DISSERTATION

THE INTENSIFICATION REVOLUTION IN DRYLAND CROPPING SYSTEMS: IMPLICATIONS FROM FIELD TO LANDSCAPE SCALE

Submitted by

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ABSTRACT

THE INTENSIFICATION REVOLUTION IN DRYLAND CROPPING SYSTEMS: IMPLICATIONS FROM FIELD TO LANDSCAPE SCALE

A global transformation in semi-arid cropping systems is occurring as dryland (non-irrigated) farmers shift from crop rotations reliant on year-long periods of bare fallow to more intensively cropped systems. Bare fallow has reduced year-to-year variability in crop yields, but it has also constrained crop productivity and, therefore, reduced carbon (C) inputs to soils. Exposure to tillage and erosion, combined with C limitation, has gradually degraded dryland soils and reduced their capacity to capture water and supply plant nutrients, requiring dryland farmers to rely on external inputs to support plant growth. However, the emergence of no-till has enabled dryland farmers to save enough water to replace bare fallows with crops, a practice called cropping system intensification. Cropping intensification has potential implications for the environment and economy of dryland agriculture as it impacts every aspect of the agroecosystem – from soil health, to weed and nutrient management, to crop yields. This dissertation seeks to unravel the economic and environmental implications of cropping system intensification at both the field and landscape scale in the US High Plains, and to understand the social dynamics underpinning this revolution.

I quantified the impacts of cropping system intensification on a range of soil health parameters on 96 dryland, no-till fields in the High Plains. Three levels of cropping system intensity – wheat-fallow, mid-intensity, and continuous – were represented along a potential evapotranspiration gradient that increases from northwestern Nebraska to southeastern Colorado.

I conducted in-depth interviews with farmers to examine the motivations, perceptions, and social interactions that influence decisions about whether and how much to intensify, and to collect detailed field histories including input use and crop yields. To scale up the implications of these field-level analyses, and to assess the current extent of the cropping revolution in the High Plains, I conducted a spatial analysis using high-resolution satellite crop data to examine changes in cropping patterns over time at the landscape scale.

I found that cropping system intensification was positively associated with soil organic carbon, aggregation, and fungal biomass, and these effects were robust amidst variability in environmental and management factors. I also found that intensified systems were associated with greater potentially mineralizable and total nitrogen (N), and arbuscular mycorrhizal fungal colonization of wheat roots, suggesting that cropping intensity enhances internal cycling of N and phosphorus (P). Continuous dryland farmers also achieved greater total crop production using fewer external inputs than wheat-fallow farmers, leading to enhanced profitability. To explain the social dynamics underpinning the cropping system revolution, I build on Carolan's application of Bourdieusian social fields to agriculture, and find several overlapping fields within Carolan's more general fields of sustainable and conventional agriculture, which are reflected in different degrees of intensification. I identify strategies for change, some of which would serve to reshape social fields, and others which leverage existing social positions and relationships, to enable farmers to overcome the barriers constraining cropping system intensification. Results from the spatial analysis suggest that, from 2008 to 2016, the High Plains witnessed a profound shift in cropping systems, as the historically dominant wheat-fallow system was replaced by intensified rotations as the dominant systems across the landscape. I estimated that these patterns over the 9-year study period increased annual grain production and annual net farm operating income, slightly reduced herbicide use, and increased C sequestration, contributing to greenhouse gas reductions. I projected each of these implications to a scenario of 100% continuous cropping adoption to estimate the potential environmental and economic impacts of cropping system intensification in the High Plains. Overall, my findings suggest that dryland cropping systems are gradually intensifying in the High Plains, and these trends are likely reversing historical negative environmental and economic trends to enhance the profitability and environmental sustainability of dryland agroecosystems.

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CHAPTER 1: INTRODUCTION

Dryland agriculture, by definition, requires farmers to cope with the uncertainty and scarcity of water. Bare fallow periods have been the primary strategy by which dryland (non-irrigated) farmers have stabilized crop yields for decades or centuries in semi-arid regions, which constitute 20% of Earth's land area (Bot et al., 2000; Koohafkan and Stewart, 2008). However, while bare fallows minimized year-to-year variability in crop yields by storing water (Greb, 1979), they also constrained overall crop production by requiring two whole years to grow a single crop (Peterson et al., 1996). Low crop productivity and diversity in crop-fallow systems have reduced soil carbon (C) by almost 60% by limiting C inputs (Bowman et al., 1990; Peterson et al., 2004), contributing to degraded soils with a hampered ability to properly store water and supply nutrients to plants (Grant et al., 2002; Shaver et al., 2002; Sherrod et al., 2003). As a result, dryland farmers today are heavily reliant on chemical sources of nutrients to support crop growth and herbicides to control weeds (Anderson, 2003). Ultimately, the dependence on costly inputs, combined with years of low commodity prices, has forced dryland farmers into a crisis of profitability. Net farm income has dropped 50% since 2013 (USDA-ERS, 2017), leading to the decline of full-time farmers and expanding farm sizes (Carolan, 2016). Meanwhile, rising consumption of herbicides and fertilizers continues to exact high environmental and human costs, evidenced by oceanic dead zones (Diaz and Rosenberg, 2008), groundwater contamination (Sebilo et al., 2013), and degradation of natural ecosystems due to atmospheric N deposition and pesticide pollution (Du et al., 2014; Simkin et al., 2016).

The story told in this dissertation is that of the effort to reverse these negative economic and environmental trends in the US High Plains, an important grain-growing region including the states of Colorado, Kansas, and Nebraska. The emergence of no-till in the 1980s enabled dryland farmers to replace year-long bare fallows (summer fallow) with a crop, a practice called cropping system intensification (Peterson et al., 1996). Since then, a cropping system “revolution” has taken place in semi-arid regions around the world, resulting in the intensification of tens of millions of dryland crop acres (Smith and Young, 2000; Maaz et al., 2017). This dissertation seeks to understand the implications of this revolution in the High Plains, one of the semi-arid regions where this revolution is taking place.

The dissertation begins with a study of the mechanisms of soil organic carbon (SOC) accrual along a gradient of cropping system intensity. Building SOC is a critical but daunting challenge in semi-arid agroecosystems. Greater SOC can enhance soil functions like nutrient provision and retention, substrate provision for biodiversity, and erosion control (Wall, 2012), but building SOC in dryland systems is constrained by high erosion rates and low C inputs (Plaza-Bonilla et al., 2015). For dryland farmers, low levels of SOC and aggregation exacerbate the risks of farming in a water-limited environment – risks that will compound with climate change (Ko et al., 2012; USGCRP, 2014). The objective of Chapter 2 is to quantify the potential for cropping system intensification to enhance SOC storage and aggregation relative to traditional crop-fallow systems and to evaluate the relative effects of management, texture, and climate on SOC levels. I hypothesized that SOC would increase with cropping system intensity directly through greater C inputs, and indirectly through greater fungal biomass and aggregation, and that these effects would be robust across a range of environmental and management contexts.

The third chapter examines whether or not cropping system intensity represents an ecologically-based weed and nutrient management strategy, and quantifies cropping system impacts on crop yields, input use, and profitability. Previous studies have suggested that intensifying from the traditional winter wheat (*Triticum aestivum*)-fallow system to a mid-intensity system like wheat-corn (*Zea mays*)-fallow increases the nitrogen (N) requirement by 44% (Kolberg et al., 1996), and that nutrient requirements may be even greater for continuous cropping systems that have eliminated summer fallow altogether (Grant et al., 2002). However, cropping system intensification also offers the opportunity to recouple C and N cycles by increasing crop production and residue inputs to soil, which may limit N losses to the environment and enable farmers to reduce fertilizer inputs (Drinkwater and Snapp, 2007; Gardner and Drinkwater; 2009). Furthermore, cropping systems reliant on long bare fallows are subject to "long fallow syndrome," wherein populations of arbuscular mycorrhizal fungi (AMF) are drastically reduced in the absence of a host plant (Thompson, 1987; Harinikumar and Bagyaraj, 1988). AMF have the potential to increase their host plants' access to immobile nutrients like phosphorus (P) (Al-Karaki et al., 2004), and thus reducing summer fallow through cropping system intensification may represent a means to increase plant P uptake with fewer P inputs. Additionally, through greater plant competition with weeds and the ability to rotate different chemicals, cropping system intensification may enable no-till farmers to control weeds using fewer herbicides (Derksen et al., 2002; Anderson, 2009; Nichols et al., 2015). The objectives of this chapter are to 1) Quantify cropping system intensity effects on microbially-mediated nutrient cycling, specifically AMF colonization of wheat roots and potentially mineralizable nitrogen (PMN), and 2) Quantify cropping system effects on herbicide and fertilizer use and crop yields, and estimate implications for profitability on working dryland farms. I hypothesized that intensification would increase the nutrient supply capacity of soil, enabling intensified farmers to achieve greater crop production while using fewer inputs, increasing profitability.

The fourth chapter presents a sociological analysis with the objective of identifying the social dynamics and barriers to adoption influencing the rate and extent of cropping system intensity on the landscape. In the High Plains, transitions to intensified cropping systems have been slow relative to other regions, and cropping systems have stratified into varying degrees of intensity (Smith and Young, 2000; Cochran et al., 2006; Hansen et al., 2012). Prior attempts to explain the wave of cropping system intensification have largely focused on simple economic rationales, and thus we lack a critical understanding of the social dynamics underlying the revolution in semi-arid cropping systems (Kaan et al., 2002; Hansen et al., 2012). I examined the motivations, perceptions, and social interactions of dryland farmers that practice different levels of cropping system intensity in the High Plains, and present strategies to facilitate transitions to intensified cropping systems.

The sociological analysis also serves to contextualize the final chapter – a spatial analysis of cropping patterns at the landscape scale, which quantifies the magnitude and characterizes the nature of cropping system intensification in the High Plains. Understanding the rate of cropping system intensification at the landscape scale is critical to estimating the economic and environmental implications of the semi-arid cropping revolution. Here, I use high-resolution satellite data to quantify dryland cropping patterns from 2008 to 2016, and I use these estimates to scale up the field-level research documented in previous chapters on soil C, herbicide use, yields, and profitability. Through this analysis, I seek to answer three questions: 1) To what extent is cropping system intensity increasing in the High Plains? 2) Which crops are replacing summer fallow and where? 3) What are the implications of observed intensification in the High Plains for C sequestration, grain yield, herbicide use, and net farm incomes?

REFERENCES

Al-Karaki, G., McMichael, B., Zak, J., 2004. Field response of wheat to arbuscular mycorrhizal fungi and drought stress. Mycorrhiza 14, 263-269.

Anderson, R., 2003. An ecological approach to strengthen weed management in the semiarid Great Plains. Advances in Agronomy 80, 33-62.

Anderson, R.L., 2009. Rotation design: A critical factor for sustainable crop production in a semiarid climate: A review. Organic Farming, Pest Control and Remediation of Soil Pollutants. Springer, pp. 107-121.

Bot, A., Nachtergaele, F., Young, A., 2000. Land resource potential and constraints at regional and country levels. . Food & Agriculture Org.

Bowman, R., Reeder, J., Lober, R., 1990. Changes in soil properties in a Central Plains rangeland soil after 3, 20, and 60 years of cultivation. Soil Science 150, 851-857.

Carolan, M., 2016. The Sociology of Food and Agriculture. Taylor & Francis.

Cochran, V., Danielson, J., Kolberg, R., Miller, P., 2006. Dryland cropping in the Canadian prairies and the US northern Great Plains. AGRONOMY 23, 293.

Derksen, D.A., Anderson, R.L., Blackshaw, R.E., Maxwell, B., 2002. Weed dynamics and management strategies for cropping systems in the northern Great Plains. Agronomy Journal 94, 174-185.

Diaz, R.J., Rosenberg, R., 2008. Spreading dead zones and consequences for marine ecosystems. science 321, 926-929.

Drinkwater, L.E., Snapp, S., 2007. Nutrients in agroecosystems: rethinking the management paradigm. Advances in Agronomy 92, 163-186.

Du, E., de Vries, W., Galloway, J.N., Hu, X., Fang, J., 2014. Changes in wet nitrogen deposition in the United States between 1985 and 2012. Environmental Research Letters 9, 095004.

Gardner, J.B., Drinkwater, L.E., 2009. The fate of nitrogen in grain cropping systems: a meta-analysis of 15N field experiments. Ecological Applications 19, 2167-2184.

Grant, C.A., Peterson, G.A., Campbell, C.A., 2002. Nutrient considerations for diversified cropping systems in the northern Great Plains. Agronomy Journal 94, 186-198.

Hansen, N.C., Allen, B.L., Baumhardt, R.L., Lyon, D.J., 2012. Research achievements and adoption of no-till, dryland cropping in the semi-arid US Great Plains. Field Crops Research 132, 196-203.

Harinikumar, K., Bagyaraj, D., 1988. Effect of crop rotation on native vesicular arbuscular mycorrhizal propagules in soil. Plant and Soil 110, 77-80.

Kaan, D.A., 2002. An economic evaluation of alternative crop rotations compared to wheat-fallow in Northeastern Colorado. Technical bul.(Colorado Agricultural Experiment Station); TB-02-1.

Ko, J., Ahuja, L.R., Saseendran, S., Green, T.R., Ma, L., Nielsen, D.C., Walthall, C.L., 2012. Climate change impacts on dryland cropping systems in the Central Great Plains, USA. Climatic Change 111, 445-472.

Kolberg, R., Kitchen, N., Westfall, D., Peterson, G., 1996. Cropping intensity and nitrogen management impact of dryland no-till rotations in the semi-arid western Great Plains. Journal of Production Agriculture 9, 517-521.

Koohafkan, P., Stewart, B.A., 2008. Water and cereals in drylands. Earthscan.

Maaz, T., Wulfhorst, J., McCracken, V., Kirkegaard, J., Huggins, D.R., Roth, I., Kaur, H., Pan, W., 2017. Economic, policy, and social trends and challenges of introducing oilseed and pulse crops into dryland wheat cropping systems. Agriculture, Ecosystems & Environment.

Nichols, V., Verhulst, N., Cox, R., Govaerts, B., 2015. Weed dynamics and conservation agriculture principles: a review. Field Crops Research 183, 56-68.

Peterson, G., Schlegel, A., Tanaka, D., Jones, O., 1996. Precipitation use efficiency as affected by cropping and tillage systems. Journal of Production Agriculture 9, 180-186.

Peterson, G.A., Westfall, D.G., 2004. Managing precipitation use in sustainable dryland agroecosystems. Annals of Applied Biology 144, 127-138.

Plaza-Bonilla, D., Arrúe, J.L., Cantero-Martínez, C., Fanlo, R., Iglesias, A., Álvaro-Fuentes, J., 2015. Carbon management in dryland agricultural systems. A review. Agronomy for Sustainable Development 35, 1319-1334.

Sebilo, M., Mayer, B., Nicolardot, B., Pinay, G., Mariotti, A., 2013. Long-term fate of nitrate fertilizer in agricultural soils. Proceedings of the National Academy of Sciences 110, 18185-18189.

Shaver, T., Peterson, G., Ahuja, L., Westfall, D., Sherrod, L., Dunn, G., 2002. Surface soil physical properties after twelve years of dryland no-till management. Soil Science Society of America Journal 66, 1296-1303.

Sherrod, L., Peterson, G., Westfall, D., Ahuja, L., 2003. Cropping intensity enhances soil organic carbon and nitrogen in a no-till agroecosystem. Soil Science Society of America Journal 67, 1533-1543.

Simkin, S.M., Allen, E.B., Bowman, W.D., Clark, C.M., Belnap, J., Brooks, M.L., Cade, B.S., Collins, S.L., Geiser, L.H., Gilliam, F.S., 2016. Conditional vulnerability of plant diversity to atmospheric nitrogen deposition across the United States. Proceedings of the National Academy of Sciences 113, 4086-4091.

Smith, E., Young, D.L., 2000. The economic and environmental revolution in semi-arid cropping in North America. Annals of Arid Zone 39, 347-362.

Thompson, J., 1987. Decline of vesicular-arbuscular mycorrhizae in long fallow disorder of field crops and its expression in phosphorus deficiency of sunflower. Australian Journal of Agricultural Research 38, 847-867.

USDA-ERS, 2017. Farm Household Income Forecast. Farm Household Well-being.

USGCRP, 2014. Climate change impacts in the United States: U.S. national climate assessment. GlobalChange.gov.

Wall, D.H., 2012. Soil ecology and ecosystem services. Oxford University Press.

CHAPTER 2: INTENSIFYING ROTATIONS INCREASES SOIL CARBON, FUNGI, AND AGGREGATION IN SEMI-ARID AGROECOSYSTEMS

2.1 Introduction

Increasing soil organic carbon (SOC) in semi-arid agroecosystems is a critical sustainability challenge for the 21st century. Semi-arid regions, defined as regions with a precipitation to potential evapotranspiration (PET) ratio of 0.2-0.5 (UN, 2011), constitute 20% of Earth's land surface and support a large agricultural population that is under increasing pressure from land degradation and desertification (Bot et al., 2000; Koohafkan and Stewart, 2008). Dryland farming in these regions depends solely on precipitation and uses no supplemental water, which presents a need for, and a challenge in increasing SOC. Warmer temperatures under future climate scenarios will further exacerbate water limitation in semi-arid climates (Ko et al., 2012; USGCRP, 2014). Dryland agriculture contributes to climate change through energy use during its life cycle (e.g., fertilizer production) and emissions of greenhouse gases from soils, but agricultural soils can also sequester carbon dioxide from the atmosphere. Not only are dryland soils an underutilized resource for enhancing C sequestration to mitigate climate change (Lal, 2004), but increasing SOC is also a key climate change adaptation strategy for dryland farmers. Increasing SOC has the potential to mitigate the risks associated with increasing water limitation by enhancing water infiltration and storage (Franzluebbers, 2002). Greater SOC can also enhance soil functions like nutrient provision and retention, substrate provision for biodiversity, and erosion control (Wall, 2012), but increasing SOC in dryland systems is constrained by high erosion rates, low C inputs, and accelerated mineralization from tillage (Plaza-Bonilla et al., 2015).

Environmental constraints on C inputs are further exacerbated by the common practice of a year-long period called summer fallow, where no crops are grown and weeds are controlled so that the soil can accumulate rainwater and increase the yield of the following crop. Summer fallow periods have historically helped stabilize wheat yields, but they are inefficient and management intensive. Precipitation storage efficiency is typically less than 35% under best management (Nielsen and Vigil, 2010), and fallow periods require frequent tillage and/or herbicides for weed control. No-till management (where weeds are controlled through herbicides instead of tillage) enhances water storage and enables dryland farmers to replace summer fallow periods with a crop, a form of cropping system intensification. Within no-till systems, cropping system intensification may increase SOC by increasing overall productivity relative to more traditional crop-fallow systems, where a crop is only grown once every two years (Sherrod et al., 2003; Peterson and Westfall, 2004). In semi-arid agroecosystems around the world, dryland farmers are undergoing transitions from crop-fallow to intensified cropping systems (Maaz et al., 2017), in what has been called a revolution in semi-arid cropping (Smith and Young, 2000). This widespread transformation in dryland agroecosystems may have significant implications for C sequestration and enhancing the resilience of dryland agriculture through gains in SOC. However, the mechanisms and extent to which cropping system intensity increases SOC independent of shifts in tillage practices are unclear and may be influenced by climate, soil type, and management strategy.

Given the limited productivity and associated amount of available C inputs to dryland soils, understanding the mechanisms and drivers of how C becomes stabilized is important to halt or reverse losses of SOC. Accrual of SOC can occur when C is protected from decomposition, either through adsorption on soil mineral surfaces, or when it is physically bound in soil aggregates (Jastrow, 1996). Protection of SOC in soil aggregates is a primary mechanism of SOC stabilization in agroecosystems and is highly sensitive to management (Tisdall and Oades, 1982). Several studies have observed the accumulation of SOC in aggregate pools as a significant driver of C accrual during the conversion from conventional till to no-till systems (Six et al., 1999). Similarly, cropping system intensification contributes greater C inputs to soil by replacing fallow periods with a growing crop, and has been associated with greater macroaggregation, microbial biomass, and SOC (Shaver et al., 2003; Peterson and Westfall, 2004; Sherrod et al., 2005; Mikha et al., 2010). Greater root and microbial biomass can enhance aggregation as roots, fungal hyphae, and microbial polysaccharides are all important stabilizing forces in the formation of aggregates (Jastrow et al., 1998).

In addition to the direct influence of greater C inputs on SOC accrual, cropping system intensification may also indirectly enhance SOC through changes in the microbial community (Fig. 2.1). Functional classifications of microbial communities enable researchers to test hypotheses relating microbial communities to ecosystem processes. For example, the relatively coarse metric of the ratio of fungi to bacteria in soil is often related to its C sequestration potential (Strickland and Rousk, 2010). A recent study using RNA and protein profiling in conjunction with phospholipid fatty acid (PLFA) analysis provided new evidence linking fungal dominance to soil C storage potential (Malik et al., 2016), implicating fungi in a central role in SOC accrual (Fig. 2.1). Fungi contribute to SOC stabilization by physically entangling soil particles in hyphae to form aggregates and by secreting cohesive substances like glycoproteins that enhance aggregation (Wilson et al., 2009). In addition to their role in aggregation, some evidence suggests that fungi may have a higher C use efficiency, and thus retain more C in the soil during the decomposition process relative to bacteria (Waring et al., 2013; Kallenbach et al., 2016; Malik et al., 2016). Still, the functional importance of the fungi:bacteria ratio remains a point of controversy due to the wide range of functional diversity across both groups, and several lines of evidence have disputed the relationship between fungal dominance and C stabilization (Rousk and Frey, 2015).

Cropping system intensity also influences soil moisture and the availability of C substrates (Farahani et al., 1998b; Sherrod et al., 2003), both of which are strong determinants of microbial community dynamics (Drenovsky et al., 2004). For example, increasing the availability of C substrates by reducing the duration of fallow periods can increase populations of arbuscular mycorrhizal fungi (Thompson, 1987; Harinikumar and Bagyaraj, 1988). Additionally, cropping system intensification is often achieved by growing a greater diversity of crops, and previous studies have linked crop diversity to higher fungi:bacteria ratios and microbial biomass (Lange et al., 2014; McDaniel et al., 2014). However, Acosta-Martinez et al. (2007) found that cropping system intensity increased microbial biomass, but not always fungal biomass, and Stromberger et al. (2007) observed few differences between the microbial community structures of different dryland crop rotations. Further investigation into the effects of no-till crop rotations on microbial communities is needed to understand the potential role of microbes as mediators of C stabilization in dryland agroecosystems.

The effects of cropping system intensity on microbial communities and aggregation as drivers of C sequestration may change based on environmental factors like soil clay content and climate (Acosta-Martinez et al., 2003; Sherrod et al., 2003; Peterson and Westfall, 2004). As SOC accrual associated with cropping system intensity has primarily been observed in controlled experimental systems, and mainly considered in combination with reduced tillage (e.g., Norton et al., 2012), further exploration is needed into the extent that cropping system intensification impacts the mechanisms of SOC stabilization independent of tillage and across a range of environmental and management factors.

I conducted a study on dryland, no-till farms and long-term experiment stations in the semi-arid Great Plains, USA that captured a wide range of crop rotations, soil textures, PET rates, and management histories. This allowed me to quantify the potential for cropping system intensification to enhance SOC storage and aggregation relative to traditional crop-fallow systems and to evaluate the relative effects of management, texture, and climate on SOC levels. I hypothesized that SOC would increase with cropping system intensity directly through greater C inputs, and indirectly through greater fungal biomass and aggregation, and that these effects would be robust across a range of environmental and management contexts (Fig. 2.1).

2.2 Materials and Methods

2.2.1 *Cropping systems*

Wheat-fallow (WF) is the dominant dryland cropping system in the semi-arid Great Plains (Hansen et al., 2012). This system consists of growing winter wheat (*Triticum aestivum*) from September to July, then fallowing for 14 months until the next wheat planting. No-till farmers in this region often reduce summer fallow frequency from one out of two years (WF), to one out of three or four years (mid-intensity; MID), by rotating winter wheat with crops like corn (*Zea mays*), sorghum (*Sorghum bicolor*), proso millet (*Panicum miliaceum*), peas (*Pisum sativum*), or sunflowers (*Helianthus annuus*). They may also eliminate summer fallow altogether via continuous cropping (CON).

2.2.2 *Study sites*

Sampling was conducted in 2015 and 2016 on 96 dryland, no-till fields in eastern Colorado and western Nebraska, representing 54 fields from working farms and 42 fields from long-term experiment stations (Fig. 2.2). Each of three levels of cropping intensity – WF (n=27), MID (n=37) and CON (n=26) – was represented along a potential evapotranspiration (PET) gradient that increased from 1368 mm yr^{-1} in northwestern Nebraska to 1975 mm yr^{-1} in southeastern Colorado (Fig. 2.2). Additionally, two 30-year old Conservation Reserve Program perennial grass plots (30-yr CRP) at the three long-term experiment stations in Colorado (Fig. 2.2) were sampled as a reference for comparison with the cropping systems (n=6). To assign a value of PET to each site, a linear equation was used based on known PETs from 6 locations as measured by open-pan evaporation (Peterson et al., 2001) and the latitude of each site. I used climate data from 1980-2010 to assign a value of 30-yr average annual precipitation to each field. Average annual precipitation at the study sites ranged from 349-472 mm and means by cropping intensity were statistically equivalent (PRISM, 2017). Five-year field histories were collected for each field (Table 2.1). I collected N fertilizer use data from working farms for the years 2010-2014 to calculate annualized fertilizer use. No field received compost or manure. All fields were under tilled WF management for several decades prior to implementation of no-till and the current crop rotation. Every field was planted to winter wheat in the fall of 2015.

2.2.3 *Soil sampling*

In the fall of 2015, soil samples from the 0-20 cm depth were taken using a corer (2 cm dia.) at 4 locations within each field that represented corners of a 10 x 10 m square and geo-referenced for later samplings. At each location, I composited 10 soil cores. Field replicates were analyzed separately for SOC and later averaged to obtain field-level means. Soil samples used to determine texture and pH were composited and analyzed by field. Soil texture was determined by hydrometer, and pH using a 1:1 slurry of soil and deionized water at Ward Laboratories in Kearney, NE. At the three long-term experiment stations in Colorado (see Fig. 2.2), samples were taken from both a summit and toeslope position in each field to examine if the differences in water availability at upland and lowland positions influenced SOC. Samples on all other fields were taken from a flat topographical position and labeled as a summit.

Additionally, soil samples were taken in spring of 2016 at the same locations as the fall sampling. The spring sampling was to a shallower 0-10 cm depth because the surface soil layers are more likely to be influenced by management practices, and surface soil physical properties, such as aggregation, can confer important functionality on water infiltration and storage (Mann, 1986; Shaver et al., 2002). A 5.5 cm slide-hammer corer was used to take one 0-10 cm depth soil sample per sampling location (4 cores per field) to assess water-stable aggregation, bulk density, SOC, total N, and PLFA. Bulk density, SOC, and total N were analyzed by sampling location (4 field replicates), and later averaged to obtain field-level means. Soils to determine water-stable aggregation and PLFA were composited and analyzed by field. Subsamples of field moist soils (20 g) were weighed, dried at 105°C, and reweighed to determine gravimetric water content. Toeslope positions at the long-term experiment stations were excluded in the spring sampling. All samples were kept on ice in coolers for 4 to 36 hours before being refrigerated at 4°C.

2.2.4 *Soil carbon & water-stable aggregates*

I determined SOC and total N on spring and fall air-dried soils. Soils were ground on a roller grinder and analyzed for total C and N on a LECO CHN-1000 auto analyzer (St. Joseph, MI). Soil inorganic C was assessed using the modified pressure calcimeter method (Sherrod et al., 2002), and subtracted from total C to determine concentrations of SOC.

A wet sieving method adapted from Elliot (1986) was performed on 80 g air-dried subsamples to separate soils into large macroaggregate (>2000 um), small macroaggregate (250-2000 um), free microaggregate (53-250 um), and free silt and clay (<52 um) size fractions. Air-dried soils were submerged in water for 5 minutes prior to sieving for slaking. Sieves were manually moved up and down 50 times over a period of 2 minutes in a shallow pan of deionized water. Each size fraction was collected in pre-weighed aluminum pans, dried in a forced-air oven at 60°C and weighed. Floating litter was decanted into a separate pan, dried, and weighed. Aggregate mean weight diameter (MWD) was calculated as an indicator of aggregate stability according to Van Bavel (1950). Very few large macroaggregates were present, so they were combined with small macroaggregates for subsequent sand correction and C analysis. Concentration of C of each size fraction was measured following similar methods as the bulk soil, described above.

A sand correction was performed on both the macroaggregate and microaggregate fractions to compare sand-free weights and C content between soils with different sand contents. Briefly, 5 g subsamples were mixed with 5% sodium hexametaphosphate and shaken to disperse aggregates. This solution was then passed through 53 um and 250 um sieves for microaggregate and macroaggregate fractions, respectively. Sand that remained on top of the sieve was dried and weighed. Sand-free C content of the aggregate fractions was calculated as:

$$\text{Sand-free \% C} = \text{\% C} \cdot (1\text{-proportion of sand})^{-1}$$

2.2.5 *Phospholipid fatty acids (PLFA)*

PLFA analyses were conducted on subsamples of the soils used to measure aggregation. Immediately upon passing through the 8 mm sieve, soils were further sieved to 2 mm and frozen at -20°C within 2 days of sampling before being freeze-dried. PLFAs were extracted and separated at Ward Labs (Kearney, NE) as described in Hamel et al. (2006). Briefly, two grams of lyophilized soil (dry weight equivalent) was extracted in 9.5 ml mixture of dichloromethane (DMC)/methanol/citrate buffer (1:2:0.8 v/v) for 1 h at 240 rpm. Then, 2.5 ml of DMC and 10ml of a saturated KCl solution were added to each tube, shaken for 5 min, and centrifuged to separate the organic fraction. Soil lipid extracts were separated in silica gel columns, transmethylated, and PLFAs were quantified by gas chromatography on an Agilent 7890A GC. The fatty acid methyl ester 18:2 w6c was used to represent fungal biomass, and 10:0 2OH, 14:0 iso, 15:0 iso, 15:0 anteiso, 16:0 iso, 16:1 w7c, 17:0 iso, 17:0 anteiso, 17:0 cyclo, and 18:1 w7c were used to represent bacterial biomass (Frostegård and Bååth, 1996). Only bacterial and fungal PLFAs (65 total biomarkers) were added to represent microbial biomass (polyunsaturated and PLFA biomarkers longer than 20 C chains were excluded from the analysis).

2.2.6 *Statistical analysis*

The relationships between cropping system intensity and SOC at 10 cm and 20 cm, aggregate MWD and C content, and microbial PLFA were tested using multiple linear regression. Models were selected using backwards selection with cropping system intensity as a categorical variable, and all management factors (# years in no-till, # years in rotation, and fertilizer use) and environmental factors (PET, % clay, pH, and slope) until all remaining terms were significant (α=0.05). To account for environmental and management factors as covariates, least-squared means for each level of cropping system intensity were generated and tested for significant pairwise comparisons. The relationships between aggregation and SOC, and between fungal biomass and macroaggregation were tested using linear regressions. Differences between C concentrations and C:N ratios of aggregate size classes were tested for significance using ANOVA, and pairwise comparisons were made using a Tukey post hoc analysis. P-values less than 0.05 were considered significant. I used R for all data analyses (R Core Team, 2013), and multiple linear regressions were generated and tested for significance using the packages lme4 (Bates et al., 2014), lsmeans (Lenth and Hervé, 2015), and lmerTest (Kuznetsova et al., 2015). Due to the small sample size, CRP fields were excluded from statistical analyses and are presented in figures for the purpose of a relative comparison only.

I used structural equation modeling (SEM) to test the fit of my hypothesized model (Fig. 2.1). SEM tests whether experimental data fit a proposed model using a covariance matrix (Kline, 2015). A non-significant ($p>0.05$) Chi-square test indicates that the model fits the data (Grace et al., 2010). SEM calculates path coefficients that represent the strength and directionality (positive or negative) of relationships between variables, and partitions variance when a response variable is connected by two or more explanatory variables. Because of this variance partitioning, removal of a path between two variables can impact path coefficients between other variables. A positive path coefficient for MID or CON reflects a positive effect of mid-intensity or continuous rotations relative to WF, respectively.

I began with the hypothesized model but found that it did not fit the data (Chi-square p-value<0.001). I refined the model by adding significant paths ($\alpha=0.05$), identified by visualizing relationships between residuals of previously unconnected variables. I also removed insignificant paths. I compared the models that significantly fit the data using Akaike information criterion.

The final model (Fig. 2.8) was significant (Chi-square p-value>0.05) and had a lower AIC than the others I compared it to. Structural equation models were created and tested using the lavaan package in R (Rosseel, 2011).

2.3 Results

Intensified cropping systems had higher SOC, aggregation, and fungal and total microbial biomass, and these trends were robust amidst variability in environmental and management conditions.

2.3.1 *SOC*

I observed greater SOC concentrations in CON relative to MID and WF rotations at both the 0-10 and 0-20 cm depths (Fig. 2.3). Cropping system intensity ($p=0.02$), PET ($p<0.001$), % clay ($p<0.001$), and slope position ($p<0.001$) explained 54% of the variability in SOC at the 0-10 depth. Similar to the shallow depth, cropping system intensity ($p<0.01$), PET ($p<0.001$), % clay ($p<0.001$), and slope position ($p<0.001$) explained 54% of the variability in SOC at the 0-20 depth. After accounting for PET, % clay, and slope as covariates, SOC concentrations in WF, MID, and CON averaged 1.09%, 1.15%, and 1.28% at 0-10 cm, and 0.92%, 0.89%, and 1.03% at 0-20 cm, respectively (Fig. 2.3). SOC levels were 17% higher in CON rotations than WF at the 0-10 cm depth, but CON was not significantly different from MID (Fig. 2.3). However, SOC concentrations in CON rotations were 16% greater than MID, and 12% greater than WF to a depth of 20 cm. SOC concentrations in CON and the less intensified rotations were about 80% and 70% of those in the 30-yr old CRP at both depths, respectively. SOC concentrations in MID rotations were similar to that of WF at both 0-10 cm and 0-20 cm depths (Fig. 2.3). There were no significant cropping intensity effects on bulk density (Table 2.1).

2.3.2 *Aggregate stability and aggregate-associated C*

Aggregate MWD and C content increased with cropping system intensity (Figs. 4, 5). Cropping system intensity ($p>0.01$), PET ($p=0.03$), and % clay ($p<0.01$) explained 30% of the variability in MWD of water-stable aggregates in the 0-10 cm depth. After accounting for PET and % clay as covariates, aggregate MWD in CON rotations was about twice as large as those in WF, and aggregate MWD in MID rotations was intermediate of the two (Fig. 2.4). Aggregate MWD in the 30-yr CRP was 4 times greater than CON rotations, and 8 times greater than WF (Fig. 2.4).

Cropping system intensity was unrelated to the sand-corrected C concentration across size classes. The C concentrations of macroaggregates were the highest, and C concentrations of the free silt and clay fraction were the lowest, regardless of cropping system intensity (Table 2.2). There were few differences in C:N ratio between aggregate size classes and ratios varied from 9-14 (data not shown), but the C:N ratio of the bulk soil was higher in CON compared to WF rotations (Table 2.1). Clay % was a significant covariate in the models of C in each aggregate size class, and # years no-till was also a covariate in the model of free silt and clay C. Total C content of the free silt and clay fraction was lower in MID rotations compared to WF, but was no different than CON rotations (Fig. 2.5A). There were no significant differences in C content of free microaggregates (Fig.5B). Total C content of macroaggregates was about 3 times greater in CON compared to WF rotations (Fig. 2.5C).

2.3.3 *Microbial PLFA*

Total PLFA concentration (a proxy for microbial biomass), the fungi:bacteria ratio, and total fungal PLFA concentration increased with cropping system intensity. There was no relationship between cropping system intensity and bacterial PLFAs (data not shown). Cropping system intensity ($p=0.04$), % clay ($p<0.01$), and pH ($p<0.01$) explained 31% of the variability in microbial biomass. Total PLFA in CON rotations was 35% greater than that of WF, and MID rotations were intermediate between the two. Total PLFA in 30-yr CRP was 1.5, 1.8, and 2.1 times greater than CON, MID, and WF rotations, respectively (Fig 6A). Cropping system intensity was the only significant predictor of the fungi:bacteria ratio ($p<0.01$, $R^2=0.18$) and total fungal PLFAs ($p<0.01$, $R^2=0.15$). CON rotations had three times higher fungi:bacteria ratios than WF and MID rotations were intermediate of the two (Fig. 2.6B). Total fungal PLFA was 3 times greater in CON rotations compared to WF, but was not significantly different from MID rotations (Fig. 2.6C).

2.3.4 *Relationships between aggregation, fungi, and SOC*

Aggregate MWD in dryland cropping systems increased linearly with fungal biomass ($MWD=119.8+4.0\times$fungal PLFA, $F=15.82$, $p<0.001$, $R^2=0.20$, Fig. 2.7A). SOC increased linearly with aggregate MWD ($SOC=0.828+0.001\times MWD$, $p<0.001$, $R^2=0.23$, Fig. 2.7B).

2.3.5 *Structural equation model*

Cropping system intensity had positive direct and indirect effects on fungal biomass, SOC, and aggregate MWD in the final structural equation model (Fig. 2.8). CON rotations had positive effects on fungal biomass, which was also positively affected by SOC. CON, but not

MID rotations, indirectly influenced aggregate MWD as mediated through increases in SOC. The positive effects of CON on fungal biomass also mediated positive effects on SOC and aggregate MWD. Soil clay content positively affected fungal biomass and SOC, and PET negatively affected SOC. Overall, the structural equation model explained 30 and 50% of the variability in aggregate MWD and SOC, respectively.

2.4 Discussion

I examined the relationship between cropping system intensity and soil properties under no-till management. My findings suggest that cropping system intensity was positively associated with SOC and aggregation, and that these trends were potentially mediated by greater fungal biomass. That these trends were present across wide ranges in soil texture, climate, and management history suggests that continuous cropping is a suitable strategy for increasing SOC, even in higher PET climates.

