THE EFFECTS OF FALL GRAZING OF COVER CROPS ON SOIL HEALTH INDICATORS AND SUBSEQUENT CORN YIELD AND QUALITY

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ABSTRACT

In the past 100 years, decoupling of crop-livestock systems has posed detrimental environmental effects through intensive agriculture. Reintegration of these systems can mitigate environmental effects and provide a means of sustainable intensification. In the upper Midwest there is an opportunity after wheat harvest in a wheat-corn rotation to plant quick growing annual cover crops. In this study, based in Central Michigan, we assessed the impact of grazing annual cover crops on soil fertility, soil carbon, soil compaction, weed population dynamics, and corn yield and quality. In this 2x4 factorial strip block design we planted two cover crop mixtures as our main plot: 1) a pure brassica mixture (PURE), and 2) a complex mixture containing brassicas, warm season grasses, cool season grasses, and legumes (MIX). Our sub-plot factor was the date of grazing: October (Oct), November (Nov), December (Dec), and a non-grazed control (NG). Cover crops were strip-grazed with lambs in the fall of 2019, 2020, and 2021. Soil measurements were collected the following spring, and then silage corn was planted. Silage corn was harvested in fall 2020, 2021, and 2022 when the crop reached 65% moisture content, and yield and forage quality were assessed. Grazing annual cover crops had no detectable impact on soil organic matter, permanganate oxidizable carbon, soil C/N ratio and soil penetration resistance. Plots grazed in Oct and Nov contained greater frequency of spring weeds when compared to the NG control which contained less spring weeds and greater frequency of live cover crop. There was also no effect of grazing on corn yield and forage quality. Spatial heterogeneity in site soil conditions resulted in high variance in summer weed biomass, particularly during the 2021 site-year. Overall, grazing annual cover crops with lambs had no negative impacts on soil health or silage corn yield and quality. This system can provide a means of sustainable intensification without reducing the productivity of corn silage rotations.

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LIST OF ABBREVIATIONS

ADF – Acid Detergent Fiber

AIC - Akaike Information Criterion

ANOVA – Analysis of Variance

CC – Cover Crop

CI – Confidence Interval

C/N – Carbon to Nitrogen Ratio

CP – Crude Protein

CSG – Cool Season Grass

DM – Dry Matter

dNDF48 – Digestible neutral detergent fiber after 48 hours

ICLS - Integrated Crop-livestock Systems

IVDMD48 – In-vitro dry matter digestibility after 48 hours.

MIX- Complex Cover Crop Mixture

NDF – Neutral Detergent Fiber

NIRS - Near-infrared Reflectance Spectroscopy

NG – Non-grazed Control Treatment

POXC- Permanganate Oxidizable Carbon

PURE – Pure Brassica Cover Crop Mixture

SE – Standard Error

SOM - Soil Organic Matter

STAB10 – Aggregate Stability after 10 Minutes

Total CN – Total Carbon and Nitrogen

WSC – Water soluble carbohydrates

WSG – Warm Season Grass

CHAPTER 1: REVIEW OF THE LITERATURE

FROM INTEGRATION TO MONOCULTURE AND BACK AGAIN

Agriculture in the United States has been trending towards monoculture crops for the past century. With this increase in monoculture there is higher dependances on external inputs and the lack of diversification in agroecosystems can put farms in precarious positions environmentally and economically during unfavorable years. Integrated crop-livestock systems (ICLS) are a set of management practices that utilize the synergistic relationships between plants and animals to produce more food, fuel, and fiber per unit area of land while maintaining or improving ecological resources. Specifically, ICLS have shown promise in improving nutrient cycling and buffering agroecosystems against climate and weather variations (Faust et al., 2018; Hilimire, 2011; Lemaire et al., 2014; Peterson et al., 2020). There are a multitude of ways livestock can be incorporated into crop production systems, all of which depend on management goals, climate, crop and livestock availability, and economics. Hilimire (2011) summarized three common typologies found in integrated crop-livestock systems. *Spatially separated* systems have animals and crops raised on a single farm, but they do not overlap spatially or temporally. Such systems usually rely on permanent pastures or the collection and application of animal manure on forage or crop fields. *Rotational* systems have crops and animals occupying a single location, but at different times. In these systems, livestock can excrete directly onto crop fields and nutrients are re-incorporated via mechanical action by the animal. *Fully combined* systems have crops and animals occupying a field at the same time. These systems require consideration of forage palatability and proper timing techniques, so that cash crops are not lost to grazing or browsing.

Integrated farms have been a keystone in management since agriculture became commonplace. All components of the farm system had to be unified in an environment devoid of the surefire efficacy of synthetic fertilizers and pesticides. Prior to tractors, livestock provided most of the working power needed to cultivate crops. In addition to draft power, grazing livestock was a reliable method of income diversification when one or more crops failed. This diversification of production was not only ecologically sound but was one of the only ways for a farm to be profitable (Ellenburg, 2000).

In the 20th century in the United States, farms began specializing in growing limited species of crops which resulted in higher yields at the cost of reduced diversity of plant and animal products produced on a single farm (Clark, 2004). This trend continues to this day. Drivers for this increased specialization are a combination of technological innovation and policy (MacDonald & McBride, 2009). In the early 1900s improvements made to tractors incentivized farmers to discontinue animal draft power for field operations. Consequently, farmers no longer needed to grow forage to feed their draft animals and could direct all grain to sales instead of diverting some for draft animal feed. The subsequent specialization of farm equipment (e.g., combines designed for harvesting corn) further drove the adoption of monocultures (Hilimire, 2011; Power and Follett, 1987). After the Great Depression, the Agricultural Adjustment Act of 1933 was passed, part of which was written to bolster rural economies by defining and investing in commodity crops such as milk, corn, cattle, wheat, and several others (Bean & Ezekiel, 1933). This further pushed farmers to specialize in specific commodity crops, and crop monocultures became the predominant farming model in the mid-20th century which continues into the present day (Plourde et al., 2013).

More recently, specialization has been further incentivized by regulating against the incorporation of animals into farming systems under the notion of human health safety. For example, in California farmers are disincentivized to plant non-crop plants to reduce the prevalence of *E. coli* bacteria in leafy greens. This is directly at odds with environmental conservation efforts to plant buffer strips or perennial pasture (Beretti & Stuart, 2008).

The increase in specialization and intensification of livestock production has resulted in a large swath of negative consequences for the environment, human health, and animal health. Ruminant animals are equipped to digest high-fiber forages, converting those nutrients into nutrient dense proteins that are otherwise inaccessible to humans (Cholewińska et al., 2020). When these animals receive a majority of their intake from corn and soy products they are at higher risk of developing digestive complications such as acidosis (Nagaraja & Lechtenberg, 2007). Animal production in large scale feedlot or confinement systems increases the risk of the transmission of zoonotic disease between animals and humans (Koyun et al., 2023). This risk of disease is further exacerbated by heavy use of prophylactic antibiotics which selects for antibiotic resistant strains of pathogens (Wegener, 2003). Large scale animal agriculture can also be a substantial cause of point source contamination for eutrophication as feedlots provide concentrated influxes of macronutrients into freshwater ecosystems where the nutrient requirements required to stimulate algal growth are low (Filip & Middlebrooks, 1976). This is particularly an issue in the Northeast and Upper Midwest United States where eutrophication is negatively impacting freshwater lakes by stimulating algal blooms which can create anoxic conditions for aquatic populations as the blooms use up all the available dissolved oxygen. Further, these blooms can produce secondary toxins that can contaminate drinking water, famously exemplified by the blooms in Lake Erie (US EPA, 2013; Xin et al., 2019).

Opportunities for ICLS

The opportunities for ICLS can be found where the requirements of animals do not directly compete with the productivity of other crops being produced. Using Hillmire's (2011) typologies some common strategies for crop-livestock reintegration can be identified beyond just applying animal manure to a crop field. One common approach is to rotate perennial pasture into a field rotation which can be both a spatially separated and a rotational system if crop and livestock phases occur simultaneously. An opportunity for full integration is grazing cash crop stover. This is advantageous because the cash crop will not be at risk for grazing damage. A riskier, but potentially beneficial approach is to graze animals at the same time as cash crops. This is most viable when the animals forage preference is not in line with the cash crop, one example of this is grazing sheep between grape vines or trees as they will typically select for grasses rather than browse shrubs (Ryschawy et al., 2021).

Grazing annual cover crops is another possible avenue of full integration. This practice can be combined into any crop rotation with a substantial fallow period with the benefit of producing high quality forage in a relatively short amount of time. Like any agricultural practice, the efficacy of grazing cover crops can be assessed on an ecological and economic level. Through this literature review and study, we aim to investigate the following questions: 1) What are the positive or negative impacts of grazing cover crops on soil health and weed ecology? And 2) Will grazing cover crops affect subsequent cash crop yield and quality, and be economically viable?

SOIL HEALTH

The concept of soil as a critical resource in the United States took hold following the Dust Bowl in the mid-20th century. From this event, stark images of failed crops, choking dust-filled winds, and abject poverty clearly demonstrated the ephemeral nature of soil health as a direct result of poor management choices. Since then, several adjectives defining soil functionality have emerged including soil tilth, soil quality, soil productivity, soil resilience, soil security, and soil degradation (Lehmann et al., 2020). Eventually it was recognized that each of these is an important facet of a much larger concept, thus ‘soil health’ was proposed as a catch-all term to capture all aspects of soil functionality (Keith et al., 2016; Lehmann et al., 2020).

Soil health can be defined as having three main pillars: 1) *Functionality* or its ability to sustain ecosystem services, not only in terms of direct functionality for humans, but also in terms of the greater ecosystem. Examples of ecosystem services are erosion prevention, agronomic production, or water quality control. 2) *Vitality* or the interactions of biotic and abiotic factors that coincide with soil functionality, such as the transmission, accessibility, and storage of water, nutrients, carbon, and soil biota. 3) *Sustainability* or the ability for a soil to continue providing its functional outputs indefinitely in a way that is economically, environmentally, and socially viable (Purvis et al., 2019).

Soil health expands beyond ecological and agricultural boundaries to include aesthetic, community building, and cultural heritage values (Janzen et al., 2021). It has also been loosely incorporated into the concept of “One Health” which emphasizes the relationships between human health, ecological health, and animal health. Soils can provide a buffer for a range of diseases and natural disasters that can impact human health. For example, biodiversity in the soil microbiome can compete with soil borne pathogens and preventing soil erosion can inhibit the movement of mobile pathogens into critical waterways (Mendes et al., 2013). Perhaps more tangibly, healthy soils and agroecosystems can transmit and hold more water which reduces the detrimental effects of floods and droughts (Lehmann et al., 2020; Keith et al. 2016).

As it pertains to agriculture, the Natural Resource Conservation Service defines soil health as “The continued capacity of soil to function as a vital living ecosystem that sustains plants, animals, and humans”. This is the most widely used definition among extension specialists and researchers as it succinctly encompasses the pillars of functionality, vitality, and sustainability. From this definition emerged the Nation Resource Conservation Service's Five Principles of Soil Health which are to: 1) minimize soil disturbance, 2) maximize soil cover, 3) maximize biodiversity, 4) maximize the presence of living roots, and 5) incorporate grazing wherever possible (Natural Resource Conservation Service, n.d.). The development, implementation, and monitoring of these principles is supported by a multitude of field and lab techniques that consider both the dynamic and static properties of soil. Static properties are those that are not meaningfully changed by management practice such as parent material and soil texture, however they are often key factors in interpreting soil data. Dynamic properties, on the other hand, are factors that can change under different forms of management under relatively short timescales. Since soil is a complicated matrix of biotic and abiotic interactions, multiple static and dynamic measurements should be taken to capture the relationship between these factors. For example, soil texture data needs to be collected to better interpret information around soil carbon because higher silt content is related to higher organic carbon content (Augustin & Cihacek, 2016). There have been several attempts to create a wholistic assessment of soil health that aim to capture the physical, biological, and chemical status of the soil (Moebius-Clune et al., 2017.). One of the most promoted ways to ensure soil health is by planting cover crops within cash crop rotations as they reinforce four of the five Principles of Soil Health. Grazing cover crops or incorporating perennial pasture can potentially promote all five.

IMPACTS OF GRAZING COVER CROPS IN CROPPING SYSTEMS

There is a large pool of research comparing the soil health and other environmental metrics in continuous row crop rotation to perennial pasture. When reflecting on the five Principles of Soil Health it is clear how perennial forage exemplifies this framework, especially when multi-species forage mixtures are grazed in place. Perennial pastures can promote healthy soils by maintaining cover, rhizodeposition and foliar deposition of carbon, and improving soil structure when compared to annual cropping systems (Augarten et al., 2023; Dhaliwal et al., 2021; Franzluebbers & Stuedemann, 2009). Similar studies have been conducted to determine if similar benefits can be seen in annual forage systems.

Often, the scientific literature uses the term forage and cover crop interchangeably regarding their role in the agroecosystem as they both provide important ecosystem services such as ground cover, pollinator attraction, and the distinction between the two is usually an economic one where forages are technically considered harvested and thus are considered a ‘second crop’ under federal insurance policy (USDA, 2019.). The lines become even more blurred when it comes to pasture as there are minimal exports of nutrients when grazing in place. In short: not all forages are cover crops and not all cover crops are pastures, but all pastures are cover crops.

Cover crops have been partitioned into different functional groups depending on their root/leaf morphology, phenology, and ecosystem niche. Ecologically, the functional groups are categorized as grasses, legumes, brassicas, and forbs, the latter of which is a catch all term for all non-legume and non-brassica forbs. These functional groups can be further categorized into more colloquial (and more marketable) groups depending on their ecosystem services: weed suppresser (plants that grow vigorously and outcompete weeds), nitrogen fixer (legume crops that can fix atmospheric nitrogen into ammonia), erosion controller (plants with fibrous root networks that hold soil in place), nitrogen/nutrient scavenger (cover crops that can explore the soil further and/or effectively uptake unutilized nutrients to be stored in plant biomass), pollinator attractor (plants that have inflorescence that will attract pollinators), and soil builders (cover crops that build organic matter by high above and below biomass production, reduce compaction, or break hardpans). The categorization of these functional groups reflects the beneficial nature of biodiversity in both natural and cultivated ecosystems and multi-species cover crop mixtures can be utilized to employ more than one ecosystem service at a time (Finney & Kaye, 2017).

One of the most promising ways that cover crops and ICLS improve agroecosystems is through the sequestration and transformation of atmospheric carbon, the closing of nutrient cycles, and stimulating microbial communities. Cover crops can sequester between 0.1 – 1.0 Mg ha^{-1} of C per year depending on the type of crop planted, the duration of planting, tillage practice (i.e., no-till vs conventional), and soil clay content (Blanco-Canqui et al., 2015; Franzluebbers & Stuedemann, 2015) No tillage systems have been shown to increase C sequestration when compared to tilled systems, especially when coupled with high biomass residues (Haddaway et al., 2017). Conventional tillage can break apart soil aggregates where occluded C is most protected from microbial decomposition (Sheehy et al., 2015). Crops that have large amounts of below ground biomass often are associated with soil C gains as decaying roots are a major source of soil C, as residues belowground are cycled by microbes, occluded into soil aggregates, and sorbed to mineral surfaces (Dungait et al., 2012; Kong et al., 2005). Longer lasting cover crops, such as perennial pasture, can accumulate more C as there is more time for above and below ground biomass to be deposited (Augarten et al., 2023). Alternatively, one study found that when comparing cover crop and non-cover crop organic and conventional systems the total C deposition was the same regardless of management practice. Carbon deposition was stratified depending on management practice where cover crop treatments hat lower amounts of subsurface deposition but higher amounts of surface deposition when compared to no cover crop treatments (Liang et al., 2022). One long term study out of Nebraska found that grass cover crops increased organic matter concentration by 11% when compared to legume cover crops and no cover crops and increasing organic matter was positively associated with aggregate stability (Blanco-Canqui & Jasa, 2019).

