH and EC scores (Figure 3.1). In 2021, the ST and MT plots had a slightly lower chemical SHI than the CT plots, and in 2022, the MT and ST plots had slightly better chemical soil health indices than the CT plots, resulting in a significant effect of treatment for the chemical SHI. As with all chemical indicators, the field section was not a significant predictor for chemical SHI. Due to relatively low variability in measured pH and EC, p-values for chemical soil health SMAF scores were quite small, though the magnitude of difference between treatments was also quite small. Given that 2021 differences were driven by small differences in pH and 2022 differences were driven by increasingly large differences in

EC, it may be relevant to continue to study the role of EC on soil health in this system. Averaging across field section, the only significant difference in chemical SHI was between the CT and MT fields in 2022, where MT fields outperformed CT fields ($p = 0.047$), with ST fields trending towards a similar relationship with CT fields ($p = 0.080$). These conflicting impacts of tillage on soil chemical health contrast with our hypothesis that chemical soil health would not vary by treatment – the role of infiltration in mitigating salinity concerns in conservation tillage warrants the use of chemical soil health indicators in semi-arid studies focused on conservation tillage.

3.1.4 Soil Nutrient Health Indicators

Plant-available P and K are used as SMAF indicators of nutrient health in agroecosystems. Phosphorous, as both raw measurement (Table 3.1) and SMAF score (Figure 3.1), were not strongly affected by treatment in either year. Given that P requirements are generally met through fertilizer additions, this is unsurprising. By system design, soil P levels are maintained at a consistent level between treatments. However, P did seem to decrease across treatments between 2021 and 2022, which may have been driven by lower fertilizer P inputs, P exports in the previous crop, and P fixation via soil mineral precipitation. The K score (Figure 3.1) was not strongly impacted by treatment, but raw measurements of plant-available K were slightly lower in the ST plots than in the other plots in 2022 (Table 3.1). However, it is worth noting that K is generally plentiful and rarely limiting in Colorado soils, indicated by the high SMAF K scores across all treatments (Figure 3.1). Notably, soil P and K were lower in block 2 than in block 1, though this effect was consistent across treatment. Given that soil P is largely driven by fertilizer inputs and soil K is largely driven by edaphic factors, this dynamic is indicative of the need to consider the impact of all management strategies and baseline environmental conditions on measured soil health, rather than attributing all differences to studied management decisions. Interestingly, P and K were generally higher in the bottom of the fields as opposed to the top, indicating that erosion and deposition of plant nutrients may play an important role in the distribution of soil health parameters in furrow-irrigated systems. Ippolito et al. (2017) noted a similar response for K, with field bottoms containing greater plant-available K as compared to field tops.

The nutrient SHI is an average of SMAF P and K scores and is presented in Figure 3.1. Treatment was not a strong predictor of nutrient SHI, though year was strongly significant. As with all nutrient indicators, field section was not a significant predictor for nutrient SHI. This is unsurprising given that P is provided by equal fertilizer treatment, while K is provided by naturally K-rich soils. These observations are consistent with our hypothesis that soil nutrient status will not be impacted by tillage treatment.

3.1.5 Overall Soil Health Index

The overall SHI is an average of all SMAF scores (Figure 3.1). Neither year, treatment, nor field section had a significant impact on overall SMAF score. Caution should be employed when interpreting the overall SHI as a complete summary of soil health in the system; edaphic factors such as pH and K are given the same weight as SOC and $_b$, which may be much more sensitive to management and indicative of changes to soil health and function. However, the combined effects of improved physical, biological, and chemical properties resulted in slightly improved overall soil health in the MT and ST treatments. In 2022 specifically, averaging across field sections, MT plots and ST plots had significantly improved overall SHI ($p = 0.026$ and $p = 0.034$, respectively). This supports several publications that have shown overall improved soil health in conservation tillage systems (Afshar et al., 2022; Deleon et al., 2020; Lewandowski & Cates, 2023; Nunes et al., 2018; Trimarco et al., 2023).

3.2 POM-C and MAOM-C.

Although we found small differences in the bulk SOC across tillage treatments, we observed interesting results when looking at C distribution in the POM and MAOM fractions. There was a tendency of more POM-C in the conservation tillage treatments where POM-C was 39% higher in MT and 20% higher in ST than in CT, respectively. This corroborates a recent finding by Gajda et al. (2021) where POM-C under conservation tillage was 28% higher in the surface soil (0-5 cm) compared to conventional tillage. However, no differences in POM-C were observed between the two tillage treatments in some depth increments below 15 cm in their study. Future studies at the current field locations should collect and fractionate soils below 15 cm to better understand how residue mixture/translocation deeper in the soil profile from our CT treatment may affect POM-C at depth.

Differences in MAOM-C across tillage treatments were smaller than POM-C (Table 3.2), similar to the findings of Aduhene-Chinbuah et al. (2022) who found no differences in MAOM-C but did observe differences in the POM-C fraction when comparing no till to conventional tillage. We show that POM-C is more sensitive to tillage which is not surprising since it is less stable than MAOM-C in aerated mineral soils (Cotrufo & Lavallee, 2022) and disturbance increases aggregate turnover (Six et al., 1998), exposing POM-C once physically protected in soil aggregates to faster microbial decomposition and loss (Gupta & Germida, 1988). Given that POM-C is more representative of biologically active C and MAOM-C is more representative of soil C with longer turnover times and low bioavailability (Leuthold et al., 2022), the observation that tillage had a larger effect on POM-C suggests that intermediary conservation tillage systems may not improve soil C sequestration compared to conventionally tilled systems that receive similar row crop management. However, conservation tillage can provide improved organic matter availability for biological activity and enhance nutrient cycling, a notion supported by the measured outcomes of BG and PMN in our study (Table 3.1).

Table 3.2. POM-C and MAOM-C (Mean ± SE) for 2021 with ANOVA tests for all parameters and two-way interactions.

	Treatment			ANOVA p-value		
	CT	MT	ST	Treatment	Section	Treatment x Section
POM-C (g C kg^{-1} soil)	1.86 ± 0.14	2.59 ± 0.30	2.24 ± 0.15	0.057	0.64	0.081
MAOM-C (g C kg^{-1} soil)	13.87 ± 1.84	16.47 ± 1.49	15.75 ± 1.32	0.81	0.71	0.81

3.3 Yield and Surface Residue

Yield was generally lower in the CT plots, with the MT and ST plots having approximately the same yield (Table 3.3). The CT plot in block 2 regularly had the lowest yield, while the CT plot in block 1 was lower than any of the conservation till plots in 2021 and the highest yielding plot in 2022. Since one of the ecosystem services provided by healthy soils is improved plant growth and crop yield (Andrews et al., 2004), competitive yield in the conservation till plots may be seen as an indicator of overall improved soil health. Overall, the relatively small difference in yields to the benefit of conservation tillage systems indicates that the system has now fully matured and competitive yields can be achieved in conservation tillage systems. This finding was also observed by Afshar et al. (2022), where yield in no-till plots had 15% greater yield than conventionally tilled plots in a similar furrow-irrigated grain corn system.

Surface crop residue is reported in Table 3.3. Generally, it appears that the MT and ST plots were similar throughout all sampling points and only meaningfully different from the CT plots at the preseason sampling. For preseason sampling in 2021 and all sampling points in 2022, treatment had a significant effect on residue, with MT and ST plots broadly exceeding CT plots yet having similar values between each other. This dynamic is logical, as the post-harvest, and post-chop-and-bale measurements of crop residue are taken after harvest and before primary tillage, when residue is mixed into the soil. The preseason measurement of crop residue captures unincorporated crop residues as the crop season begins, and thus is reflective of the primary tillage operation impacts on surface coverage. At the preseason sampling time, residue mass was ~20-30 times greater in the MT and ST plots than in the CT plots in 2021, and ~5-10 times greater in 2022.

Table 3.3. Plant Matter Characteristics (Grain Yield and Surface Residue) for 2021 and 2022

Corn Grain Yields (Mean ± SE) by Plot (Treatment + Block)						
Treatment + Block	CT Block 1	CT Block 2	MT Block 1	MT Block 2	ST Block 1	ST Block 2
2021 Yield (t ha^{-1})	13.8	12.8	14.8	14.6	14.6	14.5
2022 Yield (t ha^{-1})	12.5	10.4	12.1	11.6	11.5	11.8
Crop Residue Mass (Mean ± SE) at Time of Sampling Across all Treatments with ANOVA tests for all parameters and two-way interactions						
	Treatment			ANOVA p-value		
Residue Mass (g m^2)	CT	MT	ST	Treatment	Section	Treatment x Section
2021						
Preseason*	2.3 ± 1.1	84.5 ± 16.5	88.5 ± 12.8	< 0.001	0.085	0.045
Post-Harvest	695.2 ± 77.9	807.3 ± 141.1	826.6 ± 129.5	0.62	0.46	0.11

Post-Chop and Bale	539.9 ± 137.0	607.3 ± 113.6	651.3 ± 72.4	0.90	0.54	0.53
2022						
Preseason	12.9 ± 5.3	105.5 ± 22.1	84.4 ± 18.9	0.0058	0.19	0.38
Post-Harvest	414.3 ± 23.9	463.7 ± 33.2	543.8 ± 46.2	0.0019	0.44	0.0023
Post-Chop and Bale*	299.9 ± 40.7	442.8 ± 25.6	392.8 ± 27.6	0.023	0.14	0.56

*Outcome variable log-transformed to meet ANOVA assumptions. Mean and SE are presented untransformed.

4 CONCLUSIONS

Results suggest that in semi-arid, furrow-irrigated systems, conservation tillage is a viable way to improve soil health while maintaining high yields. Moreover, as these conservation tillage plots mature and farmer expertise improves, techniques to manage residue and maintain irrigation efficiency have allowed managers on these plots to reduce tillage intensity without major concerns over irrigation management. In particular, it seems that the strip tillage (ST) reduced $_b$ over both the MT and CT plots, but the additional soil disturbance as compared to the MT plots had some detrimental effect to soil biology, though soil biological health was still improved in the ST plots compared to the CT plots. These biological soil health improvements may in part be due to the trend towards increased POM-C in the conservation tillage plots, providing increased organic matter available for microbial communities. Combined with the consistent yield between all treatments and the slightly lower variation in yield in the MT and ST plots compared to the CT plots, our findings indicate that MT and ST plots largely outperformed CT plots in all metrics where a difference between treatments was observable. Comparing MT plots to ST plots, it appeared as though MT plots slightly outperformed ST plots across most soil health metrics, but studies have indicated the potential for strip tillage operations to provide agronomic benefits (Nowatzki et al., 2017; Wardle et al., 2015), so the use of these lighter primary tillage operations may be seen as an intermediary in transitionary systems. Ultimately, the objective of reducing tillage intensity appears to have a positive effect on soil health, but the use of intermittent strip tillage or other similar tillage techniques may help farmers make this transition while still improving soil health over conventional tillage systems. This study highlights the need for future research on how varying levels of tillage intensity impact soil health, as broad advice to “till as little as possible” likely means a different management scheme for producers in humid, rain-fed systems than it does for those in semi-arid, furrow-irrigated systems. The effect of field section (top vs. bottom) and interaction of field section with tillage treatment warrants future research on spatial distribution and erosional impacts of soil health. For WSA, PMN, P, and K, it appears as though erosion and deposition played a key role in measurements of soil health across the field, and more studies may provide valuable insight into how furrow irrigation and conservation tillage interact to impact soil health. Wide-scale adoption of conservation tillage in furrow-irrigated agriculture requires significant future research and consideration of agronomic, economic, and practical factors that influence farmer decision-making.

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CHAPTER 4 – CONNECTING THE SOIL HEALTH – WATER QUALITY NEXUS UNDER SURFACE-IRRIGATED CONSERVATION TILLAGE

ABSTRACT

Intense tillage degrades soil health, worsens soil structure, and accelerates nutrient and sediment transport to vulnerable water bodies. Unfortunately, few studies have measured both soil health and water quality under conservation tillage, particularly in semi-arid furrow-irrigated fields, limiting our understanding of tillage impacts in these systems. To address this research gap, we investigated the impact of three tillage types (conventional [CT], strip [ST], and minimum [MT]) on soil health and water quality on large research plots. Specifically, we measured ten soil health indicators under the Soil Management Assessment Framework, soil organic C fractionations, and six edge-of-field runoff water quality indicators over two years. Edge-of-field water quality was improved in ST and MT plots as compared to CT, with reductions in mean concentrations for particulate constituents (sediment and total Kjeldahl N) approaching approximately 50% or greater. Additionally, indicators of physical soil health (bulk density, water stable aggregates) and biological soil health (microbial biomass C, soil organic C, potentially mineralizable N, and particulate organic matter C) correlated to decreased concentrations of water quality pollutants. This trend was most pronounced for particulate constituents (e.g., sediment, total Kjeldahl N), which were correlated to indicators of aggregate stability. Furthermore, the lack of difference

Modified for Publication in the Journal of Environmental Quality (DOI: 10.1002/jeq2.70013)

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between ST and MT plots with regards to water quality, soil health, and crop yield indicated that there was little difference between ST and MT. In semi-arid furrow-irrigated fields, we suggest utilizing minimizing tillage practices to reduce offsite sediment loss, nutrient transport and environmental degradation.

1 INTRODUCTION

Farmers utilize reduced tillage to increase profitability, improve soil function, support plant growth, sequester C, improve water filtration and storage, improve nutrient cycling and storage, and to reduce soil degradation. The relationship between soil health and water quality has been of growing interest, as agricultural land managers develop an understanding of how management practices impact nearby water bodies (Ward et al., 1994; Smith and Warnemuende-Pappas, 2015; Mubvumba and DeLaune, 2021; Lewandowski and Cates, 2023). Conservation tillage, a set of practices that reduce tillage intensity and leave at least 30% of crop residue on the soil surface (USDA NRCS, 2016), has been shown to improve soil health and to mitigate soil erosion (Conservation Technology Information Center, 2002; Benjamin, 2003; Deleon et al., 2020). Past studies have quantified either the differences in soil health or water quality from conservation tillage, but few have connected soil health alterations to water quality changes (Lewandowski and Cates, 2023). Moreover, of those studies that have connected soil and water quality, focus has been on rain-fed systems in temperate climates with little research in semi-arid climates (e.g., Smith et al., 2020; Fermanich et al., 2023; Fortuna et al., 2023).

There is a critical need to develop techniques and research in semi-arid conservation tillage, where soil is often heavily-tilled to maintain irrigation furrows (Afshar et al., 2022), a dynamic not present in more humid, rain-fed systems. The use of tillage to maintain irrigation furrows or mitigate surface compaction (Yuan et al., 2022) must be balanced with sustainability goals. Therefore, farmers in the semi-arid US west are examining methods to reduce tillage intensity, weighing agronomic and economic considerations with ecosystem health. It is therefore surprising that there are few studies of conservation tillage and water quality in irrigated systems (Lewandowski and Cates, 2023), a notable research gap as many soil health improvements associated with conservation tillage are related to hydrology and potential water quality improvements. Concomitantly, the stabilization of soil aggregates through conservation tillage can increase water infiltration and storage, increase soil organic matter content, improve nutrient cycling, and decrease risk to surface water quality (Dalal, 1989; Tan et al., 2002; Andrews et al., 2004; Bhaduri et al., 2014; Hedayatipoor and Alamooti, 2020; Afshar et. al, 2022). In semi-arid irrigated systems, water needs are met through irrigation. For furrow-irrigated systems, which account for ~30-40% of irrigated systems in the western US (USDA, 2019), runoff is a common byproduct of attempts to distribute water across the field. In these systems, soil health indicators that correlate to soil functions should focus on mechanisms of water storage, infiltration, and resistance to erosion (Lewandowski and Cates, 2023).

Soil health frameworks, such as the Soil Management Assessment Framework (SMAF; Andrews et al., 2004), may provide tools to connect soil health to water quality. The SMAF provides ten soil health indicators split into four categories: soil physical indicators (bulk density [ρ_b] and water-stable aggregates [WSA]); soil biological indicators (β-glucosidase activity [BG], microbial biomass C [MBC], soil organic carbon [SOC], and potentially mineralizable N [PMN]); soil chemical indicators (pH and electrical conductivity [EC]); and soil nutritional indicators (plant-available P and K). Indicators are selected to be sensitive to management and related to ecosystem services such as reduced erosion and improved water infiltration; reducing soil losses while upholding soil water relations across fields are key drivers in furrow-irrigated agroecosystems.

In furrow-irrigated systems, one of the largest challenges is maintaining irrigation uniformity. Surface residue can clog furrows, causing flooding in some areas and failing to deliver water to others, potentially impacting crop yield. However, Aarstad and Miller (1981) found that even small amounts of residue (~60 kg ha^{-1}) in irrigation furrows can significantly decrease erosion and called for research and strategies to manage residue in furrow-irrigated agriculture. To address this need, the Colorado State University Agricultural Water Quality Program developed specific management practices on an 11-year ongoing conservation tillage study in northern Colorado (Wardle et al., 2015; Deleon et al., 2020). This paper builds upon the work by Trimarco et al. (in review), which evaluated soil health (using the SMAF and C fractionation) as a response to conservation tillage techniques by comparing soil health and edge-of-field runoff water quality.

The primary objective of this research is to fill the major gap in soil health research that fails to empirically connect soil properties, water quality, and soil management. Moreover, this study was performed in critically understudied semi-arid conservation tillage systems, where water storage and management are extremely important elements of soil function. Based upon existing literature, the authors formed several hypotheses to describe the soil health - water quality relations of these long-term tillage plots:

d) Water quality will be improved with decreasing levels of tillage, particularly through reduced erosion and sediment transport, but observable in all quantified water quality constituents;
e) Physical soil health indicators will be correlated to sediment, total N (TN), and P runoff;
f) Biological soil health indicators will be correlated to N runoff;
g) Runoff total P (TP) concentrations will be correlated with sediment runoff and aggregate stability; and
h) Chemical and nutritional indicators of soil health (pH, EC, and K) will not be correlated to runoff water quality.

2 MATERIALS AND METHODS

2.1 Site Description

In 2011, a two-block split-plot field experiment was established to compare three tillage methods with potential for utilization in the Central High U.S. Great Plains region: CT, MT, and ST. Prior 2011, all plots were managed under the same regime: a moldboard plowed conventional tillage system. Field operations from 2021 and 2022 can be found in Supplementary Table S1 and crop rotation details from 2011 to 2022 can be found in Supplementary Table S2. The site was at Colorado State University's Agricultural Research Development and Education Center (40°40'40"N 104°59'51"W; elevation = 1567 m). The Garret series, a fine-loamy, mixed, mesic Pachic Argiustoll (Soil Survey Staff, 2023) was the dominant soil across the plots. Mean monthly temperatures ranged from 30° C in July to 5 °C in December, with average annual precipitation of ~270 mm (Colorado Climate Center, 2023).

This site was established with guidance from a farmer-led advisory committee to determine if reduced tillage was agronomically and economically feasible within these systems. Tillage treatments were replicated within this randomized complete two-block split-plot design. Plots were ~320 m long and ~28 m wide, representing production-scale fields to evaluate real-world challenges faced in furrow-irrigated agriculture. As a production-scale field, treatments were adapted annually based on soil conditions, crop rotation and weather events. During the years considered for this paper, plots were managed under one of three tillage treatments summarized subsequently as:

a) CT plots - vertical tilled and moldboard plowed, alongside mulching, landplaning, and mild row cleaning using a ditcher and sweeper;
b) MT plots - vertical tilled with mild row cleaning operations using a ditcher and sweeper;
c) ST plots - same as the minimum till plots but with an additional strip till operation along the seedbed prior to planting. The full details of management practices can be found in Supplementary Table S1.