2.4.1 *Cropping system intensity and SOC*

Even with no-till management, summer fallow periods in WF and MID rotations can only store a maximum of 40% of total precipitation to use for the next crop, as weeds and evaporation result in significant water losses during this time (Peterson et al., 1996). Replacing summer fallow periods with crops increases the overall precipitation use efficiency of cropping systems (Farahani et al., 1998a), and as a result, cropping system intensification can potentially double the total amount of crop residue production relative to WF (Peterson and Westfall, 2004). Although I did not measure C input, others have found that SOC concentrations often mirror the gradient of C inputs associated with cropping system intensity and perennial grasslands (Sherrod et al., 2005; Engel et al., 2017). However, I observed that differences in SOC were only observed when summer fallow was completely eliminated, as there were no significant differences in SOC in MID rotations compared to WF (Fig. 2.3). The structural equation model offers one possible mechanistic explanation for the observed differences in SOC between the cropping systems. The final model suggested that the effects of PET and clay more strongly influenced SOC than the additional C inputs associated with MID rotations relative to WF, but that the combined direct and indirect effects of CON rotations on SOC were enough to overcome these environmental factors (Fig. 2.8). Additionally, greater precipitation use efficiency in CON systems may also contribute to more SOC by maintaining soil water content at lower levels throughout the year relative to less intense rotations, thereby reducing organic matter decomposition (Paustian et al., 2000).

Assuming that all fields started at similar levels of SOC at the initiation of the present crop rotation, continuous cropping was associated with an average increase of 0.08 Mg C $ha^{-1}yr^{-1}$ relative to WF at the 0-10 cm depth. This is consistent with prior estimates of C accrual (0.07-0.16 Mg C $ha^{-1}yr^{-1}$) associated with dryland cropping system intensification (Peterson and Westfall, 2004; Engel et al., 2017). In addition to the host of agronomic benefits associated with greater SOC, the total C emissions associated with the life cycle of dryland no-till grain production in the U.S. is on average 0.1 Mg C ha^{-1} yr^{-1} (Nelson et al., 2009), and thus the conversion to no-till continuous cropping from no-till wheat-fallow has the potential to offset the C footprint of grain production in these systems through enhanced soil C sequestration. I also observed 0.11% higher SOC concentrations to a depth of 20 cm in CON compared to MID rotations. Relative to less intensified rotations, CON systems more closely mimic natural regional ecosystems by supporting a growing crop every year, and this similarity is evident in

SOC and other soil properties that are closer to the levels of the 30-yr CRP in continuously cropped soils. However, I note that levels of SOC in CON soils are still much lower than the grassland soils, suggesting that significant constraints to SOC accumulation exist in annual cropping systems.

2.4.2 *Aggregates*

Cropping system intensity and associated higher C inputs were associated with higher water-stable aggregation and macroaggregate-associated C (Figs. 4 and 5). Higher intensity rotations may contribute to aggregation through enhanced above and belowground C inputs, and greater residue cover on the soil surface protects aggregates from the destructive forces of wind and rain (Kong et al., 2005; Mulumba and Lal, 2008). Annually cropped semi-arid agroecosystems are particularly susceptible to erosion due to constraints on plant production and residue cover, and the greater amount of aggregation in the 30-yr CRP relative to the cropping systems reflects the contributions of perennial roots and constant plant cover to enhancing soil structure.

I predicted a co-increase in both SOC and aggregate MWD in intensified rotations as the greater level of C inputs become physically protected in aggregates, which would contribute to the accumulation of SOC (Fig. 2.1). Additionally, as SOC is a driver of aggregation, I predicted and observed a similar positive effect of SOC on aggregate MWD (Tisdall and Oades, 1982; Figs. 2.1, 2.8). I was surprised to find that removing the direct effect of aggregate MWD on SOC improved the fit of the structural equation model, but I found that SOC directly and indirectly increases MWD as mediated through soil fungi, as is discussed in section 2.4.3 (Fig. 2.8). A more sensitive indicator of aggregate-associated C may have revealed a more direct influence of aggregation on SOC in the structural equation model. Others have suggested that increases in microaggregates within macroaggregates are responsible for a majority of the difference in SOC between differently managed cropping systems (Six and Paustian, 2014). I did not isolate microaggregates within macroaggregates here due to low overall aggregation across the study sites.

I note that analyzing the relationship between SOC and aggregate MWD independent of environmental and management factors revealed a positive relationship, as expected (Fig. 2.7B). This is in agreement with others who have examined the relationship between SOC and aggregate MWD, highlighting aggregation as an important C stabilization mechanism in dryland soils (Kemper and Koch, 1966; Shaver et al., 2003). Greater proportions of C-rich macroaggregates in CON rotations contributed to higher SOC concentrations in the bulk soil relative to WF (Fig. 2.3, Fig. 2.5C, Table 2.2). The C:N ratio was similar across aggregate size classes and mineral fractions, and generally ranged from 9-14. These low values relative to fresh inputs suggest that much of the occluded C in macroaggregates is likely derived from more highly processed C (Elliott, 1986; Jastrow, 1996). Plant C inputs can become stabilized as microbial byproducts of decomposition and tightly bind to soil mineral surfaces (Cotrufo et al., 2013). An accumulation of this microbially processed C in macroaggregates (and possibly in microaggregates within macroaggregates) is potentially responsible for much of the observed difference in SOC between different cropping systems.

2.4.3 *Microbial Communities*

I found that both fungal biomass and fungi:bacteria ratio increased with cropping system intensity (Fig. 2.6), lending support for the interpretation of fungi:bacteria as a soil property that influences C dynamics and is sensitive to management (De Vries et al., 2012; Waring et al., 2013). Rotation intensity likely influenced the fungal community through changes in the soil C:N ratio, and overall greater SOC (Waring et al., 2013; Malik et al., 2016). Past studies have drawn links between greater N fertilizer use, lower soil C:N, and lower fungal dominance in soils (Bardgett and McAlister, 1999; Waring et al., 2013). Annualized N fertilizer use from 2010-2014 was no greater in intensified systems (Table 2.1) despite achieving greater total crop production compared to WF (Rosenzweig et al., in review), and this likely contributed to the observed increase in soil C:N with cropping system intensity (Table 2.1). Thus, it is possible that lower nutrient availability in intensified systems may be contributing to the observed increase in fungi:bacteria (Strickland and Rousk, 2010). Additionally, intensification is most often achieved through growing a greater diversity of crops, and crop diversity, independent of intensity, can enhance microbial biomass, fungal dominance, and ultimately SOC (Lange et al., 2014; McDaniel et al., 2014). I did not observe a relationship between pH and fungi:bacteria, possibly due to a lack of variability in pH across sites (Table 2.1). I also found little influence of the number of years in no-till on microbial communities, possibly because most sites had at least 5 years of no-till management, which is long enough to exert a considerable effect relative to conventional tillage (Rosenzweig et al., 2016).

The influence of cropping system intensity on SOC and aggregation may be attributed to the role of fungi in mediating the processes that control these properties. The relationship between SOC and fungi is circular, as fungi contribute to C sequestration, and soils with more SOC also tend have more fungi (Strickland and Rousk, 2010). My findings from the structural equation model suggest that SOC and fungi positively influenced each other, supporting the idea of a positive feedback loop (Fig. 2.8). While the link between fungi, C use efficiency, and SOC remains controversial (Strickland and Rousk, 2010), it is generally accepted that fungi contribute to SOC accrual by enhancing the physical protection of organic matter in aggregates (Six et al., 2006). The positive linear relationship between fungal biomass and aggregate MWD corroborates past work examining the link between them (Fig. 2.7A). Aggregation also has important feedbacks with overall agroecosystem performance in semi-arid climates, as aggregates create pore space and improve the soil's ability to reduce surface runoff (and erosion) and capture water (Shaver et al., 2003), which is the most limiting factor to crop production in dryland systems.

2.4.4 *Conclusions*

While WF remains the most common cropping system in the semi-arid High Plains, this and other semi-arid regions around the world are undergoing a profound transition to intensified dryland cropping systems, and thus it is critical to understand the implications of this transformation (Hansen et al., 2012; Maaz et al., 2017). I found different levels of SOC, aggregation, and fungal biomass between different levels of cropping system intensity and tested hypothesized links between them. Overall, my results suggest that cropping system intensity, independent of tillage, increases SOC both directly, through greater C inputs to soil, and indirectly, through effects on microbial communities and aggregation. I observed these relationships to be remarkably robust across a wide climatic gradient, and amidst variability in soil texture and management history. These results corroborate others who have found greater aggregation and SOC in more intensely cropped systems, but also shed new light on the central role that fungi may play in C storage in dryland agroecosystems. I found that the elimination of summer fallow in semi-arid cropping systems has the potential to offset the C emissions associated with no-till grain production, and provide gains in SOC that will contribute to the long-term success of dryland agriculture.

Table 2.1. Soil properties and management histories of study sites. Means are reported for each variable, with the range of values presented below each mean in parentheses. Values for annualized N fertilizer use and soil C:N represent means ± standard error. Soils were sampled to 20-cm depth with the exception of values for bulk density and soil C:N, which are for the 0-10 cm depth. Letters represent significant differences between cropping systems (α=0.05). Annualized N fertilizer use is the total amount of N fertilizer applied over the years 2010-2014 divided by 5. WF=wheat-fallow; MID=rotations with fallow every 3 or 4 years; CON=continuous rotations; 30-yr CRP=Conservation Reserve Program shortgrass prairie strips restored 30 years ago (n=6).

Cropping system	**Clay (%)**	**Silt (%)**	**Sand (%)**	**pH**	**Years no-till**	**Years in rotation**	**N fertilizer (kg N ha^{-1} yr^{-1})**	**Bulk Density (g $^{cm-3}$)**	**Soil C:N**
WF	22 (10-31)	39 (19-50)	41 (22-71)	7.4 (6.3-8.5)	20 (0-30)	23 (10-50)	31±1.0a	1.0 (0.8-1.3)	8.7±0.1b
MID	21 (8-36)	36 (15-60)	43 (22-76)	7.3 (6.3-8.3)	19 (1-30)	19 (3-30)	45±1.6a	1.0 (0.8-1.2)	8.9±0.1a b
CON	22 (8-40)	38 (17-53)	40 (14-75)	7.4 (6.3-8.3)	22 (4-30)	21 (5-30)	34±1.4a	1.0 (0.8-1.2)	9.2±0.1a
30-yr CRP	18 (8-25)	33 (21-46)	49 (29-71)	7.7 (7.2-8.2)	30	30	NA	0.9 (0.8-1.0)	10.3±0.2

Table 2.2. Sand-corrected C concentration (%) of water-stable aggregate and mineral size fractions. Values represent means ± standard error. Letters represent significant differences between size fractions within cropping system intensity. No significant cropping system differences were observed. WF=wheat-fallow (n=20); MID=rotations with fallow every 3 or 4 years (n=29); CON=continuous rotations (n=18); 30-yr CRP=Conservation Reserve Program shortgrass prairie strips restored 30 years ago (n=6).

	WF	**MID**	**CON**	**30-yr CRP**
Macroaggregates (>250 um)	2.40±0.03a	2.46±0.03a	2.24±0.05a	2.23±0.08
Microaggregates (53-250um)	1.65±0.02b	1.86±0.02b	2.12±0.04ab	2.92±0.07
Silt and Clay (0-53 um)	1.13±0.02c	1.09±0.01c	1.17±0.02b	0.97±0.02

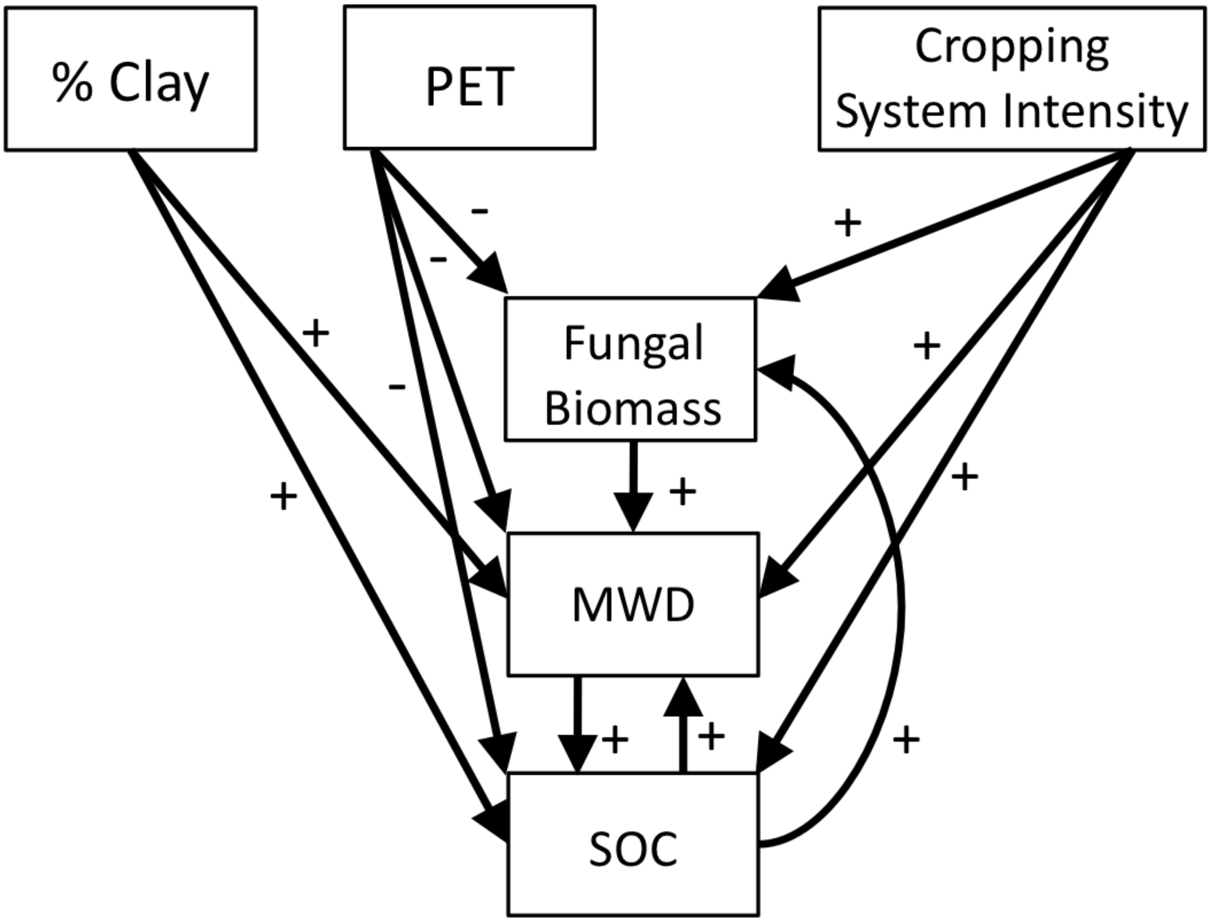

Figure 2.1. Hypothesized relationship between cropping system intensity and the mechanistic drivers of SOC accrual as affected by climate, soil texture, and management. Plus and minus signs denote positive and negative relationships, respectively. MWD=Mean weight diameter of water-stable aggregates, PET=Potential evapotranspiration, SOC=Soil organic carbon.

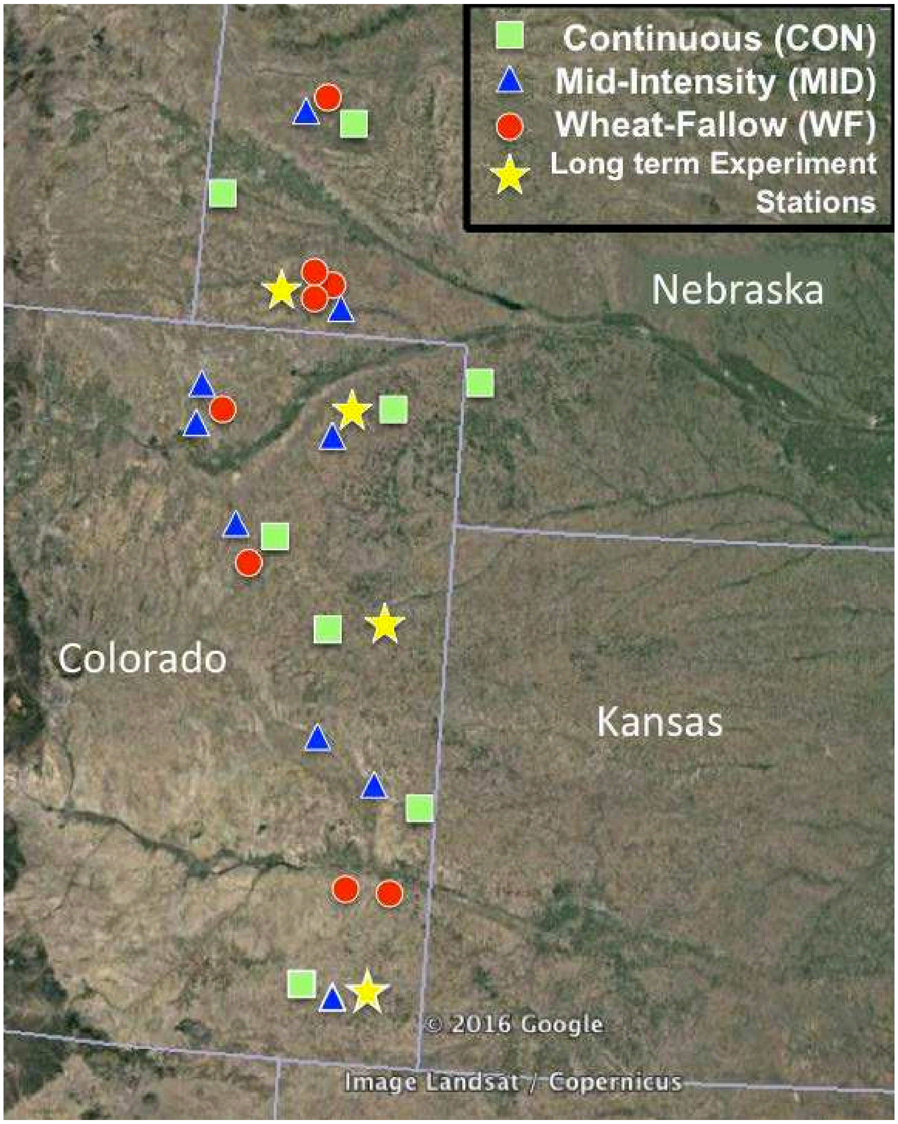

Figure 2.2. Study locations color-coded by cropping system intensity. Multiple fields per location were sampled, and all three levels of cropping system intensity were present at each of the experiment stations.

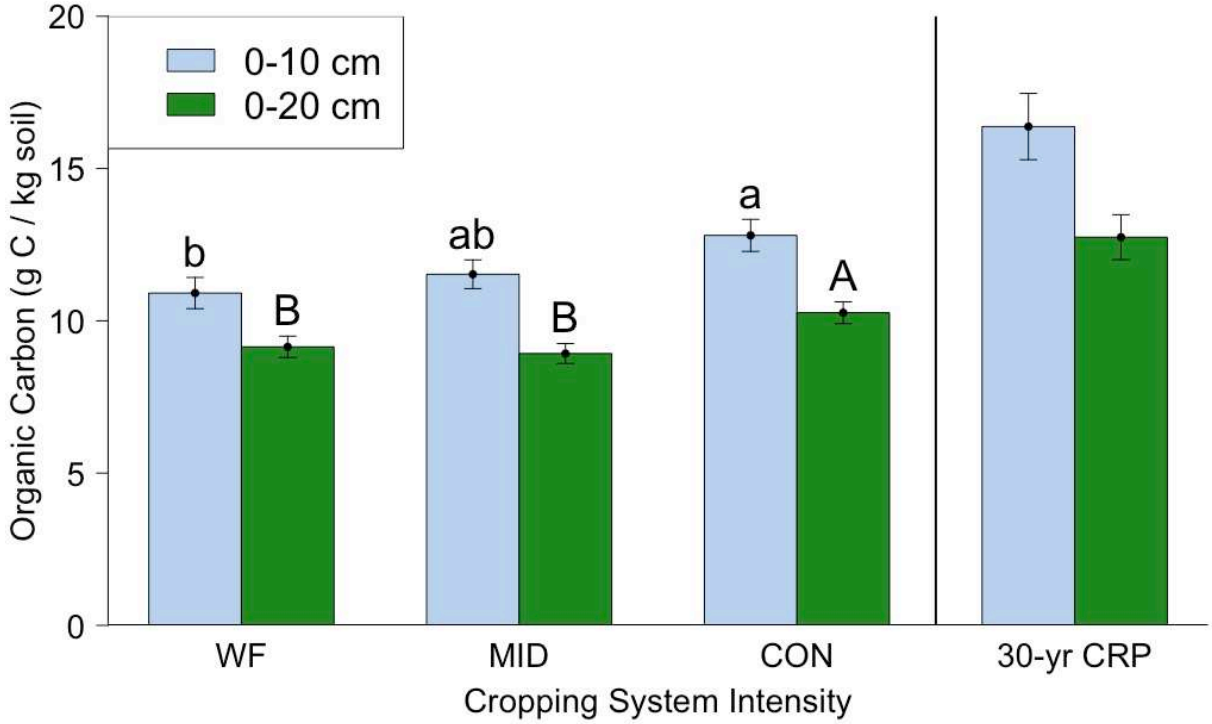

Figure 2.3. Cropping system intensity effects on SOC concentration in the bulk soil. Bar heights and error bars represent model generated least-squared means ± standard error. Significant covariates in models at both depths were % clay, PET, and slope position. Lower case letters represent significant differences between treatments at the 0-10 cm depth ($p<0.05$), and upper case letters represent significant differences between treatments at the 0-20 cm depth ($p<0.05$). WF=wheat-fallow (n=27); MID=rotations with fallow every 3 or 4 years (n=37); CON=continuous rotations (n=26); 30-yr CRP=Conservation Reserve Program shortgrass prairie strips restored 30 years ago (n=6).

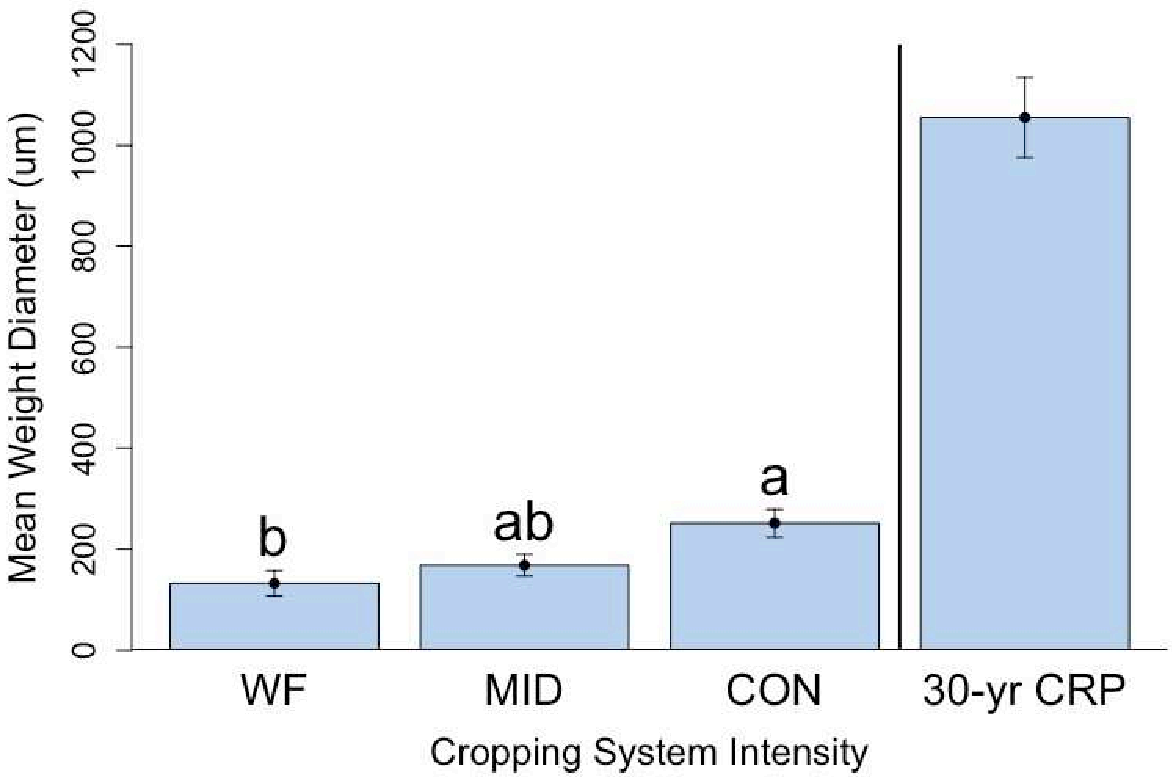

Figure 2.4. Cropping system intensity effects on water stable aggregation assessed via mean weight diameter in surface soils (0-10 cm depth). Bar heights and error bars represent model generated least-squared means ± standard error. PET and % clay were significant covariates in the model. Letters represent significant differences between treatments (p<0.05). WF=wheat-fallow (n=20); MID=rotations with fallow every 3 or 4 years (n=29); CON=continuous rotations (n=18); 30-yr CRP=Conservation Reserve Program shortgrass prairie strips restored 30 years ago (n=6).

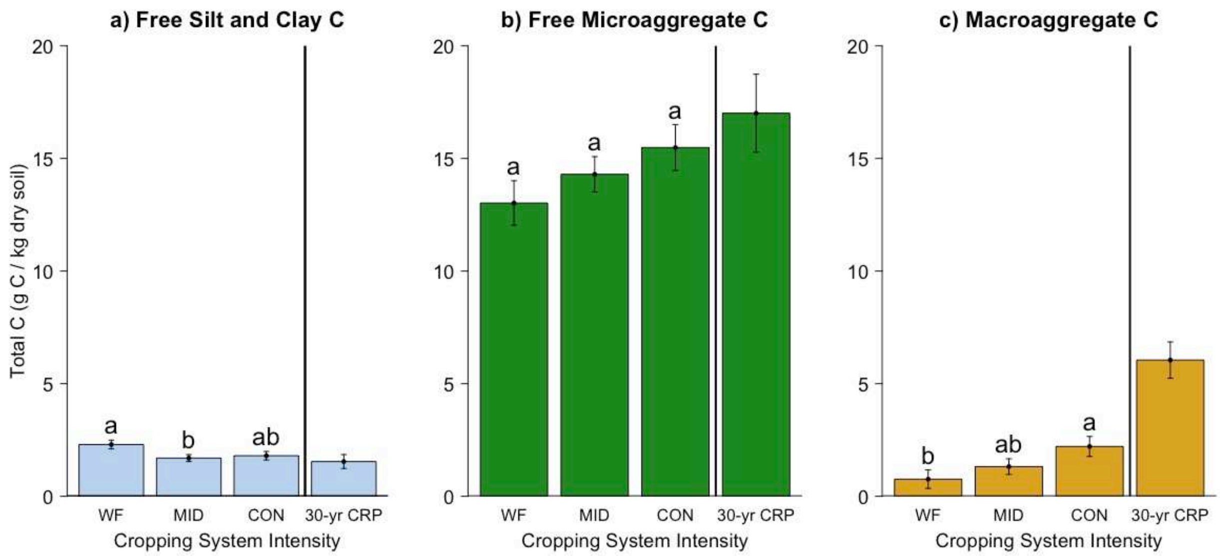

Figure 2.5. Total C content of the: a) free silt and clay, b) free microaggregate, and c) macroaggregate fractions for surface soils (0-10 cm depth). Bar heights and error bars represent model-generated least-squared means + standard error. Letters represent significant differences in C between cropping system intensities. Clay % was a significant covariate in the models for each of the size classes, and years no-till was also a covariate in the model of the free silt and clay fraction. WF=wheat-fallow (n=20); MID=rotations with fallow every 3 or 4 years (n=29); CON=continuous rotations (n=18); 30-yr CRP=Conservation Reserve Program shortgrass prairie strips restored 30 years ago (n=6).

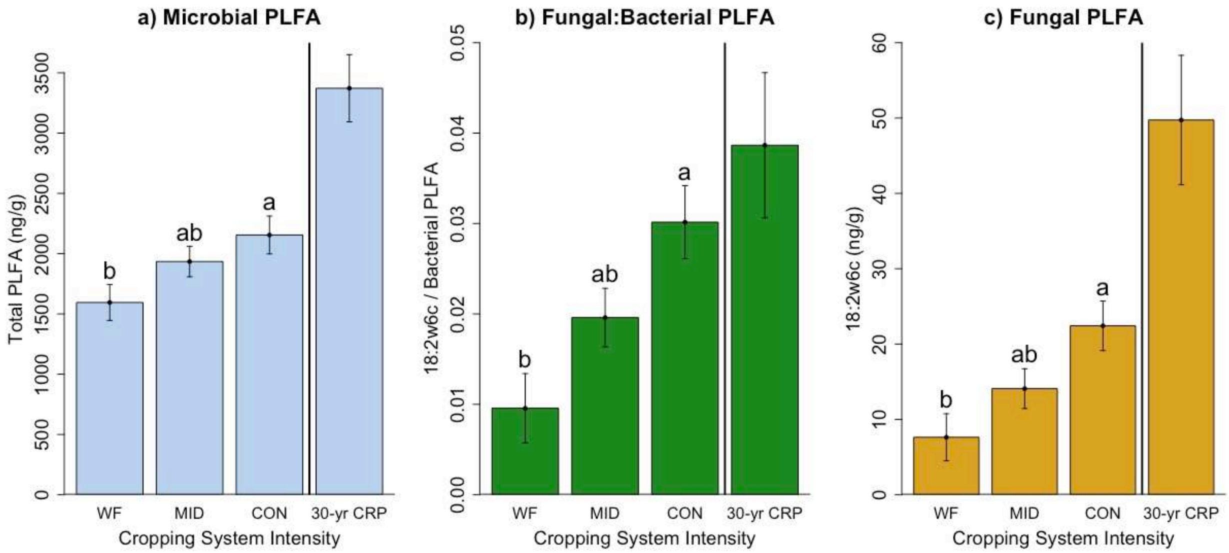

Figure 2.6. Cropping system intensity effects on a) Microbial PLFA, b) Fungal:bacterial PLFA ratio, and c) Fungal PLFA. Bar heights and error bars represent model generated least-squared means $\pm$ standard error. Clay % and pH were significant covariates in the model of total PLFA abundance. Letters represent significant differences between treatments ($p<0.05$). WF=wheat-fallow (n=20); MID=rotations with fallow every 3 or 4 years (n=29); CON=continuous rotations (n=18); 30-yr CRP=Conservation Reserve Program shortgrass prairie strips restored 30 years ago (n=6).

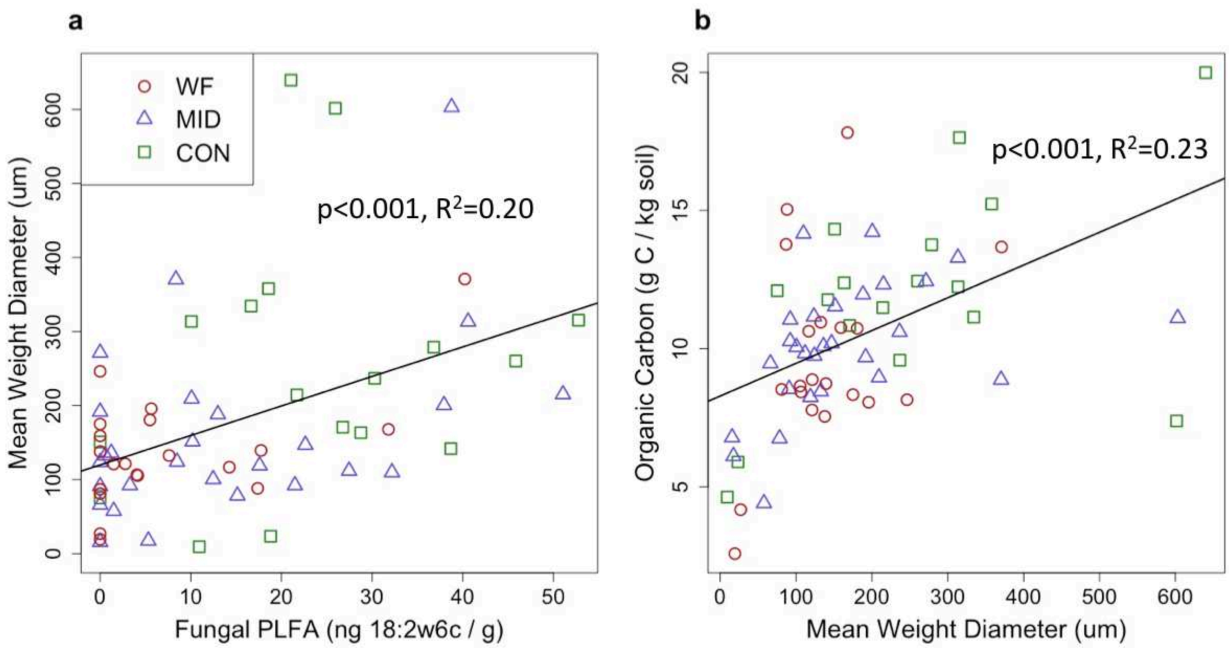

Figure 2.7. a) Relationship between fungal biomass and MWD of water-stable aggregates. MWD=119.78+3.99*fungal PLFA (p<0.001, R0.20), and b) relationship between MWD of water-stable aggregates and SOC concentrations to 10 cm depth. SOC=8.28+0.01*MWD ($p<0.001$, $R^2=0.23$). WF=wheat-fallow (n=20); MID=rotations with fallow every 3 or 4 years (n=29); CON=continuous rotations (n=18); 30-yr CRP=Conservation Reserve Program shortgrass prairie strips restored 30 years ago (n=6).

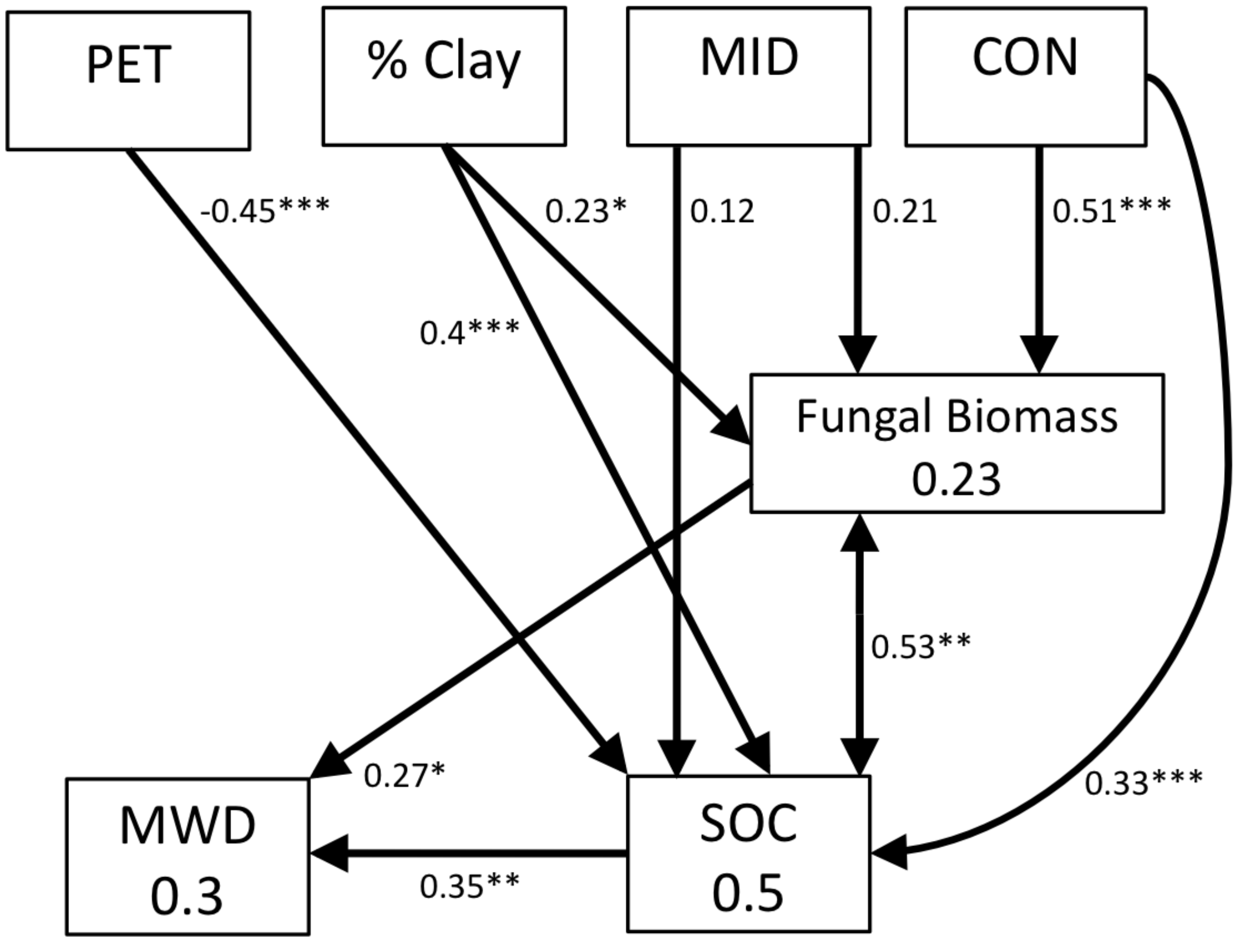

Figure 2.8. Structural equation model. PET=Potential evapotranspiration, MID=rotations with fallow every 3 or 4 years, CON=continuous rotations, Fungal Biomass=Abundance of the fungal PLFA 18:2 w6c, MWD=Mean weight diameter of water-stable aggregates, SOC=Soil organic C concentration. Magnitudes of MID and CON effects are relative to WF as the intercept. Numbers in boxes represent R^2 values and numbers along the arrows are standardized path coefficients, which represent the magnitude of the effect. Significance of each pathway is denoted by ***p<0.001, **p<0.01, *p<0.05, ·p<0.1.

REFERENCES

Acosta-Martinez, V., Mikha, M.M., Vigil, M.F., 2007. Microbial communities and enzyme activities in soils under alternative crop rotations compared to wheat–fallow for the Central Great Plains. Applied Soil Ecology 37, 41-52.

Acosta-Martinez, V., Zobeck, T., Gill, T., Kennedy, A., 2003. Enzyme activities and microbial community structure in semiarid agricultural soils. Biology and Fertility of Soils 38, 216-227.

Bardgett, R.D., McAlister, E., 1999. The measurement of soil fungal: bacterial biomass ratios as an indicator of ecosystem self-regulation in temperate meadow grasslands. Biology and Fertility of Soils 29, 282-290.

Bates, D., Maechler, M., Bolker, B., Walker, S., 2014. lme4: Linear mixed-effects models using Eigen and S4. R package version 1, 1-23.

Bot, A., Nachtergaele, F., Young, A., 2000. Land resource potential and constraints at regional and country levels. . Food & Agriculture Org.