Grazing can increase net primary productivity (NPP) of plants (Brewer & Gaudin, 2020). This can stimulate root growth and root exudate secretion, thus increasing C deposits depending on the environment and the intensity of grazing. Implementing direct grazing onto perennial pasture or cover crops can ‘kick start’ decomposition of plant litter and stimulate the growth of decomposers (Bardgett & Wardle, 2003; Holland et al., 1996). Grazing, including grazing cover crops, can have variable effects on different pools of C. One study shows that grazing cover crops can reduce the amount of particulate organic matter when compared to not grazing (Anderson et al., 2022).

Soil microbes are important for overall agroecosystem health because they often act as mediators in many key biogeochemical reactions (Hinsinger et al., 2009; Jacoby et al., 2017). Specifically, they can transform soil C pools, access nutrients otherwise inaccessible to plants, and fix atmospheric nitrogen into ammonia (Mylona et al., 1995; Schimel & Schaeffer, 2012). Because of this, soil scientists are interested in what populations of microbes are in the soil, what their biomass is, and how much C they have access to. Cover crops can stimulate microbial populations by depositing soil C and establishing symbiotic relationships with both soil bacteria and fungi. A metanalysis by Kim et al. (2020) demonstrates a significant increase in soil microbial activity, diversity, and abundance under cover crop management. Permanganate oxidizable carbon (POXC) is a metric used as a proxy for semi-labile forms of soil C that may be accessible to soil microbes and mineralizable C is a more direct measurement of labile carbon by invoking microbial respiration, both of these methods are relatively sensitive to short term management changes (Culman et al., 2012; Haney et al., 2008). One site managed under ICLS for 30 years in a corn/soy/cover crop rotation found higher labile forms of C in order of grazed pasture > ICLS > conventional (Dhaliwal et al., 2021). Several studies out of Georgia investigate the relationship between tillage, cover crops, and grazing. They show that biologically active fractions of C and N are highly stratified under no-till conditions, and grazing cover crops can increase biologically active C and N under no-till conditions near the surface (Franzluebbers & Stuedemann, 2008, 2015)

Soil Compaction, Structure, and Erosion

Planting cover crops can reduce wind and water erosion when compared to bare soil, but grasses are more effective than broadleaves and legumes due to their fibrous root network, slower decomposition rate, and usually taller height (Blanco-Canqui et al., 2015). One common metric to assess a soil's ability to withstand wind and water erosion is aggregate stability. Aggregates form through a combination of biotic factors (i.e. root/microbe exudates, mechanical binding from root hairs and mycelium, and bioturbation) and abiotic factors (i.e. freeze/thaw cycles, polyvalent metal ions as binding agents, and clay flocculation) (Amézketa, 1999; Lehmann et al., 2017). Soil aggregates are responsible for partitioning soil carbon pools, microbial populations, and mediating water fluxes through the soil profile (Dungait et al., 2012; Puget et al., 2000). Aggregates can be divided into microaggregates (2μm - 250 μm) and macroaggregates (>250 μm), the latter of which is more likely to be transformed from land management practices (Oades & Waters, 1991). There are multiple ways to study aggregate stability depending on the intent of the research question. Mean weight diameter via wet or dry sieving through a nest of different sieve sizes can be done to understand the distribution of the aggregate size classes. A fraction of 1-2-mm dry soil aggregates can be weighed then wetted and oscillated to determine the fraction of water stable aggregates (Amézketa, 1999). A more recent methodology out of the Soil Health Institute measures aggregate stability by photographing >4-mm diameter aggregates initially submerged in water and again after 10 minutes to compare the pre-slaking and post-slaking area via image analysis (Fajardo et al., 2016).

Aggregate stability is positively related to C input and total soil organic carbon (Augustin & Cihacek, 2016). Cover crops can increase the presence of polysaccharides in soils, a common class of compounds that contribute to soil aggregation (Liu et al., 2005). A potting study found a 5% increase in water stable aggregates in most cover crop treatments just before termination and again 30 days after incorporation. This suggests that cover crop residues can improve aggregate stability through decomposing leaf litter and/or roots (Stegarescu et al., 2021). A 3-year field study out of Nebraska investigating the relationship between corn stover removal, cover crop planting, and nutrient management on soil structure found that the removal of stover reduced dry aggregate stability by 34%. It was also found that in the upper 2.5cm of soil the mean weight diameter of aggregates was greater in the plots treated with cover crops. They conclude that while cover crops may help ameliorate aggregate stability, other management practices can overshadow their benefits, in this case the removal of corn stover (Blanco-Canqui et al., 2014). Another study by the same author found that grass cover crops can increase the mean weight diameter of aggregates at 0- to 7.5-cm in depth in fields managed under no-till for 12 years. Grasses increased the percentage of macroaggregates by 45% at 0- to 7.5-cm depth and by 31% from 7.5- to 15-cm depth (Blanco-Canqui & Jasa, 2019).

When cover crops or pasture are grazed the effects on aggregate stability are varied. A study out of the southeast US found that there was no effect of grazing on aggregate stability, but was instead reduced by conventional tillage when compared with no tillage (Franzluebbers &

Stuedemann, 2008). One study out of Argentina compares the effects of grazing and tillage on soil physical properties in a wide range of cropping systems. They determined that tillage has a greater effect on soil physical properties than grazing with no tillage showing greater proportions of macroaggregates, higher populations of fungi, and higher bulk density (Quiroga et al., 2009).

Much of the current research on grazing effects on aggregate stability is emerging out of the semi-arid grasslands of China and Mongolia. A study out of an arid steppe environment in Mongolia measured macro-aggregate stability (10-20mm) using tensile strength under no grazing or continuous grazing and found that the continuously grazed sections had higher tensile strength. However, they attribute this increase in stability to the mechanical compaction of soil into larger clods via hoof action (Wiesmeier et al., 2012). A grassland study out of Inner Mongolia investigating the effects of grazing exclusion and grazing rotation on microaggregate and macroaggregate fractions, and water stable aggregates determined that moderate exclusion increases fine aggregate stability without impacting macroaggregates. However, it should be noted that some exclusion and rotational grazing sites had different initial vegetation and soil characteristics, which the authors acknowledge in their paper (Dong et al., 2022).

Soil compaction is perhaps the most contentious issue when implementing grazing into cropping systems. While there is no specific literature documenting farmer apprehension around soil compaction and ICLS, it is anecdotally understood among extension specialists. Cover crops on their own have been shown to improve soil compaction through two modes of action: 1) deposition of carbon into the soil and 2) the physical reworking of soil structure by cover crop roots. The latter of which is perhaps the most touted benefit of brassica cover crops with large roots that can break apart hardpans (Chen & Weil, 2011). However, the greatest benefit can be achieved when mixing multiple cover crop functional groups together such as tap-rooted brassicas with fibrous-rooted grasses (Snapp et al., 2005).

Implementation of grazing into cover cropping systems is less universally beneficial than cover crops on their own when it comes to compaction. One study found a 7 to 15% increase in soil penetration resistance when compared to no grazing (Faé et al., 2009). Another study out of Illinois found a similar result where integrating livestock resulted in significantly higher penetration resistance when compared to continuous corn at 921 kPa and 655 kPa respectively (Maughan et al., 2009). None of these studies report compaction that impedes plant root growth when averaged across each respective study. However, there are some individual site-years that have detrimentally high compaction. While compaction can occur, most authors attribute freeze-thaw cycles to alleviating compaction between seasons.

Weed Biomass

Cover crops are known to suppress weeds through shading or outcompeting them for nutrients and can suppress weeds for cash crops if cash crops are planted one to three weeks after cover crop termination (Osipitan et al., 2018). There are several factors that play into their effectiveness in weed suppression, but one modeling study in Pennsylvania determined growing degree days, cover crop type, and cover crop biomass are the most important factors for both spring and fall weeds (Baraibar et al., 2018). Selection of cover crops for weed control requires consideration of germination rate and biomass accumulation.

It is hypothesized that incorporating livestock into cover crop systems can suppress weeds through trampling action or actively grazing weed biomass at critical points during their phenology. However, the opposite effect can occur if livestock graze cover crops too early in their growth cycle, thus limiting their ability to outcompete weeds. Tracy & Davis (2009) compared weed biomass and population dynamics between continuous corn, corn stover, and annual cover crops being grazed by cattle and found that the primary weed suppressing effect was from rotation of crops rather than the integration of livestock.

Cash Crop and Animal Yields

Cover crops on their own have shown to either not impact or potentially increase cash crop yields (Daryanto et al., 2018). One study out of Georgia found an increase in cotton lint yield under cover crop grazing with an average return of \$81 ha^{-1} (Schomberg et al., 2014). Another out of Illinois shows an increase in corn yield under grazed winter cover crops when compared to continuous corn, 11.5 Mg ha^{-1} and 10.8 Mg ha^{-1} respectively (Maughan et al., 2009). Sometimes the results are not as clear for example, one study out of Tennessee found that grazing winter wheat reduced overall corn grain yield but increased grain test weight when compared to a no grazing (Curtis & Buttrey, 2018).

In terms of animals gains most studies find that the gain in kg day^{-1} is adequate for their respective production systems (Planisich et al., 2021). However, there may be lost opportunity costs and greater need for management when incorporating cover crops into these systems (Drewnoski et al., 2018).

SUMMARY AND CONCLUSIONS

From an environmental perspective, grazing cover crops can sequester carbon more reliably when coupled with a no-till system. Cover crops on their own deposit carbon into the soil while grazing can enhance carbon deposition. Grazing cover crops can also enhance or impede the weed suppressing effects of cover crops. When they are grazed too early, the cover crops will not be able to compete with early weed species. Cover crops can improve the stability of macroaggregates and incorporating livestock may promote or have no effect on macroaggregate formation. Often, environmental controls will determine the efficacy of a particular management practice including soil type, climate, and previous and current management practices. In most of these studies the change is relatively small or non-existent in the short term, but longer-term studies show consistent increases of most soil health metrics. To maximize the benefits from cover crops, other methods of conservation agriculture must be employed at the same time including reduced tillage and incorporating light to moderate grazing over the long-term.

The effects of grazing cover crops depend heavily on environment, management practice, and cover crop objectives. Purely from an economic standpoint, grazing cover crops will likely have no significant impact on cash crop yield and quality. Leasing cover crop acreage to livestock producers as pasture may help crop growers recover management costs of the cover crops. An alternative approach is individual farmers diversifying their own income by raising their own livestock. However, the current policy and incentive landscape heavily disincentivizes the coupling of livestock and crops.

Overall, it appears that incorporating livestock grazing on cover crops into cropping systems will likely not harm those agroecosystems and potentially provide benefits over the long time. It is clear that more long-term research needs to be done to better understand how these systems perform. An agroecosystem, like any ecosystem, is liable to change in both the short and long term.

CHAPTER 2: EFFECTS OF FALL COVER CROP GRAZING ON CORN SILAGE YIELD, FORAGE QUALITY, AND SOIL HEALTH

INTRODUCTION

Integrated crop livestock systems (ICLS) have been a part of agriculture since its inception and continue to be used around the world (Entz et al., 2005; Hilimire, 2011). However, in the United States there has been a marked decoupling of crop and livestock systems starting with the advent of tractors in the 20th century removing the need for animal draft power from farms, and thus the forages required to feed them. The Agricultural Adjustment Act then created the commodity-based system of agriculture we see to this day (Bean & Ezekiel, 1933; Hilimire, 2011; MacDonald & McBride, 2009). This decoupling combined with the use of artificial fertilizers resulted in an intensification of both crop and livestock systems (MacDonald & McBride, 2009). While intensification has vastly increased the amount of food produced per unit area, it has resulted in detrimental environmental effects from soil degradation to pollution of water ways (Alhameid et al., 2017; Filip & Middlebrooks, 1976; Khan & Mohammad, 2014; Koyun et al., 2023; Tsiafouli et al., 2015). In the interest of producing the same, or more, product per unit land area, there is an incentive to integrate livestock back into cropping systems with one potential method being grazing annual cover crops (Entz et al., 2005; Peterson et al., 2020; Russelle et al., 2007).

Cover crops on their own can reduce the need for fertilizer inputs through the deposition of plant matter, nitrogen fixation from leguminous species, reduction of soil nutrient losses via nutrient scavenging, and improvement of soil structure with fibrous root systems and soil carbon deposition (Blanco-Canqui, 2018; Blanco-Canqui et al., 2015; Liu et al., 2005). Furthermore, cover crops add a layer of protection to the soil by reducing evaporation of water from the soil surface, reducing the impacts of erosion via wind and water, and improving the stability of soil aggregates (Amézketa, 1999; Blanco-Canqui et al., 2014; Blanco-Canqui, 2018; Liu et al., 2005). Healthy soil aggregates provide habitat for key microbes, improve soil aeration, transmit moisture and gases throughout the profile, and occlude soil organic matter (Amézketa, 1999; Oades & Waters, 1991; Puget et al., 2000). Adding grazing to cover crops can potentially stimulate microbial communities and increase microbially derived C and N, especially when used in conjunction with no-till systems (Franzluebbers & Stuedemann, 2009, 2015). However, the effects of grazing cover crops on soil aggregate stability are mixed, with some studies citing an improvement in water stable aggregates and increased macroaggregate frequency (Maughan et al., 2009; Quiroga et al., 2009), while others demonstrated no difference in macroaggregate stability (Franzluebbers & Stuedemann, 2008).

Cover crops have been shown to reduce soil compaction, especially mixtures that contain species with fibrous and tap-rooted systems (Blanco-Canqui et al., 2015; Blanco-Canqui & Jasa, 2019; Liu et al., 2005). Soil compaction from hoof traffic is an often-mentioned concern with grazed cover crops, with soil penetration resistance and bulk density being the standard assessment of compaction. Implementing grazing increased soil bulk density in one trial (Quiroga et al., 2009) but had no impact on bulk density in others (Anderson et al., 2022; Franzluebbers & Stuedemann, 2008; Liebig et al., 2012). Several studies have shown a 9% to 30% increase in soil penetration resistance when incorporating livestock across a range of environments and production systems (De Andrade Bonetti et al., 2019; Dhakal et al., 2022; Maughan et al., 2009).

The impact of cover crop grazing on subsequent crop yields is also of interest considering grazing can be beneficial or damaging to agroecosystems depending on how grazing was implemented (Fan et al., 2021; Kelly et al., 2021; Schuster et al., 2016). Grazing cover crops did not alter subsequent yields of soybeans (*Glycine max L.*) in Missouri (Rushing et al., 2023), or corn (*Zea mays L.*) and soybeans in Nebraska (Anderson et al., 2022), but decreased yields of cotton (*Gossypium hirutum L.*) lint in Georgia (Schomberg et al., 2014). In Georgia, adding livestock to cover cropping systems had no positive or negative effect on corn grain, soybean, and wheat (*Triticum aestivum L.*) yield, but implementing no-till systems increased yield of corn and soybean (Franzluebbers & Stuedemann, 2014). A second study from Missouri showed that soil compaction caused by grazing cattle (*Bos taurus L.*) on cover crops reduced corn yield by 0.4 Mg ha^{-1} but had no effect on soybean yields (Dhakal et al., 2022).

In terms of animal production, grazing cover crops can produce cattle and sheep (*Ovis aries L.*) with economically acceptable market weights and carcass characteristics, acceptable meat sensory quality, and a positive return on financial investment (Franzluebbers & Stuedemann, 2014; Macaluso et al., 2020; Ryschawy et al., 2021). This integration can be achieved on a local or regional scale through addition of owned livestock to row crop operations or through land lease arrangements where row crop producers lease cover crops to livestock owners for grazing (Higgins, 2017), a practice which can help pay for cover crop establishment. Implementation of ICLS in this way can be a cost-effective way of sustainably intensifying production.

