

Continuous living cover: Adaptive strategies for putting regenerative agriculture into practice

Edited by

Jacob Jungers, Jose G. Franco, Ashley Conway,
Carol Williams and E. Britt Moore

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Continuous living cover: Adaptive strategies for putting regenerative agriculture into practice

Topic editors

Jacob Jungers — University of Minnesota Twin Cities, United States

Jose G. Franco — U.S. Dairy Forage Research Center, Agricultural Research Service (USDA), United States

Ashley Conway — University of Missouri, United States

Carol Williams — University of Wisconsin-Madison, United States

E. Britt Moore — University of North Carolina Wilmington, United States

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EDITED AND REVIEWED BY
Timothy Bowles,
University of California, Berkeley, United States

*CORRESPONDENCE
Jacob M. Jungers
✉ junge037@umn.edu

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Editorial: Continuous living cover: adaptive strategies for putting regenerative agriculture into practice

Evelyn C. Reilly¹, Ashley Conway-Anderson², Jose G. Franco³,
Jacob M. Jungers^{4*}, E. Britt Moore⁵ and Carol Williams⁶

¹Green Lands Blue Waters, University of Minnesota Twin Cities, St. Paul, MN, United States, ²Center for Agroforestry, University of Missouri, Columbia, MO, United States, ³U.S. Dairy Forage Research Center, Agricultural Research Service (USDA), Madison, WI, United States, ⁴Department of Agronomy and Plant Genetics, University of Minnesota Twin Cities, St. Paul, MN, United States, ⁵Department of Environmental Sciences, University of North Carolina Wilmington, Wilmington, NC, United States, ⁶Department of Plant and Agroecosystem Sciences, University of Wisconsin-Madison, Madison, WI, United States

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Editorial on the Research Topic

Continuous living cover: adaptive strategies for putting regenerative agriculture into practice

Introduction

Continuous Living Cover (CLC) is a term used to describe agricultural systems that include year-round vegetative cover above ground and living roots below ground. Examples of CLC include agroforestry, perennial biomass, perennial forages and grazing lands, perennial grains, and systems of summer and winter annuals and cover crops managed to maximize soil coverage (Jewett and Schroeder, 2015; Chrisman et al., 2021). Continuous Living Cover offers a framework for studying and implementing agricultural strategies that keep land in production while maintaining or enhancing soil and water quality in the long term. These strategies promote a diversified agricultural landscape and can be combined in myriad ways to help farmers achieve both economic and environmental goals.

Strategies for achieving CLC addressed in this Research Topic include spring planted winter cereal rye (*Secale cereale* L.) interseeded with soybeans (*Glycine max* (L.) Merr.) (Brockmueller et al.), pennycress (*Thlaspi arvense* L.) relay-cropped with soybeans (Gesch et al.), silvopasture systems (Mayerfeld et al.), perennial grains (Chamberlain et al.; Pinto et al.; Reilly et al.; Cureton et al.; Mulla et al.), perennial forages (Chamberlain et al.; McPheeters et al.), perennial grasslands (Audia et al.; Wepking et al.; Rissman et al.), and cover crops (Ingram; Koehler-Cole et al.; Myers and Wilson; Nichols and MacKenzie; Thompson et al.). While CLC can be employed on a global scale, most of the research in this Research Topic was conducted in the context of the predominant cropping systems in the Midwestern United States, but conclusions are suitable for broader geographies and agroecological systems.

History of the CLC concept

Continuous Living Cover strategies have been used since ancient times. Virgil's (c 29 BCE) writings reference diverse annual rotations, legume cover crops, animal integration and reduced tillage, noting their beneficial effects on soil (Mackail, 1950). North American Indigenous agriculture has long integrated perennial and annual polycultures, intercropping, animals, and agroforestry (Salmón, 2012; Carlisle, 2022; Nabhan et al., 2022; Kapayou et al., 2023). Benefits of these agricultural practices include stabilizing crop yields over time, soil health enhancement, crop pest and pathogen management, and weed reduction, all of which have been reported since at least 1939 (Blake, 1939) and supported by scientific literature since at least the 1980s (Lewandowski, 1987; Rossier and Lake, 2014; Mueller et al., 2019), though they have been observed by practitioners for much longer.

In the early 2000s, a coalition of partners across the U.S. Upper Midwest, including the Green Lands Blue Waters steering committee, was looking for a term to convey the sustainable agriculture practices and goals they wanted to promote. They initially used "continuous cover" and "conservation cover" before arriving at "Continuous Living Cover", which became an umbrella term around which others began organizing (Aaron Reser, personal communication, June 25, 2023; Jeff Berg, personal communication, June 30, 2023). The term appears in titles and keywords of scientific literature from 2010 (Jordan and Warner, 2010), and in U.S. government agency funding and support beginning slightly later (e.g., SARE, 2014).

The benefits of CLC systems in the U.S. Upper Midwest are relatively well-documented (Feyereisen et al., 2006; Basche and DeLonge, 2017; Franco et al., 2018, 2021a; Liebig et al., 2018; Jungers et al., 2019; Reilly et al.), and a renewed interest in them has been brought about by the continuing dominance of low diversity, input-intensive cropping systems and the adverse impacts associated with them.

Rationale

There is an urgent need for agriculture systems that keep land in production while preserving soil and water quality, providing wildlife habitat, and limiting greenhouse gas emissions. In the U.S. Upper Midwest, summer annual row crops have replaced much of the historical native forests and prairies (Schulte et al., 2006; Liebman and Schulte, 2015) that built deep soils and supported diverse ecosystems. The current agricultural paradigm is supported by federal policy, notably crop insurance, along with well-developed infrastructure and supply chains, technical assistance, industry interests, and dominant narratives about American agriculture (Boody et al., 2005; Jordan et al., 2007). For example, agricultural subsidies totaled \$276.1 billion from 1995 to 2021, the majority of which supported a few annual commodity crops including corn (*Zea mays* L.), soybeans, wheat, and cotton (*Gossypium hirsutum* L.) (EWG, 2023). While modern row crop agriculture produces high yields, it also results in negative externalities which are well-documented and widespread (Boody and DeVore, 2006; Davis et al., 2012; Liebman and Schulte, 2015).

Rates of soil erosion from farm fields in the U.S. Midwest are 10–1,000 times higher than natural systems (Quarrier et al.,

2023), resulting in the loss of an estimated ~57.6 billion tons of soil over the past 150 years (Thaler et al., 2022), as well as large losses of soil organic carbon (Sanford et al., 2012; Sanderman et al., 2017). Widespread nitrogen fertilizer continues to contribute to the hypoxic zone in the Gulf of Mexico (Rabalais and Turner, 2019), nitrate leaching into groundwater, and formation of the potent greenhouse gas nitrous oxide (Wang and Li, 2019), threatening human health, ecosystem function, and long-term climate stability. Globally, the food system is the largest driver of biodiversity loss and continues to threaten species as land is converted to agricultural uses (Williams et al., 2020; Knapp and Sciarretta, 2023). Consolidation has also led to fewer, larger farms and decreased diversity of farm owners (USDA, 2019; Congressional Research Service, 2021).

Continuous Living Cover systems offer an evidence-based avenue to address these challenges. They facilitate longer periods of crop growth that maximize solar energy use, minimize erosion and nutrient loss, support greater wildlife diversity, incorporate more crop and livestock species, and provide socioeconomic benefits such as diversified income streams (Boody et al., 2005; Jordan et al., 2007; Davis et al., 2012; Tamburini et al., 2020). In addition, by increasing soil organic matter, CLC systems can increase soil water retention, conferring greater resilience to floods and droughts that are becoming more common due to climate change (Hatfield and Dold, 2017; Lal, 2020; Berdeni et al., 2021). Some practices, especially agroforestry and managed grazing, can increase soil organic carbon and could be avenues for agricultural carbon sequestration (Becker et al., 2022; Mayer et al., 2022). Several articles in this Research Topic further describe ecosystem-scale soil, water, and habitat benefits from CLC strategies (Audia et al.; Reilly et al.; Chamberlain et al.; Wepking et al.). There is also evidence that diversified CLC systems can improve agronomic outcomes including yield, yield stability, and weed and pest suppression (Davis et al., 2012; Isbell et al., 2017; Tamburini et al., 2020).

Scientific basis for CLC

The science of CLC is firmly rooted in ecology. Soil ecosystems require energy and nutrient inputs, the means for nutrient cycling and nutrient loss minimization, and protection from degradative forces. Inputs must be of a biochemical diversity commensurate with the diverse types of ecophysiology and ecological life strategies found in these systems. In short, the scientific basis of CLC is supported by four foundational concepts: functional biodiversity, rhizosphere activity, year-round surface cover, and minimal disturbance.

Functional biodiversity

Functional biodiversity is the collective of organismal and ecological traits that increase overall ecosystem service provisions, resistance, and resilience (Tilman et al., 1997, 2014; Loreau et al., 2001; Hooper et al., 2005). A growing body of literature speaks to the importance of functional biodiversity to agroecosystems. Adding to the functional biodiversity of cropping systems has been shown to enhance productivity (Franco et al., 2015), yield stability

(Khan and McVay, 2019; Franco et al., 2021b), and substantially increase soil health metrics (McDaniel et al., 2014; Costa et al., 2018; Sprunger et al., 2020). Articles in this Research Topic also highlight how functional biodiversity can increase crop pest suppression [Brockmueller et al.; Bruce et al.(b)], retain nutrients (Wepking et al.), and augment soil water retention (Nichols et al., 2022; Moore, 2023).

Rhizosphere activity

Temporal and spatial expansion of the rhizosphere, along with associated rhizodeposition, microbial activity, and nutrient cycling, have been shown to support soil health and ecosystem functioning (Neumann, 2007; Moore et al., 2014; Reilly et al.). Root exudates seem to disproportionately influence soil microbial community composition (Dennis et al., 2010) and soil organic matter cycling (Sokol et al., 2019) more so than shoot or root decomposition. Kelly et al. (2022) found that crop root exudates were a main factor in determining soil microbial community composition, as well as nitrogen cycling. Other microbes such as arbuscular mycorrhizal fungi that inhabit the rhizosphere are also critical in nutrient cycling and enhancing crop resiliency in response to abiotic stressors (Begum et al., 2019). Another example from research in this Research Topic showed that a perennial grain crop had higher root biomass compared to annual crops, and that this root biomass was likely associated with nitrate leaching reductions in the perennial crop (Reilly et al.).

Year-round surface cover

Year-round cover on the soil surface substantially attenuates wind and water erosion. Incorporation of living cover, such as perennial grass (Acharya et al., 2019) and agroforestry systems (Sauer et al., 2021), have been shown to be effective in reducing sediment transport compared to conventional row crop systems. Additionally, dead or decomposing cover, such as crop residues, can also reduce erosion (Kaspar and Singer, 2011) and improve soil structural stability (Kahlon et al., 2013).

Minimal disturbance

Minimizing disturbance, namely tillage, facilitates functional biodiversity, rhizosphere activity, and perennial surface cover. Soil structure (Kahlon et al., 2013), soil ecological community composition (Mathew et al., 2012), and water flow (Zhang et al., 2017) can vary significantly as a function of tillage. As such, no-tillage and reduced tillage management systems serve to facilitate many of the soil ecosystem services detailed herein.

Challenges and barriers to adoption

While Continuous Living Cover strategies offer many environmental benefits, adoption has been slow. For instance, although cover crop usage has increased by 50% from 2012 (4.2

million ha) to 2017 (6.3 million ha), cover crops were used on only 3.9% of total U.S. cropland (USDA, 2019).

Some challenges are related to the climate. In the U.S. Upper Midwest and other cold climates, the short growing season and limited planting window after harvest of summer annual crops have necessitated research on cover crop interseeding, which has yet to produce consistent results, limiting its use by growers. Even in corn silage production systems, which have a shorter seeding-to-harvest window than corn harvested for grain, cover crops should generally be planted on or before September 15 to provide the greatest benefits (Feyereisen et al., 2006), leaving little time for establishment and biomass production.

Slow adoption is also a result of lack of policy support and incentives (Rissman et al.), as well as limited availability of technical assistance (Cureton et al.). For example, while the United States Department of Agriculture (USDA) Natural Resources Conservation Service (NRCS) encourages year-round cover through practices like cover cropping and intercropping, the Risk Management Agency imposes varying planting limitations for insurance eligibility (NRCS, 2014, 2019; RMA, 2019). Further, though cover crop cost share funding is sometimes available, it may not adequately compensate the farmer for the cost of seed, planting, and potential yield reductions, meaning that implementation may entail personal income loss (Plastina et al., 2018). When CLC practices are incentivized, adoption increases. For example, participation in an incentive program doubled average cover crop acreage among farmers in the Northeastern United States, a region with similar climatic challenges to the U.S. Upper Midwest (Chami et al., 2023).

Another factor is variability and trade-offs in on-farm performance due to regional or other factors, a topic addressed by several of the articles in this Research Topic. For example, Brockmueller et al. observed more variability in yields of organic soybeans with an interseeded rye living mulch compared to the tilled control. Effective weed suppression depended on having enough soil moisture for sufficient rye biomass production, thus, soil moisture influences the success of this CLC practice. Bruce et al.(a) demonstrated that cover crops and reduced tillage management of organic squash (*Cucurbita pepo* L.) resulted in trade-offs: weed pressure was reduced, but yield was also reduced and there was a similar negative outcome on pest pressure. Other work by Bruce et al.(b) shows how living cover crop mulches can reduce both pest and weed pressures, but may also reduce crop yield. Similarly, soybean-pennycress relay systems show promise, but require more regional adaptation research (Gesch et al.).

Also addressed by this literature is one of the challenges for broader CLC adoption, the fact that the factors that affect the scope, extent, or rate of improvement are not well-identified, so it is difficult to predict conditions for the greatest success. Modeling helps to illustrate these dynamics. Grass bioenergy crops strategically integrated into an Iowa watershed could provide ecosystem services, but projected watershed-wide revenues ranged from −\$44.2 to \$128.8 million (Audia et al.). This variability in outcomes, whether it is the magnitude of improvement or simply trade-offs between positive and negative effects, is a key limiting factor in widespread adoption (Ingram) because it creates a high-risk decision-making environment for producers, compounded in some cases by increased management needs. For

example, top concerns reported by non-adopter farmers in a national farmer survey about adopting cover crops were related to variability in system performance (Myers and Wilson). Potential reduction in crop yields and economic returns, and poor stand establishment were second only to the additional time and labor needed to manage an integrated system with cover crops.

A theme that appears throughout this Research Topic is concern about on-farm performance being compounded by the disconnect between institutional resources and support as research attempts to gain a deeper understanding of the nuances in performance of these systems. Koehler-Cole et al. note a significant discrepancy in outcomes in cover crop research between controlled, replicated researcher-led trials and “real world” performance in farmer-led trials, indicating a need for more of the latter. A survey of Wisconsin farmers using cover crops also identified needs for more regionally-specific information, which is deeply entwined with availability of research, as well as better contextualized data—the “story” behind the numbers (Ingram). Though these challenges exist, CLC practices can also be implemented with few apparent downsides. For example, intercropping Kernza (*Thinopyrum intermedium* [Host] Barkworth & D.R. Dewey), perennial grain with legumes increased forage value without decreasing grain yield (Pinto et al.). Reduced tillage didn’t affect the profitability of conventional or organic systems (Pearsons et al.), and occasional tillage could reduce herbicide reliance without harming soil health when combined with cover crops and perennial grains (McPheeters et al.).

The existence of successful CLC systems, ongoing challenges, and the growing interest among farmers (Mayerfeld et al.) underscores the need for continued research efforts to assess which factors influence outcomes under different conditions, as well as for improved policy and technical assistance to encourage adoption and manage risk. This requires building a deeper understanding of agroecological interactions in order to provide practitioners with nuanced recommendations, which can help generate more reliable performance and make the increased effort a worthwhile investment.

Putting CLC into action

Implementing multifunctional agriculture systems built on CLC practices will require ongoing research, consistent communication of technical information to producers, development of relevant enterprises to support sustainable commercialization, and reshaping public policy and opinion (Boody and DeVore, 2006; Jordan and Warner, 2010; Liebman and Schulte, 2015; Jordan et al., 2016). Each article in this Research Topic offers insight from a different perspective into how CLC adoption could be expanded.

Foundational research continues to demonstrate how CLC can achieve the goals of many different stakeholders (Chamberlain et al.; Reilly et al.; Mayerfeld et al.). As research on CLC crops and strategies advances, the findings can be translated into applied practices and tested by researchers and early-adopter growers to determine how to integrate them into conventional cropping systems (Gesch et al.; Koehler-Cole et al.).

Underutilized strategies can help identify research needs (Nichols and MacKenzie), which in some cases should be expanded to on-farm experimentation at a range of scales (Koehler-Cole et al.).

As more empirical data are generated from experiments and on-farm studies, researchers can model where to best promote specific CLC practices for optimized economic and agronomic outcomes (Audia et al.). Innovative strategies such as remote sensing can pinpoint hotspots of adoption, providing useful insights (Thompson et al.). Throughout the development and testing process, researchers also must measure the economic and environmental implications of CLC implementation (e.g., Pearsons et al.; Pinto et al.).

Grower adoption and successful marketing of CLC crops requires effective, ongoing communication between farmers, researchers, intermediaries, technical service providers, policy makers, and food processors (Jordan et al.; Conway). Empirical data and models are important for guiding policy recommendations to support grower adoption of CLC (Mulla et al.; Thompson et al.). Early partnerships are also critical to prioritize research goals and ensure that new CLC practices are deployed in scenarios with high likelihood of success (Mayerfeld et al.).