Cotrufo, M.F., Wallenstein, M.D., Boot, C.M., Denef, K., Paul, E., 2013. The Microbial Efficiency-Matrix Stabilization (MEMS) framework integrates plant litter decomposition with soil organic matter stabilization: do labile plant inputs form stable soil organic matter? Global Change Biology 19, 988-995.

De Vries, F.T., Liiri, M.E., Bjørnlund, L., Bowker, M.A., Christensen, S., Setälä, H., Bardgett, R., 2012. Land use alters the resistance and resilience of soil food webs to drought. Nature Climate Change 2, 276-280.

Drenovsky, R., Vo, D., Graham, K., Scow, K., 2004. Soil water content and organic carbon availability are major determinants of soil microbial community composition. Microbial Ecology 48, 424-430.

Elliott, E., 1986. Aggregate structure and carbon, nitrogen, and phosphorus in native and cultivated soils. Soil science society of America journal 50, 627-633.

Engel, R.E., Miller, P.R., McConkey, B.G., Wallander, R., 2017. Soil Organic Carbon Changes to Increasing Cropping Intensity and No-Till in a Semiarid Climate. Soil Science Society of America Journal 81, 404-413.

Farahani, H., Peterson, G., Westfall, D., 1998a. Dryland cropping intensification: A fundamental solution to efficient use of precipitation. Advances in Agronomy (USA).

Farahani, H., Peterson, G., Westfall, D., Sherrod, L., Ahuja, L., 1998b. Soil water storage in dryland cropping systems: the significance of cropping intensification. Soil Science Society of America Journal 62, 984-991.

Franzluebbers, A., 2002. Water infiltration and soil structure related to organic matter and its stratification with depth. Soil and Tillage Research 66, 197-205.

Frostegård, Å., Bååth, E., 1996. The use of phospholipid fatty acid analysis to estimate bacterial and fungal biomass in soil. Biology and Fertility of Soils 22, 59-65.

Grace, J.B., Anderson, T.M., Olff, H., Scheiner, S.M., 2010. On the specification of structural equation models for ecological systems. Ecological Monographs 80, 67-87.

Hansen, N.C., Allen, B.L., Baumhardt, R.L., Lyon, D.J., 2012. Research achievements and adoption of no-till, dryland cropping in the semi-arid US Great Plains. Field Crops Research 132, 196-203.

Hamel, C., Hanson, K., Selles, F., Cruz, A.F., Lemke, R., McConkey, B., Zentner, R., 2006. Seasonal and long-term resource-related variations in soil microbial communities in wheat-based rotations of the Canadian prairie. Soil Biology and Biochemistry 38, 2104-2116.

Harinikumar, K., Bagyaraj, D., 1988. Effect of crop rotation on native vesicular arbuscular mycorrhizal propagules in soil. Plant and Soil 110, 77-80.

Jastrow, J., 1996. Soil aggregate formation and the accrual of particulate and mineral-associated organic matter. Soil Biology and Biochemistry 28, 665-676.

Jastrow, J., Miller, R., Lussenhop, J., 1998. Contributions of interacting biological mechanisms to soil aggregate stabilization in restored prairie. Soil Biology and Biochemistry 30, 905-916.

Kallenbach, C.M., Frey, S.D., Grandy, A.S., 2016. Direct evidence for microbial-derived soil organic matter formation and its ecophysiological controls. Nature Communications 7.

Kline, R.B., 2015. Principles and practice of structural equation modeling. Guilford publications.

Ko, J., Ahuja, L.R., Saseendran, S., Green, T.R., Ma, L., Nielsen, D.C., Walthall, C.L., 2012. Climate change impacts on dryland cropping systems in the Central Great Plains, USA. Climatic Change 111, 445-472.

Kong, A.Y., Six, J., Bryant, D.C., Denison, R.F., Van Kessel, C., 2005. The relationship between carbon input, aggregation, and soil organic carbon stabilization in sustainable cropping systems. Soil Science Society of America Journal 69, 1078-1085.

Koohafkan, P., Stewart, B.A., 2008. Water and cereals in drylands. Earthscan.

Kuznetsova, A., Brockhoff, P.B., Christensen, R.H.B., 2015. Package ‘lmerTest’. R package version 2.

Lal, R., 2004. Carbon sequestration in dryland ecosystems. Environmental Management 33, 528-544.

Lange, M., Habekost, M., Eisenhauer, N., Roscher, C., Bessler, H., Engels, C., Oelmann, Y., Scheu, S., Wilcke, W., Schulze, E.-D., 2014. Biotic and abiotic properties mediating plant diversity effects on soil microbial communities in an experimental grassland. PloS one 9, e96182.

Lenth, R.V., Hervé, M., 2015. lsmeans: Least-squares means. R package version 2, 20-23.

Maaz, T., Wulfhorst, J., McCracken, V., Kirkegaard, J., Huggins, D.R., Roth, I., Kaur, H., Pan, W., 2017. Economic, policy, and social trends and challenges of introducing oilseed and pulse crops into dryland wheat cropping systems. Agriculture, Ecosystems & Environment.

Malik, A.A., Chowdhury, S., Schlager, V., Oliver, A., Puissant, J., Vazquez, P.G., Jehmlich, N., von Bergen, M., Griffiths, R.I., Gleixner, G., 2016. Soil fungal: bacterial ratios are linked to altered carbon cycling. Frontiers in microbiology 7.

Mann, L., 1986. Changes in soil carbon storage after cultivation. Soil Science 142, 279-288.

McDaniel, M., Tiemann, L., Grandy, A., 2014. Does agricultural crop diversity enhance soil microbial biomass and organic matter dynamics? A meta-analysis. Ecological Applications 24, 560-570.

Mikha, M.M., Benjamin, J.G., Vigil, M.F., Nielson, D.C., 2010. Cropping intensity impacts on soil aggregation and carbon sequestration in the central Great Plains. Soil Science Society of America Journal 74, 1712-1719.

Mulumba, L.N., Lal, R., 2008. Mulching effects on selected soil physical properties. Soil and Tillage Research 98, 106-111.

Nelson, R.G., Hellwinckel, C.M., Brandt, C.C., West, T.O., De La Torre Ugarte, D.G., Marland, G., 2009. Energy use and carbon dioxide emissions from cropland production in the United States, 1990–2004. Journal of Environmental Quality 38, 418-425.

Nielsen, D.C., Vigil, M.F., 2010. Precipitation storage efficiency during fallow in wheat-fallow systems. Agronomy Journal 102, 537-543.

Norton, J.B., Mukhwana, E.J., Norton, U., 2012. Loss and recovery of soil organic carbon and nitrogen in a semiarid agroecosystem. Soil Science Society of America Journal 76, 505-514.

Paustian, K., Six, J., Elliott, E., Hunt, H., 2000. Management options for reducing CO2 emissions from agricultural soils. Biogeochemistry 48, 147-163.

Peterson, G., Schlegel, A., Tanaka, D., Jones, O., 1996. Precipitation use efficiency as affected by cropping and tillage systems. Journal of Production Agriculture 9, 180-186.

Peterson, G., Westfall, D., Peairs, F., Sherrod, L., Poss, D., Gangloff, W., Larson, K., Thompson, D., Ahuja, L.R., Koch, M., Walker, C., 2001. Sustainable Dryland Agroecosystem Management. Technical bulletin (Colorado Agricultural Experiment Station); TB-01-2.

Peterson, G.A., Westfall, D.G., 2004. Managing precipitation use in sustainable dryland agroecosystems. Annals of Applied Biology 144, 127-138.

Plaza-Bonilla, D., Arrúe, J.L., Cantero-Martínez, C., Fanlo, R., Iglesias, A., Álvaro-Fuentes, J., 2015. Carbon management in dryland agricultural systems. A review. Agronomy for Sustainable Development 35, 1319-1334.

PRISM, 2017. PRISM Climate Data.

Rosenzweig, S.T., Carson, M.A., Baer, S.G., Blair, J.M., 2016. Changes in soil properties, microbial biomass, and fluxes of C and N in soil following post-agricultural grassland restoration. Applied Soil Ecology 100, 186-194.

Rosseel, Y., 2011. lavaan: an R package for structural equation modeling and more Version 0.4-9 (BETA). Ghent University.

Rousk, J., Frey, S.D., 2015. Revisiting the hypothesis that fungal-to-bacterial dominance characterizes turnover of soil organic matter and nutrients. Ecological Monographs 85, 457-472.

Shaver, T., Peterson, G., Ahuja, L., Westfall, D., Sherrod, L., Dunn, G., 2002. Surface soil physical properties after twelve years of dryland no-till management. Soil Science Society of America Journal 66, 1296-1303.

Shaver, T., Peterson, G., Sherrod, L., 2003. Cropping intensification in dryland systems improves soil physical properties: regression relations. Geoderma 116, 149-164.

Sherrod, L., Dunn, G., Peterson, G., Kolberg, R., 2002. Inorganic carbon analysis by modified pressure-calcimeter method. Soil Science Society of America Journal 66, 299-305.

Sherrod, L., Peterson, G., Westfall, D., Ahuja, L., 2003. Cropping intensity enhances soil organic carbon and nitrogen in a no-till agroecosystem. Soil Science Society of America Journal 67, 1533-1543.

Sherrod, L., Peterson, G., Westfall, D., Ahuja, L., 2005. Soil organic carbon pools after 12 years in no-till dryland agroecosystems. Soil Science Society of America Journal 69, 1600-1608.

Six, J., Elliott, E., Paustian, K., 1999. Aggregate and soil organic matter dynamics under conventional and no-tillage systems. Soil Science Society of America Journal 63, 1350-1358.

Six, J., Frey, S., Thiet, R., Batten, K., 2006. Bacterial and fungal contributions to carbon sequestration in agroecosystems. Soil Science Society of America Journal 70, 555-569.

Six, J., Paustian, K., 2014. Aggregate-associated soil organic matter as an ecosystem property and a measurement tool. Soil Biology and Biochemistry 68, A4-A9.

Smith, E., Young, D.L., 2000. The economic and environmental revolution in semi-arid cropping in North America. Annals of Arid Zone 39, 347-362.

Strickland, M.S., Rousk, J., 2010. Considering fungal: bacterial dominance in soils–methods, controls, and ecosystem implications. Soil Biology and Biochemistry 42, 1385-1395.

Stromberger, M., Shah, Z., Westfall, D., 2007. Soil microbial communities of no-till dryland agroecosystems across an evapotranspiration gradient. Applied Soil Ecology 35, 94-106.

Team, R.C., 2013. R: A language and environment for statistical computing. R foundation for statistical computing, Vienna, Austria.

Thompson, J., 1987. Decline of vesicular-arbuscular mycorrhizae in long fallow disorder of field crops and its expression in phosphorus deficiency of sunflower. Australian Journal of Agricultural Research 38, 847-867.

Tisdall, J.M., Oades, J.M., 1982. Organic matter and water-stable aggregates in soils. Journal of Soil Science 33, 141-163.

UN, 2011. Global drylands: a UN system-wide response. Environment Management Group of the United Nations Geneva. Retrieved from http://www. unccd. int/Lists/SiteDocumentLibrary/Publications/Global_Drylands_Full_Report. pdf.

USGCRP, 2014. Climate change impacts in the United States: U.S. national climate assessment. GlobalChange.gov.

Van Bavel, C., 1950. Mean weight-diameter of soil aggregates as a statistical index of aggregation. Soil Science Society of America Journal 14, 20-23.

Wall, D.H., 2012. Soil ecology and ecosystem services. Oxford University Press.

Waring, B.G., Averill, C., Hawkes, C.V., 2013. Differences in fungal and bacterial physiology alter soil carbon and nitrogen cycling: Insights from meta-analysis and theoretical models. Ecology letters 16, 887-894.

Wilson, G.W., Rice, C.W., Rillig, M.C., Springer, A., Hartnett, D.C., 2009. Soil aggregation and carbon sequestration are tightly correlated with the abundance of arbuscular mycorrhizal fungi: results from long-term field experiments. Ecology letters 12, 452-461.

CHAPTER 3: ECOLOGICALLY BASED NUTRIENT AND WEED MANAGEMENT IN DRYLAND CROPPING SYSTEMS BENEFITS ECONOMICS AND ENVIRONMENT

3.1 Introduction

Global agricultural production has risen steadily over the past several decades and must continue to keep up with rising food demand. At the same time, production practices must shift to reduce nutrient losses and external chemical inputs that threaten ecosystem function and economic performance (Carolan, 2016; Hunter et al., 2017). Rising nutrient requirements associated with increasing crop production have largely been addressed through greater additions of synthetic fertilizers (Tilman et al., 2001). Increasing consumption and costs of nitrogen (N) and phosphorus (P) fertilizer, in addition to rising herbicide use, place an economic burden on farmers, reflected in the widening disparity between low commodity prices and high input costs (Liebman et al., 2001; van der Ploeg, 2006; Fuglie et al., 2007; Carolan, 2016). Net farm incomes since 2013 have declined 50% (ERS, 2017), leading to the decline of full-time farmers and expanding farm sizes (Carolan, 2016). Meanwhile, synthetic N and P fertilizer and herbicides continue to exact high environmental and human costs, evidenced by oceanic dead zones (Diaz and Rosenberg, 2008), groundwater contamination (Sebilo et al., 2013), and degradation of natural ecosystems due to atmospheric N deposition and pesticide pollution (Du et al., 2014; Simkin et al., 2016). Reducing the need for external inputs would reduce impacts on environmental health and increase farmer profits if yields could be sustained. However, N and P are the most limiting nutrients in agricultural production (Chapin, 1991), and the onset of herbicide resistant weeds is driving greater use of even more expensive and herbicides with greater toxicity levels (Mortensen et al., 2012). The challenge, then, is to identify an alternative strategy for maintaining or increasing agricultural production while sustainably managing nutrients and weeds.

This challenge is being played out in an accelerated timeframe in the semi-arid dryland cropping systems of the US High Plains, just one of many semi-arid regions around the world that are undergoing a profound shift in cropping system management (Smith and Young, 2000; Maaz et al., 2017). The rise of no-till has enabled sufficient water savings to reduce the need for year-long bare fallow periods, called summer fallow, that have traditionally characterized dryland cropping systems (Hansen et al., 2012). Large and rapid gains in crop productivity are being realized as dryland farmers replace summer fallow with crops, a practice called cropping system intensification. One of the tradeoffs of these productivity gains is that costly herbicides are required to replace tillage as the primary form of weed control. Additionally, researchers have suggested that even a slight increase in cropping system intensity may increase fertilizer requirements. For example, Kolberg et al. (1996) estimated that in the High Plains, intensifying from the traditional winter wheat (*Triticum aestivum*)-fallow system to a mid-intensity system like wheat-corn (*Zea mays*)-fallow increases the N requirement by 44%. Nutrient requirements may be even greater for continuous cropping systems that have eliminated summer fallow altogether (Grant et al., 2002). As tens of millions of acres in semi-arid regions around the world are intensifying, including millions in the High Plains (Smith and Young, 2000; Hansen et al., 2012; Maaz et al., 2017), there is a need to determine the impact of cropping system intensity on herbicide and fertilizer use in no-till systems.

The challenge of meeting rapidly increasing crop nutrient requirements while mitigating environmental harm necessitates an ecologically based nutrient management strategy. Despite theoretically higher nutrient requirements, cropping intensification may represent such a strategy, as it is associated with greater crop production and subsequent soil carbon (C) inputs, which offers the opportunity to recouple C and N cycles (Drinkwater and Snapp 2007). Cropping system intensification can increase crop production and residue returns to soil by up to 100% (Peterson and Westfall, 2004), which may provide a larger N sink and build reservoirs of organic N that are released more slowly over time compared to highly mobile mineral N pools (Sherrod et al., 2003; Gardner and Drinkwater, 2009). Most applied inorganic fertilizer N is at risk of loss to the environment, as major grain crops only take up ~40% of applied N in the year of application (Gardner and Drinkwater, 2009). The remaining N can leach or run off into waterways leading to degradation of aquatic ecosystems or escape as NH_3 or potent greenhouse gases like N_2O that contribute to climate change (Galloway et al., 2003). When the breakdown of organic material provides most plant N (i.e., when C and N cycles are coupled), more N is retained in the soil because organic N that is mineralized in the rhizosphere has a much lower risk of loss (Gardner and Drinkwater, 2009). Thus, tighter N cycling in agroecosystems limits wasteful and harmful environmental losses of N and may enable farmers to reduce fertilizer use (Drinkwater et al., 1998).

Intensified cropping systems may also stimulate plant uptake of N and P relative to wheat-fallow through changes in soil microbial communities. Wheat-fallow, which is reliant on only one crop and 14 months of bare fallow every other year, represents a highly simplified cropping system. Simplified cropping systems reduce the capacity of the soil to supply plant-available nutrients, and generate greater reliance on inorganic fertilizer (Drinkwater and Snapp, 2007). A lack of internal nutrient cycling capacity creates a “fertilizer treadmill,” wherein mineral fertilizer use begets a greater dependence on mineral fertilizer (Drinkwater and Snapp, 2007). The long summer fallow period that is still prevalent in many dryland cropping systems imposes C limitations on the microbial community that may inhibit its growth and activity. I previously observed that microbial biomass was 20% and 35% greater in mid-intensity and continuous cropping systems in the High Plains relative to wheat-fallow, respectively (Chapter 2), and I suspect that the alleviation of C limitation has significant impacts for plant nutrient availability via impacts on the microbial community. In particular, arbuscular mycorrhizal fungi (AMF) can increase plant nutrient uptake, but are susceptible to "long fallow syndrome" wherein populations of AMF are drastically reduced in the absence of a host plant (Thompson, 1987; Harinikumar and Bagyaraj, 1988). AMF form a symbiotic relationship with most crop types, in which AMF colonize the host plant's roots and trade water and nutrients in return for C (Auge, 2001; Allen, 2007). AMF are particularly effective at increasing their host plant's access to immobile nutrients like P, and higher colonization rates achieved through inoculation have been attributed to increased plant P concentrations and grain yields in semi-arid cropping systems (Al-Karaki et al., 2004). Increasing the availability of C substrates through cropping system intensification may increase populations of AMF and associated benefits to nutrient accessibility, and potentially provide a means of reducing dependence on P fertilizer.

In addition to the potential impacts of agricultural simplification on fertilizer dependency, a lack of crop production and diversity may also foster herbicide dependency (Anderson, 2009). Herbicide use is on the rise globally, indicated by a 15-fold increase in glyphosate consumption over the past two decades (Benbrook, 2016). No-till systems, in particular, require high rates of herbicide application, a reality that is only becoming more severe as weeds continue to develop herbicide resistance (Young, 2006). However, the combination of no-till and cropping intensification maintains surface residue and increases plant competition with weeds, potentially providing an ecological alternative to synthetic herbicide (Derksen et al., 2002; Anderson, 2009; Nichols et al., 2015).

There are serious tradeoffs with risk and yield associated with cropping system intensification in water-limited environments. For example, farmers growing a continuous crop rotation eliminate summer fallow before growing wheat, which can incur a substantial wheat yield penalty and increase the risk of crop failure (Vigil and Nielsen, 1997; Holman et al., 2016). Thus, a wide suite of soil health, yield, and economic impacts of cropping system intensification need to be elucidated so farmers can be fully informed when making difficult crop management decisions. I coupled detailed management and yield data with a suite of soil analyses on working dryland, no-till farms and long-term experiment stations in the High Plains states of Colorado and Nebraska to address the following objectives: 1) Quantify cropping system intensity effects on microbially-mediated nutrient cycling, specifically AMF colonization and potentially mineralizable N (PMN), 2) Quantify cropping system effects on herbicide and fertilizer use; 3) Quantify cropping system intensity effects on crop yields and estimate implications for profitability. I hypothesized that intensification would increase the nutrient supplying capacity of soil, enabling intensified farmers to achieve greater crop production while using fewer inputs, increasing profitability.

3.2 Methods

3.2.1 *Cropping systems*

Wheat-fallow (WF) has traditionally been the dominant dryland cropping system in the semi-arid High Plains (Hansen et al., 2012). This system consists of growing winter wheat from September to July, then bare fallowing for 14 months (summer fallow) until the next wheat planting. No-till farmers in this region often reduce summer fallow frequency from one out of two years (WF), to one out of three or four years (mid-intensity; MID), by rotating winter wheat with crops like corn, sorghum (*Sorghum bicolor*), proso millet (*Panicum miliaceum*), peas (*Pisum sativum*), or sunflowers (*Helianthus annuus*). They may also eliminate summer fallow altogether via continuous cropping (CON), most often through a diverse crop rotation.

3.2.2 *Study sites*

Soil, plant, and management data was gathered from 96 dryland, no-till fields in eastern Colorado and western Nebraska, representing 54 fields from working farms and 42 fields from long-term experiment stations (Fig. 3.1). Each of three levels of cropping intensity – WF (n=27), MID (n=37) and CON (n=26) – were represented along a potential evapotranspiration (PET) gradient that increased from 1368 mm yr^{-1} in northwestern Nebraska to 1975 mm yr^{-1} in southeastern Colorado (Fig. 3.1). To assign a value of PET to each site, a linear equation was used based on known PETs from 6 locations as measured by open-pan evaporation (Peterson et al., 2001) and the latitude of each site. All sites receive 350-450 mm of annual precipitation. At the three long-term experiment stations in Colorado (see Fig. 3.1), total soil N was assessed from both a summit and toeslope position in each plot. Samples on all other fields were taken from a flat topographical position and labeled as a summit. All fields were under tilled WF management for several decades prior to implementation of no-till and the current crop rotation. Every field was planted to winter wheat in the fall of 2015. See Chapter 2 for more detailed site descriptions.

3.2.3 *Soil and plant sampling and analysis*

Soil and plant sampling to assess total N, PMN, AMF colonization of wheat roots, and wheat P concentration was conducted in May and June of 2016 to coincide with wheat heading. AMF colonization is typically highest at wheat heading due to optimal soil temperatures and ample photosynthate supply to roots (Al-Karaki et al., 2004). A 5.5 cm slide-hammer corer was used to take one 0-10 cm depth soil sample from each of four points that form the corners of a 10 x 10 m square on each field and geo-referenced for later samplings. Soils were composited and analyzed for bulk density, texture, and pH as described in Chapter 2. Total N was determined from all 96 fields, while PMN, AMF colonization, and wheat P concentration were determined from a subset of fields (n=51) that were growing Byrd variety winter wheat. About 5-10 whole winter wheat plants from each sampling location were harvested for AMF colonization and P concentration analyses.

I determined total N and PMN on air-dried soils. Soils to assess total N were ground on a roller grinder and analyzed on a LECO CHN-1000 auto analyzer (St. Joseph, MI). PMN was determined using a 7-day anaerobic incubation (Drinkwater et al., 1996). Briefly, soils were sieved to 2 mm and divided into two groups for initial extraction and incubation. Initial extraction soils (5g) were shaken with 50mL 2M KCl for 1 hr, filtered, and extracts were frozen for later analysis. Incubated soils were added to 10 mL of deionized water, flushed with N_2 gas for 1 minute, capped, and stored at 30°C for 7 days before being extracted similar to the initial extraction soils. Samples were analyzed for NH_4^+-N and NO_3^--N colorimetrically using a microplate reader (Biotek, VT) using methods described by Finney et al. (2015), and PMN was calculated as the concentration of NH_4^+-N in the incubated sample minus the NH_4^+-N concentration in the initial extraction sample.

Intact wheat plants were kept cool after harvest, and roots were separated from aboveground biomass. Aboveground biomass was dried at 30°C and shipped to Ward Laboratories in Kearney, NE for analysis of P concentration. Wheat roots were rinsed, cleared with 10% KOH, acidified with 2% HCl, and stained with Trypan blue within 24 hrs following the methods of Phillips and Hayman (1970) and Koske and Gemma (1989). Stained roots were then stored in vials of deionized water at 4°C for subsequent AMF colonization analysis. AMF colonization was assessed following the gridline intersect method (McGonigle et al., 1990). Four replicates for each field were analyzed and later averaged to obtain field-level means of % colonization.

3.2.4 *Input use and partial enterprise budgets*

For each working farm field whose operator was willing to report data (n=42), I collected yearly data from 2010-2014 on N and P fertilizer use, glyphosate, 2,4-D, and dicamba use, and yields for each crop from 2010 to 2015. No field received compost or manure, and amounts of nutrients applied other than N and P were negligible. The net operating income for each cropping system was calculated using partial enterprise budgets. Fertilizer prices for each year from the National Agricultural Statistics Service (NASS, 2017) were converted to $/ha of N and P and multiplied by the amount of N and P applied each year in kg/ha to obtain fertilizer expenditures in $/ha. Amounts of herbicide were converted to acid equivalents (AE)/ha, and multiplied by the 2017 cost of each herbicide in $/AE calculated from University of Nebraska to obtain herbicide expenditures in $/ha (UNL, 2017). Additional expenditures included planting and seed costs, herbicide and fertilizer application costs, and harvest costs, and were estimated for each crop in each year using custom rates as reported by Colorado State University (CSU, 2017). The cost of pea seed was derived from Burgener et al. (2005). Based on results of interviews with the farmers in this study, I assumed that each summer fallow period required 4 separate applications of herbicide.

Annualized grain production was calculated by dividing the total amount of grain production from 2010 to 2014 by 5. To calculate revenues generated from crop production, yields were multiplied by crop prices for each crop in each year (NASS, 2011; 2015; 2017). Net farm operating income was calculated as revenues minus expenses for each year, excluding fixed costs, and then divided by 5 to calculate annualized net operating income in $/ha/yr. Additionally, to assess the wheat yield penalty of continuous cropping relative to summer fallowing, I collected wheat yield data from a broader set of fields (5 to 10 fields per farmer) from 2012 to 2015, and recorded whether the preceding year was summer fallow or cropped.

3.2.5 *Statistical analysis*

The relationships between cropping system intensity and annualized grain yield, wheat yield penalty, fertilizer and herbicide use, net operating income, total N, PMN, and % AMF colonization were tested using multiple linear regression. Models were selected using backwards selection with cropping system intensity as a categorical variable, and all management factors (# years in no-till and # years in rotation) and environmental factors (PET, % clay, pH, and slope) until all remaining terms were significant. To account for environmental and management factors as covariates, least-squared means for each level of cropping system intensity were generated and tested for significant pairwise comparisons. I chose α=0.1 to determine significance because of the inherent variability that cannot be accounted for in an observational study of this magnitude (e.g., weather variability, management decisions not captured in interviews, etc.). The relationships between % AMF colonization and plant P, and between % AMF colonization and PET, were tested using linear regressions. I used R for all data analyses (R Core Team, 2013), and multiple linear regressions were generated and tested for significance using the packages lme4 (Bates et al., 2014), lsmeans (Lenth and Hervé, 2015), and lmerTest (Kuznetsova et al., 2015).

3.3 Results

Overall, I found support for my hypothesis that cropping system intensification in general, and continuous cropping in particular, would enhance the N and P supply capacity of soil by increasing total N and PMN, and fostering AMF colonization correlated with enhanced P uptake. Farmers practicing continuous cropping reduced herbicide use compared to WF, and achieved greater annualized grain production despite applying similar total amounts of fertilizer. Simultaneous production gains and reductions in input use translated into greater estimated net farm operating income for intensified cropping systems.

3.3.1 *Soil and plant analyses*

I observed positive effects of cropping system intensity on total soil N and PMN at a depth of 0-10 cm. Cropping system intensity ($p=0.04$), PET ($p<0.001$), % clay ($p<0.001$), and slope position ($p<0.001$) explained 59% of the variability in total N. After accounting for PET, % clay, and slope as covariates, total N stocks were 12% higher in CON rotations than WF ($p=0.04$), but CON was not significantly different from MID (Fig. 3.2a). Cropping system intensity ($p<0.01$), % clay ($p<0.001$), and an intensity-by-clay interaction ($p=0.02$) explained 47% of the variability in PMN. After accounting for % clay and the intensity-by-clay interaction as covariates, PMN was 30% higher in CON rotations than WF ($p=0.06$), but CON was not significantly different from MID (Fig. 3.2b). The trends and magnitude of the differences in PMN between intensities are similar when the intensity-by-PET interaction is removed from the model (data not shown), indicating that the interpretation of these differences is not confounded by the interaction. Analyzing the interaction between intensity and % clay revealed that % clay had a strong positive effect on PMN in WF rotations ($R^2=0.62$, $p<0.01$), and a slight positive effect in MID rotations ($R^2=0.21$, $p=0.06$), but no effect in CON rotations.

AMF colonization increased with cropping system intensity (Fig. 3.3a), and was negatively impacted by PET (Fig. 3.3b). Cropping system intensity ($p=0.001$), PET ($p=0.47$), and an intensity-by-PET interaction ($p<0.01$) explained 59% of the variability in % AMF colonization. PET was retained in the model due to the significant interaction between PET and intensity. After accounting for PET and the intensity-by-PET interaction as covariates, MID and CON rotations had roughly 2 and 3 times more colonization than WF ($p=0.02$, $p<0.001$), respectively (Fig. 3.3a). Additionally, CON rotations had 54% more AMF colonization than MID rotations ($p=0.01$). The trends and significant differences in AMF colonization between intensities are similar when the intensity-by-PET interaction is removed from the model (data not shown), indicating that the interpretation of these differences is not confounded by the interaction. However, analyzing the interaction between intensity and PET revealed that PET had a stronger negative effect on % AMF colonization in CON rotations ($p<0.001$, $R^2=0.63$) than in MID rotations ($R^2=0.22$, $p=0.03$), and PET had no effect on AMF colonization in WF rotations (Fig. 3.3b). Wheat aboveground biomass P concentrations positively increased with AMF colonization ($R^2=0.16$, $p=0.03$, Fig. 3.3c).

3.3.2 *Inputs, yields, and net operating income*

Annualized fertilizer use from 2010 to 2014 was similar between cropping system intensities, as a result of smaller amounts of fertilizer applied per crop in CON rotations. Cropping system intensity ($p=0.002$) and PET ($p<0.001$) explained 55% of the variability in annualized N fertilizer use. MID rotations applied about 59% more N fertilizer per year (18 kg N/ha/yr) than both CON and WF rotations ($p=0.007$; Fig. 3.4a). In CON rotations, N applied per crop was about 22 and 34 kg N/ha less than WF ($p=0.05$) and MID rotations ($p<0.001$), respectively (Fig. 3.4b). Annualized P fertilizer use and P applied per crop did not differ by cropping system intensity (Fig. 3.4c, 3.4d).

Glyphosate, 2,4-D, and dicamba use from 2010 to 2014 decreased substantially with cropping system intensity (Fig. 3.5). There were no significant covariates in the models of glyphosate and dicamba use by intensity, but pH ($p<0.01$) was a significant covariate in the model of 2,4-D. CON rotations used 50% the amount of glyphosate ($p=0.07$, Fig. 3.5a), 20% the amount of 2,4-D ($p<0.001$), and 32% the amount of dicamba ($p=0.03$) applied in WF rotations. Additionally, MID rotations used 57% ($p<0.01$) and 82% ($p=0.049$) the amount of 2,4-D and dicamba as WF rotations, respectively.

Annualized grain yields from 2010 to 2015 increased with cropping system intensity, despite a wheat yield reduction associated with the elimination of summer fallow (Fig. 3.6). Cropping system intensity ($p<0.001$), N fertilizer use ($p=0.01$), and PET ($p<0.01$) explained 71% of the variability in annualized grain yield. After accounting for N fertilizer and PET as covariates, MID and CON rotations produced 46% and 60% ($p<0.01$) more grain per year than WF, respectively. I separated all wheat yields into two cropping treatments based on whether wheat was preceded by summer fallow or continuous cropped. Cropping treatment ($p<0.001$) and

PET ($p<0.001$) explained 36% of the variability in wheat yields. After accounting for PET as a covariate, on average across 2012 to 2015, wheat that followed a crop yielded 29% less than summer fallowed wheat ($p<0.001$).

I observed positive effects of cropping system intensity on annualized net farm operating income from 2010 to 2014 (Fig. 3.7). Cropping system intensity ($p=0.04$) and PET ($p<0.001$) explained 61% of variability in net farm operating income. After accounting for PET as a covariate, net profits of CON rotations were an estimated \$47/ha/yr (80%) more than WF ($p=0.06$), and MID rotations made \$42/ha/yr (70%) more than WF ($p=0.08$). There was no difference in net operating income between CON and MID rotations.

3.4 Discussion

The results demonstrate that cropping system intensification represents an ecologically based and economically sound strategy for meeting crop nutrient requirements and managing weeds. This seems to hold true across a wide range of soil types, climates, and management styles present throughout the semi-arid High Plains, although the magnitude of some of the benefits of intensification appear to decrease as PET increases.

Enhanced crop production in intensified cropping systems helps to increase soil N retention, indicated by greater total N stocks despite similar levels of fertilizer N input relative to WF (Fig. 3.2). In addition to greater crop production serving as a larger N sink, previous research has demonstrated that greater C inputs to soil associated with cropping system intensification lead to more microbial biomass (Sherrod et al., 2005; Chapter 2). However, increased soil and microbial N stocks do not necessarily signify increased nutrient recycling. Higher soil C:N ratios (Chapter 2) and lower soil nitrate in intensified systems (Westfall et al., 1996) indicate that N limitation increases with intensification, and suggests that N is being rapidly assimilated into microbial biomass and soil organic matter where it may not necessarily be readily available for plant uptake. However, I found higher PMN in continuous systems relative to WF, indicating that the additional N being retained in the soil is potentially available for plant uptake. Enhanced crop diversity can stimulate microbial biomass and nutrient cycling activity (Davis et al., 2012; King and Hofmockel, 2017), and greater cropping system intensity is typically associated with higher crop diversity. Thus, the results suggest that continuous cropping leads to more N in organic pools, which ultimately becomes available through mineralization, suggesting that C and N cycles are coupled in continuous cropping systems. Enhanced retention and internal cycling of N enables continuous crop farmers to realize high levels of crop production while applying less N fertilizer, indicated by the significantly lower amount of N applied per crop and similar or greater annualized grain yield in continuous relative to WF and mid-intensity rotations. Although total N concentrations were higher in continuously cropped soils, and P concentrations were similar between the cropping system intensities, I am unable to determine whether or not farmers who practice continuous cropping are mining soil N or P, as I did not calculate nutrient budgets.

Cropping system intensity also influences nutrient availability via effects on AMF. I found that increasing the presence of a plant host by reducing or eliminating summer fallow greatly increased AMF colonization of winter wheat, which was related to enhanced P uptake. This finding is in agreement with others who have demonstrated the deleterious effect of bare fallows on AMF ((Thompson, 1987; Harinikumar and Bagyaraj, 1988) and subsequent effects on plant P status (Thompson, 1987). Bowman and Halvorson (1997) also found that continuous cropping increased winter wheat P concentrations relative to WF, but they attributed this observation to greater P recycling from plant residues. The findings suggest that AMF are also a critical component of greater P uptake in continuous cropping systems. The beneficial effects of greater P uptake may help plants maintain growth during drought, as the availability of P is greatly reduced when soil moisture is low (Viets Jr., 1972), and greater P uptake can improve plants' drought response due to effects on stomatal conductance and root growth (Auge, 2001). AMF likely also improve precipitation use efficiency in dryland agroecosystems by increasing aggregation, thereby improving water capture (Shaver et al., 2002; 2003; 2012; Gianinazzi et al., 2010; Chapter 2). The importance of AMF to P uptake becomes clear in light of a lack of a relationship between P fertilizer use and plant P concentration. AMF, cropping system intensity, and PET were the only significant factors in models of plant P, and P fertilizer use was not correlated with plant P (data not shown).

In addition to cropping intensity, previous studies have also demonstrated the benefits of input reduction and no-till on populations of AMF (Munyanziza et al., 1997; Corkidi et al., 2002; Gottshall et al., 2017). Verzeaux et al. (2016) found that no-till and the absence of N fertilization benefited AMF populations and led to similar rates of N uptake and aboveground biomass in winter wheat relative to conventional N fertilization. I found no relationship between AMF colonization and N or P fertilization or the number of years in no-till, possibly due to relatively low inputs in dryland systems relative to temperate systems where these relationships are often examined. Additionally, even though all fields in this study had been in no-till management for at least 5 years, most wheat-fallow systems had very low AMF colonization (<6%). These results suggest that cropping system intensification, independent of input use and tillage system, is a necessary condition for enhancing AMF colonization in dryland agroecosystems without the use of inocula. However, the magnitude of the benefit of intensification decreased as PET increased.

This is likely due to overall smaller plants in hotter and drier climates, which would reduce the supply of C to AMF, and more severe water limitations that restrict AMF growth (Auge, 2001).

While summer fallow can stabilize wheat yields from year to year, it requires an extensive amount of herbicide or tillage to control weeds. Herbicide costs often make up 20-30% of a producer's input costs in the Great Plains (Derksen et al., 2002), and there is growing recognition of herbicides' role in ecosystem degradation and danger to human health (Liebman et al., 2001; Gilliom et al., 2006; Mortensen et al., 2012). Early studies on weed management in dryland cropping systems sought to control weeds via reduced tillage, but found that weeds persisted in summer fallow regardless of tillage management (Fenster et al., 1968). More recently, the importance of crop rotation, and specifically the synergistic effects of combined crop rotation and no-till effects have come to light (Anderson, 2009). Cropping system intensification in this study enabled no-till, mid-intensity and continuous farmers to reduce consumption of three of the most prolific herbicides in dryland grain production by 30% and 60%, respectively, relative to no-till wheat-fallow (Fig. 3.5). At the same time, the results lend support to mounting evidence that ecological approaches to weed management can provide effective weed control and be more profitable than herbicide-only based approaches (Anderson, 2003; Pimentel et al., 2005; Liebman et al., 2008; Davis et al., 2012).

The differences in profitability between the cropping systems, reported as net operating income, reflect a larger trend in the agricultural economy. For 50 years, the cost of inputs has risen while commodity prices have stagnated (Fuglie et al., 2007), placing a "double squeeze" on farmer profits (van der Ploeg, 2006; Carolan, 2016). Continuous farmers, and mid-intensity farmers to a lesser degree, were able to increase yields while reducing inputs. This enabled them to act on both sides of the double squeeze, which led to greater profitability (Fig. 3.7). Often, profitability of dryland cropping systems in the High Plains is conflated with wheat yields, likely due to the historic significance of wheat in this region (Nielsen et al., 2015; Holman et al., 2016). However, although I found a 29% wheat yield reduction associated with continuous cropping, these results demonstrate that wheat yields were not synonymous with profitability due to the expenses of maintaining summer fallow. The multi-year, systems-based approach demonstrates the benefits provided by growing more crops rather than maximizing yield of a single crop.