Study Objectives

In the upper Midwest, winter wheat offers an ideal opportunity for integrating ICLS into row crops because land is generally fallow from wheat harvest to spring corn planting (Ghimire et al., 2019). Cover crops planted immediately after wheat harvest have enough time to produce grazable forage in fall and early winter and this is ideal for finishing spring-born lambs.

In this study we evaluated the feasibility of silage corn production using two cover crop mixtures planted after wheat and grazed from October through December, with silage corn planted the following spring. Objectives were: 1) evaluate lamb performance and carcass quality when finished for slaughter on cover crops, 2) evaluate impact of cover crop grazing on soil health in the short term, and 3) assess if cover crop grazing reduces the yield and forage quality of silage corn planted subsequent to the cover crop. Objective 1 was addressed by Macaluso (2020). This thesis will focus on the second and third objectives.

MATERIALS AND METHODS

Site Description

The study site was located in East Lansing, Michigan (42.684057 N, -84.478667 W; 270 m elevation). The site was located on glacial till and outwash deposits and the soil series present in descending order of land area are: Marlette fine sandy loam (fine-loamy, mixed, semiactive, mesic Oxaqudalf), Marlette loam (fine-loamy, mixed, semiactive, Oxyaqudalf), Colwood-Brookston loams (fine-loamy, mixed, active, mesic Typic Endoaquolls and fine-loamy, mixed, superactive, mesic Typic Argiaquolls respectively), and Conover loam (fine-loamy, mixed, active, mesic Aquic Hapludalf). The climate is a temperate forest biome with a 30-year average annual high temperature of 14.9 C, an average annual low of 4.8 C, with the typical temperature range falling between –8.0 to 28.0 C, and average annual precipitation of 872 mm per year (U.S. Climate Normals, 2021) Monthly precipitation totals and mean/min/max air temperatures were obtained from a weather station near the study site (Michigan Automated Weather Network, n.d.).

Experimental Design

The study was conducted from 2019 through 2022. The study alternated annually between two adjacent fields (Field 1 and Field 2). Field 1 was grazed in 2019 and 2021 with corn grown in 2020 and 2022, and Field 2 was grazed in 2020 with corn grown in 2021. Crop management for both fields are summarized in Table 2.1. Because of Covid-19 limitations, no field data for corn was collected in 2020 but soil health measurements were taken in the spring following grazing in all three years (2020-2022). Experimental design was a strip block design with three site-years, three replications within each site-year, and a split block treatment arrangement (Fig. 2.1). Main plot factor was two cover crop treatments randomized within blocks and planted in strips measuring approximately 12 by 213 m with long dimensions running east-west. The cover crop mixtures were: 1) PURE, a pure brassica mixture consisting of tillage radish (*Raphanus sativus* L.), turnip (*Brassica rapa subsp. Rapa* L.), and rape (*Brassica napus* L.); and 2) MIX, tillage radish, turnip, and rape, oats (*Avena sativa* L.), cereal rye (*Secale cereale* L.), pearl millet (*Pennisetum glaucum* L.), Japanese millet (*Echinohloa esculenta* L.), field pea (*Pisum sativum* L.), and berseem clover (*Trifolium alexandrinum* L.). The subplot factor was the time of cover crop grazing: 1) grazed in October (OCT); 2) November (NOV); or 3) December (DEC) versus 4) a non-grazed control (NG). Grazing date subplots were not randomized among replications—they began at one end of the field and progressed west to east across the length of the main plots as the grazing paddocks were advanced over time. There were 24 experimental units per site-year (two cover crop levels x four graze date levels x three replications) for a total of 72 experimental plots over the three site-years.

Crop Management

Cover crops were planted in late summer as soon as possible following wheat grain harvest and manure incorporation (Table 2.1). Seeding rates and varieties of the cover crop mixtures are summarized in Table 2.2, and seeding dates and grazing dates are shown in Table 2.1. The grazing phase of the research was described by Macaluso (2020). In brief, grazing treatments were applied by strip-grazing each experimental unit with five lambs for one week with a daily forage dry matter allowance based on 9-10% of lamb body weight, and a targeted consumption of 50% of the available forage. Lambs were excluded from grazed areas after that single week and cover crops were allowed to regrow as weather allowed.

Corn was seeded using a conventional corn planter in June of 2020, May of 2021, and June of 2022 in 76-cm spaced rows with the hybrids ‘1040AMXT’ in 2020 and 2022 and ‘B02V87AMX’ in 2021. During the 2021 and 2022 growing seasons, corn was monitored for black cutworm (*Agrotis ipsilon* [Hufnagel]) and European corn borer (*Ostrinia nubilalis* [Hübner]) using a set of two pheromone traps, one per species of moth, towards the western end of the field. Each trap was placed during the time of peak flight for the respective worm species and moths were counted each week until no individuals were found in the pheromone traps. Corn chlorophyl content was monitored using a chlorophyll meter (SPAD-502Plus, Konica Minolta, Ramsey, NJ) as a proxy for nitrogen status at the VT and R1 stages of growth during the 2022 study year only. Readings were taken on the ear leaf with 30 readings per study plot.

Annual weeds were collected at the time of harvest, hand-sorted by species, and counted. After collection, weed biomass was weighed fresh and then oven dried at 65 C and weighed again for biomass on a dry matter basis. Shortly after corn was sampled for biomass, ten corn stalks per plot were monitored for tar spot (*Rhystisma acerinum L.*) and rust spot (*Puccinia sorhi L.*, and *Puccinia polysora L.*) by visually estimating the percentage of the ear leaf covered by each disease.

Corn Yield and Forage Quality

Corn was harvested for silage when it reached 65-75% moisture content. We determined moisture content by harvesting 10 random corn stalks from the field at the reproductive stage, recording their fresh weight, drying at 65 C for two days and calculating moisture content from the difference. At harvest, fresh corn biomass was collected by cutting 3-m sections of whole plants from two adjacent rows at 15 cm above the soil surface in each plot. Cut plants were weighed fresh using a sling scale. From the fresh biomass sample on each plot, ten randomly selected stalks were chopped through a woodchipper (MTD, Valley City, OH). A sub-sample of the chopped corn was collected by random grab, weighed and oven-dried at 65 C for two days. Overall dry matter yield was calculated by multiplying the fresh biomass by the dry matter concentration from the chopped corn sub-sample. Following harvest, yield stand counts were collected in each plot.

After drying, the chopped whole corn samples were sequentially processed through a 2-mm screen in a Wiley mill (Thomas Scientific, Swedesboro, NJ) and then a 1-mm screen in a cyclone mill (Udy Corporation, Fort Collins, CO). Forage nutritive composition was determined on the ground samples using near-infrared spectrometer (NIRS, Model DS2500, FOSS North America, Silver Springs, MN) using a fresh corn silage equation developed by the NIRS Forage and Feed Consortium (NIRSC, Berea, KY).

Soil Health

Soil samples were collected from the upper 15-cm of soil using a 2.54-cm diameter sampling probe. Ten randomized subsamples were collected per plot, deposited into a bucket, and homogenized by hand. Fresh samples were sieved through a 6-mm sieve and air-dried for storage. A subsample of soil from each plot was separated using a sample splitter and sent to A&L Great Lake Laboratories (Fort Wayne, Indiana) for a standard soil test analysis including pH, soil organic matter (SOM, loss on ignition), P, K, Mg, and Ca concentration (Meilich extractant). Total carbon, nitrogen, and C:N ratio were determined using a Leco TruMac C/N dry combustion analyzer (Wright & Bailey, 2001). Soil compaction was measured at 2.5-cm intervals from 0 to 46 cm each spring using a penetrometer (Spectrum Technologies, Inc., Field Scout SC 900, Aurora, IL). Five subsamples were taken per plot per year and averaged to create a single profile for each experimental unit. Bulk density was collected with a 5-cm-diameter sampling cup. Three subsamples were collected from the upper 5 cm of soil in each plot and placed into one sample bag per plot. Moist soil was weighed immediately following collection and then processed through a 6-mm sieve to extract stones which were weighed separately. A subsample of the sieved soil was collected and placed into an oven at 105 C for 24 h to obtain moisture content. Bulk density in Mg m^{-3} was then calculated correcting for moisture content and stones (Corbin and Robertion, 2019).

Permanganate oxidizable carbon (POXC) was determined using methods outlined by Weil et al. (2003) and Culman et al. (2012) as a proxy for labile carbon in the soil that is accessible to microbes. Two 500-µl aliquots were taken per sample and analyzed using a microplate reader (Bio Tek Synergy H1 Multimode Reader, Agilent, Santa Clara, CA).

Infiltration rate was measured using the metal single-ring infiltrometer method during the third year of the study (Reynolds, 2007). Three infiltration rate subsamples were collected within each plot, avoiding locations that had been visibly influenced by traffic from people, animals, or harvest machinery as those conditions were not representative of the overall field. Aggregate stability samples were collected using a 2.5-cm-diameter soil probe with 10 subsamples per plot. Wet aggregate stability was calculated using a modified version of the Soil Health Institute Image Analysis for Aggregate Stability protocol to account for a sub-optimal sampling strategy (Fajardo et. al, 2016). We calculated wet aggregate stability after 10 minutes of slaking (STAB10) by dividing the initial area by the final area of soil aggregates. A larger STAB10

Index indicates more stable aggregates that resist disintegration in water. Three aggregates were placed into each petri dish using three petri dishes per plot resulting in a total of 9 aggregates per plot (as per Soil Health Institute protocol recommendations). Aggregates were chosen based on having a diameter greater than 4 mm and exhibiting no obvious signs of breakage or shearing from the soil probe. The overall image analysis was performed the same way as the methods outlined by the Soil Health Institute. We made modifications to the protocol by using a ring stand to hold the camera in place, and we developed a for-loop in R to analyze the images in batches for more efficient and consistent output (Appendix).

Relative ground cover was measured in each plot in the spring using a 100-point step transect method distributed over approximately the entire plot area. The presence of cover crops, weeds, plant residue, or bare soil was recorded at the toe point of the shoe for each step. The percentage of each ground cover type was calculated by dividing the number of observations in each ground cover category by the total number of observations. Ground cover transects were done in lieu of biomass sampling because high spatial variability of biomass would have required a prohibitive number of clipped quadrats to provide representative estimates.

Statistical Analysis

Statistical analysis was conducted using a combination of the nlme package (v3.1-152; Pinheiro et al., 2021) in RStudio (RStudio Team, 2020) and PROC MIXED in SAS (SAS Institute, 2015). The cover crop, grazing date, and site-year were considered fixed effects while rep was considered a random effect. ANOVAs were conducted with a type II sum of squares and tested using an F statistic. Levene’s test of unequal variance was done using the CAR package in R and unequal variance models were fitted to variables where Levene’s test values were $P < 0.05$ for a given study factor. Multiple comparisons of corn silage and soil health metrics were done using Tukey's Honest Significant Difference (HSD) and significance was declared at $P < 0.05$. Soil data collected with the penetrometer was treated as a repeated measure along depths. An autoregressive structure with equal variance was selected as the best fitting model that achieved convergence because soil penetration resistance was not independent from depth. Analysis of soil compaction was done by slicing the data at each depth and comparing each factor using Fishers LSD. A stepwise AIC-based regression using the olsrr package in R was applied to aggregate stability with spring cover crop regrowth, SOM, soil C, soil N, C/N ratio, POXC, spring weed ground cover, spring residue cover, sand, silt, and clay added as model parameters. Selection of final model parameters was based on the lowest AIC values and highest adjusted R^2 after each step (Hebbali, 2017).

Weed data was initially analyzed in a similar way to the rest of the parameters; however high variance between plots made it difficult to separate environmental heterogeneity from treatment effects, and weed populations shifted between years. Weed count data was therefore standardized to a Simpsons Diversity index to study the populations present during each study year using the following equation (Travlos et al., 2018):

$$D = 1 - \sum_{i=1}^{S} p_i^2$$

Where D is Simpson's Diversity index, p_i is the proportion of weed species with respect to the total number of individuals, and S is the number of species. This produces a number between 0 and 1 where 0 represents less plant diversity and 1 represents greater plant diversity.

RESULTS

Weather

Monthly total and 30-year normal precipitation are in Fig. 2.2 and average mean/max/min air temperature data are in Fig 2.3. Precipitation was 11%, 13%, 13%, and 36% less than normal in 2019, 2020, 2021, and 2022 respectively (Fig. 2.2). Precipitation during the grazing phase was greatest in October for each study year at 130 mm (64% higher than normal), 70 mm (11% less than normal), and 96 mm (22% higher than normal) for 2019, 2020, and 2021 respectively. In November, precipitation was 57%, 46%, and 58%less than normal for 2019, 2020, and 2021 respectively. Monthly precipitation in December exceeded the 30-year normal by 91% in 2019 and was less than normal in 2020, 2021, and by 5%, and 6%, respectively. During the corn phase in 2021, monthly precipitation from Jun to Aug was close to the 30-year normal while monthly precipitation during the corn phase in 2022 was, on average, 37% below the 30-year normal throughout the growing season. Average yearly temperatures (Fig. 2.3) did not deviate from the 30-year average (9.6 C) where the hottest year was 2021 with an average temperature of 9.9 C and the coolest was 2019 with an average temperature of 8.5 C.

S*oil Fertility and Health Measurements*

Soil texture and fertility conditions for each field are summarized in Table 2.3. Values for Field 1 are the average across 2020 and 2022 while Field 2 is reported for the 2021 site-year. Fields had similar soil texture, but Field 1 had better pH and soil nutrient concentrations for growing corn than Field 2.

Results from ANOVA of all soil carbon and nitrogen measurements are summarized in Table 2.4. The main effects of cover crop, grazing date, and site year for soil organic matter, soil C, soil N, and POXC are recorded in Table 2.5 because there were no observed interaction effects for these variables. Most soil carbon and nitrogen measurements showed unequal variance across site-year and models were refitted accordingly. There were no differences (P > 0.05) among cover crop, grazing date, or site-year factors for SOM, and soil N. Averaged across both cover crop and grazing date factors, POXC was 16% higher in Field 2 in 2021 (772 mg kg^{-1}) when compared to Field 1 in 2020 (619 mg kg^{-1}) and 2022 (689 mg kg^{-1}) (P = 0.04). C/N ratio (P = 0.01) was 9% greater in Field 2 (2021) than the average of both site-years in Field 1 (2020 and 2022). There was a grazing date by site-year interaction observed for soil C/N ratio (Table 2.4, 2.6). Soil C/N ratio was greater in plots that were grazed in December in 2021 (12.0) compared to all other plots and site years except NG and plots grazed in November in 2021, and NG plots and plots that were grazed in October in 2020 (P = 0.01). There was a significant cover crop by grazing date interaction for soil C (P = 0.05) and soil C/N ratio (P = 0.01), however no differences were detected when conducting mean separations and contrasts (P > 0.05).

Aggregate stability (STAB10 Index) had a three-way interaction of cover crop, grazing date, and site year (P = 0.03) (Table 2.4, Fig. 2.4). Non-grazed plots and plots grazed in November planted with both MIX and PURE in 2021, and PURE plots grazed in October in 2021, had a greater aggregate stability (ranging from 0.73 to 0.76) when compared to MIX plots grazed in December in 2020 and 2022, and NG plots in 2022 (ranging from 0.46 to 0.49). Despite the interaction, STAB10 index was numerically greater for PURE than MIX for all comparisons except Dec 2021 and cover crop main effects means across grazing date and site year were 0.66 and 0.61 for PURE and MIX respectively (P = 0.04). Since Field 2 had a greater C/N ratio and POXC than Field 1 (in addition to the fact that all other measurements pertaining to soil C and N trended higher in Field 2), we aimed to better understand what variables had the greatest impact on soil aggregate stability. A stepwise AIC-based regression was applied to aggregate stability where C/N ratio (AIC = -31, Adj R^2 = 0.81) showed the greatest impact on aggregate stability and the addition of spring weeds improved the model slightly (AIC = -32, Adj R^2 = 0.85).