Conclusion

The articles in this Research Topic span a range of disciplines, describe several topics in agronomic and environmental quality research, and address several key factors for implementation: identifying and addressing research needs; shaping policy and program supports for CLC; and equipping the people and entities central to the transition. The Research Topic compiles research that represents current work and needs around CLC, but perhaps more importantly, it aims to define and establish the concept in the scientific literature. Although there are barriers to establishing CLC systems that are practically and economically viable and accessible to all farmers, CLC strategies offer a pathway to mitigate and perhaps avoid some of the worst harms caused by the dominant agricultural system in the U.S. Upper Midwest. Exciting opportunities are emerging in current research and through innovative partnerships. Pairing new science with an openness to learning more from historical and Indigenous approaches, CLC holds promise to create an agriculture that supports resilient farms, ecosystems, and rural communities.

Author contributions

ER: Conceptualization, Project administration, Writing – original draft, Writing – review & editing. AC-A: Conceptualization, Project administration, Writing – original draft, Writing – review & editing. JF: Conceptualization, Project administration, Writing – original draft, Writing – review & editing. JJ: Conceptualization, Project administration, Writing – original draft, Writing – review & editing. EM:

Conceptualization, Project administration, Writing – original draft, Writing – review & editing. CW: Conceptualization, Project administration, Writing – original draft, Writing – review & editing.

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References

- Acharya, B. S., Blanco-Canqui, H., Mitchell, R. B., Cruse, R., and Laird, D. (2019). Dedicated bioenergy crops and water erosion. *J. Env. Qual.* 48, 485–492. doi: 10.2134/jeq2018.10.0380
- Basche, A., and DeLonge, M. (2017). The impact of continuous living cover on soil hydrologic properties: a meta-analysis. *Soil Sci. Soc. Am. J.* 81, 1179–1190. doi: 10.2136/sssaj2017.03.0077
- Becker, A. E., Horowitz, L. S., Ruark, M. D., and Jackson, R. D. (2022). Surface-soil carbon stocks greater under well-managed grazed pasture than row crops. *Soil Sci. Soc. Am. J.* 86, 758–768. doi: 10.1002/saj2.20388
- Begum, N., Qin, C., Ahanger, M. A., Raza, S., Khan, M. I., Ashraf, M., et al. (2019). Role of arbuscular mycorrhizal fungi in plant growth regulation: implications in abiotic stress tolerance. *Front. Plant Sci.* 10, 1068. doi: 10.3389/fpls.2019.01068
- Berdeni, D., Turner, A., Grayson, R. P., Llanos, J., Holden, J., Firbank, L. G., et al. (2021). Soil quality regeneration by grass-clover leys in arable rotations compared to permanent grassland: effects on wheat yield and resilience to drought and flooding. *Soil Till. Res.* 212, 105037. doi: 10.1016/j.still.2021.105037
- Blake, S. F. (1939). A new variety of *Iva ciliata* from Indian rock shelter in the south-central United States. *Rhodora* 41, 81–86.
- Boody, G., and DeVore, B. (2006). Redesigning agriculture. *Bioscience* 56, 839. doi: 10.1641/0006-3568(2006)56(839:RA)2.0.CO;2
- Boody, G., Vondracek, B., Andow, D. A., Krinke, M., Westra, J., Zimmerman, J., et al. (2005). Multifunctional agriculture in the United States. *Bioscience* 55, 27. doi: 10.1641/0006-3568(2005)055(0027:MAITUS)2.0.CO;2
- Carlisle, L. (2022). *Healing Grounds: Climate, Justice, and the Deep Roots of Regenerative Farming*. Washington, DC: Island Press.
- Chami, B., Niles, M. T., Parry, S., Mirsky, S. B., Ackroyd, V. J., and Ryan, M. R. (2023). Incentive programs promote cover crop adoption in the northeastern United States. *Agric. Env. Lett.* 8, e20114. doi: 10.1002/acl.20114
- Chrisman, S., Cureton, C., Hayden, D., Iutzi, F., Meier, E., Moore, E. B., et al. (2021). *Our Journey to a Transformed Agriculture through Continuous Living Cover*. Green Lands Blue Waters. Available online at: <https://greenlandsbluewater.org/wp-content/uploads/2021/08/OurJourneyToTransformedAgThruCLC-GLBW2021.pdf> (accessed September 18, 2023).
- Congressional Research Service (2021). *Racial Equity in U.S. Farming: Background in Brief*. Congressional Research Service. Available online at: <https://crsreports.congress.gov/product/pdf/R/R46969> (accessed September 30).
- Costa, O. Y. A., Raaijmakers, J. M., and Kuramae, E. E. (2018). Microbial extracellular polymeric substances: ecological function and impact on soil aggregation. *Front. Microbiol.* 9, 1636. doi: 10.3389/fmicb.2018.01636
- Davis, A. S., Hill, J. D., Chase, C. A., Johanns, A. M., and Liebman, M. (2012). Increasing cropping system diversity balances productivity, profitability and environmental health. *PLoS ONE* 7, e47149. doi: 10.1371/journal.pone.0047149
- Dennis, P. G., Miller, A. J., and Hirsch, P. R. (2010). Are root exudates more important than other sources of rhizodeposits in structuring rhizosphere bacterial communities?: Root exudates and rhizosphere bacteria. *FEMS Microbiol. Ecol.* 72, 313–327. doi: 10.1111/j.1574-6941.2010.00860.x
- EWG (2023). *Total Commodity Programs, United States*. Washington, DC: Environmental Working Group. Available online at: <https://farm.ewg.org/progdetail.php?fips=00000&progcode=totalfarm®ionname=theUnitedStates> (accessed September 30).
- Feyereisen, G. W., Wilson, B. N., Sands, G. R., Strock, J. S., and Porter, P. M. (2006). Potential for a rye cover crop to reduce nitrate loss in southwestern Minnesota. *Agron. J.* 98, 1416–1426. doi: 10.2134/agronj2005.0134
- Franco, J. G., Berti, M. T., Grabber, J. H., Hendrickson, J. R., Nieman, C. C., Pinto, P., et al. (2021a). Ecological intensification of food production by integrating forages. *Agronomy* 11, 2580. doi: 10.3390/agronomy11122580
- Franco, J. G., Duke, S. E., Hendrickson, J. R., Liebig, M. A., Archer, D. W., and Tanaka, D. L. (2018). Spring wheat yields following perennial forages in a semiarid no-till cropping system. *Agron. J.* 110, 2408–2416. doi: 10.2134/agronj2018.01.0072
- Franco, J. G., Gramig, G. G., Beamer, K. P., and Hendrickson, J. R. (2021b). Cover crop mixtures enhance stability but not productivity in a semi-arid climate. *Agron. J.* 113, 2664–2680. doi: 10.1002/agj2.20695
- Franco, J. G., King, S. R., Masabni, J. G., and Volder, A. (2015). Plant functional diversity improves short-term yields in a low-input intercropping system. *Agric. Ecosyst. Environ.* 203, 1–10. doi: 10.1016/j.agee.2015.01.018
- Hatfield, J. L., and Dold, C. (2017). “Climate variability effects on agriculture land use and soil services,” in *Soil Health and Intensification of Agroecosystems*, eds M. M. Al-Kaisi, and B. Lowery (London: Academic Press), 25–50.
- Hooper, D. U., Chapin, F. S., Ewel, J. J., Hector, A., Inchausti, P., Lavorel, S., et al. (2005). Effects of biodiversity on ecosystem functioning: a consensus of current knowledge. *Ecol. Monogr.* 75, 3–35. doi: 10.1890/04-0922
- Isbell, F., Adler, P. R., Eisenhauer, N., Fornara, D., Kimmel, K., Kremen, C., et al. (2017). Benefits of increasing plant diversity in sustainable agroecosystems. *J. Ecol.* 105, 871–879. doi: 10.1111/1365-2745.12789
- Jewett, J. G., and Schroeder, S. (2015). *Continuous Living Cover Manual*. Green Lands Blue Waters. Available online at: https://greenlandsbluewater.org/wp-content/uploads/2019/08/CLC_Manual_FULL-1.pdf (accessed September 15, 2023).
- Jordan, N., Boody, G., Broussard, W., Glover, J. D., Keeney, D., McCown, B. H., et al. (2007). Sustainable development of the agricultural bio-economy. *Science* 316, 1570–1571. doi: 10.1126/science.1141700
- Jordan, N., and Warner, K. D. (2010). Enhancing the multifunctionality of US agriculture. *Bioscience* 60, 60–66. doi: 10.1525/bio.2010.60.1.10
- Jordan, N. R., Dorn, K., Runck, B., Ewing, P., Williams, A., Anderson, K. A., et al. (2016). Sustainable commercialization of new crops for the agricultural bioeconomy. *Elementa* 4, 000081. doi: 10.12952/journal.elementa.000081
- Jungers, J. M., DeHaan, L. H., Mulla, D. J., Sheaffer, C. C., and Wyse, D. L. (2019). Reduced nitrate leaching in a perennial grain crop compared to maize in the Upper Midwest, USA. *Agric. Ecosyst. Environ.* 272, 63–73. doi: 10.1016/j.agee.2018.11.007
- Kahlon, M. S., Lal, R., and Ann-Varughese, M. (2013). Twenty two years of tillage and mulching impacts on soil physical characteristics and carbon sequestration in Central Ohio. *Soil Till. Res.* 126, 151–158. doi: 10.1016/j.still.2012.08.001

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- Kapayou, D. G., Herrigty, E. M., Hill, C. G., Camacho, V. C., Nair, A., Winham, D. M., et al. (2023). Reuniting the three sisters: collaborative science with Native growers to improve soil and community health. *Agric. Hum. Values* 40, 65–82. doi: 10.1007/s10460-022-10336-z
- Kaspar, T. C., and Singer, J. W. (2011). *The Use of Cover Crops to Manage Soil*. Lincoln, OR: USDA ARS/UNL Faculty.
- Kelly, C., Haddix, M. L., Byrne, P. F., Cotrufo, M. F., Schipanski, M., Kallenbach, C. M., et al. (2022). Divergent belowground carbon allocation patterns of winter wheat shape rhizosphere microbial communities and nitrogen cycling activities. *Soil Biol. Biochem.* 165, 108518. doi: 10.1016/j.soilbio.2021.108518
- Khan, Q. A., and McVay, K. A. (2019). Productivity and stability of multi-species cover crop mixtures in the Northern Great Plains. *Agron. J.* 111, 1817–1827. doi: 10.2134/agronj2018.03.0173
- Knapp, J., and Sciarretta, A. (2023). Agroecology: protecting, restoring, and promoting biodiversity. *BMC Ecol. Evol.* 23, s12862-023-02140-y. doi: 10.1186/s12862-023-02140-y
- Lal, R. (2020). Soil organic matter and water retention. *Agron. J.* 112, 3265–3277. doi: 10.1002/agj2.20282
- Lewandowski, S. (1987). Diohe'ko, the three sisters in seneca life: implications for a native agriculture in the finger lakes region of New York State. *Agric. Hum. Values* 4, 76–93. doi: 10.1007/BF01530644
- Liebig, M. A., Hendrickson, J. R., Franco, J. G., Archer, D. W., Nichols, K., and Tanaka, D. L. (2018). Near-surface soil property responses to forage production in a semiarid region. *Soil Sci. Soc. Am. J.* 82, 223–230. doi: 10.2136/sssaj2017.07.0237
- Liebman, M., and Schulte, L. A. (2015). Enhancing agroecosystem performance and resilience through increased diversification of landscapes and cropping systems. *Elementa* 3, 000041. doi: 10.12952/journal.elementa.000041
- Loreau, M., Naeem, S., Inchausti, P., Bengtsson, J., Grime, J. P., Hector, A., et al. (2001). Biodiversity and ecosystem functioning: current knowledge and future challenges. *Science* 294, 804–808. doi: 10.1126/science.1064088
- Mackail, J. W. (1950). *Virgil's Works: The Aeneid, Eclogues, Georgics*. New York: NY: The Modern Library, 297.
- Mathew, R. P., Feng, Y., Githinji, L., Ankumah, R., and Balkcom, K. S. (2012). Impact of no-tillage and conventional tillage systems on soil microbial communities. *Appl. Environ. Soil Sci.* 2012, 1–10. doi: 10.1155/2012/548620
- Mayer, S., Wiesmeier, M., Sakamoto, E., Hübner, R., Cardinael, R., Kühnel, A., et al. (2022). Soil organic carbon sequestration in temperate agroforestry systems – A meta-analysis. *Agric. Ecosyst. Environ.* 323, 107689. doi: 10.1016/j.agee.2021.107689
- McDaniel, M. D., Tiemann, L. K., and Grandy, A. S. (2014). Does agricultural crop diversity enhance soil microbial biomass and organic matter dynamics? A meta-analysis. *Ecol. Appl.* 24, 560–570. doi: 10.1890/13-0616.1
- Moore, E. B. (2023). Challenges and opportunities for cover crop mediated soil water use efficiency enhancements in temperate rain-fed cropping systems: a review. *Land* 12, 988. doi: 10.3390/land12050988
- Moore, E. B., Wiedenhoef, M. H., Kaspar, T. C., and Cambardella, C. A. (2014). Rye cover crop effects on soil quality in no-till corn silage-soybean cropping systems. *Soil Sci. Soc. Am. J.* 78, 968–976. doi: 10.2136/sssaj2013.09.0401
- Mueller, N. G., White, A., and Szilagyi, P. (2019). Experimental cultivation of eastern north america's lost crops: insights into agricultural practice and yield potential. *J. Ethnobiol.* 39, 549. doi: 10.2993/0278-0771-39.4.549
- Nabhan, G. P., Colunga-GarcíaMarín, P., and Zizumbo-Villarreal, D. (2022). Comparing wild and cultivated food plant richness between the arid american and the mesoamerican centers of diversity, as means to advance indigenous food sovereignty in the face of climate change. *Front. Sustain. Food Syst.* 6, 840619. doi: 10.3389/fsufs.2022.840619
- Neumann, G. (2007). "Root exudates and nutrient cycling," in *Nutrient Cycling in Terrestrial Ecosystems*, eds P. Marschner, and Z. Rengel (Berlin: Springer Berlin Heidelberg), 123–157.
- Nichols, V. A., Moore, E. B., Gailans, S., Kaspar, T. C., and Liebman, M. (2022). Site-specific effects of winter cover crops on soil water storage. *Agrosyst. Geosci. Environ.* 5, e20238. doi: 10.1002/agj2.20238
- NRCS (2014). *NRCS Conservation Practice Standard Cover Crop (Code 340) – Cover Crop*. United States Department of Agriculture. Available online at: https://www.nrcs.usda.gov/sites/default/files/2022-09/Cover_Crop_340_CPS.pdf (accessed September 18).
- NRCS (2019). *NRCS Cover Crop Termination Guidelines*. United States Department of Agriculture. Available online at: https://www.nrcs.usda.gov/sites/default/files/2022-09/Termination_Guidelines_Designated_6.28_10.24am_%28002%29.pdf
- Plastina, A., Liu, F., Sawadgo, W., Miguez, F., and Carlson, S. (2018). Partial budgets for cover crops in Midwest row crop farming. *J. Am. Soc.* 90–106.
- Quarrier, C. L., Kwang, J. S., Quirk, B. J., Thaler, E. A., and Larsen, I. J. (2023). Pre-agricultural soil erosion rates in the midwestern United States. *Geology* 51, 44–48. doi: 10.1130/G50667.1
- Rabalais, N. N., and Turner, R. E. (2019). Gulf of Mexico hypoxia: past, present, and future. *Limnol. Oceanogr. Bull.* 28, 117–124. doi: 10.1002/lob.10351
- RMA (2019). *Cover Crops and Federal Crop Insurance*. Risk Management Agency, United States Department of Agriculture. Available online at: <https://www.rma.usda.gov/en/Fact-Sheets/National-Fact-Sheets/Cover-Crops-and-Crop-Insurance> (accessed September 18).
- Rossier, C., and Lake, F. (2014). *Indigenous traditional ecological knowledge in agroforestry*. *Agroforestry Notes*, 44. U.S. Department of Agriculture, United States Forest Service, National Agroforestry Center. Available online at: <https://www.fs.usda.gov/nac/assets/documents/agroforestrynotes/an44g14.pdf> (accessed October 2).
- Salmon, E. (2012). *Eating the landscape: American Indian stories of food, identity, and resilience*. Tucson, AZ: University of Arizona Press.
- Sanderman, J., Hengl, T., and Fiske, G. J. (2017). Soil carbon debt of 12,000 years of human land use. *Proc. Natl. Acad. Sci. U. S. A.* 114, 9575–9580. doi: 10.1016/j.pnas.1706103114
- Sanford, G. R., Posner, J. L., Jackson, R. D., Kucharik, C. J., Hedtcke, J. L., and Lin, T.-L. (2012). Soil carbon lost from Mollisols of the North Central U.S.A. with 20 years of agricultural best management practices. *Agric. Ecosyst. Environ.* 162, 68–76. doi: 10.1016/j.agee.2012.08.011
- SARE (2014). *Integrating Continuous Living Cover Into Farming Systems Through Professional Development*. College Park, MD: Sustainable Agriculture Research and Education Program, National Institute of Food and Agriculture, U.S. Department of Agriculture.
- Sauer, T. J., Dold, C., Ashworth, A. J., Nieman, C. C., Hernandez-Ramirez, G., Philipp, D., et al. (2021). "Agroforestry practices for soil conservation and resilient agriculture," in *Agroforestry and Ecosystem Services*, eds R. P. Udawatta, and S. Jose (Cham: Springer International Publishing), 19–48.
- Schulte, L. A., Asbjornsen, H., Liebman, M., and Crow, T. R. (2006). Agroecosystem restoration through strategic integration of perennials. *J. Soil Water Conserv.* 61, 164A–169A.
- Sokol, N. W., Kuebbing Sara, E., Karlsen-Ayala, E., and Bradford, M. A. (2019). Evidence for the primacy of living root inputs, not root or shoot litter, in forming soil organic carbon. *New Phytol.* 221, 233–246. doi: 10.1111/nph.15361
- Sprunger, C. D., Martin, T., and Mann, M. (2020). Systems with greater perenniality and crop diversity enhance soil biological health. *Agric. Environ. Lett.* 5, e20030. doi: 10.1002/acl2.20030
- Tamburini, G., Bommarco, R., Wanger, T. C., Kremen, C., Van Der Heijden, M. G. A., Liebman, M., et al. (2020). Agricultural diversification promotes multiple ecosystem services without compromising yield. *Sci. Adv.* 6, eaba1715. doi: 10.1126/sciadv.aba1715
- Thaler, E. A., Kwang, J. S., Quirk, B. J., Quarrier, C. L., and Larsen, I. J. (2022). Rates of historical anthropogenic soil erosion in the Midwestern United States. *Earth's Fut.* 10, e2021EF002396. doi: 10.1029/2021EF002396
- Tilman, D., Isbell, F., and Cowles, J. M. (2014). Biodiversity and ecosystem functioning. *Ann. Rev. Ecol. Syst.* 45, 471–493. doi: 10.1146/annurev-ecolsys-120213-091917
- Tilman, D., Knops, J., Wedin, D., Reich, P., Ritchie, M., and Siemann, E. (1997). The Influence of functional diversity and composition on ecosystem processes. *Science* 277, 1300–1302. doi: 10.1126/science.277.5330.1300
- USDA (2019). *2017 Census of Agriculture: Summary and State Data*. United States Department of Agriculture. Available online at: www.nass.usda.gov/Publications/AgCensus/2017/Full_Report/Volume_1,_Chapter_1_US/usv1.pdf (accessed October 1).
- Wang, Z. H., and Li, S. X. (2019). "Nitrate N loss by leaching and surface runoff in agricultural land: a global issue (a review)," in *Advances in Agronomy*, ed D. L. Sparks (London: Elsevier), 159–217.
- Williams, D. R., Clark, M., Buchanan, G. M., Ficitola, G. F., Rondinini, C., and Tilman, D. (2020). Proactive conservation to prevent habitat losses to agricultural expansion. *Nat. Sustain.* 4, 314–322. doi: 10.1038/s41893-020-00656-5
- Zhang, M., Lu, Y., Heitman, J., Horton, R., and Ren, T. (2017). Temporal changes of soil water retention behavior as affected by wetting and drying following tillage. *Soil Sci. Soc. Am. J.* 81, 1288–1295. doi: 10.2136/sssaj2017.01.0038