Row cleaning and cultivation was employed in all plots to facilitate irrigation water management. Fertilizer additions were based on Colorado State University recommendations from composite spring soil tests sent to American Agricultural Laboratory, Inc. (McCook, Nebraska, USA). In 2021, all plots received equivalent N inputs (~180 kg ha^{-1} N), the CT and MT plots received ~34 kg P ha^{-1} while the ST plots received 25 kg P ha^{-1}. In 2021, the CT plots received all fertilizer as a mixed DAP-urea broadcast in early April. The MT plots received an early April DAP-urea broadcasting accounting for half of the N and all the season’s P, with a mid-June banding UAN application providing the remaining N needs. The ST plots received an early April ammonium phosphate-urea ripper-shanked banding application at a depth of ~15-20 cm for 38% of the N and all P (with supplemental Black Label Zn and Reax K) and the rest of N needs in a mid-June UAN banding application. In 2022, all plots received equivalent fertilizer inputs: a banded Optistart Gold pre-plant application of ~2 kg N ha^{-1} and ~6 kg P ha^{-1} in late April, followed by an early June Maximum N-Pact foliar application of ~3 kg N ha^{-1} and a banded UAN application of ~71 kg N ha^{-1} a week later. Irrigation water sourced from a groundwater well (~27.4 m depth) was delivered from a concrete lateral ditch via 3.8 cm aluminum siphon tubes to every other furrow. For 2021 and 2022, the field was planted with a flexible dent corn variety (DeKalb DKC 47-54RIB or Pioneer P9489Q, respectively) at 88,950 seeds ha^{-1} on 76.2 cm row spacing.

2.2 Sampling and Analysis

2.2.1 Soil Sampling and Analysis

Full details of soil sampling and soil health analysis are described in Trimarco et al. (In review). In brief, composite soil samples were taken to a depth of 15 cm in every plot using a 2.5 cm diameter soil probe on the shoulder of a seedbed. Each composite soil sample, from a 3 m radius centered around a GPS-located point, was composed of ~30 cores. All soil samples were placed into a plastic bucket, thoroughly mixed, and transferred to a plastic bag which was then sealed and placed in a cooler with ice packs. Additionally, a Madera probe (Evett et al., 2022; Precision Machine Co., Lincoln, Nebraska, USA) was used to take an intact ~61 cm^3 bulk density (ρ_b) core to a depth of 15 cm on the shoulder of the sampled furrow. In 2021, two composite samples were taken in each plot: each in row 15 of 36 (moving west to east), one ~30 m upslope of the bottom of the field and another ~30 m downslope of the top of the field. In 2022, this was repeated, but the addition of the same sample setup was also obtained in row 19 to capture four samples per plot and to have a sample taken within the row (i.e., row 19) sampled for outflow water quality.

Soil samples were immediately returned to the laboratory and processed following the SMAF (Andrews et al., 2004) with detailed protocols described in Trimarco et al. (In review). In brief, ρ_b was determined by drying an intact Madera probe core at 105 °C until completely dry. Water stable aggregates were determined with a Yöder sieving machine according to Kemper and Rosenau (1986). β-glucosidase activity was determined according to Green et al. (2007). Microbial biomass C was determined via a 4-day chloroform fumigation method (Hobbie, 1998; Beck et al., 1997). Soil organic C was determined by measuring total C on a dry combustion VELP Dumas Elemental Analyzer (VELP Scientifica, Usmate Velate, Italy), and subtracting inorganic C (Nelson and Sommers, 1996), as determined by the pressure transducer method (Sherrod et al., 2002). Potentially mineralizable N was quantified as the difference of the sum of 2M KCl extractable NO_3-N and NH_4-N following a 28-day incubation minus NO_3-N and NH_4-N concentrations from non-incubated soil (Mulvaney, 1996; Curtin and McCallum, 2004), as measured on a Genesys 10S UV-VIS spectrophotometer at 540 nm and 610 nm, respectively. Soil pH (Thomas, 1996) and EC (Rhoades, 1996) were determined in a 1:1 soil solution following a 2 h shaking period and, for EC, an additional centrifuging operation. Plant-available P and K were determined via Olsen extraction (Olsen et al., 1954) and measured on an inductively coupled plasma-optical emission spectrophotometer (ICP-OES) at 213.617 nm and 766.490 nm, respectively. In addition to the soil health indicators used in the SMAF, soils sampled in 2021 were processed for C fractionation according to Cotrufo et al. (2019). Briefly, soils were dispersed with sodium hexametaphosphate and fractionated according to size through a 53 μm sieve, and C was measured on a VELP Dumas Elemental Analyzer to determine mineral associated organic matter-C (MAOM-C) and particulate organic matter-C (POM-C). Time and labor constraints prevented C fractionation in 2022.

2.2.2 Water Sampling and Analysis

Irrigation was provided by siphon tubes from the concrete ditch supplying well water at the north end of the field, with water transmitted to the south end through furrows. Irrigation water was applied for approximately 12 hr per event. Irrigations occurred approximately every 14 to 20 days, based on crop need. Irrigation events were sampled on June 23rd, July 9th, July 30th, August 12th, August 25th, and September 11th of 2021, and June 30th, July 13th, July 26th, and August 25th of 2022 (the final irrigation occurred on August 25th in block 1 and August 29th on block 2 due to water management needs). This constituted 6 sampled events in 2021 and 4 events in 2022. For each irrigation, block 2 was irrigated ~1-3 days after block 1 was irrigated due to water limitations. An irrigation event on August 10th of 2022 was not sampled due to furrow management challenges after a heavy hail event resulted in accumulation of corn leaves in the irrigation furrows.

In 2021, after planting and prior to the first irrigation, 7" trapezoidal flumes were installed 10 m from the bottom of the field in row 15 of each plot. Berms were established 5 m up-field from the flume to combine flow from two irrigated furrows (15 and 17) to ensure that flow was adequate for sampling and to increase sampling area in each plot. Samples were collected by hand at the outfall of each flume ensuring for the capture of moving water rather than standing water. HDPE plastic bottles (1 L) were filled completely at predetermined intervals (described below) for each of six irrigation events in 2021. Every grab sample was duplicated in the field as a second sample taken immediately after the first and for all analysis involving water analytes, duplicate samples were averaged together. Samples were taken at first flush (at initiation of runoff), 1 hr after first flush, and at the end of the irrigation (approximately 12 hr after irrigation start or 10 hr after runoff initiation). The three water samples were combined by vigorously shaking all 1 L bottles and then pouring them into a 4 L composite bottle.

In 2022, the same flumes were installed 10 m from the bottom of the field in row 17 of each plot shortly after planting and before the first irrigation. In 2022, five total irrigation runoff events were captured, though one of these was removed from analysis due to irrigation management challenges. As in 2021, berms were established 5 m up-field from the flume to combine flow from rows 17 and 19. However, instead of hand sampling, water samples were obtained with low-cost automated water samplers developed by the Agricultural Water Quality Program at Colorado State University (Brown et al., 2023). Every hour, beginning when flow height in the flume exceeded 1.27 cm, a 200 mL sample was collected and composited into two 2 L plastic bottles, with one serving as the duplicate. Identical to 2021, the composite water samples were analyzed for the same constituents using identical methodologies.

All composite samples were vigorously shaken and split into smaller 250 mL bottles, then delivered to ALS Laboratories in Fort Collins, Colorado for nitrate-N (NO_3-N; EPA Method 300.0 Revision 2.1), total Kjeldahl N (TKN; Standard Method 4500-NH_3 G11), ortho-phosphate P (Ortho-P; EPA Method 300.0 Revision 2.1), TP (EPA Method 365.2), and selenium (Se; EPA

Method 200.8) analyses. The decision to analyze these constituents was based on the most common nutrients of concern in runoff water (N and P), as well as Se and total suspended solids (TSS), which have been identified as constituents of concern in Colorado's South Platte River Basin (Colorado Department of Public Health & Environment, 2023), the basin into which return flow irrigation water from this site drains. At the start and end of each irrigation event, an inflow sample was obtained from the supplying irrigation ditch and analyzed for the same constituents as listed above. Inflow concentrations can be found in Supplementary Table S3.

Furthermore, one split bottle was retained for TSS concentration, determined in-house (EPA Method 160.2). Briefly, a DI-rinsed, oven-dried at 105 °C, and pre-weighed glass microfiber filter was secured to a Buchner funnel. Samples were vigorously shaken and poured into a 500 mL beaker and stirred continuously with a magnetic stir bar. Three successive 10 mL volumes of the sample were transferred to the filter paper and then vacuum filtered. Then, the filter paper was removed, oven-dried at 105 °C, and weighed to determine the mass of residue left on the filter.

2.2.3 Statistical Analysis

Concentrations of all constituents were evaluated in a linear mixed model with tillage treatment and sampling year as fixed factorial predictors, irrigation event as a fixed ordinal predictor, and block as a random factorial predictor. Analysis of variance (ANOVA) was performed with the Kenward-Rogers degree of freedom method to limit homoscedasticity concerns (Kenward & Roger, 1997), log-transformations were performed as necessary to meet ANOVA model assumptions of residual normality and homoscedasticity, and a Tukey adjustment

(Tukey, 1949) was applied for multiple comparisons when the ANOVA was significant. Statistical significance was evaluated at an $\alpha \leq 0.05$.

To assess the correlation of soil health to water quality, the mean of each soil health indicator for each plot was averaged to the plot level and assigned as the soil health status all year. For example, for all of 2021, the average of ρ_b taken during soil sampling in the CT plot in block 1 was assumed to be ρ_b in this plot whenever a water quality sample was generated. This method assumes that the soil characteristics at the time of sampling are the same as that during water sampling, which was necessary because repeated soil health sampling at every irrigation event was too laborious and expensive. Linear mixed effect models were created with the soil health indicator as a fixed continuous predictor, year as a fixed factorial predictor, irrigation event as a fixed ordinal factor, and block as a random factorial predictor. The same process was repeated using POM-C and MAOM-C as soil health indicators, but these models excluded year as they were only performed for 2021 samples. Outcome variables were log-transformed as needed to meet model assumptions of normality and homoscedasticity and ANOVA was performed with the Kenward-Rogers degree of freedom method to limit homoscedasticity concerns (Kenward & Roger, 1997). Statistical significance was evaluated at an $\alpha \leq 0.05$. Ortho-P was not considered in any analysis as it was detected in only two events. Event 6 in 2021 was removed from NO_3-N and TN analysis due to expired holding time for NO_3-N samples resulting in outliers.

It is worth noting the gap between edge-of-field measurements and instream water quality impairments and to limit our inference of downstream water quality from this study. Even when runoff load or concentration is high, the contribution to river water quality may be relatively small. However, the aggregation of many fields may significantly threaten river water quality, particularly when rivers have low flow and agriculture composes a large portion of the landscape. In many systems, subsurface lateral flow may significantly impact river water quality, but this mechanism was not measured in this study. However, given the lack of precipitation and fine soil texture in this system, deep percolation and lateral flow of nutrients is likely slow and not a major contributor to river water quality. Further research should attempt to connect landscape scale conservation practices with changes in river water quality by studying the multiple pathways by which contamination reaches receiving water bodies. Findings in this study should be interpreted in the context of how soil and water interact at the soil-water interface rather than the quantity of constituents of concern being transmitted to waterbodies.

3 RESULTS AND DISCUSSION

3.1 Water Quality

Runoff water quality, evaluated as the concentration of five key water constituents (TSS, Se, TKN, NO_3-N, TN, TP), is presented in Figure 4.1 as boxplots split by treatment. Limitations in the quality of flow measurements led us to only consider constituent concentrations, rather than loads. A decrease in concentration should logically lead to decreased loads if flow in furrows is generally consistent between plots. Deleon (2017) observed decreased outflow in the ST and MT plots as compared to the CT plots in a prior study of the same plots in 2015-2016, whereas Driscoll (2013) observed (insignificantly) decreased outflow in the CT plots compared to the MT or ST plots in 2011-2012. While samples only were taken when outflow water depth in the flume exceed 1.27 cm, which may limit the effect of very low flow resulting in very high constituent concentration, findings about concentration should be interpreted in the context of how soil and water interact at the soil-water interface rather than the total quantity of constituents of concern being transmitted to waterbodies.

For no water quality analytes were ST and MT significantly different (TSS: $p = 0.71$; Se: $p = 0.99$; TKN: $p = 0.78$; TN: $p = 0.43$; NO_3-N, and TP not evaluated as ANOVA $p > 0.05$). However, when differences were present, CT increased runoff constituent concentrations as compared to conservation tillage practices. The CT plots had a greater TSS concentration than both the MT and ST plots (CT v. MT: $p = 0.0023$; CT v. ST: $p < 0.001$), with the same pattern observed for TKN and TN (TKN: CT v. MT: $p = 0.011$, CT v. ST: $p = 0.0018$; TN: CT v. MT: $p = 0.045$ and CT v. ST: $p = 0.0015$). For Se, CT was greater than ST but not MT (CT v. MT: $p = 0.057$ and CT v. ST: $p = 0.041$). For NO_3-N and TP, no differences existed between treatments (NO_3-N: $p = 0.42$ and TP: $p = 0.27$). For all analytes, irrigation event was a significant predictor of analyte concentration.

This supports our hypothesis that conservation tillage leads to decreased constituent runoff, though it is interesting to observe that MT, the most conservative tillage method, was not significantly different than ST. Despite these benefits, conservation tillage in irrigated agriculture is not without costs, namely crop residue management. Strip tillage attempts to bridge the gap between no-till and conventional tillage. Nowatzki et al. (2017) reported that strip till methods paired with row cleaning allow for precise management of residue to optimize soil temperature, moisture retention, and surface coverage to maintain competitive yields with conventionally tilled fields while reducing soil disturbance. The additional strip till operation in the ST plots compared to the MT plots did not have a negative effect on water quality.

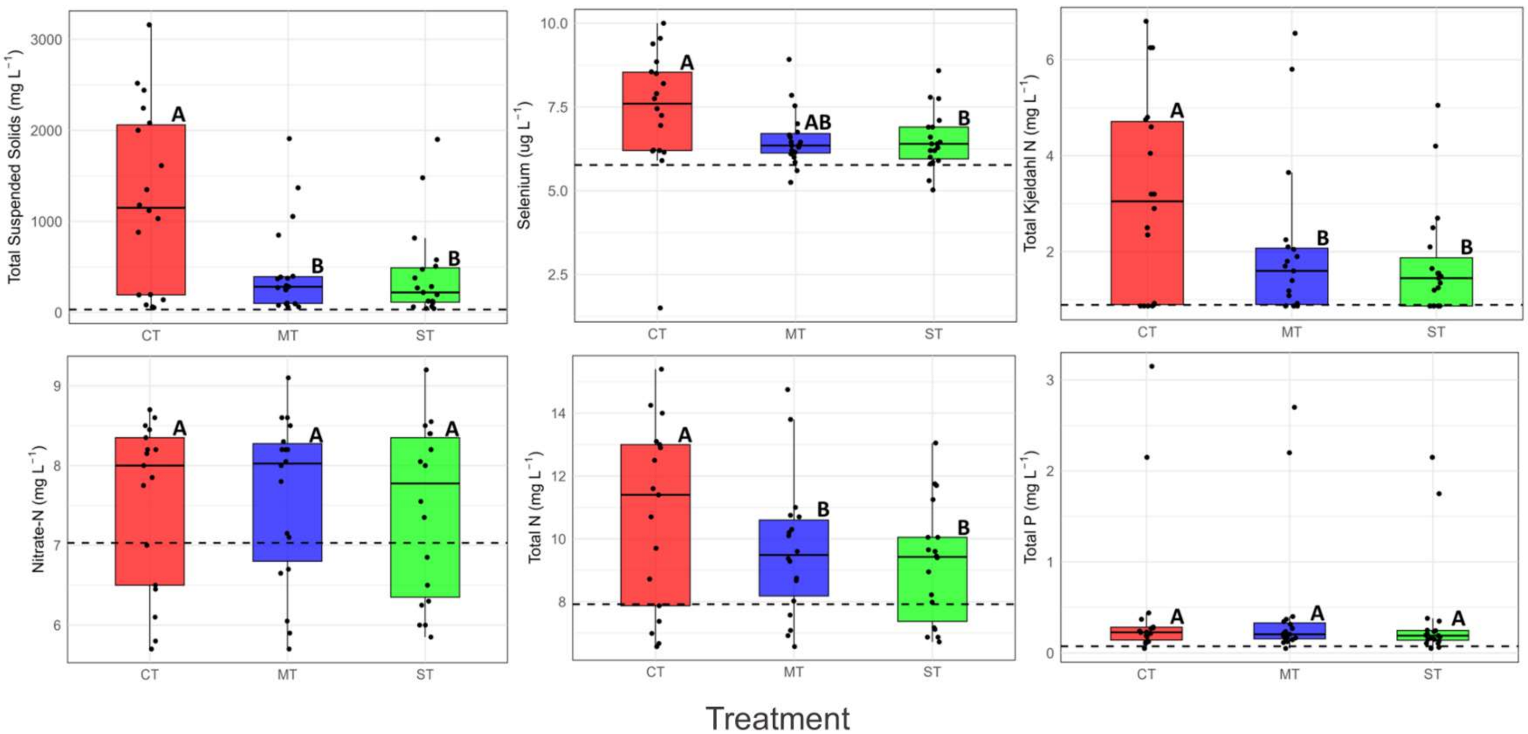

Figure 4.1. Boxplots of each water quality analyte by treatment (CT=conventional tillage; MT=minimum tillage; ST=strip tillage). Significantly different treatment means are indicated by different letters. Models were log-transformed to meet assumptions of normality and homoscedasticity as necessary for TSS, TKN, and TP. For all analytes, irrigation event was a significant predictor ($p < 0.05$). For context, dashed lines are shown at the mean for inflow water quality constituent concentrations for each analyte across year and irrigation event.

Thus, producers in furrow-irrigated systems may be able to use strip tillage to loosen the seed bed and ensure proper emergence (Nowatski et al., 2017) without excessive risk of water quality degradation. These results are similar to those of Benham et al. (2007) who showed that both strip till and no-till reduced nutrient loading in surface water runoff compared to conventionally tilled fields but were not significantly different from each other. In contrast, other studies have observed differences in erosion under rainfall or rainfall simulations between no-till and reduced-till (analogous to our MT treatment; van Vlietl et al., 1993; Smith and Warnemuende-Pappes, 2015). This lack of consistency between studies comparing reduced-till and no-till warrants further research on the role that marginal tillage practices have on water quality and highlights the importance of site-specific evaluation.

3.2 Soil Health – Water Quality Connections

The results of mixed models for soil health and water quality connections are presented in Table 4.1. The concentration of runoff TSS was positively correlated to ρ_b, EC, and soil P, and negatively correlated to WSA, MBC, SOC, PMN, pH and POM-C. In this system, when ρ_b and EC are relatively high, and when MBC, SOC, PMN, POM-C, and WSA are relatively low, greater TSS in runoff may occur. Higher ρ_b and EC and lower MBC, SOC, PMN, POM-C, and WSA indicate less healthy soils. Increased SOC has been correlated to aggregate stability in several studies (e.g., Rahimi et al., 2000; Iheshiulo et al., 2024), and SOC is known to increase under conservation tillage compared to conventional tillage (Six et al., 1999; 2000; Luo et al., 2010). Six et al. (2000) suggested macroaggregate turnover is decreased in no-till sites compared to conventionally tilled sites, and that a portion of POM is protected within macroaggregates.