We used soil infiltration rate as a baseline metric for soil hydrologic function while gravimetric soil moisture content was obtained as a parameter for bulk density. Soil infiltration rate was only measured in the spring of 2022 and there was no difference ($P > 0.05$) for any of our treatments (data not shown). On average, it took 364 seconds for 2.5 cm of water to infiltrate; however, infiltration rate was highly variable across the entire field. Gravimetric soil moisture content, collected simultaneously with bulk density, did not differ across treatments ($P > 0.05$), and ranged from 15 to 25% (data not shown).

We utilized both bulk density and soil penetration resistance as measurements for soil compaction through grazing. There was no difference ($P > 0.05$) in spring soil bulk density regardless of fall cover crop and grazing date treatments, with a grand mean of 1.37 g/cm^3 across all treatments (data not shown). ANOVA results for soil penetration resistance are shown in Table 2.7. Cover crop and grazing date treatments did not affect penetration resistance (PR) ($P >$ 0.05) (Fig. 2.5-A, 2.5-B). While grazing date did not significantly impact soil penetration resistance, there is a numerical trend where plots that were grazed had slightly elevated penetration resistance when compared to the NG control from 0-12 cm (Fig. 2.5-B). There was a significant site-year by depth interaction (Table 2.7, Fig. 2.5-C). Penetration resistance readings taken in the spring of 2020 were greater than 2021 and 2022 across all depths with the average penetration resistance of 1844 kPa in 2020, 938 kPa in 2021, and 956 kPa in 2022 ($P = 0.001$).

Spring Ground Cover, Weed Diversity, and Botanical Composition

ANOVA results for spring ground cover composition and spring weed diversity were complex with frequent interactions among factors (Table 2.8). Total spring ground cover (Fig. 2.6) was less for PURE than MIX for all grazing dates but did not differ between cover crops for the ungrazed NG control (Fig 2.6-B, cover crop by grazing date interaction, P = 0.006). The PURE cover crop treatment had less total spring ground cover than MIX in 2020 and 2022, but not in 2021 (Fig. 2.6-A, cover crop by site-year interaction, P = 0.,003).

The impact of treatments on live cover crop regrowth (i.e. ground cover) in spring (Fig 2.7) was also complex. The cover crop by grazing date interaction (P = 0.006) is shown in Fig. 2.7-A. Within grazing dates, there were never differences in cover crop regrowth between PURE and MIX but the numerical rank of PURE and MIX was inconsistent within grazing date treatments. Within the PURE treatment, ungrazed plots had greater spring regrowth than all grazed PURE plots, but ungrazed MIX only had greater spring regrowth than Nov-grazed MIX plots. The cover crop by site-year interaction (P =0.03, Fig. 2.7-B) indicated that both cover crop treatments consistently had more regrowth in 2020 and 2021 than in 2022, but with numerical changes in rank between them each year. The grazing date by site-year interaction (P = 0.001, Fig. 2.7-C) also indicated consistently less spring regrowth in 2022 than in previous years. Within site-years, non-grazed plots had more spring regrowth than plots grazed in Nov in 2020, more than plots grazed in Dec in 2021, and did not differ from grazed plots in 2022.

The proportion of plant residue had significant cover crop by site-year and grazing date by site-year interactions (P = 0.003 and 0.004 respectively) (Table 2.8). Spring residue cover did not differ between cover crop treatments in 2020 but residue was greater for MIX than PURE in 2021 and 2022 (Fig. 2.8-A). Spring residue cover was greater for MIX in 2022 than for any other cover crop-site-year combination. Grazing date effects on spring residue cover also differed across years (grazing date by site-year interaction, P = 0.004), such that residue cover did not differ among grazing dates in 2020, was greatest for plots grazed in December in 2021, and least for plots grazed in October in 2022.

The main effects of grazing date and site-year influenced the proportion of weeds in spring ground cover (P = 0.01 for both) (Table 2.8) Plots that were never grazed and plots grazed in December had the lowest proportion of spring weeds (7% and 8% respectively) when compared to plots grazed in October and November (19% and 15% respectively; P = 0.01, P = 0.04) (Fig. 2.9-A). Field 1 in 2022 had the highest proportion of weeds (21%) when compared to 2020 (Field 1) and 2021 (Field 2) (9 and 7% respectively; P = 0.02, P = 0.01) (Fig. 2.9-B).

Simpsons Diversity Index for spring weeds had a 3-way interaction of year, cover crop, and grazing date (P = 0.04, Table 2.8), however when multiple comparisons and contrasts were run there was no difference between any treatment. Therefore, we are interpreting the main effect along grazing date (P = 0.004). The greatest weed diversities were observed in plots grazed in October (0.60; P = 0.01) and November (0.57, P = 0.04) when compared to plots grazed in December (0.40).

Corn Growth: Yield, Pests, and Summer Weeds

ANOVAs for corn yield, height, ear percentage, tar spot, rust spot, weed biomass, and Simpsons Diversity Index of weeds at harvest are reported in Table 2.9 and means for these variables are summarized in Table 2.10. Corn biomass yield on a dry matter basis was not different between treatments, and it ranged from 14.79 Mg ha^{-1} in PURE plots that were not grazed to 17.74 Mg ha^{-1} in PURE plots grazed in October. There was also no difference observed in corn stalk height across cover crop, grazing date, and site-year. There was a three-way interaction of cover crop, grazing date, and site year for ear weight as a percentage of dry biomass (Table 2.9, *P* = 0.02). PURE plots grazed in December in 2021 had lower ear percentage (30%) when compared to non-grazed PURE plots grazed in 2021 (46%). There was no difference in corn leaf nitrogen at VT and RI growth stage during the 2022 study year (data not shown, average 61.5 and 58.8 SPAD values respectively, $P > 0.05$).

There was no difference in incidence of tar spot or rust across cover crop, grazing date, or site-year factors ($P > 0.05$; Table 2.9, Table 2.10). There was no difference in weed biomass at harvest detected for cover crop, grazing date, or site-year ($P > 0.05$). However, there was a high coefficient of variation for weed biomass across cover crop, grazing date, and site year (164, 131, and 106 respectively). On average, Field 2 in 2021 had 595 kg ha^{-1} of weed biomass while Field 1 in 2022 had 12 kg ha^{-1} (Table 2.9, Table 2.10). There was a significant difference in Simpsons Diversity Index ($P = 0.01$) for weeds at harvest with plots grazed in October showing a greater weed diversity index (0.54) when compared to the NG plots (0.31).

Corn Forage Quality

The corn forage quality ANOVA results are summarized in Table 2.11 and main effect means are in Table 2.12. Crude protein, acid detergent fiber, ash, and fat concentration did not differ across any treatment factors ($P > 0.05$). Neutral detergent fiber ($P = 0.03$), and neutral detergent fiber digestibility after 48 hours ($P = 0.02$) were higher in 2022 (443 g kg^{-1}, and 303 g kg^{-1} respectively) when compared to 2021 (389 g kg^{-1}, and 257 g kg^{-1} respectively). Starch had a significant cover crop and grazing date interaction where it was only greater in MIX plots grazed in November (36.76%) when compared to PURE plots grazed in November (30.71%) (Fig 2.10). Water soluble carbohydrates (Table 2.13) had a significant grazing date x site-year interaction ($P = 0.05$) where the least amount of WSC in corn biomass was found in plots grazed in October and December in Field 2 in 2021 (61.8 g kg^{-1} for both) when compared to plots grazed in December in Field 1 in 2022 (81.1 g kg^{-1}) (P = 0.01 for both comparisons).

DISCUSSION

We used this study as an opportunity to investigate the impacts of grazing on annual cover crops on the greater agroecosystem. A comprehensive field approach improved understanding of biotic and abiotic factors to evaluate the potential of cover crop grazing as an integrated-crop livestock system. Herein we discuss the implications of the results and how they fit into the pre-existing literature of grazing annual cover crops.

Soil Health and Fertility

There was no effect of cover crop mixture or grazing date on spring soil penetration resistance. However, we did observe a numerical trend of greater soil penetration resistance in plots that were grazed when compared to the non-grazed control. Soil penetration readings collected in 2020 showed higher penetration resistance throughout the whole soil profile than in 2021 and 2022. COVID restrictions prevented us from obtaining bulk density and soil moisture data in 2020, so we are unable to verify the dry soil hypothesis or use bulk density to verify soil penetration readings. Another impact of COVID restrictions was that we were unable to collect corn yield and quality data in 2020, and thus we could not determine whether there was a compaction effect on silage corn yield and quality for that year. Nevertheless, across all treatment means penetration resistance never exceeded the 2000 kPa threshold that is considered to impede both root elongation and the output of lateral roots (Atwell, 1993; Colombi & Walter, 2016). Further, the fields were tilled 2 to 4 weeks prior to corn planting. Therefore, we expect that the observed numerical trend within grazing treatments for penetration resistance were unlikely to affect corn growth and indeed, we detected no corn yield differences. At approximately 20 - 25 cm depth, compaction was close to the 2000 kPa threshold. This depth was near the typical plow pan layer for chisel plowing and was likely to be an artefact from years of continuous tillage. Our overall soil compaction results were not in line with the 15% greater penetration resistance after grazing winter annual forages found by Fae et al. (2009). Like Fae, we did not find any difference in subsequent crop yield because of compaction. Schomberg et al. (2014) associated their lower cotton yields after cover crop grazing with soil compaction that occurred from grazing during rainy conditions. In regions where the ground freezes in winter, the combination of frost action and tillage is expected to eliminate grazing-induced compaction before corn is planted (Leuther & Schlüter, 2021).

The consistency of in SOM, soil C, soil N, and C/N across cover crop and grazing date treatments is not unexpected given the short duration of this trial, with each field subjected to the crop rotation for only one to two years of the rotation sequence. It is generally accepted that consistent application of a given management practice over the timescale of years to decades is required to measurably change soil C (Alhameid et al., 2017; Crews & Rumsey, 2017; Dhaliwal et al., 2021). For example, Franzluebbers and Stuedemann (2015) detected a reduction in microbially available C and N when combining ICLS with tilled systems only after seven years.

We had hypothesized that POXC was the test most likely to detect short-term changes in soil C pools because it is considered a proxy for microbially available C and particulate organic C. Since POXC represents a labile form of carbon it has been known to be affected by management practice more than other measures of soil carbon (Culman et al., 2012; Wade et al., 2020). However, we were not able to detect any changes in POXC across the cover crop and grazing date factors. Similar studies using POXC and other methods of detecting short term transformations in labile carbon found an increase of microbially available and/or labile carbon within no-till systems combining ICLS, cover cropping alone, and cover cropping combined with ICLS (Dhaliwal et al., 2021; Franzluebbers & Stuedemann, 2015). Changes in labile carbon can be difficult to detect in land-uses that involve disturbance (i.e., tillage), and often any changes detected in agriculture systems come from no-till systems. Some possible reasons for the difficulty in detecting short-term changes include limited study durations, conventional tillage disrupting microbial populations, or application of manure overshadowing any potential treatment effects (Franzluebbers & Stuedemann, 2008, 2015; Haddaway et al., 2017). Our findings also corroborate with recent studies that refute the hypothesis that POXC can reasonably detect truly microbially available forms of carbon. Hurisso et al. (2016) propose that POXC better represents soil carbon pools that represent long-term soil carbon sequestration such as carbon sequestered by reduced soil disturbance. Woodings & Margenot (2023) outright refute POXC as a proxy for most carbon compounds understood as labile. In their assay they find that permanganate solution readily oxidizes lignin and aromatic compounds that are typically categorized as recalcitrant forms of carbon. They propose that assigning permanganate oxidation to “labile” or “recalcitrant” pools of soil carbon can result in misleading interpretations.

Our grazing date factor was confounded with spatial variation of soil type in the fields. This spatial heterogeneity is reflected in the higher C, N, POXC, and C/N ratio observed in Field 2 (2021). The higher C/N ratio and higher clay content found in the November and NG plots in 2021 may simply reflect poor drainage and different litter decomposition dynamics in saturated soils (Gabriel & Kellman, 2014) typical of Colwood-Brookston loams which happened to be the predominant soil type in those plots. Typically, this soil is excellent for crop production (Official Series Description - COLWOOD Series, n.d.) but tends to be poorly drained, and these fields were not equipped with tile drainage thus making them more susceptible to saturation and surface ponding.

Soil aggregate stability is an important factor in soil health due to its role in soil aeration, nutrient cycling, and carbon sequestration (Amézketa, 1999). Furthermore, wet aggregate stability can be used to better understand how resistant a soil is to weathering (Moebius-Clune et al., 2017.). Most literature investigating the relationship between tillage and grazing on aggregate stability shows that tillage practice has a greater positive impact than grazing because reducing soil disturbance tends to improve soil aggregate stability. When compared to grazing, conventional tillage disrupts soil structure more severely (Franzluebbers & Stuedemann, 2008; Sheehy et al., 2015; Young & Ritz, 2000). Cover crops on their own can improve aggregate stability (Anderson et al., 2022; Blanco-Canqui et al., 2014; Liu et al., 2005), but in our study it was not possible to assess the effects of cover cropping in isolation because space limitations precluded inclusion of a no-cover-crop control treatment. The main drivers for promoting aggregate stability are biotic in nature, particularly root exudation and microbial biproducts (Amézketa, 1999; Franzluebbers et al., 2000; Lehmann et al., 2017). Since our study was conducted using conventional tillage, we suspect that mycorrhizal hyphal networks were broken apart, soil microbes were disturbed, and aggregates were mechanically broken apart. We hypothesized that the complex mixture (MIX) with its varied functional groups and root structures would improve soil aggregation over the pure brassica (PURE) mixture, and that grazing would stimulate soil C deposition and therefore result in the production of macroaggregates (Canarini et al., 2019; Holland et al., 1996). Our results were complex, with a three-way interaction among cover crop, grazing date, and site-year factors. It is difficult to assess what impact grazing date had on aggregate stability as there is no unifying trend among the grazing date level between site years except for the fact that plots grazed in December planting with MIX in 2020 and 2022 were consistently lower. One numerical trend that was clear was that the all-brassica PURE cover crop treatment had 8% greater wet aggregate stability than the MIX treatment containing more diverse species. This is the opposite of our hypothesis and is in conflict with literature demonstrating that increased plant species diversity usually improves soil aggregation. Soil aggregation is in-part mediated by soil microbial populations and plant communities, specifically through the release of exoenzymes, and polysaccharides (Amézketa, 1999; Blanco-Canqui & Lal, 2004a; Oades & Waters, 1991). Further, plant litter can also play a role in soil aggregation, namely a higher litter-derived C:N ratio will lead to greater aggregate stability (Blanco-Canqui & Lal, 2004b; Hewins et al., 2017). The sparse research that has been done on brassica effects on aggregate stability show that they are associated with lower aggregate stability when compared to grasses (Stegarescu et al., 2020, 2021). One possible explanation for our results is that our brassica mixtures contained a significant proportion of turnip and radish and the high polysaccharide content found in the tubers of these plants could partially explain why the brassica mixtures produced a greater mean weight diameter (MWD) of soil aggregates in the short term as explained by Abiven et al., (2007). They observe a higher flux of microbial biomass, polysaccharides, and greater soil fungal hyphal lengths in less than 20 days after depositing and slowly wetting brassica material (cauliflower which resulted in a higher short-term aggregate MWD. Abiven et al. propose that the quick decomposition of brassica material and the resulting influx of polysaccharides and microbial biomass are the main drivers in their aggregate MWD measurements. While we did not record the rate of wetting of our plots nor MWD, the assumed quick decomposition of brassica materials in the spring may have played a small role in our aggregate stability results. The site-year effect on the three-way interaction was due in part to the greater aggregate stability observed in Field 2 (2021) when compared to Field 1 (2020 and 2022). One key point to note is that Field 2 also had greater measurements that pertained to soil carbon and nitrogen (POXC, C/N ratio, soil C, and soil N) than Field 1. When we encountered this difference in soil characteristics between fields, we ran a stepwise regression that determined soil C/N ratio explained most of the variability for aggregate stability in our samples (Adj $R^2 = 0.81$) with spring weeds improving the model by a small amount (Adj $R^2 = 0.85$). Based on our three-way interaction and stepwise regression, we conclude that soil heterogeneity was the primary driver for wet aggregate stability between site years while brassica decomposition may have contributed to the greater aggregate stability observed in PURE plots.