3.4.1 *Conclusion*

Addressing the global economic and environmental challenges associated with rising fertilizer and herbicide use requires the establishment of internal capacity in agroecosystems to supply nutrients and control weeds. This capacity has been all but lost in simplified agroecosystems, including those reliant on summer fallow that once dominated many semi-arid climates. However, semi-arid regions around the world are witnessing a "revolution" in dryland cropping (Smith and Young, 2000), in which no-till is enabling farmers to intensify and diversify crop rotations. I found support for the hypothesis that continuous cropping in the High Plains can increase N retention and cycling and P uptake by plants, mediated by increased associations with AMF. This enhanced capacity to supply nutrients in continuously cropped soils enabled continuous dryland farmers to achieve more grain production using the same amount of fertilizers compared to those practicing wheat-fallow. Additionally, the combination of no-till and reduced summer fallow enabled intensified farmers to drastically reduce herbicide use. Overall, I conclude that cropping system intensification represents an opportunity to achieve more grain production while managing weeds and nutrients with fewer external inputs, leading to greater profitability and better environmental outcomes.

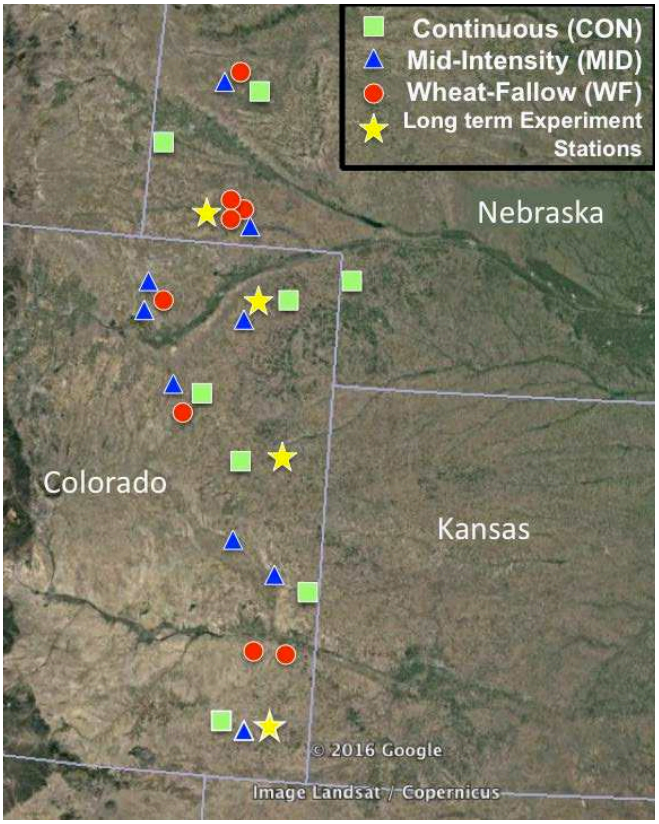

Figure 3.1. Study locations color-coded by cropping system intensity. Multiple fields per location were sampled, and all three levels of cropping system intensity were present at each of the experiment stations.

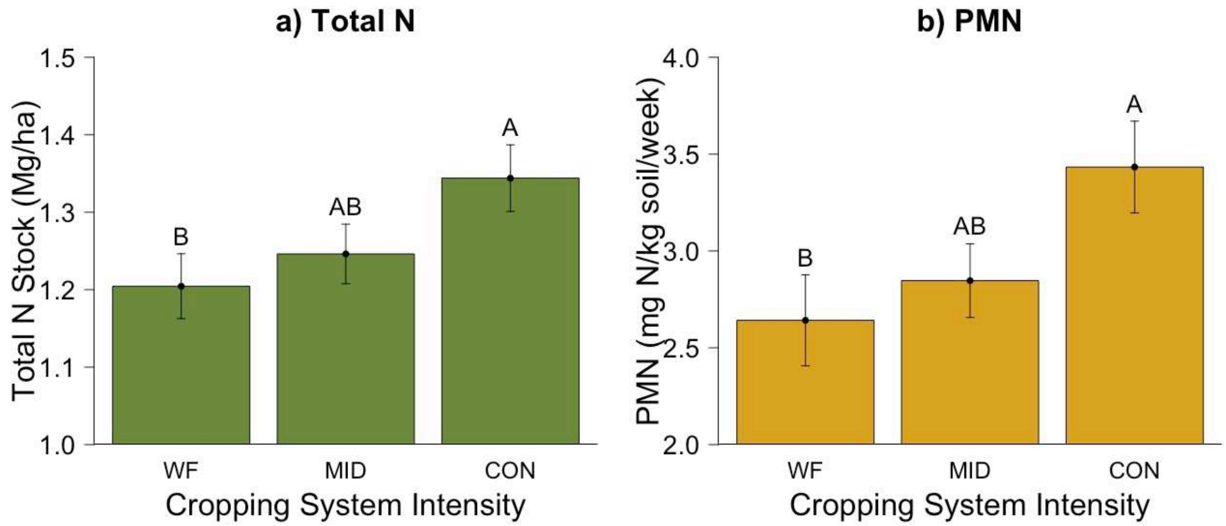

Figure 3.2. Cropping system intensity effects on a) Total N, and b) potentially mineralizable N (PMN). Bar heights and error bars represent model generated least-squared means ± standard error. Clay, pH, PET, and slope were significant covariates in the model of total N, and clay was a significant covariate in the model of PMN. Letters represent significant differences between treatments (p<0.1). WF=wheat-fallow; MID=rotations with summer fallow every 3 or 4 years; CON=continuous rotations.

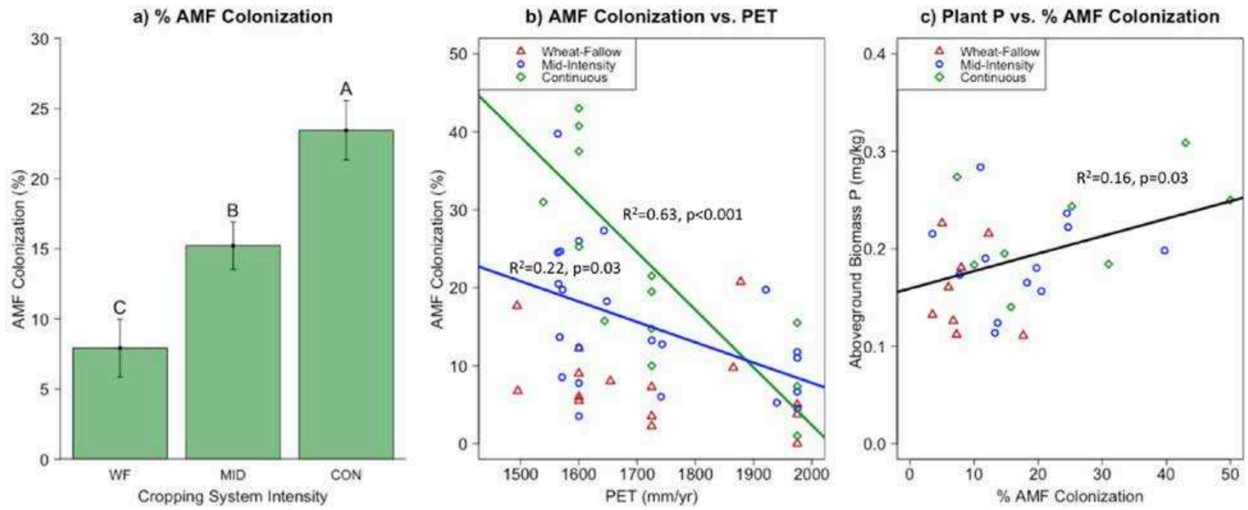

Figure 3.3. a) Cropping system effects on % AMF colonization of winter wheat roots, b) the relationship between % AMF colonization and potential evapotranspiration (PET), separated by cropping system intensity, and c) the relationship between wheat P concentration and % AMF colonization, separated by cropping system intensity. Bar heights and error bars represent model generated least-squared means ± standard error. PET and a PET-by-intensity interaction were significant covariates in the model of % AMF colonization. Letters represent significant differences between treatments ($p<0.1$). WF=wheat-fallow; MID=rotations with summer fallow every 3 or 4 years; CON=continuous rotations.

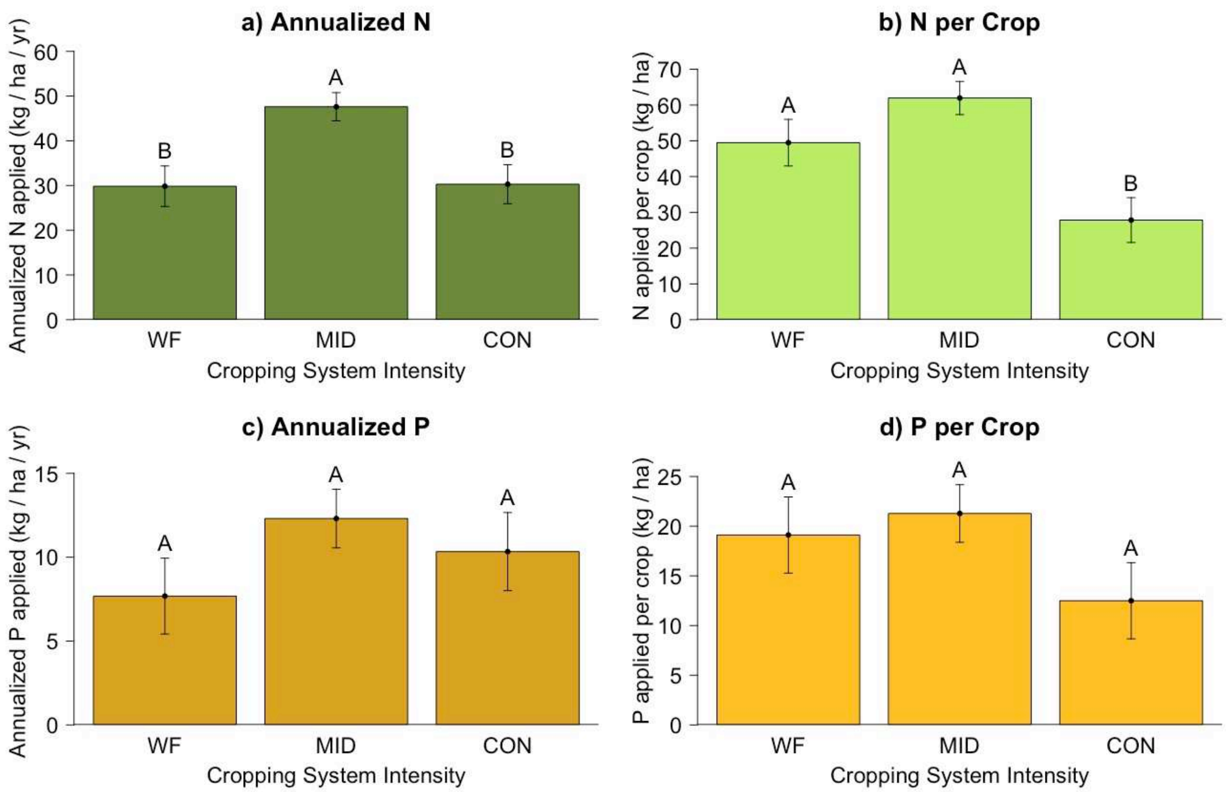

Figure 3.4. Fertilizer nitrogen (N) and phosphorus (P) use by cropping system intensity from 2010 to 2014. Annualized N and P use was calculated as the total amount of N and P applied over the 5-year period divided by 5. N and P applied per crop were calculated as the total amount of N and P applied over the 5-year period divided by the number of years in which a crop was planted. Bar heights and error bars represent model generated least-squared means $\pm$ standard error. PET was a significant covariate in the model of annualized N, N per crop, and annualized P. Letters represent significant differences between treatments ($p<0.1$). WF=wheat-fallow; MID=rotations with summer fallow every 3 or 4 years; CON=continuous rotations.

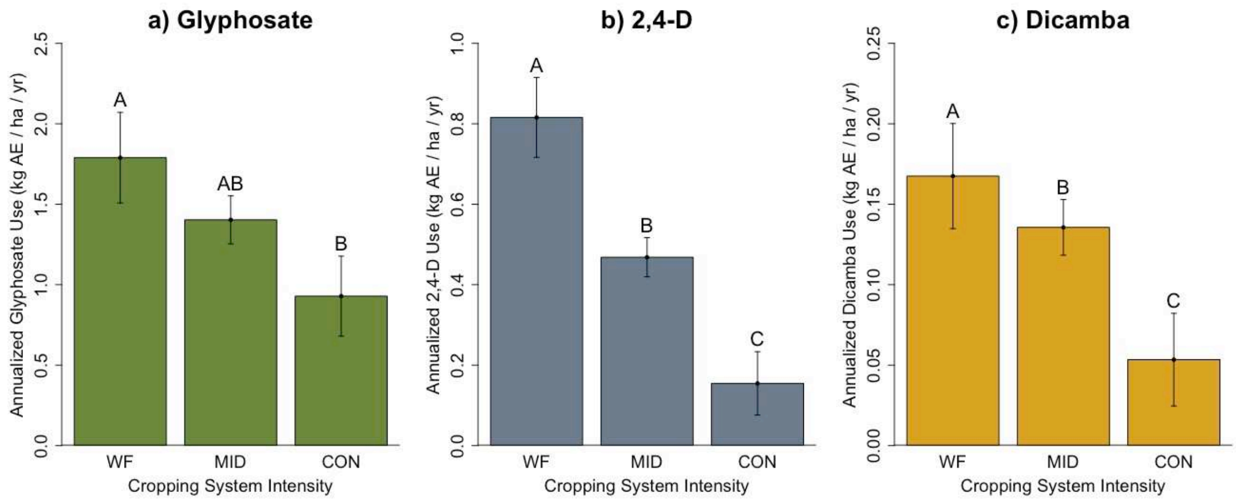

Figure 3.5. Cropping system intensity effects on acid equivalents (AE) of a) glyphosate, b) 2,4-D, and c) dicamba applied from 2010 to 2014. Bar heights and error bars represent model generated least-squared means $\pm$ standard error. pH was a significant covariate in the model of 2,4-D use. Letters represent significant differences between treatments ($p<0.1$). WF=wheat-fallow; MID=rotations with summer fallow every 3 or 4 years; CON=continuous rotations.

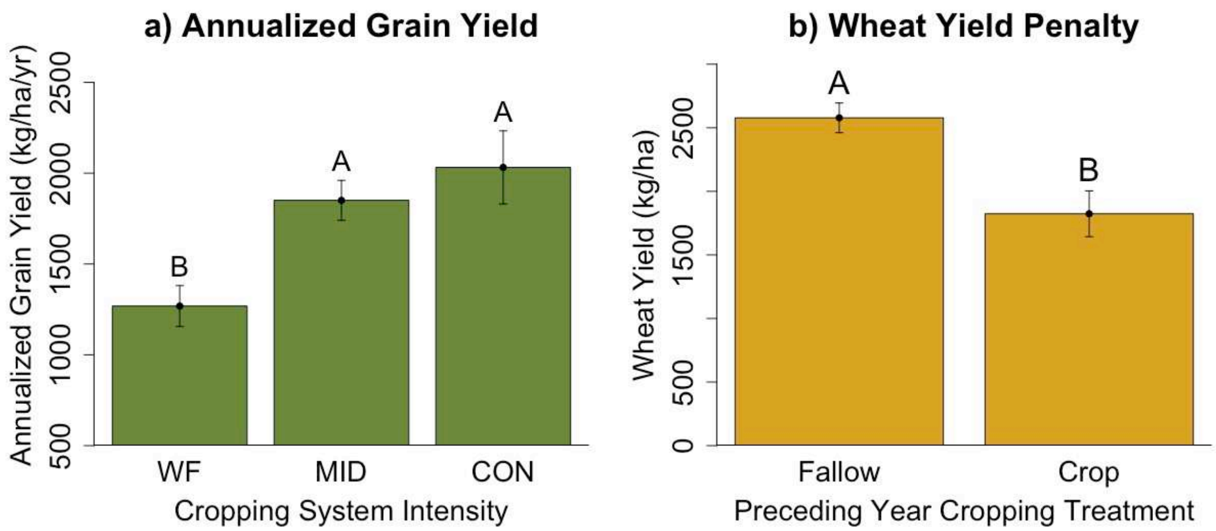

Figure 3.6. Cropping system intensity effects on a) annualized grain yield from 2010 to 2015, and b) continuous cropping effects on wheat yields relative to summer fallow from 2012 to 2015. Annualized grain yield was calculated as the total amount of grain production from 2010 to 2015 divided by 6. Bar heights and error bars represent model generated least-squared means $\pm$ standard error. Annualized N fertilizer applied and PET were significant covariates in the model of annualized grain yield, and PET was a significant covariate in the model of wheat yields. Letters represent significant differences between treatments ($p<0.1$). WF=wheat-fallow; MID=rotations with summer fallow every 3 or 4 years; CON=continuous rotations.

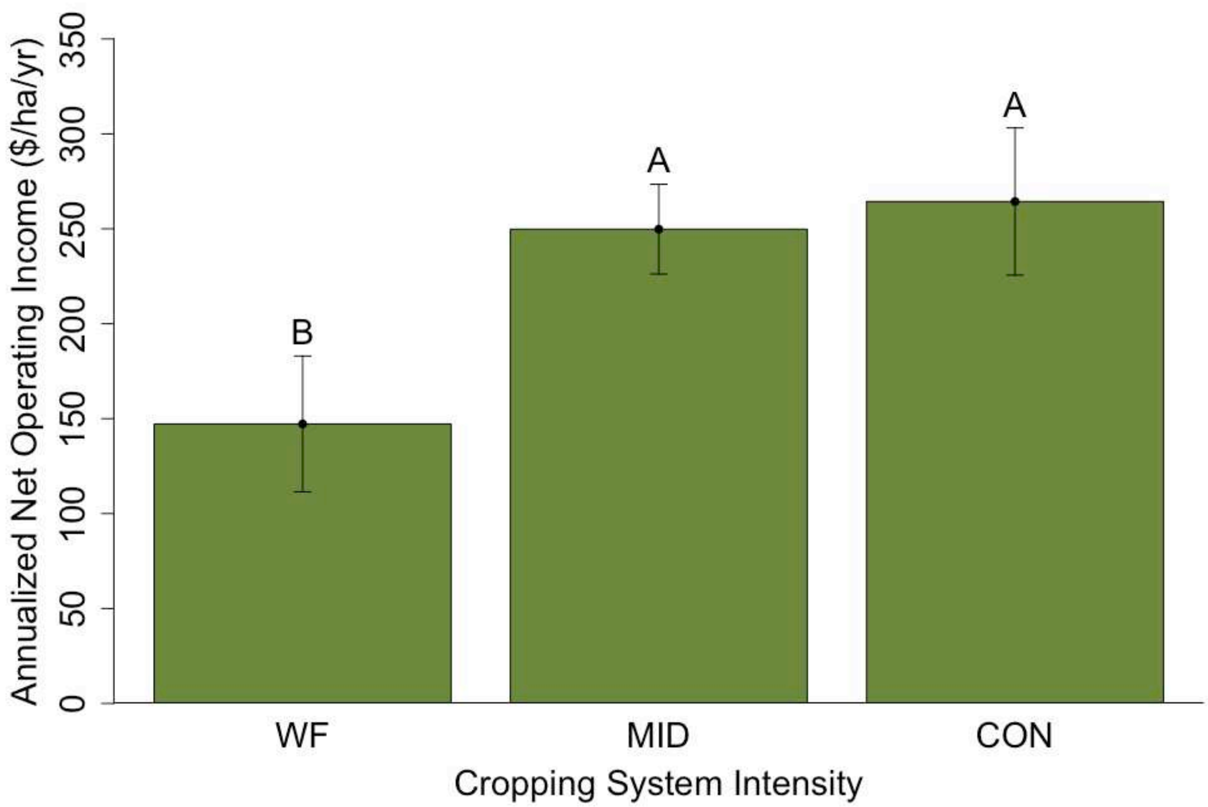

Figure 3.7. Cropping system intensity effects on annualized net operating income in $ from 2010 to 2014. Annualized net operating income was calculated as the net operating income over the 5-year period divided by 5. Bar heights and error bars represent model generated least-squared means ± standard error. PET was a significant covariate in the model net operating income. Letters represent significant differences between treatments (p<0.1). WF=wheat-fallow; MID=rotations with summer fallow every 3 or 4 years; CON=continuous rotations.

REFERENCES

Al-Karaki, G., McMichael, B., Zak, J., 2004. Field response of wheat to arbuscular mycorrhizal fungi and drought stress. Mycorrhiza 14, 263-269.

Allen, M.F., 2007. Mycorrhizal fungi: highways for water and nutrients in arid soils. Vadose Zone Journal 6, 291-297.

Anderson, R., 2003. An ecological approach to strengthen weed management in the semiarid Great Plains. Advances in Agronomy 80, 33-62.

Anderson, R.L., 2009. Rotation design: A critical factor for sustainable crop production in a semiarid climate: A review. Organic Farming, Pest Control and Remediation of Soil Pollutants. Springer, pp. 107-121.

Augé, R.M., 2001. Water relations, drought and vesicular-arbuscular mycorrhizal symbiosis. Mycorrhiza 11, 3-42.

Bates, D., Maechler, M., Bolker, B., Walker, S., 2014. lme4: Linear mixed-effects models using Eigen and S4. R package version 1, 1-23.

Benbrook, C.M., 2016. Trends in glyphosate herbicide use in the United States and globally. Environmental Sciences Europe 28, 3.

Bowman, R., Halvorson, A., 1997. Crop rotation and tillage effects on phosphorus distribution in the central Great Plains. Soil Science Society of America Journal 61, 1418-1422.

Burgener, P.A., Baltensperger, D.D., Feuz, D.M., 2005. Alternative Crop Budgets and Decision Making. University of Nebraska Lincoln - Digital Commons.

Carolan, M., 2016. The Sociology of Food and Agriculture. Taylor & Francis.

Chapin III, F.S., Vitousek, P.M., Van Cleve, K., 1986. The nature of nutrient limitation in plant communities. The American Naturalist 127, 48-58.

Corkidi, L., Rowland, D.L., Johnson, N.C., Allen, E.B., 2002. Nitrogen fertilization alters the functioning of arbuscular mycorrhizas at two semiarid grasslands. Plant and Soil 240, 299-310.

CSU, 2017. Crop Enterprise Budgets.

Davis, A., Hill, J., Chase, C., Johanns, A., Liebman, M., 2012. Increasing Cropping System Diversity Balances Productivity. Profitability and Environmental.

Derksen, D.A., Anderson, R.L., Blackshaw, R.E., Maxwell, B., 2002. Weed dynamics and management strategies for cropping systems in the northern Great Plains. Agronomy Journal 94, 174-185.

Diaz, R.J., Rosenberg, R., 2008. Spreading dead zones and consequences for marine ecosystems. science 321, 926-929.

Drinkwater, L.E., Cambardella, C.A., Reeder, J.D., Rice, C.W., 1996. Potentially mineralizable nitrogen as an indicator of biologically active soil nitrogen. Methods for assessing soil quality, 217-229.

Drinkwater, L.E., Snapp, S., 2007. Nutrients in agroecosystems: rethinking the management paradigm. Advances in Agronomy 92, 163-186.

Drinkwater, L.E., Wagoner, P., Sarrantonio, M., 1998. Legume-based cropping systems have reduced carbon and nitrogen losses. Nature 396, 262-265.

Du, E., de Vries, W., Galloway, J.N., Hu, X., Fang, J., 2014. Changes in wet nitrogen deposition in the United States between 1985 and 2012. Environmental Research Letters 9, 095004.

Fenster, C., Domingo, C., Burnside, O., 1969. Weed control and plant residue maintenance with various tillage treatments in a winter wheat-fallow rotation. Agronomy Journal 61, 256-259.

Finney, D.M., Eckert, S.E., Kaye, J.P., 2015. Drivers of nitrogen dynamics in ecologically based agriculture revealed by long-term, high-frequency field measurements. Ecological Applications 25, 2210-2227.

Fuglie, K., MacDonald, J.M., Ball, V.E., 2007. Productivity growth in US agriculture.

Galloway, J.N., Aber, J.D., Erisman, J.W., Seitzinger, S.P., Howarth, R.W., Cowling, E.B., Cosby, B.J., 2003. The nitrogen cascade. Bioscience 53, 341-356.

Gardner, J.B., Drinkwater, L.E., 2009. The fate of nitrogen in grain cropping systems: a meta-analysis of 15N field experiments. Ecological Applications 19, 2167-2184.

Gianinazzi, S., Gollotte, A., Binet, M.-N., van Tuinen, D., Redecker, D., Wipf, D., 2010. Agroecology: the key role of arbuscular mycorrhizas in ecosystem services. Mycorrhiza 20, 519-530.

Gilliom, R.J., Barbash, J.E., Crawford, C.G., Hamilton, P.A., Martin, J.D., Nakagaki, N., Nowell, L.H., Scott, J.C., Stackelberg, P.E., Thelin, G.P., 2006. Pesticides in the nation's streams and ground water, 1992-2001. US Geological Survey.

Gottshall, C.B., Cooper, M., Emery, S.M., 2017. Activity, diversity and function of arbuscular mycorrhizae vary with changes in agricultural management intensity. Agriculture, Ecosystems & Environment 241, 142-149.

Grant, C.A., Peterson, G.A., Campbell, C.A., 2002. Nutrient considerations for diversified cropping systems in the northern Great Plains. Agronomy Journal 94, 186-198.

Hansen, N.C., Allen, B.L., Baumhardt, R.L., Lyon, D.J., 2012. Research achievements and adoption of no-till, dryland cropping in the semi-arid US Great Plains. Field Crops Research 132, 196-203.

Harinikumar, K., Bagyaraj, D., 1988. Effect of crop rotation on native vesicular arbuscular mycorrhizal propagules in soil. Plant and Soil 110, 77-80.

Holman, J., Roberts, T., Maxwell, S., 2016. Fallow Replacement Crop (Cover Crops, Annual Forages, and Short-Season Grain Crops) Effects on Wheat and Grain Sorghum Yields. Kansas Agricultural Experiment Station Research Reports 2, 2.

Hunter, M.C., Smith, R.G., Schipanski, M.E., Atwood, L.W., Mortensen, D.A., 2017. Agriculture in 2050: Recalibrating Targets for Sustainable Intensification. BioScience 67, 386-391.

King, A.E., Hofmockel, K.S., 2017. Diversified cropping systems support greater microbial cycling and retention of carbon and nitrogen. Agriculture, Ecosystems & Environment 240, 66-76.

Kolberg, R., Kitchen, N., Westfall, D., Peterson, G., 1996. Cropping intensity and nitrogen management impact of dryland no-till rotations in the semi-arid western Great Plains. Journal of Production Agriculture 9, 517-521.

Koske, R., Gemma, J., 1989. A modified procedure for staining roots to detect VA mycorrhizas. Mycological research 92, 486-488.

Kuznetsova, A., Brockhoff, P.B., Christensen, R.H.B., 2015. Package ‘lmerTest’. R package version 2.

Lenth, R.V., Hervé, M., 2015. lsmeans: Least-squares means. R package version 2, 20-23.

Liebman, M., Gibson, L.R., Sundberg, D.N., Heggenstaller, A.H., Westerman, P.R., Chase, C.A., Hartzler, R.G., Menalled, F.D., Davis, A.S., Dixon, P.M., 2008. Agronomic and economic performance characteristics of conventional and low-external-input cropping systems in the central Corn Belt. Agronomy Journal 100, 600-610.

Liebman, M., Mohler, C.L., Staver, C.P., 2001. Ecological management of agricultural weeds. Cambridge University Press.

Maaz, T., Wulfhorst, J., McCracken, V., Kirkegaard, J., Huggins, D.R., Roth, I., Kaur, H., Pan, W., 2017. Economic, policy, and social trends and challenges of introducing oilseed and pulse crops into dryland wheat cropping systems. Agriculture, Ecosystems & Environment.

McGonigle, T., Miller, M., Evans, D., Fairchild, G., Swan, J., 1990. A new method which gives an objective measure of colonization of roots by vesicular—arbuscular mycorrhizal fungi. New phytologist 115, 495-501.

Mortensen, D.A., Egan, J.F., Maxwell, B.D., Ryan, M.R., Smith, R.G., 2012. Navigating a critical juncture for sustainable weed management. BioScience 62, 75-84.

Munyanziza, E., Kehri, H., Bagyaraj, D., 1997. Agricultural intensification, soil biodiversity and agro-ecosystem function in the tropics: the role of mycorrhiza in crops and trees. Applied Soil Ecology 6, 77-85.

Nichols, V., Verhulst, N., Cox, R., Govaerts, B., 2015. Weed dynamics and conservation agriculture principles: a review. Field Crops Research 183, 56-68.

Nielsen, D.C., Vigil, M.F., 2005. Legume green fallow effect on soil water content at wheat planting and wheat yield. Agronomy journal 97, 684-689.

Peterson, G., Westfall, D., Peairs, F., Sherrod, L., Poss, D., Gangloff, W., Larson, K., Thompson, D., Ahuja, L.R., Koch, M., Walker, C., 2001. Sustainable Dryland Agroecosystem Management. Technical bulletin (Colorado Agricultural Experiment Station); TB-01-2.

Peterson, G.A., Westfall, D.G., 2004. Managing precipitation use in sustainable dryland agroecosystems. Annals of Applied Biology 144, 127-138.

Phillips, J.M., Hayman, D., 1970. Improved procedures for clearing roots and staining parasitic and vesicular-arbuscular mycorrhizal fungi for rapid assessment of infection. Transactions of the British mycological Society 55, 158IN116-161IN118.

Pimentel, D., Hepperly, P., Hanson, J., Douds, D., Seidel, R., 2005. Environmental, energetic, and economic comparisons of organic and conventional farming systems. BioScience 55, 573-582.

Sebilo, M., Mayer, B., Nicolardot, B., Pinay, G., Mariotti, A., 2013. Long-term fate of nitrate fertilizer in agricultural soils. Proceedings of the National Academy of Sciences 110, 18185-18189.

Shaver, T., Peterson, G., Ahuja, L., Westfall, D., Sherrod, L., Dunn, G., 2002. Surface soil physical properties after twelve years of dryland no-till management. Soil Science Society of America Journal 66, 1296-1303.

Shaver, T., Peterson, G., Ahuja, L.R., Westfall, D., 2013. Soil sorptivity enhancement with crop residue accumulation in semiarid dryland no-till agroecosystems. Geoderma 192, 254-258.

Shaver, T., Peterson, G., Sherrod, L., 2003. Cropping intensification in dryland systems improves soil physical properties: regression relations. Geoderma 116, 149-164.

Sherrod, L., Peterson, G., Westfall, D., Ahuja, L., 2003. Cropping intensity enhances soil organic carbon and nitrogen in a no-till agroecosystem. Soil Science Society of America Journal 67, 1533-1543.

Sherrod, L., Peterson, G., Westfall, D., Ahuja, L., 2005. Soil organic carbon pools after 12 years in no-till dryland agroecosystems. Soil Science Society of America Journal 69, 1600-1608.

Simkin, S.M., Allen, E.B., Bowman, W.D., Clark, C.M., Belnap, J., Brooks, M.L., Cade, B.S., Collins, S.L., Geiser, L.H., Gilliam, F.S., 2016. Conditional vulnerability of plant diversity to atmospheric nitrogen deposition across the United States. Proceedings of the National Academy of Sciences 113, 4086-4091.

Smith, E., Young, D.L., 2000. The economic and environmental revolution in semi-arid cropping in North America. Annals of Arid Zone 39, 347-362.

Team, R.C., 2013. R: A language and environment for statistical computing. R foundation for statistical computing, Vienna, Austria.

Thompson, J., 1987. Decline of vesicular-arbuscular mycorrhizae in long fallow disorder of field crops and its expression in phosphorus deficiency of sunflower. Australian Journal of Agricultural Research 38, 847-867.

Tilman, D., Fargione, J., Wolff, B., D'Antonio, C., Dobson, A., Howarth, R., Schindler, D., Schlesinger, W.H., Simberloff, D., Swackhamer, D., 2001. Forecasting agriculturally driven global environmental change. Science 292, 281-284.

UNL, 2017. 2017 Approximate Retail Price ($) Per Unit of Selected Herbicides for Field Crops.

USDA-ERS, 2017. Farm Household Income Forecast. Farm Household Well-being.

USDA-NASS, 2013. Crop Values 2012 Summary.

USDA-NASS, 2015. Crop Values 2014 Summary.

USDA-NASS, 2017a. Crop Values 2016 Summary.

USDA-NASS, 2017b. Quick Stats.

Van der Ploeg, J.D., 2006. Agricultural production in crisis. Handbook of rural studies, 258-277.

Verzeaux, J., Roger, D., Lacoux, J., Nivelle, E., Adam, C., Habbib, H., Hirel, B., Dubois, F., Tetu, T., 2016. In winter wheat, no-till increases mycorrhizal colonization thus reducing the need for nitrogen fertilization. Agronomy 6, 38.

Vietz, F., 1972. Water deficits and nutrient availability. Water deficits and plant growth 101, 217-239.

Westfall, D., Havlin, J., Hergert, G., Raun, W., 1996. Nitrogen management in dryland cropping systems. Journal of Production Agriculture 9, 192-199.

Young, B.G., 2006. Changes in Herbicide Use Patterns and Production Practices Resulting from Glyphosate-Resistant Crops1. Weed Technology 20, 301-307.

CHAPTER 4: THE SEMI-ARID CROPPING REVOLUTION: A CASE OF CONTESTED TRUTHS AND CONFLICTING IMAGINARIES

4.1 Introduction

Agriculture in the High Plains—a sub-region of the Great Plains, including eastern parts of Colorado, Wyoming, and New Mexico, and also western parts of the Midwest states of Nebraska, Kansas, and South Dakota—is undergoing a transformation. For almost 90 years since the Dust Bowl struck this region, dryland (rainfed) farmers have largely relied on one type of cropping system to stay viable in a climate characterized by water scarcity and unpredictability. The wheat-fallow system involves growing winter wheat for 10 months, followed by a 14-month period without a crop called summer fallow. Wheat-fallow was universally adopted after the Dust Bowl in part because high-power tractors and improved weed control implements facilitated maintenance of summer fallow periods, and also in response to recommendations from the Soil Conservation Service (now the Natural Resources Conservation Service) to alternate strips of wheat and fallow to impede rampant soil erosion (Greb, 1979; Helms, 1990). Precipitation accumulates in the soil during the year long summer fallow period, which enables farmers to mitigate some of the risk of wheat failure due to drought (Peterson et al., 1996). However, while wheat-fallow provides some degree of yield and income stability, it can reduce the productivity and profitability of dryland farmers because it requires two years to grow a single crop (Kaan, 2002; Peterson and Westfall, 2004). Low levels of crop productivity and diversity combined with over a century of tillage have left soils degraded, further exacerbating the risks of a profession already fraught with uncertainty. But while wheat-fallow still remains a ubiquitous practice in the High Plains, farmers are transitioning toward more economically and environmentally beneficial cropping systems.

Exponential growth in the adoption of no-till over the last two decades has enabled vast numbers of dryland farmers to transition to intensified cropping systems, which are crop rotations that have reduced the frequency of summer fallow, to the point in some cases of eliminating it entirely (Hansen et al., 2012). Transitions to intensified systems were rapid in Canada and the Northern Great Plains following the advent of no-till, but the High Plains have been slower to follow suit (Smith and Young, 2000; Cochran et al., 2006; Hansen et al., 2012). Due to the potential for cropping system intensification to reverse century-old trends in soil degradation and revitalize the profitability of dryland agriculture in this region, there is a need to understand the factors driving or inhibiting transitions to intensified cropping systems in the High Plains.

Scholars are only beginning to examine farm-level social dynamics of the transformation to intensified and diversified cropping systems (Bennett and Cattle, 2013; Carlisle, 2016; Ingram et al., 2016; Basche and Roesch-McNally, 2017) which some have called the "semi-arid cropping revolution" (Smith and Young, 2000). Meanwhile, another body of literature seeks to understand this revolution through an analysis of macro scale forces like policies and markets. Maaz et al. (2017) largely attribute increasing crop diversity in Canada and Australia to policy and market changes and research advancements, although Smith and Young (2000) found that market prices and policy had little relationship to summer fallow frequency in Canada and the Northern Great Plains. There is also a sizable body of scholarship that attributes the growing wave of cropping system intensification to simple economic or agronomic rationales, assuming that most producers transition mainly to increase profitability (Kaan, 2002; Hansen et al., 2012).

While perceived profitability has been found to correlate with adoption of intensified cropping systems (Saltiel et al., 1994), such perceptions are highly dependent on social interactions and information networks (Carlisle, 2016). Additionally, while it is generally accepted that a mid-intensity rotation (e.g., wheat-corn-fallow) increases net economic returns relative to wheat-fallow, the economics of continuous cropping are much less straightforward because of different input requirements and increased risk of crop failure (Dhuyvetter et al., 1996; Kaan, 2002; Aiken et al., 2013). To my knowledge, there has been no attempt to explain the sociological drivers and constraints contributing to cropping system intensification trends in the High Plains.

While the economic risks can increase with continuous cropping, these systems tend to provide the greatest soil health and environmental benefit (Shaver et al., 2003; Sherrod et al., 2003; Acosta-Martinez et al., 2007). Notably, continuous rotations have the potential to offset the carbon dioxide emissions associated with the life cycle of no-till grain production through enhanced soil carbon sequestration, which contributes to simultaneous climate change adaptation and mitigation (Chapter 2). Additionally, as intensification is most often achieved by increasing the diversity of crops in rotation, a wide range of benefits from risk mitigation to better weed control and reduced pesticide inputs typically accompanies greater cropping intensity (Derksen et al., 2002). These complex trade-offs and interactions between risk, profitability, and soil health suggest that the decision about whether and how much to intensify is more complicated than the prevailing narratives around profitability acknowledge. An understanding of the social dynamics influencing the degrees of cropping system intensity on the landscape could help identify strategies to overcome the barriers constraining intensification.

No-till is a prerequisite for successful cropping system intensification (Peterson et al., 1996). While barriers that prevent the adoption of no-till are relevant to the broader agricultural shift occurring on the landscape, these barriers have been extensively studied and reviewed (Nowak, 1983; Lyon et al., 2004; Knowler and Bradshaw, 2007), and thus I focus solely on the decisions related to cropping system intensification once no-till adoption has already occurred. In contrast to the adoption of discrete technologies or practices, transitions from wheat-fallow to intensified cropping systems represent a more complicated shift in farming system, and thus may suffer from slow diffusion as predicted for complex innovations (Pannell, 1999; Padel, 2001; Rogers, 2003; Blesh and Wolf, 2014).