Aggregate stability increases infiltration and decreases sediment runoff (Barthès and Roose, 2002; Nimmo, 2004), a notion supported by the relationship between TSS and both WSA and POM-C. Lavallee et al. (2020) noted that POM is more sensitive to tillage disruption of macroaggregates than MAOM. . In a companion study, Trimarco et al. (In review) observed that SOC was ~10-20% greater in the MT and ST plots. Interestingly, the increase of MAOM-C in the MT and ST plots (2.6 and 1.9 g C kg^{-1} soil, respectively) was greater in magnitude than the increase of POM-C (0.7 and 0.4 g C kg^{-1} soil, respectively). However, because POM-C levels were lower in the system, the percentage change of POM-C exceeded that of MAOM-C (POM-C increased by 39% in the MT plots and 20% in the ST plots, whereas MAOM-C only increased by 19% and 14%, respectively). This is perhaps why Trimarco et al. (In review) observed a near-significant effect of tillage on POM-C but not MAOM-C on the same dataset used in this study, and that while POM-C was negatively correlated to TSS, TKN, and TN, MAOM-C was not correlated to any water quality constituent. Water stable aggregation was about ~55% greater in MT and ~28% greater in ST compared to CT in 2021 (with similar trends in 2022; Trimarco et al., In review), but this difference was not statistically significant, demonstrating the high level of variability. We observed both an effect of tillage on TSS and a negative correlation between TSS and WSA, indicating that variability in WSA limited our capacity to detect a significant difference. Soils with more stable macroaggregates ought to have increased WSA and POM-C, and decreased TSS runoff. Together, these relationships indicate that the reduced tillage intensity in the conservation tillage plots enhanced macroaggregate stabilization, which in turn supports POM-C stabilization, microbial community activity (as proxied by MBC, SOC, and PMN) and decreased sediment transport, even if WSA was not affected by treatment (Lavallee et al., 2020; Trimarco et al., In review). Unfortunately, the WSA method used in this study does not

Table 4.1. Mixed model outcomes for soil health indicators and water quality constituents.

	Water Quality Analytes											
	TSS*		Se		TKN		NO_3-N		TN		TP*	
Soil Health Indicators	Indicator	Year	Indicator	Year	Indicator	Year	Indicator	Year	Indicator	Year	Indicator	Year
b	**+, p = 0.001, R^2 = 0.66**	0.12	+, p = 0.052, R^2 = 0.31	0.88	**+, p = 0.007, R^2 = 0.64**	**0.009**	+, p = 0.80, R^2 = 0.84	**< 0.001**	**+, p = 0.009, R^2 = 0.80***	**< 0.001**	+, p = 0.56, R^2 = 0.64	**0.0030**
WSA	**-, p = 0.031, R^2 = 0.56**	**< 0.001**	-, p = 0.95, R^2 = 0.24	0.10	-, p = 0.13, R^2 = 0.58	**< 0.001**	-, p = 0.95, R^2 = 0.84	**< 0.001**	-, p = 0.30, R^2 = 0.76*	**< 0.001**	+, p = 0.64, R^2 = 0.64	**< 0.001**
BG	+, p = 0.95, R^2 = 0.55	**< 0.001**	-, p = 0.18, R^2 = 0.26	0.64	+, p = 0.86, R^2 = 0.57	**< 0.001**	+, p = 0.30, R^2 = 0.84	**< 0.001**	+, p = 0.68, R^2 = 0.76*	**< 0.001**	+, p = 0.95, R^2 = 0.64	**< 0.001**
MBC	**-, p = 0.043, R^2 = 0.57**	**< 0.001**	-, p = 0.20, R^2 = 0.25	0.070	-, p = 0.102, R^2 = 0.59	**< 0.001**	+, p = 0.61, R^2 = 0.84	**< 0.001**	-, p = 0.2, R^2 = 0.77*	**< 0.001**	+, p = 0.96, R^2 = 0.64	**< 0.001**
SOC	**-, p = 0.047. R^2 = 0.57**	**< 0.001**	+, p = 0.86, R^2 = 0.24	0.20	-, p = 0.17, R^2 = 0.59	**< 0.001**	-, p = 0.99, R^2 = 0.84	**< 0.001**	-, p = 0.33, R^2 = 0.77*	**< 0.001**	+, p = 0.65, R^2 = 0.64	**< 0.001**
PMN	**-, p = 0.041, R^2 = 0.62**	**< 0.001**	-, p = 0.23, R^2 = 0.29	0.20	-, p = 0.099, R^2 = 0.61	**< 0.001**	-, p = 0.79, R^2 = 0.84	**< 0.001**	-, p = 0.10, R^2 = 0.79*	**< 0.001**	-, p = 0.98, R^2 = 0.64	**< 0.001**
pH	**-, p < 0.001, R^2 = 0.76**	**0.003**	**-, p = 0.028, R^2 = 0.30**	0.27	**-, p < 0.001, R^2 = 0.72**	0.29	-, p = 0.97, R^2 = 0.84	**< 0.001**	**-, p < 0.001, R^2 = 0.83**	**0.38**	-, p = 0.53, R^2 = 0.64	0.062
EC	**+, p < 0.001, R^2 = 0.69**	**< 0.001**	**+, p = 0.004, R^2 = 0.38**	**0.001**	**+, p = 0.002, R^2 = 0.66**	**< 0.001**	+, p = 0.84, R^2 = 0.84	**< 0.001**	**+, p = 0.011, R^2 = 0.80***	**< 0.001**	+, p = 0.60, R^2 = 0.64	**0.011**
P	**+, p = 0.001, R^2 = 0.57**	0.62	+, p = 0.64, R^2 = 0.24	0.56	**+, p = 0.007, R^2 = 0.60**	0.12	+, p = 0.53, R^2 = 0.84	**< 0.001**	+, p = 0.002, R^2 = 0.76	**0.006**	-, p = 0.76, R^2 = 0.64	**0.007**
K	+, p = 0.45, R^2 = 0.55	**< 0.001**	+, p = 0.30, R^2 = 0.25	0.18	+, p = 0.53, R^2 = 0.58	**< 0.001**	+, p = 0.13, R^2 = 0.85	**< 0.001**	+, p = 0.15, R^2 = 0.77*	**< 0.001**	+, p = 0.96, R^2 = 0.64	**< 0.001**
MAOM-C	-, p = 0.31, R^2 = 0.25	-	+, p = 0.62, R^2 = 0.22	-	-, p = 0.34, R^2 = 0.40	-	-, p = 0.57 R^2 = 0.38	-	-, p = 0.31, R^2 = 0.38	-	+, p = 0.91, R^2 = 0.54	-
POM-C	**-, p < 0.001, R^2 = 0.40**	-	-, p = 0.088, R^2 = 0.22	-	**-, p < 0.001, R^2 = 0.47***	-	+, p = 0.56, R^2 = 0.39	-	**-, p = 0.003, R^2 = 0.48***	-	-, p = 0.80, R^2 = 0.54	-

The directionality of relationship between soil health indicators and water quality concentration (+ or -), p-value of the ANOVA test of SHI, and Marginal R^2 for the total fit are presented in the Indicator column (n = 56, except for NO_3 and TN models, where n = 53. Kenward Rogers degrees of freedom were approximated as ~52 for all models except NO_3 and TN models, which were approximated as ~49). The p-value for the ANOVA test of sampling year is presented in the Year column. ANOVA p-values for Event are not presented here, as every single model found this to be significant. TSS and TP were log-transformed in all models using these as outcome variables, and TN was transformed for all models except pH and extractable soil P. Significant effects are bolded. b = bulk density; WSA = water stable aggregates; BG = β-glucosidase activity; MBC = microbial biomass carbon; SOC = soil organic carbon; PMN = potentially mineralizable nitrogen; EC = electrical conductivity; P = plant-available P; K = plant-available potassium; POM-C = Particulate organic matter C; MAOM-C = Mineral associated organic matter C; TSS = Total suspended solids; Se = Selenium; TKN = Total Kjeldahl N; NO_3-N = Nitrate-N.

fractionate aggregates by size; a more precise method may be warranted to study the relationship between aggregate size and sediment transport. Sampling year was a significant predictor in almost all TSS models, excluding ρ_b and P. This is indicative that either sampling methodology or year-to-year variations in field conditions had a strong impact on TSS concentration.

Runoff water Se concentrations were correlated with two soil health indicators: negatively with pH and positively with EC, though there was a trend towards a positive correlation with ρ_b. Mean Se concentration in CT (7.4 ± standard error 0.5 μg L^{-1}) exceeded that of MT or ST (6.5 ± 0.2 μg L^{-1} for both), and irrigation inflow water was closer to conservation tillage treatments (5.8 ± 0.55 μg L^{-1}), indicating that the majority of Se came from the irrigation water itself. While Se can occur in dissolved and particulate forms, it tends to be in the dissolved forms of selenite (SeO_3^{2-}) and selenate (SeO_4^{2-}) in water (Paikaray, 2016). Soils with lower pH tended to have higher runoff Se concentrations, contrary to studies that have reported lower pH facilitating increased organic matter binding and decreased Se mobility (Paikaray, 2016). However, all pH values fell between 7.9 and 8.4, a range unlikely to have a meaningful impact on sorption-dissolution dynamics. While NO_3-N, a dissolved constituent, was not impacted by treatment, Se was significantly lower in ST than CT. This is surprising, given that the primary forms of Se in water are the dissolved anions selenite/selenate. However, there may be fewer anion exchange sites in the CT treatments as a function of the (insignificantly) decreased POM-C and MAOM-C in the CT plots compared to the ST or MT plots (Sparks, 2003; Trimarco et al., In review). Decreased anion exchange capacity could result in decreased sorption of selenite/selenate to organic matter (Sparks, 2003), resulting in Se enrichment in runoff in the CT plots. Additionally, increased anion concentration in solution has been shown to increase selenite/selenate mobilization (Kumar and Riyazuddin, 2011). This may explain the positive correlation between EC (a measure of dissolved ions) and Se runoff. Even in this system, where EC was too low to impact crop growth (~ 0.6 – 1.3 ds m^{-1}; Waskom et al., 2012), it may have played a role in water quality: more saline soils with less anion exchange sites had increased Se mobilization and runoff. However, without Se load data or speciation, it is not possible to compare the contribution from irrigation water and from the soil or to determine the mechanism by which ST reduced Se runoff compared to CT. All plots and irrigation inflow (Supplementary Table S3) regularly exceeded the South Platte River Basin warm river water quality standard of 4.6 µg L^{-1} (CDPHE, 2024), indicating that in the South Platte River Basin, conservation tillage is unlikely to strongly impact river Se. Still, further research should focus on understanding the impact of conservation tillage on Se and other less studied water quality constituents.

Both dissolved and particulate N runoff were quantified in this study. Total Kjeldahl N is the largely particulate form of N composed of organic N and ammonium/ammonia (U.S. EPA, 1993), while NO_3-N is a dissolved form of N, and TN is the sum of TKN and NO_3-N. Total Kjeldahl N concentration had a positive correlation with ρ_b, EC, and extractable P, a negative correlation with pH and POM-C, and negative correlations trending towards significance with PMN and MBC. NO_3-N did not have a significant correlation with any soil health indicator but was correlated to sampling year in every model, which is likely indicative of the annual changes to irrigation inflow NO_3-N (~8 mg L^{-1} in 2021 and ~6 mg L^{-1} in 2022, see Supplementary Table S3). This dynamic prompted further investigation, and a linear mixed model for NO_3-N with tillage treatment (including inflow samples as a "tillage treatment") and sampling year as fixed effects and block as random effect found no significant difference between any tillage treatment and inflow NO_3-N, indicating that tillage and soil health did not impact NO_3-N runoff. Total N followed a similar pattern as TKN, indicating that the impact of tillage on TN runoff is determined by TKN runoff. The correlation of TKN with POM-C is notable as POM-C accumulation is associated with macroaggregate stability (Lavallee et al., 2020). Soils that protect their POM in macroaggregates see less organic N in POM runoff as TKN and more organic N remains in the soil as PMN. We would expect to see the same dynamic with MAOM-C as MAOM-C is also protected via occlusion in microaggregates. However, with no difference in MAOM-C across the treatments, it seems plausible that MAOM-C is saturated in this system (Cotrufo et al., 2019; Lavallee et al., 2020). Soils that have more macroaggregate protection logically lose less sediment (evidenced by the TSS-WSA relationship), and this sediment is likely rich in NH_4^+ (a component of TKN) bound to soil surfaces (Sparks, 2003). While the relationship between WSA and TKN was not significant, a method of WSA evaluation that fractionates aggregates by size may elucidate the relationship between POM stored in macroaggregates, increased macroaggregate turnover in intensely tilled systems, and runoff of organic N. Together, these relationships indicate that N runoff is largely mediated by soil health characteristics that support protection of organic N and NH_4^+-rich sediment.

Total P was not correlated with any soil health indicator, and all differences in TP seemed to be driven by year or irrigation event. This contradicts the hypothesis that TP runoff is correlated to WSA. However, investigation of correlations between TSS and TP using a linear model found that TP was highly correlated to TSS ($p < 0.001$, $R^2 = 0.48$, TP and TSS were log-transformed to meet model assumptions). Positive correlations between TSS and TP have been documented thoroughly, and studies have indicated that particulate P dominates runoff P, particularly in furrow-irrigated systems (Turner et al., 2004a; Turner et al., 2004b; Bjorneberg et al., 2006), a dynamic driven by tight associations between soil particles and P (Sparks, 2003; Jalali and Kolachchi, 2009). Alongside the lack of ortho-P detection, this indicates that TP runoff is almost entirely driven by sediment-bound P in this system. Multiple studies from Turner and colleagues (Turner et al., 2004a; Turner et al., 2004b) highlighted correlations between Olsen-extractable P and dissolved reactive P (indicated by ortho-P in this study), but these studies reported Olsen P measurements ~10x as great as those observed in our study. The low concentration in this study may explain why Olsen P was not correlated to runoff TP: depletion of Olsen P reduced ortho-P runoff to below detection limit. Given the large number of studies that have documented decreased P runoff in conservation tillage systems (e.g., Benham et al., 2007; those cited within Osmond et al., 2019; Mubvumba et al., 2023) and that TSS was significantly impacted by tillage treatment, the lack of significance between tillage practice and TP concentration is surprising. This may indicate that conservation tillage has a positive impact on erosional loss of sediment, but this impact is dulled with regards to P loss. Perhaps the addition of pre-season fertilizer P on all treatments limits the potential for conservation tillage to reduce P runoff, though we expected reduced erosion to reduce TP runoff. The potential for conservation tillage to reduce runoff P losses should continue to be studied, particularly with methods that have low detection limits for ortho-P to determine the role of partitioning in P runoff.

From a soil health perspective, it is worth noting the positive correlation between ρ_b and TSS, TKN, and TN runoff (with a trend towards significance for Se). This observation contrasts with studies that have reported increased ρ_b or decreased sediment and nutrient transport in systems with decreased tillage (Stein et al., 1986; van Vliet et al, 1993; Benham et al., 2007; Smith and Warnemunede-Pappas, 2015; Nunes et at., 2018; Afshar et al., 2022; Yuan et al., 2022). However, most of these studies measured either ρ_b or water quality, not both, and compared no-till and moldboard-plowed systems without comparison of intermediate tillage.

Trimarco et al. (In review) found that ST had decreased ρ_b compared to CT, with MT falling in between the two. Castellini et al. (2020) observed that minimum tillage resulted in decreased ρ_b and increased infiltration compared to no-tillage, indicating that tillage effects on ρ_b are site-specific and depend on soil texture or management. As a measure of compaction, ρ_b represents not only pore space but also the pore continuity that plays an important role in infiltration-runoff relationship, with more compacted soils having decreased infiltration (Bagnall et al., 2022; Basset et al., 2023) and consequently more runoff. Furthermore, Özdemir et al. (2022) documented that more compacted soils have less stable aggregates. Lavallee et al. (2020) proposed that aggregate stability is key to protection of POM-C. These together indicate that intensive tillage compacts the soil even at shallow depths (0 – 15 cm), breaks up aggregates, and releases the sediment and organic N within these aggregates to be eroded, all of which is supported by the observations in this study.

In this study, EC was significantly correlated to TSS, Se, TKN, and TN runoff concentrations possibly because less compacted soils in conservation tillage plots allow infiltration to leach salt from the root zone and produces less sediment and nutrient-rich runoff. Aarstad and Miller (1981) documented that in conservation tillage systems increased surface residue slows water flow and encourages infiltration. Increased residue in the ST and MT plots may also contribute to the negative relationship between biological soil health and water quality constituent concentrations. Soil pH was correlated to TSS, Se, TKN, and TN. However, the range of pH values was very small (7.9 – 8.4), so this may be more indicative of the tillage treatment effect on these analytes and the observations from Trimarco et al. (In review) that the MT and ST treatments had a slightly higher pH (< 0.15 mean difference) rather than any mechanistic effect of pH on water quality constituents.

Several related factors contributed to improved soil health and water quality in the ST and MT plots relative to the CT plots, which we attribute to less intense tillage leaving more crop residue on the soil surface and limiting destruction of soil structure. Reduced tillage decreased compaction (ρ_b) and increased aggregate stability (WSA), which may have increased water infiltration and leached salts from the root zone (EC). Within the stable aggregates, POM was protected and resulted in less organic N, sediment, and sediment-bound ammonium runoff (TKN and TSS), and increased PMN and POM-C levels, which likely worked in tandem to support microbial community activity (BG and MBC). The residue left on the field was then likely degraded by these more active microbial communities, contributing further to aggregate stabilization, decreased ρ_b, and the continuation of this cycle. These dynamics have all been observed separately in conservation tillage, but seldom together in a single system to correlate multiple elements of ecosystem health and soil-water interactions. To better assess potential correlation between soil health parameters and water quality outcomes, future investigation should focus on highly relevant evaluation, such as WSA through mean weight diameter, the use of in situ infiltrometers, or estimates of nutrient retention capacity.

Our hypothesis that decreasing tillage (CT > ST > MT) would improve soil health and water quality was generally supported. However, ST and MT were never significantly different, indicating that the additional strip till operation in the ST plots compared to the MT plots had no adverse water quality impact. Compared to CT, ST and MT reduce fuel, labor, and equipment use, reduce soil erosion, increase soil organic matter, and conserve soil moisture through surface residue coverage (Wardle et al., 2015; Nowatzki et al., 2017). Compared to MT, ST has been shown to support plant establishment, particularly when spring warming of the seedbed is a concern, though, in the study of the same site (Trimarco et al., 2024), yields were not different between treatments. Therefore, we propose that conservation tillage in semi-arid furrow-irrigated agroecosystems is an improvement from conventional tillage in terms of soil health, water quality, and decreased input costs. If farmers find a need to strip till to manage residue or improve seedbed conditions, it appears they can do so without sacrificing soil health or water quality, but in our study minimum tillage minimized inputs and maintained consistent yield, so may have been the most profitable. This finding may prove valuable for farmers utilizing conservation tillage in furrow irrigated systems.

CONCLUSIONS

This research is one of few efforts to empirically connect soil health and water quality in agricultural systems. This direct empirical approach allowed us to identify that aggregate stability and protection of POM-C is an important factor in runoff of sediment and nutrients in semi-arid furrow-irrigated agroecosystems. Overall, conservation tillage (minimum and strip till) improved physical and biological soil health and reduced transport of sediment, Se, and TKN. Conservation tillage may further prove effective in managing surface residue and soil compaction to the benefit of soil health and water quality without sacrificing crop yield (Trimarco et al., in review). It was particularly key that the additional tillage operation performed in ST plots relative to MT plots had no negative effect on soil physical health or water quality, though it slightly reduced soil biological health. This indicates that a farmer may choose to manage under MT or ST with little consequence to environmental health, though adding marginal tillage practices such as strip tillage should be to address an agronomic need (i.e., seed emergence or soil compaction) at the cost of additional inputs. Future work should continue to pair soil health with water quality to better understand the mechanisms driving overall environmental health under a broader range of ecosystems.