Spring Ground Cover

Relationships among treatment factors for spring botanical composition components were complex. In our assessment of the total covered ground in the spring following cover crop grazing (the sum of live cover crops, live weeds, and plant residue) we found that the MIX plots had the greatest proportion of total cover when compared to PURE plots across grazed treatments (Oct, Nov, and Dec). However, there was no difference between cover crop treatments in total cover for the non-grazed control. One possible explanation for this is that brassicas tend to have a higher digestibility especially when compared to mature grasses with high ADF (Maxin et al., 2020). In our plots we noticed lambs give grazing preference to the more palatable forages such as brassica leaves and tubers, and new grass leaves. This selectiveness resulted in a greater proportion of coarse stemmy plant material being left behind in plots planted with MIX and this persisted as residue the following spring. In our site-year by cover crop and site-year by grazing date interaction for total cover there was less live ground cover observed in 2022. The spring of 2022 was below the 30-year normal in rainfall which may have resulted in less live cover crop growth and weeds.

In our analysis of the proportion of live cover-crop, PURE had more live cover crop compared to MIX in the NG control and plots that were grazed in Nov and Dec. When cover crops did survive, they were mostly rye, rape, and turnip. Even though berseem clover is able to overwinter, spring populations were low. While grazing early in fall reduced the overwintering biomass from cover crops compared to non-grazed covers, grazed cover crop still produced live cover crop biomass the following May.

In our analysis of spring residue, we found that MIX left behind the most plant residue when compared to PURE in plots that were grazed later in the season. These cover crops were most effective at maintaining springtime cover when they were never grazed. Plots planted with MIX tended to have more residue possibly because of the presence of grass species that have a higher C/N ratio and decompose slower when compared to brassicas (Jahanzad et al., 2016). Dead cover crop and weed residues can provide some benefits such as reducing water loss through evaporation (Klocke et al., 2009). Furthermore, live cover crops and residue can protect the soil against wind and water erosion by physically trapping soil particles (Blanco-Canqui et al., 2015). In the case of the present study, residue accumulation can be a potential benefit of our multi-species mixture if the desired management outcome is maintaining soil cover. However, large amounts of grass residue may harbor diseases that are detrimentally compatible with corn and other *Poaceae* crops (Stapleton et al., 2010) or make it difficult to plant the following year by inhibiting seed to soil contact.

Spring weeds were different across the main effect of grazing date and site-year. The greatest proportion of weeds were found in plots grazed in October (19%) and the fewest found in the NG control (7%). This difference is likely caused by the removal of the cover crop canopy through grazing earlier in the season thus allowing for winter annual weeds to germinate and compete the following spring (Baraibar et al., 2018; MacLaren et al., 2019). Studies out of Brazil found that weed seed bank density and weeds per meter squared were reduced with a higher grass sward height while another found that increasing the intensity of grazing reduced weed seed bank size (Schuster et al., 2016, 2018). A study out of a semi-arid organic system found that integrating grazing in an organic reduced-till system increased weed biomass (Larson et al., 2021). In our study, the presence of spring weeds are strongly associated with the amount of time a cover crop was given to accumulate biomass in the fall and coincides with the expectation that greater cover crop biomass is most effective at weed suppression (MacLaren et al., 2019). The type of mixture in our study did not impact weed suppression possibly because both accumulated biomass rapidly. In terms of site-year, we observed the greatest proportion of weeds in the spring of 2022 (Field 1) which may be the result of the lower-than-normal rainfall in the spring of that year selecting for the emergence of drought resistant weed populations.

Corn Yield and Quality

Cover crop treatment had no impact on corn yield and quality. This conflicted with our hypothesis that the implementation of a multi-species cover crop mix would improve silage corn yield, corn quality and soil health by providing a wider range of ecosystems services (Blanco-Canqui et al., 2015; Daryanto et al., 2018). Even though there were no differences between treatments detected in our corn yields, there was a high variance in dry matter yield in plots planted in Field 2 in 2021 with the lowest recorded yield being 8.94 Mg ha^{-1} and the highest being 21.30 Mg ha^{-1}. Based on our assessment of soil conditions, individual plot yields, coefficients of variance, and percent ear yield, we determined this high variation to be a result of spatial heterogeneity. Plots placed in the poorly drained loam simultaneously exhibited high soil clay proportion, higher C/N ratio, poor corn yields, and high weed biomass at harvest. This spatial heterogeneity likely also explains the greater percent ear yield and lower corn stalk heights observed in the NG PURE plots planted in 2021. Corn that was stunted in saturated plots did not produce much vegetative biomass when compared to other plots thus resulting in a greater proportion of ear in yield. There were also significant differences between site-years for neutral detergent fiber, and digestible neutral detergent fiber after 48 hours where both were higher in 2022 than in 2021. One possible explanation is that the poor yields observed in some plots in 2021 reduced the overall fiber content when averaged across the field. Our results are similar to those found in the literature where integrating grazing had no effect on subsequent cash crop yield (Curtis & Buttrey, 2018; Maughan et al., 2009) and contradicted those where there were yield losses (Anderson et al., 2022; Schomberg et al., 2014). As for forage quality, our results were slightly above the target ranges for NDF and ADF, and under the average reported values for CP when comparing to a recent 2022 variety trial (Singh, 2022). NDF values were 100 g kg^{-1} over the average for silage corn producers while ADF was about 10 g kg^{-1} over the average. CP values were 10 g kg^{-1} less than the average range (Allen et al., 2015).

Another observation likely influenced by spatial variation in soil type is the high biomass of annual grasses found in the plots with Colwood-Brookston at the time of corn harvest, particularly foxtail millet and barnyard grass (*Setaria italica L.* and *Echinochloa crus-galli L.* respectively). These weeds are resilient and can grow in more marginal conditions such as dry or poorly drained soils and possibly contributed to the high variance in corn dry matter yield observed between individual plots in 2021 (Nadeem et al., 2020; Sung et al., 1987). Since weed populations were different each year, we used Simpsons Diversity Index to quantify the diversity of weeds present ranging from 0 - 1 with 1 indicating a higher diversity. Overall, plots grazed in October had a greater diversity (0.54) while NG plots showed less diversity (0.31). We found that Simpsons Diversity Index values had an inverse relationship with summer weed biomass. This is in agreement with a study investigating the relationship between weed diversity and cereal grain yield where they found that a greater weed diversity reduced the severity of yield losses (Adeux et al., 2019). This relationship may also explain the substantial impacts on individual plot yields we observed in 2021 that contained exclusively annual grasses. Grazing may have had a role in shifting weed species diversity in the several ways. First, grazing may have interrupted the life cycle of summer annual weeds by consuming or trampling plants during their seeding period in the early fall thus reducing recruitment of new plants. Second, grazing removed the canopy cover during the cover crop phase and may allow for more diverse weed species to emerge because early canopy cover is what determines effective weed suppression (Brennan & Smith, 2005; Lawley et al., 2012). We suspect that the greatest effect on summer weed biomass was from spatial heterogeneity with a possible secondary effect of grazing. It is likely that the high weed biomass observed at corn harvest was from a population of plants whose lifecycle evaded the grazing the previous fall. Furthermore, weed management through grazing or mowing may require more than one pass to be effective and each plot in our study was only grazed once (Mainardis et al., 2020; Zimdahl, 2018). Weed biomass and species diversity did not differ among the two cover crop mixture treatments, which indicates that the complexity of our mixtures did not have a meaningful effect on weed suppression. This may have been because both mixtures contained fast-growing brassica species that produced biomass relatively quickly (Macaluso, 2020).

Our results are notable in that there were no detectable benefits to using a more complex cover crop mixture over a simple one other than the total covered ground observed in the spring. Complex mixtures are more expensive than lower diversity cover crop mixture and more difficult to manage (Florence & McGuire, 2020), through our assessment we determine that producers can get substantial benefits even with a simple cover crop mixture. These results contradict the current consensus in the literature that increased cover crop biodiversity and complexity result in better soil health characteristics (Finny & Kaye, 2017; Saleem et al., 2020). However, our complex mixture contained nine cover crop species, some of which performed poorly which may indicate that there are diminishing returns in the number of species added to a mixture. A simplification of the complex cover crop mixture (MIX) by only including a species from each cover crop functional group (legumes, grasses, and other broadleaves) may have mitigated some of the redundancy.

CONCLUSION

Fall grazing of cover crop mixtures did not reduce silage corn yield or negatively impact forage quality. Cover crop mixtures and grazing date did not affect labile soil carbon, SOM, or C/N ratio within the short timeframe of this study. Further, we did not observe any impacts on soil penetration resistance regardless grazing date or cover crop mixture. The numerical trend of elevated compaction within grazed plots is not severe enough to dissuade farmers from adopting this management style. However, longer term research needs to be conducted on grazing cover crops to determine potential accumulation or losses of soil carbon, in particular how labile and recalcitrant carbon is transformed in both the short and long term.

Because grazing animals directly impact and transform biomass, grazing had a large impact on springtime plant populations. Grazing earlier in the fall (Oct or Nov) resulted in greater spring weed ground cover while grazing late (Dec) or not grazing at all left ample cover crop biomass to compete with weed species. We also found that the complexity of cover crop mixture did not play a role in weed suppression. Since both mixtures contained brassicas that vigorously accumulate biomass, they were able to achieve canopy closure at a similar rate.

Grazing cover crops in the fall had little impact on weed biomass at corn harvest but does impact weed species diversity. Grazing cover crops early in the fall increased weed species diversity the following summer during corn growth, while not grazing the cover crop at all reduced summer weed species diversity. More research must be conducted to determine exactly how grazing can impact weed phenology of winter and summer annuals, particularly assessments of soil seed banks within management systems. Another approach may involve the planting of weed mixes of differing functional groups and assessing their responses to grazing. Grazing cover crops also had no major impact on corn yield and quality regardless of the date grazed or cover crop mixture. Most variation in weed biomass and corn yield in our study was attributed to spatial heterogeneity of soil conditions.

As a management tool, grazing cover crops appears to be a feasible way to sustainably intensify operations without jeopardizing corn yields and soil health in the short term. We have also determined that the complexity of the cover crop mixture did not impact any of the corn yield and quality, and soil health metrics which means that farmers do not need to invest in expensive and complicated cover crop mixtures to achieve substantial benefits. These conclusions are drawn based on our findings in the present study and with the companion piece written by Macaluso 2020 that demonstrate grazing cover crops is an effective method of finishing spring-born lambs

TABLES

TABLE 2.1 Wheat, corn, and cover crop planting dates, manure applications, supplemental N, and herbicide applications in two fields in East Lansing, MI from 2019 - 2022

Action	Date	Application Rate
FIELD 1		
Wheat Harvest	Aug 01, 2019	-
Manure Application[a]	Aug 05, 2019	154 N, 66 P, 81 K (kg ha^{-1})
Cover Crop Planting	Aug 17, 2019	-
Grazing Start	Oct 21, 2019	-
Grazing End	Dec 17, 2019	-
Corn Planting	Jun 02, 2020	32000 plants ha^{-1} population
Corn Harvest	Sep 09, 2020	-
Manure Application[a]	Oct 05, 2020	141 N, 12 P, 73 K (kg ha^{-1})
Herbicide	May 13, 2021	1.16 g ha^{-1} a.i. Pyrafulfotole 26.73 g ha^{-1} a.i. Bromoxynil Octanoate 25.74 g ha^{-1} a.i. Bromoxynil Heptanoate
Manure Application[a]	Jul 17, 2021	137 N, 52 P, 76 K
Cover Crop Planting		-
Grazing Start	Oct 08, 2021	-
Grazing End	Dec 02, 2021	-
Manure Application[a]	Jun 13, 2022	116 N, 45 P, 64 K (kg ha^{-1})
Corn Planting	Jun 20, 2022	32000 population
Fertilizer Application	Jun 20, 2022	3.59 N, 10.70 P, 3.59 K
Herbicide	Jul 19, 2022	0.42 kg ha^{-1} a.e. Glyphosate, N-(phsphonomethyl) glycine 56.12 g ha^{-1} a.i. Mesotrione
Fertilizer	Jul 20, 2022	116.43 kg ha^{-1} N
Corn Harvest	Oct 06, 2022	-
FIELD 2		
Wheat Harvest	Jul 20, 2020	-
Manure Application[a]	Jul 27, 2020	155 N, 19 P, 83 K (kg ha^{-1})
Cover Crop Planting	Jul 30, 2020	-
Grazing Start	Oct 12, 2020	-
Grazing End	Dec 09, 2020	-

TABLE 2.1 (cont'd)

Manure Application[a]	Apr 27, 2021	224 N, 27 P, 12 K (kg ha^{-1})
Corn Planting	May 04, 2021	32000 plants ha^{-1} population
Fertilizer	May 04, 2021	3.59 N, 10.70 P, 3.59 K
Herbicide	Jun 11, 2021	9.20 kg ha^{-1} a.e. Glyphosate, N-(phosphonomethyl) glycine 56.12 g ha^{-1} a.i. Mesotrione
Fertilizer	Jun 15, 2021	59.87 kg ha^{-1} Foliar N
Corn Harvest	Sep 14, 2021	
Manure Application[a]	Oct 11, 2021	82 N, 15 P, 73 K (kg ha^{-1})

[a] Manure incorporated within 24-hr of application
a.i. – Active ingredient
a.e. – Acid equivalent

TABLE 2.2 Cover crop seeding rates for all-brassica (PURE) and complex mixtures (MIX) planted in August in 2019, 2020, and 2021 in East Lansing, MI (Modified from Macaluso, 2020)

			Seed Mixture					
		Functional	PURE			MIX		
Species	**Variety**	**Group**	2019	2020	2021	2019	2020	2021
			--------------------------kg/ha------------------------------					
Rape	Winfred	forb	4.4	3.4	3.4	1.3	1.5	1.5
Radish	Tillage	forb	7.8	6.4	6.4	2.4	3.1	3.1
Turnip	Purple Top	forb	4.4	3.4	3.4	1.2	1.4	1.4
Pearl Millet	not stated	WSG [a]	-	-	-	1.1	1.5	1.5
Japanese Millet	not stated	WSG	-	-	-	1.0	1.6	1.6
Berseem Clover	Frosty	legume	-	-	-	2.3	2.7	2.7
Field Pea	4010	legume	-	-	-	11.8	13.5	13.5
Oats	Bob	CSG [b]	-	-	-	15.9	17.7	17.7
Rye	Hazlet	CSG	-	-	-	7.7	8.1	8.1

[a] WSG, warm season grass.
[b] CSG, cool season grass.