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EDITED BY
Ashley Conway,
University of Missouri, United States

REVIEWED BY
Rajani Srivastava,
Banaras Hindu University, India
Ajit Singh,
University of Nottingham Malaysia
Campus, Malaysia

*CORRESPONDENCE
Ben Brockmueller
brockmueller@wisc.edu

†PRESENT ADDRESS
Léa Vereecke,
Rodale Institute, Madison, WI,
United States

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Spring-seeded winter rye living mulches enhance crop biodiversity and promote reduced tillage organic soybeans

Ben Brockmueller^{1*}, Nicole E. Tautges², Léa Vereecke^{1†} and Erin M. Silva¹

¹Department of Plant Pathology, University of Wisconsin, Madison, WI, United States, ²Michael Fields Agricultural Institute, East Troy, WI, United States

As recognition increases of the benefits of reducing soil disturbance to preserve soil health, there is mounting interest in developing innovative methods of using cover crops as living mulches to control weeds in organic grain systems. Spring-planted winter cereal rye (*Secale cereale* L.) interseeded with soybeans (*Glycine max.* [L.] Merr.) is a promising, yet untested, living mulch system because rye exhibits vigorous growth in the early spring during the critical weed free period, but then dies back as the soybean canopy matures. The objectives of this study were to compare a rye living mulch system with a tilled “organic business-as-usual” control, and to understand the risks and benefits associated with delaying soybean planting date to manage the weed seed bank prior to establishment of rye and soybeans. Three treatments including (1) a June-planted rye and soybean living mulch system, (2) a June planted tilled control, and (3) a May planted tilled control, were compared in terms of weed prevalence and soybean grain yield in a randomized complete block experimental design with four replications implemented across 3 site years from 2019 to 2020. Interseeding rye as a living mulch resulted in consistently higher weed pressure as compared to tilled controls. Increased weed pressure in May- over June-planted controls in 2 of 3 site years indicate planting date influences weed dynamics. Rye biomass was positively correlated with soybean yield ($R^2 = 0.76$, $r = 0.87$, $p < 0.05$) and negatively correlated with weed biomass ($R^2 = 0.63$, $r = -0.79$, $p < 0.05$). Under optimal conditions where rye biomass was maximized, interseeding rye adequately suppressed weeds without reducing soybean yields as compared to tilled controls. However, under drier conditions with lower rye production, increased weed pressure and reduced yields emphasize the risks associated with living mulch systems.

KEYWORDS

organic agriculture, living mulch, interseeding, winter cereal rye, soybeans

Introduction

Organic agriculture provides an alternative management paradigm that limits environmental externalities through the adoption of practices that promote ecosystem services; most notably soil health (Tuck et al., 2013; Reganold and Wachter, 2016; Muller et al., 2017). However, concerns over organic grain production systems’ dependence

on tillage to control weeds, terminate cover crops, manage disease, and incorporate crop residues has prompted substantial research and innovation efforts to advance the development of reduced or no-till organic cropping systems (Carr, 2017; Silva and Delate, 2017; Silva and Vereecke, 2019). In recent years, advances in no-till organic soybean production have arisen through the strategic use of fall-planted cover crops, most commonly winter cereal rye, to suppress weed growth in place of tillage (Silva, 2014; Silva and Delate, 2017; Vincent-Caboud et al., 2019). Winter rye is typically seeded in September in the Upper Midwest (Silva, 2014; Silva and Delate, 2017) to reach the recommended minimum threshold of 8,000 kg ha⁻¹ of biomass for consistent weed suppression (Mirsky et al., 2013; Vincent-Caboud et al., 2019) before being terminated with a roller crimper in the spring to create a thick mulch layer to cover the soil. Meeting these early rye planting date requirements to achieve adequate weed suppression can be challenging following corn, which almost always appears before soybeans in rotations and is often not harvested for grain until November in the Upper Midwest. Therefore, organic producers are seeking adaptive management strategies to achieve weed suppression while maintaining reduced tillage systems *without* relying on fall-planted covers. Interseeding winter rye simultaneously with soybeans as a living mulch has been demonstrated as an alternative weed control approach that minimizes soil disturbance in organic soybean production (Thelen et al., 2004; Uchino et al., 2009; Nelson et al., 2011), would not rely on operational timing in the fall, and could even enable stale seedbed techniques in the spring prior to planting.

Living mulches maintain both weed suppression and cash crop yield when selected living mulch species have morphological and physiological differences from cash crops that limit competition for light, nutrients, and water (Verret et al., 2017; Bhaskar et al., 2021). Winter rye is an ideal living mulch for soybeans because its life cycle is complementary to soybean's. Due to winter rye's vernalization requirement, it will not set seed when spring sown (Bàrberi, 2002; Uchino et al., 2009), precluding issues from rye seedbank or soy seed lot contamination. Winter cereal rye creates a living mulch that is highly competitive with the early season germinating weeds through light interception, soil resource competition, and allelopathic effects (Brainard and Bellinder, 2004; Reberg-Horton et al., 2005; Datta et al., 2017; Vollmer et al., 2020; Bhaskar et al., 2021). However, winter rye begins to senesce as temperatures increase and soybean demand for water and nutrients intensifies, thereby potentially limiting the competitive effects of the living mulch on soybean grain yield (Robinson and Dunham, 1954; Ateh and Doll, 1996; Thelen et al., 2004).

While spring-seeding winter rye as a living mulch in soybean production systems was initially proposed in the 1950's, knowledge on agronomic best management practices remains limited. Explorations into variations on rye seeding rate (Ateh and Doll, 1996; Nelson et al., 2011), soybean seeding rate

(Thelen et al., 2004), rye and soybean planting dates (Thelen et al., 2004; Nelson et al., 2011), and soybean row spacings (Nelson et al., 2011) have been examined in the literature. High soybean seeding rates and narrow rows have been recommended as cultural weed control strategies that can result in earlier canopies and greater direct competition with weeds (Holshouser and Whittaker, 2002; Mortensen et al., 2012; Datta et al., 2017). However, planting soybeans on narrow rows increases risk as it precludes the opportunity to perform cultivation if weed control from the rye living mulch becomes inadequate (Uchino et al., 2009; Nelson et al., 2011). Planting rye prior to soybean planting has been seen as impractical as rye growth competes strongly with soybeans for resources (Robinson and Dunham, 1954; Uchino et al., 2009). Seeding rye in the weeks following soybean planting have limited impacts on soybean vigor and grain yield; however, delayed seeding prohibits winter rye's niche of competing with the early season weeds thereby requiring additional soil disturbance to control weeds prior to rye interseeding (Uchino et al., 2009).

A previous experience at the University of Wisconsin has indicated that there is value to delaying rye and soybean planting dates later than typically seeded under standard organic management practices to decrease the risk from rye vernalization occurring as well as to lower the weed seed bank through additional stale seed bedding prior to planting (Rasmussen, 2004; Boyd et al., 2017). However, delaying planting to reduce weed populations represents a potential trade off as delaying planting dates beyond mid-May in the Upper Midwest can result in the loss of yield potential (Pedersen and Lauer, 2004; De Bruin and Pedersen, 2008; Hu and Wiatrak, 2012). Therefore, there remains a need to elucidate the dynamics of winter rye and soybean planting dates to determine agronomic management systems that balance weed suppression with soybean production.

Given the potential utility of spring-seeded rye as a reduced tillage weed control measure in organic agriculture, further research is required to optimize management and address yield losses (Uchino et al., 2009; Nelson et al., 2011). The objectives of this study were to understand potential risks and benefits from (1) interseeding winter rye with soybeans and (2) delaying soybean planting date within the context of a spring-seeded rye system on weed prevalence and soybean production in the Upper Midwest to further guide adaptation of organic weed management strategies.

Materials and methods

Field history and site description

A field experiment to examine spring-seeded winter rye with soybeans was implemented at the Arlington Agricultural Research Station (AARS) (43°30'N, 89°34'W) in Columbia

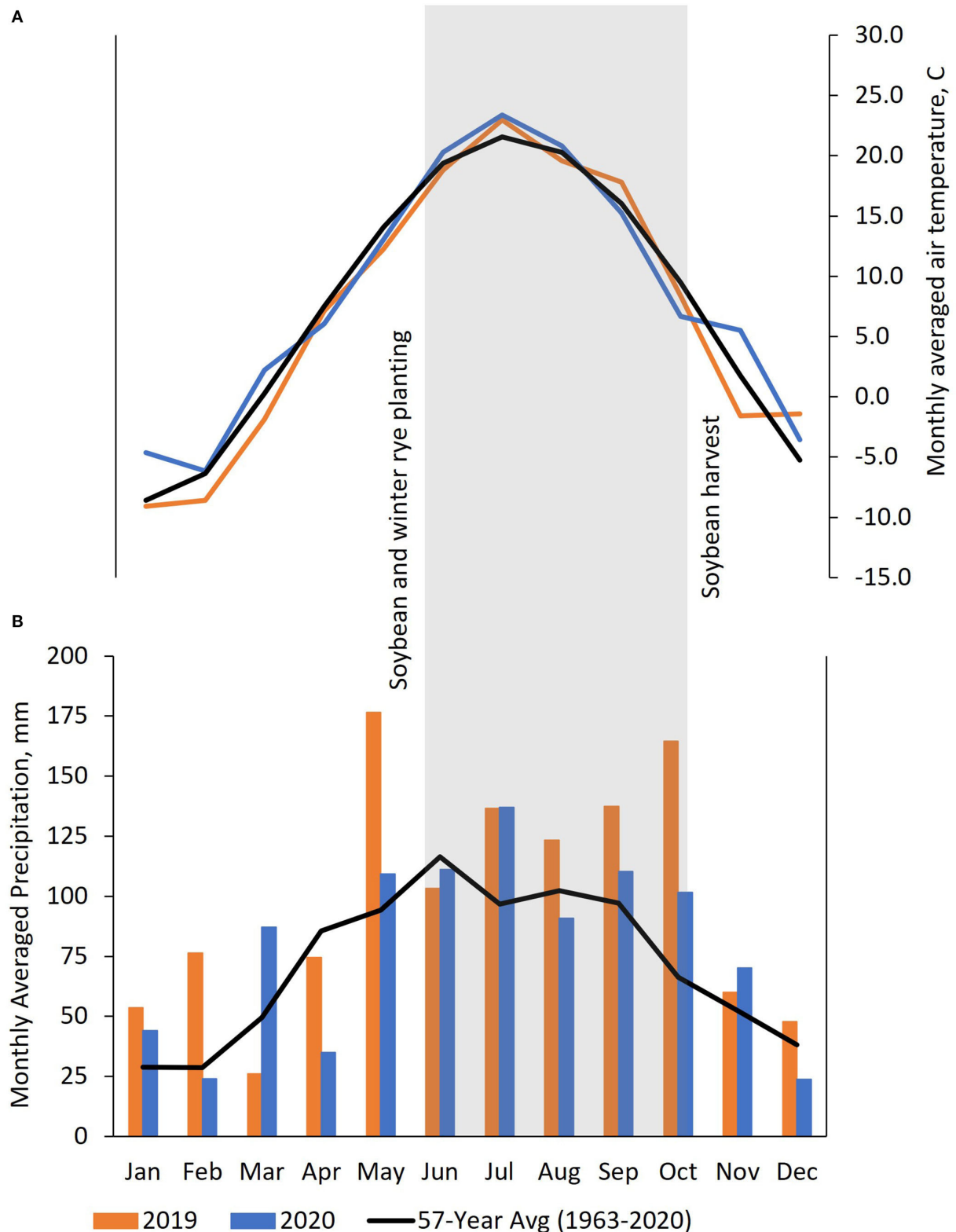


FIGURE 1

(A) Monthly averaged air temperatures (°C) and (B) precipitation (mm) plotted against 57-year averages (1963–2020) at the Arlington Agricultural Research Station, Arlington, WI, 2019–2020. Area shaded in gray signifies the active growing season of soybean during the 2 study years.

County, WI in 2019–2020. The experimental location contained a Plano series silt loam soil (Fine-silty, mixed, superactive, mesic Typic Argiudolls) (USDA, NRCS, 2021). These trials were implemented following a corn silage crop on certified organic land (Midwest Organic Services Agency, Viroqua, WI). Weather data for the research site was acquired from the National Weather Service and accessed through the Wisconsin State Climatology Office (<https://www.aos.wisc.edu/~sco/>). Plots were established under rainfed conditions without irrigation in all site years. Cumulative annual precipitation was 1,180 mm and 944 mm in 2019 and 2020, respectively, as compared against the 57-year historical average of 861 mm. In-season precipitation trends from June through August were above the historical average by 43.6 mm in 2019 and 23.5 mm in 2020 (Figure 1).

A third site year was conducted at the Michael Fields Agricultural Institute's (MFAI) research farm (42°48'N, 88°26'W) in Walworth County, WI in 2020. The MFAI location contained a St. Charles silt loam soil (Fine-silty, mixed, superactive, mesic Typic Hapludalfs) (USDA, NRCS, 2021). The MFAI experimental site land has been certified organic since 2017, and the previous crop before soybeans in this experiment was an alfalfa + grass mix. Precipitation was only recorded in-season at the MFAI site. Monthly cumulative precipitation of 52.1 mm in June, 78.7 mm in July, and 55.9 mm in August were recorded. These observed values fall 74.5, 156, and 274 mm below the established 7-year site history (2014–2020) averages for June, July, and August, respectively, indicating abnormally dry conditions during the winter rye and soybean growing season.

Treatments and experimental design

A randomized complete block design with three treatments examining the presence or absence of winter rye as a living mulch and soybean planting date were replicated four times in all site years. Treatments contrasted a winter cereal rye living mulch ("June Rye + Soy") with (1) a May-planted

tilled soybean control using standard weed control practices of tine weeding and inter-row cultivation ("May Control"), and to a (2) June-planted tilled soybean control that allows for additional stale seedbed preparation to manage the weed seed bank prior to in-season tine weeding and inter-row cultivation ("June Control") (Figure 2). A May-planted living mulch with soybean treatment was excluded from the experimental design based on previous experience at the University of Wisconsin indicating the early seeding of rye increases risk of rye vernalization and higher weed pressures due to fewer opportunities to manage the weed seed bank prior to planting. Plot size was 68 × 5 m in 2019 and lengthened to 137 m in 2020.