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CHAPTER 5 – ECONOMIC IMPACTS OF VARIOUS PAYMENT SCHEMES FOR CARBON SEQUESTRATION IN CONSERVATION TILLAGE

SUMMARY

Conservation tillage has increasingly been purported to be a critical tool in sequestering soil C on a global scale. The creation of voluntary C offset markets to address the externality of carbon emissions has allowed private corporations or funders to support conservation tillage projects and alleviate upfront costs. This may lower the barriers to adoption of conservation tillage in some agroecosystems, but the significant agronomic challenges associated with reducing tillage in semi-arid furrow-irrigated systems has resulted in sluggish adoption. Reflecting on a 13-year conservation tillage study in furrow-irrigated plots in northern Colorado, USA, we utilize a partial enterprise budget analysis to compare two key payment schemes on the basis of cost-effectiveness, distributional effects, and breakeven prices for sequestered carbon: flat rate payments analogous to participation in the voluntary carbon market and frontloaded payments analogous to cost sharing or federal grants. We found that the 13-year net present value of each operation was highest for strip tillage ($1,790,382 at a 30% farmer discount rate) and lowest for minimum tillage ($1,574,552), with conventional tillage falling between the two ($1,669,112). Results indicate that the high discount rates amongst farmers compared to project funders allow for significant efficiency gains and cost-effectiveness using frontloaded payments. For example, at a 30% farmer discount rate, average annual cash payments required for minimum tillage to be equivalent to conventional tillage fell from ~$47 to ~$18-28, depending on the specific payment design. However, due to low modelled C sequestration rates, breakeven prices for minimum tillage still were far above hypothetical carbon offset credits, implying a

farmer would need to be paid between \$230 and \$538 per ton CO_{2e} to justify transitioning from conventional tillage to minimum tillage. Furthermore, measured sequestration in the minimum tillage scenario far underperformed modelled sequestration, making carbon-based payments even more unfeasible. Strip tillage, on the other hand, outperformed conventional tillage and minimum tillage in terms of net present value of production, total profitability, and sequestration, indicating that no payment was necessary to switch from conventional to strip tillage. Overall, the findings support the notion that as carbon sequestration potential is variable and requires further research in semi-arid systems, a more holistic approach should be used to quantify and compensate the benefits and costs of conservation tillage. Consequently, if the goal of payments to farmers is to increase adoption of conservation practices, frontloading payments and supporting development of technical expertise may be a more advisable pathway for future policy design.

1 INTRODUCTION

We are rapidly approaching the critical threshold of 2 °C of global warming from greenhouse gas emissions. In an attempt to slow or reverse this trend, agricultural lands face increasing pressure to sequester soil carbon at a landscape scale. Of the potential ways to store carbon in soils, one of the most prominent involves transitioning farms from intense tillage systems to reduced or no till. The theory behind these practices is that by reducing microbial access to organic matter and exposure of stored organic matter to the atmosphere, we can slow the rate of decomposition while maintaining or increasing the rate of organic matter input (Six et al., 1999). A number of studies across a wide variety of landscapes report that soil carbon increases as a consequence of reduced tillage, though the degree to which soils can sequester large quantities of carbon varies by soil type, climate, and existing management conditions (Bhattacharyya et al., 2022; Cotrufo et al., 2021; Deleon et al., 2020; Blanco-Canqui et al., 2013). Some estimates indicate that reducing tillage intensity across large landscapes could result in the removal of as much as 1 ton C ha^{-1} $year^{-1}$ in the top 30cm (Ogle et al., 2019). Consequently, there is significant interest in agricultural lands to mitigate climate change pressures and to serve as generators of carbon offsets in the voluntary carbon market (VCM) and other incentive-based policy mechanisms or disclosure protocols.

However, there are a number of technical and financial barriers to reducing tillage intensity. A study from the McKinsey & Company consulting firm identified technical expertise, improvements in profit margins, and revenue generation from sustainable management to be the primary barriers to adoption of conservation agriculture practices (Fiocco et al., 2024). The technical barriers to adoption of reduced till may be particularly challenging in semi-arid irrigated cropping systems (Trimarco et al., in review). Tillage is used to manage soil conditions, facilitate irrigation management, and improve crop establishment (Nowatzki et al., 2017; Wardle et al., 2015) and the need to manage irrigation flow across the field makes conservation tillage increasingly difficult in surface irrigated systems. While long-term reductions in tillage have been shown to improve soil health in these systems (Trimarco et al., in review) and have positive effects on yield and profitability across agroecoystems in the long-term (Nunes et al., 2018), there may be significant yield reductions associated with the first few years of reduced tillage (Halvorson et al., 2006). Furthermore, these reduced tillage systems may require additional equipment or technical expertise (Wardle et al., 2015). The combination of these factors makes conversion to reduced or no till a decision that often results in decreased short-term profitability.

However, the benefit to society of reduced emissions from conservation tillage can be rather large in magnitude (Rennert & Kingdon, 2022). Trimarco et. al (in review) further indicated considerable improvements to soil health and water quality under conservation tillage after a decade of conversion, indicating that non-market benefits include more than simply sequestered carbon. Therefore, payment to farmers is a critical component in adopting reduced or no till at scale for the betterment of both carbon capture and ecosystem health. A number of incentive schemes are available to help farmers adopt conservation practices. In this study, we evaluate the potential of two prominent categories of compensation - VCMs and government conservation contracts – to understand how these tools can help to improve conservation tillage adoption.

In VCMs, third-party agencies such as IndigoAg (https://www.indigoag.com/), facilitate contracts between farmers and carbon offset purchasers (Downey, 2023). The purchasers of these carbon offsets are typically large corporations interested in claiming reduced emissions as part of an internal effort or marketing campaign. The third-party helps to aggregate together multiple farmers into a pool of carbon offsets which can be generated and sold, facilitating an easier transaction while taking a percentage of the payment. For the carbon credits to become verified, the project design and outcome must be verified by a carbon "standard", or an organization tasked with ensuring emissions goals are met and accurately recorded. These organizations create protocols, or scientifically-founded guidelines for projects to generate carbon offsets. For example, IndigoAg utilizes the Soil Enrichment Protocol, which was created by the Climate Action Reserve (a standard organization) to generate carbon credits from agricultural lands (Pape, 2024).

There are other incentive programs available to help farmers adopt conservation practices. Both federal and state agencies distribute public sources of funding available to producers hoping to adopt conservation practices on their farm (*List of Agricultural*). Government programs such as the Environmental Quality Incentives Program (EQIP) and the Conservation Stewardship Program (CSP) provide funding to help farmers adopt reduced tillage (Government Programs, 2021). These programs typically involve contracts between the farmer and the federal government (typically through the US Department of Agriculture's Natural Resource Conservation Service) and provide a combination of financial assistance and technical assistance to help farmers make this challenging transition. For example, the EQIP program offers farmers in Colorado ~$22 per acre to convert to reduced or no till in 2025 and contracts may last up to 10 years (Environmental Quality Incentives Program, 2025). The Conservation Stewardship Program has a contract minimum of $4,000 for a suite of various conservation practices related to water quality, soil health, and greenhouse gas management (*List of Agricultural Funding*). Alternatively, some contracts offer cost-sharing, which attempts to lower the upfront costs of a structural change to production practices as a barrier to adoption. However, these may be privy to the cyclical nature of administrations with differing perspectives on combating climate change. In contrast, VCMs are susceptible to the weaknesses of any private sector market with limited regulation and oversight but rely on public opinion and desire for corporations to take responsibility for climate change. Recent backlash surrounding legitimacy and reliability has seriously hurt public trust in VCMs (Harvey, 2024). This uncertainty necessitates a critical look at carbon offset trading, particularly when payment to vulnerable farming populations is part of the business structure.

Yet, the very existence of VCMs and incentive programs implies a market failure as they serve as a way to internalize otherwise external costs of agricultural production. Indeed, when we consider that every metric ton of CO_2 represents an externality that contributes to global climate change and subsequent damage, the estimated global social cost of carbon emissions is roughly $75 per metric ton CO_{2e} (2019 $USD), assuming a 2.5% discounting rate (Rennert & Kingdon, 2022). However, this is not what we would pay a farmer to sequester a ton of $CO_{2e.}$ Rather, after taking out fees and the third-party takes its cut, farmers get paid a fraction of this: roughly $10-30 (Fulwider et al., n.d.; Sellars et al., 2021). This is not only less than the value society gains but is also less than what a farmer would truly require as a breakeven price, which was estimated to be $34-43 per ton CO_2e ($USD 2010; Sellars et al., 2021).

This discrepancy in the payment from carbon offsets and the challenges that farmers face in adopting reducing tillage is exacerbated by the potential difference between the timing of these costs and benefits. Different contracts offer different payment schedules, but many involve annual payments of an agreed upon amount based upon some projected (and later verified) estimation of sequestered carbon (Fulwider et al., n.d.; Whiting et al., 2023). However, this approach may ignore the effect of differing discount rate on the value of these transactions to each party. Studies of discount rate in the US have indicated that discount rates can range from 20-40% amongst farmers (Wuepper et al., 2023), a shockingly high discount rate compared to that of carbon project funders. The federal government frequently uses two discount rates – 3% and 7% - for policy analysis (Li & Pizer, 2021). The impact of the difference between these social and private discount rates may be underutilized in policy design.

To study the impact of various policies to compensate conservation tillage, we created a partial budget analysis of a 13-year conservation tillage study managed at Colorado State University's Agricultural Research Development and Education Center (ARDEC). At this site, three tillage regimes (conventional till [CT], minimum till [MT], and strip till [ST]) were established on field-scale plots to study the soil health and water quality implications of long-term conservation tillage. Utilizing estimates of machinery operation and input costs, yield, and both modelled and measured soil C sequestration, we modelled the impact of two different payment schemes – a flat-rate payment for sequestering C every year, and a frontloaded payment scheme akin to cost-sharing – on system profitability, net present value, and break-even price for sequestered carbon. We formed the following hypotheses:

- How does the profitability of the three tillage regimes compare over time?
- How are different tillage regimes sensitive to changes in input cost?
- Will a flat-rate payment for sequestered carbon approach farmer breakeven prices for reduced tillage?
- What is the distributional effect of frontloading payments compared to a flat-rate payment?

2 MATERIALS AND METHODS

2.1 Site Description

In 2011, a 13-acre field at Colorado State University's ARDEC facility (40°40'40"N 104°59'51"W; elevation = 1567 m) was converted from an intensively tilled row crop rotation to a series of six research plots exploring three unique tillage regime - CT, MT and

ST – each replicated twice. Conventional tillage was primarily managed under moldboard plow, MT under vertical tillage, and ST under a combination of vertical and strip tillage. It is worth noting that the conservation tillage techniques were adapted over time. Originally, MT was managed with extremely minimal tillage for two years, but significant challenges with seed emergence prompted additions of vertical tillage. The full schedule of all field operations is available upon request. The dominant soil series at the site was the Garret series, a fine-loamy, mixed, mesic Pachic Argiustoll (Soil Survey Staff, 2023). Mean monthly temperatures peaked at 30° C in July and reached a minimum of 5 °C in December. Average annual precipitation of ~270 mm necessitated surface irrigation to maintain high yields in a corn-dominated rotation (Colorado Climate Center, 2023). The site was originally converted and managed under guidance of the Colorado Corn Grower's Association as a testing site to develop technique for conservation tillage in furrow-irrigated row cropping systems. The unique challenges of managing residue in irrigation furrows has made conservation tillage adoption sluggish in the semi-arid west, so the development of regional expertise and techniques is critical in improving adoption rates.

2.2 Partial Enterprise Budget Analysis

To assess the production value of these tillage regimes, a partial enterprise budget analysis was built for each tillage regime from 2011 to 2023. The key benefits and costs of this budget analysis included the following: machinery operation costs, fertilizer inputs, pesticide inputs, seed inputs, harvested grain or silage, and harvested crop residue. The analysis in this report is a close approximation of the actual management over the 12-year study period. A management scheme was reconstructed for each tillage regime using detailed management records maintained by research station staff and simplified slightly to establish a minimum set of equipment needed for each regime. Simplifications were minor, such as assuming that all operations in a regime were performed by the same tractor of the maximum horsepower needed for the most intense field operation, rather than the set of several tractors available to research station staff, or the combining of one strip tiller in the early stages of the experiment and a different but similar strip tiller in the late stages of the experiment into a single strip tiller used continuously. In general, these minor changes should serve to simplify and streamline the simulation to improve the clarity of presented results. To better represent the scale of these farm operations, all costs and benefits were simulated as if the field was 500 acres, and equipment size was adjusted to reflect that a larger field could allow for use of a larger, more efficient implement. The budget analysis is presented as if at year 0 (i.e. 2011), the farmer simultaneously purchased all new equipment for their chosen tillage regime, utilizing bank loans (inflation-adjusted interest rate of 5.5%) to do so. This may be slightly unrealistic, as farmers making the decision to transition to conservation tillage are likely do so with an established management schedule and must instead purchase additional equipment (such as a strip tiller). To address this scenario, we also performed a more "conservative" scenario where the farmer choosing to utilize conservation tillage not only purchases all required conservation tillage equipment, but also purchases a moldboard plow, landplaner, and cultimulcher – the three primary pieces of equipment that differentiate an MT or ST equipment set from the CT equipment set. The only additional costs of these equipment come from ownership and paying off bank loans, as it is assumed this equipment is "wasted" and sits unused. This more conservative scenario is used for the majority of analysis in this study, unless otherwise noted. All costs and benefits are presented in 2023 USD (adjusted using the US Bureau of Labor Statistics' Consumer Price Index Calculator available at https://www.bls.gov/data/inflation_calculator.htm) unless otherwise noted. In general, attempts were made to source pricing of both costs and benefits based on regional or national averages to broaden the applicability of the study and to limit the bias that operating a university research farm may have on prices received or paid. When possible, prices were obtained from the United States Department of Agriculture's National Agricultural Statistics Service (USDA NASS, available at https://www.nass.usda.gov/index.php). However, when such prices were unavailable or not easily proxied, prices from farm management records were used supplementarily.

Machinery operation costs were estimated using the Machinery Cost Calculator from Iowa State University Extension and Outreach. This tool estimates annual operating costs for a piece of equipment and the tractor it requires. The inputs to this tool include the original purchase price, current list price, age of machine when purchased, current age, annual hours or acres of use and the years of ownership remaining for each implement and the tractor used in the tillage regime. It further requires the acreage used, tractor horsepower, accumulated hours of use on the tractor, field capacity of the implement, diesel price, labor rate, and interest rate. Using these inputs, the calculator estimates the capital recovery (interest and depreciation), taxes, insurance, housing costs, repair costs, fuel and lubrication costs, and labor costs of a single implement for a single year (Edwards, 2015). Purchase price and salvage price of machinery was estimated using a combination of the 2023 Illinois Farm Business Management Handbook (Lattz & Schnitkey, 2023b) or when this was not sufficient, by calling local equipment dealers and obtaining estimated list price quotes, with purchase price calculated as 85% of list price (detailed quotes available upon request). Effective field capacity for each implement was obtained from either the University of Illinois' Farm Business Management Handbook (Lattz & Schnitkey, 2023a) or calculated using the following equation (Hanna, 2016):

$$Effective\ Field\ Capacity\ (ac\ per\ hr) = width\ (ft) * speed\ (mph) * \left(\frac{5280\,\frac{ft}{mi}}{43{,}560\ \frac{ft^2}{ac}} \right) * field\ efficiency\ (\%)$$

Interest rate was modelled at the default 5.50% for the Machinery Cost Calculator tool. The engine horsepower varied by treatment: the CT regime required a 295 HP tractor to operate the moldboard plow, while the MT and ST regimes required only a 245 HP tractor. Diesel price and labor cost were estimated at $2.50/gallon and $20/hr in the calculator respectively and later adjusted to reflect realistic costs. These costs were estimated using the Machinery Cost Calculator, but at times a single implement was used twice in a single crop year. To address this, an average annual usage (total hours used over the life of the project divided by 13 years) was used to calculate interest, depreciation, taxes, insurance, housing, and repair costs, and the individual fuel and labor costs of a single 500 acre pass were used to assess machinery operation costs. For example, the moldboard plow was used slightly less than once per year, resulting in an average annual use of 462 acres per year, which was used to calculate interest, depreciation, taxes, insurance, housing, and repair costs, as these are largely dependent on annual wear and tear and maintenance. However, a single 500 acre pass was used to estimate the fuel, lubrication, and labor cost of operating the moldboard plow in any given instance and multiplied by the number of passes utilized in any given year. Later, fuel and lubrication costs were adjusted using the NASS Survey's Diesel Index Price from 2011 to 2024 based on 2011 prices and the price paid for diesel in 2011 under the same survey (USDA National Agricultural Statistics Service). Labor costs were similarly adjusted to assume that all labor was performed at Colorado's minimum wage (U.S. Department of Labor). Custom operations were performed for all harvesting and hauling events and field operations where over the course of the 13-year period it was deemed unlikely for a farmer to purchase equipment based off the infrequency of use. This included ripping/subsoiling, planting of barley and winter wheat using a drill seeder, planting of dry beans, the occasional discing operation in the CT regime, the three times that a strip tiller was used in the MT regime to band apply fertilizer and the two rare use of a dry fertilizer spreader cart in the MT and ST regimes (MT and ST primarily used liquid fertilizer on the back of a cultivator or strip tiller, whereas CT additionally more regularly used a spreader cart for dry fertilizer and so was modelled as bearing the full ownership costs of the spreader cart).

Fertilizer prices were similarly estimated by utilizing the NASS' national survey of fertilizer prices from 2011-2024 (USDA National Agricultural Statistics Service). For fertilizers that were not listed in this survey, a fertilizer index provided by the same survey was combined with estimated list prices obtained by contacting local chemical dealers to estimate the cost of various fertilizer inputs over the experiment period. Pesticide prices were obtained by contacting local chemical dealers and adjusted using the same fuel index as was used to adjust diesel prices, as theoretically prices of fuel, an input to pesticide, should influence pesticide price and no pesticide index was readily available. This may induce some uncertainty in our model findings, but as pesticide and seed use did not vary across treatments, it should not influence any conclusions made about our model. Seed prices were determined using the NASS seed index (USDA National Agricultural Statistics Service) and the purchase price of a flexible grain/silage corn variety of 2022 to adjust corn seed prices in 2011, 2012, 2013, 2014, 2015, 2017, 2020, 2021, 2022, and 2023, when grain or silage corn was grown. Purchase prices from farm records for barley, dry beans, and winter wheat for 2016, 2018, and 2019 respectively were used.

Yield of each crop under each treatment was monitored as part of the ongoing research on the site and used as the estimated yield in this simulation. In 2018, dry beans failed entirely due to hail and disease and were not recovered with crop insurance. This event is included in the simulation to incorporate some risk of lost yield as part of the model. In 2019, winter wheat yield data was recorded for all of the plots as a whole, but plot-specific (and therefore treatment-specific) yield was not gathered. To estimate the treatment effect on yield, a linear relationship between year and the percent above or below CT yield was assumed for MT and ST using all other years. The linear regression equations for MT and ST were used to calculate the contribution of each treatment to total yield across the plots so that the total yield was consistent with farm records and the treatment effect on yield followed annual trends. Commodity prices received were gathered from the United States Department of Agriculture's Economic Research Service database of cost and return estimates of corn and barley for the Northern Great Plains region (U.S. Department of Agriculture, Economic Research Service). Prices received for silage corn and winter wheat were obtained from farm records due to a lack of availability. As part of the management on this site, we further removed ~50% of crop dry matter after harvest across all treatments and sold the bales either to a local market or the livestock team at the research facility. Market prices for these bales were obtained from farm records and used to estimate the additional income from selling this feed. While the total weight of these bales was noted, the treatment split of these bales was rarely recorded (in fact only for barley in 2016). Consequently, for all years in which grain corn or winter wheat was grown and bales were harvested, the share of this bale harvest attributable to each treatment was assumed to be proportional to the yield in each treatment and estimated accordingly. Harvest benefit was quantified as the sum of all harvest products (silage in silage corn years, grain plus bales in grain corn years, for example).