TABLE 2.3 Average soil texture, fertility, and pH for Field 1 (averaged across 2020 and 2022 site-years) and Field 2 (averaged across the 2021 site year) for soil collected in the spring following fall grazing in East Lansing, MI

Field	Sand	Silt	Clay	P	K	Mg	Ca	pH
	----------(%)--------			--------(mg kg^{-1})-------				
Field 1	56	34	10	80	149	165	1015	6.15
Field 2	55	36	9	55	91	200	985	5.96

TABLE 2.4 Analysis of variance for soil carbon and nitrogen measurements taken in the spring following cover crop grazing from 2020 – 2022 in East Lansing, MI

Model	**SOM**	**C**	**N**	**C/N Ratio**	**POXC** [a]	**STAB10** [b]
Cover Crop (CC)	NS	NS	NS	NS	NS	0.04
Grazing Date (GD)	NS	NS	NS	NS	NS	NS
CC x GD	NS	0.05	NS	0.01	NS	NS
Site-year (year)	NS	NS	NS	0.01	0.04	0.01
Year x CC	NS	NS	NS	NS	NS	NS
Year x GD	NS	NS	NS	0.01	NS	NS
Year x CC x GD	NS	NS	NS	NS	NS	0.03

[a] NG represents non-grazed control
[b] POXC represents permanganate oxidizable carbon

TABLE 2.5 Soil carbon and nitrogen measurements taken the spring following grazing in East Lansing, MI for the main effects of cover crop mix, grazing date, and site-year

	SOM	C	N	POXC [c]
	-----------($g\ kg^{-1}$) ----------			($mg\ kg^{-1}$)
Cover Crop				
PURE	27.8 a	15.6 a	1.44 a	699 a
MIX	27.6 a	15.7 a	1.46 a	687 a
CV [a]	21	21	15	17
Grazing Date				
Oct	25.4 a	14.5 a	1.38 a	670 a
Nov	29.9 a	16.3 a	1.52 a	718 a
Dec	28.2 a	16.0 a	1.46 a	693 a
NG [b]	27.2 a	15.7 a	1.44 a	692 a
CV	20	20	14	16
Site Year				
Field 1 - 2020	25.9 a	14.3 a	1.30 a	619 b
Field 2 - 2021	32.6 a	18.7 a	1.60 a	772 a
Field 1 - 2022	24.5 a	13.9 a	1.40 a	689 ab
CV	15	13	11	14

Note: Within columns, means followed by the same letter are not considered different Tukey HSD ($P > 0.05$)

[a] CV – Coefficient of variation

[b] NG represents non-grazed control

[c] POXC represents permanganate oxidizable carbon

TABLE 2.6 Interaction matrix for grazing date and site-year levels for soil C/N ratio measured the spring following cover crop grazing in 2020, 2021, and 2022in East Lansing, MI

	Oct	**Nov**	**Dec**	**NG[a]**
	------------------(g kg^{-1})---------------------			
Field 1 - 2020	10.6 abc	10.5 bc	10.5 bc	10.9 abc
Field 2 - 2021	10.7 bc	11.2 abc	12.0 a	11.6 ab
Field 1 - 2022	10.0 c	10.3 bc	10.0 c	10.2 c

Note: Within rows and columns, means followed by the same letter are not considered different Tukey HSD (*P* > 0.05)

TABLE 2.7 Analysis of variance for soil penetration resistance in the spring following fall cover crop grazing in 2020, 2021, and 2022

Model	Penetration Resistance
Grazing date (GD)	NS
Cover Crop (CC)	NS
Depth [a]	0.001
Site-year (SY)	0.01
GD x CC	NS
GD x Depth	NS
CC x Depth	NS
GD x SY	NS
CC x SY	NS
Depth x SY	0.001

Note: P values were considered significant based on an $\alpha < 0.05$

[a] Depth was considered a repeated measure in analysis

TABLE 2.8 Analysis of variance for spring ground cover measured using 100-step transects the spring following cover crop grazing from 2020 to 2022 in East Lansing, MI

Model	Total Cover	Cover Crop	Residue	Weeds	Diversity Index [a]
Cover Crop (CC)	0.001	NS	NS	NS	NS
Grazing Date (GD)	0.03	0.001	NS	0.01	0.004
CC x GD	0.006	0.006	NS	NS	NS
Site-year (year)	NS	0.001	NS	0.01	NS
Year x CC	0.003	0.03	0.003	NS	NS
Year x GD	NS	0.001	0.004	NS	NS
Year x CC x GD	NS	NS	NS	NS	0.04

Note: P values were considered significant based on an $\alpha < 0.05$

[a] Simpsons Diversity Index of weed populations measured in the spring

TABLE 2.9 Corn yield, height, percent ear, disease, and weed measurements at corn harvest in 2021 and 2022 following fall grazing in East Lansing, MI

Model	**Dry Yield**	**Corn Height**	**% Ear** [a]	**Tar**	**Rust**	**Weed Biomass**	**Diversity Index** [b]
Cover Crop (CC)	NS	NS	NS	NS	NS	NS	NS
Grazing Date (GD)	NS	NS	NS	NS	NS	NS	0.01
CC x GD	NS	NS	NS	NS	NS	NS	NS
Site-year (year)	NS	NS	NS	NS	NS	NS	NS
Year x CC	NS	NS	NS	NS	NS	NS	NS
Year x GD	NS	NS	NS	NS	NS	NS	NS
Year x CC x GD	NS	NS	0.02	NS	NS	NS	NS

Note: P values were considered significant based on an $\alpha < 0.05$

[a] Percentage of dry matter yield that is from ear weight

[b] Simpsons diversity index for weeds collected at corn harvest

TABLE 2.10 Main effect of cover crop, grazing date, and site-year for corn yield, height, disease, and weed measurements at corn harvest following fall cover crop grazing in 2021 and 2022 in East Lansing, MI

	Dry Yield	Corn Height	Tar	Rust	Weed Biomass	Diversity Index [a]
	(Mg ha^{-1})	**(cm)**	**(% leaf area)**		**(kg ha^{-1})**	
Cover Crop						
PURE	16.49 a	201 a	1.29 a	5.19 a	312 a	0.44 a
MIX	16.59 a	203 a	1.77 a	5.60 a	294 a	0.42 a
CV [b]	12	10	141	87	164	41
Grazing Date						
Oct	17.37 a	207 a	2.37 a	5.37 a	145 a	0.54 a
Nov	16.25 a	195 a	1.21 a	7.75 a	438 a	0.38 ab
Dec	17.93 a	209 a	1.22 a	5.02 a	127 a	0.47 ab
NG	15.35 a	197 a	1.32 a	3.45 a	503 a	0.31 b
CV	11	9	120	78	131	38
Site-Year						
2021 - Field 2	16.75 a	186 a	0.37 a	7.42 a	595 a	0.49 a
2022 - Field 1	16.34 a	218 a	2.69 a	3.37 a	12 a	0.37 b
CV	11	7	122	58	106	40

Note: Within columns of a given factor, means followed by the same letter are not significantly different according to Tukey's HSD ($P < 0.05$)

[a] Simpsons diversity index for weeds collected at corn harvest

[b] Coefficient of variation

TABLE 2.11 Analysis of variance for corn forage quality measured at corn harvest in 2021 and 2022 following fall grazing

Model	Crude Protein	aNDF[a]	ADF[b]	Starch	WSC	Fat	Ash	IVTDMD 48[c]	NDFD 48[d]
Cover Crop (CC)	NS	NS	NS	NS	NS	NS	NS	NS	NS
Grazing Date (GD)	NS	NS	NS	NS	NS	NS	NS	NS	NS
CC x GD	NS	NS	NS	0.05	NS	NS	NS	NS	NS
Site-year (year)	NS	0.03	NS	NS	NS	NS	NS	NS	0.02
Year x CC	NS	NS	NS	NS	NS	NS	NS	NS	NS
Year x GD	NS	NS	NS	NS	0.04	NS	NS	NS	NS
Year x CC x GD	NS	NS	NS	NS	NS	NS	NS	NS	NS

Note: P values were considered significant based on an $\alpha < 0.05$

[a] aNDF – Amylase neutral detergent fiber

[b] ADF – Acid detergent fiber

[c] IVTDMD48 – In-vitro dry matter digestibility after 48 hours

[d] NDFD48 – Neutral detergent fiber digestibility after 48 hours

[e] WSC – Water soluble carbohydrates

TABLE 2.12 Main effect of cover crop, grazing date, and site-year on corn forage quality measurements taken shorty after corn harvest in 2021 and 2022 following fall cover crop grazing

	Crude Protein	**aNDF[a]**	**ADF[b]**	**Fat**	**Ash**	**IVTDMD 48[c]**	**NDFD 48[d]**
	------------------------------------ ($g\ kg^{-1}$)----------------------------------						
Cover Crop							
PURE	73.1 a	423 a	214 a	29.5 a	30.7 a	870 a	284 a
MIX	73.0 a	409 a	207 a	30.2 a	30.5 a	875 a	276 a
Grazing Date							
Oct	73.8 a	413 a	209 a	30.0 a	30.1 a	872 a	278 a
Nov	71.6 a	417 a	210 a	30.1 a	29.8 a	875 a	282 a
Dec	73.3 a	412 a	209 a	29.6 a	30.8 a	871 a	275 a
NG	73.5 a	421 a	213 a	29.8 a	31.6 a	872 a	283 a
Site-Year							
2021 - Field 2	70.0 a	389 b	202 a	29.9 a	30.5 a	875 a	257 b
2022 - Field 1	76.0 a	443 a	218 a	29.8 a	30.7 a	870 a	303 a

Note: Within columns of a given factor, means followed by the same letter are not significantly different according to Tukey's HSD ($P < 0.05$)

[a] aNDF – Amylase neutral detergent fiber

[b] ADF – Acid detergent fiber

[c] IVTDMD48 – In-vitro dry matter digestibility after 48 hours

[d] NDFD48 – Neutral detergent fiber digestibility after 48 hours

TABLE 2.13 Interaction of site-year and grazing date on water soluble carbohydrates (WSC) for corn harvested in 2021 and 2022

	Oct	**Nov**	**Dec**	**NG**
	-------------------WSC [a] (g kg) ------------			
Field 2 - 2021	61.8 b	68.1 ab	61.8 b	71.3 ab
Field 1 - 2022	74.4 ab	68.0 ab	81.1 a	73.8 ab

Note: Within columns, means followed by the same letter are not significantly different according to Tukey's HSD (P < 0.05)

[a] WSC – Water soluble carbohydrates

[b] NG – Non-grazed control

FIGURES

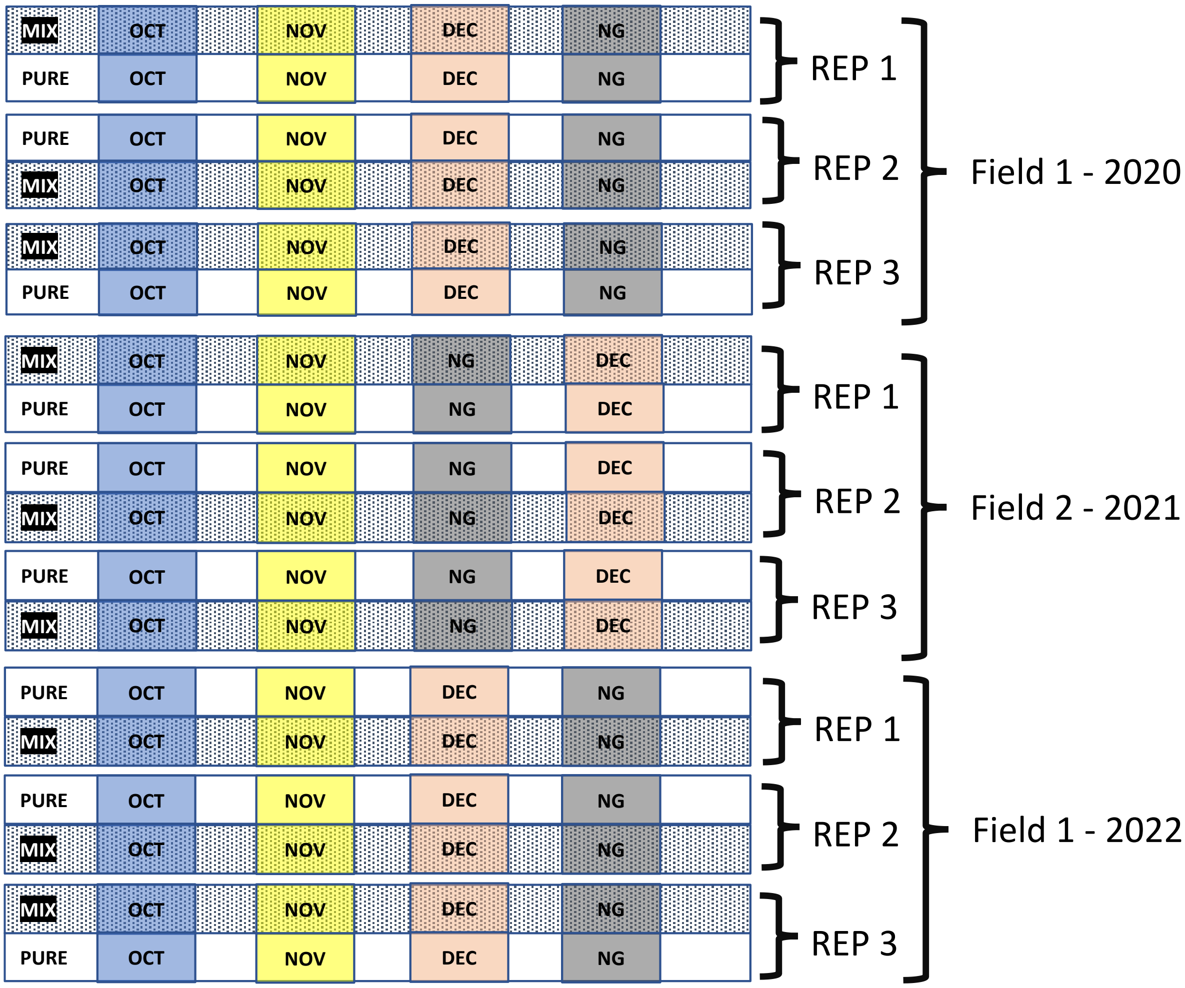

FIGURE 2.1 Experimental design of the fall cover crop grazing study in East Lansing, MI from 2019 to 2022 with three replicates of the strip block design with two levels of crop mixture as the main plot: 1) a pure brassica (PURE), and 2) a complex mixture with multiple functional groups including brassicas, warm season grasses, cool season grasses, and legumes (MIX). The sub-plot is the month the plot was grazed: October, November, December, and a non-grazed control (NG).