Crop management

Organic fertility sources were not applied either before or after planting during the duration of the study in all site years. Pre-plant cultivation occurred as a stale seedbed technique to reduce the weed seedbank prior to crop establishment (Travlos et al., 2020) using a field cultivator (Sunflower Manufacturing, Beloit, KS) at AARS and a disc + finisher at the MFAI site, from late April until planting (Table 1). Winter rye and soybeans were seeded on the same day with rye planted immediately following soybean sowing (Table 1). Soybeans (Viking O.1706N) were planted (JD 1750, John Deere, Moline, IL) 3.8-cm deep on 76-cm row spacing according to the treatment structure at 531,265 seeds ha⁻¹. Winter rye was interseeded with a no-till drill 2-cm deep on 19-cm row spacing to maintain four rows of rye between each soybean row. A seeding rate of 4.9 million seeds ha⁻¹ using Spooner (University of Wisconsin, Madison, WI) and Aroostook (Soil Conservation Service Plant Materials Center et. al, Big Flatts, NY) winter rye varieties in 2019 and 2020, respectively, in accordance with Wisconsin interseeded spring rye seeding rates as described by Ateh and Doll (1996). Tine weeding and rotary hoeing were performed 1 to 2 times as necessary to adequately disrupt emerging weed seedlings with minimal soil disturbance

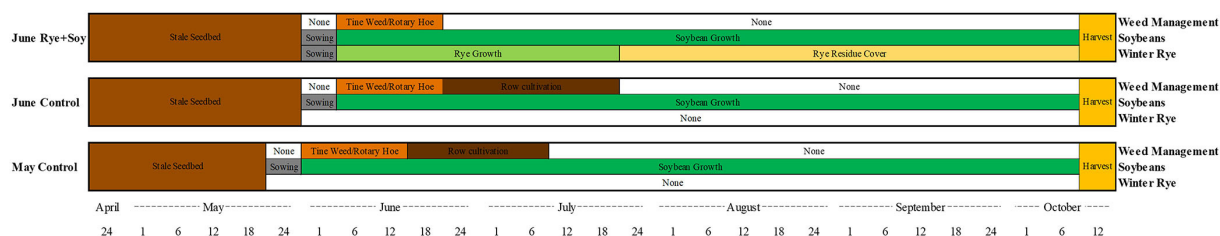


FIGURE 2

Cropping systems diagram illustrating conceptual differences and timing of weed control activities, soybean growth, and winter rye cover when managed under three experimental treatments (June Rye + Soy, June Control, and May Control) as implemented at the Arlington Agricultural Research Station, Arlington, WI and the Michael Fields Agricultural Institute, East Troy, WI, 2019–2020.

TABLE 1 Frequency of pre-plant and post-plant tillage operations for spring-seeded rye and tilled control treatments located at the Arlington Agricultural Research Station (AARS) in Arlington, WI and the Michael Fields Agricultural Institute (MFAI), East Troy, WI, 2019–2020.

Site year	Treatment	Pre-plant cultivation	Planting	Post-plant tine weeding and rotary hoe	Post-plant row cultivation	Soybean harvest
AARS 2019	June Rye + Soy [†]	4x	Rye: June 5 Soybeans: June 5	2x	–	November 24
	June Control	4x	Soybeans: June 5	3x	3x	November 24
	May Control	2x	Soybeans: May 23	2x	4x	November 24
AARS 2020	June Rye + Soy	4x	Rye: June 2 Soybeans: June 2	1x	–	October 9
	June Control	4x	Soybeans: June 2	3x	5x	October 9
	May Control	3x	Soybeans: May 22	3x	5x	October 9
MFAI 2020	June Rye + Soy	3x	Rye: June 1 Soybeans: June 1	1x	–	October 22
	June Control	3x	Soybeans: June 1	1x	2x	October 22
	May Control	2x	Soybeans: May 25	1x	3x	October 22

[†] Treatment Abbreviations: June Rye + Soy (rye interseeded with June-planted soybeans), May Control (tilled soybeans without rye seeded in May), June Control (tilled soybeans without rye seeded in June).

(Table 1) following soybean and rye sowing. However, no mechanical methods of weed control were utilized until the rye two-leaf growth stage (Zadoks 12) and soybean cotyledon stage (VC) to minimize damage to emerging crops (Zadoks et al., 1974). Rye growth then was allowed to control weeds without the use of mechanical weed management tools after initial tine weeding and rotary hoe passes were completed in the June Rye + Soy treatment. In contrast, the no-rye control treatments followed typical weed control management practices for organic soybeans by receiving 2–5 post-plant tillage operations using a field cultivator to achieve adequate weed control until canopy closure of soybean rows. All treatments were harvested for soybean grain upon reaching crop maturity (Table 1).

Data collection

Individual weeds in each plot were counted and separated by grasses and broadleaves as a measure indicating weed abundance. At the AARS location in 2019 and 2020, weed abundance was recorded in the early August by counting all weed plants in three locations per plot using a frame of 0.25 m². At MFAI 2020, weed abundance was recorded on July 1 at two locations per plot using a frame of 0.25 m². In early August at all site years, broadleaf and grass weeds and rye biomass were cut at ground level at the time of rye senescence, and fractions were separated upon collection to understand the community structure of present weeds. Biomass samples were dried in a forced air oven at 60°C until completely dry. Soybean plant stand was determined by counting all soybeans on 1/1,000 of an acre of row length in three locations per plot in mid-July. At AARS, the plots

were machine harvested with yield determined by weighing harvested soybean grain and correcting to 13% moisture from harvested moisture as determined by a soil moisture meter (Dickey-John GAC 2500). At MFAI, soybean grain yields were determined by cutting all soybean plants within a 1-m² quadrat per plot, hand threshing grain from pods, and weighing grain. The yields reported were adjusted to 13% moisture.

Data analysis

Data analysis was performed with RStudio statistical software, version 4.1.0 (R Core Team, 2021). The data was separated and analyzed by individual site year due to significant differences in site year when pooled. A two-way ANOVA linear model tested differences between total weed, grass, and broadleaf biomass as well as soybean stand counts and grain yield. Assumptions of normality and homogeneity of variances were confirmed through examination of residual plots using the ggResidpanel package (Goode and Rey, 2019), Levene's test, and the Shapiro–Wilk normality test. All weed biomass measurements were log(1 + x) transformed to validate the models. The total weed, grass, and broadleaf abundance was analyzed using a negative binomial generalized linear model through the MASS package (Venables and Ripley, 2002). The mean separation was performed using Tukey's HSD at $p < 0.05$ using the agricolae package (de Mendiburu, 2021) while Pearson's correlations were examined using the hmisc package (Harrel and Dupont, 2021). Principle component (PC) analysis was performed using the FactoMineR package (Lê et al., 2008) to calculate the PC scores as well as loadings for each parameter.

TABLE 2 Analysis of variance and treatment means of biomass (dry matter basis) and abundance of total, broadleaf, and grass weeds located at the Arlington Agricultural Research Station (AARS) in Arlington, WI and the Michael Fields Agricultural Institute (MFAI), East Troy, WI, 2019–2020.

Site year	Treatment	Total weed abundance [‡]	Grass weed abundance	Broadleaf weed abundance	Total weed biomass	Grass weed biomass	Broadleaf weed biomass	Total rye biomass
				weeds m ⁻²	kg ha ⁻¹			
AARS	June Rye + Soy [†]	16.0 a [§]	6.75 a	9.25 a	293 a	26.0 a	267 a	2206
2019	June Control	1.38 b	0.63 b	0.75 b	31.6 a	2.88 a	28.8 a	—
	May Control	13.6 a	7.13 a	6.50 a	181 a	132 a	48.4 a	—
AARS	June Rye + Soy	96.3 a	68.7 a	27.7 a	1557 a	1092 a	465 a	949
2020	June Control	0.33 b	0.33 b	0.00 b	33.0 b	0.00 b	33.0 b	—
	May Control	1.67 b	0.00 b	1.67 b	223 ab	0.00 b	223 b	—
MFAI	June Rye + Soy	156 a	61.6 a	94.5 a	2366 a	1425 a	941 b	278
2020	June Control	28.1 c	1.20 c	26.9 c	2458 a	213 a	2245 a	—
	May Control	105 b	43.7 b	61.6 b	2827 a	290 a	2536 a	—
Source				Pr > f				
Treatment (Trt)		***	***	*	**	***	*	—
Site Year (SY)		***	**	***	***	***	***	***
Trt*SY		NS	NS	NS	*	**	*	—

[‡] Total, grass, and broadleaf weed abundance was analyzed using a negative binomial generalized linear model while total, grass, and broadleaf weed biomass used a log transformed linear model with back transformed means presented.

[†] Treatment abbreviations: June Rye + Soy (rye interseeded with June-planted soybeans), May Control (tilled soybeans without rye seeded in May), June Control (tilled soybeans without rye seeded in June).

[§] Means within each column followed by a letter are significantly different at $p < 0.05$ using the Tukey-Kramer procedure of mean separation. Mean separation was performed within each individual site year.

*Significant at the 0.05 probability level.

**Significant at the 0.01 probability level.

***Significant at the 0.001 probability level.

Results

Effect on weed abundance and biomass

There was an interaction between location and year for weed abundance. At 2 site years, total weed abundance was greatest in the June Rye + Soy treatment and was similar to the May Control at AARS 2019 (Table 2). The June Control treatment experienced the lowest total weed abundance at AARS 2019 and MFAI 2020, and was similar to the May Control at AARS 2020 (Table 2). In breaking out the counts by species, the weed abundance counts for grass and broadleaf weeds were statistically similar to total weed abundance results among treatments, and both grass and broadleaf weeds were present at similar levels within a treatment (Table 2).

Fewer differences were observed among treatments in total weed biomass compared to abundance counts, likely due to the high variance often encountered when measuring weed biomass. The only differences in total weed biomass among treatments was detected at AARS 2020, where weed biomass was greater in the June Rye + Soy than the June Control, and the May Control was similar to both treatments (Table 2). In examining grass vs. broadleaf biomass, the only notable difference from the abundance results occurred at MFAI 2020, where broadleaf weed biomass was lower in the June Rye

+ Soy than the May and June Controls (the opposite trend as that observed from weed abundance; Table 2). Although weed biomass was not statistically different between the May and the June Control treatments, at all 3 site years weed biomass trended higher in the May Control (Table 2), similar to trends observed in previous studies (in this case, unusually high variability precluded detection of statistical differences; Supplementary Figures 1, 2).

A principal component (PC) analysis revealed that study location primarily separated along PC 1, with the AARS locations associated with higher soybean yields and stand counts, and the MFAI 2020 location strongly affected by weed biomass, particularly broadleaves (Figure 3). Treatment effects separated mainly along PC 2, with the tilled control treatments not strongly associating with any particular measured variables (but weakly associated with broadleaf weed biomass), and the June Rye + Soy weakly associated with grass weed biomass (Figure 3).

Soybean plant stand

Significant differences in soybean plant stand among treatments were noted in 2 of 3 site years (Table 3). At AARS 2019, higher soybean plant stands were recorded in June Rye

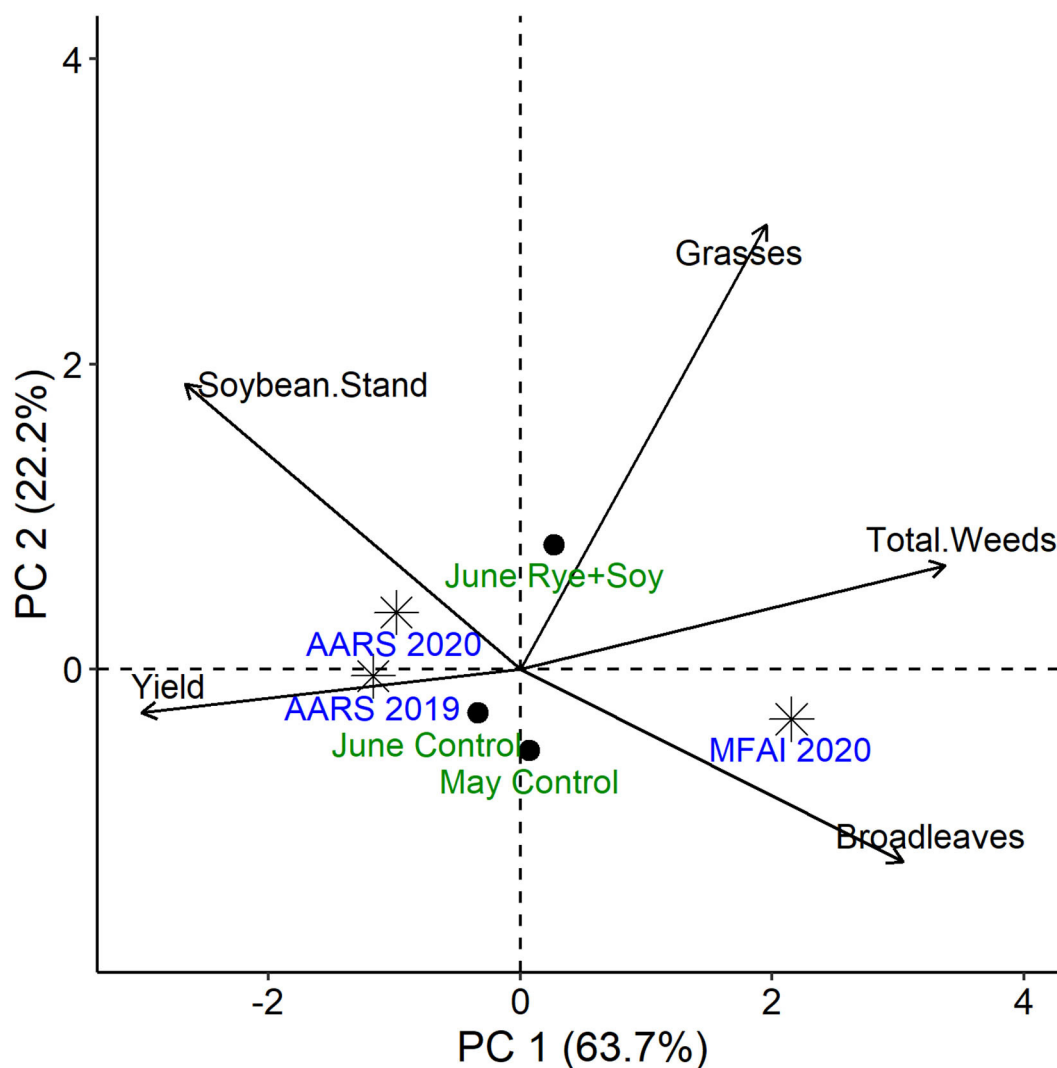


FIGURE 3

Principal component (PC) plot superimposed with eigenvectors of treatments and site years in relation to measures of weed biomass (kg ha^{-1}), soybean plant stand (plants ha^{-1}), and soybean grain yield (Mg ha^{-1}). Treatment abbreviations: June Rye + Soy (rye interseeded with June-planted soybeans), May Control (tilled soybeans without rye seeded in May), June Control (tilled soybeans without rye seeded in June). Site year abbreviations: AARS, Arlington Agricultural Research Station; MFAI, Michael Fields Agricultural Institute.

+ Soy as compared to June Control. May Control exhibited lower soybean stand in relation to either of the June-planted treatments at MFAI 2020 (Table 3), due to relatively wet and cold soil conditions persisting into the end of May in that site year. The Pearson correlation coefficients indicated that soybean stand counts were significantly related to broadleaf weed abundance indicating that soybean stands are important for competing with germinating broadleaf weeds (Table 4). In contrast, soybean stand counts were not correlated with grass weed abundance. Similar correlation trends were observed between the soybean stand counts and the weed biomass, as for weed abundance (Table 4).

Rye biomass

Winter rye biomass production was strongly affected by site year (Table 2) with AARS 2020 and MFAI 2020 achieving only 43 and 12.6% of the rye biomass observed at AARS 2019. The relationship between the rye biomass and the weed biomass was best fit with a quadratic polynomial regression indicating that achieving adequate rye growth is essential for effective weed control (Figure 4A). Both biomass and abundance of broadleaf and grass weed components showed negative correlations with rye biomass with the exception of broadleaf weed abundance ($p < 0.1$; Table 4).

TABLE 3 Analysis of variance and mean soybean stand counts and soybean grain yield, shown with standard deviation, at the Arlington Agricultural Research Station (AARS), Arlington, WI and Michael Fields Agricultural Institute (MFAI), East Troy, WI, 2019–2020.

Site year	Treatment	Soybean stand plants ha ⁻¹	Grain yield Mg ha ⁻¹
AARS	June Rye + Soy [†]	443.133 ± 17.831 a [§]	3.85 ± 0.32 a
2019	June Control	395.772 ± 13.807 b	3.55 ± 0.12 a
	May Control	411.422 ± 27.181 ab	3.85 ± 0.21 a
AARS	June Rye + Soy	433.876 ± 20.546 a	3.24 ± 0.64 b
2020	June Control	449.114 ± 16.244 a	3.88 ± 0.36 ab
	May Control	421.314 ± 24.300 a	4.60 ± 0.19 a
MFAI	June Rye + Soy	359.244 ± 88.256 a	1.85 ± 0.54 b
2020	June Control	321.515 ± 86.884 a	3.10 ± 0.39 a
	May Control	178.802 ± 14.549 b	2.23 ± 0.30 ab
	Source	Pr > f	
	Treatment (Trt)	***	***
	Site Year (SY)	***	***
	Trt*SY	**	***

[†] Treatment abbreviations: June Rye + Soy (rye interseeded with late-planted soybeans), May Control (tilled soybeans without rye seeded in May), June Control (tilled soybeans without rye seeded in June).

[§] Means within each column followed by a letter are significantly different at $p < 0.05$ using the Tukey-Kramer procedure of mean separation. Mean separation was performed within each individual site year.

*Significant at the 0.05 probability level.

**Significant at the 0.01 probability level.

***Significant at the 0.001 probability level.

TABLE 4 Pearson correlation coefficients (r values) between the soybean grain yield (Mg ha⁻¹), the soybean stand (plants ha⁻¹), and the rye biomass (kg ha⁻¹) with measures of weed parameters located at the Arlington Agricultural Research Station in Arlington, WI and the Michael Fields Agricultural Institute, East Troy, WI, 2019–2020.

Weed measurements	Agronomic measurements		
	Soybean yield	Soybean stand	Rye biomass
Total Weed Biomass (kg ha ⁻¹)	−0.69***	−0.55***	−0.71**
Grass Biomass (kg ha ⁻¹)	−0.48**	−0.05 ^{NS}	−0.64*
Broadleaf Biomass (kg ha ⁻¹)	−0.56***	−0.69***	−0.70*
Total Weed Abundance (weeds m ⁻²)	−0.74***	−0.52**	−0.77**
Grass Weed Abundance (weeds m ⁻²)	−0.57***	−0.17 ^{NS}	−0.62*
Broadleaf Weed Abundance (weeds m ⁻²)	−0.59***	−0.59***	−0.50 ^{NS}

^{NS} signifies non-significant results at the $p < 0.05$ probability level.