In each year, profit was assessed as the sum of all harvest benefits minus costs accrued during that cropping period. Cropping period was defined as beginning immediately after all harvest operations (combining, shredding stalks, and baling, for example) and ending at the next harvest. This has a few notable consequences. First, tillage events or machinery operations that occur in the autumn or winter of after harvest are rolled into the next cropping period's costs, a method that more evenly distributes the costs of machinery operations than using calendar years would. Second, when indexing prices received or paid is used with input costs, the annual average index of the calendar year is used for all prices that year. For example, a moldboard plow event in November of 2022 and a cultimulching event in March of 2023 both use the fuel index of 2023, as 2023 is the year in which our crop is grown and harvested and most costs are accrued. Third, the beginning and end of cropping periods vary by year and by crop. In general, corn dominated our cropping periods and harvest tended to occur at the end of October or in early November. However, occasionally harvest would occur in early December or late November. To keep all cropping periods a consistent length of time, we elected to begin and end all cropping periods on November 16th. This required us to occasionally "time-shift" a handful of operations to slide benefits and costs of harvest into the more representative cropping period. We time-shifted the chop and bale operations from

2011 (November 29th), 2014 (December 2nd), and 2015 (November 23rd), back to November 16th of their respective year to include them as part of the cropping period of the crop they represent. We similarly time-shift a moldboard plow event, vertical tillage, and cultimulching from November 10th-12th 2022 up to November 16th 2022 to incorporate them as part of 2023's operational costs, since they represent soil management for the 2023 crop. We did not time-shift the planting of winter wheat in September of 2018, leaving this as a 2018 cost since the time-shift would have been much more substantial than that of the other events that were time-shifted. This method was chosen to better represent the decisions and annual budget evaluations farmers make when considering management alternatives. A farmer who grew grain corn in the summer of 2022 would likely view all the soil management operations prior to planting and during growth as incurred costs in for the 2022 cropping period and the harvest of this crop (regardless if it happens on December 2nd or November 2nd) as a benefit of the 2022 cropping period. This approach maintains a 365-day cropping period that as accurately as possible captures all costs and benefits of a growing a crop to harvest.

2.3 COMET-Farm Simulations

The COMET-Farm tool (*COMET-Farm v5.*0, 2024) was utilized to assess the potential C sequestration of switching from CT to MT or ST. COMET-Farm is an online tool used to estimate the potential greenhouse gas emissions from various farm management scenarios by sending the user-built schedules to a DayCent algorithm and modelling soil, water, plant, and atmosphere relations over the simulation period. We entered all harvest, planting, tillage and fertilizer events for each tillage regime for 13 years into the COMET-Farm platform, allowing the program to auto-generate irrigation events. We then compared the estimated soil C sequestration for each regime. The full details of the COMET-Farm model can be found at https://comet-farm.com/documentation. It is worth emphasizing that the Terms and Conditions for COMET-Farm make it clear that COMET-Farm is not to be used as "an accounting framework for emission reduction crediting or trading" as the "provided results [do not] constitute an offset protocol or full lifecycle GHG analysis". However, the Climate Action Reserve's Soil Enrichment Protocol (*U.S. Soil Enrichment Protocol v1.1*, 2024) specifically mentions that COMET-Farm model may be appropriate for project planning, but thorough soil sampling is required for project verification, and COMET-Farm appears to be used as a quantification tool for participation in VCMs (*Ranchers in Marin County Consider Carbon Credits*, 2024). The use of COMET-Farm in this report is not intended to be part of an offset protocol or to engage in any carbon crediting. Rather, it is used to estimate the potential for sequestration under various tillage scenarios to compare the benefits and costs of various compensation policies, if model estimates were verified as accurate. Were a farmer to enroll in a carbon crediting contract, they may use a model such as COMET-Farm to estimate their potential sequestered soil C and possible payments, but the outcomes of this model could not be used to independently generate carbon credits and verification of sequestered C is necessary. In this work, we consider the perspective of a farmer hoping to gain compensation for switching to conservation tillage and we compare the distributional effects and cost-effectiveness of various payment plans analogous to carbon offsets and federal conservation grants using both modelled sequestration and measured sequestration.

2.4 Soil C Measurements

In summer of 2024, soil organic C was measured in each plot in a manner broadly consistent with the methods presented in Trimarco et al. (in review). In brief, soil samples were collected using a 2.5 cm diameter step probe to a depth of 30 cm and split into two depths: 0-15 cm and 15-30 cm. Approximately 10 cores were sampled within a 3-m radius centered around a GPS located point consistent with Trimarco et al. (in review), composited by depth, transferred to a plastic Ziploc bag, sealed, and placed in a dry cooler with ice packs. At the center of this sampling radius, an additional intact core was taken for ρ_b determination at each depth using a Madera probe (Evett et al., 2022; Precision Machine Co., Lincoln, Nebraska, USA) and a bucket augur. Soils for bulk density (ρ_b) were placed in tin cans and immediately sealed with tape to limit evaporative losses. Two sampling radii were established at the top and bottom of each plot for a total of four sampling radii in each plot.

Upon returning to the lab, ρ_b and moisture content were determined by immediately weighing moist cores stored in tin cans, drying at 105 ^{o}C for 24 hr, then weighing and repeating until a consistent dry weight was obtained. Soils were powder ground and total soil C was measured via dry combustion on a VELP Dumas Elemental Analyzer (VELP Scientifica, Usmate Velate, Italy; Nelson and Sommers, 1996) using 0.100 g of this powder-ground air-dried soil, while inorganic C was determined via the pressure transducer method (Sherrod et al., 2002) using 1.00 g of powder-ground air-dried soil. Soil organic C concentration was determined by subtracting inorganic C from total C. Soil organic C stock was determined by the following equation:

$$SOC\ Stock\ in\ a\ given\ sample\ thickness\ (tonne\ C\ ac^{-1})$$
$$= SOC\ (mg\ g^{-1}) * Bulk\ Density\ (g\ cm^{-3}) * Sample\ Thickness\ (cm) * 0.1 * 2.47\ ac\ ha^{-1}$$

The total SOC stock in each plot was considered to be the average of the sum of SOC stock from the 0-15cm and 15-30cm depths. The carbon dioxide equivalent (CO_{2e}) of a ton of SOC stock was 3.67 tonnes CO_2.

2.5 Benefit-Cost Comparison of Compensation Schemes

To calculate the net present value of each tillage regime, we treated the simulation as if the farmer were making the decision to choose a tillage regime in 2011 and reflecting on the value of that decision in 2023. We therefore discount future benefits from 2011 forward and consider the net present value in terms of 2023 $USD. Net present value was calculated according to the following equation:

$$Net\ Present\ Value = \sum_{t=0}^{12} \frac{R_t}{(1+i)^t};$$

where t is the number of years after the experiment started (2011 = 0, 2012 = 1, … 2023 = 12), R_t is the cash flow in year t (as either a benefit or cost), and i is the discount rate.

While most private companies have discount rates around 10-20% (Calhoun & Harkins, 2021) and government policies are evaluated at 3% and 7% discount rates (Li & Pizer, 2021), surveys and designed experiments with farmers indicate that private discount rates amongst farmers range from 20% to 40% (Duqette et al., 2012; Wuepper et al., 2023). We therefore evaluate the effect of varying farmer private discount rates from 20% to 40% and payer discount rates of 3% and 7% on net present value, break-even price, and policy design. We performed a sensitivity analysis on the net present value of farm operations to the farmer utilizing three discount rates (20%, 30%, and 40%) and with varying change in input costs of fuel, fertilizer, pesticide, and seed (-25%, no change, +25%) to understand how uncertainty in input costs could impact farmer decision making and risk tolerance.

Based upon the results of this NPV simulation and the estimated C sequestration from the COMET-Farm simulations, we estimated the minimum breakeven price per ton CO_{2e} in order for a farmer to see the NPV of the MT and ST regime plus carbon offset payments be equivalent to the NPV of the CT regime. Breakeven price was estimated under three different payment schemes: 1) where an even cash payment is paid to the farmer each year from year 1 to 13; 2) where 50% of the total cash payment required is paid in the first year, 20% each in years two and three, and 10% in year four; and 3) where 30% of the total cash is paid in the first year, 15% in the second year, 10% in the third year, and 5% in each subsequent year.

In the following discussion, we attempt to visualize the benefits and costs of each payment scheme and tillage regime through the lens of a farmer planning the next 13 years of their operation beginning in 2011. We ask what payment scheme provides the critical compensation to the farmer with timing and distributional considerations. We further plan for our funding schemes in a manner analogous to the way a carbon offset project may be funded: with COMET-Farm modelled sequestration forming the basis of carbon offset payments and end-of-project direct soil C measurements to verify sequestration. In reality, the carbon offset protocols such as the Soil Enrichment Protocol (*U.S. Soil Enrichment Protocol v1.1*, 2024) require verification sampling roughly every 5 years in part to ensure project compliance. However, the experimental plots studied in this research were not planned with carbon offsetting in mind and this monitoring was not performed. Regardless, compliance and recording of tillage operations is not a concern with this project, so soil C measurements are largely used to verify sequestration. Throughout the discussion, we refer to the entity compensating the farmer to switch to conservation tillage as the “payer”. In this scenario, the payer could represent a corporation or third party carbon offset contractor compensating the farmer through the voluntary carbon market or the payer could represent a government funding agency such as the NRCS EQIP program. For this discussion, we will refer to them as the “payer” and make references to price points for both carbon credits that a purchaser could make in the voluntary carbon market and for acreage contracts that federal grants provide to farmers. As carbon markets mature and develop, it is certainly possible that government entities enter into direct carbon trading agreements with farmers, but for this report the primary comparison is between payment structures commonly facilitated by various types of payers, not types of payers themselves. All calculations were performed in Microsoft Excel.

3 RESULTS AND DISCUSSION

3.1 Soil C Sequestration

The COMET-Farm model estimated that from 2011 to 2023, the CT regime would have sequestered 0.077 tonnes CO_{2e} ac^{-1} $year^{-1}$, MT would have sequestered 0.16 tonnes CO_{2e} ac^{-1} $year^{-1}$, and ST would have sequestered 0.13 tonnes CO_{2e} ac^{-1} $year^{-1}$, indicating that MT and ST would have sequestered 0.080 and 0.054 tonnes CO_{2e} ac^{-1} $year^{-1}$ over the baseline, respectively.

However, direct soil C measurements indicated different outcomes. At the end of the experiment, the CT plots held 4.29 ± 0.23 (mean ± standard error) tonnes CO_{2e} of SOC stock per acre compared to the 4.36 ± 0.20 tonnes and 5.32 ± 0.21 tonnes CO_{2e} in the MT and ST plots respectively. This represents a sequestration rate of 0.0057 tonnes CO_{2e} per acre per year in the MT plots and 0.079 tonnes CO_{2e} per acre per year in the ST plots. For the ST plots, this is somewhat close to what the COMET-Farm model predicted with the model underpredicting actual sequestration by ~32%, but the MT plots severely underperformed compared to model simulations, achieving only ~7% of the modelled sequestration rate. The reasons for the severe underperformance warrant further research in the future. If tillage was intense enough in the MT regime to deviate from the model, that may explain some of the underperformance, but the ST regime outperformed the MT regime in terms of actual sequestration while receiving more intensive tillage, so this explanation is somewhat dubious. It may be that decreased yield in the MT plots resulted in decreased plant matter in those plots and hence decreased organic matter inputs, whereas the highly competitive yield in the ST plots (see Supplementary Table S.4) maintained higher levels of organic matter inputs to the soil. Regardless of the cause of this discrepancy, the payment plans presented below are reported in the context of modelled sequestration rates, as this is the structure of most carbon offset contracts. The underperformance of the MT regime is discussed further at the end of this section.

3.2 Partial Enterprise Budget Analysis

Estimates of the net present value for each tillage regime are presented in Table 5.1. One of the most important findings presented in this table is that at the end of the 13 year experimental period, the net present value of the ST regime outperformed the CT regime at all three evaluated discount rates and input costs. One of the intentions of this simulation was to identify the breakeven price for sequestered soil carbon for the MT and ST regimes, but the results below suggest that no such payment is necessary for ST to outperform CT. This is contrary to our hypothesis that both conservation tillage regimes would require compensation to reach the NPV of the CT regime. This discrepancy may be due to the fact that CT heavily outperformed MT in terms of yield in the early years of the experiment, with MT having as much as 23% decreased yield in 2012 and 16% decreased yield in 2013. Meanwhile, ST had reductions of yield of less than 11% for every year, and the decreased yield in the early years tended to have a smaller effect than the decreased input costs. Consequently, as we further explore the data in this simulation, we will focus our compensation schemes on transitioning farmers from CT to MT and the required payments to do so. The effect of these payments will still be presented for the ST regime, but "custom" payment schedules for the ST regime are not presented here.

Table 5.1. Net Present Value of the Three Tillage Regimes (CT, MT, and ST) at Various Discount Rates and Input Costs Under Two Simulation Scenarios

Conventional Tillage							
Scenario #1 – Baseline				Scenario #2 – Conservation Pays Extra			
	Farmer Discount Rate				Farmer Discount Rate		
Input Cost	40%	30%	20%	Input Cost	40%	30%	20%
-25%	$1,573,898	$1,841,534	$2,281,527	-25%	$1,573,898	$1,841,534	$2,281,527
No Change	$1,427,154	$1,669,112	$2,064,835	No Change	$1,427,154	$1,669,112	$2,064,835
+25%	$1,280,410	$1,496,689	$1,848,142	+25%	$1,280,410	$1,496,689	$1,848,142
Minimum Tillage							
Scenario #1 – Baseline				Scenario #2 – Conservation Pays Extra			

	Farmer Discount Rate				Farmer Discount Rate		
Input Cost	40%	30%	20%	Input Cost	40%	30%	20%
-25%	$1,484,286	$1,748,802	$2,196,280	-25%	$1,465,234	$1,726,659	$2,169,021
No Change	$1,355,607	$1,596,695	$2,003,452	No Change	$1,336,554	$1,574,552	$1,976,194
+25%	$1,226,927	$1,444,587	$1,810,625	+25%	$1,207,874	$1,422,444	$1,783,366
Strip Tillage							
Scenario #1 – Baseline				Scenario #2 – Conservation Pays Extra			
	Farmer Discount Rate				Farmer Discount Rate		
Input Cost	40%	30%	20%	Input Cost	40%	30%	20%
-25%	$1,686,671	$1,965,774	$2,428,913	-25%	$1,667,619	$1,943,631	$2,401,654
No Change	$1,557,134	$1,812,525	$2,234,344	No Change	$1,538,081	$1,790,382	$2,207,085
+25%	$1,427,597	$1,659,275	$2,039,775	+25%	$1,408,544	$1,637,132	$2,012,516

The MT and ST regimes were less sensitive to change in input costs than the CT regime. For example, in scenario #1 at a 30% farmer discount rate, a 25% increase in input costs decreased NPV of MT and ST regimes by $152,108 (-9.5%) and $153,250 (-8.5%) respectively, whereas the same change in input costs resulted in a loss of $172,423 (-10.3%). This is somewhat expected, as many of the tillage events unique to the CT regime (specifically the moldboard plow) are fuel-intensive, so changes in fuel cost will have a larger impact on NPV under CT. This may be viewed as a positive aspect of the MT and ST regimes; uncertainty about the future price of input costs, especially those directly or indirectly tied to fuel prices, may be mitigated by decreasing the inputs into the system.

One important consequence of high farmer discount rates is a potential mistiming of benefits from incentive programs. Payments that accrue at flat rate each year or increase over time (such as is common in carbon markets) pay farmers equivalent or greater cash as the NPV of this cash sharply decreases. Essentially, this system asks farmers to take a large risk when costs are -

discounting-wise - the highest, only to spread out or delay payments until the benefits of these payments are minimized or fail to address early risk-taking. Furthermore, corporations and governments face far lower discount rates than individual farmers; upfront costs are not weighted as heavily for these large, multi-generational organizations. Calhoun & Harkins (2021) note that most corporations use discount rates of 12-20%, but a well-diversified, large, consistently growing company (like many funding large carbon offset projects) may face a discount rate below 10%. Regardless of the exact discount rate, paying a large sum up front is not nearly as detrimental to the long-term net present value of operations to the buyers of carbon offsets as it is to the farmers who take on the risk of switching tillage techniques. We therefore consider policy design that frontloads payments to farmers to match upfront costs with compensation to more directly address barriers to adoption.

Based on the NPV's calculated in Table 5.1, we present the necessary cash payments, breakeven prices per ton of modelled sequestered CO_{2e}, and distributional effects of the three payment policies (Table 5.2). For brevity, moving forward we present only these data from payment policies under the more "conservative" scenario #2, where the MT and ST regimes must pay off ownership costs of unused machinery. As ST did not require any additional compensation to convert from CT, the values provided in Table 5.2 represent payment plans to supplement the MT regime. In these payment plans, the cash required to compensate NPV is calculated using the Solver tool in Microsoft Excel with the cash splits described and the sum of the NPV of these cash payments equaling the difference between MT and CT NPV. The breakeven price represents this cash payment divided by the total estimated tonnes of sequestered CO_{2e} over CT, and the cash paid per acre per year is this cash payment divided by 500 acres and by 13 years, with neither of these values discounted.

Table 5.2. Breakeven Price, Cash Paid, and Payer's NPV of Three Payment Plans Under Varying Farmer Discount Rates with Modelled Sequestration

	Farmer Discount Rate		
	40%	30%	20%
Breakeven Price ($ per tonne CO_{2e})			
Flat Payment	$676	$583	$425
50-20-20-10 Payments	$230	$230	$206
25-15-10-5… Payments	$371	$352	$289
Cash Paid			
Flat Payment	$351,723	$303,606	$221,012
50-20-20-10 Payments	$119,668	$119,677	$107,099
25-15-10-5… Payments	$197,876	$183,248	$150,235
Cash Paid by Acre Per Year			
Flat Payment	$54.11	$46.71	$34.00
50-20-20-10 Payments	$18.41	$18.41	$16.48
25-15-10-5… Payments	$30.44	$28.19	$23.11
Payer's NPV (3% Discount Rate)			
Flat Payment	$296,368	$255,823	$186,228
50-20-20-10 Payments	$116,581	$100,842	$104,336
25-15-10-5… Payments	$172,029	$163,442	$133,997
Payer's NPV (7% Discount Rate)			
Flat Payment	$241,950	$208,850	$152,034
50-20-20-10 Payments	$112,875	$82,326	$101,019
25-15-10-5… Payments	$151,266	$143,715	$117,824

Table 5.2 provides a couple key insights about the impact and distributional effects of these payment policies. At a 30% farmer discount rate, switching from a flat payment to the 50-20-20-10 payment structure saved the payer \$183,929 in cash and \$154,981 of NPV (at a 3% payer discount rate) while maintaining equivalent NPV back to the farmer. Furthermore, the cash paid was delivered in a critical time period when the farmer faced high levels of risk and uncertainty.