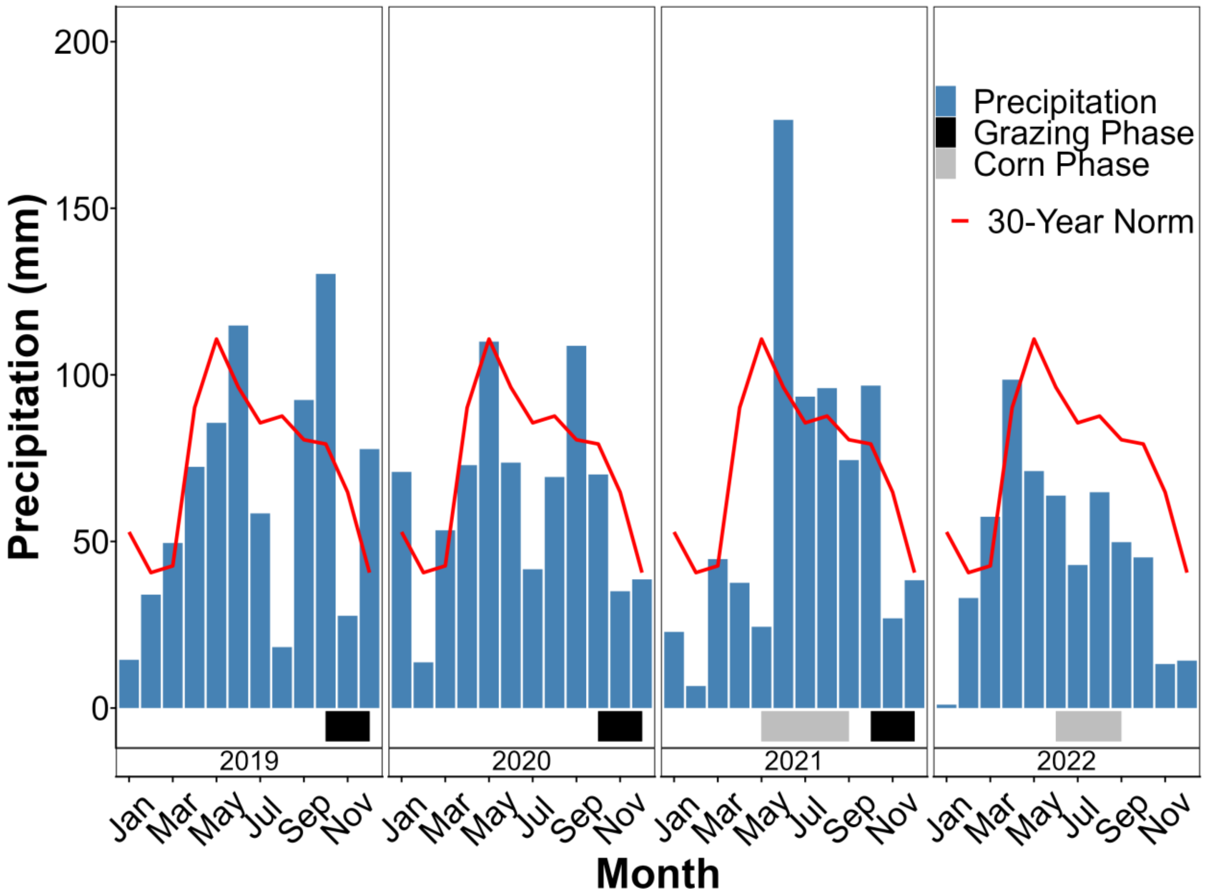

FIGURE 2.2 Local monthly total and 30-year (1991-2020) average precipitation for East Lansing, Michigan from 2019 to 2022. The black and gray rectangles represent the cover crop grazing phases and the corn silage phases of this study, respectively.

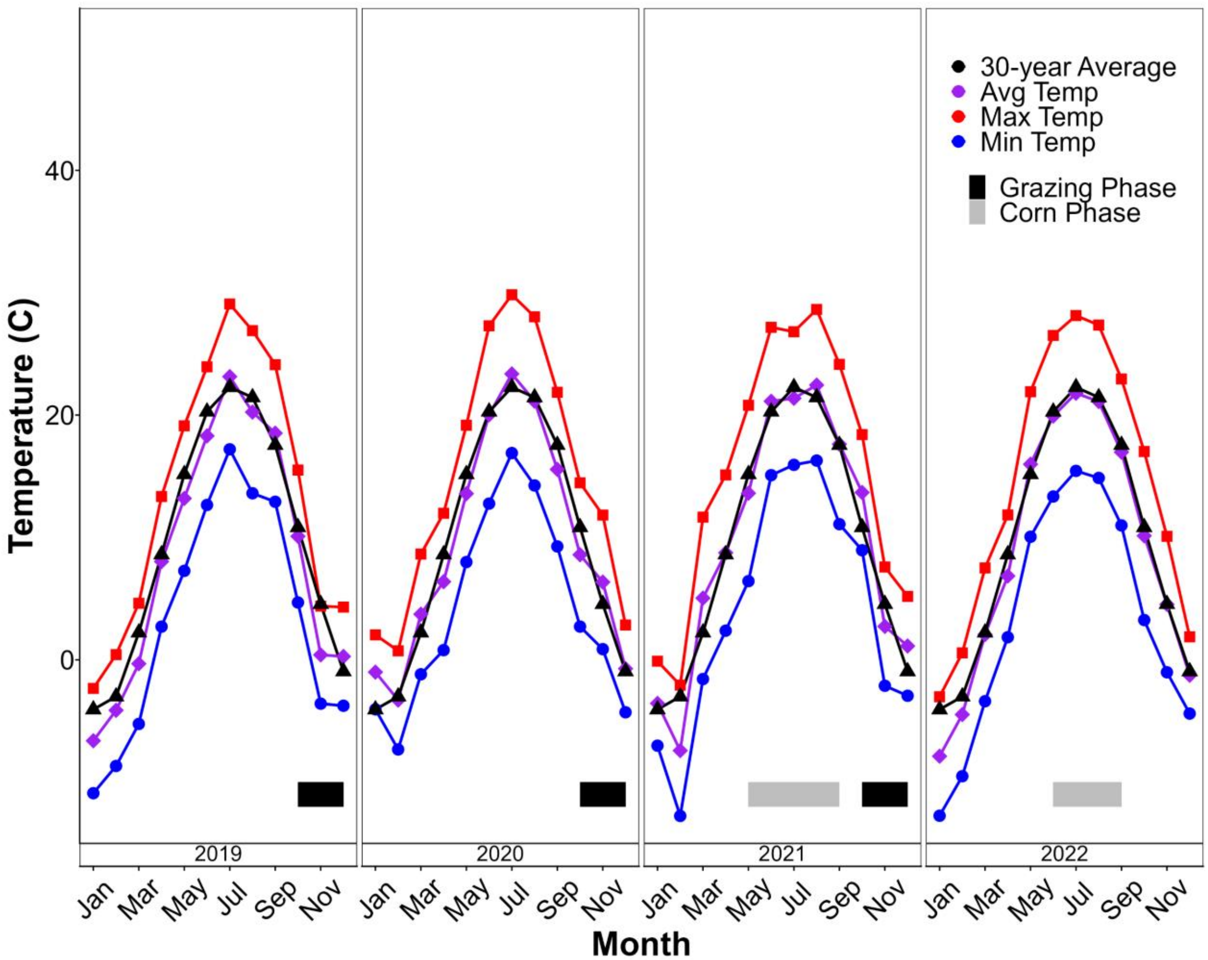

FIGURE 2.3 Monthly minimum, maximum, and average air temperature in East Lansing, Michigan plotted with the 30-year norm (1991-2020) for average air temperature. The black and gray rectangles represent the cover crop grazing phases and the corn silage phases of this study respectively.

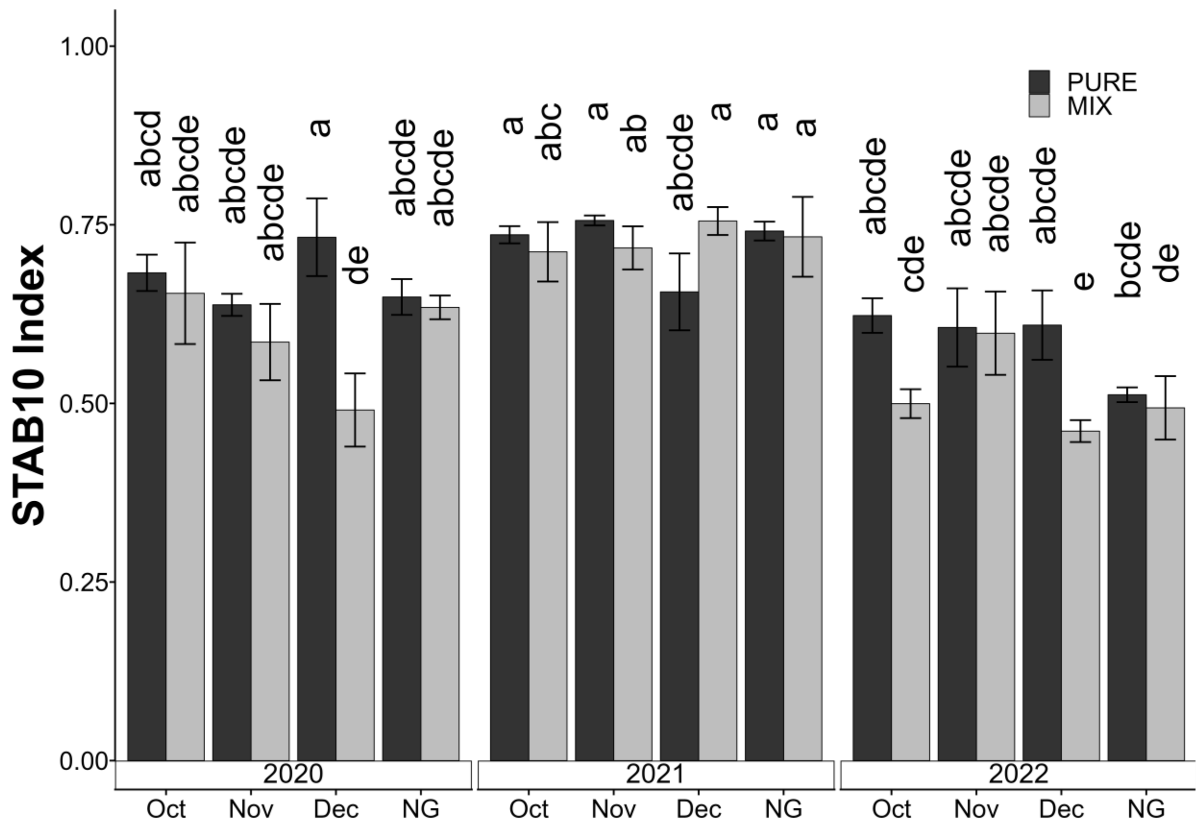

FIGURE 2.4 Three-way interaction of site-year, cover crop, and grazing date factors for soil aggregate stability (STAB10) measured in spring 2020-2022 following fall cover crop grazing in East Lansing, MI. Error bars represent standard error. Bars that share the same letter are not considered different based on Tukey's HSD (*P* > 0.05).

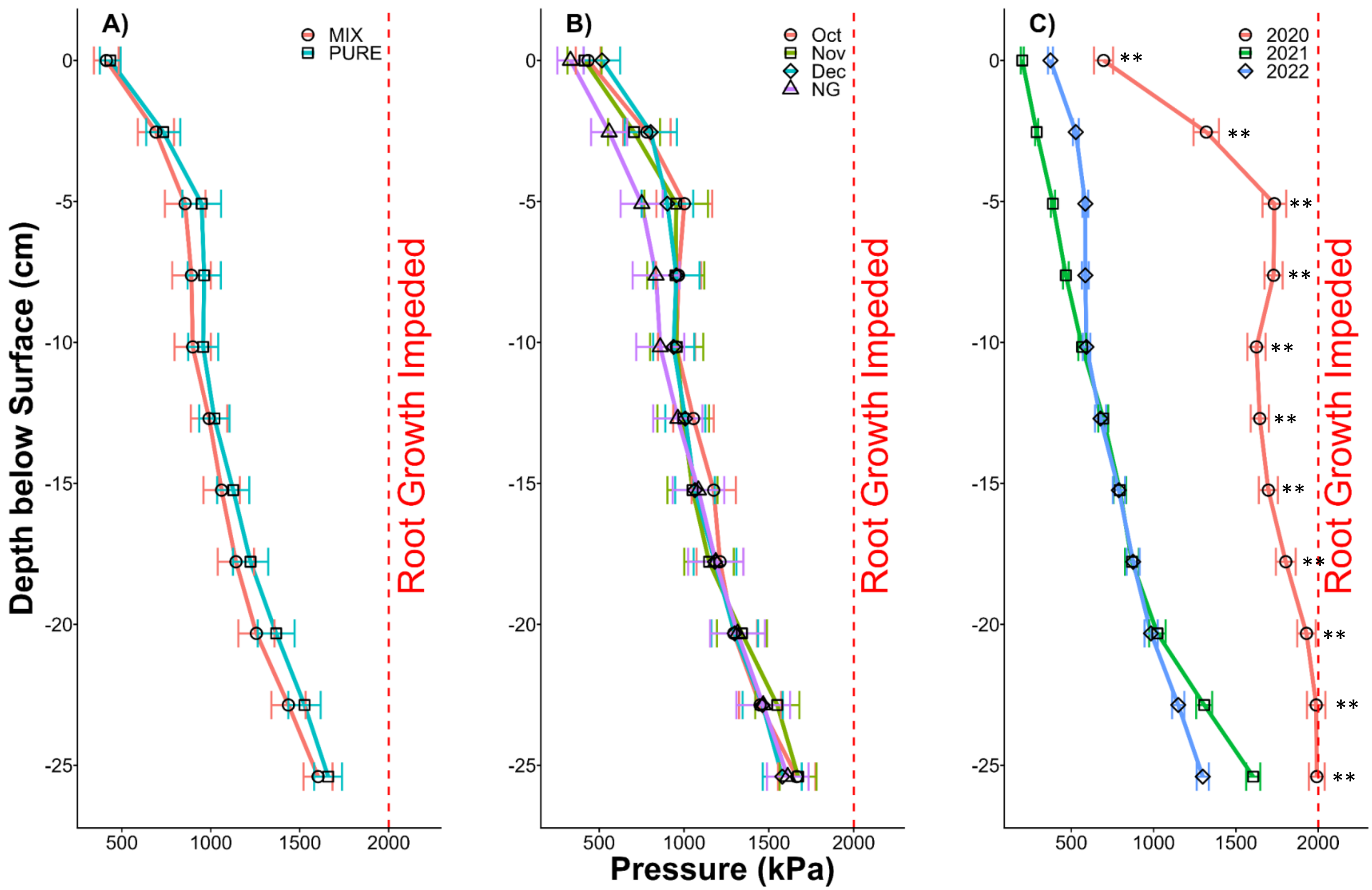

FIGURE 2.5 Spring soil penetration resistance 0 to 30 cm in depth following fall cover crop grazing in East Lansing, MI, for cover crop mixture by depth (A), grazing date by depth (B), and site-year by depth (C). Error bars represent standard error. The dashed line indicates the threshold that may impede plant root development. **Significant at the $P = 0.01$ value.

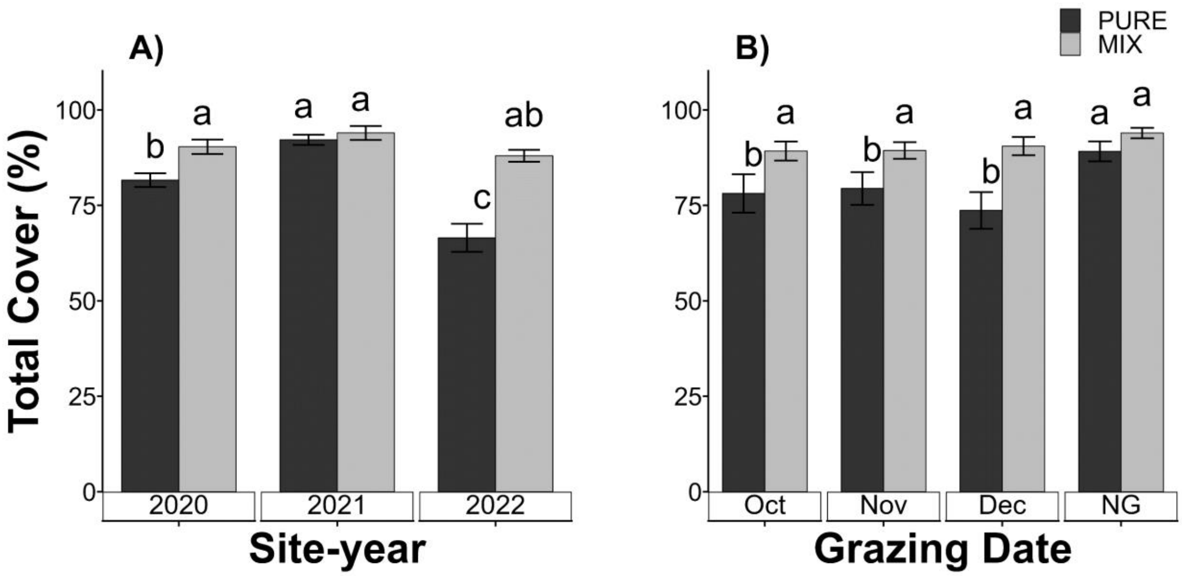

FIGURE 2.6 Proportion of total ground covered in spring following fall cover crop grazing in East Lansing, MI in 2020, 2021 and 2022 for the interactions of cover crop by site-year (A) and grazing date by cover crop (B). Error bars represent standard error. Bars that share the same letter are not considered different based on Tukey's HSD (*P* > 0.05).