*Significant at the 0.05 probability level.

**Significant at the 0.01 probability level.

***Significant at the 0.001 probability level.

Soybean yield

Soybean yields were significantly different among treatments in 2 of 3 site years with no difference in yield reported at AARS 2019 (Table 3). At AARS 2020, soybean grain yields were greater in the May Control than the June Rye + Soy, whereas yields in the June Control were similar to the other two treatments (Table 3). At MFAI 2020, soybean grain yields were greatest in the June Control and lowest in the June Rye + Soy, whereas grain yields in the May Control were similar to the other two treatments (Table 3). Soybean grain yield exhibited a quadratic

relationship with rye biomass (Figure 4B) indicating higher soybean yields as rye biomass increased ($R^2 = 0.76$, $p < 0.001$) likely achieved through improved weed suppression. Soybean yield was most strongly correlated with total weed abundance but exhibited negative correlations with all measured indicators of weed prevalence (Table 4).

Discussion

While it appears that rye successfully controlled weeds to an extent, total weed abundance remained higher for June Rye

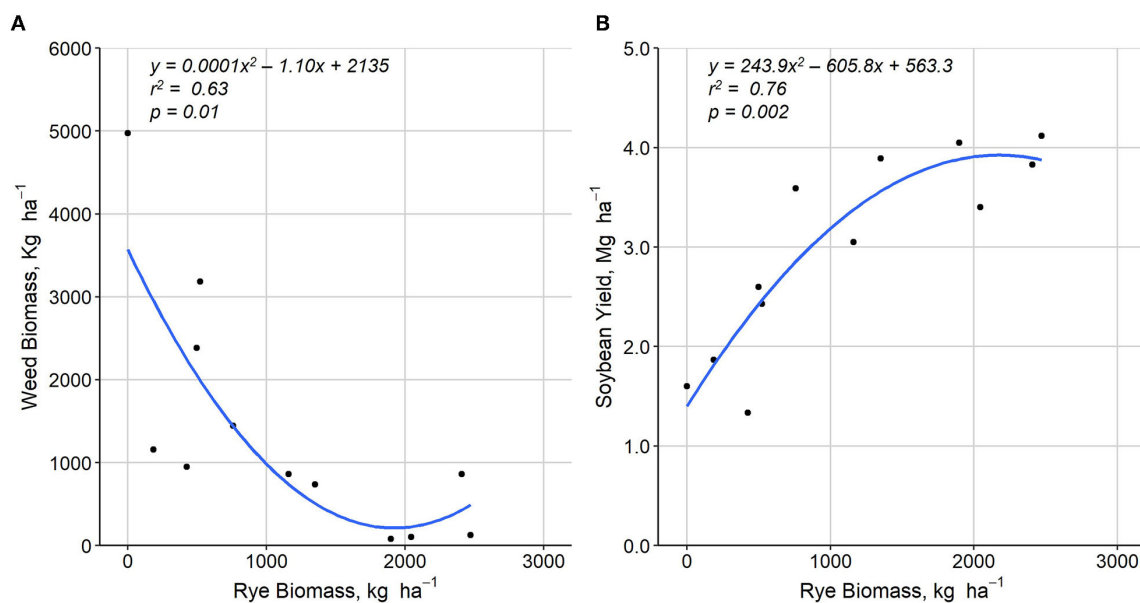


FIGURE 4

Effects of winter rye biomass on (A) weed biomass and (B) soybean yield measured at the Arlington Agricultural Research Station in Arlington, WI and the Michael Fields Agricultural Institute, East Troy, WI, 2019–2020.

+ Soy at all 3 site years matching reports of greater weed prevalence for rye living mulch systems seen in the literature (Robinson and Dunham, 1954; Ateh and Doll, 1996). Weed pressure and rye biomass in this study varied substantially by location likely as a combined result of soil moisture conditions and weed seed bank. Lower rye biomass at MFAI 2020 as compared to other site years was likely a result of dry conditions following rye planting, giving the rye a slow start, whereas the high rye biomass observed at AARS 2019 was likely driven by well above average precipitation throughout the growing season. Rye and weed biomass observed at MFAI 2020 match observations from Nelson et al. (2011) who saw rye dry weights below 400 kg ha⁻¹ resulting in heavy weed pressure exceeding 2,400 kg ha⁻¹ and soybean yields below 2.0 Mg ha⁻¹. In contrast Geddes and Gulden (2021) observed rye mulch exceeding 1,000 kg ha⁻¹ that reduced volunteer canola stands without lowering soybean yield in 2 of 3 site years. These results fit with the trends observed in the present study where a strong positive relationship between rye biomass and soybean yield coupled with a negative relationship between rye biomass and weed biomass suggest that achieving adequate rye biomass is essential to controlling weeds in this system. The previous studies have noted that when rye seeding rates are increased to obtain greater ground cover, soybean yield drops due to greater competition for soil moisture (Ateh and Doll, 1996; Nelson et al., 2011). Therefore, under high precipitation conditions as observed at AARS 2019, the moisture competition from rye may be mitigated while providing substantial biomass to

suppress weeds. Following these results, when spring and the early summers are predicted, we would recommend growers not use rye as a living mulch for soybeans.

Study results indicate that a winter rye living mulch may provide better suppression against broadleaf weeds as opposed to grass weeds indicating that the type of dominant weed present may impact the weed control ability of rye. In this study, interseeding rye was associated with higher levels of grass weeds as compared to broadleaf weeds. Furthermore, at MFAI 2020 which experienced relatively higher proportions of broadleaf weeds as compared to grasses, June Rye + Soy reduced broadleaf weed biomass even though higher broadleaf abundance was observed. This effect was less pronounced for grasses indicating a greater propensity for rye and soybean companion crops to together compete with broadleaf weed types as opposed to grasses. Interestingly, measures of weed abundance proved to be a more sensitive indicator of differences in weed pressure as opposed to weed biomass in this study. While this is likely partially affected by high variability in weed biomass sampling, it may also reflect an ability of rye to reduce weed growth potential through allelopathic effects. Allelopathy in winter rye has been well-established showing that rye's allelochemicals can reduce both weed germination and growth (Barnes and Putnam, 1983; Schulz et al., 2013; Grint et al., 2022). The previous research has suggested that broadleaf weeds tend to exhibit greater sensitivity to the benzoxazinoid allelochemicals produced by rye forming a major component of their weed suppression ability (Barnes and Putnam, 1986; Gavazzi et al., 2010; Schulz et al., 2013).

Soybean yield was not significantly different between the June Control and the May Control in any of the locations. With the exception of MFAI 2020, June Control yielded numerically lower than May Control at AARS 2019 and AARS 2020 indicating the potential for a loss of yield potential as indicated in the literature (Pedersen and Lauer, 2004; De Bruin and Pedersen, 2008; Hu and Wiatrak, 2012). The numeric yield increase of June Control over May Control at MFAI 2020 may be due to a higher weed abundance and lower plant stands reported with May Control. This indicates that while it is likely that yield potential is being lost by delaying planting date, the potential for higher weed pressure due to fewer stale seedbed passes may offset potential increases in yield potential by pushing planting dates earlier. Therefore, earlier planting dates must be balanced with the ability to reduce the weed seed bank prior to planting through stale seedbed practices.

In this study, establishing winter rye as a living mulch showed both neutral and negative effects on soybean yield throughout the course of this study which match the generally negative (Ateh and Doll, 1996; Thelen et al., 2004; Nelson et al., 2011) and generally neutral effects (Robinson and Dunham, 1954; Geddes and Gulden, 2021) observed on soybean yield by year when interseeding rye as a living mulch as compared to a chemically managed or tilled control. Yield reductions when utilizing rye as a living mulch have been attributed to competition for soil moisture (Thelen et al., 2004; Uchino et al., 2009; Geddes and Gulden, 2021) and insufficient weed control (Nelson et al., 2011) which are in accordance with results obtained from the present study. This system proved most successful when under high precipitation conditions that maximized rye biomass growth without competing with soybeans for available moisture, as observed at AARS 2019.

Conclusion

Rye was unable to provide as consistent or as effective weed control as the tilled controls evidenced by higher weed abundance throughout the study. However, the negative relationship observed between rye biomass and weed biomass coupled with the positive affect of rye biomass on soybean yields suggest that under high precipitation conditions rye biomass can be maximized without inducing competitive effects for soil moisture. June Control trended toward lower soybean grain yields as compared to May Control indicating a yield potential loss by pushing the planting dates into June. However, consistently higher weed abundance in May Control represents a tradeoff of maximizing yield potential with achieving adequate weed control. Therefore, delaying establishment of a winter rye living mulch with soybeans until June can aid in reducing weed populations through stale seedbed techniques. Ultimately, successful implementation of a rye living mulch system,

demonstrated specifically at AARS 2019, was achieved by June planting rye and soybeans under high precipitation conditions that allowed substantial rye growth to reduce weed populations without competing with soybeans for soil moisture. This study confirms the potential for spring-seeded rye with reduced tillage as an effective adaptive management approach to increase crop biodiversity without compromising soybean yield in organic crop production systems. However, these results also underscore the risks associated with intensifying agricultural management and minimizing mechanical weed control.

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

Author contributions

LV and NT implemented field trials and collected data. BB and NT provided data analysis and drafted manuscript with contributions from ES and LV. All authors provided substantial intellectual contributions and have agreed upon the final presented results.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

The Supplementary Material for this article can be found online at: <https://www.frontiersin.org/articles/10.3389/fsufs.2022.926606/full#supplementary-material>

References

- Ateh, C. M., and Doll, J. D. (1996). Spring-planted winter rye (*Secale cereale*) as a living mulch to control weeds in soybean (*Glycine max*). *Weed Technol.* 10, 347–353. doi: 10.1017/S0890037X00040070
- Bärberi, P. (2002). Weed management in organic agriculture: are we addressing the right issues? *Weed Res.* 42, 177–193. doi: 10.1046/j.1365-3180.2002.00277.x
- Barnes, J. P., and Putnam, A. R. (1983). Rye residues contribute weed suppression in no-tillage cropping systems. *J. Chem. Ecol.* 9, 1045–1057. doi: 10.1007/BF00982210
- Barnes, J. P., and Putnam, A. R. (1986). Evidence for allelopathy by residues and aqueous extracts of rye (*Secale cereale*). *Weed Sci.* 34, 384–390. doi: 10.1017/S0043174500067035
- Bhaskar, V., Westbrook, A. S., Bellinder, R. R., and DiTommaso, A. (2021). Integrated management of living mulches for weed control: a review. *Weed Technol.* 35, 856–868. doi: 10.1017/wet.2021.52
- Boyd, N. S., Brennan, E. B., and Fennimore, S. A. (2017). Stale seedbed techniques for organic vegetable production. *Weed Technol.* 20, 1052–1057. doi: 10.1614/WT-05-109.1
- Brainard, D., and Bellinder, R. (2004). Weed suppression in a broccoli-winter rye intercropping system. *Weed Sci.* 2, 281–290. doi: 10.1614/WS-03-031R
- Carr, P. M. (2017). Guest editorial: conservation tillage for organic farming. *Agriculture* 7:19. doi: 10.3390/agriculture7030019
- Datta, A., Ullah, H., Tursun, N., Pornprom, T., Knezevic, S., and Chauhan, B. (2017). Managing weeds using crop competition in soybean [*Glycine max* (L.) Merr.]. *Crop Prot.* 95, 60–68. doi: 10.1016/j.cropro.2016.09.005
- De Bruin, J. L., and Pedersen, P. (2008). Soybean seed yield response to planting date and seeding rate in the Upper Midwest. *Agron. J.* 100, 696–703. doi: 10.2134/agronj2007.0115
- de Mendiburu, F. (2021). *agricolae: Statistical Procedures for Agricultural Research*. Available online at: <https://CRAN.R-project.org/package=agricolae>
- Gavazzi, C., Schulz, M., Marocco, A., and Tabaglio, V. (2010). Sustainable weed control by allelochemicals from rye cover crops: from the greenhouse to field evidence. *Allelop. J.* 25, 259–274. doi: 10.4081/ija.2013.e5
- Geddes, C. M., and Gulden, R. H. (2021). Wheat and cereal rye inter-row living mulches interfere with early season weeds in soybean. *Plants* 10:2276. doi: 10.3390/plants10112276
- Goode, K., and Rey, K. (2019). *ggResidpanel: Panels and Interactive Versions of Diagnostic Plots Using 'ggplot2'*. R package version 0.3.0. Available online at: <https://CRAN.R-project.org/package=ggResidpanel>
- Grint, K. R., Arneson, N. J., Oliveira, M. C., Smith, D. H., and Werle, R. (2022). Cereal rye cover crop terminated at crop planting reduces early-season weed density and biomass in Wisconsin corn-soybean production. *Agrosyst. Geosci. Environ.* 5:e20245. doi: 10.1002/ag2.20245
- Harrel, F. E. Jr., and Dupont, C. (2021). *Hmisc: Harell miscellaneous*. R package version 4.6-0. Available online at: <https://hbiostat.org/R/Hmisc>
- Holshouser, D. L., and Whittaker, J. P. (2002). Plant population and row-spacing effects on early soybean production systems in the Mid-Atlantic USA. *Agron. J.* 94, 603–611. doi: 10.2134/agronj2002.6030
- Hu, M., and Wiatrak, P. (2012). Effect of planting date on soybean growth, yield, and grain quality. *Agron. J.* 104, 785–790. doi: 10.2134/agronj2011.0382
- Lê, S., Josse, J., and Huisson, F. (2008). FactoMineR: a package for multivariate analysis. *J. Stat. Softw.* 25, 1–18. doi: 10.18637/jss.v025.i01
- Mirsky, S. B., Ryan, M. R., Teasdale, J. R., Curran, W. S., Reberg-Horton, C. S., Spargo, J. T., et al. (2013). Overcoming weed management challenges in cover crop-based organic rotational no-till soybean production in the Eastern United States. *Weed Technol.* 27, 193–203. doi: 10.1614/WT-D-12-00078.1
- Mortensen, D. A., Egan, J. F., Maxwell, B. D., Ryan, M. R., and Smith, R. G. (2012). Navigating a critical juncture for sustainable weed management. *BioScience* 62, 75–84. doi: 10.1525/bio.2012.62.1.12
- Muller, A., Schader, C., Scialabba, N. E., Brüggemann, J., Isensee, A., Erb, K., et al. (2017). Strategies for feeding the world more sustainably with organic agriculture. *Nat. Commun.* 8:1290. doi: 10.1038/s41467-017-01410-w
- Nelson, K. A., Smeda, R. J., and Smoot, R. L. (2011). Spring-interseeded winter rye seeding rates influence weed control and organic soybean yield. *Int. J. Agron.* 2011, 1–7. doi: 10.1155/2011/571973
- Pedersen, P., and Lauer, J. G. (2004). Soybean growth and development response to rotation sequence and tillage system. *Agron. J.* 96, 1005–1012. doi: 10.2134/AGRONJ2004.1005
- R Core Team (2021). *R: A Language and Environment for Statistical Computing*. R Found. Stat. Comput. Vienna, Austria. <https://www.R-project.org/>
- Rasmussen, I. A. (2004). The effect of sowing date, stale seedbed, row width and mechanical weed control on weeds and yields of organic winter wheat. *Weed Res.* 44, 12–20. doi: 10.1046/j.1365-3180.2003.00367.x
- Reberg-Horton, S. C., Burton, J. D., Danehower, D. A., Ma, G., Monks, D. W., Murphy, J. P., et al. (2005). Changes over time in the allelochemical content of ten cultivars of rye (*Secale cereale* L.). *J. Chem. Ecol.* 31, 179–193. doi: 10.1007/s10886-005-0983-3
- Reganold, J. P., and Wachter, J. M. (2016). Organic agriculture in the twenty-first century. *Nat. Plants* 2:15221. doi: 10.1038/nplants.2015.221
- Robinson, R., and Dunham, R. (1954). Companion crops for weed control in soybeans 1. *Agron. J.* 46, 278–281. doi: 10.2134/agronj1954.00021962004600060010x
- Schulz, M., Marocco, A., Tabaglio, V., Macias, F. A., and Molinillo, J. M. G. (2013). Benzoxazinoids in rye allelopathy—From discovery to application in sustainable weed control and organic farming. *J. Chem. Ecol.* 39, 154–174. doi: 10.1007/s10886-013-0235-x
- Silva, E. M. (2014). Screening five fall-sown cover crops for use in organic no-till crop production in the Upper Midwest. *Agroecol. Sustain. Food Syst.* 38, 748–763. doi: 10.1080/21683565.2014.901275
- Silva, E. M., and Delate, K. (2017). A decade of progress in organic cover crop-based reduced tillage practices in the Upper Midwestern USA. *Agriculture* 7:44. doi: 10.3390/agriculture7050044
- Silva, E. M., and Verecke, L. (2019). Optimizing organic cover crop-based rotational tillage systems for early soybean growth. *Org. Agr.* 9, 471–481. doi: 10.1007/s13165-019-00243-9
- Thelen, K. D., Mutch, D. R., and Martin, T. E. (2004). Utility of interseeded winter cereal rye in organic soybean production systems. *Agron. J.* 96, 281–284. doi: 10.2134/agronj2004.2810
- Travlos, I., Gazoulis, I., Kanatas, P., Tsekoura, A., Zannopoulos, S., and Papastylianou, P. (2020). Key factors affecting weed seeds' germination, weed emergence, and their possible role for the efficacy of false seedbed technique as weed management practice. *Front. Agron.* 2:1. doi: 10.3389/fagro.2020.00001
- Tuck, S. L., Winqvist, C., Mota, F., Ahnström, J., Turnbull, L. A., and Bengtsson, J. (2013). Land-use intensity and the effects of organic farming on biodiversity: a hierarchical meta-analysis. *J. Appl. Ecol.* 51, 746–755. doi: 10.1111/1365-2664.12219
- Uchino, H., Iwama, K., Jitsuyama, Y., Yodate, T., and Nakamura, S. (2009). Yield losses of soybean and maize by competition with interseeded cover crops and weeds in organic-based cropping systems. *Field Crops Res.* 113, 342–351. doi: 10.1016/j.fcr.2009.06.013
- USDA, NRCS (2021). *Web Soil Survey*. Available online at: <http://websoilsurvey.sc.egov.usda.gov/> (accessed October 30, 2021).
- Venables, W. N., and Ripley, B. D. (2002). *Modern Applied Statistics With S, Fourth edition*. New York, NY: Springer.
- Verret, V., Gardarin, A., Pelzer, E., Médiène, S., Makowski, D., and Valantin-Morison, M. (2017). Can legume companion plants control weeds without decreasing crop yield? A meta-analysis. *Field Crops Res.* 204, 158–168. doi: 10.1016/j.fcr.2017.01.010
- Vincent-Caboud, L., Casagrande, M., David, C., Ryan, M. R., Silva, E. M., and Peigné, J. (2019). Using mulch from cover crops to facilitate organic no-till soybean and maize production. A review. *Agron. Sustain. Dev.* 39:45. doi: 10.1007/s13593-019-0590-2
- Vollmer, K. M., Besancon, T. E., Carr, B. L., VanGessel, M. J., and Scott, B. A. (2020). Spring-seeded cereal rye suppresses weeds in watermelon. *Weed Technol.* 34, 42–47. doi: 10.1017/wet.2019.102
- Zadoks, J. C., Chang, T. T., and Konzak, C. F. (1974). A decimal code for the growth stages of cereals. *Weed Res.* 14, 415–421. doi: 10.1111/j.1365-3180.1974.tb01084.x