Perhaps one of the most notable findings presented in Table 5.2 is that the breakeven price for a CO_{2e} ton of sequestered soil C across farmer discount rates and payment plans far exceeds that of realistic payments for sequestration. Current carbon prices range between \$20-50 per ton CO_{2e}, but the breakeven prices calculated above are roughly 10x as large. A recent study by Wang et al. (2024) similarly found that current carbon payments are too low to sufficiently motivate farmers to switch practices in the US Midwest, but even the prices presented in that study are well below \$100 per ton of CO_{2e}. There is scant research on the decision-making behavior of farmers in the semi-arid western US, but the breakeven prices in Table 5.2 indicate that current carbon offset prices may be even less impactful to farmers in the region. This may be in part due to the low levels of carbon sequestration feasible in semi-arid systems where precipitation is heavily limited (Ogle et al., 2019). Furthermore, many carbon offset programs assume far more conservative tillage than may be technically feasible in furrow irrigated systems. In these furrow-irrigated systems, additional operations are required to manage crop residue on the soil surface to ensure water can easily move to the end of the field (Wardle et al., 2015), which may be a contributing factor to the limited adoption of no till in the western US. These secondary tillage operations disturb the soil and limit the rate of C sequestration, putting a cap on the potential for carbon offset generation on these fields. Carbon offset programs may be better suited for farmers in temperate climates where potential sequestration rates are higher due to climatic conditions and technical sequestration rates are not limited by the need for furrow management. The disparity between carbon payments and breakeven prices may be framed an other way: a farmer receiving a flat rate payment of \$45 ton^{-1} CO_{2e} would need to sequester ~1 ton ac^{-1} yr^{-1} using our breakeven cost estimates, over 10x what was modelled using COMET-Farm. Ogle et al. (2019) emphasized that the gap between theoretical and technical sequestration potential, particularly in dry regions, is a reason to consider no-till (and logically, reduced tillage in general) a soil health- and water quality-focused management technique first. In semi-arid furrow-irrigated systems, farmers may not be able to successfully enough carbon offsets to compensate for the upfront costs of switching to conservation tillage.

Indeed, it appears that C sequestration is so limited in our MT regime – which aimed to keep tillage as minimal while maintaining irrigation furrows – that breakeven prices far exceed even the social cost of carbon, with most estimates falling below \$100 per ton CO_{2e} (Rennert & Kingdon, 2022). However, Rennert et al. (2024) recently showed that newer models indicate \$185 (\$44-\$413 in 2020\$) per ton of CO_{2e} may be more appropriate and even approaches the breakeven prices in our most aggressively frontloaded scenario. However, given the wealth of cost-efficient climate solutions presented by organizations such as Project Drawdown (drawdown.org), paying over \$200 per ton CO_{2e} seems neither efficient nor cost-effective. When we further note that measured sequestration in our MT regime vastly underperformed compared to our model simulations, this gap between breakeven price and achievable carbon sequestration becomes even more concerning. It is notable however that front loading payments, particularly with our 50-20-20-10 payment schedule, cut the breakeven price by as much as >50%. It is not entirely unreasonable to assume that if in the MT regime C sequestration was greater than what was achieved in this simulation or if upfront yield was more competitive, a breakeven price closer to realistic carbon offset payments more achievable. In the likely scenario where breakeven prices are too high, front loading payments appears to be an effective way to wrangle farmer willingness-to-accept closer to a funder's willingness-to-pay.

One considerable downside of front loading payments is the risk of losing permanence of carbon emissions or farmers choosing to unenroll in the program partway through the process. This risk becomes greater the more frontloaded a payment schedule is; the 50-20-20-10 program above provides no direct incentive for a farmer to maintain reduced tillage practices past the fourth year and a farmer could certainly choose to revert back to conventional tillage practices, causes the soil to re-emit the carbon that was supposedly sequestered. A funding agency or payer could sign the farmer to a contract mandating a particular set of tillage practices for 13 years while only paying them for the first few years, but this has significant political challenges and while it is designed to provide payment to farmers when they need it most, it could have the appearance of being overly legislative or restrictive. These concerns prompted the assessment of the 25-15-10-5… plan, where farmers are disproportionately paid in the front half of the contract but there is still some financial incentive to maintain conservation practices as the project matures. This may face similar challenges when farmers see decreasing payments in the late stages of the contract, but by year 5 of our simulation the cash profit of any given year in the MT regime was roughly equivalent to the cash profit of the CT regime in the same year and by year 7 to 9, regularly outpaced CT profitability (see Supplementary Table S4). In this simulation, we do not explicitly model farmer's decision-making after the first year, but if a farmer had access to this information and could decide to return to CT or maintain MT practices, they would identify that MT is more profitable than the CT alternative even with minimal or no outside compensation. The objective of our compensation policies should then be to guide farmer decision-making to this critical point in the conservation systems or to get them "over the hump" to a time when MT is more profitable than their alternatives.

Given this policy objective, a criterion for our policy design from the farmer's perspective becomes whether or not a carbon payment or conservation incentive advances the timeline to the point where conservation tillage is more profitable sooner. Without carbon credit payments, the (non-discounted) cumulative profits from the MT regime consistently exceeded the CT regime beginning in 2022 (Supplementary Table S4). If we were to calculate the cash value a somewhat generous carbon offset payment of $40 per modelled ton CO_{2e} per year at $20,814 and frontload that payment under our 25-15-10-5… payment plan (i.e. 25% of that cash is paid in the first year, 15% in the second year, and so forth), we only move the year in which cumulated MT profit exceed cumulated CT profit by one year. This is not an entirely insignificant amount of time but would likely not be enough to truly impact farmer decision-making and risk tolerance. If we consider the same payment scenario under a 30% farmer discount rate, the payment plan leaves the NPV of MT $83,820 lower than that of CT, a mere $10,740 improvement from our original simulation at 30% farmer discount rate where we did not pay a farmer to switch at all. Flat-rate payments under the same conditions generated an $87,851 deficit in NPV for MT, indicating that indeed frontloading payments made some impact on resultant NPV, but that carbon credits alone are insufficient to offset the early upfront costs of switching to MT. Even if we consider the less conservative scenario where a farmer doesn't have to pay off bank loans for unused equipment and receives carbon credit payments under our 25-15-10-5… scheme, the NPV of that MT regime still falls below that of CT by $68,500. It is worth noting that ST outperformed CT in terms of both profitability and NPV in every scenario presented above. Without any carbon offset payments and with a 30% farmer discount rate, the NPV of ST exceeded CT by $143,413 and the inclusion of carbon offsets for modeled ST sequestration following our 25-15-10-5… scheme increased this difference in NPV to $146,055 in ST's favor. Given the above evidence, we must conclude that carbon offsets are not enough to compensate a farmer in our simulation to switch from MT to ST.

However, the findings presented in Table 5.2 do shed light on an important price point. Using our 25-15-10-5… pricing scheme, the cash payment required to compensate a farmer to switch from CT to MT was between $150,000 - $200,000, depending on farmer discount rate. When we divide that cash payment across 500 acres and 13 years, we find that the average annual payment ranges from $23-30 per acre per year, a comparable number to the annual payments of $24.31 ($23.04 2023 USD) per acre put forth by the NRCS EQIP program for fiscal year 2025 in Colorado (*Payment Schedules*, 2025). While EQIP contracts tend to be relatively short and would not frequently go for as long as 13 years of continuous reduced tillage, the convergence of this required compensation point is of note. This prompted us to perform an additional comparison of the distributional effects of frontloading compensation at the ~$28 per acre per year price point observed at the 30% discount rate. In this scenario, we set aside an amount of cash exactly equal to the amount of cash required to offset the loss of NPV for a farmer practicing MT if the farmer has a 20% discount rate ($183,248) and paid this in two ways, as a flat cash payment of $28.19 per acre per year or following our 25-15-10-5… scheme. The distributional impacts of these simulations are presented in Table 5.3. We also identify in our data the year in which MT NPV begins to regularly exceed NPV on a year-to-year basis. For both payment structures, we determined that year 4 was the key moment when it became more profitable (in terms of NPV) to be practicing MT rather than CT, regardless of discount rate. This is notably one year earlier than without any compensation at all. We similarly find that if we were to consider just the cumulative profits of each scenario, both payment plans advanced the year in which MT exceeded CT by one year – from year 12 to year 11. It is worth noting again that the NPV of cumulative profits never exceeds that of the CT regime, except in the case of the frontloaded payment plan at 20% discount rate – a feature of this policy design.

Table 5.3. Distributional Effects of Front-Loading Payments on a Per-Acre Basis

	Farmer Discount Rate		
	40%	30%	20%
Flat Rate Payment			
Total NPV Paid (Payer's Perspective – 3% Discount)	$154,408	$154,408	$154,408
Total NPV Paid (Payer's Perspective – 7% Discount)	$126,056	$126,056	$126,056
Total NPV Received (Farmer's Perspective)	$48,715	$59,066	$76,671
NPV Compared to CT (Farmer's Perspective)	-$41,886	-$35,494	-$11,970
MT NPV Begins to Exceed CT NPV	Year 4	Year 4	Year 4
Frontloaded 25-15-10-5… Payment			
Total NPV Paid (Payer's Perspective – 3% Discount)	$163,442	$163,442	$163,442
Total NPV Paid (Payer's Perspective – 7% Discount)	$143,715	$143,715	$143,715
Total NPV Received (Farmer's Perspective)	$86,088	$94,560	$108,119
NPV Compared to CT (Farmer's Perspective)	-$4,522	$0	$19,478
MT NPV Begins to Exceed CT NPV	Year 4	Year 4	Year 4

Thanks to high discount rates among farmers, frontloading the cash payment of $183,248 using the 25-15-10-5… payment plan provided significantly more benefit to the farmer at the cost of relatively little additional cost to the payer. For example, at a 30% farmer discount rate and a 7% payer discount rate, switching from a flat payment to frontloaded payment increased the NPV of payer costs by $17,659 (a 14% increase), but increased farmer benefit by $35,494 (a 60% increase). A more frontloaded payment plan could further expand this benefit discrepancy to increase total NPV but should be balanced by need to ensure compliance and permanence of reduced emissions.

Frontloading compensation for conservation tillage appears to provide a cost-effective way to help farmers overcome the early barriers to adoption. We therefore select two candidate policies to compensate our hypothetical farmer to switch from CT to MT: a flat-rate carbon credit payment of equal cash payments each year and a frontloaded payment plan (following our 25-15-10-5…) plan that may be analogous to a NRCS EQUP contract. In this simulation, both payment plans are designed to bring MT NPV up to CT NPV with a 30% farmer discount rate. We compare the distributional effects, total benefit and cost, and the cost paid per metric ton of emissions reductions of these two plans, alongside the business-as-usual scenario of CT and the alternatives of unfunded ST and MT in Table 5.4. We further consider the avoided diesel emissions of each conservation tillage scenario. Decreasing the frequency and intensity of field operations logically decreases diesel emissions, and the Machinery Cost Calculator utilized for the partial enterprise budget analysis allows use to estimate diesel use over the life of the project.

Table 5.4. Comparison of Various Policy Plans and Management Scenarios at a 30% Farmer Discount Rate

	CT	Unfunded ST	Unfunded MT	MT + Flat Rate	MT + Frontload
Farmer NPV without Compensation	$1,669,112	$1,790,382	$1,574,552	$1,669,112	$1,669,112
Cash Paid to Match CT NPV	-	-	-	$293,367	$183,248
Average Annual Cash Paid per Acre	-	-	-	$45.13	$28.19
Payer's NPV of Costs (3% DR)	-	-	-	$247,195	$163,442
Payer's NPV of Costs (7% DR)	-	-	-	$201,807	$143,715
Sequestered C (COMET-Farm; tonnes CO_{2e})	-	351	520	520	520
Sequestered C (Measured; tonnes CO_{2e})	-	513	37	37	37
Breakeven Price (Cash $ modelled ton^{-1} CO_{2e})	-	-	-	$564	$352
Payer's NPV Per Tonne Modelled Sequestered C (3% DR)	-	-	-	$475	$314
Payer's NPV Per Tonne Modelled Sequestered C (7% DR)	-	-	-	$388	$276
Social Benefit of Modelled Sequestered C ($55 ton^{-1} CO_{2e})	-	$19,305	$28,600	$28,600	$28,600
Social Benefit of Measured Sequestered C ($55 ton^{-1} CO_{2e})	-	$22,572	$1,628	$1,628	$1,628
NPV Direct Benefits – Costs	$1,669,112	$1,809,687	$1,574,552	$1,450,517	$1,534,270

(3% Payer NPV, Modelled Sequestration)					
NPV Direct Benefits – Costs (7% Payer NPV, Modelled Sequestration)	\$1,669,112	\$1,809,687	\$1,574,552	\$1,495,905	\$1,553,997
NPV Direct Benefits – Costs (3% Payer NPV, Measured Sequestration)	\$1,669,112	\$1,812,954	\$1,576,180	\$1,328,985	\$1,412,738
NPV Direct Benefits – Costs (7% Payer NPV, Measured Sequestration)	\$1,669,112	\$1,812,954	\$1,576,180	\$1,374,373	\$1,432,465
Avoided Diesel Emissions from Machinery Operations (tonnes CO_{2e})	-	445	461	461	461
Payer's NPV ton^{-1} CO_{2e} Total Modelled Emission Reductions (Fuel + COMET-Farm; 3% Discount Rate)	-	-	-	\$252	\$167
Payer's NPV ton^{-1} CO_{2e} Total Modelled Emission Reductions (Fuel + COMET-Farm; 7% Discount Rate)	-	-	-	\$206	\$146

Results indicate that the most profitable decision pathway is also the pathway which provides the highest overall social benefit: an unfunded conversion to ST from CT. At virtually every decision point, discount rate, and variation in input cost, ST outperformed CT in terms of NPV to the farmer while still sequestering large quantities of soil C. Consider further that when direct soil measurements are taken into account, the ST plots outperformed model estimates and even had much higher sequestration than the MT plots. Attempts to model the breakeven price for ST (i.e. the cash price paid per metric ton of sequestered soil C-CO_{2e}) were unsuccessful, as our model indicated that there was no need to fund transition to ST at all. This somewhat alleviates the concerns that the additional operations in the ST plots sacrificed additional carbon sequestration for agronomic benefits to improve seed emergence and yield. It's not clear exactly why ST sequestration outperformed model expectations while MT critically underperformed. Perhaps the consistently higher yield in ST than MT provided enough organic matter to the plots to improve sequestration rates but given the relatively high level of residue left on the field after harvest (~50%), it is surprising that MT underperformed model expectations. However, we do recognize that the results presented in this manuscript are from one case study and should not detract from the large body of evidence that strip tillage and similar secondary tillage techniques can limit potential sequestration rates or organic matter storage (Butkevičienė et al., 2023; Debska et al., 2020; Lal & Kimble., 1997; Martin-Lammerding et al., 2013).

Despite the limitations of comparing conservation tillage techniques in this case study, there is still significant merit to comparing the distributional impacts of the two proposed payment schemes to move a farmer from CT to MT. Moving from a flat-rate payment scheme to a frontloaded payment scheme reduced the NPV of the payer's costs from $247,195 to $163,442, a 34% decrease, while maintaining consistent NPV from the farmer's perspective. While this increase in total benefits would appear to be purely positive, the structure of payment schemes various political and legal implications. From a contract or policy design perspective, it is valuable to the payer or the recipient of carbon credits to ensure permanence of sequestration or at least of management, a reason that IndigoAg cites as the primary driver behind their backloaded payment scheme (*How much can I earn from carbon farming?*). The idea behind this backloaded payment scheme is that a farmer receives the payments for the first year of C sequestration in installments over 5 years: 50% the first year, 20% the second year, and 10% in the third, fourth, and fifth years. If the farmer were to unenroll in the program partway through, all future payments would be forfeit, including the share of already sequestered C that has not yet been paid for. This type of payment scheme provides insurance to the payer that management practice is maintained, but this comes at the cost of paying increasingly greater cash payouts to farmers as the present value of those payments decreases due to the intrinsically high discounting rate amongst farmers. The impact observed in this study - more backloaded payments resulting in decreased efficiency of payer NPV translating to farmer NPV – would be even worse if cash payments started low in increased over time. This would result in either more money being needed to given to farmers in order to match lost NPV from transitioning to conservation tillage or current payments failing to financially support farmers as necessary. The design of these payment schemes ought to continue to play close attention to distributional impacts to prioritize farmer welfare, cost efficiency, and total benefits-cost across varying discount rate of both farmers and payers. The critical need to compare discount rates between farmers and payers will become more critical as carbon markets mature and the actors in them change and develop.

Perhaps one of the most interesting dynamics identified in this study is the consistent observation that across discount rates and payment schemes, the breakeven price to compensate the MT regime on per tonnes CO_{2e} sequestered basis was considerably higher than the social cost of carbon. This gap is exacerbated when we consider the actual soil C sequestered in the MT regime as opposed to the sequestration that was modelled with COMET-Farm. For example, under the frontloaded payment scheme at a 30% farmer discount rate and a 3% payer discount rate (Table 5.4), the breakeven price per ton CO_{2e} of modelled sequestration was $354 (for a payer NPV of $314). When we consider the measured sequestration, this price increases to an alarming $4,953 ($4,417 payer NPV). Given that the COMET-Farm model is based on a distribution of many fields, it is a better tool for policy planning and decision-making than the measurements of a single case study. Still, that MT so critically underperformed compared to the model highlights the need for increased research on the potential for C sequestration in arid and semi-arid climates (Ghimire et al., 2024; Ogle et al., 2019; Plaza-Bonilla et al., 2015).

The consistency at which our breakeven price so far exceeded the social cost of carbon and common carbon offset payments is perhaps an indicator that carbon markets are not an effective tool for compensating conversion to conservation tillage in semi-arid furrow-irrigated systems. However, as many have espoused, the benefits of transitioning to conservation tillage far exceed greenhouse gas mitigation (Ogle et al., 2019; Plaza-Bonilla et al., 2015; Trimarco et al., in review, a,b). Conservation tillage has repeatedly been shown to improve both soil health and water quality. The organic matter left on the field in reduced tillage systems may decrease the need for nutrient inputs (Al-Kaisi & Guzman, 2007), limiting both the direct cost and the emissions from chemical fertilizer. Conservation tillage has demonstrated mixed impacts on infiltration (Baumhardt et al., 2012; Dalal, 1989; Deleon et al., 2020), but in semi-arid systems conservation tillage is largely thought to result in long-term improvements in water storage due to improved soil structure and organic matter storage (Plaza-Bonilla et al., 2015). The MT and ST plots studied in this simulation also appear to produce cleaner runoff water from irrigations than their CT counterparts (Trimarco et al., in review b), a notion that is supported by a wealth of literature (Aarstad and Millerr, 1981; Uribe et al., 2018; Deleon et al., 2020). Elucidating the value of clean water is ultimately dependent on the pollutant measured, the risk to receiving waterbodies, and the use of the water, but studies suggest extremely high costs and economic downturn when regional water quality becomes a concern (Bridging the Gap, 2024; Cassidy et al., 2023). Regardless, conservation of water quantity and quality undoubtedly has an economic value and reduced tillage may provide conservation of both in furrow-irrigated systems by improving water storage in water-limited regions and preventing pollution at the source. Indeed, the water quality implications of conservation tillage may be particularly large in furrow-irrigated systems where tillage intensity at the soil surface is critical aspect of both residue management and water quality (Wardle et al., 2015).