In summary, this paper seeks to generate a greater understanding of the drivers and constraints governing cropping system intensification in the High Plains. The argument is ground in in-depth interviews with dryland, no-till farmers throughout the High Plains representing three levels of cropping system intensity from wheat-fallow to continuous. From these insights I suggest strategies for social change that can facilitate farmer transitions to intensified cropping systems. The argument is theoretically animated by Carolan's (2005) earlier application of Bourdieu, specifically the latter's concept of "field" to better understand tensions created by contested worldviews and practices in agriculture.

4.2 Briefly Setting the Conceptual Stage

In contrast to individualistic and rationalistic approaches to behavior and perception, a focus on Bourdieusian fields de-centers the individual as an analytic unit, looking instead toward the social and embodied nature of action as a source of knowledge, understanding, and affect (Bourdieu, 1977). Central to this literature is the idea that social activity, much of it mundane, not only reproduces worlds but also creates them. This research tradition seeks to apprehend from whence deeply held understandings—or "spirit" (Weber, 1993), "field" (Bourdieu, 1977), “ethic” (Appadurai, 2013), “imaginary” (Anderson, 1983; Taylor, 2004)—arise, noting that while “understanding makes the practice possible, it is also true that the practice largely carries the understanding” (Strauss 2006:330).

Practices are guided by what Bourdieu (1990:66) has called the “practical sense,” which refers to patterns rooted within the respective communities of an individual. Practices should not, however, be reduced to merely learned rules of “appropriate” behavior. Practices can also engender a “feel for the game” (Bourdieu 1990:66) that is notably disruptive to the status quo, leading to imaginaries that have the potential to provoke resistance and change (Carolan 2016). Behavioral change, then, when viewed through this lens, emphasizes *acts* and *social networks of trust and knowledge* for understanding why agents do and think as they do.

One way to think about this approach is through the analysis provided in *Modern Social Imaginaries* (Taylor 2004). Taylor (2004), building on Anderson (1983), explains in this work how social imaginaries are conceptions of the world held by “ordinary people” (Taylor 2004:106) that are carried by, and which in turn legitimate, practices that are forward looking, informing not only how things usually go but also how they ought to go. In this sense, then, fields (or what Taylor refers to as imaginaries) are deeply consequential, shaping actors’ motivations, values, identities, and understandings of what is and what ought to be.

Such approaches have been applied widely in the critical agrifood literature, from the “good farmer” literature (e.g., Burton 2004; Sutherland and Burton 2011) to more specifically Carolan’s (2005) application of the Bourdieusian field concept to understanding contesting fields in agriculture. In that latter’s piece, a heuristic was developed to help think through the social processes at work that cause, for example, some farmers to look at an agro-ecological space and see a poorly managed field, while other farmers can look at the exact same space and see a highly valuable management practice in place. This chapter builds on this tradition and Carolan's piece in particular to understand those competing agro-imaginaries as they play out in the High Plains among wheat farmers.

4.3 Methods

The design of this study and the questionnaire were approved by the Colorado State University Institutional Review Board. Interviews with 23 farmers were conducted in 2015 and 2016 following the precepts of grounded theory (Glaser and Strauss, 1967), with slight alteration. Although this method does not presuppose an understanding of the phenomenon at hand, but rather allows relevant themes to emerge throughout the interview process, the first interviews were inspired by the semi-structured, open-ended interview questionnaire used by Blesh and Wolf (2014; Appendix 1). Relevant themes were coded into applicable categories, which were constantly compared throughout the course of data collection, enabling higher order themes to emerge. Each interview lasted between one and two hours. Sampling was concluded when new themes ceased to emerge.

Each level of cropping system intensity – wheat-fallow, mid-intensity, and continuous – was present in each of three climatic zones (north, middle, and south) located in eastern Colorado and western Nebraska (Table 4.1). The northern zone contained farmers in Nebraska, while the middle and southern zones contained farmers in northeastern and southeastern Colorado, respectively. All primary operators identified as male. Almost every farmer either grew up on a farm that practiced wheat-fallow or practiced it himself before transitioning to the present crop rotation. Interviewees were identified by snowball sampling, as each participant was asked to suggest at least one other farmer who fit the criteria. All interviewed farmers practice dryland agriculture (no irrigation) with <500 mm annual precipitation and use no-till practices. By seeking out farmers across a wide geographical range along the full spectrum of cropping system intensity, I was able to access a variety of social networks, thereby overcoming a common shortfall in snowball sampling (Gilbert, 1995). I captured a wide range of farm sizes (Table 4.1) and ages, although age data was not formally recorded.

4.4 Results

Applications of Bourdieu's theory of practice to agriculture have evolved from a singular field of farming (e.g., Raedeke et al., 2003) to the distinct yet overlapping fields of conventional and sustainable agriculture (Carolan, 2005). Carolan justified the construction of multiple fields by noting that arguments for a singular farming field "conceptually oversimplify the diverse array of attitudinal proclivities, cultural rules and resources, and objective network constraints that exist within the production realm of agriculture" (2005:395). In attempting to understand the factors that influence the occurrence of summer fallow in the High Plains, I build on this analysis and find a complex set of distinctions between social networks, information sources, perceptions, and practices, which necessitates further reconstruction of the social (sub-) fields in agriculture. In doing so, I identify numerous overlapping social fields within the broader fields of conventional and sustainable agriculture (Figure 4.1). Gleaning from the findings of the abovementioned interviews, I "map" the sub fields of *no-till wheat-fallow, no-till mid-intensity*, and *soil health practitioners* against Carolan's (2005) more general fields of conventional and sustainable agriculture. This tentative analytic and conceptual exercise is explained (and justified) further in the following discussion. Three emergent themes from the research reveal the forces that shape and are shaped by the social landscape of dryland agriculture: 1) the emergence of a soil health paradigm, 2) perceptions of risk and profitability, 3) crop insurance policy. Discussion of each theme is followed by strategies for facilitating transitions to intensified cropping systems, based on the insights that emerged from the interviews.

4.4.1 *Theme #1: The Emerging Soil Health Paradigm*

In 2013, the Natural Resources Conservation Service (NRCS) launched the first soil health initiative in the US. The "Unlock the Secrets of the Soil" campaign, as it was called, laid the foundation for the national conversation about soil health. Soil health is a management philosophy centered on four principles: 1) Keep the soil covered as much as possible, 2) Disturb the soil as little as possible, 3) Keep plants growing throughout the year, and 4) Diversify as much as possible using crop rotation and cover crops (NRCS, 2013). While many producers likely followed these principles for decades if not centuries prior to 2013, NRCS codified and formalized these few simple rules, and the movement that has since emerged established an infrastructure and platform for the dissemination of the soil health philosophy. In the years since 2013, farmers and ranchers, food and agribusinesses, environmental organizations, and, to a lesser extent, universities have joined the movement to spread the soil health ideology through farming conferences, articles, videos, films, and personal communication. I found evidence of an impact of this larger movement on the ideologies and practices of dryland farmers in the High Plains. Some farmers strongly identified with the soil health philosophy, and this new way of thinking manifested in transitions to continuous cropping systems and innovations beyond cropping system intensification like the integration of livestock and cover crops.

Several of the soil health tenets are relevant to decisions about summer fallowing. I found adherence to these principles, specifically the tenets related to maximizing diversity and time with a growing plant, to be a strong motivation underlying transitions to continuous crop rotations.

> My goals are just to diversify myself even more… and return more to my land. – M9, continuous
>
> We're going to try to have no fallow periods even between crops. We're trying to go to continuous root all the time. You know, our philosophy is… I want to keep a growing crop year round. – M6, continuous

One of the overarching beliefs held by proponents of the soil health paradigm is that mimicking nature through ecologically-based practices can reduce reliance on chemical herbicides and fertilizers. I found that soil health practitioners translated this belief into practice, resulting in differences in input use between continuous and less intensified farmers. I expected even greater fertilizer use in continuously cropped systems, as intensified producers replace fallow periods that don't require any fertilizer with crops that do. But, I found that annualized fertilizer use between 2010 and 2014 was over 40% lower in continuous rotations relative to mid-intensity, which can be attributed to continuous farmers applying about half the amount of N fertilizer per crop of less intensified farmers (Chapter 3).

Continuous farmers also use about 50% of the glyphosate used by wheat-fallow farmers, and less than 20% and 40% of the amount of 2,4-D and dicamba, respectively, constituting substantial reductions in the three most prolific herbicides used in grain production (Chapter 3). Mid-intensity producers often cited better weed control as a motivation for intensifying, but for reasons other than increasing plant competition. Many mid-intensity producers rotate wheat with Round-up Ready crops like corn, which enables them to control the weeds that are typically a problem in wheat.

> SR: You said that throwing in corn would be the best for that piece of ground. Why is that?
> M3, mid-intensity: Just weed control. You'd be able to throw a different herbicide at it.

Soil health practitioners, on the other hand, were more likely to acknowledge the ecological, rather than the chemical, means by which cropping system intensification helps control weeds. Replacing fallow periods with a crop increases plant competition with weeds, and thus reduces the amount of herbicide required (Derksen et al., 2002).

> Palmer amaranth is starting to show up here, and where it's raising hell is in edible beans. So that was the reason I put peas in the rotation because that will give us another crop in the rotation that will outcompete Palmer. The peas will choke it out before it ever gets going. – N4, continuous

As I will discuss, whether or not producers acknowledge the ability for ecological processes to profitably and effectively replace chemical inputs depends on whose knowledge they trust, and consequently I find that a gradient of input use is aligned along the social organization of dryland agriculture, as illustrated by the continuum at the bottom of Figure 4.1.

A critical source of motivation for soil health practitioners in the High Plains lies in the stories of farmers in wetter climates who have successfully transitioned to highly diverse continuous cropping systems. A number of high profile dryland farmers have become important figures in the soil health movement, and several of them were mentioned by the soil health practitioners in this study as inspiration for transitioning to a continuous crop rotation. These high profile farmers communicate their stories through the infrastructure of the soil health movement: conferences, videos, magazine articles, and farmer-to-farmer communication. The specific outcomes that soil health practitioners are inspired to achieve are often related to reductions in chemical inputs or improvements in soil health.

> I'd love to be the Gabe Brown of Eastern Colorado. I'd love to have his yields and no chemicals and no fertilizers. That would be perfect. – M6, continuous

Several knowledge claims circulate within soil health networks that underpin the belief that cropping system intensification and diversification can reduce the need for chemical fertilizer. An example is the claim that growing a greater diversity of plants will foster more microbial diversity in the soil, and that microbial diversity will enhance nutrient cycling and the amount of soil nutrients available for plants. Several farmers referred to this claim as the "diversity aboveground equals diversity belowground" rationale, and cited it as a reason why they strive to grow as great a diversity of crops as possible.

> If you want to build healthy soil and build a functioning soil I think you have to have a diversity of root exudates and a diversity of food aboveground, which will equate to diversity belowground, which will ultimately improve the function of the soil. – M9, continuous

"Mainstream" (i.e. university and government) agronomists often communicate the benefits of cropping system intensification in terms of better weed control or greater overall crop production. These benefits are easy to communicate because they are easy to "see" (e.g., weed-free rows), and have long been associated with desirable outcomes in production agriculture (Burton, 2004). In contrast, soil health benefits like greater microbial activity are more epistemologically distant, or harder to "see" (Carolan, 2006a), and thus they often make less compelling cases to intensify. However, the awareness about complex soil processes illustrated by the farmer in the quote above represents a deeper knowledge about soil health that motivated his transition to a diverse and continuous crop rotation. Unlike those embedded in (sub-) fields located on the left side of Figure 4.1, this knowledge enabled him to "see" the soil health benefits of continuous cropping. This "vision" helped override the privileging of logics that celebrate the epistemologically "near" benefits of, say, short-term profit maximization but that otherwise result in long-term detrimental effects, such as upon unseen microbial communities.

Soil health claims like "diversity aboveground equals diversity belowground" were mentioned almost exclusively by soil health practitioners. This is not to say that farmers outside of the soil health field have never been exposed to such knowledge claims. For example, farmers across the spectrum of conventional and sustainable agriculture attend the regional High Plains No-till Conference, at which soil health practitioners or proponents often give presentations and explain how they believe soil health management practices affect the intricate workings of soil function. But the science of soil health, like the science of human health, is complex and often unresolved, and obtaining knowledge about the soil system requires a great deal of trust in "experts" that make such knowledge claims. Trust serves to reduce the complexity and uncertainty present in the science of soil health, and enables farmers to "know" how the soil functions (Luhmann, 1979). On the other hand, a lack of trust in those who claim "diversity aboveground equals diversity belowground" may inhibit the motivations of other farmers to diversify in the hopes of reducing fertilizer use. As this and several other claims about soil health are still a matter of debate in mainstream scientific research, it appears that the emergence of soil health social networks has created a distinct set of knowledge in this social field.

The farmers who cited soil health rationales as motivation for eliminating summer fallow place their trust in different sources of information than the inhabitants of other social fields, which results in the emergence of contested truths. Most importantly, conventional farmers and even other farmers in the field of sustainable agriculture believe that summer fallow is a necessary component of a profitable dryland farming system in the High Plains, while soil health practitioners largely believe that fallow is a detriment to profitability. One mid-intensity farmer referred to the producers who eliminated fallow as "a group of cover crop-livestock-holistic-soil health-it doesn't matter we're going to prove this works no matter if it's economical or not," illustrating the common belief among dryland farmers that summer fallow is an economic necessity, and soil health practices are economically detrimental.

The soil health practitioners identified nationally recognized leaders in the soil health movement as sources of information, including an NRCS agent, a high-profile farmer and rancher from North Dakota, and several private sector soil scientists. This differs from the information sources of other inhabitants of the field of sustainable agriculture, and also those practicing conventional agriculture, who tended to receive their information from local land grant universities, neighboring farmers, and product dealers. Soil health practitioners were more likely to distrust mainstream research, while less intensified producers use it to justify their use of summer fallow.

> For years, I guess all the old-timers, my dad and all the older [farmers], they said that wheat-fallow is the most profitable system. And my dad read a study that I think K-State did that said they did the feasibility on different cropping systems and they came out with the same conclusion. – S4, wheat-fallow

> [A researcher] did 5 years of research with wheat behind peas and wheat summer fallow and I saw his research because one day he gave me hell about it when I was giving a talk and said that I shouldn't be telling guys that yields would be

> pretty good. And I said, ‘Well… that's been our experience and that's been the experience of guys that I've talked to that have raised peas is that their wheat yields are staying pretty consistent.’ ‘Well I've got research that shows it don't’ and he's like, ‘my research showed that it costs you about 20 bushel wheat yield per acre.’ Well OK [researcher], you're a shitty farmer. I don’t know, I've heard a lot of guys say a lot of shit that comes out of [that research station] they just throw it out the window because their research and other research don't jive… And he's like, ‘this is God's law: you're going to lose a lot of yield if you grow peas.’ There's another 50 farmers around the room that will disagree with you but… so I don't even pay any attention to researchers anymore really. – N4, continuous

This is not to say that the mainstream scientific consensus stands in opposition to the merits of soil health practices, although mainstream researchers are less likely to make the broad claims of the potential impacts of these practices compared to the proponents of soil health. As such, soil health practitioners tend to trust each other’s knowledge and that of private scientists who are unified in their support of the soil health principles.

Whether or not knowledge claims are accepted as truth ultimately depends on the degree of trust placed in the source of the information, and the decision about who to trust is inextricably linked to identity (Carolan and Bell, 2003). Many farmers in this study who practice continuous cropping in part construct their sense of self around their ties to the soil health philosophy. The most enduring behavior change is realized when a person’s identity becomes intertwined with the behavior itself (e.g., “I am the type of farmer who practices continuous cropping”; e.g., Burton 2004). In reforming their sense of self in conjunction with transitions to continuous cropping, soil health practitioners also reform their networks of trust, what they know, and ultimately reconstitute the dominant habitus. One farmer said he didn’t trust the research from a local experiment station because “they're not friends of soil health” (M6, continuous farmer). The farmer’s mistrust of the experiment station’s research was driven by the perception that the researchers are biased against the soil health philosophy, and not, say, the scientific rigor with which the research was conducted. This judgment of the researchers as opposed to the research itself illustrates the point of Carolan and Bell (2003) that "whose science you trust will… ultimately determine which one represents the truth." Carolan (2006b) noted that the rise of sustainable agriculture as a legitimate farm management system could be attributed to the fact that "more people now trust sustainable agriculture and the proponents of sustainable agriculture to be speaking the truth." The same can be said for the emergence of the soil health paradigm. In other words, by trusting soil health proponents, soil health practitioners have a lower threshold for what they deem to be "true enough" about the benefits of soil health practices. In this case, soil health practitioners reject the long-standing "truth" that summer fallow is a necessity in a semi-arid climate.

4.4.2 *Theme #2: Perceptions of Risk and Profitability*

Challenges in adopting ecologically based systems are often framed as difficulties in overcoming the chasm between choosing to promote human and ecosystem wellbeing vs. choosing to maximize profit (Morel and Léger, 2016). However, cropping system intensification can simultaneously improve soil health (Sherrod et al., 2005) and increase net profitability by 25-40% compared to WF (Kaan, 2002), and thus bridge the chasm of profit/environment. But I found that farmers inhabiting different social fields have different definitions and perceptions of "profitability", and thus the potential benefits of cropping system intensification can be interpreted in contrasting ways. This is less to suggest that farmers, or at least some farmers, are not interested in having profitable operations. The point, rather, is that profitability can mean wildly different things depending on the (sub-) field one is embedded in—e.g., short-term economic profit maximization vs. long term social, ecological, and economic profit optimization (Flora and Curtiss, 2014).

These perceptions ultimately influence the chosen degree of cropping system intensification. Additionally, farmers do not make decisions concerning adoption of risky agricultural practices based on the intent of profit maximization alone (Shapiro et al., 1992), but they instead integrate their perceptions of risk and profitability (Anderson and Dillon, 1992). I identified three distinct strategies for navigating the interactions between profitability and risk: risk avoidance, yield maximization, and input reduction (Figure 4.2).

Cropping system intensification in the semi-arid High Plains involves considerable risk and uncertainty. It is a region with greater day-to-day and year-to-year weather variability than almost any other region in the country (Nolan Doesken, personal communication). For dryland farmers, there is considerable risk that each investment in the seeds, chemicals, and labor required to plant a crop will be lost to drought, hail, or fire. WF farmers avoid risk through infrequent crop plantings, and reduce risk of losses to drought by summer fallowing to increase the chances of a successful wheat crop.

> You could almost argue that wheat-fallow in some circumstances is more profitable than a more intensive cropping rotation. In this part of the state, where it's more arid and I have less moisture, the reality is that if you grow 2 crops out of 3 years, that's that much more moisture you're pulling out. So as long as the weather's providing enough moisture, it all works great. But if your shortfall is on moisture, just going out there and doing one wheat crop, that has really good genetics that can grow really good yield every other year, there is an argument to kinda stay towards that. My experience is summer crops tend to be more erratic on their success, because rainfall in the summer months can be erratic... And so statistically, guys bet on their wheat crop year in and year out because it's kind of a known quantity and they know what to expect. – S5, WF

Many WF farmers' perceptions of profitability are inseparable from the perceived risk of drought. But while WF minimizes yield risk, farmers that only rely on one type of crop are highly vulnerable to market risk. Low wheat prices leave WF farmers with no other profitable options, and while they may have consistent wheat yields, their profitability is highly influenced by volatile markets. WF often results in the smallest amount of total crop production compared to the other cropping systems, and provides the least benefit to soil health, which may in fact increase susceptibility to yield risk in the long-term as soil health declines. But as Bowman and Zilberman (2013:44) note, "A risk-averse farmer… may be less likely to adopt new technologies, even if they are likely to reduce his susceptibility to risk or increase productivity or income over the long run." Profitability simply represents the difference between monetary costs and returns, but it is clear that the time-scale on which farmers perceive profitability determines the strategies they will employ. Heavily focused on short-term risks, WF farmers choose yield consistency and risk avoidance over profitability or long-term risk mitigation through improvements in soil health.

Additionally, thinking about profitability beyond the myopic (neoliberal) understanding that reduces "it" to economic capital, toward a stance recognizing all capitals—in no particular order: social, cultural, natural, built, financial, human, and political—illuminates the relevance of social networks and communities of practice upon decision-making, as various types of capital have different exchange rates depending on the field that one inhabits. WF farmers are willing to trade in natural capital in the form of soil health in order to achieve consistent (at least in the short term) returns of economic and other forms of capital derived from a high-yielding crop. The symbolic and cultural capital embodied by a large wheat crop has high value in the social fields of conventional agriculture, as many farmers and bankers confer the qualities of a "good farmer" on those with impressive yields (Burton, 2004). Thus, yield risk embodies other risks that extend beyond the threat of economic losses to those influencing social and cultural status.

Mid-intensity producers capitalize on yield gains associated with cropping system intensity without increasing risk to wheat yields. I call this the yield maximization strategy, whereby farmers rotate different crops to reduce disease and weed pressure and reduce market/price risks while maintaining summer fallow periods, which combine to support higher wheat yields. They also add summer crops that have the potential to increase total grain yields by 75% relative to WF (Peterson and Westfall, 2004). A typical response from farmers when asked about their motivation for transitioning from wheat-fallow to a mid-intensity rotation was, "Economics. Trying to get more crops - 2 crops out of the 3 years instead of 1 crop every 2 years" (M8, mid-intensity). Wheat yield drag, the penalty to wheat yield associated with moisture reductions from eliminating summer fallow, is the primary barrier impeding continuous cropping among yield maximizers. There is an obvious economic cost associated with lower wheat yields.

> Well my last experience with [continuous cropping] is that it cost me about $200,000. Just because the peas weren't yielding that great and completely destroyed my wheat yields and it even lessened my corn yields 2 years later. – M3, mid-intensity

However, even among mid-intensity producers there was a diversity of reasons for not tolerating wheat yield drag. While some farmers' concerns over low wheat yields are mainly economic, others rationalize the use of summer fallow from the standpoint of protecting natural capital, which indicates a division of mid-intensity farmers in both the conventional and sustainable agricultural fields (Figure 4.1).

> M1, mid-intensity: I planted cover crops strips in a fallow field a couple of different years… and it was 18 bushels less [wheat] production where I had the cover crop vs. where I just no-tilled, summer fallowed it. And then the other issue that crept up in that situation was then the residue from that wheat crop, it was a lot poorer wheat crop, that residue – the hard wind broke that off. I actually ended up having to go apply manure to the cover crop strips to keep the topsoil from blowing away.
> SR: It's the opposite effect of what you wanted.
> M1: That scared us pretty bad. Our whole entire rotation depends on that dense, heavy wheat residue... And the best way we've found to produce that is with summer fallow wheat.

While yield maximizers may strive for the simultaneous accrual of natural and economic capital, adherence to this productivist model is characterized by a high input, high output strategy (Wilson, 2001; Burton, 2004). The viability of high-input business models as a means of maximizing profitability is waning, as the cost of inputs like fertilizer, seeds, and herbicide has risen steadily since the 1960s while commodity prices have stagnated or decreased (Carolan, 2017). These looming trends have placed a “double squeeze” on farmer profits (Van der Ploeg, 2006).

I found that farmers practicing continuous cropping, and especially soil health practitioners, have found a potential strategy to overcome the double squeeze. They utilize a low-input model to increase profitability, demonstrated by much lower herbicide use compared to the other cropping systems (Chapter 3). I previously discussed how the practice of continuous cropping helps achieve herbicide reductions, and also noted that soil health practitioners were inspired by others who have managed to drastically reduce input use. But it’s largely the way continuous farmers perceive the costs and risks of summer fallowing that motivate transitions to continuous cropping systems.

> You start penciling out this wheat-summer fallow system. You got all this money tied up in fertilizer and herbicides and all that, and then you spend the whole year killing things [weeds] so you have $50/acre tied up in your wheat crop before you even put it in the ground. So I was trying to eliminate that. And that's how I kinda found these cover crops and stuff because I was trying to eliminate that fallow period after millet or any of those periods where I had to spray or do something. – M9, continuous

In stark contrast to WF and mid-intensity producers, who utilize summer fallow to mitigate yield risk, continuous farmers view summer fallow as an even riskier strategy. Summer fallowing can be an expensive practice, especially for no-till farmers. It takes multiple herbicide spray operations with costly chemicals to prevent weeds from growing during fallow periods. These spray operations are becoming even more expensive due to herbicide resistant weeds, which force farmers to use more expensive or greater quantities of herbicides. Therefore, summer fallow is an economic investment that improves the likelihood, but does not guarantee the success of the following wheat crop, as any number of natural or economic disasters can make a wheat crop unprofitable. One farmer said, “I can't have a fallow period, I'm just throwing money into a black hole with no guarantees” (M6, continuous). Additionally, summer fallow only stores about 25-30% of the precipitation that falls (Peterson et al., 1996), and thus continuous farmers perceive this to be a waste of the region’s most precious resource.

> There’s this concept that, well because we're dry in the arid west that [summer fallow] is the only way I can build soil moisture – that I can't grow a crop every year. But it really is a fallacy… I mean even the ARS studies they're showing a 30% water storage efficiency. So it's like OK so like you get 10 inches of rain this year and you're just going to give 7 inches of it up just for the hell of it and you're going to try to get 3 inches in the ground. I mean there’s no way I know, that 30% efficiency can sound like a good idea. But in the case of wheat-fallow that's what they keep advocating. – S8, continuous.

Still, farmers practicing continuous cropping assume enormous risks to wheat yield in the short-term, potentially sacrificing economic capital for natural capital in the form of soil health. Continuous cropping simultaneously exacerbates and mitigates yield risk relative to the WF strategy (Dhuyvetter et al., 1996). The risk of a failed wheat crop due to drought is high because reserves of soil moisture are consistently low (Aiken et al., 2013). However, by leveraging crop diversity, continuous farmers may be less susceptible overall to sweeping losses to drought, hail, or fire because different crops grow at different times of the year (Di Falco and Perrings, 2005; Bowman and Zilberman, 2013). Additionally, price/market risk is mitigated through diversification. As explained by one farmer: "I don't have all my eggs in one basket. If the price of wheat is terrible at least you have some of the other crops to fall back on" (N2, continuous).

Continuous producers, and those practicing sustainable agriculture in general, have a long-term view of profitability as they consider the potential for enhanced resilience conferred by soil health benefits. Continuous rotations improve soil structure relative to less intensified rotations (Shaver et al., 2003), which can help the soil collect and store more water from precipitation. Practitioners of sustainable agriculture see losses of natural capital like degraded soil structure directly translating to economic losses.

> In that instance when I seen that lake form in that [neighbor's] field, that showed a water infiltration problem. It didn't rain an awful lot in that particular field, it just rained enough that that soil was so poor that it could not hold the rain that came… And I think that's how important moisture is – you have to utilize it. It doesn't matter what falls from the sky, it matters what ultimately infiltrates your field and isn't stuck in a lagoon or ponded up in the middle of your field. I've been trying to build soil health… to try to accept rain events that come very rapidly. It's hard to tell what the future will bring but I think if you can prepare yourself now for events down the road – so the moisture I am getting now I'm building – growing as many plants and as diverse plants as I can to build that soil so that if I do come to a very dry time hopefully I've built my soils enough and raised my

> organic matter and still have my soils covered that I can better make it through that dry period because my soils will be more resilient. – M9, continuous.

This quote demonstrates long-term vision and planning. The farmer assumes greater risk in the short term by continuous cropping and growing cover crops in the hopes of acquiring natural capital that can protect future profits.

In addition to risk management concerns, producers identified a lack of profitable crops other than wheat as a barrier impeding their motivation to intensify.

> The big thing to me is if I could find a crop that was suited to our area to put in that summer fallow period that would be economical. If you could make money on that crop, I'd be willing to give up wheat yield in order to eliminate that fallow period. – M1, mid-intensity
>
> If local price of wheat is $4, that's a money loser. There's no way you can raise it for $4. But again, that's one of the things that is prompting us talking about doing more milo and less fallow. But, I don't know, the price of milo's terrible too. I mean it's down, it's almost $3 probably. So neither one of the prices are good. And either one of them is probably going to be a money loser – S4, wheat-fallow

Mounting economic, environmental, and market challenges will further threaten the viability of wheat monocultures, and the mitigation of risks afforded by crop diversification will likely be an important adaptation strategy (Lin, 2011; Maaz et al., 2017). Additionally, with herbicide resistant weeds and macroeconomic forces driving up the cost of inputs, there is now a stronger economic case than ever for an input reduction strategy, and I may see greater abandonment of the traditional productivist model in the pursuit of profitability. One farmer observed the consequences for those who do not adapt to changing environmental or economic conditions:

> When it got so dry in the 2000's, everybody thought they should keep summer fallowing… You know it's funny. All the guys my age that used to farm… they're all gone. They're all working for the railroad or they're crop advisers at Simplot. – N4, continuous

4.4.3 *Theme #3: Crop Insurance*

The high risks inherent in dryland agriculture make federal crop insurance an influential factor in agricultural decision-making in the High Plains. In a shift in focus from income support to risk management, the 2014 Farm Bill ended direct payments and increased subsidies for crop insurance programs. Studies based on older farm bills suggest that crop insurance payments may have an effect on the profitability rankings of different dryland cropping systems (Nail et al., 2007). While the impact of more recent policy changes has yet to be quantified, farmers interviewed in this study generally viewed crop insurance policies as a barrier to cropping system intensification.

The clearest policy impediment to continuous cropping is that it is not an insurable practice in every county in the High Plains (Figure 3.4). In other words, farmers in counties not designated for continuous cropping by the Risk Management Agency (RMA) are unable to insure wheat, usually their most profitable crop, unless there is a year-long summer fallow period preceding wheat planting. The only exception is that a farmer can apply for an individualized crop insurance plan if they can produce at least a 5-year yield history under continuous cropping, which requires producing crops for at least 5 years with no insurance. Many mid-intensity producers cited this prohibitive policy as a reason for not even considering continuous cropping, as they were unwilling to take risks with uninsurable wheat. One farmer said that if continuous cropping was an insurable practice in his county, "we would look into it… but that's the thing, there's no way you can really do it if it's not insurable because it's a really huge gamble" (M7, mid-intensity). Additionally, even in counties that are insurable for continuous cropping, farmers often have to pay significantly higher premiums for continuously cropped vs. summer fallowed wheat. Higher premiums for lower guarantees are presumably justified because of the greater assumed wheat yield risk associated with continuous cropping. However, not all farmers practicing continuous cropping agree that it is riskier than summer fallowing.

> We still to this day are being penalized in paying double premium because I don't fallow before our wheat. Even though we've had 20 years of success, they won't change the policy. I couldn't insure it for quite a while and they eventually allowed us to insure it but it was, they would cut your APH in half, your guarantee in half and double your premium when I first did that. So I just dropped my insurance for 5 years. I came back with actual production history and proved my yields so they couldn't cut it in half. And then I took insurance again, but I'm still paying a higher premium. – N2, continuous

Despite these impediments, some farmers still practice continuous cropping throughout the full extent of the High Plains, either by paying higher premiums or by foregoing crop insurance completely, and relying on diversification to mitigate risk. Successful cropping system intensification requires the diversification of crop rotations, where wheat is rotated with one or more different kinds of crops. But many dryland farmers have never grown crops other than wheat, and thus they haven't established proven yield histories for those crops. If a farmer hasn't established a proven yield history on a given crop, they are insured at a transitional yield, or T-yield, based on the county average yield for that crop. Many farmers said that their county's T-yields are so low that the economic risk of growing a new crop is too great, even if they were to collect crop insurance.

> The other barrier to switching to milo in a rotation is most farmers that are doing wheat-fallow, if they want to grow milo, they have to take the T-yield, or whatever the county average is. And most of those T-yields are set at below economic values and so it's a really big risk. I saw this in my dad's operation, is he could go out there and plant a thousand acres of milo but because he had very little history, he had no crop insurance to back it up. So if he's going to his banker and saying, 'I'm going to go plant a thousand acres of milo because CSU extension says I'm going to make more money,' the banker says, 'well what if it fails?' 'Well I guess I'll lose this many thousands of dollars' and so the banker's like, 'well why would you do that? Stick with wheat, you've got the insurance to back it up.' And so crop insurance, I saw that at my dad's, we never were really able to make that transition to a more intensive milo rotation because crop insurance was probably the final straw that just could not, the risk was just too big. – S5, WF

It can take several years to establish a proven yield history that is higher than the county T-yield. The risks incurred in the transition years can be difficult to justify to a banker or a landlord, especially for young farmers or those with little fluid capital. Low T-yields make it difficult to try new crops, which is an impediment to cropping system intensification.

4.5 Suggested Strategies for Change

Based on insights gleaned from the above three emergent themes, how might we leverage this new understanding to motivate and enable farmers to innovate? I offer the following strategies specific to the case of facilitating cropping system intensification, but also with potential implications for sustainable agriculture more broadly.

4.5.1 *Strategy #1: Lessons from the Soil Health Movement*

A lack of successful examples of continuous cropping in the High Plains constrains motivations to intensify. In effectively communicating stories of successful intensification and diversification, and establishing networks of trust with scientists that are open to these practices, the soil health movement has catalyzed a transformative departure from the traditional practice of dryland farming. The findings support others who have shown that farmers are highly influenced by their peers who have used an agricultural practice with success (Warner, 2008). Demonstrating successful continuous cropping in the High Plains may contribute to greater use of this system throughout the region. Targeted outreach, specifically by "recruiting" highly networked non-intensified farmers or product dealers to adopt the soil health philosophy, may also be an effective means of normalizing soil health practices. Additionally, as mainstream researchers tend to have more cultural capital with farmers outside of the soil health field, it will be important to keep an open mind about what constitutes a viable practice by considering the successes that some farmers have achieved with a soil health approach. Likewise, there is a need for more mainstream research to test the claims made by soil health proponents. Establishing mainstream scientific support for the soil health practices and principles that hold up under empirical scientific inquiry could serve to reshape the dominant social field by associating conventional networks of trust with knowledge claims about soil health.

4.5.2 *Strategy #2: Profitability*

The soil health movement catalyzed transitions to continuous cropping within a group of producers that take a long-term view of profitability and place a high value on natural capital. But soil health practices like continuous cropping may not be appropriate for all dryland farmers unless new innovations can reduce the risk and enhance the short-term profitability of these emerging systems. Breeding efforts and market development for important regional rotational crops like millet, grain sorghum (called milo), or field peas could increase the short-term profitability of intensified cropping systems, and consequently catalyze more transitions to intensified and diversified systems, as has been observed in Canada, Australia, and most recently in the inland Pacific Northwest (Maaz et al., 2017). In lieu of strong markets for traditional commodities, several soil health practitioners have begun integrating livestock and grazed cover crops (or forages) into their crop rotations to generate another income stream. They also grow non-traditional crops for this region, which allows them to enter alternative markets. Studying and communicating the potential for livestock integration and alternative crops to enhance profitability may be another way to facilitate intensification, as some farmers said they would be willing to assume added risks if they could make more money.

Additionally, there is a need to better understand and communicate the potential for intensified cropping systems to enhance profitability by enabling input reductions. Linking abstract concepts like soil health to highly visible outcomes like herbicide payments could serve to make the effects of soil health practices more tangible and help to overcome epistemic barriers to sustainable agriculture (Carolan, 2006a). This would require a shift in the way mainstream research is conducted and interpreted, from defining the success of an agronomic practice from a short-term productivist view to a longer-term view of profitability and multi-capital optimization. As input reduction is likely to become more advantageous over time relative to yield maximization, the benefits of this profitability strategy are increasingly persuasive.

4.5.3 *Strategy #3: Crop Insurance*

The barriers imposed by crop insurance are linked to the policies that focus on specific commodities or practices. In 2015, RMA began offering whole-farm revenue protection in every county in every state in the country, signaling a shift toward diversity-friendly insurance policies that enable more freedom in crop management. However, I observed that whole-farm insurance itself might be impeded by barriers to adoption. Only one farmer interviewed in this study utilized this new policy (S8, continuous), whereas everyone else had crop-specific revenue protection policies. While potential barriers to whole-farm insurance adoption did not emerge as a topic of inquiry in the bulk of the interviews, one farmer cited over-complexity and privacy issues as a hindrance to adoption (N2, continuous).

Changes in crop insurance policy alone may not be enough to foster agricultural innovation. Crop insurance provides an important safety net, but it is inherently risk averse and unwelcoming to new practices due to requirements for established yield and income histories. Establishing and supporting other financial incentives, such as conservation payments and similar programs, may be a more desirable route for encouraging innovation.

4.6 Conclusion

Moving beyond simple economic rationales to explain a growing wave of cropping system intensification, I provide evidence for a social dynamic shaping the degree to which farmers are willing to intensify. For almost a century, the dominant social body of dryland agriculture ascribed to an imaginary that embedded the use of summer fallow in dryland agricultural practice. The soil health movement served to socially construct a new imaginary that rejects this traditional notion, and gave rise to a new social field of soil health practitioners with different values, perceptions, and knowledges than other dryland farmers. In contrast to the motivations of soil health practitioners to reduce inputs and build natural capital, many other farmers have transitioned to mid-intensity cropping systems for productivist motivations, citing evidence from mainstream research that replacing summer fallow with crops increases total food production. Still, the perceived risks associated with cropping system intensification, reinforced by crop insurance policies squared in the traditional imaginary, prevent many dryland farmers from moving beyond a wheat-fallow rotation.

To facilitate innovation in sustainable agriculture generally, and cropping system intensification specifically, I suggest strategies like "recruiting" highly networked individuals to the soil health philosophy, or changing the mindset of mainstream agronomic researchers to be more inclusive of long-term viewpoints and profitability strategies beyond yield maximization, which would help to reshape the conventional social fields through existing networks of trust. Additionally, I suggest strategies such as maximizing short-term profitability of intensified cropping systems through market development, which would leverage our understanding of farmer perceptions in their existing social positions to facilitate intensification. A combination of these two types of strategies may help fit sustainable agricultural practices into existing imaginaries, and form new imaginaries that drive sustainable innovation.