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EDITED BY

Jose G. Franco,
Agricultural Research Service (USDA),
United States

REVIEWED BY

May M. Wu,
Argonne National Laboratory (DOE),
United States
David Archer,
Agricultural Research Service (USDA),
United States

*CORRESPONDENCE

Ellen Audia
ellen.audia@siu.edu

[†]These authors share last authorship

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Measuring changes in financial and ecosystems service outcomes with simulated grassland restoration in a Corn Belt watershed

Ellen Audia *, Lisa A. Schulte [†] and John Tyndall[†]

Department of Natural Resource Ecology and Management, Iowa State University, Ames, IA, United States

While provisioning ecosystem services generated through agricultural production are high, this often comes at the expense of other ecosystem services. Approaches that support both farm income and a balanced array of ecosystem services are needed. We employed a landscape modeling approach to demonstrate the financial and ecosystem service outcomes of strategically restoring grassland cover within a Corn Belt agricultural watershed. We assessed potential changes associated with a “Baseline” land use scenario and two alternative scenarios for the Grand River Basin (Iowa and Missouri, USA). In a “Buffered” scenario we simulated the impacts of replacing cropland within 20m of streams with restored native grassland cover. In a “Productivity-based” scenario we simulated the replacement of annual row crops on poorly performing croplands with native grassland cover. Grasslands comprised 0.4% of the Baseline scenario. Grassland was expanded to 0.8% of the watershed in the Buffered scenario, reducing annual nutrient and sediment loss by 1.44%, increasing soil carbon sequestration by 0.12% over 10 years, and increasing pollinator abundance by 0.01%. The estimated annual value of these enhancements was \$1.7 million for nitrogen reduction, \$0.1 million for phosphorus reduction, \$0.5 million for sediment reduction, and \$1.3 million for soil carbon sequestration. Grassland comprised 4.9% of the watershed in the Productivity-based scenario, reduced annual nutrient and sediment loss by 11.50%, increased soil carbon sequestration by 1.13% over 10 years, and increased pollinator abundance by 0.42%. The estimated annual value of enhancements was \$18 million for nitrogen reduction, \$1.4 million for phosphorus reduction, \$2.5 million for sediment reduction, and \$14 million for soil carbon sequestration. We also calculated the value of grassland biomass for a potential energy market. The benefit of producing and selling grassland biomass ranged -\$445 to \$1,291 ha⁻¹ yr⁻¹. Scaled to the watershed, annual revenues ranged -\$7.3 million to \$21.1 million for the Buffered scenario and -\$44.2 million to \$128.8 million for the Productivity-based scenario. This study was the first to quantify changes in revenue and the value of ecosystem services associated with grassland restoration in the Grand River Basin and can help inform discussion among watershed stakeholders.

KEYWORDS

continuous living cover, perennial agriculture, bioenergy, environmental benefits, InVEST, natural resource economics, water quality

Introduction

Ecosystem goods and services are the benefits humans derive from nature, which include provisioning (e.g., food), regulating (e.g., water purification), cultural (e.g., recreation and spirituality), and supporting services (e.g., soil formation and nutrient cycling) (MEA, 2005). Agroecosystems have been traditionally designed to produce the provisioning ecosystem goods of food, feed, forage, fiber, bioenergy, and pharmaceuticals (Power, 2010). In the past 60 years, global cropland expansion and green technologies such as high-yielding cultivars, fertilizers and pesticides, and mechanization have enabled cereal yields to increase by 280 percent (Ritchie and Roser, 2019). However, the low diversity, high input agricultural systems that work to maximize crop yields also tend to have negative environmental impacts, including on soil health, water quality, and wildlife habitat, among other impacts (Foley et al., 2005; Power, 2010; Asbjornsen et al., 2014; Liebman and Schulte, 2015). For instance, of the 585 impaired waterbodies in Iowa most are related to bacteria, fish kills, and algal growth, all of which are largely due to agricultural runoff (Iowa Department of Natural Resources, 2020). Another example of this is the hypoxic zone in the Gulf of Mexico, which is a result of nutrient loss from agricultural lands within the Mississippi River Basin (Gulf Hypoxia Action Plan, 2008), and impairs the Gulf region's ability to provide seafood and support tourism. Agriculture's ability to take advantage of emerging environmental markets, such as for flood control, clean water, and carbon reduction can also be limited as it is difficult to accurately measure and value the effects of agricultural conservation practices on ecosystem services and facilitate payments to landowners (Reed, 2020).

To sustain agriculture's traditional role, shore up its unintended negative environmental impact, and support the expansion into new roles, efforts are being made to strategically restore native perennial grassland within the U.S. Corn Belt's annual crop matrix (e.g., Glover et al., 2010; DeLuca and Zabinski, 2011; Hirsh et al., 2013; Schulte, 2014; Zhou et al., 2014). There is a growing body of research supporting the need to maintain continuous living cover in agroecosystems and understand the economic feasibility of doing so (e.g., Schulte et al., 2006, 2017; Meehan et al., 2013; Asbjornsen et al., 2014; Bonner et al., 2014; Zilverberg et al., 2014; Zimmerman et al., 2019).

Perennials can maintain ecosystem services while contributing to existing and emerging markets such as bioenergy, outdoor recreation (e.g., agritourism), hunting, and nutrient and carbon crediting (Meehan et al., 2013; Zilverberg et al., 2014; John and McIsaac, 2017; Powell et al., 2018; Ha and Wu, 2022; Zimmerman et al., 2022).

Grassy feedstocks from restored perennial grassland are being promoted for renewable fuel production. These grassy feedstocks can be grown in areas where annual row crop

production is chronically less profitable and/or areas of high conservation value (Gelfand et al., 2013; Meehan et al., 2013; Brandes et al., 2016; Schulze et al., 2016; Mishra et al., 2019; Khanna et al., 2021; Martinez-Feria et al., 2022). Converting low-yielding cropland to grassland cover has the potential to improve the overall profitability of farm fields (Bonner et al., 2014; Brandes et al., 2018; Nair et al., 2018). The cost of grassland establishment and management tends to be lower than for cash crops, and depending on local or regional market development, perennial systems may out compete annual systems in terms of productivity and profitability (Tilman et al., 2006; Gelfand et al., 2013; Manatt et al., 2013; Brandes et al., 2016). New markets for bioenergy grassland crops could subsequently foster emerging ecosystem and or commodity markets and the ecosystem services associated with grassland systems provide widespread public benefits, such as climate regulation, water purification, and recreational services, some of which can be monetized (Johnson et al., 2012; Meehan et al., 2013; Mishra et al., 2019).

Our goal with this research was to evaluate one method of agroecosystem perennialization – strategically restoring and/or reconstructing grassland composed of native species as a biomass crop (hereafter referred to as bioenergy grassland) – to jointly expand agricultural markets and enhance ecosystem service outcomes in the Grand River Basin (GRB), located in Iowa and Missouri, USA. The GRB was chosen for this study because it represents an agriculturally dominated watershed contributing to water quality impairments in the Mississippi River Basin. While grasslands have a variety of different uses (e.g., grazing land, hay, bedding), the GRB and surrounding region has for decades hosted projects that seek to simultaneously meet bioenergy production and conservation goals through perennial grassland restoration (e.g., Shepherd, 2000; Austin, 2011; Butler, 2019; Prairie Lands, 2022). Our specific objectives for the GRB were to (1) create alternative land use scenarios that meet both bioenergy development and conservation goals; (2) assess the potential ecosystem service outcomes associated with current land use and alternative land use scenarios, specifically impacts to nutrient and sediment retention, carbon sequestration, and pollinator abundance; (3) determine potential private financial and public economic outcomes associated with the current land use and alternative scenarios; and (4) inform agricultural and natural resource decision-making.

Methods

Study area

The Grand River Basin is located in southwest Iowa (38%) and northwest Missouri (62%), USA. The entire watershed has an area of 20,460 km², most of which lies in Missouri

and entirely within the Dissected Till Plains (Pitchford and Kerns, 1999; NRCS, 2006). The topography of the basin is mostly composed of rolling and undulating uplands dissected by broad, flat stream valleys (Pitchford and Kerns, 1999). Shales, sandstones, and limestones underlie the watershed and the predominating soils are silt loams and silty clay loams derived from glacial drift and loess (Pitchford and Kerns, 1999; NRCS, 2006). Based on a weather station in the central area of the basin, Gentry County, average temperatures in the GRB for the period 2000–2022 ranged from 5.3 to 17.6, and annual precipitation ranged from 0 cm to 112 cm (<http://agebb.missouri.edu/weather/stations/>).

Land use in the basin is estimated to be 30% cropland, 44% pasture, 17% forest, 3.7% water/wetland, 4.2% urban, and 0.6% grassland and shrubland (Figure 1). Over 200,000 hectares of the watershed is enrolled in the USDA Conservation Reserve Program (USDA FSA, 2020).

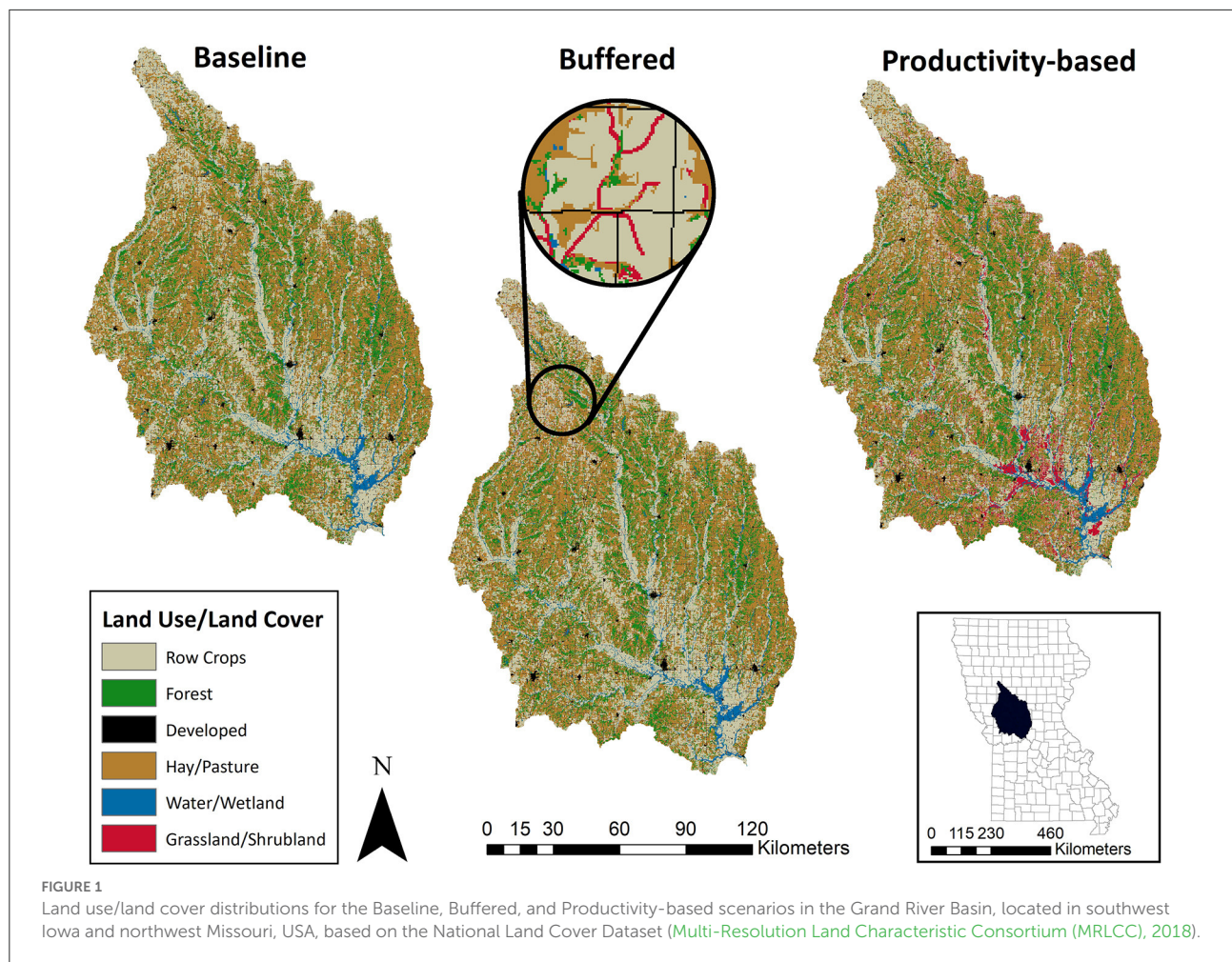
Major environmental impairments within the GRB include water quality degradation and habitat loss (Pitchford and Kerns, 1999). The Section 303 (d) list of impaired waters (category 5) in Iowa includes 585 impairments, six of which occur in the GRB and are bacterial and biological in character (Iowa Department of Natural Resources, 2020). The list in Missouri includes 481 impairments, 11 of which occur in the GRB, with bacteria, dissolved oxygen, mercury in fish, and heavy metals being the reasons for impairment (MoDNR, 2020). Excess nitrogen and phosphorus are also listed as impairments, and 20% of all pollutants come from non-point sources (MoDNR, 2020). Missouri and Iowa are major contributors to nutrient loading in the Mississippi/Atchafalaya River Basin (MARB) (Alexander et al., 2008), with agricultural land contributing between 50–60% of the nutrients (MoDNR, 2014). Iowa and Missouri have both pledged to reduce nutrient loading in response to the 2008 Gulf Hypoxia Action Plan (MoDNR, 2014; Iowa Nutrient Reduction Strategy (INRS), 2017), to shrink the hypoxic dead zone that occurs there (Gulf Hypoxia Action Plan, 2008). Various agricultural best management practices and green infrastructure have been encouraged and implemented to help control nutrient and sediment loss (MoDNR, 2014; Iowa Nutrient Reduction Strategy (INRS), 2017).

Habitat quality and loss are also of concern in the GRB. Native Midwestern landscapes are considered critically endangered as >50% of the native vegetation in the region has been converted to other vegetation types (Hoekstra et al., 2005). Loss of vegetation richness and simplification of landscapes can negatively influence soil formation, erosion control, water retention, nutrient cycling, and habitat quality all of which impact plant and animal biodiversity (Schulte et al., 2006; Power, 2010). In Iowa and Missouri, native prairie has been replaced mainly by agricultural and urban land (Iowa Department of Natural Resources, 2015; Missouri Department of Conservation, 2015). Such land use change is the primary threat to plant and animal biodiversity within grasslands (Hirsh et al.,

2013; Iowa Department of Natural Resources, 2015; Missouri Department of Conservation, 2015). This landscape conversion in Iowa and Missouri has led to very low numbers or even extirpation of many native wildlife species (Iowa Department of Natural Resources, 2015; Missouri Department of Conservation, 2015). Sedimentation, and nutrient loading from runoff, and channelization and levee construction in streams cause aquatic and riparian habitat degradation throughout Iowa and Missouri as well (Iowa Department of Natural Resources, 2015; Missouri Department of Conservation, 2015).