The externalities of heavy tillage are not limited to increased carbon emissions, so when carbon markets do not completely address this market failure, we ought to explore tools that internalize more than just carbon emissions into the market. Funding from programs like the NRCS EQIP offer cost-sharing and per-acre payments pay for a much wider variety of conservation practices than those aimed at carbon sequestration. Indeed, the stated objectives of the EQIP program range from water conservation to nutrient management and beyond. It is exceedingly difficult to estimate the social benefits of clean water and healthy soils, but attempts to do so have supported managing for these ecosystem goals (Wallander et al., 2016). Moreover, the cash payments per acre for reduced tillage offered by the EQIP program in Colorado are much closer to the breakeven prices required in our simulations, particularly when those cash payments are frontloaded. Given that EQIP is a federally funded program, it likely faces lower discount rates than private purchasers of carbon offsets and would find upfront payments comparatively less concerning.

We therefore suggest that in semi-arid furrow-irrigated systems, federal grants and frontloaded payment schemes are an optimal compensation scheme compared to carbon crediting. Upfront costs associated with switching to conservation tillage are not adequately addressed in settings where carbon sequestration may be limited due to technical or agronomic challenges. Moreover, an emphasis on carbon sequestration alone in these systems may fail to recognize and value the significant ecosystem service improvements conferred by conservation tillage, particularly those centered around water resource conservation. Government compensation schemes under programs like the NRCS EQIP may provide contractual flexibility to frontload payments, build knowledge networks, and develop technical expertise to improve adoption of conservation tillage. Comparatively, funders aiming to purchase carbon offsets as inexpensively as possible while maintaining higher certainty in sequestration rate may find more cost-effective projects in other geographic regions or in other sectors entirely. The divergence between carbon offset program payments and breakeven prices in this simulation is contrasted by observation that the NRCS EQIP's willingness to pay ~$23 (2025$) per acre per year for reduced tillage approaches the per acre breakeven price under frontloaded payments ($28.19, Table 5.4). Furthermore, the use of federal grants and extension programs is more supportive of network building and dissemination of technical expertise, which may even be a larger barrier than financial incentives in scenarios where a moderate tillage technology (such as strip till) can attain improvements in soil health, water quality, and C sequestration without limiting yield. Regardless of the specific funding source, an approach that focuses on early barriers to adoption and farmer welfare appears to provide better cost-effectiveness and efficiency than traditional carbon offset market schemes.

Therefore, we propose that a more socially beneficial approach to carbon offset payments is to preferentially weigh payments towards the front of the adoption scheme or employ a cost-sharing mechanism. In this scenario, farmers are paid when the risks are highest to lower the barrier to adoption, and payments slowly taper off as the farmer both develops their technical expertise and approaches a more profitable equilibrium. As the project matures, the farmer is likely to find that the payments are not necessary to maintain this system, but to ensure permanence, contracts may be set up such that farmers maintain reduced tillage for at least 10 years. Generally, if it can be assumed that if the long-term profitability of farm operations can be improved under reduced tillage, and as a consequence of decreased inputs, risk can be further reduced, farmers are likely to transition to this new, low-risk, high-or-equivalent-profitability equilibrium.

4 CONCLUSIONS

Based on the findings of Trimarco et al. (in review) and a litany of others, the ecosystem benefits of conservation tillage are numerous. However, in semi-arid furrow-irrigated systems, the limitations on carbon sequestration appear to prevent carbon offsetting to be a sufficient tool to correct market failures of heavy tillage. Extremely high breakeven costs per ton of sequestered CO_{2e} (perhaps brought on my very limited yield in the early years of the study in the MT regime) indicated that carbon offsetting alone is not sufficient to push a farmer to MT from CT. In these scenarios, taking a more holistic view of the benefits conferred by conservation tillage – particularly those that are unique to furrow-irrigated or water-stressed fields – may prove valuable to estimating benefits and costs of converting from conventional to conservation tillage.

Funding schemes that prioritize a broader set of ecosystem services may find value in facilitating conservation tillage contracts, and if they do, they ought to consider various payment schemes to offset the financial burden to farmers. The frontloaded payment schemes to offset farmer costs in this simulation provided significant cost-savings as opposed to flat payments, and the concerns about permanence of practice may be somewhat mitigated by the observation that by roughly 5-6 years after conversion both ST and MT became more profitable than the CT on a year-to-year basis, even without compensation. It is notable that ST not only outperformed MT in terms of measured sequestered C, but furthermore far outperformed the long-term profitability and NPV of CT, even without compensation for practice change and generated the highest overall net benefit. When taking into consideration the observations in the series of studies on these plots quantifying the improvements in soil health and water quality attributable to both MT and ST (Trimarco et al., in review), ST appears to bring the best balance of soil health and water quality improvements, long-term economic benefits, and ease of agronomic management. While the authors recognize the limitations in drawing conclusions from a single case study, these finding support the notion that conservation tillage techniques such as strip tillage do not necessarily represent a sacrifice to environmental health for economic or agronomic gain. Further research should continue to study the broader ecosystem and economic impacts of various tillage techniques, particularly in agroecosystems with unique management challenges such as furrow irrigation. Policy-makers and funders should further continue to explore a diverse set of compensation tools to help address environmental concerns and market failures in a cost-effective and equitable manner, ensuring that the distribution of benefits and costs is carefully considered in policy design and implementation.

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CHAPTER 6 – CONCLUSIONS

As the semi-arid west faces the challenges of agricultural production in a changing climate, sustainable intensification has become increasingly important. We must find a way to produce more food on less land using less resources while securing farmer welfare. The balancing of these goals would be a challenge in any agroecosystem and it is only made more difficult by the climate of the semi-arid west. Extended dry spells and unpredictable precipitation result in large swathes of agricultural land relying on irrigation to meet production goals. The impacts of irrigation contribute significantly to the environmental degradation caused by agriculture in the region. The nutrients and sediment carried off of farm fields through erosion contribute to worsening water quality, an issue that has gained national attention. Concurrently, intensifying drought in arid and semi-arid regions of the world has put increasing pressure on global water resources. The western US is not immune to these concerns, and rapid withdrawals of groundwater threaten the long-term sustainability of the region's surface water and groundwater resources.

Amongst these concerns, farmers in semi-arid western US have increasingly examined ways to improve water conservation – in terms of both quantity and quality – on their fields. These critical BMPs may provide a blueprint for sustainable management, but their broader ecosystem impact necessitates further research. I identified, for example, that while transitioning to sprinkler irrigation eliminated runoff on our fields, our research farm became considerably more saline, posing a threat to productivity.

While the studies I performed occurred on entirely separate fields and with different research goals, they highlight a common theme: the desire to find a more sustainable way to manage soil health and water quality in furrow-irrigated systems. The research presented here represents two divergent paths: one where a farmer changes the method of their irrigation to improve water use efficiency and one where a farmer utilizes conservation tillage to decrease pollutant runoff.

Conversion from furrow to sprinkler irrigation proved fruitful in terms of improving both soil health and water quality. While I did not directly monitor water quality changes on these fields, the change from furrow to sprinkler irrigation inherently reduces runoff concerns as runoff volume is significantly reduced, if not eliminated entirely (as was the case at the private producer's farm). A notable soil health improvement at both sites was the homogenization of soil health across the field. Under furrow irrigation, erosion and deposition dynamics resulted in highly heterogenous soil health across the field. However, when these fields switched to sprinkler irrigation, a notable homogenization was observed, even when average soil health did not increase. Homogenization may be seen as a positive soil health outcome as fields that are more homogenous may require less intense management planning and could receive more consistent inputs with more consistent yields, potentially leading to efficiency gains. However, the transition to sprinkler irrigation is not without its risks. Deficit irrigation has the potential to result in soil salinization, particularly in regions where salinization of water resources is becoming more of a concern. While water quality concerns were addressed on these fields, the continued monitoring of soil health is a critical element of future ecosystem function. Balancing soil health and water quality with agronomic needs is an essential part of sustainable intensification.

The conservation tillage site at Kerbel is illustrative of the complexity in managing soil and water in furrow-irrigated systems. In furrow-irrigation systems, conservation tillage adoption has been sluggish. Tillage is used to not only develop a suitable seedbed for planting, but to manage residue in furrows and ensure proper distribution of irrigation. However, there may still be tillage techniques with decreased intensity that allow farmers to improve soil health and water quality while still managing for these agronomic goals. At the Kerbel Tillage site, I identified significant soil health and water quality improvements under strip tillage and minimum tillage 10 years after converting to conventional tillage. The most notable improvements were to soil biological and physical health, where it appears that decreased tillage intensity resulted in protection of organic matter in soil aggregates, which in turn seemed to lead to improved runoff water quality on conservation tillage plots particularly with respect to particulate constituents (sediment and total Kjeldahl nitrogen). Strip tillage did not appear to be significantly deleterious to soil health or water quality compared to minimum tillage, indicating that if one could gain significant agronomic benefit from strip tillage, it may be advisable to do so.

Fortunately, conservation tillage also has the potential to sequester significant amounts of SOC, enabling farmers to utilize voluntary carbon markets to compensate the upfront costs associated with adoption. However, the economic simulations in this study should cast skepticism that carbon offset markets are a good fit in the semi-arid west. Converting from conventional tillage to minimum tillage required prices for carbon credits that – by orders of magnitude – exceeded current market prices for sequestered carbon. Conversely, strip tillage was found to be more profitable and provide more present value at virtually every decision point compared to conventional tillage without sacrificing soil health or water quality.

One could mistakenly conclude that strip tillage is the "best" regime in every sense, but I urge a more holistic look at a farmer's individual needs and goals. Indeed, strip tillage outperformed minimum tillage in our case study, but the broader takeaway is that current carbon offset schemes may be insufficient to address the barriers to adoption of conservation tillage. Cost-sharing, development of technical expertise, and the matching of financial support to farmer time preference are critical to improving adoption rates. It seems that frontloading payments is an attractive pathway to meet these goals. This may clash with the financial objectives and policy design of carbon offset contracts, but it is a better fit for contracts similar to those presented by federal grants (such as the NRCS EQIP). I argue that if, as a society, we want to help farmers adopt more sustainable practices, we must provide timely financial support and technical expertise to help farmers reach competitive yield and profits in conservation systems. Furthermore, if we value these sustainable practices for ecosystem health improvements beyond carbon sequestration – namely improvements in soil health and water conservation – then agencies that can better measure, validate, and value these benefits can appropriately price them. This may be why programs such as the NRCS EQIP are willing to pay much higher prices for conservation practices than private purchasers of carbon credits; public agencies make decisions based off public benefit whereas private corporations largely aim for the cheapest way to offset carbon emissions. In this author's view, sustainable intensification is possible with the support of robust policies to support land manger decision-making and reduce barriers to adoption.

We must continue to explore the broader ecosystem health and socioeconomic impacts of conservation agriculture across various agroecosystems. Approaches that consider not only soil health or water conservation, but how these aspects of ecosystem health interact with each other and with farmer decision-making is tantamount to future sustainability goals. Managing for healthy soils and clean water while improving or maintain productivity is possible, even in semi-arid systems, but we must pay careful attention to the tradeoffs of each management decision. Doing so will require diligent monitoring, technical support, and adaptation. Policies and compensation schemes that put the farmer at the forefront may provide a more equitable and resilient pathway towards regional sustainability while helping society meet its environmental goals. Research and policy development should center on farmer decision-making and incorporate regional context. We strongly urge researchers, community leaders, and policy-makers to continue to study how to achieve and scale up sustainable management in water-limited environments. Sustainable and resilient soil management is indeed in our future, so long as work tirelessly to make it a reality.

APPENDIX

Table S1. Field Operations for the 2021 and 2022 Field Seasons by Treatment.

2021 Field Season Operations			
Date	**Conventional Till**	**Minimum Till**	**Strip Till**
10/22/2020	Verti-Till (7.5 cm)	Verti-Till (7.5 cm)	Verti-Till (7.5 cm)
3/8/2021	Verti-Till (7.5 cm)	Verti-Till (7.5 cm)	Verti-Till (7.5 cm)
3/9/2021	Moldboard Plow (30 cm)		
3/11/2021	Mulch		
3/11/2021	Landplane		
3/12/2021	Mulch		
4/7/2021	Mulch		
4/8/2021	Spread dry fertilizer		
4/9/2021	Condition seedbeds	Condition seedbeds	Condition seedbeds
4/9/2021	Cultivate	Cultivate	
4/12/2021	Cultipack		
4/13/2021	Cultipack	Spread dry fertilizer	Cultivate
4/13/2021			Strip Till (25 cm) + Band liquid fertilizer
4/27/2021	Plant corn	Plant corn	Plant corn
5/25/2021	Spray pesticides	Spray pesticides	Spray pesticides
6/10/2021	Spray micronutrients	Spray micronutrients	Spray micronutrients
6/18/2021	Cultivate and clean rows	Cultivate and clean rows	Cultivate and clean rows
6/18/2021		Side-dress liquid fertilizer	Side-dress liquid fertilizer
7/23/2021	Apply miticide and fertilizer	Apply miticide and fertilizer	Apply miticide and fertilizer

10/22/2021	Harvest grain	Harvest grain	Harvest grain
10/29/2021	Shred and windrow	Shred and windrow	Shred and windrow
10/29/2021	Chop and bale residue	Chop and bale residue	Chop and bale residue
Total Operations	20	14	14
2022 Field Season Operations			
Date	**Conventional Till**	**Minimum Till**	**Strip Till**
11/23/2021	Verti-Till (7.5 cm)	Verti-Till (7.5 cm)	Verti-Till (7.5 cm)
11/24/2021	Verti-Till (7.5 cm)		
11/29/2021	Subsoiler (25 cm)	Subsoiler (25 cm)	Subsoiler (25 cm)
11/30/2021	Moldboard plow (30 cm)		
11/30/2021	Mulch		
12/3/2021	Landplane		
12/3/2021	Mulch		
4/11/2022	Cultivate	Verti-Till (7.5 cm)	Verti-Till (7.5 cm)
4/11/2022			Strip till (25 cm)
4/11/2022	Condition seedbeds	Condition seedbeds	Condition seedbeds
4/11/2022	Spread dry fertilizer	Spread dry fertilizer	Spread dry fertilizer
4/12/2022	Cultipack	Cultipack	Cultipack
4/29/2022	Plant corn	Plant corn	Plant corn
4/29/2022	Apply Optistart	Apply Optistart	Apply Optistart
6/7/2022	Spray pesticide	Spray pesticide	Spray pesticide
6/13/2022	Cultivate and clean rows	Cultivate and clean rows	Cultivate and clean rows
6/13/2022	Apply liquid fertilizer	Apply liquid fertilizer	Apply liquid fertilizer
10/24/2022	Harvest grain	Harvest grain	Harvest grain
11/2/2022	Shred and windrow	Shred and windrow	Shred and windrow

11/2/2022	Chop and bale residue	Chop and bale residue	Chop and bale residue
Total Operations	19	14	15

Supplementary Table S1 adapted from Trimarco et al. (in review). Primary tillage operations include tillage depth in parantheses. Other operations (i.e., mulching, seedbed conditioning, cultivating, cultipacking, row cleaning, and landplaning) occur primarily at the surface or with shallower disturbance than the primary tillage operation.

*Verti-till uses shallow-angled flat disks to chop crop residue and decrease compaction while maintain surface residue coverage.

**Subsoiler or deep ripper is used to break deep compaction without the full inversion of moldboard plowing.

***Mulching breaks clods and smooths the soil surface.

****Cultipacking breaks clods and smooths the soil surface.

*****Strip tillage involves a band of tillage in the seedbed using coulters, a ripper shank, and a roller (Deleon et al., 2020; Nowatzki et al., 2017; Wardle et al., 2015).

Table S3. Crop Planted by Year

Year	Crop
2011	Grain corn (*Zea mays*)
2012	Grain corn (*Zea mays*)
2013	Silage corn (*Zea mays*)
2014	Grain corn (*Zea mays*)
2015	Grain corn (*Zea mays*)
2016	Barley (*Hordeum vulgare*)
2017	Silage corn (*Zea mays*)
2018	Dry beans (*Phaseolus vulgaris*)
2019	Winter wheat (*Triticum aestivum)*
2020	Grain corn (*Zea mays*)
2021	Grain corn (*Zea mays*)
2022	Grain corn (*Zea mays*)
2023	Silage corn (*Zea mays*)

Table S2. Inflow Water Quality Concentration by Year (mg L^{-1} Mean ± SE)

	TSS	Se	TKN	NO_3-N	TN	TP
2021	29.8 ± 2.7	0.0055 ± 0.0001	0.87 ± 0.00	8.0 ± 0.1	8.8 ± 0.1	0.054 ± 0.003
2022	36.0 ± 2.7	0.0060 ± 0.0002	0.87 ± 0.00	6.6 ± 0.2	7.5 ± 0.2	0.098 ± 0.028

Table S4. Costs and Benefits of Production by Tillage Regime – 2011 to 2023

Year	Machinery Operation Costs	Fertilizer Costs	Pesticide + Seed Costs	Total Costs	Yield Benefits	Profit
Conventional Tillage						
2011	$190,368	$80,688	$75,063	-$346,119	$745,237	$399,118
2012	$169,076	$85,553	$78,606	-$333,235	$1,012,832	$679,597
2013	$267,028	$56,410	$85,515	-$408,954	$782,453	$373,499
2014	$142,449	$14,691	$68,649	-$225,788	$437,136	$211,348
2015	$166,163	$63,129	$74,313	-$303,604	$557,296	$253,692
2016	$99,671	$20,979	$32,080	-$152,731	$632,062	$479,332
2017	$281,427	$47,718	$77,235	-$406,379	$980,335	$573,956
2018	$103,892	$49,377	$60,279	-$213,548	$0	-$213,548
2019	$120,685	$0	$28,012	-$148,697	$500,642	$351,944
2020	$121,343	$49,310	$75,172	-$245,825	$530,170	$284,346
2021	$181,047	$51,817	$75,891	-$308,755	$761,866	$453,112
2022	$153,489	$106,552	$90,792	-$350,832	$687,881	$337,049
2023	$224,237	$141,217	$98,822	-$464,276	$594,500	$130,224
Total	$2,220,875	$767,440	$920,428	-$3,908,743	$8,222,412	$4,313,669
Minimum Tillage - Scenario #1 - No Extra Costs						
Year	Machinery Operation Costs	Fertilizer Costs	Pesticide + Seed Costs	Total Costs	Yield Benefits	Profit
2011	$121,830	$80,688	$75,063	-$277,580	$740,879	$463,299
2012	$108,595	$85,553	$78,606	-$272,754	$775,526	$502,772
2013	$197,887	$56,410	$85,515	-$339,812	$653,876	$314,064
2014	$110,789	$14,691	$68,649	-$194,128	$404,624	$210,496
2015	$110,147	$55,183	$74,313	-$239,643	$538,173	$298,531
2016	$96,312	$20,979	$32,080	-$149,371	$633,330	$483,960
2017	$254,179	$52,308	$77,235	-$383,721	$952,366	$568,645
2018	$81,205	$59,354	$60,279	-$200,838	$0	-$200,838
2019	$68,108	$0	$28,012	-$96,120	$477,912	$381,793
2020	$96,142	$49,310	$75,172	-$220,624	$496,596	$275,973
2021	$120,729	$56,470	$75,891	-$253,090	$837,498	$584,408
2022	$128,496	$106,552	$90,792	-$325,840	$710,435	$384,595
2023	$187,837	$141,217	$98,822	-$427,876	$606,100	$178,224
Total	$1,682,254	$778,715	$920,428	-$3,381,396	$7,827,316	$4,445,921
Strip Tillage - Scenario #1 - No Extra Costs						
Year	Machinery Operation Costs	Fertilizer Costs	Pesticide + Seed Costs	Total Costs	Yield Benefits	Profit