Table 4.1. Characteristics of farmers interviewed.
N: Northern zone; M: Middle zone; S: Southern zone; WF: Wheat-fallow

Farmer ID	Cropping System Intensity	Acres Farmed
N1	Continuous	4600
N2	Continuous	16000
N3	WF, Mid-intensity	9000
N4	Continuous	3500
N5	WF	2000
N6	WF	2000
N7	WF, Mid-intensity	3400
M1	Mid-intensity	12000
M2	Mid-intensity	10000
M3	Mid-intensity	20000
M4	WF	9000
M5	Continuous	4000
M6	Continuous	5500
M7	WF, Mid-intensity	7000
M8	Mid-intensity	3600
M9	Continuous	1600
M10	Continuous	1980
S1	Mid-intensity	9900
S2	Mid-intensity	21000
S3	Continuous	4000
S4	WF	47500
S5	WF	10000

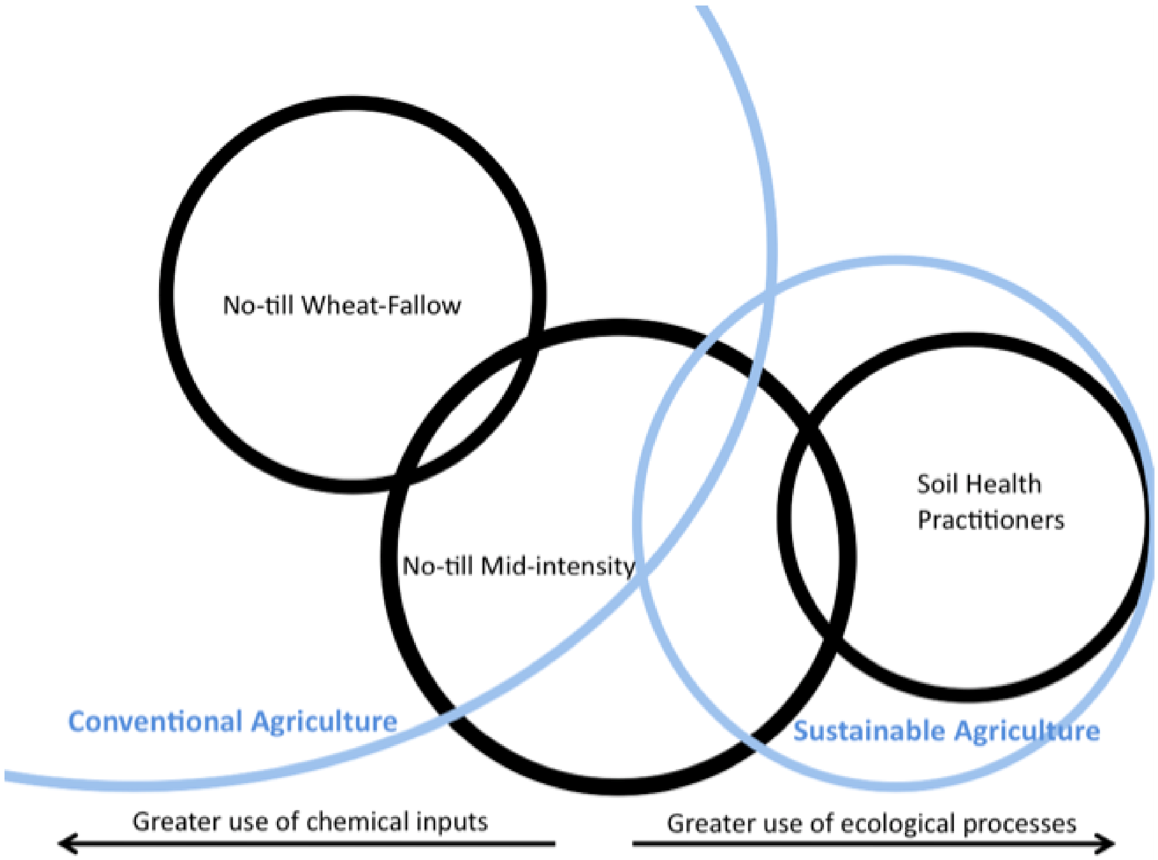

Figure 4.1. Heuristic model overlaying Carolan's (2005) fields with distinct sub fields of practice among dryland farmers in the High Plains.

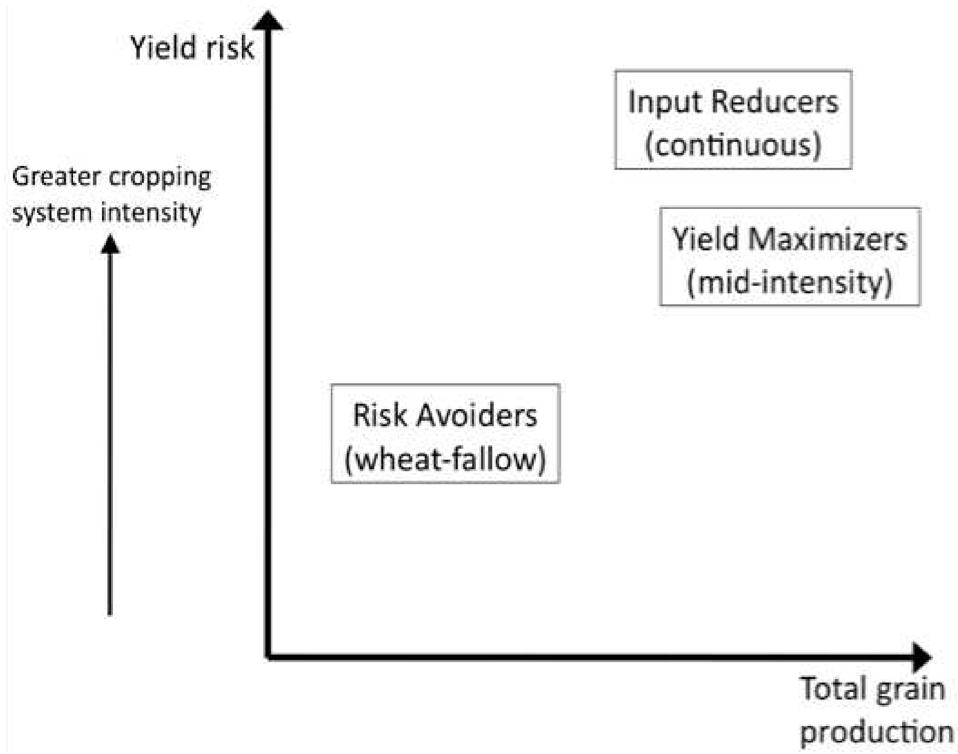

Figure 4.2. Three different profitability strategies and their comparative relationships to grain production and yield risk.

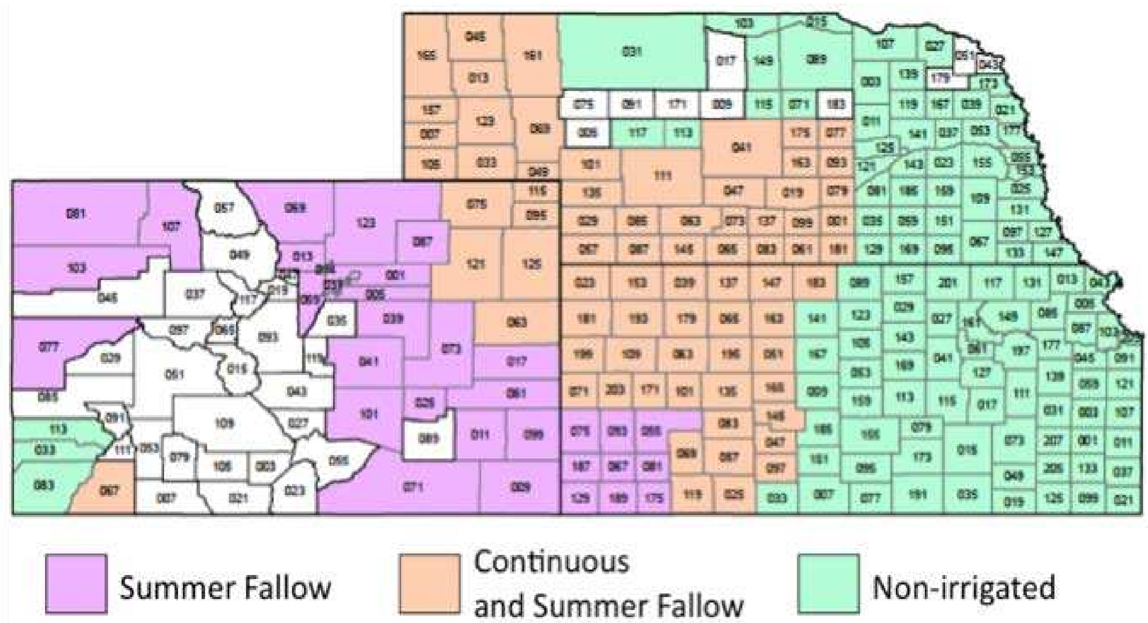

Figure 4.3. Allowable crop insurance practices by county as designated by the Risk Management Agency.

REFERENCES

Acosta-Martinez, V., Mikha, M.M., Vigil, M.F., 2007. Microbial communities and enzyme activities in soils under alternative crop rotations compared to wheat–fallow for the Central Great Plains. Applied Soil Ecology 37, 41-52.

Acosta-Martinez, V., Zobeck, T., Gill, T., Kennedy, A., 2003. Enzyme activities and microbial community structure in semiarid agricultural soils. Biology and Fertility of Soils 38, 216-227.

Aguilar, J., Gramig, G.G., Hendrickson, J.R., Archer, D.W., Forcella, F., Liebig, M.A., 2015. Crop species diversity changes in the United States: 1978–2012. PloS one 10, e0136580.

Aiken, R., O'Brien, D., Olson, B., Murray, L., 2013. Replacing fallow with continuous cropping reduces crop water productivity of semiarid wheat. Agronomy Journal 105, 199-207.

Anderson, B., 1983. Imagined communities: Reflections on the origin and spread of nationalism. Verso Books.

Anderson, J.R., Dillon, J.L., 1992. Risk analysis in dryland farming systems. Food & Agriculture Org.

Anderson, R., 2003. An ecological approach to strengthen weed management in the semiarid Great Plains. Advances in Agronomy 80, 33-62.

Appadurai, A., 2013. The future as cultural fact: Essays on the global condition. London and New York: Verso.

Arnold, S., Dileo, J., Takushi, T., 2014. Colorado Greenhouse Gas Inventory – 2014 Update Including projections to 2020 and 2030. Colorado Department of Public Health and Environment.

Bardgett, R.D., McAlister, E., 1999. The measurement of soil fungal: bacterial biomass ratios as an indicator of ecosystem self-regulation in temperate meadow grasslands. Biology and Fertility of Soils 29, 282-290.

Basche, A.D., Roesch-McNally, G.E., 2017. Research topics to scale up cover crop use: Reflections from innovative Iowa farmers. Journal of Soil and Water Conservation 72, 59A-63A.

Bates, D., Maechler, M., Bolker, B., Walker, S., 2014. lme4: Linear mixed-effects models using Eigen and S4. R package version 1, 1-23.

Bennett, J.M., Cattle, S., 2013. Adoption of soil health improvement strategies by Australian farmers: I. Attitudes, management and extension implications. The Journal of Agricultural Education and Extension 19, 407-426.

Blesh, J., Wolf, S.A., 2014. Transitions to agroecological farming systems in the Mississippi River Basin: toward an integrated socioecological analysis. Agriculture and Human Values 31, 621-635.

Bot, A., Nachtergaele, F., Young, A., 2000. Land resource potential and constraints at regional and country levels. . Food & Agriculture Org.

Bourdieu, P., 1977. Outline of a Theory of Practice. Cambridge university press.

Bowman, M., Zilberman, D., 2013. Economic factors affecting diversified farming systems. Ecology and Society 18.

Burton, R.J., 2004. Seeing through the ‘good farmer's’ eyes: towards developing an understanding of the social symbolic value of ‘productivist’behaviour. Sociologia Ruralis 44, 195-215.

Carlisle, L., 2016. Factors influencing farmer adoption of soil health practices in the United States: a narrative review. Agroecology and Sustainable Food Systems 40, 583-613.

Carolan, M., 2016. The Sociology of Food and Agriculture. Taylor & Francis.

Carolan, M., 2017. More-than-Active Food Citizens: A Longitudinal and Comparative Study of Alternative and Conventional Eaters. Rural Sociology 82, 197-225.

Carolan, M.S., 2005. Barriers to the adoption of sustainable agriculture on rented land: An examination of contesting social fields. Rural Sociology 70, 387-413.

Carolan, M.S., 2006a. Do you see what I see? Examining the epistemic barriers to sustainable agriculture. Rural Sociology 71, 232-260.

Carolan, M.S., 2006b. Social change and the adoption and adaptation of knowledge claims: Whose truth do you trust in regard to sustainable agriculture? Agriculture and human values 23, 325-339.

Carolan, M.S., Bell, M.M., 2003. In truth we trust: discourse, phenomenology, and the social relations of knowledge in an environmental dispute. Environmental Values, 225-245.

CDWR, 2017. Geographic Information System Data. Colorado's Decision Support Systems.

Cochran, V., Danielson, J., Kolberg, R., Miller, P., 2006. Dryland cropping in the Canadian prairies and the US northern Great Plains. AGRONOMY 23, 293.

Conant, R.T., Ogle, S.M., Paul, E.A., Paustian, K., 2011. Measuring and monitoring soil organic carbon stocks in agricultural lands for climate mitigation. Frontiers in Ecology and the Environment 9, 169-173.

Cotrufo, M.F., Wallenstein, M.D., Boot, C.M., Denef, K., Paul, E., 2013. The Microbial Efficiency Matrix Stabilization (MEMS) framework integrates plant litter decomposition with soil organic matter stabilization: do labile plant inputs form stable soil organic matter? Global Change Biology 19, 988-995.

De Vries, F.T., Liiri, M.E., Bjørnlund, L., Bowker, M.A., Christensen, S., Setälä, H., Bardgett, R., 2012. Land use alters the resistance and resilience of soil food webs to drought. Nature Climate Change 2, 276-280.

Dennehy, K., Litke, D., McMahon, P., 2002. The High Plains Aquifer, USA: groundwater development and sustainability. Geological Society, London, Special Publications 193, 99-119.

Derksen, D.A., Anderson, R.L., Blackshaw, R.E., Maxwell, B., 2002. Weed dynamics and management strategies for cropping systems in the northern Great Plains. Agronomy Journal 94, 174-185.

Dhuyvetter, K., Thompson, C., Norwood, C., Halvorson, A., 1996. Economics of dryland cropping systems in the Great Plains: A review. Journal of Production Agriculture 9, 216-222.

Di Falco, S., Perrings, C., 2005. Crop biodiversity, risk management and the implications of agricultural assistance. Ecological economics 55, 459-466.

Diaz, R.J., Rosenberg, R., 2008. Spreading dead zones and consequences for marine ecosystems. science 321, 926-929.

Drenovsky, R., Vo, D., Graham, K., Scow, K., 2004. Soil water content and organic carbon availability are major determinants of soil microbial community composition. Microbial Ecology 48, 424-430.

Drinkwater, L.E., Snapp, S., 2007. Nutrients in agroecosystems: rethinking the management paradigm. Advances in Agronomy 92, 163-186.

Drinkwater, L.E., Wagoner, P., Sarrantonio, M., 1998. Legume-based cropping systems have reduced carbon and nitrogen losses. Nature 396, 262-265.

Du, E., de Vries, W., Galloway, J.N., Hu, X., Fang, J., 2014. Changes in wet nitrogen deposition in the United States between 1985 and 2012. Environmental Research Letters 9, 095004.

Elliott, E., 1986. Aggregate structure and carbon, nitrogen, and phosphorus in native and cultivated soils. Soil science society of America journal 50, 627-633.

Engel, R.E., Miller, P.R., McConkey, B.G., Wallander, R., 2017. Soil Organic Carbon Changes to Increasing Cropping Intensity and No-Till in a Semiarid Climate. Soil Science Society of America Journal 81, 404-413.

Farahani, H., Peterson, G., Westfall, D., 1998a. Dryland cropping intensification: A fundamental solution to efficient use of precipitation. Advances in Agronomy (USA).

Farahani, H., Peterson, G., Westfall, D., Sherrod, L., Ahuja, L., 1998b. Soil water storage in dryland cropping systems: the significance of cropping intensification. Soil Science Society of America Journal 62, 984-991.

Flora, C.B., Curtiss, C.F., 2014. Social Sustainability of Cellulosic Energy Cropping Systems. Cellulosic Energy Cropping Systems, 315-333.

Franzluebbers, A., 2002. Water infiltration and soil structure related to organic matter and its stratification with depth. Soil and Tillage Research 66, 197-205.

Frostegård, Å., Bååth, E., 1996. The use of phospholipid fatty acid analysis to estimate bacterial and fungal biomass in soil. Biology and Fertility of Soils 22, 59-65.

FSA, 2017. Crop Acreage Data.

Galloway, J.N., Aber, J.D., Erisman, J.W., Seitzinger, S.P., Howarth, R.W., Cowling, E.B., Cosby, B.J., 2003. The nitrogen cascade. Bioscience 53, 341-356.

Gilbert, N., 1995. Researching social life. Sage, Thousand Oaks, CA.

Gilliom, R.J., Barbash, J.E., Crawford, C.G., Hamilton, P.A., Martin, J.D., Nakagaki, N., Nowell, L.H., Scott, J.C., Stackelberg, P.E., Thelin, G.P., 2006. Pesticides in the nation's streams and ground water, 1992-2001. US Geological Survey.

Glaser, B., Strauss, A., 1967. The discovery of grounded theory: Strategies for qualitative research. Aldine, Chicago, IL.

Gotelli, N.J., Chao, A., 2013. Measuring and estimating species richness, species diversity, and biotic similarity from sampling data. Encyclopedia of Biodiversity 5, 195-211.

Grace, J.B., Anderson, T.M., Olff, H., Scheiner, S.M., 2010. On the specification of structural equation models for ecological systems. Ecological Monographs 80, 67-87.

Greb, B., 1979. Reducing drought effects on croplands in the west-central Great Plains.

Guerrero, B., Wright, A., Hudson, D., Johnson, J., Ammoson, S., 2010. The economic value of irrigation in the Texas Panhandle.

Hamel, C., Hanson, K., Selles, F., Cruz, A.F., Lemke, R., McConkey, B., Zentner, R., 2006. Seasonal and long-term resource-related variations in soil microbial communities in wheat-based rotations of the Canadian prairie. Soil Biology and Biochemistry 38, 2104-2116.

Han, W., Di, L., Yagci, A.L., Yang, Z., 2013. Exploring continuous corn cropping patterns and their relationship with geographic factors. Agro-Geoinformatics (Agro-Geoinformatics), 2013 Second International Conference on. IEEE, pp. 490-494.

Hansen, N.C., Allen, B.L., Baumhardt, R.L., Lyon, D.J., 2012. Research achievements and adoption of no-till, dryland cropping in the semi-arid US Great Plains. Field Crops Research 132, 196-203.

Harinikumar, K., Bagyaraj, D., 1988. Effect of crop rotation on native vesicular arbuscular mycorrhizal propagules in soil. Plant and Soil 110, 77-80.

Helms, D., 1990. Conserving the Plains: The soil conservation service in the Great Plains. Agricultural History 64, 58-73.

Ingram, J., Mills, J., Dibari, C., Ferrise, R., Ghaley, B.B., Hansen, J.G., Iglesias, A., Karaczun, Z., McVittie, A., Merante, P., 2016. Communicating soil carbon science to farmers: Incorporating credibility, salience and legitimacy. Journal of Rural Studies 48, 115-128.

Jastrow, J., 1996. Soil aggregate formation and the accrual of particulate and mineral-associated organic matter. Soil Biology and Biochemistry 28, 665-676.

Jastrow, J., Miller, R., Lussenhop, J., 1998. Contributions of interacting biological mechanisms to soil aggregate stabilization in restored prairie. Soil Biology and Biochemistry 30, 905-916.

Kaan, D.A., 2002. An economic evaluation of alternative crop rotations compared to wheat-fallow in Northeastern Colorado. Technical bul.(Colorado Agricultural Experiment Station); TB-02-1.

Kallenbach, C.M., Frey, S.D., Grandy, A.S., 2016. Direct evidence for microbial-derived soil organic matter formation and its ecophysiological controls. Nature Communications 7.

Kline, R.B., 2015. Principles and practice of structural equation modeling. Guilford publications.

Knowler, D., Bradshaw, B., 2007. Farmers' adoption of conservation agriculture: A review and synthesis of recent research. Food policy 32, 25-48.

Ko, J., Ahuja, L.R., Saseendran, S., Green, T.R., Ma, L., Nielsen, D.C., Walthall, C.L., 2012. Climate change impacts on dryland cropping systems in the Central Great Plains, USA. Climatic Change 111, 445-472.

Kong, A.Y., Six, J., Bryant, D.C., Denison, R.F., Van Kessel, C., 2005. The relationship between carbon input, aggregation, and soil organic carbon stabilization in sustainable cropping systems. Soil Science Society of America Journal 69, 1078-1085.

Koohafkan, P., Stewart, B.A., 2008. Water and cereals in drylands. Earthscan.

Kuznetsova, A., Brockhoff, P.B., Christensen, R.H.B., 2015. Package 'lmerTest'. R package version 2.

Lal, R., 2004. Carbon sequestration in dryland ecosystems. Environmental Management 33, 528-544.

Lange, M., Habekost, M., Eisenhauer, N., Roscher, C., Bessler, H., Engels, C., Oelmann, Y., Scheu, S., Wilcke, W., Schulze, E.-D., 2014. Biotic and abiotic properties mediating plant diversity effects on soil microbial communities in an experimental grassland. PloS one 9, e96182.

Lenth, R.V., Hervé, M., 2015. lsmeans: Least-squares means. R package version 2, 20-23.

Lin, B.B., 2011. Resilience in agriculture through crop diversification: adaptive management for environmental change. BioScience 61, 183-193.

Luhmann, N., 1979. Trust and power. United Kingdom: John Wiley and Sons, Inc.

Lyon, D., Bruce, S., Vyn, T., Peterson, G., 2004. Achievements and future challenges in conservation tillage. Proceedings of the 4th international crop science congress, pp. 1-19.

Maaz, T., Wulfhorst, J., McCracken, V., Kirkegaard, J., Huggins, D.R., Roth, I., Kaur, H., Pan, W., 2017. Economic, policy, and social trends and challenges of introducing oilseed and pulse crops into dryland wheat cropping systems. Agriculture, Ecosystems & Environment.

Malik, A.A., Chowdhury, S., Schlager, V., Oliver, A., Puissant, J., Vazquez, P.G., Jehmlich, N., von Bergen, M., Griffiths, R.I., Gleixner, G., 2016. Soil fungal: bacterial ratios are linked to altered carbon cycling. Frontiers in microbiology 7.

Mann, L., 1986. Changes in soil carbon storage after cultivation. Soil Science 142, 279-288.

McDaniel, M., Tiemann, L., Grandy, A., 2014. Does agricultural crop diversity enhance soil microbial biomass and organic matter dynamics? A meta-analysis. Ecological Applications 24, 560-570.

Mikha, M.M., Benjamin, J.G., Vigil, M.F., Nielson, D.C., 2010. Cropping intensity impacts on soil aggregation and carbon sequestration in the central Great Plains. Soil Science Society of America Journal 74, 1712-1719.

Morel, K., Léger, F., 2016. A conceptual framework for alternative farmers' strategic choices: the case of French organic market gardening microfarms. Agroecology and Sustainable Food Systems 40, 466-492.

Mulumba, L.N., Lal, R., 2008. Mulching effects on selected soil physical properties. Soil and Tillage Research 98, 106-111.

Nail, E.L., Young, D.L., Schillinger, W.F., 2007. Government subsidies and crop insurance effects on the economics of conservation cropping systems in eastern Washington. Agronomy journal 99, 614-620.

NDNR, 2017. Landuse Data. Nebraska Department of Natural Resources.

Nelson, R.G., Hellwinckel, C.M., Brandt, C.C., West, T.O., De La Torre Ugarte, D.G., Marland, G., 2009. Energy use and carbon dioxide emissions from cropland production in the United States, 1990–2004. Journal of Environmental Quality 38, 418-425.

Nielsen, D.C., Vigil, M.F., 2010. Precipitation storage efficiency during fallow in wheat-fallow systems. Agronomy Journal 102, 537-543.

Norton, J.B., Mukhwana, E.J., Norton, U., 2012. Loss and recovery of soil organic carbon and nitrogen in a semiarid agroecosystem. Soil Science Society of America Journal 76, 505-514.

Nowak, P.J., 1983. Obstacles to adoption of conservation tillage. Journal of Soil and Water Conservation 38, 162-165.

NRCS, 2013. Unlock the secrets in the soil. USDA.

Ogle, S.M., Breidt, F.J., Paustian, K., 2005. Agricultural management impacts on soil organic carbon storage under moist and dry climatic conditions of temperate and tropical regions. Biogeochemistry 72, 87-121.

Padel, S., 2001. Conversion to organic farming: a typical example of the diffusion of an innovation? Sociologia ruralis 41, 40-61.

Pannell, D., 1999. Social and economic challenges in the development of complex farming systems. Agroforestry Systems 45, 395-411.

Parton, W.J., Gutmann, M.P., Merchant, E.R., Hartman, M.D., Adler, P.R., McNeal, F.M., Lutz, S.M., 2015. Measuring and mitigating agricultural greenhouse gas production in the US Great Plains, 1870–2000. Proceedings of the National Academy of Sciences 112, E4681-E4688.

Paustian, K., Six, J., Elliott, E., Hunt, H., 2000. Management options for reducing CO2 emissions from agricultural soils. Biogeochemistry 48, 147-163.

Peterson, D., Whistler, J., Egbert, S., Brown, J.C., 2011. Mapping irrigated lands by crop type in Kansas. Pecora 18 proceedings, November, 14-17.

Peterson, G., Schlegel, A., Tanaka, D., Jones, O., 1996. Precipitation use efficiency as affected by cropping and tillage systems. Journal of Production Agriculture 9, 180-186.

Peterson, G., Westfall, D., Peairs, F., Sherrod, L., Poss, D., Gangloff, W., Larson, K., Thompson, D., Ahuja, L.R., Koch, M., Walker, C., 2001. Sustainable Dryland Agroecosystem Management. Technical bulletin (Colorado Agricultural Experiment Station); TB-01-2.

Peterson, G.A., Westfall, D.G., 2004. Managing precipitation use in sustainable dryland agroecosystems. Annals of Applied Biology 144, 127-138.

Plaza-Bonilla, D., Arrúe, J.L., Cantero-Martínez, C., Fanlo, R., Iglesias, A., Álvaro-Fuentes, J., 2015. Carbon management in dryland agricultural systems. A review. Agronomy for Sustainable Development 35, 1319-1334.

Plourde, J.D., Pijanowski, B.C., Pekin, B.K., 2013. Evidence for increased monoculture cropping in the Central United States. Agriculture, ecosystems & environment 165, 50-59.

PRISM, 2017. PRISM Climate Data.

Raedeke, A.H., Green, J.J., Hodge, S.S., Valdivia, C., 2003. Farmers, the practice of farming and the future of agroforestry: an application of Bourdieu's concepts of field and habitus. Rural Sociology 68, 64-86.

Reitsma, K.D., Clay, D.E., Clay, S.A., Dunn, B.H., Reese, C., 2016. Does the US cropland data layer provide an accurate benchmark for land-use change estimates? Agronomy Journal 108, 266-272.

Robertson, P.G., Gross, K.L., Hamilton, S.K., Landis, D.A., Schmidt, T.M., Snapp, S.S., Swinton, S.M., 2014. Farming for ecosystem services: An ecological approach to production agriculture. BioScience 64, 404-415.

Rogers, E.M., 2003. Diffusion of innovations. Free Press. New York, 551.

Rosenzweig, S.T., Carson, M.A., Baer, S.G., Blair, J.M., 2016. Changes in soil properties, microbial biomass, and fluxes of C and N in soil following post-agricultural grassland restoration. Applied Soil Ecology 100, 186-194.

Rosseel, Y., 2011. lavaan: an R package for structural equation modeling and more Version 0.4-9 (BETA). Ghent University.

Rousk, J., Frey, S.D., 2015. Revisiting the hypothesis that fungal-to-bacterial dominance characterizes turnover of soil organic matter and nutrients. Ecological Monographs 85, 457-472.

Sahajpal, R., Zhang, X., Izaurralde, R.C., Gelfand, I., Hurtt, G.C., 2014. Identifying representative crop rotation patterns and grassland loss in the US Western Corn Belt. Computers and electronics in agriculture 108, 173-182.

Saltiel, J., Bauder, J.W., Palakovich, S., 1994. Adoption of sustainable agricultural practices: Diffusion, farm structure, and profitability. Rural sociology 59, 333-349.

Sebilo, M., Mayer, B., Nicolardot, B., Pinay, G., Mariotti, A., 2013. Long-term fate of nitrate fertilizer in agricultural soils. Proceedings of the National Academy of Sciences 110, 18185-18189.

Shapiro, B.I., Brorsen, B.W., Doster, D.H., 1992. Adoption of double-cropping soybeans and wheat. Journal of Agricultural and Applied Economics 24, 33-40.

Shaver, T., Peterson, G., Ahuja, L., Westfall, D., Sherrod, L., Dunn, G., 2002. Surface soil physical properties after twelve years of dryland no-till management. Soil Science Society of America Journal 66, 1296-1303.

Shaver, T., Peterson, G., Sherrod, L., 2003. Cropping intensification in dryland systems improves soil physical properties: regression relations. Geoderma 116, 149-164.

Sherrod, L., Dunn, G., Peterson, G., Kolberg, R., 2002. Inorganic carbon analysis by modified pressure-calcimeter method. Soil Science Society of America Journal 66, 299-305.

Sherrod, L., Peterson, G., Westfall, D., Ahuja, L., 2003. Cropping intensity enhances soil organic carbon and nitrogen in a no-till agroecosystem. Soil Science Society of America Journal 67, 1533-1543.

Sherrod, L., Peterson, G., Westfall, D., Ahuja, L., 2005. Soil organic carbon pools after 12 years in no-till dryland agroecosystems. Soil Science Society of America Journal 69, 1600-1608.

Simkin, S.M., Allen, E.B., Bowman, W.D., Clark, C.M., Belnap, J., Brooks, M.L., Cade, B.S., Collins, S.L., Geiser, L.H., Gilliam, F.S., 2016. Conditional vulnerability of plant diversity to atmospheric nitrogen deposition across the United States. Proceedings of the National Academy of Sciences 113, 4086-4091.

Six, J., Elliott, E., Paustian, K., 1999. Aggregate and soil organic matter dynamics under conventional and no-tillage systems. Soil Science Society of America Journal 63, 1350-1358.

Six, J., Frey, S., Thiet, R., Batten, K., 2006. Bacterial and fungal contributions to carbon sequestration in agroecosystems. Soil Science Society of America Journal 70, 555-569.

Six, J., Paustian, K., 2014. Aggregate-associated soil organic matter as an ecosystem property and a measurement tool. Soil Biology and Biochemistry 68, A4-A9.

Smith, E., Young, D.L., 2000. The economic and environmental revolution in semi-arid cropping in North America. Annals of Arid Zone 39, 347-362.

Stern, A.J., Doraiswamy, P.C., Hunt, E.R., 2012. Changes of crop rotation in Iowa determined from the United States Department of Agriculture, National Agricultural Statistics

Service cropland data layer product. Journal of Applied Remote Sensing 6, 063590-063590.

Strauss, C., 2006. The imaginary. Anthropological theory 6, 322-344.

Strepanovic, S.C., C.; Santra, D.; Adesemoye, T., 2016. Field pea production: Rotational costs and benefits. University of Nebraska Lincoln.

Strickland, M.S., Rousk, J., 2010. Considering fungal: bacterial dominance in soils–methods, controls, and ecosystem implications. Soil Biology and Biochemistry 42, 1385-1395.

Stromberger, M., Shah, Z., Westfall, D., 2007. Soil microbial communities of no-till dryland agroecosystems across an evapotranspiration gradient. Applied Soil Ecology 35, 94-106.

Sutherland, L.A., Burton, R.J., 2011. Good farmers, good neighbours? The role of cultural capital in social capital development in a Scottish farming community. Sociologia Ruralis 51, 238-255.

Taylor, C., 2004. Modern social imaginaries. Duke University Press.

Team, R.C., 2013. R: A language and environment for statistical computing. R foundation for statistical computing, Vienna, Austria.

Thompson, J., 1987. Decline of vesicular-arbuscular mycorrhizae in long fallow disorder of field crops and its expression in phosphorus deficiency of sunflower. Australian Journal of Agricultural Research 38, 847-867.

Tilman, D., 1999. Global environmental impacts of agricultural expansion: the need for sustainable and efficient practices. Proceedings of the National Academy of Sciences 96, 5995-6000.

Tilman, D., Fargione, J., Wolff, B., D'Antonio, C., Dobson, A., Howarth, R., Schindler, D., Schlesinger, W.H., Simberloff, D., Swackhamer, D., 2001. Forecasting agriculturally driven global environmental change. Science 292, 281-284.

Tisdall, J.M., Oades, J.M., 1982. Organic matter and water-stable aggregates in soils. Journal of Soil Science 33, 141-163.

UN, 2011. Global drylands: a UN system-wide response. Environment Management Group of the United Nations Geneva. Retrieved from http://www. unccd. int/Lists/SiteDocumentLibrary/Publications/Global_Drylands_Full_Report. pdf.

USDA-ERS, 2017. Farm Household Income Forecast. Farm Household Well-being.

USDA-NASS, 2017a. Croplands Data Layer.

USDA-NASS, 2017b. Quick Stats.

USGCRP, 2014. Climate change impacts in the United States: U.S. national climate assessment. GlobalChange.gov.

Van Bavel, C., 1950. Mean weight-diameter of soil aggregates as a statistical index of aggregation. Soil Science Society of America Journal 14, 20-23.

Van der Ploeg, J.D., 2006. Agricultural production in crisis. Handbook of rural studies, 258-277.

Wall, D.H., 2012. Soil ecology and ecosystem services. Oxford University Press.

Waring, B.G., Averill, C., Hawkes, C.V., 2013. Differences in fungal and bacterial physiology alter soil carbon and nitrogen cycling: Insights from meta-analysis and theoretical models. Ecology letters 16, 887-894.

Warner, K.D., 2008. Agroecology as participatory science: Emerging alternatives to technology transfer extension practice. Science, Technology, & Human Values 33, 754-777.

Weber, M., 1993. The sociology of religion. Beacon Press.

Wilson, G.A., 2001. From productivism to post-productivism… and back again? Exploring the (un) changed natural and mental landscapes of European agriculture. Transactions of the Institute of British Geographers 26, 77-102.

Wilson, G.W., Rice, C.W., Rillig, M.C., Springer, A., Hartnett, D.C., 2009. Soil aggregation and carbon sequestration are tightly correlated with the abundance of arbuscular mycorrhizal fungi: results from long-term field experiments. Ecology letters 12, 452-461.

Yates, J., Smith, J., Pate, J., Weinheimer, J.A., Dudensing, R., Johnson, J., 2010. Regional economic impact of irrigated versus dryland agriculture in the Texas High Plains. Beltwide Cotton Conference Proceedings.

Young, B.G., 2006. Changes in Herbicide Use Patterns and Production Practices Resulting from Glyphosate-Resistant Crops1. Weed Technology 20, 301-307.

CHAPTER 5: LANDSCAPE-SCALE CROPPING TRANSFORMATION IN THE HIGH PLAINS: ECONOMIC AND ENVIRONMENTAL IMPLICATIONS

5.1 Introduction

A global revolution in semi-arid cropping systems is underway, with implications for the environment and economies of semi-arid regions (Smith and Young, 2000; Maaz et al., 2017). Semi-arid regions constitute 20% of Earth's land surface and are home to nearly 1 billion people (UNSO/UNDP, 1997). These large and growing populations have placed enormous pressure on dryland (non-irrigated) farms of semi-arid regions and the natural resources on which they depend. As much as 40% of agricultural land globally suffers from severe soil degradation (Koohafkan and Stewart, 2008), and dryland soils are especially vulnerable due to water limitations that constrain crop productivity and exacerbate soil erosion (Bot et al., 2000). Crop production and soil health are further constrained in semi-arid cropping systems by a year long period called summer fallow, where no crops are grown and weeds are controlled, commonly practiced in dryland agriculture in many regions. While summer fallow reduces risk of crop failure by accumulating precipitation in the soil, decades of crop rotations with frequent summer fallow have led to degraded soils and management systems dependent on large and increasing amounts of chemical inputs (Peterson et al., 1998; Derksen et al., 2002; Chapter 3). However, over the last several decades, technological advancements have enabled farmers to intensify cropping systems by replacing summer fallow with a crop, which has the potential to reverse these detrimental economic and environmental trends.

Typically, agricultural intensification is supported by a greater use of chemical inputs. For example, the observed doubling in global agricultural production from the 1960s to the

1990s coincided with almost a 7-fold increase in nitrogen (N) inputs and a 3-fold increase in pesticide use, with cascading effects for the environment and human health (Tilman, 1999; et al., 2001; Galloway et al., 2003; Gilliom et al., 2006). In contrast, semi-arid cropping intensification can increase crop production and residues returned to the soil while reducing chemical inputs (Chapter 3). As a result, reducing summer fallow frequency can simultaneously contribute to desirable environmental and economic outcomes (Kaan et al., 2002; Shaver et al., 2002; Sherrod et al., 2003; Peterson and Westfall, 2004; Chapter 3). However, the extent of recent intensification patterns, and the potential impacts of broader adoption, have not been characterized at regional and landscape scales.

The transformation in semi-arid cropping systems has been documented in several regions around the world, including Australia, Canada, and the Northern Great Plains (Smith and Young, 2000; 2003; Hansen et al., 2012; Maaz et al., 2017), but the extent of the revolution has not been quantified in the High Plains of the US – an important grain growing region in the Western Great Plains including the states of Colorado, Kansas, and Nebraska. Since the Dust Bowl in the 1930s, most dryland farmers in the High Plains have grown a rotation of winter wheat and summer fallow to mitigate the risk of farming in a highly variable and dry climate. However, many no-till farmers are transitioning from wheat-fallow to a mid-intensity rotation (e.g., wheat-corn-fallow), and some have even eliminated summer fallow completely through diverse crop rotations, a practice called continuous cropping. Continuous cropping likely provides a greater environmental and economic benefit relative to mid-intensity and wheat-fallow (Chapter 3), but previous estimates of intensification have only been derived from changes in individual crop or fallow acreages, and thus we lack an estimate of cropping systems-level changes. For example, Hansen et al. (2012) report a reduction in summer fallow acres by

70% in the US Northern Great Plains since 1990, and Maaz et al. (2017) report similar reductions in several Canadian provinces, suggesting that millions of hectares are being intensified. A better understanding of the extent and degree of cropping system intensification could provide policymakers with information about the economic and environmental implications of these transformations, and place these shifts into the broader context of the global cropping revolution.