Modeling framework

We employed a five-step methodological approach. We first identified two contrasting land use scenarios – a “Buffered” and a “Productivity-based” scenario – to compare with current land use in the GRB, based on the 2016 National Land Cover Dataset (Multi-Resolution Land Characteristic Consortium (MRLCC), 2018). Descriptions of each NLCD land cover type can be found in the Supplementary material (Supplementary Table S1). For the Buffered scenario, we identified all row-cropped areas within 20 m of a perennial stream and shifted them to bioenergy grassland. For the Productivity scenario, we shifted land use from row crops to bioenergy grassland in areas that have low corn and soybean yield potential based on the National Commodity Crop Productivity Index (NCCPI; Dobos et al., 2012), which we obtained from the gSSURGO database (gSSURGO Database, 2020). Cropland with a NCCPI value of < 0.5 was shifted to bioenergy grassland. We chose 0.5 as the cutoff point because those soils would likely have a history of chronic economic loss. Dobos et al. (2008) present a bivariate fit regression of corn yield from 35 different states and NCCPI and indicate that expected corn yield ranged from 40 bushels per acre to about 160 bushels per acre with an average of about 110 bushels per acre. For context, from 2016 to 2022 the average breakeven corn yield in six Iowa counties (Black Hawk, Fremont, Hamilton, Mills, Tama, and Wright) was 190 bushels per acre (Iowa State University Extension and Outreach (ISUEO), 2022). Furthermore, Li et al. (2016) used crop insurance data from Midwestern states including Iowa and Missouri to indicate that crop insurance losses increase at low to medium NCCPI value ranges and begins to decrease after the 0.65 NCCPI value. Bioenergy grassland was defined as a planted system based on a conservation-oriented prairie seed mix suitable for the region. Specifically, average seed mix costs were based on regional prices for mesic pollinator 10/30 (10 grasses/30 forbs) plantings designed to provide supportive environments for honeybees and butterflies. Second, we used the Integrated Valuation of Ecosystem Services and Tradeoffs model (InVEST; Natural Capital, Project, 2019) model to estimate ecosystem service outcomes in the GRB, including nutrient delivery ratios, sediment delivery, carbon



storage, and pollinator abundance, based on current land use and the Buffered and Productivity-based land-use scenarios. Third, we conducted a comprehensive net present value financial assessment associated with the bioenergy grassland as a source of herbaceous biomass feedstocks. Fourth, ecosystem service and net present value outputs were then combined in a social benefit-to-cost analysis. Finally, we created maps and other graphics to demonstrate bioenergy grassland, ecosystem service, and financial opportunities. Detailed descriptions of data, models, and analysis procedures can be found in the [Supplementary material](#).

Results

Land use/land cover change

The total amount of cropland decreased by 0.4% in the Buffered and 4.5% in the Productivity-based scenarios. A total of 7,743 ha of row-cropped land was converted to grassland vegetation in the Buffered scenario and 91,274 ha in the Productivity-based scenario ([Supplementary Table S2](#)).

Approximately 1.2% of the Buffered area overlapped with the Productivity-based area. These areas of overlap are dispersed throughout the GRB, but occur mainly in the northern and southern portions (see [Supplementary Figure S1](#) in the [Supplementary material](#)). It should be noted that our scenarios did not account for any additional shifts in land use that may well accompany the changes in cropped land that we present. These changes could include converting pasture or non-agricultural land (e.g., treed areas, fence rows, conservation land) to cropped land or adopting less diverse crop rotations. Such changes would likely have impacts on both the economic and environmental outcomes of our overall analysis ([Bonham et al., 2006](#); [Fleming, 2014](#)).

Ecosystem services

Results indicate that integration of bioenergy grassland into either perennial stream buffers or low-yielding cropland increased ecosystem service outcomes ([Table 1](#)). In the Buffered scenario, annual sediment retention increased by 86,088 Mg (0.72% reduction in loss compared to baseline), annual total

TABLE 1 Baseline, buffered, and productivity-based scenario outputs for ecosystem services generated by InVEST (v 3.7.0) in the Grand River Basin, Iowa and Missouri, USA.

Ecosystem service	Baseline	Buffered	Productivity-based
Phosphorous export	2,367	2,355	2,234
Nitrogen export	14,973	14,938	14,594
Sediment export	12,007,418	11,921,329	11,602,739
Sequestered carbon	107,061,426	107,188,520	108,274,417
Pollinator abundance (0–1)	0.1098	0.1099	0.111

Phosphorus, nitrogen, and sediment export represent annual values, carbon sequestration (Mg C) represents sequestration over 10-years, and pollinator abundance is indexed. All other values are in Mg.

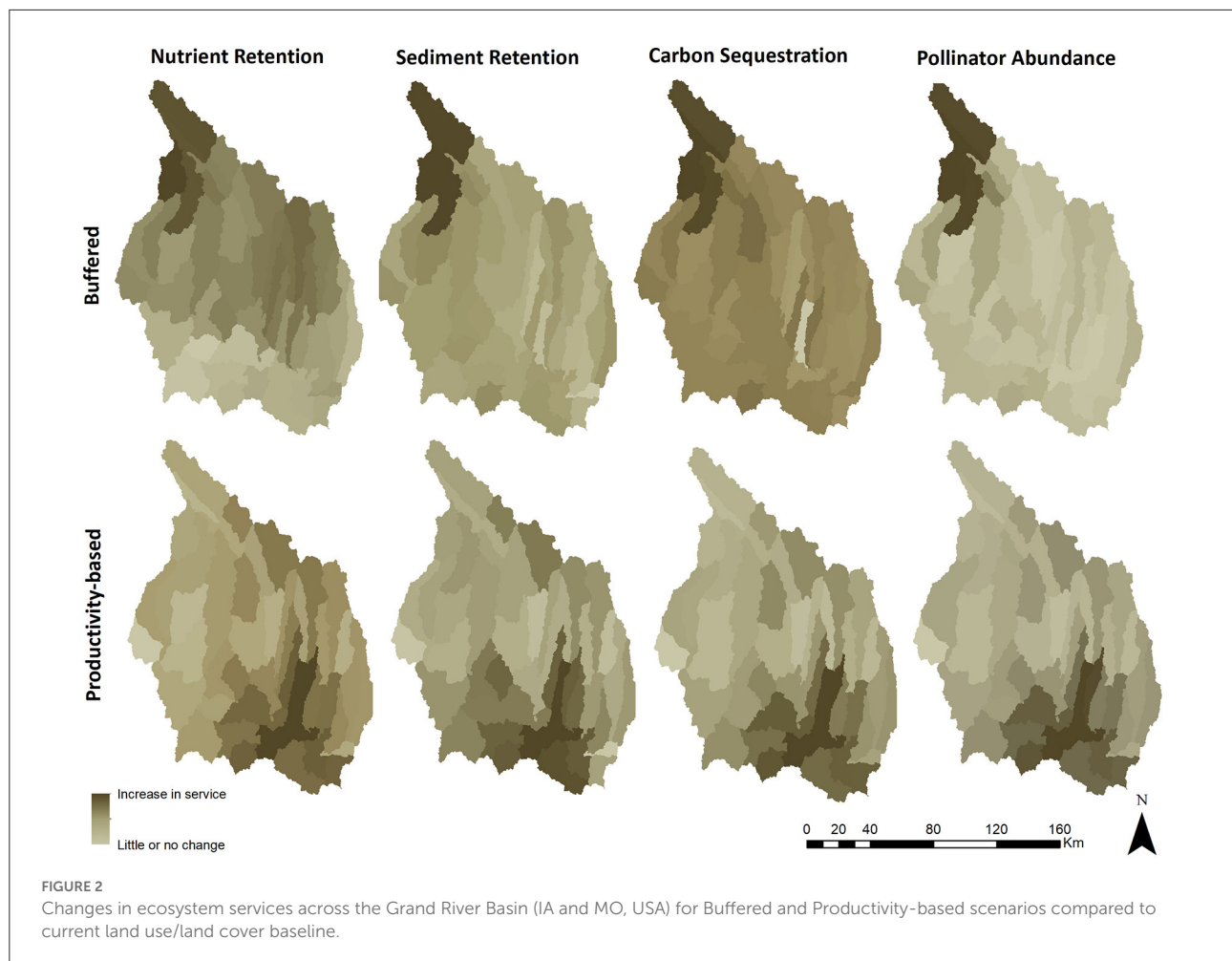
nitrogen retention increased by 35 Mg (0.23% reduction), annual phosphorus retention increased by 11 Mg (0.49% reduction), carbon sequestration increased by 127,093 Mg C over 10 years (0.12% increase), and pollinator abundance increased by 0.01%. In the Productivity-based scenario, relative to baseline, annual sediment retention increased by 404,678 Mg (3.37% reduction), annual nitrogen retention increased by 379 Mg (2.53% reduction), annual phosphorous retention increased by 132 Mg (5.59% reduction), carbon sequestration increased by 1,212,990 Mg C over 10 years (1.13% increase), and pollinator abundance increased by 0.42%. Across all modeled services, the Productivity-based scenario facilitated a greater increase in ecosystem services than the Buffered scenario at the watershed level (Table 1). In the Buffered scenario, the greatest enhancement of ecosystem services occurred in the northern portion of the GRB, while in the Productivity-based scenario the greatest enhancement of ecosystem services occurred in the south-central portion of the GRB and along the Grand River (Figure 2). These relationships are likely caused by the dominance of cropland in the south and along the Grand River, and the high concentration of perennial streams intersecting cropland in the north (Supplementary Figure S1).

We also calculated annual ecosystem service outcomes for each scenario per-ha of restored native grassland (Figure 3). In the Buffered scenario, phosphorous loss was reduced by 1.48 kg ha⁻¹, nitrogen loss was reduced by 4.54 kg ha⁻¹, sediment loss was reduced by 12.26 tons ha⁻¹, and carbon storage increased by 1.64 Mg ha⁻¹ yr⁻¹. In the Productivity-based scenario, phosphorous loss was reduced by 1.45 kg ha⁻¹, nitrogen loss was reduced by 4.15 kg ha⁻¹, sediment loss was reduced by 4.89 tons ha⁻¹, and carbon storage increased by 1.33 Mg ha⁻¹ yr⁻¹.

Economic valuation

We estimated baseline field-level per-ha annualized costs (2022 USD; Supplementary Table S9) and net revenue associated with producing a bioenergy grassland crop in Iowa and Missouri

assuming three discount rates, three different yields, and eight different farmgate selling prices for the biomass. Our cost assessment methods are similar to those utilized in studies that examined perennial cover establishment as either an in-field practice or in riparian areas (e.g., Roberts et al., 2009; Bravard et al., 2022) as well as part of biomass production systems (James et al., 2010; Manatt et al., 2013). The costs and net revenues for bioenergy grassland crop varied depending on scenario due to differences in the opportunity cost of land and the assumed biomass yield which impacted harvesting costs. Opportunity costs of land were calculated by using area-weighted land rent estimates for the counties in the GRB which are largely determined by relevant soil productivity measures; the Corn Suitability Rating in Iowa and the National Commodity Crop Productivity Index in Missouri (Massey and Brown, 2021; Plastina et al., 2022). As noted in Bravard et al. (2022) crop productivity indices are significant factors in determining area land rent as they provide a comparative numerical ranking of soil quality relative to producing a base crop (corn in this case). The average baseline annualized cost of establishing, managing and harvesting a bioenergy grassland crop over a 10-year management horizon in the GRB was estimated to be \$592 ha⁻¹ for the Buffered scenario, and \$588 ha⁻¹ for the productivity scenario (using a real discount rate of 5%; Supplementary Table S10). Production costs also varied with discount rates (7–9%) from \$591 ha⁻¹–\$705 ha⁻¹ (Supplementary Table S10). Here we discuss results that use a 10-year average real discount rate of 5% (Table 2). The results from the other two discount prices can be found in the Supplementary material (Supplementary Tables S12, S13). Breakeven prices vary somewhat between the Buffered scenario and the Productivity-based scenario. For the Buffered scenario, results indicate that a biomass selling price above \$88 Mg⁻¹ will produce positive net revenue given yields ≥ 6.7 Mg ha⁻¹, a selling price above \$50 Mg⁻¹ will produce positive net revenue given yields ≥ 13.5 Mg ha⁻¹, and a selling price above \$35 Mg⁻¹ will produce positive net revenue given yields greater than or equal to 20.2 Mg ha⁻¹ (Table 2). For the Productivity-based scenario, results indicate that a biomass selling price above \$88 Mg⁻¹ will produce positive net revenue given yields ≥ 6.7 Mg ha⁻¹, a selling price above \$49 Mg⁻¹ will produce positive net revenue given yields ≥ 13.5 Mg ha⁻¹, and a selling price above \$35 Mg⁻¹ will produce positive net revenue given yields ≥ 20.2 Mg ha⁻¹ (Table 2). These results reflect assumed static establishment, management and harvesting costs for an acre of bioenergy grassland in Iowa and Missouri (Supplementary Table S9), a range of potential but unmeasured yield outcomes, and the assumption that biomass harvest would not occur until year three of the analytical horizon. We also assume that current average land rent is an adequate measure of the opportunity cost of land. The reality is that actual opportunity costs of land may be higher or lower than land rent in any given year and is dependent upon commodity and production prices (Tyndall et al., 2013).



Nevertheless, because commodity markets are volatile and difficult to predict, somewhat more temporally stable land rent markets are often deemed an acceptable proxy (Tyndall and Roesch, 2014). Because land rent is largely based on inherent soil productivity, the relative comparative findings among scenarios should remain unchanged (Zimmerman et al., 2019). Ultimately, costs and revenue vary temporally and spatially depending on site-level conditions, production practices, weather, policy, and market conditions for biomass but also various inputs such as land, fertilizer, labor, and seed (Tyndall et al., 2021). Individual biomass systems may also experience additional costs associated with prolonged establishment periods or maintaining the health of any given stand of biomass. Some farmers may experience costs associated with whole stand reestablishment in the case of crop failure or hazard damage due to weather (Liu et al., 2011). Furthermore, in the productivity scenario the in-field costs of grass harvest, baling and on-site transportation and storage would likely be highly variable due to the heterogeneous scale, shape and patchy nature of the biomass systems presented (Nair et al., 2017). Likewise, accessibility in riparian areas can be complicated by moisture conditions during harvest periods

and the care often needed to minimize harvest impact in hydrologically sensitive areas (Erdozain et al., 2020). Ultimately the spatial fragmentation of the biomass systems presented in our study would likely carry additional, but unaccounted for logistical and or environmental costs associated with harvesting and on-site handling of biomass materials (Ferrarini et al., 2017).

Total potential biomass yields in the GRB ranged from 58,075 to 174,226 Mg yr⁻¹ in the Baseline scenario, 110,149 to 330,446 Mg yr⁻¹ in the Buffered scenario, and 671,900 to 2,015,701 Mg yr⁻¹ in the Productivity-based scenario. Given these yields and assuming farmgate selling prices between \$22 and \$99 Mg⁻¹, total biomass net annual revenue ranged from -\$7,311,371 and \$21,119,430 in the Buffered scenario, and -\$44,227,025 to \$128,827,270 in the Productivity scenario (Table 3).

We estimated the potential economic value generated from ecosystem service enhancement for each modeled scenario in the GRB. The annualized value for carbon sequestration within the GRB is the highest among all analyzed ecosystem services in both scenarios. In the Buffered scenario, we estimate the annualized value of nitrogen reduction to be

TABLE 2 Per hectare annualized net revenue of establishing, harvesting and selling bioenergy grassland biomass in Iowa and Missouri, USA given yield and scenario dependent annualized production costs at a 5% discount rate.

		Bioenergy grassland biomass price (\$ Mg ⁻¹)							
	Yield (Mg ha ⁻¹)	22	33	44	55	66	77	88	99
Buffered	6.7	(\$445)	(\$371)	(\$297)	(\$224)	(\$150)	(\$76)	(\$3)	\$71
	13.5	(\$378)	(\$230)	(\$81)	\$67	\$216	\$364	\$513	\$661
	20.2	(\$268)	(\$46)	\$176	\$399	\$621	\$843	\$1,065	\$1,287
Productivity-based	6.7	(\$441)	(\$367)	(\$294)	(\$220)	(\$146)	(\$73)	\$1	\$75
	13.5	(\$375)	(\$226)	(\$78)	\$71	\$219	\$368	\$516	\$665
	20.2	(\$264)	(\$42)	\$180	\$402	\$624	\$847	\$1,069	\$1,291

Prices are in 2022 US\$. Yield scenarios were based on [Tilman et al. \(2006\)](#), [James et al. \(2010\)](#), [Manatt et al. \(2013\)](#), and [Nichols et al. \(2014\)](#). Parentheses indicate a negative number.

TABLE 3 Total annual net revenue generated from each modeled scenario in the Grand River Basin assuming yields of 6.7, 13.5, and 20.2 Mg ha⁻¹, farmgate selling prices between \$22 and \$99 Mg⁻¹, and yield and scenario dependent annualized production costs.

Price (\$ Mg ⁻¹)	Buffered			Productivity-based		
	6.7 Mg ha ⁻¹	13.5 Mg ha ⁻¹	20.2 Mg ha ⁻¹	6.7 Mg ha ⁻¹	13.5 Mg ha ⁻¹	20.2 Mg ha ⁻¹
22	(\$7,311)	(\$6,175)	(\$4,325)	(\$44,227)	(\$37,298)	(\$26,382)
33	(\$6,100)	(\$3,752)	(\$690)	(\$36,836)	(\$22,516)	(\$4,209)
44	(\$4,888)	(\$1,328)	\$2,945	(\$29,445)	(\$7,734)	\$17,964
55	(\$3,676)	\$1,095	\$6,580	(\$22,054)	\$7,048	\$40,136
66	(\$2,465)	\$3,518	\$10,215	(\$14,663)	\$21,829	\$62,309
77	(\$1,253)	\$5,941	\$13,850	(\$7,273)	\$36,611	\$84,482
88	(\$42)	\$8,365	\$17,485	\$118	\$51,393	\$106,655
99	\$1,170	\$10,788	\$21,119	\$7,509	\$66,175	\$128,827

Prices are in 2022 US\$ (thousands). Yield scenarios were based on [Tilman et al. \(2006\)](#), [James et al. \(2010\)](#), [Manatt et al. \(2013\)](#), and [Nichols et al. \(2014\)](#). Parentheses indicate a negative number.