2011	$130,382	$80,688	$75,063	-$286,133	$788,818	$502,685
2012	$123,963	$85,553	$78,606	-$288,122	$966,349	$678,227
2013	$227,328	$56,410	$85,515	-$369,253	$763,548	$394,295
2014	$115,141	$14,691	$68,649	-$198,480	$400,149	$201,669
2015	$102,644	$55,183	$74,313	-$232,140	$536,808	$304,667
2016	$98,851	$20,979	$32,080	-$151,910	$581,508	$429,598
2017	$263,366	$52,308	$77,235	-$392,909	$1,033,478	$640,569
2018	$76,052	$59,354	$60,279	-$195,685	$0	-$195,685
2019	$71,809	$0	$28,012	-$99,821	$482,969	$383,148
2020	$99,467	$49,310	$75,172	-$223,948	$476,601	$252,653
2021	$112,926	$71,234	$75,891	-$260,050	$831,404	$571,354
2022	$131,515	$106,552	$90,792	-$328,859	$697,279	$368,420
2023	$178,840	$147,744	$98,822	-$425,406	$572,750	$147,344
Total	$1,732,284	$800,005	$920,428	-$3,452,716	$8,131,661	$4,678,944

Minimum Tillage - Scenario #2 - Paying Off Extra Machinery Costs						
Year	Machinery Operation Costs	Fertilizer Costs	Pesticide + Seed Costs	Total Costs	Yield Benefits	Profit
2011	$129,262	$80,688	$75,063	-$285,013	$740,879	$455,866
2012	$114,149	$85,553	$78,606	-$278,309	$775,526	$497,217
2013	$202,871	$56,410	$85,515	-$344,796	$653,876	$309,081
2014	$115,387	$14,691	$68,649	-$198,726	$404,624	$205,898
2015	$114,444	$55,183	$74,313	-$243,940	$538,173	$294,233
2016	$100,368	$20,979	$32,080	-$153,427	$633,330	$479,904
2017	$258,035	$52,308	$77,235	-$387,578	$952,366	$564,788
2018	$84,900	$59,354	$60,279	-$204,532	$0	-$204,532
2019	$71,678	$0	$28,012	-$99,691	$477,912	$378,222
2020	$99,633	$49,310	$75,172	-$224,114	$496,596	$272,482
2021	$124,215	$56,470	$75,891	-$256,576	$837,498	$580,921
2022	$132,132	$106,552	$90,792	-$329,476	$710,435	$380,959
2023	$192,212	$141,217	$98,822	-$432,251	$606,100	$173,849
Total	$1,739,286	$778,715	$920,428	-$3,438,429	$7,827,316	$4,388,888

Strip Tillage - Scenario #2 - Paying Off Extra Machinery Costs						
Year	Machinery Operation Costs	Fertilizer Costs	Pesticide + Seed Costs	Total Costs	Yield Benefits	Profit
2011	$137,815	$80,688	$75,063	-$293,566	$788,818	$495,253
2012	$129,518	$85,553	$78,606	-$293,677	$966,349	$672,672
2013	$232,312	$56,410	$85,515	-$374,237	$763,548	$389,312
2014	$119,739	$14,691	$68,649	-$203,078	$400,149	$197,071
2015	$106,942	$55,183	$74,313	-$236,438	$536,808	$300,370
2016	$102,907	$20,979	$32,080	-$155,966	$581,508	$425,542

2017	$267,223	$52,308	$77,235	-$396,765	$1,033,478	$636,712
2018	$79,747	$59,354	$60,279	-$199,379	$0	-$199,379
2019	$75,380	$0	$28,012	-$103,392	$482,969	$379,577
2020	$102,957	$49,310	$75,172	-$227,439	$476,601	$249,162
2021	$116,413	$71,234	$75,891	-$263,537	$831,404	$567,867
2022	$135,151	$106,552	$90,792	-$332,495	$697,279	$364,784
2023	$183,215	$147,744	$98,822	-$429,781	$572,750	$142,969
Total	$1,789,317	$800,005	$920,428	-$3,509,749	$8,131,661	$4,621,912

Table S5. Field Operations Used in Economic Models and COMET-Farm – 2011 to 2023

Date	Conventional Tillage	Strip Tillage	Minimum Tillage
1/1/2011		Strip Till	
1/1/2011	Deep Ripper		
1/1/2011	Moldboard Plow		
3/3/2011	Harrow		
3/3/2011	Harrow		
3/15/2011	Landplane		
3/15/2011	Landplane		
4/2/2011	Ditch		
4/5/2011	Cultipack		
5/4/2011	Plant	Plant	Plant
5/4/2011	Fertilizer Application	Fertilizer Application	Fertilizer Application
5/9/2011			Clean Rows
5/16/2011	Pesticide Application	Pesticide Application	Pesticide Application
6/23/2011	Fertilizer Application	Fertilizer Application	Fertilizer Application
6/27/2011	Cultivate	Cultivate	Cultivate
11/15/2011	Harvest	Harvest	Harvest
11/29/2011	Chop and Windrow	Chop and Windrow	Chop and Windrow
11/29/2011	Bail	Bail	Bail
11/30/2011	Disc		
3/5/2012	Moldboard Plow		
3/8/2012	Harrow		

3/8/2012	Harrow		
3/14/2012	Landplane		
3/14/2012	Landplane		
3/14/2012		Clean Furrows	
3/16/2012		Strip Till	
4/13/2012	Ditch		
4/13/2012	Condition Seedbeds		
5/1/2012			Clean Rows
5/3/2012	Plant	Plant	Plant
5/3/2012	Fertilizer Application	Fertilizer Application	Fertilizer Application
5/10/2012	Pack Furrows		
5/10/2012	Pesticide Application	Pesticide Application	Pesticide Application
6/4/2012	Pack Furrows (Pipe)	Pack Furrows (Pipe)	
6/11/2012		Clean Rows	Clean Rows
6/20/2012	Pesticide Application	Pesticide Application	Pesticide Application
6/26/2012	Fertilizer Application	Fertilizer Application	Fertilizer Application
6/26/2012	Cultivate	Cultivate	Cultivate
11/1/2012	Harvest	Harvest	Harvest
11/15/2012	Chop and Windrow	Chop and Windrow	Chop and Windrow
11/15/2012	Bail	Bail	Bail
12/8/2012	Moldboard Plow		
12/8/2012		Strip Till	
3/11/2013	Landplane		
3/29/2013	Condition Seedbeds	Condition Seedbeds	Condition Seedbeds
3/29/2013	Cultipack	Cultipack	Cultipack
5/13/2013	Fertilizer Application	Fertilizer Application	Fertilizer Application
5/14/2013	Plant Silage Corn	Plant Silage Corn	Plant Silage Corn
6/2/2013	Pesticide Application	Pesticide Application	Pesticide Application
6/19/2013	Fertilizer Application/Cultivate	Fertilizer Application/Cultivate	Fertilizer Application/Cultivate
7/24/2013	Pesticide Application	Pesticide Application	Pesticide Application
9/30/2013	Harvest Silage Corn	Harvest Silage Corn	Harvest Silage Corn
10/2/2013	Vertical Disc		

10/9/2013	Moldboard Plow		
11/18/2013	Cultimulch		
3/20/2014	Cultimulch		
3/27/2014	Landplane		
4/3/2014	Vertical Ripper	Vertical Ripper	Vertical Ripper
4/11/2014	Cultimulch		
4/11/2014	Seedbed Conditioner		
4/11/2014	Cultimulch		
5/6/2014	Plant Grain Corn	Plant Grain Corn	Plant Grain Corn
6/3/2014	Pesticide Application	Pesticide Application	Pesticide Application
6/25/2014	Cultivate	Cultivate	Cultivate
12/2/2014	Harvest Grain Corn	Harvest Grain Corn	Harvest Grain Corn
12/2/2014	Shred Stalks	Shred Stalks	Shred Stalks
12/2/2014	Bale	Bale	Bale
3/23/2015	Disc		
3/23/2015			Vertical Till
3/23/2015	Moldboard Plow		
3/27/2015	Cultimulch	Cultimulch	Cultimulch
4/13/2015		"Orthoman 1 tripper (A)" - Strip Till	"Orthoman 1 tripper (A)" - Strip Till
4/13/2015		Fertilizer Application w/strip till	Fertilizer Application w/ strip till
4/14/2015	Seedbed condition		
4/14/2015	Cultimulch		
4/14/2015	Fertilizer Application		
4/30/2015	Plant Grain Corn	Plant Grain Corn	Plant Grain Corn
6/9/2015	Pesticide Application	Pesticide Application	Pesticide Application
6/24/2015	Cultivate	Cultivate	Cultivate
6/24/2015		Pipe Ditch	Pipe Ditch
11/16/2015	Harvest Grain Corn	Harvest Grain Corn	Harvest Grain Corn
11/23/2015	Shred Stalks	Shred Stalks	Shred Stalks
11/23/2015	Bale Stalks	Bale Stalks	Bale Stalks
11/25/215	Disc		
12/11/2015	Deep Ripper	Deep Ripper	Deep Ripper
12/11/2015	Vertical Till	Vertical Till	Vertical Till
12/11/2015			Vertical Till
12/11/2015	Moldboard Plow		

2/22/2016	Cultimulch		
2/24/2016	Landplane		
3/1/2016	Fertilizer Application	Strip Till + Fertilizer Application*	Fertilizer Application
3/2/2016	Condition Seedbeds	Condition Seedbeds	Condition Seedbeds
3/2/2016	Cultimulch		
3/2/2016	Cultipack	Cultipack	Cultipack
3/3/2016	Plant Barley	Plant Barley	Plant Barley
5/5/2016	Pesticide Application	Pesticide Application	Pesticide Application
7/25/2016	Harvest Barley	Harvest Barley	Harvest Barley
7/26/2016	Bale Straw	Bale Straw	Bale Straw
9/2/2016	Pesticide Application	Pesticide Application	Pesticide Application
9/3/2016	Pesticide Application	Pesticide Application	Pesticide Application
3/10/2017	Moldboard Plow		
3/10/2017	Cultimulch		
3/14/2017	Cultimulch		
3/14/2017	Landplane		
4/10/2017	Cultimulch w/ Pre-plant Fertilizer		
4/10/2017		Pre-Plant Fertilizer	Pre-Plant Fertilizer
4/13/2017		Strip Till + Fert	Strip Till + Fert
4/13/2017	Condition Seedbeds	Condition Seedbeds	Condition Seedbeds
4/13/2017	Cultimulch		
4/14/2017			Cultimulch
4/19/2017	Plant Silage Corn	Plant Silage Corn	Plant Silage Corn
6/21/2017		Sidedress Fert	Sidedress Fert
6/21/2017	Cultivate	Cultivate	Cultivate
10/13/2017	Harvest Silage Corn	Harvest Silage Corn	Harvest Silage Corn
11/21/2017	Vertical Till		
3/23/2018	Moldboard Plow		
3/26/2018	Cultimulch		
4/3/2018	Cultimulch		
4/3/2018	Landplane		
4/18/2018	Cultimulch		
5/1/2018	Fertilizer Application		
5/15/2018	Cultivate	Cultivate	Cultivate

5/15/2018	Pesticide Application	Pesticide Application	Pesticide Application
5/17/2018	Corrugator #1	Corrugator #1	Corrugator #1
5/18/2018		Strip Till + Fertilize	Strip Till + Fertilize
5/18/2018		Cultipack	Cultipack*
5/18/2018	Plant Dry Beans	Plant Dry Beans	Plant Dry Beans
6/25/2018	Cultivate	Cultivate	Cultivate
6/26/2018	Pipe Ditch	Pipe Ditch	Pipe Ditch
7/17/2018	Cultivate	Cultivate	Cultivate
8/30/2018	Knife Dry Beans	Knife Dry Beans	Knife Dry Beans
9/4/2018	Harvest Dry Beans	Harvest Dry Beans	Harvest Dry Beans
9/7/2018	Vertical Till	Vertical Till	Vertical Till
9/7/2018	Vertical Till	Vertical Till	Vertical Till
9/10/2018	Severe Hail Damage	Severe Hail Damage	Severe Hail Damage
9/14/2018	Cultimulch	Cultimulch	Cultimulch
9/14/2018	Condition Seedbeds	Condition Seedbeds	Condition Seedbeds
9/14/2018	Cultipack	Cultipack	Cultipack
9/14/2018	Fertilizer Application	Fertilizer Application	Fertilizer Application
9/15/2018	Drill Wheat	Drill Wheat	Drill Wheat
4/24/2019	Pesticide Application	Pesticide Application	Pesticide Application
5/31/2019	Pesticide Application	Pesticide Application	Pesticide Application
6/26/2019	Pesticide Application*	Pesticide Application	Pesticide Application
6/27/2019	Pesticide Application*	Pesticide Application	Pesticide Application
7/18/2019	Pesticide Application	Pesticide Application	Pesticide Application
7/19/2019	Pesticide Application	Pesticide Application	Pesticide Application
7/23/2019	Pesticide Application	Pesticide Application	Pesticide Application
8/2/2019	Harvest Wheat	Harvest Wheat	Harvest Wheat
8/3/2019	Chop + Bale Straw	Chop + Bale Straw	Chop + Bale Straw
10/7/2019	Pesticide Application	Pesticide Application	Pesticide Application
11/8/2019	Vertical Till		
11/20/2019	Subsoil Rip		
11/23/2019	Moldboard Plow		

11/24/2019	Cultimulch		
11/25/2019	Landplane		
4/1/2020		Strip Till	
4/1/2020			Row Clean
4/1/2020	Fertilizer Application	Fertilizer Application	Fertilizer Application
4/9/2020	Condition Seedbeds		
4/9/2020	Cultipack		
4/28/2020	Plant Grain Corn	Plant Grain Corn	Plant Grain Corn
4/30/2020	Pesticide Application	Pesticide Application	Pesticide Application
6/2/2020	Pesticide Application	Pesticide Application	Pesticide Application
6/17/2020	Pesticide Application	Pesticide Application	Pesticide Application
6/18/2020		Cultivate	Cultivate
6/19/2020	Pipe Ditcher	Pipe Ditcher	Pipe Ditcher
10/9/2020	Harvest Grain Corn	Harvest Grain Corn	Harvest Grain Corn
10/21/2020	Shred Stalks	Shred Stalks	Shred Stalks
10/21/2020	Bale	Bale	Bale
10/22/2020	Vertical Till	Vertical Till	Vertical Till
3/8/2021	Vertical Till	Vertical Till	Vertical Till
3/9/2021	Moldboard Plow		
3/11/2021	Cultimulch		
3/11/2021	Landplane		
3/12/2021	Cultimulch		
4/7/2021	Cultimulch		
4/8/2021	Fertilizer Application		
4/9/2021	Condition Seedbeds	Condition Seedbeds	Condition Seedbeds
4/9/2021	Cultivate		Cultivate
4/12/2021	Cultipack		
4/13/2021	Cultipack		
4/13/2021			Fertilizer Application
4/13/2021		Condition Seedbeds	
4/13/2021		Strip Till + Fertilizer Application	
4/27/2021	Plant Grain Corn	Plant Grain Corn	Plant Grain Corn
5/25/2021	Pesticide Application	Pesticide Application	Pesticide Application

6/10/2021	Fertilizer Application	Fertilizer Application	Fertilizer Application
6/18/2021	Cultivate	Cultivate	Cultivate
6/18/2021	Clean Furrows	Clean Furrows	Clean Furrows
6/18/2021		Fertilizer Application	Fertilizer Application
7/23/2021	Pesticide and Fertilizer Application	Pesticide and Fertilizer Application	Pesticide and Fertilizer Application
10/22/2021	Harvest Grain	Harvest Grain	Harvest Grain
10/29/2021	Shred and Chop	Shred and Chop	Shred and Chop
10/29/2021	Bale	Bale	Bale
11/23/2021	Vertical Till	Vertical Till	Vertical Till
11/25/2021	Vertical Till		
11/29/2021	Subsoil Rip	Subsoil Rip	Subsoil Rip
11/30/2021	Moldboard Plow		
11/30/2021	Cultimulch		
12/3/2021	Landplane		
12/3/2021	Cultimulch		
4/11/2022	Cultivate		
4/11/2022		Vertical Till	Vertical Till
4/11/2022		Strip Till	
4/11/2022	Condition Seedbeds	Condition Seedbeds	Condition Seedbeds
4/11/2022	Fertilizer Application	Fertilizer Application	Fertilizer Application
4/12/2022	Cultipack	Cultipack	Cultipack
4/29/2022	Plant Grain Corn w/ Optistart Gold	Plant Grain Corn w/ Optistart Gold	Plant Grain Corn w/ Optistart Gold
6/7/2022	Pesticide Application	Pesticide Application	Pesticide Application
6/13/2022	Cultivate	Cultivate	Cultivate
6/13/2022	Clean Furrows	Clean Furrows	Clean Furrows
6/13/2022	Fertilizer Application	Fertilizer Application	Fertilizer Application
10/24/2022	Harvest Grain	Harvest Grain	Harvest Grain
11/2/2022	Shred and Windrow	Shred and Windrow	Shred and Windrow
11/2/2022	Chop and Bale	Chop and Bale	Chop and Bale
11/10/2022	Moldboard Plow		
11/10/2022	Vertical Till	Vertical Till	Vertical Till
11/11/2022	Cultimulch		

3/16/2023	Cultimulch		
3/20/2023	Landplane		
4/13/2023	Cultimulch		
4/13/2023	Condition Seedbeds		Condition Seedbeds
4/13/2023	Fertilizer Application		Fertilizer Application
4/14/2023	Cultipack		Cultipack
4/14/2023		Strip Till + Fertilizer Application	
5/2/2023	Fertilizer Application	Fertilizer Application	Fertilizer Application
5/2/2023	Plant Silage Corn	Plant Silage Corn	Plant Silage Corn
5/31/2023	Pesticide Application	Pesticide Application	Pesticide Application
6/26/2023	Fertilizer Application	Fertilizer Application	Fertilizer Application
6/28/2023	Cultivate	Cultivate	Cultivate
6/28/2023	Pipe Ditch	Pipe Ditch	Pipe Ditch
9/8/2023	Harvest Silage Corn	Harvest Silage Corn	Harvest Silage Corn
9/12/2023	Vertical Till	Vertical Till	Vertical Till
9/12/2023	Fertilizer Application	Fertilizer Application	Fertilizer Application
9/13/2023	Condition Seedbeds	Condition Seedbeds	Condition Seedbeds
9/13/2023	Cultipack	Cultipack	Cultipack

ABBREVIATIONS

ANOVA = analysis of variance; ARDEC = Agricultural Research Development and Education Center (at Colorado State University); AVRC = Arkansas Valley Research Center; BG = β-glucosidase activity; BMP = best management practice; CSP = Conservation Stewardship Program; CT = conventional till; EC = electrical conductivity; K = potassium; MAOM-C = mineral associated organic matter C; MBC = microbial biomass carbon; MT = minimum till; NPV = net present value; NO_3-N = Nitrate-N; NRCS EQIP = Natural Resources Conservation Service Environmental Quality Incentives Program; P = phosphorous; P_b = bulk density; PNG = ρ-nitrophenyl-β-glucopyranoside; PMN = potentially mineralizable nitrogen; POM-C = particulate organic matter C; Se = Selenium; SHI = Soil Health Index; SMAF = Soil Management Assessment Framework; SOC = soil organic carbon; ST = strip till; THAM = TRIS (hydroxymethyl) aminomethane; TKN = total Kjeldahl N; TP = total phosphorous; TSS = total suspended solids; USDA = United States Department of Agriculture; VCM = voluntary carbon market; WSA = water stable aggregates.