Agricultural communities across the United States, including those in the High Plains, are facing a crisis of profitability. From 2013 to 2017, net farm income declined 50%, and most farmers today cannot survive without off-farm income and government subsidies (Carolan, 2016; ERS, 2017). Over the past several decades, farmers have increasingly paid more for inputs like equipment and chemicals, while commodity prices have remained stagnant or declined, a phenomenon called the “double squeeze” (van der Ploeg, 2006). Given the pressure on profits, summer fallow creates an economic burden due to the high cost of herbicides required to manage weeds in no-till systems, and gross incomes that are constrained by an entire year without crop production. However, total grain production can be enhanced by 60-75% in intensified systems relative to wheat-fallow (Peterson et al., 2004), and substantial herbicide reductions can be achieved through increased plant competition with weeds and the ability to rotate different herbicides in intensified systems (Derksen et al., 2002; Chapter 3). Additionally, I found that, despite achieving greater crop production, continuous crop farmers in the High Plains apply similar amounts of N fertilizer as traditional wheat-fallow farmers, leading to 80% greater net income (Chapter 3). Thus, the intensification of cropping systems may provide an opportunity to act on both sides of the double squeeze – to reduce inputs while enhancing crop production.

Due to the large amounts of land dedicated to dryland crop production, even proportionately small shifts in cropping patterns can translate to significant environmental impacts (Lal, 2004). Scaling field-level estimates of C sequestration to the landscape level can inform US and international climate policy aimed at increasing terrestrial C offsets (Conant et al., 2011). Parton et al. (2015) estimated that the Great Plains region (which encompasses the High Plains) only accounts for 5% of US agricultural greenhouse gas emissions, but also found that this region has the potential to become a net C sink through the adoption of sustainable agricultural technologies, particularly through reduced tillage. Intensified cropping systems increase soil organic carbon (SOC) beyond the effects of reducing tillage through greater crop-derived C input to soil (Sherrod et al., 2003; Chapter 2). The analysis by Parton et al. (2015), however, did not include the C sequestration benefits of cropping system intensification, and thus may have underestimated the region's potential to mitigate climate change.

I conducted a spatial analysis of cropping patterns in the High Plains from 2008-2016 to answer three questions: 1) To what extent is cropping system intensity increasing in the High Plains? 2) Which crops are replacing summer fallow and where? 3) What are the implications of observed intensification in the High Plains for C sequestration, grain yield, herbicide use, and net farm incomes? Assessing cropping system intensification patterns and potential implications in the High Plains will generate insights that will help us understand the broader semi-arid cropping revolution occurring around the world.

5.2 Methods

5.2.1 *Study area*

The study area consists of the main semi-arid dryland cropping counties in the High Plains states of Colorado, Kansas, and Nebraska. Counties were selected if a majority of the area receives less than <500 mm average annual precipitation, estimated using rasterized data from PRISM (2017). Average annual precipitation from 1980-2010 for the study area is 432 mm (PRISM, 2017). The study years included a representative range of wetter and drier than normal years. Annual precipitation in the study period (2008-2016) ranged from 266 mm in 2012 to 605 mm in 2015 (PRISM, 2017). All selected counties had at least 2000 acres in dryland wheat production as of 2016.

5.2.2 *Data*

All crop acreage and crop rotation analysis was performed using the Croplands Data Layer (CDL) provided by the National Agricultural Statistics Service (NASS). The CDL is a geo-referenced raster data layer using a combination of satellite imagery and extensive agricultural ground truthing (NASS, 2017). The CDL contains agricultural land cover data for the contiguous United States, and has been used to estimate crop acreage, crop rotations, and rotational diversity over time (Stern et al., 2012; Plourde et al., 2013; Yang et al., 2013; Sahajpal et al., 2014). CDL maps for Colorado, Kansas, and Nebraska for the years 2008 to 2016 were analyzed using arcGIS. CDLs from 2008 and 2009 were generated at 56 m resolution, and were reprocessed to a 30 m resolution to match the resolution of the other years.

The CDL has low accuracy in classifying non-agricultural land-uses (NASS, 2017), and thus the 2011 National Land Cover Database (NLCD) was used to mask-out non-agricultural classes and grasslands, as others have done to isolate croplands specifically (e.g., Plourde et al., 2013). Spatial irrigation data was used to mask-out irrigated acreage in Colorado using data from 2010 (and 2015 and 2016 only for the Republican River Basin) available from Colorado Decision Support Systems through the Colorado Department of Natural Resources (CDWR, 2017). Nebraska drylands were isolated by masking irrigated lands using data from 2005 available from the Nebraska Department of Natural Resources (NDNR, 2017), and similarly for Kansas using data centered around 2007 provided by the Kansas Applied Remote Sensing Program (Peterson et al., 2011).

5.2.3 *Crop acreage*

Crop pixel counts were converted to acres then hectares. I compared CDL estimates of crop acreage to Farm Service Agency (FSA) data from 2011 to 2016 (FSA, 2017). CDL data typically underestimate acres of crops relative to FSA or NASS data (Johnson, 2013), but accuracies in this study were high for major crops like corn (97%), winter wheat (92%), and sorghum (84%). Accuracies were lower for minor crops like millet (71%) and peas (42%), suggesting that there is much more land in production for these crops than estimated here (FSA, 2017).

While I removed much of the grasslands from the dataset using the NLCD, I found that many fallow acres were incorrectly classified as grasslands or open space in 2008, but classification improved gradually through subsequent years. Others have encountered similar confusions between grasslands and croplands (e.g., Reitsma et al., 2016), necessitating a means of correctly identifying these classes. I obtained more accurate assessments of fallow land (92% accuracy compared to FSA data) by classifying any fallow, grassland, or open space classes as fallow only if they were rotated with other crops. I calculated the effective number of crop species (ENCS) for each year from 2008 to 2016 as an index of crop species diversity (Gotelli and Chao, 2013) as described in Aguilar et al. (2015).

5.2.4 *Crop rotations*

Cropping system intensities were assessed in 3-year sequences, and new rasters were generated denoting the 3-year cropping system intensity for each pixel (Figure 5.2). Crops were divided into three classes: winter wheat, fallow (including grassland and open space classes rotated with crops as previously discussed), and other crops. Crop-fallow was designated as all sequences with 2 occurrences of fallow or a wheat-fallow-wheat sequence, mid-intensity was designated as all sequences with 1 occurrence of fallow, and continuous was designated as all sequences with 0 occurrences of fallow. Sequences with 3 occurrences of fallow were classified as grassland or open space, and removed from subsequent analysis. The rasters of cropping system intensity were compared for each 3-year rotation window to produce maps of changes in cropping system intensity through time (Figure 5.2). Acreage for each crop in each year was calculated from the number of pixels in the CDL for each crop class (Figure 5.3). Crop footprint maps were developed for each 3-year rotation window by including pixels if they grew a particular crop at any point in the 3-year period (Figure 5.4).

5.2.5 *C sequestration, herbicide use, grain production, and net income analyses*

Concurrent studies measured the differences in SOC, annualized grain yield, and input use between the different cropping system intensities in the High Plains across the geographic extent of the study area examined here (Chapters 2 and 3). Data used for the spatial and regional analysis in this study include least-squared means and standard errors generated by statistical models of cropping system intensity and significant soil, management, and climate covariates (Table 5.1). Sampling procedures for each metric are described in Chapters 2 and 3. I obtained regional assessments of C sequestration, herbicide use, grain production, and net income by multiplying the least-squared means and standard errors for each cropping system by the number of hectares in each cropping system for each year.

5.3 Results

Substantial levels of cropping system intensification occurred from 2008 to 2016 as summer fallow was replaced by alternative (non-wheat) crops. This regional transformation in cropping systems is likely associated with increases in soil C sequestration and crop production, and decreases in herbicide use, contributing to higher net farm incomes.

5.3.1 *Changes in cropland*

Dry cropland in the study area increased from 3.9 to 4.0 million hectares from 2008 to 2016. Crop-fallow was the dominant dryland cropping system in 2008, consisting of 2.1 million hectares, or 53% of dry cropland (Figure 5.1). By 2016, however, crop-fallow only represented 42% of dry cropland, with mid-intensity and continuous rotations constituting 43% and 15% of dry cropland, respectively. Over the 9-year period, land in mid-intensity rotations increased by 0.3 million hectares, and land in continuous rotations increased by 0.2 million hectares, while land in crop-fallow decreased by 0.4 million hectares (Figure 5.1, Figure 5.2). The extra 0.1 million hectares in intensified rotations likely came from land taken out of the Conservation Reserve Program. Transitions to continuous rotations were most pronounced in northeastern

Colorado and southwestern Nebraska, whereas transitions to mid-intensity rotations were most pronounced in western Kansas (Figure 5.2).

Reductions in summer fallow from 1.8 to 1.3 million hectares (48% to 33% of dry cropland) from 2008 to 2016 were largely matched by increases in alternative (non-wheat) crops (Figure 5.3a). Four crops (corn, sorghum, millet, and field peas) increased by nearly 0.4 million hectares, from 17% of dry cropland in 2008 to 26% by 2016 (Figure 5.3), while increases in winter wheat remained proportional to overall increases in dry cropland. Corn and sorghum made up the vast majority of alternative crop acreage, although pea and millet acreage also increased substantially over the 9-year period (Figure 5.3b). Surprisingly, sunflower acreage decreased by about a third. There was a great deal of interannual variability in crop acreage. Crop prices were surprisingly unrelated to annual trends in crop acreage (data not shown). I suspect that large drops in acreage, as observed in corn and sorghum in 2014, are mostly related to abnormally wet conditions that prevented planting (PRISM, 2017).

Alternative crops were adopted in distinct sub-regions within the High Plains (Figure 5.4). Growth in sorghum production largely occurred in western Kansas and southeastern Colorado, and replaced much of the corn production in these regions. Corn was largely adopted further north (Figure 5.4). Millet adoption mostly occurred in mid and northeastern Colorado and the southern Nebraska panhandle, and peas were adopted almost exclusively in northeastern Colorado and Nebraska (Figure 5.4). I found a slight increase in ENCS from 2008 to 2014, indicating increasing diversity in dryland cropping systems (Figure 5.5).

5.3.2 *Implications of cropping system intensification*

Cropping intensity corresponds to decreases in herbicide use. I found that continuously cropped farmers applied roughly 50%, 80%, and 60% less glyphosate, 2,4-D, and dicamba than wheat-fallow farmers, respectively, across this region (Chapter 3). Scaling up these estimates to the landscape level suggests modest reductions in the use of three common herbicides (glyphosate, 2,4-D, and dicamba) were achieved from 2008 to 2016, despite increases in overall cropland. Annual glyphosate use decreased by 70 Mg acid equivalent (AE) (1%) (Figure 5.6a), 2,4-D use decreased by 150 Mg AE (6%) (Figure 5.6b), and dicamba use fell 7 Mg AE (1%) from 2008 to 2016 (Figure 5.6c).

Using farmer-reported yield histories from 2010-2015, I found that mid-intensity and continuous rotations had 46% and 60% greater annualized grain production relative to wheat-fallow, respectively, after accounting for the effects of fertilizer use and climate (as described in Chapter 3). Expanding these estimates to the landscape scale suggests larger gains in crop production were realized than can be accounted for by increases in dry cropped acres alone (Figure 5.6d). Production increases associated with land conversion to cropland were 0.17 Tg grain (3%) from 2008 to 2016. Including the annualized yield differences between cropping system intensities suggests an increase of 0.53 Tg grain production (9%) from 2008 to 2016 (Figure 5.6d).

Comparisons of partial enterprise budgets from 2010 to 2014 (Chapter 3) demonstrate the potential implications of cropping system intensification for regional economies. I found that annualized net incomes were 70% and 80% higher in mid-intensity and continuous crop rotations compared to wheat-fallow, respectively (Chapter 3). Scaling these results to the landscape level suggests that that annual net operating incomes in dryland agriculture increased by $80 million

(10%) from 2008 to 2016 (Figure 5.6e). For comparison, total net farm cash income in the study area, including irrigated land and livestock, was $2.6 billion in 2012 according to NASS census data (NASS, 2017).

I concurrently assessed soil organic carbon (SOC) stocks to 10 cm on 96 dryland no-till fields across the full extent of the study area (Chapter 2). After accounting for potential evapotranspiration (PET) and soil clay content as covariates, I found that no-till mid-intensity rotations stored on average 0.02 Mg C/ha/yr and no-till continuous rotations stored 0.08 Mg C/ha/yr relative to no-till wheat-fallow (Chapter 2). Scaling these results to the landscape-level trends in cropping system intensity suggests that annual soil C sequestration increased 38% from 2008 to 2016, with levels of annual C sequestration from dryland agriculture increasing from 63 Gg C/yr to 88 Gg C/yr, or 0.32 $MMTCO_2e$, by 2016 (Figure 5.6f).

We predicted theoretical maximum grain production (Figure 5.6d), and net income gains (Figure 5.6e), maximum C sequestration (Figure 5.6f), and theoretical minimum herbicide usage (Figure 5.6a, 5.6b, 5.6c) by calculating these metrics based on a scenario of 100% adoption of no-till, continuous cropping practices. Theoretical minimum glyphosate usage (at 100% continuous adoption) is 3760 Mg AE/yr, or about 60% of the current level (Figure 5.7a). Minimum 2,4-D usage is 640 AE/yr, or 28% of current levels (Figure 5.7b). Minimum dicamba usage is 240 Mg AE/yr, less than half of current levels (Figure 5.7c). Theoretical maximum grain production is 8.23 Tg grain/yr (Figure 5.7e), or about 24% of current levels. The theoretical maximum net operating income based on 100% continuous no-till suggests that over $1 billion/yr could be generated before fixed costs, representing a 26% increase over today's operating incomes. If all dryland crop acreage transitioned to continuous rotations, annual C sequestration would surpass 310 Gg C/yr relative to a baseline of 100% no-till wheat-fallow, or 223 Gg C/yr above current levels (Fig. 5.6f).

5.4 Discussion

Our results suggest there are high rates of cropping system intensification and diversification, even in the volatile and dry climate of the High Plains. Notably, since 2008, the dominant dryland cropping system by land area changed from wheat-fallow to mid-intensity rotations. This crossover is a historic landmark for agriculture in this region, as it almost certainly represents the first widespread transformation in dryland cropping systems in the High Plains since the Dust Bowl. Additionally, as no-till is usually a prerequisite for cropping system intensification (Peterson et al., 1996), increasing intensity likely reflects increasing no-till adoption, which suggests that no-till adoption in the High Plains is higher than previously reported. There are no recent studies of no-till adoption in the High Plains, but estimates from 2004 suggested about 20% of total cultivated land in this region was under no-till management (CTIC, 2011; Hansen et al., 2012). Based on our estimates of cropping system intensification, it is now likely that over half of dryland acres in the High Plains are under no-till or reduced-till management relative to conventionally tilled wheat-fallow.

Trends towards increasing cropping system intensity likely reflect numerous shifts that are encouraging or enabling farmers to intensify. First, as previously mentioned, high no-till adoption rates are likely enabling intensification for many dryland farmers. Technological diffusion often follows an “S” curve, with slow initial adoption largely due to lack of information about the technology, followed by exponential growth as innovators and early adopters generate and spread information about emerging technologies (Rogers, 2003). Since no-till and intensification in the High Plains began, largely in the 1980s, farmers and researchers have learned a great deal about these innovations and their agronomic and economic benefits. Long-term cropping systems experiments have been critical in generating information about intensification, demonstrating the potential to build soil structure and SOC (Shaver et al., 2002; Sherrod et al., 2003), maximize precipitation use efficiency leading to greater crop production (Peterson et al., 1996; Farahani et al., 1998), and generate more income (Kaan et al., 2002). However, no-till is becoming more difficult due to the increasing pervasiveness of herbicide resistant weeds, which require greater amounts and more expensive herbicides to control (Young, 2006). Intensified and diversified crop rotations offer a potential solution to increasing herbicide costs through ecologically-based weed management that reduces reliance on herbicides (Anderson, 2004).

Second, transitions to continuous cropping are linked to the emerging soil health movement, which has been effective in disseminating stories of intensified and diversified dryland farmers who have been able to increase yields while using fewer chemical inputs (Chapter 4). Coupled with greater understanding of the soil health benefits of intensification, the opportunity to capitalize on input reduction as a profitability maximization strategy likely motivated and enabled many producers to continuous crop (Carlisle, 2016; Chapter 4). Still, continuous rotations only represent 15% of the dry cropped area, and are increasing at a slower rate than mid-intensity cropping systems (Figure 5.1). Continuous cropping is still considered high risk by many farmers, highlighting the need for more research and outreach related to continuous cropping (Nielsen, unpublished data).

Third, genetic improvements in alternative crops have created more economically viable choices for dryland farmers in the High Plains. For example, new drought-resistant corn varieties have improved corn yields and survival in the High Plains, enabling its use as a staple crop, though it remains one of the more risky crops in dryland rotations under variable weather conditions. Breeding research and alternative crop development has been found to be a key driver of intensification in other regions, such as the development of canola in Canada and Australia (Maaz et al., 2017). Lastly, improved access to markets and local infrastructure for processing is likely a key driver of increases in sorghum and pea adoption in particular. New pea processing facilities in western Nebraska likely drove the observed exponential increases in pea adoption in the region (Strepanovik, 2016). These large sub-regional increases in pea adoption, as well as sorghum and millet, are likely underlying the trend toward increasing diversity in the broader High Plains region. Still, there are many barriers – social, economic, political, and environmental – that are likely to impede intensification to some degree (Carlisle, 2016; Chapter 4). To assess the extent to which farmers are able to overcome these barriers, the simple procedure developed in this study for measuring cropping system intensity can track the progress of the transformations moving forward by annually tracking cropping patterns. This analysis can be applied to quantify the extent of the cropping system revolution in other regions as well, although it requires high-resolution spatial crop data that is not available in every country.

Cropping system intensification represents a win-win strategy for environment and economy, and these results suggest that there are significant regional economic benefits of the widespread transformation. Improving the economic outcome of dryland agriculture is increasingly critical as water resources for irrigation in the High Plains are dwindling while competition for water from growing urban areas is increasing (Dennehy et al., 2002), and thus the region is likely to see more frequent conversions from irrigated to dryland agriculture. Economic estimates suggest trends could result in several billion dollars in lost revenue in the

Texas High Plains (Guerrero et al., 2010; Yates et al., 2013). Therefore, boosting dryland economic prospects will be critical to maintaining economic stability in the region. I estimated that the effect of intensification alone (comparing net operating income at 100% wheat-fallow vs. 100% continuous crop adoption) represents roughly 23% of the study region's entire net farm income, and thus cropping system intensification may be a key strategy for protecting the High Plains agricultural economy in the face of growing water scarcity. The economic benefits of intensification are associated with greater grain production and lower herbicide use, both of which have societal benefits beyond economics.

Trends toward increasing crop rotational diversity in the High Plains run counter to increasing monoculture in the broader Central US region (Plourde et al., 2013). Increasing crop diversity associated with intensification also provides some degree of economic stability. Diversification helps stabilize year-to-year incomes, as it mitigates profit risk, which includes yield risk and price risk for inputs and outputs (Di Falco and Perrings, 2005; Bowman and Zilberman, 2013). The ENCS of about 3 suggests that 3 crops (wheat, corn, and sorghum) dominate the majority of dryland crop production in the study region (Aguilar et al., 2015), but an increase of 0.4 from 2008 to 2016 means minor crops like millet and peas are increasingly playing important roles in High Plains agriculture.

The environmental benefits of changes in cropping patterns over the 9-year study period were modest, but theoretical maximum levels of C sequestration suggest large potential impacts should intensification continue. While there are no emissions reduction goals in Kansas or Nebraska, an executive order in Colorado set statewide emissions reduction goals of 26% by 2025 relative to 2005 levels (Hickenlooper, 2017). Colorado agriculture produced 9 million metric tons of CO_2 greenhouse gas equivalents ($MMTCO_2e$), or about 7% of the state's emissions in 2005 (Arnold et al., 2014). Our estimates of C sequestration due to intensification alone as of 2016 are roughly 0.32 $MMTCO_2e$/yr relative to a baseline of no-till wheat-fallow. These C sequestration rates contribute over 3.5% emissions reductions in the agricultural sector of Colorado. However, the theoretical C sequestration rate at 100% continuous crop represent emissions reductions is 0.82 $MMTCO_2e$/yr above today's levels, which is almost a tenth of the greenhouse gas emissions in the agricultural sector of Colorado. While it is highly unlikely that this level of adoption would be achieved by 2025, and considering these estimates are also derived from land including counties in Kansas and Nebraska, these results demonstrate the potential for agricultural soils to contribute towards emissions reduction goals. Furthermore, soil C sequestration not only acts in the service of climate change mitigation, but also provides agroecosystem services associated with soil organic matter. Soil functions like water capture and storage, erosion control, and nutrient supply contribute to agricultural resilience and have the potential to improve farm economies (Wall et al., 2012; Robertson et al., 2014).

These are likely conservative estimates of actual C sequestration for several reasons. First, estimates of SOC stocks reported in Chapter 2 are likely underestimating actual SOC stocks, considering bulk density measurements were lower than previously reported from long-term experimental sites. For example, average bulk density measurements used from long-term experimental sites ranged from 0.91-1.11 g/cm^2 while Sherrod et al. (2005) estimated the same fields to be 1.24-1.59 g/cm^2. Second, calculating C sequestration relative to a baseline of 100% no-till wheat-fallow does not account for C accrual associated with no-till. Ogle et al. (2005) and Parton et al. (2015) attribute C sequestration gains of 0.1 Mg C/ha/yr associated with no-till adoption in dry temperate agroecosystems independent of cropping patterns. As no-till adoption usually occurs in conjunction with cropping system intensification, actual C sequestration associated with trends in cropping system intensification are likely much higher. Thus, 100% adoption of continuous no-till cropping systems would lead to C sequestration rates of over 822 Gg C/yr, or 3 $MMTCO_2e$/yr, relative to a baseline of tilled wheat-fallow. These SOC gains are also only for the top 10 cm, although accrual likely occurs at least to 20 cm depth, further suggesting that these may be underestimates of C sequestration.

5.4.1 *Conclusion*

Recent evidence suggests that semi-arid dryland cropping systems around the world are intensifying. However, this evidence was based on estimates of changes in annual crop and fallow acreages, and thus we lacked an understanding of the degree of system-level intensification (mid-intensity vs. continuous) and the implications for the environment and economy. I conducted a spatial analysis of multi-year cropping patterns to quantify cropping system intensification in the High Plains in order to understand the rate at which dryland farmers are intensifying and where they are intensifying, and to estimate the potential environmental and economic outcomes. I found that the High Plains, despite having an extremely dry and volatile climate that likely constrains intensification, saw a profound shift in the dominant cropping system within the last few years; intensified cropping systems now make up the majority of dry croplands in the region. I estimated that this transformation has the potential to contribute to greenhouse gas mitigation targets, increase food production, and generate millions of dollars for farmers in the High Plains. As satellite data becomes more widely available, this analysis could be extended to other semi-arid regions around the world to provide policymakers with critical information about the implications of the semi-arid cropping revolution.

Table 5.1. Field-scale values used to scale up implications of cropping system intensity to the landscape level. Data are model-generated least-squared means $\pm$ SE. PET=Potential evapotranspiration. AE=Acid Equivalent

	Wheat-fallow	Mid-intensity	Continuous	Covariates
Glyphosate (kg AE/ha/yr)	1.79$\pm$0.28	1.42$\pm$0.15	0.93$\pm$0.25	-
2,4-D (kg AE/ha/yr)	0.82$\pm$0.10	0.47$\pm$0.05	0.15$\pm$0.08	pH
Dicamba (kg AE/ha/yr)	0.17$\pm$0.03	0.14$\pm$0.02	0.05$\pm$0.03	-
Annualized Grain Yield (Mg/ha/yr)	1.27$\pm$0.11	1.85$\pm$0.11	2.03$\pm$0.20	N fertilizer use, PET
Annualized Net Income ($/ha/yr)	147$\pm$36	250$\pm$24	264$\pm$39	PET
SOC stock (Mg/ha)	10.65$\pm$0.41	11.10$\pm$0.38	12.30$\pm$0.42	PET, % Clay, Slope

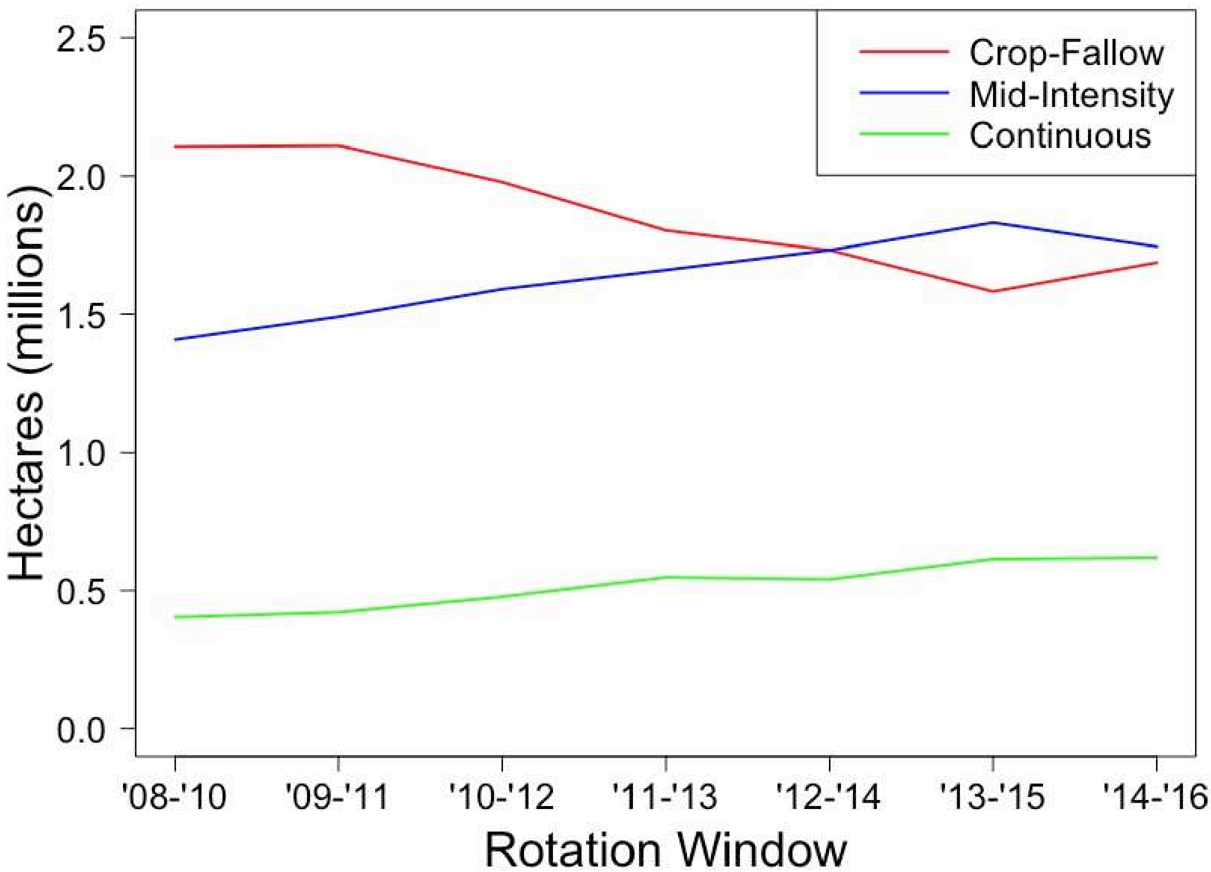

Figure 5.1. Patterns in dryland cropping system intensity in 3-year rotation windows from 2008 to 2016 in the High Plains region of Colorado, Kansas, and Nebraska.

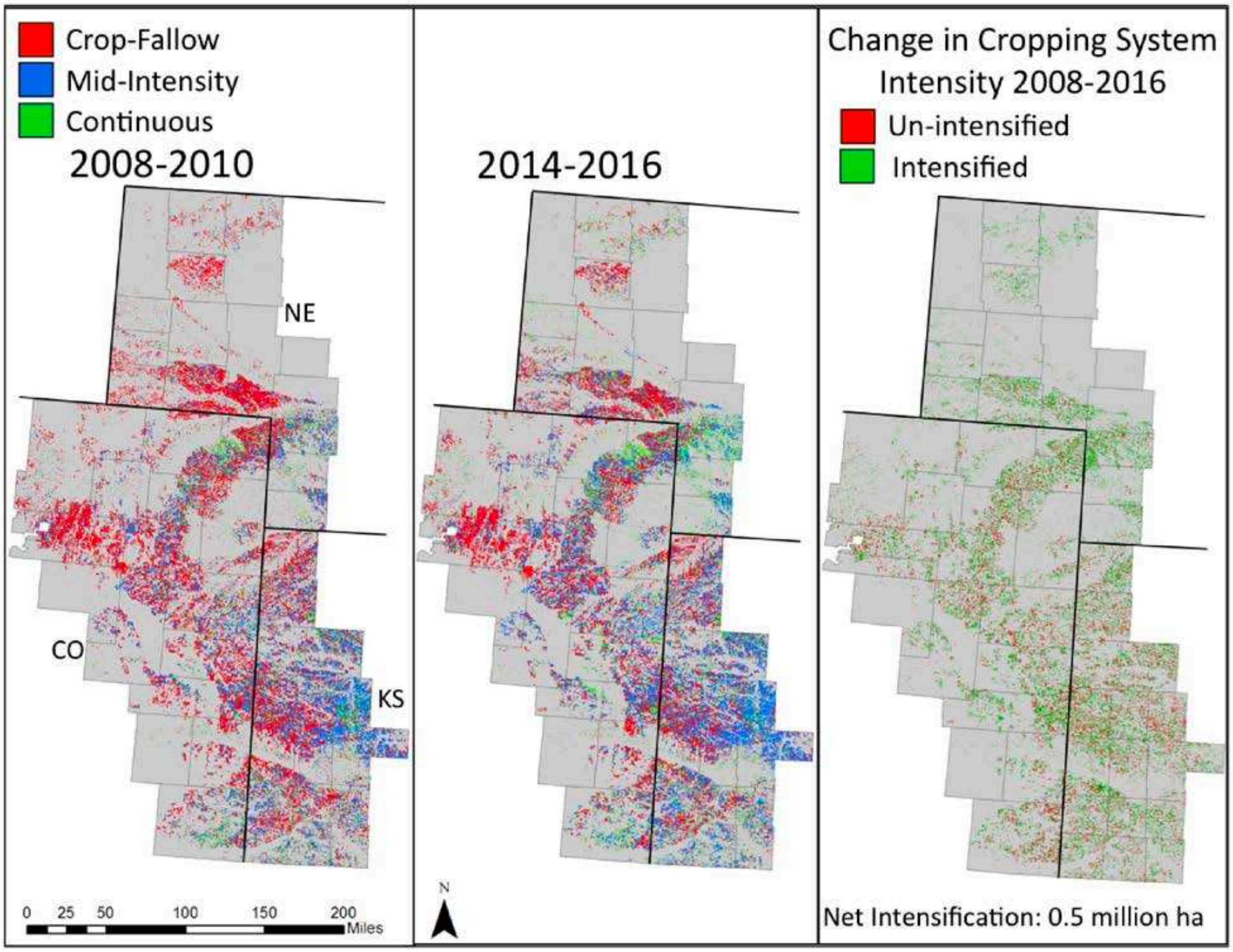

Figure 5.2. Maps of dryland cropping system intensity derived from 3-year rotation windows starting in 2008 (left panel) and 2014 (middle panel), and the change in cropping system intensity between the two rotation windows (right panel) in the High Plains region of Colorado, Kansas, and Nebraska.

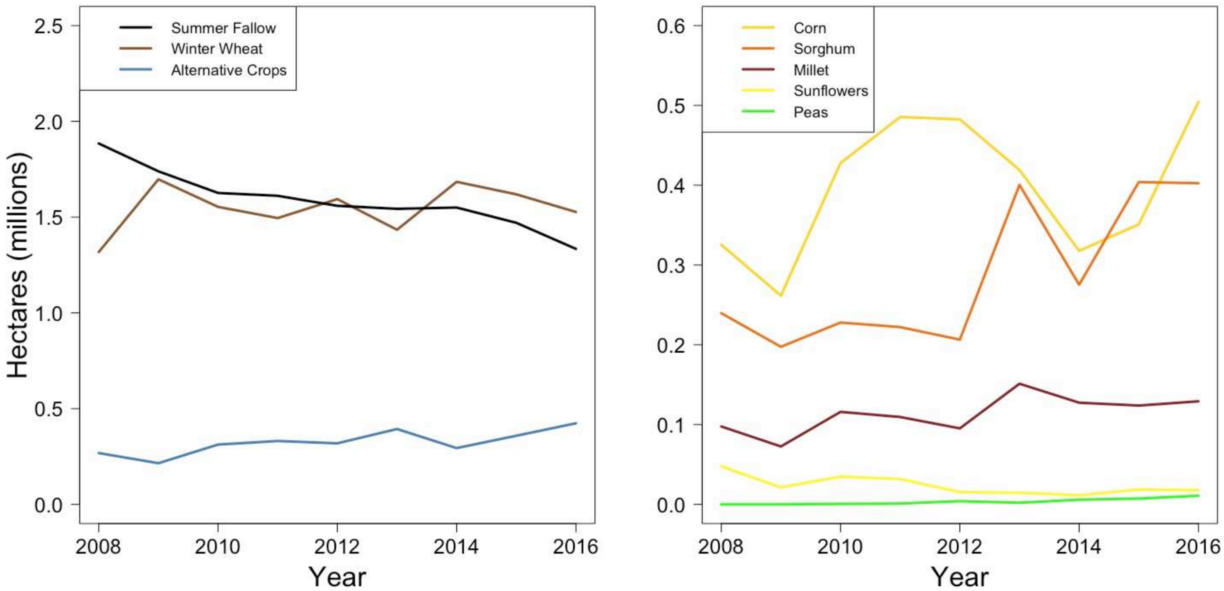

Figure 5.3. Dryland area growing various crops or under summer fallow management from 2008 to 2016 in the High Plains region of Colorado, Kansas, and Nebraska. Data derived from the NASS Croplands Data Layer.

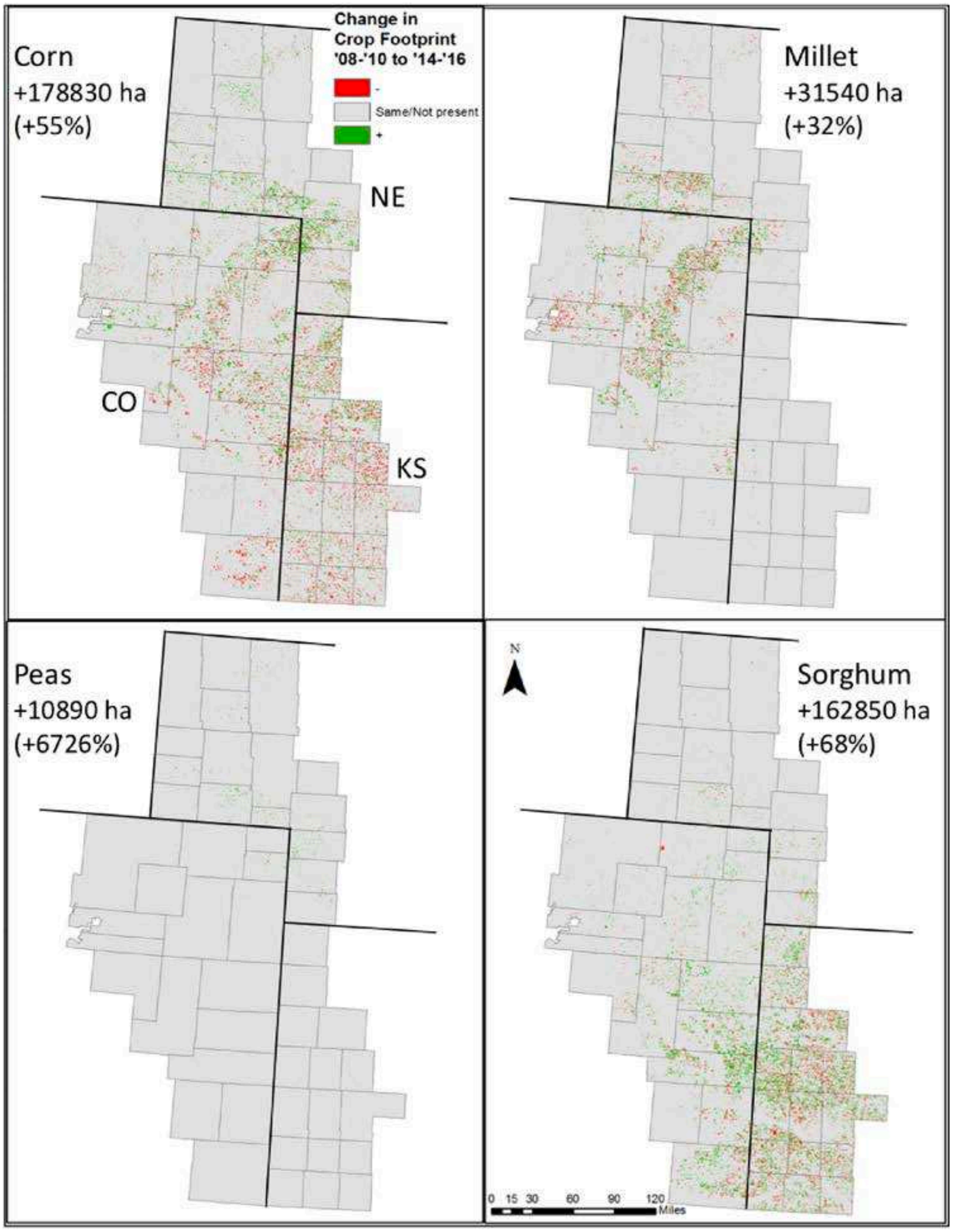

Figure 5.4. Change in footprints of alternative dryland crops from 2008-2010 to 2014-2016 in the High Plains region of Colorado, Kansas, and Nebraska. Data represent net changes in cropland followed by the % change in parentheses.