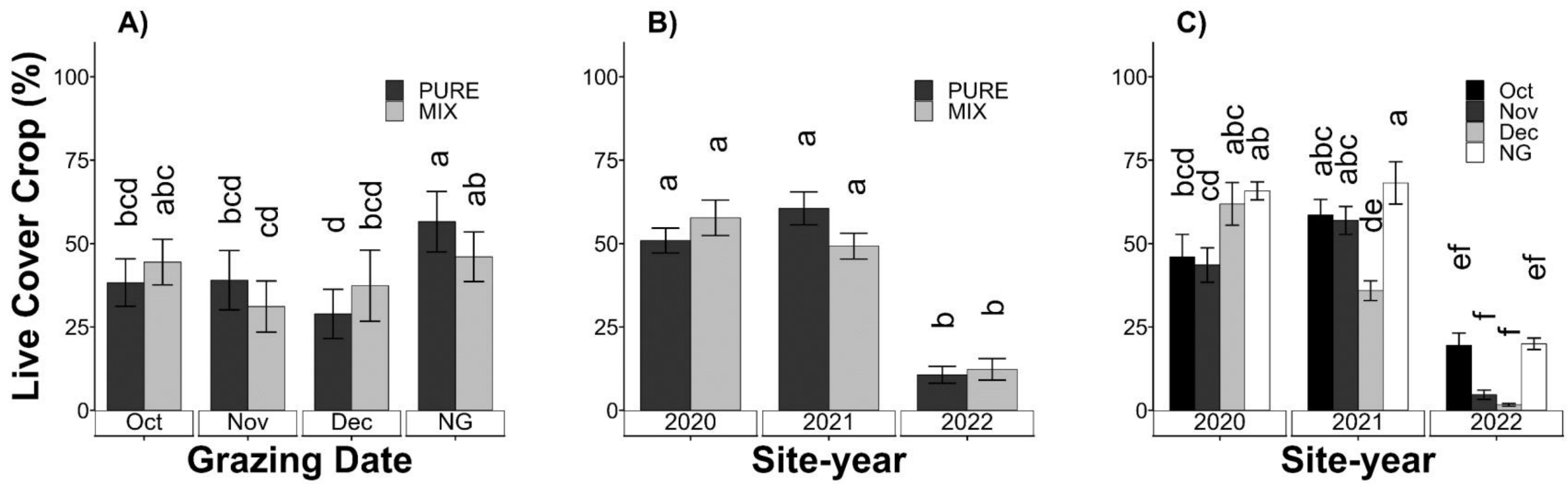

FIGURE 2.7 Proportion of ground covered by live cover crop in spring following fall cover crop grazing in East Lansing, MI in 2020, 2021 and 2022 for the interactions of cover crop by site-year (A) and grazing date by cover crop (B), and grazing date by site-year (C). Error bars represent standard error. Bars that share the same letter are not considered different based on Tukey's HSD (P > 0.05).

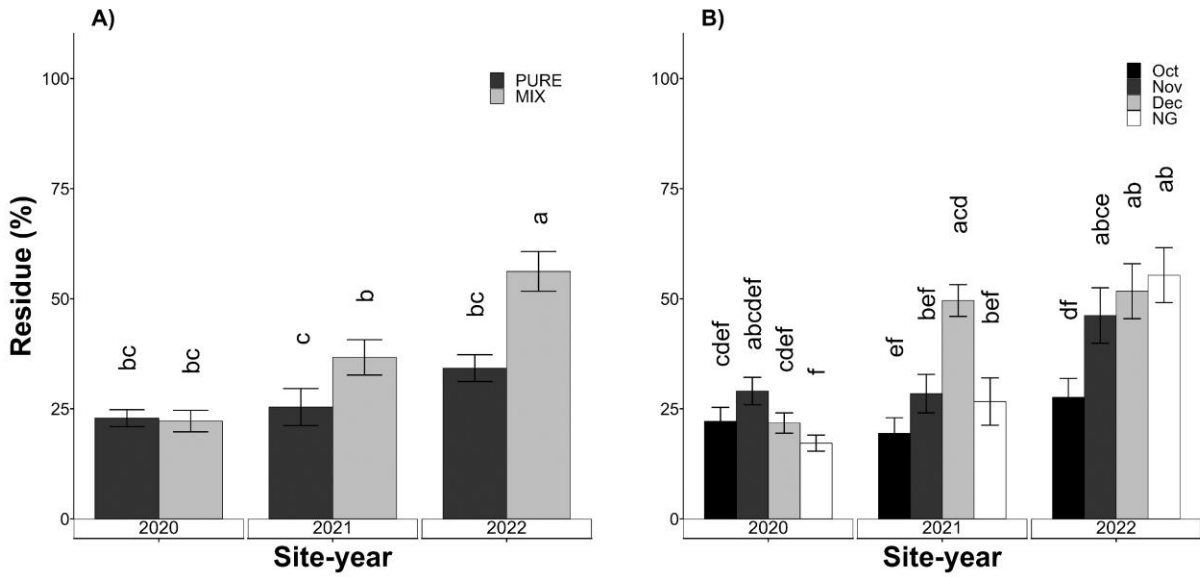

FIGURE 2.8 Proportion of ground covered by residue in spring following fall cover crop grazing in East Lansing, MI in 2020, 2021 and 2022 for the interaction of cover crop by site-year (A) and grazing date by site-year (B). Error bars represent standard error. Bars that share the same letter are not considered different based on Tukey's HSD ($P > 0.05$).

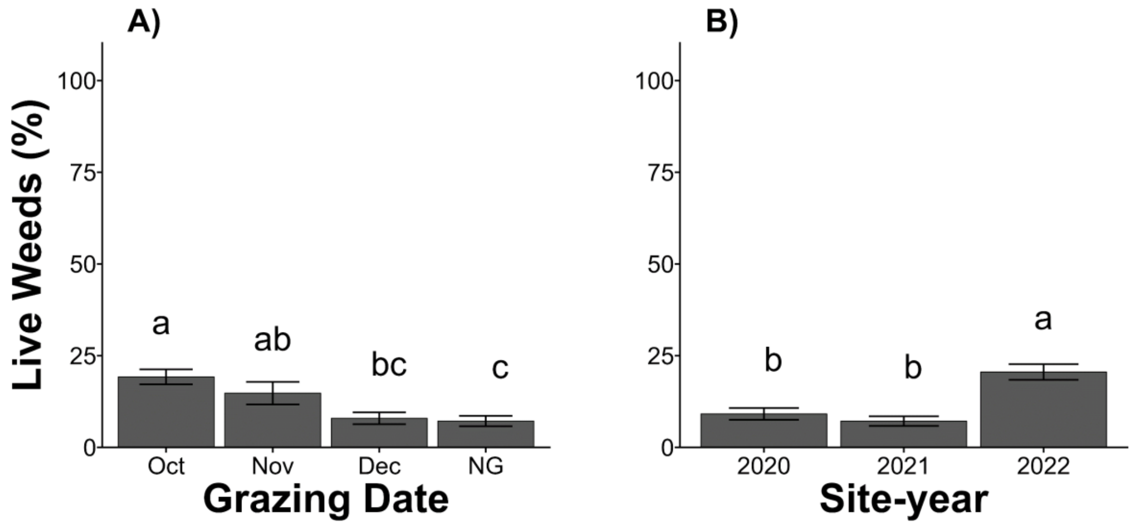

FIGURE 2.9 Proportion of ground covered by live weeds in spring following fall cover crop grazing in East Lansing, MI in 2020, 2021 and 2022 for the main effects of grazing date (A) and site-year (B). Error bars represent standard error. Bars that share the same letter are not considered different based on Tukey's HSD (P > 0.05).

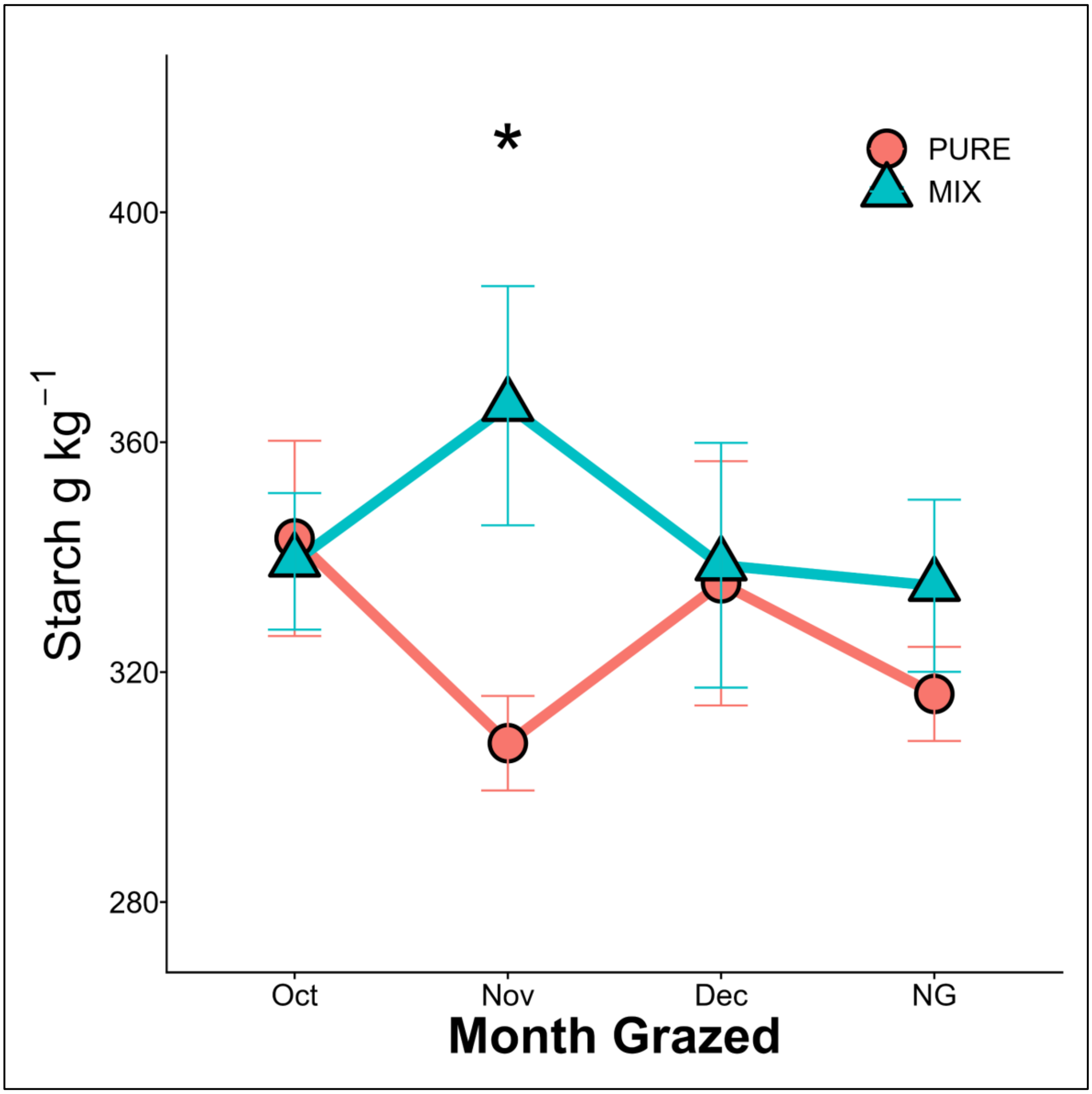

FIGURE 2.10 Cover crop by grazing date interaction plots of corn starch content measured shortly after corn harvest in the summers of 2021 and 2022. All results are on a dry matter basis. Error bars represent standard error. Significance is denoted by asterisks ($P < 0.05$). *Significant at the $P = 0.05$ value.

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APPENDIX

Code for aggregate stability image analysis in R:

```
#Set wd where your pictures are in, in my case 'WAS Photos'
dir_path="C:/Users/dk186/OneDrive/Documents/MSU Masters Project/CCS Stats/WAS
Photos/CCS 3/Upside down"
#Create a list of the file names that are required to run the loop, print to
make sure its correct
files=list.files(path=dir_path, pattern='\\.jpg$', full.names=TRUE)
#Create a dataframe to store our results in with a columns for Sample ID and
Ag Stability Index (STAB10)
results <- data.frame('Sample_ID'=character(),
'STAB10'=numeric(),stringsAsFactors = FALSE)
#~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~
#LOOP FOR IMAGE ANALYSIS
#~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~~
#FOR THIS TO WORK YOUR IMAGES MUST BE NAMED IN SUCH A WAY THAT THEY ARE NEXT
TO EACH OTHER IN THE FILE
#FOR EXAMPLE: [SAMPLEID]-pre and [SAMPLEID]-post
for (i in seq(1, length(files), 2)){
  #Using gsub to delete the directory from the sample name (i.e. delete
everything before the / and replace it with nothing)
  file_name_a=gsub(".*/",'',files[i])
  file_name_b=gsub(".*/",'',files[i+1])

  #This line removes the pre and post designation and stores the sample ID on
its own in a list
  sample_id=gsub("-pre.jpg|-post.jpg", '', file_name_a)

  #Read in each image in the couplet
  img_a=readImage(files[i])
  img_b=readImage(files[i+1])

  #Cropping image a and B - this will need to be tweaked depending on your
setup
  crop_a=img_a[900:2400,500:2400,]
  crop_b=img_b[900:2400,500:2400,]
  #Make sure to plot each crop to make sure you are capturing all the
aggregates and removing clutter
    #For example petri dish edges and sample-IDs written on the light
  #plot(crop_a)
  #plot(crop_b)

  #Set to grayscale
  colorMode(crop_a)=Grayscale
  colorMode(crop_b)=Grayscale

  #This separates the foreground from the background using Ostu method for
thersholding and is needed to create a binary image
```

```
  threshold_a=otsu(crop_a)
  threshold_b=otsu(crop_b)

  #Create a binary image using the threshold values from before and applying
them to the grayscale image
  binary_a=EBImage::combine(mapply(function(frame,th)frame<=th,
                                getFrames(crop_a),threshold_a,SIMPLIFY =
FALSE))
  binary_b=EBImage::combine(mapply(function(frame,th)frame<=th,
                                  getFrames(crop_b),threshold_b,SIMPLIFY =
FALSE))
  plot(binary_a)
  plot(binary_b)

  #This slices the array along the 3rd dimension (frames) and sums all of
values of the object (area)
  area_a=apply(binary_a, MARGIN = 3,sum,na.rm=T)
  area_b=apply(binary_b, MARGIN = 3,sum,na.rm=T)

  #Was index is the initial area over the final, in my files the after (-
post) image comes before the before (-pre)
    #If you name your photo's [SAMPLEID]-a and [SAMPLEID]-b it will be
area_a/area_b
  WASIndex=mean(area_b/area_a)
  print(sample_id)
  print(WASIndex)

  #Using our dataframe from before, rbind will place the respective sample ID
and STAB10 index into each column
  results=rbind(results, data.frame('Sample_ID' = sample_id,
'STAB10'=WASIndex, stringsAsFactors = FALSE))
}
#Printing our data frame with its values
print(results)
write_xlsx(results, "C:/Users/dk186/OneDrive/Documents/MSU Masters
Project/CCS Stats/WAS Photos//CCS Agstab Output.x
```