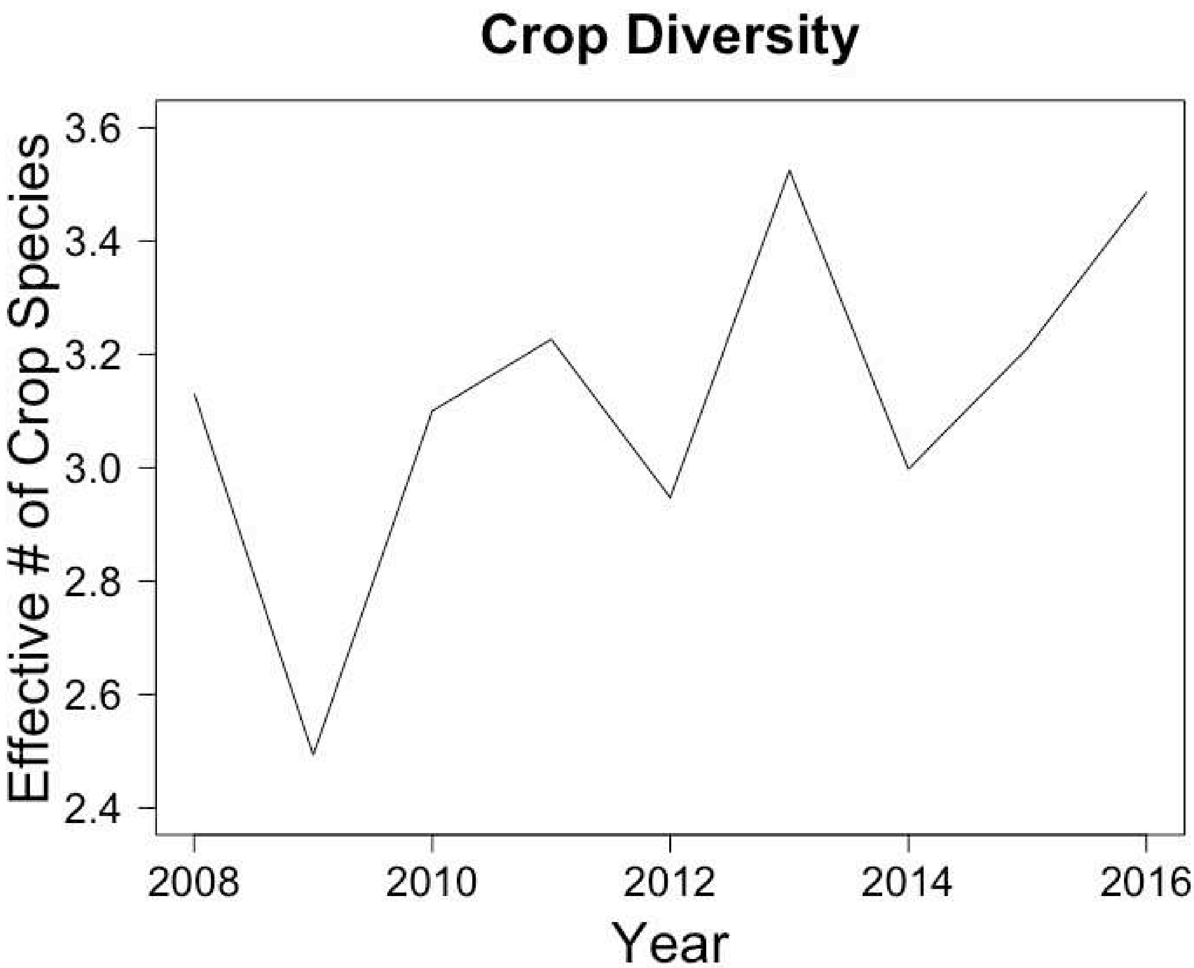

Figure 5.5. Dryland crop diversity, calculated as the Effective Number of Crop Species, from 2008 to 2016 in the High Plains region of Colorado, Kansas, and Nebraska. Data derived from the NASS Croplands Data Layer.

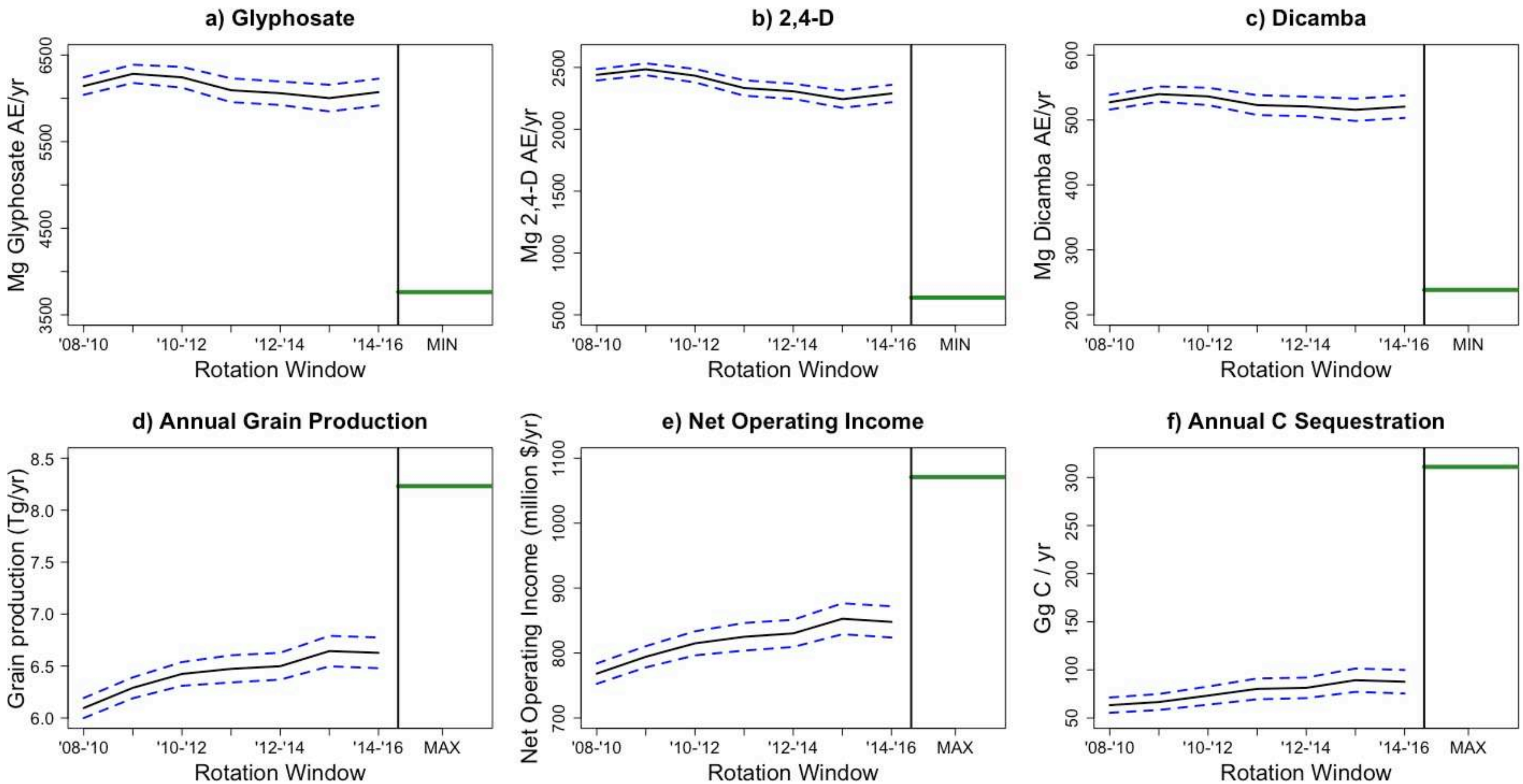

Figure 5.6. Landscape-scale estimates of a) Glyphosate use, b) 2,4-D use, c) Dicamba use, d) Annual Grain Production, e) Annual Net Income, and f) C sequestration in the High Plains region of Colorado, Kansas, and Nebraska for each 3-year rotation window from 2008 to 2016. Data denote means (solid lines) ± standard error (dashed lines), and green solid lines represent the theoretical maximum or minimum level for each metric based on 100% continuous crop adoption.

REFERENCES

Acosta-Martinez, V., Mikha, M.M., Vigil, M.F., 2007. Microbial communities and enzyme activities in soils under alternative crop rotations compared to wheat–fallow for the Central Great Plains. Applied Soil Ecology 37, 41-52.

Acosta-Martinez, V., Zobeck, T., Gill, T., Kennedy, A., 2003. Enzyme activities and microbial community structure in semiarid agricultural soils. Biology and Fertility of Soils 38, 216-227.

Aiken, R., O'Brien, D., Olson, B., Murray, L., 2013. Replacing fallow with continuous cropping reduces crop water productivity of semiarid wheat. Agronomy Journal 105, 199-207.

Anderson, B., 1983. Imagined communities: Reflections on the origin and spread of nationalism. Verso Books.

Anderson, J.R., Dillon, J.L., 1992. Risk analysis in dryland farming systems. Food & Agriculture Org.

Appadurai, A., 2013. The future as cultural fact: Essays on the global condition. London and New York: Verso.

Bardgett, R.D., McAlister, E., 1999. The measurement of soil fungal: bacterial biomass ratios as an indicator of ecosystem self-regulation in temperate meadow grasslands. Biology and Fertility of Soils 29, 282-290.

Basche, A.D., Roesch-McNally, G.E., 2017. Research topics to scale up cover crop use: Reflections from innovative Iowa farmers. Journal of Soil and Water Conservation 72, 59A-63A.

Bates, D., Maechler, M., Bolker, B., Walker, S., 2014. lme4: Linear mixed-effects models using Eigen and S4. R package version 1, 1-23.

Bennett, J.M., Cattle, S., 2013. Adoption of soil health improvement strategies by Australian farmers: I. Attitudes, management and extension implications. The Journal of Agricultural Education and Extension 19, 407-426.

Blesh, J., Wolf, S.A., 2014. Transitions to agroecological farming systems in the Mississippi River Basin: toward an integrated socioecological analysis. Agriculture and Human Values 31, 621-635.

Bot, A., Nachtergaele, F., Young, A., 2000. Land resource potential and constraints at regional and country levels. . Food & Agriculture Org.

Bourdieu, P., 1977. Outline of a Theory of Practice. Cambridge university press.

Bowman, M., Zilberman, D., 2013. Economic factors affecting diversified farming systems. Ecology and Society 18.

Burton, R.J., 2004. Seeing through the ‘good farmer's’ eyes: towards developing an understanding of the social symbolic value of ‘productivist’behaviour. Sociologia Ruralis 44, 195-215.

Carlisle, L., 2016. Factors influencing farmer adoption of soil health practices in the United States: a narrative review. Agroecology and Sustainable Food Systems 40, 583-613.

Carolan, M., 2016. The Sociology of Food and Agriculture. Taylor & Francis.

Carolan, M., 2017. More-than-Active Food Citizens: A Longitudinal and Comparative Study of Alternative and Conventional Eaters. Rural Sociology 82, 197-225.

Carolan, M.S., 2005. Barriers to the adoption of sustainable agriculture on rented land: An examination of contesting social fields. Rural Sociology 70, 387-413.

Carolan, M.S., 2006a. Do you see what I see? Examining the epistemic barriers to sustainable agriculture. Rural Sociology 71, 232-260.

Carolan, M.S., 2006b. Social change and the adoption and adaptation of knowledge claims: Whose truth do you trust in regard to sustainable agriculture? Agriculture and human values 23, 325-339.

Carolan, M.S., Bell, M.M., 2003. In truth we trust: discourse, phenomenology, and the social relations of knowledge in an environmental dispute. Environmental Values, 225-245.

Cochran, V., Danielson, J., Kolberg, R., Miller, P., 2006. Dryland cropping in the Canadian prairies and the US northern Great Plains. AGRONOMY 23, 293.

Cotrufo, M.F., Wallenstein, M.D., Boot, C.M., Denef, K., Paul, E., 2013. The Microbial Efficiency Matrix Stabilization (MEMS) framework integrates plant litter decomposition with soil organic matter stabilization: do labile plant inputs form stable soil organic matter? Global Change Biology 19, 988-995.

De Vries, F.T., Liiri, M.E., Bjørnlund, L., Bowker, M.A., Christensen, S., Setälä, H., Bardgett, R., 2012. Land use alters the resistance and resilience of soil food webs to drought. Nature Climate Change 2, 276-280.

Derksen, D.A., Anderson, R.L., Blackshaw, R.E., Maxwell, B., 2002. Weed dynamics and management strategies for cropping systems in the northern Great Plains. Agronomy Journal 94, 174-185.

Dhuyvetter, K., Thompson, C., Norwood, C., Halvorson, A., 1996. Economics of dryland cropping systems in the Great Plains: A review. Journal of Production Agriculture 9, 216-222.

Di Falco, S., Perrings, C., 2005. Crop biodiversity, risk management and the implications of agricultural assistance. Ecological economics 55, 459-466.

Diaz, R.J., Rosenberg, R., 2008. Spreading dead zones and consequences for marine ecosystems. science 321, 926-929.

Drenovsky, R., Vo, D., Graham, K., Scow, K., 2004. Soil water content and organic carbon availability are major determinants of soil microbial community composition. Microbial Ecology 48, 424-430.

Drinkwater, L.E., Snapp, S., 2007. Nutrients in agroecosystems: rethinking the management paradigm. Advances in Agronomy 92, 163-186.

Drinkwater, L.E., Wagoner, P., Sarrantonio, M., 1998. Legume-based cropping systems have reduced carbon and nitrogen losses. Nature 396, 262-265.

Du, E., de Vries, W., Galloway, J.N., Hu, X., Fang, J., 2014. Changes in wet nitrogen deposition in the United States between 1985 and 2012. Environmental Research Letters 9, 095004.

Elliott, E., 1986. Aggregate structure and carbon, nitrogen, and phosphorus in native and cultivated soils. Soil science society of America journal 50, 627-633.

Engel, R.E., Miller, P.R., McConkey, B.G., Wallander, R., 2017. Soil Organic Carbon Changes to Increasing Cropping Intensity and No-Till in a Semiarid Climate. Soil Science Society of America Journal 81, 404-413.

Farahani, H., Peterson, G., Westfall, D., 1998a. Dryland cropping intensification: A fundamental solution to efficient use of precipitation. Advances in Agronomy (USA).

Farahani, H., Peterson, G., Westfall, D., Sherrod, L., Ahuja, L., 1998b. Soil water storage in dryland cropping systems: the significance of cropping intensification. Soil Science Society of America Journal 62, 984-991.

Flora, C.B., Curtiss, C.F., 2014. Social Sustainability of Cellulosic Energy Cropping Systems. Cellulosic Energy Cropping Systems, 315-333.

Franzluebbers, A., 2002. Water infiltration and soil structure related to organic matter and its stratification with depth. Soil and Tillage Research 66, 197-205.

Frostegård, Å., Bååth, E., 1996. The use of phospholipid fatty acid analysis to estimate bacterial and fungal biomass in soil. Biology and Fertility of Soils 22, 59-65.

Gilbert, N., 1995. Researching social life. Sage, Thousand Oaks, CA.

Glaser, B., Strauss, A., 1967. The discovery of grounded theory: Strategies for qualitative research. Aldine, Chicago, IL.

Grace, J.B., Anderson, T.M., Olff, H., Scheiner, S.M., 2010. On the specification of structural equation models for ecological systems. Ecological Monographs 80, 67-87.

Greb, B., 1979. Reducing drought effects on croplands in the west-central Great Plains.

Hamel, C., Hanson, K., Selles, F., Cruz, A.F., Lemke, R., McConkey, B., Zentner, R., 2006. Seasonal and long-term resource-related variations in soil microbial communities in wheat-based rotations of the Canadian prairie. Soil Biology and Biochemistry 38, 2104-2116.

Hansen, N.C., Allen, B.L., Baumhardt, R.L., Lyon, D.J., 2012. Research achievements and adoption of no-till, dryland cropping in the semi-arid US Great Plains. Field Crops Research 132, 196-203.

Harinikumar, K., Bagyaraj, D., 1988. Effect of crop rotation on native vesicular arbuscular mycorrhizal propagules in soil. Plant and Soil 110, 77-80.

Helms, D., 1990. Conserving the Plains: The soil conservation service in the Great Plains. Agricultural History 64, 58-73.

Hickenlooper, J.W., 2017. Executive Order D 2017-015 Supporting Colorado's Clean Energy Transition. Office of the Governor.

Ingram, J., Mills, J., Dibari, C., Ferrise, R., Ghaley, B.B., Hansen, J.G., Iglesias, A., Karaczun, Z., McVittie, A., Merante, P., 2016. Communicating soil carbon science to farmers: Incorporating credibility, salience and legitimacy. Journal of Rural Studies 48, 115-128.

Jastrow, J., 1996. Soil aggregate formation and the accrual of particulate and mineral-associated organic matter. Soil Biology and Biochemistry 28, 665-676.

Jastrow, J., Miller, R., Lussenhop, J., 1998. Contributions of interacting biological mechanisms to soil aggregate stabilization in restored prairie. Soil Biology and Biochemistry 30, 905-916.

Kaan, D.A., 2002. An economic evaluation of alternative crop rotations compared to wheat-fallow in Northeastern Colorado. Technical bul.(Colorado Agricultural Experiment Station); TB-02-1.

Kallenbach, C.M., Frey, S.D., Grandy, A.S., 2016. Direct evidence for microbial-derived soil organic matter formation and its ecophysiological controls. Nature Communications 7.

Kline, R.B., 2015. Principles and practice of structural equation modeling. Guilford publications.

Knowler, D., Bradshaw, B., 2007. Farmers' adoption of conservation agriculture: A review and synthesis of recent research. Food policy 32, 25-48.

Ko, J., Ahuja, L.R., Saseendran, S., Green, T.R., Ma, L., Nielsen, D.C., Walthall, C.L., 2012. Climate change impacts on dryland cropping systems in the Central Great Plains, USA. Climatic Change 111, 445-472.

Kong, A.Y., Six, J., Bryant, D.C., Denison, R.F., Van Kessel, C., 2005. The relationship between carbon input, aggregation, and soil organic carbon stabilization in sustainable cropping systems. Soil Science Society of America Journal 69, 1078-1085.

Koohafkan, P., Stewart, B.A., 2008. Water and cereals in drylands. Earthscan.

Kuznetsova, A., Brockhoff, P.B., Christensen, R.H.B., 2015. Package 'lmerTest'. R package version 2.

Lal, R., 2004. Carbon sequestration in dryland ecosystems. Environmental Management 33, 528-544.

Lange, M., Habekost, M., Eisenhauer, N., Roscher, C., Bessler, H., Engels, C., Oelmann, Y., Scheu, S., Wilcke, W., Schulze, E.-D., 2014. Biotic and abiotic properties mediating plant diversity effects on soil microbial communities in an experimental grassland. PloS one 9, e96182.

Lenth, R.V., Hervé, M., 2015. lsmeans: Least-squares means. R package version 2, 20-23.

Lin, B.B., 2011. Resilience in agriculture through crop diversification: adaptive management for environmental change. BioScience 61, 183-193.

Luhmann, N., 1979. Trust and power. United Kingdom: John Wiley and Sons, Inc.

Lyon, D., Bruce, S., Vyn, T., Peterson, G., 2004. Achievements and future challenges in conservation tillage. Proceedings of the 4th international crop science congress, pp. 1-19.

Maaz, T., Wulfhorst, J., McCracken, V., Kirkegaard, J., Huggins, D.R., Roth, I., Kaur, H., Pan, W., 2017. Economic, policy, and social trends and challenges of introducing oilseed and pulse crops into dryland wheat cropping systems. Agriculture, Ecosystems & Environment.

Malik, A.A., Chowdhury, S., Schlager, V., Oliver, A., Puissant, J., Vazquez, P.G., Jehmlich, N., von Bergen, M., Griffiths, R.I., Gleixner, G., 2016. Soil fungal: bacterial ratios are linked to altered carbon cycling. Frontiers in microbiology 7.

Mann, L., 1986. Changes in soil carbon storage after cultivation. Soil Science 142, 279-288.

McDaniel, M., Tiemann, L., Grandy, A., 2014. Does agricultural crop diversity enhance soil microbial biomass and organic matter dynamics? A meta-analysis. Ecological Applications 24, 560-570.

Mikha, M.M., Benjamin, J.G., Vigil, M.F., Nielson, D.C., 2010. Cropping intensity impacts on soil aggregation and carbon sequestration in the central Great Plains. Soil Science Society of America Journal 74, 1712-1719.

Morel, K., Léger, F., 2016. A conceptual framework for alternative farmers' strategic choices: the case of French organic market gardening microfarms. Agroecology and Sustainable Food Systems 40, 466-492.

Mulumba, L.N., Lal, R., 2008. Mulching effects on selected soil physical properties. Soil and Tillage Research 98, 106-111.

Nail, E.L., Young, D.L., Schillinger, W.F., 2007. Government subsidies and crop insurance effects on the economics of conservation cropping systems in eastern Washington. Agronomy journal 99, 614-620.

Nelson, R.G., Hellwinckel, C.M., Brandt, C.C., West, T.O., De La Torre Ugarte, D.G., Marland, G., 2009. Energy use and carbon dioxide emissions from cropland production in the United States, 1990–2004. Journal of Environmental Quality 38, 418-425.

Nielsen, D.C., Vigil, M.F., 2010. Precipitation storage efficiency during fallow in wheat-fallow systems. Agronomy Journal 102, 537-543.

Norton, J.B., Mukhwana, E.J., Norton, U., 2012. Loss and recovery of soil organic carbon and nitrogen in a semiarid agroecosystem. Soil Science Society of America Journal 76, 505-514.

Nowak, P.J., 1983. Obstacles to adoption of conservation tillage. Journal of Soil and Water Conservation 38, 162-165.

NRCS, 2013. Unlock the secrets in the soil. USDA.

Padel, S., 2001. Conversion to organic farming: a typical example of the diffusion of an innovation? Sociologia ruralis 41, 40-61.

Pannell, D., 1999. Social and economic challenges in the development of complex farming systems. Agroforestry Systems 45, 395-411.

Paustian, K., Six, J., Elliott, E., Hunt, H., 2000. Management options for reducing CO2 emissions from agricultural soils. Biogeochemistry 48, 147-163.

Peterson, G., Schlegel, A., Tanaka, D., Jones, O., 1996. Precipitation use efficiency as affected by cropping and tillage systems. Journal of Production Agriculture 9, 180-186.

Peterson, G., Westfall, D., Peairs, F., Sherrod, L., Poss, D., Gangloff, W., Larson, K., Thompson, D., Ahuja, L.R., Koch, M., Walker, C., 2001. Sustainable Dryland Agroecosystem Management. Technical bulletin (Colorado Agricultural Experiment Station); TB-01-2.

Peterson, G.A., Westfall, D.G., 2004. Managing precipitation use in sustainable dryland agroecosystems. Annals of Applied Biology 144, 127-138.

Plaza-Bonilla, D., Arrúe, J.L., Cantero-Martínez, C., Fanlo, R., Iglesias, A., Álvaro-Fuentes, J., 2015. Carbon management in dryland agricultural systems. A review. Agronomy for Sustainable Development 35, 1319-1334.

PRISM, 2017. PRISM Climate Data.

Raedeke, A.H., Green, J.J., Hodge, S.S., Valdivia, C., 2003. Farmers, the practice of farming and the future of agroforestry: an application of Bourdieu's concepts of field and habitus. Rural Sociology 68, 64-86.

Rogers, E.M., 2003. Diffusion of innovations. Free Press. New York, 551.

Rosenzweig, S.T., Carson, M.A., Baer, S.G., Blair, J.M., 2016. Changes in soil properties, microbial biomass, and fluxes of C and N in soil following post-agricultural grassland restoration. Applied Soil Ecology 100, 186-194.

Rosseel, Y., 2011. lavaan: an R package for structural equation modeling and more Version 0.4-9 (BETA). Ghent University.

Rousk, J., Frey, S.D., 2015. Revisiting the hypothesis that fungal-to-bacterial dominance characterizes turnover of soil organic matter and nutrients. Ecological Monographs 85, 457-472.

Saltiel, J., Bauder, J.W., Palakovich, S., 1994. Adoption of sustainable agricultural practices: Diffusion, farm structure, and profitability. Rural sociology 59, 333-349.

Sebilo, M., Mayer, B., Nicolardot, B., Pinay, G., Mariotti, A., 2013. Long-term fate of nitrate fertilizer in agricultural soils. Proceedings of the National Academy of Sciences 110, 18185-18189.

Shapiro, B.I., Brorsen, B.W., Doster, D.H., 1992. Adoption of double-cropping soybeans and wheat. Journal of Agricultural and Applied Economics 24, 33-40.

Shaver, T., Peterson, G., Ahuja, L., Westfall, D., Sherrod, L., Dunn, G., 2002. Surface soil physical properties after twelve years of dryland no-till management. Soil Science Society of America Journal 66, 1296-1303.

Shaver, T., Peterson, G., Sherrod, L., 2003. Cropping intensification in dryland systems improves soil physical properties: regression relations. Geoderma 116, 149-164.

Sherrod, L., Dunn, G., Peterson, G., Kolberg, R., 2002. Inorganic carbon analysis by modified pressure-calcimeter method. Soil Science Society of America Journal 66, 299-305.

Sherrod, L., Peterson, G., Westfall, D., Ahuja, L., 2003. Cropping intensity enhances soil organic carbon and nitrogen in a no-till agroecosystem. Soil Science Society of America Journal 67, 1533-1543.

Sherrod, L., Peterson, G., Westfall, D., Ahuja, L., 2005. Soil organic carbon pools after 12 years in no-till dryland agroecosystems. Soil Science Society of America Journal 69, 1600-1608.

Simkin, S.M., Allen, E.B., Bowman, W.D., Clark, C.M., Belnap, J., Brooks, M.L., Cade, B.S., Collins, S.L., Geiser, L.H., Gilliam, F.S., 2016. Conditional vulnerability of plant diversity to atmospheric nitrogen deposition across the United States. Proceedings of the National Academy of Sciences 113, 4086-4091.

Six, J., Elliott, E., Paustian, K., 1999. Aggregate and soil organic matter dynamics under conventional and no-tillage systems. Soil Science Society of America Journal 63, 1350-1358.

Six, J., Frey, S., Thiet, R., Batten, K., 2006. Bacterial and fungal contributions to carbon sequestration in agroecosystems. Soil Science Society of America Journal 70, 555-569.

Six, J., Paustian, K., 2014. Aggregate-associated soil organic matter as an ecosystem property and a measurement tool. Soil Biology and Biochemistry 68, A4-A9.

Smith, E., Young, D.L., 2000. The economic and environmental revolution in semi-arid cropping in North America. Annals of Arid Zone 39, 347-362.

Strauss, C., 2006. The imaginary. Anthropological theory 6, 322-344.

Strickland, M.S., Rousk, J., 2010. Considering fungal: bacterial dominance in soils–methods, controls, and ecosystem implications. Soil Biology and Biochemistry 42, 1385-1395.

Stromberger, M., Shah, Z., Westfall, D., 2007. Soil microbial communities of no-till dryland agroecosystems across an evapotranspiration gradient. Applied Soil Ecology 35, 94-106.

Sutherland, L.A., Burton, R.J., 2011. Good farmers, good neighbours? The role of cultural capital in social capital development in a Scottish farming community. Sociologia Ruralis 51, 238-255.

Taylor, C., 2004. Modern social imaginaries. Duke University Press.

Team, R.C., 2013. R: A language and environment for statistical computing. R foundation for statistical computing, Vienna, Austria.

Thompson, J., 1987. Decline of vesicular-arbuscular mycorrhizae in long fallow disorder of field crops and its expression in phosphorus deficiency of sunflower. Australian Journal of Agricultural Research 38, 847-867.

Tisdall, J.M., Oades, J.M., 1982. Organic matter and water-stable aggregates in soils. Journal of Soil Science 33, 141-163.

UN, 2011. Global drylands: a UN system-wide response. Environment Management Group of the United Nations Geneva. Retrieved from http://www. unccd. int/Lists/SiteDocumentLibrary/Publications/Global_Drylands_Full_Report. pdf.

USGCRP, 2014. Climate change impacts in the United States: U.S. national climate assessment. GlobalChange.gov.

Van Bavel, C., 1950. Mean weight-diameter of soil aggregates as a statistical index of aggregation. Soil Science Society of America Journal 14, 20-23.

Van der Ploeg, J.D., 2006. Agricultural production in crisis. Handbook of rural studies, 258-277.

Wall, D.H., 2012. Soil ecology and ecosystem services. Oxford University Press.

Waring, B.G., Averill, C., Hawkes, C.V., 2013. Differences in fungal and bacterial physiology alter soil carbon and nitrogen cycling: Insights from meta-analysis and theoretical models. Ecology letters 16, 887-894.

Warner, K.D., 2008. Agroecology as participatory science: Emerging alternatives to technology transfer extension practice. Science, Technology, & Human Values 33, 754-777.

Weber, M., 1993. The sociology of religion. Beacon Press.

Wilson, G.A., 2001. From productivism to post-productivism… and back again? Exploring the (un) changed natural and mental landscapes of European agriculture. Transactions of the Institute of British Geographers 26, 77-102.

Wilson, G.W., Rice, C.W., Rillig, M.C., Springer, A., Hartnett, D.C., 2009. Soil aggregation and carbon sequestration are tightly correlated with the abundance of arbuscular mycorrhizal fungi: results from long-term field experiments. Ecology letters 12, 452-461.

CHAPTER 6: SUMMARY

This dissertation documented the soil health and management implications of cropping system intensification across a wide range of environmental and management variability in the High Plains, and scaled up these results to estimate the potential environmental and economic impacts at the landscape level. I hypothesized that SOC would increase with cropping system intensity directly through greater C inputs, and indirectly through greater fungal biomass and aggregation, and that these effects would be robust across a range of environmental and management contexts. Results from comparisons of SOC between cropping systems supported this hypothesis, and suggest that continuous cropping has the potential to provide gains in SOC and improved soil structure that will help offset C emissions and enhance the resilience of dryland agroecosystems. Continuous rotations had 17% and 12% higher SOC concentrations than wheat-fallow in 0-10 cm and 0-20 cm depths, respectively. Aggregate stability in continuous rotations was about twice that in wheat-fallow rotations, and fungal biomass was three times greater in continuous rotations than wheat-fallow, but was not significantly different from mid-intensity rotations. Using structural equation modeling, I observed that continuous cropping, potential evapotranspiration, % clay content, and fungal biomass together explained 50% of the variability in SOC, and that SOC appears to enhance aggregation directly and as mediated through increases in fungal biomass. Overall, the model suggests that cropping system intensity increases SOC both directly, through greater C inputs to soil, and indirectly, by increasing fungal biomass and aggregation. Further research should examine what pools are accumulating the added C in intensified systems, and determine the origin and availability of accrued C. Is the accumulated C the result of microbial byproducts that have tightly bound to mineral surfaces (Cotrufo et al., 2013), or is it protected as particulate organic matter in aggregates? Addressing these questions would help explain the fate of C and generate insights for managing these pools.

I hypothesized that continuous cropping systems would also have an enhanced capacity to provide plant-available nutrients relative to wheat-fallow, enabling continuous farmers to achieve greater crop production with the same or fewer inputs, leading to increased profitability. The results supported this hypothesis. Total N and PMN were 12% and 30% greater in continuous rotations relative to wheat-fallow, respectively, suggesting that internal N cycling was stimulated in continuous systems. Additionally, mid-intensity and continuous rotations had roughly 2 and 3 times more AMF colonization than wheat-fallow, respectively, and AMF colonization was positively correlated with plant P concentration. Others have explained greater winter wheat P concentrations in continuous rotations relative to wheat-fallow through greater P recycling from plant residues (Bowman and Halvorson, 1997), but my findings suggest that AMF are also a critical component of greater P uptake in continuous cropping systems. These cropping intensity effects on nutrient cycling corresponded with 22 and 34 kg/ha less N fertilizer applied per crop in continuous cropping relative to wheat-fallow and mid-intensity rotations, respectively, despite the same or greater total crop production. Additionally, continuous farmers used less than half the total herbicide as wheat-fallow farmers. Continuous rotations net profits were an estimated $47/ha/yr (80%) more than wheat-fallow, and mid-intensity rotations net profits were $42/ha/yr (70%) more than wheat-fallow. These results suggest that cropping system intensification represents an opportunity to achieve more grain production while managing weeds and nutrients with fewer external inputs, leading to greater profitability and better environmental outcomes. Further research should examine which preceding crops foster the most AMF colonization in wheat, or enable the greatest reductions in herbicide and fertilizer use. Additionally, the contribution of the microbial community to higher PMN in continuous systems is unclear. Do higher PMN levels simply suggest greater N accumulation in soil, or is this observation related to a more active microbial community that is better able to cycle N?

In examining the motivations underlying farmers' decisions about whether or not to intensify, I build on Carolan's (2005) application of Bourdieusian social fields to agriculture to find several overlapping fields within Carolan's more general fields of sustainable and conventional agriculture, which are reflected in different degrees of intensification. I identified three main themes that help explain the occurrence of summer fallow on the landscape. First, the emergence of the soil health movement connected a group of farmers to a paradigm that is incongruous with the practice of summer fallow, motivating them to transition to continuous cropping systems. Networks of farmers and experts that ascribe to the soil health philosophy have generated a new truth that summer fallow is no longer a necessity in the High Plains. Second, I identified three distinct strategies for navigating the complex interactions between profitability and risk inherent in the decision about cropping system intensification: risk avoidance, yield maximization, and input reduction. Wheat-fallow farmers avoid yield risk through frequent summer fallow by avoiding crop growth during unpredictable summer climatic conditions, and by maintaining adequate levels of soil moisture. Mid-intensity farmers maximize crop yields by incorporating a summer crop and applying large amounts of fertilizer, while also maintaining summer fallow to maximize subsequent wheat yields. Continuous farmers absorb a 29% wheat yield reduction by eliminating summer fallow, but they make up for this economic loss by reducing inputs and factoring in the economic gain of growing an additional crop. Lastly, crop insurance presents numerous barriers to cropping system intensification that prevent continuous cropping in some counties. I recommend strategies for change to increase cropping system intensification, including conducting soil health research at mainstream institutions, and supporting breeding and marketing efforts to improve the profitability of alternative crops. Future research should examine the effects of the recently introduced whole-farm crop insurance program on the willingness of farmers to adopt sustainable agricultural practices.

To assess the progression of the semi-arid cropping revolution in the High Plains, and to scale up the economic and environmental implications of the field-level analyses, I conducted a spatial analysis using satellite data to examine changes in cropping patterns over time and space at the landscape scale. I use these estimates to scale up my previous field-level research in this region on soil C, herbicide use, yields, and profitability. Over the 9-year study period, the historically dominant wheat-fallow system was replaced by intensified rotations as the dominant systems by land area. This shift coincided with a 0.5 million-hectare decline in summer fallow and a concurrent increase in alternative (non-wheat) crops, resulting in higher crop diversity across the region. I estimate that, from 2008 to 2016, these patterns resulted in a 0.53 Tg (9%) increase in annual grain production, an $80 million (10%) increase in annual net farm operating income, slight reductions in herbicide use, and an 88 Gg C/yr increase in C sequestration that corresponds to greenhouse gas reductions of 0.32 million metric tons of CO_2 equivalents per year ($MMTCO_2e$/yr). I project each of these implications to a scenario of 100% intensification and estimate that, relative to 2016 levels, herbicide use would be reduced by more than half, grain production would increase by 25%, net operating income would increase by $223 million (26%), and greenhouse gases would be reduced by an additional 0.8 $MMTCO_2e$/yr. Further analyses should incorporate estimates of no-till adoption to obtain more accurate estimates of C sequestration associated with the cropping revolution, as no-till is usually a prerequisite for intensification (Peterson et al., 1996) and sequesters substantial amounts of C (Parton et al., 2015).

In summary, these results suggest that cropping system intensification, in conjunction with no-till, represents a win-win management strategy for the farmers and the environment. Greater crop production in continuous rotations helps alleviate the C limitations that have hindered soil health in simplified systems, and enables the regeneration of key soil functions. Greater crop diversity and improved soil health associated with intensification reduces dryland farmers' dependency on external inputs for the management of weeds and nutrients, and allows them to realize greater crop production and profitability. At the same time, soil C sequestration and reduced inputs help mitigate climate change while limiting the loss of harmful chemicals to the environment. Future research should examine the potential for maximizing crop diversity within continuous crop rotations to further enhance these benefits.

REFERENCES

Bowman, R., Halvorson, A., 1997. Crop rotation and tillage effects on phosphorus distribution in the central Great Plains. Soil Science Society of America Journal 61, 1418-1422.

Carolan, M.S., 2005. Barriers to the adoption of sustainable agriculture on rented land: An examination of contesting social fields. Rural Sociology 70, 387-413.

Cotrufo, M.F., Wallenstein, M.D., Boot, C.M., Denef, K., Paul, E., 2013. The Microbial Efficiency Matrix Stabilization (MEMS) framework integrates plant litter decomposition with soil organic matter stabilization: do labile plant inputs form stable soil organic matter? Global Change Biology 19, 988-995.

Parton, W.J., Gutmann, M.P., Merchant, E.R., Hartman, M.D., Adler, P.R., McNeal, F.M., Lutz, S.M., 2015. Measuring and mitigating agricultural greenhouse gas production in the US Great Plains, 1870–2000. Proceedings of the National Academy of Sciences 112, E4681-E4688.

Peterson, G., Schlegel, A., Tanaka, D., Jones, O., 1996. Precipitation use efficiency as affected by cropping and tillage systems. Journal of Production Agriculture 9, 180-186.

APPENDIX 1: SOCIOLOGICAL INTERVIEW QUESTIONNAIRE

1) Tell me about your farming operation.
2) How did you become involved in agriculture?
3) How did the [transition] come about? How did you get [here] from wheat-fallow (or whatever came before)?
4) Was anything particularly daunting or challenging? What were the barriers (or risks) to transitioning to a new type of farming? How did you deal with that? What lessons came out of the experience?
5) What resources were helpful/available to you during the transition?
6) Do you consider the transition complete? Do you approach your farm differently now? How do you think about the work that you do?
7) What role, if any, does crop insurance play in your decisions about what crops to plant? In what other ways does crop insurance affect the way you manage your farm?
8) Does continuous cropping affect the crop insurance premium for wheat? If so, by what percent?
9) Who do you talk to now? When and where do you talk with others about farming? Are there groups that form based on different views about farming? Are there only certain people you can talk with about farming? Do you feel free to say what you want to say? Do some people judge what you do?
10) Do you think it’s important for other growers to intensify and diversify rotations? If yes, are you involved in teaching/mentoring others?

11) Do you open your farm for demonstrations? Are these formal or neighborly visits (or both)? How does it work? Do you visit other farms?

12) What is your history of involvement with federal government programs? What, if any, programs do you currently participate in? Do you have any ideas for policy changes at the level of USDA, NRCS, etc. that would incentivize [transitioning]? What would those incentives look like?

14) How do you feel the transition affects your short-term profitability? Long-term? Do you foresee any challenges in the future that may affect the profitability of your operation? Do you think you will make any changes to deal with those challenges? Have you noticed any other impacts of the transition? (Probe: on farm)

15) What advice would you offer to other farmers considering [a transition]?

16) Is there anything else you would like to say? Are there things I might do differently when I talk with farmers about these issues?

17) Is there anyone else you recommend I talk with for this project?