\$1,700,529, phosphorus reduction to be \$124,089, sediment reduction to be \$524,921, and carbon sequestration to be \$1,348,229. For the Productivity-based scenario, we estimate the annualized value of nitrogen reduction to be \$18,330,016, phosphorus reduction to be \$1,428,914, sediment reduction to be \$2,467,500, and carbon sequestration to be \$14,294,696. Across all modeled services, the Productivity-based scenario generated more ecosystem service related revenue, but also converted a much larger area out of annual crops. We therefore calculated ecosystem service related revenue per hectare of restored native grassland as well ([Figure 3](#)). In the Buffered scenario, we estimated the annualized value of nitrogen reduction to be \$220 ha⁻¹, phosphorous reduction to be \$16 ha⁻¹, sediment reduction to be \$68 ha⁻¹, and carbon sequestration to be \$174 ha⁻¹, for a combined annual ecosystem service value of \$478 ha⁻¹. For the Productivity-based scenario, we estimated the annualized value of nitrogen reduction to be \$201 ha⁻¹, phosphorous reduction to be \$16 ha⁻¹, sediment reduction to be \$27 ha⁻¹, and carbon sequestration to be \$157 ha⁻¹, for a combined annual ecosystem service value of \$401 ha⁻¹.

Discussion

Ecosystem services

We constructed and modeled two land-use scenarios for the GRB using InVEST (v 3.7.0) to compare ecosystem service enhancement relative to a baseline scenario. Ecosystem services were enhanced in both the Buffered and Productivity-based scenarios, with greater enhancement in the Productivity-based scenario at the watershed scale. This was primarily due to the variation in area converted to bioenergy grassland in each scenario. Perennial vegetation has been proven to reduce nutrient and sediment loss from agricultural fields, store carbon, and increase pollination services ([DeLuca and Zabinski, 2011](#); [Meehan et al., 2013](#); [Asbjornsen et al., 2014](#); [Schulte et al., 2017](#); [Zimmerman et al., 2019](#)), and thus the per unit ecosystem service values assigned to perennial land uses in this model were based on empirical data.

Multiple empirical and modeling studies have determined that increasing perennial vegetation on a landscape will help reduce nutrient and sediment loss to waterbodies (e.g., [Dabney](#)

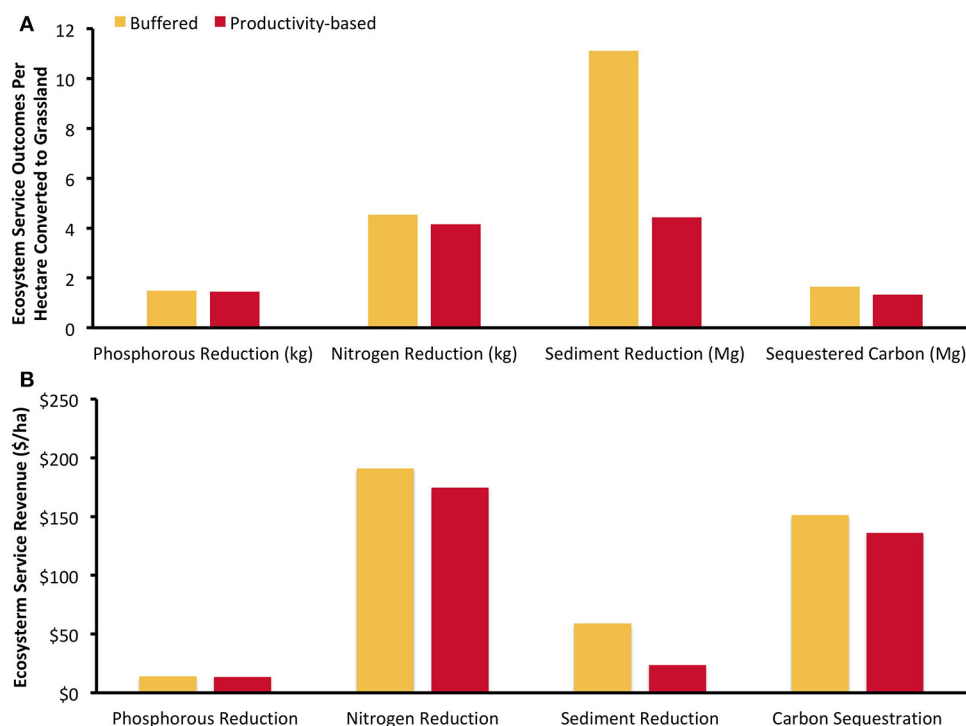


FIGURE 3

Per hectare annualized ecosystem service outcomes in each alternative scenario simulated in the Grand River Basin, Iowa and Missouri, USA (A); phosphorus, nitrogen, and sediment values are loss reduction while the carbon value is an increase. Annualized ecosystem service related revenue also shown (B).

et al., 2001; Vaché et al., 2002; Helmers et al., 2012; Asbjornsen et al., 2014; Zhou et al., 2014; Schulte et al., 2017). Perennial vegetation, with its abundant and complex root system increases the ability of agricultural land to slow, filter, and store water (Hernandez-Santana et al., 2013; Gutierrez-Lopez et al., 2014; Zhou et al., 2014). These findings support our results that increasing grassland cover on a landscape will reduce nutrient and sediment loss to waterbodies. We found that the Productivity-based scenario, which simulated land use change from row crops to bioenergy grassland on a greater number of hectares, more greatly reduced nutrient and sediment loss than the Buffered scenario at the watershed scale. Although this relationship might be true in a modeling context, after a certain point more perennial vegetation on a landscape does not necessarily lead to more or measurable water quality benefits. For example, when planting prairie strips within row-cropped catchments in Iowa, Schulte et al. (2017) found little difference in water quality measures between fields with 10% prairie vs. 20% prairie indicating that impacts were not a function of area converted. Dosskey et al. (2002), Dosskey et al. (2005) recorded similar findings regarding the effects of perennial vegetation on water quality. There can also be significant lags associated with the implementation of conservation practices and water quality benefits (Meals et al., 2010), which we did not incorporate in this

analysis. Our results, thus, represent reductions in nutrients and sediment independent of time.

When all ecosystem services were measured per unit area of restored grassland, the Buffered scenario performed more efficiently than the Productivity-based scenario. In terms of nutrient and sediment reduction, this is likely due to how grassland was restored on the landscape as well as how InVEST measures nutrient and sediment export. InVEST in part determines nutrient and sediment export of a given pixel in the landscape by measuring the size (or length), slope, and vegetation's retention efficiency and maximum retention efficiency (nutrient export) or C-factor (sediment export) of the area upslope (or downslope) from that pixel to understand the nutrient delivery ratio or soil erosion probability of that pixel. The difference in placement of native grassland between the Buffered and Productivity-based scenario is likely the main reason for the difference in area based nutrient and sediment reduction. While field scale and downstream water quality management practices are spatially explicit in their effectiveness (Rao et al., 2009; Schulte et al., 2017), the area-based differences in nutrient reduction between scenarios were small ($<1 \text{ kg ha}^{-1}$). The difference in per hectare sediment reduction is large (7 tons ha^{-1}), and should be more carefully considered. Multiple factors related to site history, topography, and soil

affect whether an in-field or edge-of-field practice will be most effective at generating conservation outcomes on any particular field (Tomer et al., 2013). However, multiple studies have found that while perennial vegetation applied within fields keeps nutrients and soil in place (Helmert et al., 2012; Zhou et al., 2014), also placing it within riparian areas can further enhance water quality (Vaché et al., 2002; Dabney et al., 2006). It should be noted that approximately 1.2% of the buffered area was located in areas of low productivity that were also converted to herbaceous vegetation in the Productivity-based scenario. These areas of overlap were mostly located in the northern and southern portions of the GRB (Supplementary Figure S1), and less in the central portion.

Compared to other studies, the reductions in nitrogen and phosphorous we present in the GRB for each land use scenario are conservative. For example, the Iowa Nutrient Reduction Strategy determined that streamside buffers have the ability to reduce phosphorous by 18% and nitrate-nitrogen by 7%, and within-field perennial crops could reduce phosphorous by 29% and nitrate-nitrogen by 18% (depending on area of land converted) (Iowa Nutrient Reduction Strategy (INRS), 2017). Helmert et al. (2011) found restoring only 5% more of a catchment to prairie strips resulted in over 70% reductions in total nitrogen and phosphorous. In terms of per-hectare reductions, Tyndall et al. (2013) cite that prairie strips can reduce phosphorous by 7.32 kg ha^{-1} and nitrogen by 31.93 kg ha^{-1} . These values are much higher than the per hectare values we present in our scenarios: the Productivity-based scenario restored 4.5% of the GRB to grassland (91,274 ha), resulting in a 2.5–5.6% increase in nutrient retention of the landscape. We do not model the use of prairie strips in our analysis, which are likely better at reducing nutrients due to their spatially explicit placement within field vs. within-watershed as in our case.

Polasky et al. (2011) also used InVEST to simulate multiple alternate land cover scenarios for the state of Minnesota, in which one scenario restored marginal agricultural lands and 100 m of lands along all streams to pre-settlement vegetation (i.e., open water, restored forest, restored grassland, restored wetland, and unknown restored cover). Polasky et al. (2011) found that restoring only ~3% of the land in the state could reduce phosphorous export by 34%. One difference between their study and ours is that they restored lands to pre-settlement vegetation, which included a mix of vegetation types including grassland, wetland, savanna, woodland, and forest, while we restored lands to perennial grassland. Regardless, InVEST modules are designed only to understand the effects of land management (see Hamel, 2014 for model assessment) and results are often sensitive to geographic location, spatial scale, input data (i.e., precipitation, export coefficients), and the resolution of those data sets (Salata et al., 2017; Benez-Secanho and Dwivedi, 2019). For example, Redhead et al. (2018) found that InVEST's nutrient reduction model performed well in terms of the relative

magnitude of nitrogen and phosphorus export, but could over or underestimate actual nutrient export by as much as 65%.

Our findings suggest the need for either land use conversion to a much greater amount of continuous living cover and/or the need to include and combine multiple practices (e.g., prairie strips) in order to meet water quality targets for Iowa and Missouri (see Zimmerman et al., 2019). However, incorporating more spatially explicit practices such as prairie strips into our modeling framework and/or using a more robust model like SWAT would only be possible for an investigation considering a different spatial extent, given the distribution of stream gauge stations and monitoring data available in the GRB. Further analyses are also needed that incorporate impacts to subsurface flow and tile drainage through the use of conservation practices that impact those flows (i.e., wetlands, bioreactors, etc.).

Carbon storage was also enhanced with both scenarios compared to the Baseline. The higher amount of carbon stored in the Productivity-based scenario at the watershed level is again due to the larger relative area of bioenergy grassland. Others have demonstrated the potential for perennial vegetation to store carbon below-ground (DeLuca and Zabinski, 2011; Liebman et al., 2014). Compared to certain row crops, perennial species have much more extensive root systems that stay in the ground all year allowing for higher levels of belowground biomass and carbon storage (DeLuca and Zabinski, 2011; Whitmore et al., 2015). For instance, Guzman and Al-Kaisi (2010) found that tallgrass prairie remnant in Iowa had 86% greater belowground carbon storage than a corn-soy rotation, and Glover et al. (2010) found perennial grasslands in Kansas to have 43 Mg ha^{-1} more soil carbon than annual wheat fields. The per unit values of belowground carbon storage assigned to each land cover type in this analysis were based on empirical data that demonstrate this difference. Since InVEST simply aggregates these per pixel values, scenarios with more potential to store carbon will have greater sequestration as is seen in the Productivity-based scenario. When considered per hectare, the Buffered scenario stored slightly more carbon ($\sim 0.3 \text{ Mg ha}^{-1}$). It should be stated that InVEST does not incorporate changes in the rate of carbon storage through time and, eventually, the rate decreases as an equilibrium is reached. The timeline for when this equilibrium is reached depends on climate, soil type, land use, and other factors and occurs over time horizons >50 years (Paustian and Cole, 1998). In this analysis we converted depleted cropland in which soil carbon stocks are likely diminished compared to soils that have never been cultivated. Therefore, in the relatively short 10-year period in which we simulated carbon storage the relationship between increased grassland vegetation and carbon storage was likely linear (Paustian and Cole, 1998). Still, because estimates from this portion of the analysis simply represent potential, we did not consider eventual equilibrium within the landscape. Regardless, with support from a wide body of research (Power, 2010; e.g., Guzman and Al-Kaisi, 2010; DeLuca

and Zabinski, 2011; Whitmore et al., 2015), this analysis reveals opportunities to increase the carbon storage potential of the GRB's annual row-crop matrix through the implementation of perennial grassland.

Pollinator abundance as a function of habitat suitability was also modeled in the GRB under each land use scenario. The index of pollinator abundance in the baseline scenario was very low at only 0.1098. Although the pollinator abundance index increased in both alternative scenarios, gains were minor (Table 1). We included three bee genera (*Lasioglossum*, *Melissodes*, and *Agopostemon*) that are commonly found in Midwestern agricultural landscapes like the GRB, as well as Monarch butterflies (*Danaus plexippus*) as a charismatic insect in conservation need. Monarchs have declined by 80% in North America over the past two decades due to habitat loss (Iowa Monarch Conservation Consortium (IMCC), 2018). Low numbers of pollinators have been reported in highly disturbed and fragmented landscapes (Winfree et al., 2009; Potts et al., 2010; Kennedy et al., 2013) of the U.S. Corn Belt. Foraging resources are often lacking in agriculturally dominated landscapes (Hellerstein et al., 2017) like the GRB (Table 1), and it is therefore not surprising that pollinator abundance estimates were low in the Baseline scenario. The marginal change in pollinator abundance between alternative scenarios and Baseline reflect the relatively small – even if strategic – land use changes incorporated into our scenarios. They also might lie in the complex relationships between pollinators and the landscape that InVEST cannot capture. Models that relate land cover and pollinator abundance across large landscapes are often simplified and limited, a possible result of data coarseness or an incomplete understanding of ecological relationships (Cunningham et al., 2018; Sharp et al., 2018). Many pollinators respond to fine scale landscape features that are not reflected in existing LULC data and/or are challenging to incorporate into models (Sharp et al., 2018).

The InVEST pollinator model assumes resource and nesting availability is uniform across each land cover type (Sharp et al., 2018). In reality, land cover types can vary greatly across space and time due to differences in soil, climate, topography, and management decisions. Additional experimental work is needed to inform landscape and watershed estimations. However, not having enough perennial vegetation on the landscape or in the right places might also explain these marginal changes in abundance. Bennett et al. (2014), using different methods, also measured pollinator abundance in southeast Michigan under two land use scenarios and found that converting 600,000 ha (70%) of agricultural land to perennial bioenergy crops could increase pollinator abundance by 600% in some areas. This might suggest that in order to more greatly increase pollinator abundance in the GRB, more than 91,274 ha of bioenergy grassland would need to be planted, and likely in arrangements that better support their food and nesting needs. Further

analyses should compare InVEST to other pollinator models and experiment with more land use scenarios and arrangements.

The greatest opportunities for ecosystem service enhancement occurred primarily in the south-central part of the GRB when targeting low productivity cropland, and in the northern part of the GRB when targeting riparian areas of perennial streams. These spatial relationships exist because most of the row-cropped land in the GRB is along the Grand River and concentrated around its outlet at the bottom of the basin, and the majority of the perennial streams that intersect row-cropped lands are in the northern part of the basin (Figure 2). Incorporating bioenergy grassland within these areas will enhance ecosystem services, and in the Productivity-based scenario it will do so while minimizing costs. Although they are extensive within the GRB (Supplementary Table S2), we did not explore converting pasture or haylands within the GRB to bioenergy land uses. Including these lands in the analysis would likely change the flow of ecosystem services across the basin and offer different opportunities for conservation.

Economic valuation

Private and public economic benefits quantified in this analysis revealed opportunities for enhancement. We demonstrate that the Productivity-based scenario has the capacity to generate more revenue privately than the Buffered scenario, as well as generate more watershed level ecosystem services. While markets for grassland biomass and for ecosystem services are not yet robust, our results indicate that there is potential for both in the GRB. It should be noted, however, that similar land uses such as hay land and land enrolled in CRP may be in competition with bioenergy grasslands in the GRB. Thus, the interests of stakeholders of various land use options should be considered by partnerships working toward creating bioeconomic opportunities. Current data suggest that, in 2016, much of the existing grassland/shrubland in the GRB was likely enrolled in CRP, and a significant portion (44.1%; Supplementary Table S2) of the watershed was devoted to hay and pasture land. Within row-crop plantings of bioenergy grassland may be able to merge well into these existing markets, and as CRP contracts expire or become limited new opportunities for bioenergy grassland could emerge.

Private economic benefits related to the production and sale of bioenergy grassland biomass on low-yielding row-cropped lands in the GRB exist in certain market contexts. We determined that the costs of producing a hectare of low-input bioenergy grassland on low-yielding row-cropped land in the study region ranged from \$588 to \$712 annually (depending on expected yield and scenario), and could create a positive net revenue depending on yield and biomass price. For example, in the productivity scenario a positive net revenue occurs at a

yield of 6.7 Mg ha⁻¹ and selling prices at or above \$88 Mg⁻¹. For higher expected yields, lower biomass prices will result in a positive net revenue (Table 2). Based on a sensitivity analysis relative to discount rates, the findings did not vary substantially across higher rate of return expectations (e.g., 7 and 9% real rate of return; Supplementary Tables S12, S13). In this analysis we defined bioenergy grassland to be a mix of perennial native grasses and forbs suitable for conservation outcomes but also annual harvest. Tilman et al. (2006) reported yields of multi-species grassland biomass on degraded lands with low fertility to be 3.7 Mg ha⁻¹, which is significantly lower than yields of mixed species that we suggest could create revenue. However, some studies have found higher yields of mixed perennial species with the use of fertilizer. For example, Daigh et al. (2015) found biomass yields of prairie planted in experimental plots in central Iowa to be 5 Mg ha⁻¹ without fertilizer and 7.6 Mg ha⁻¹ with fertilizer. Nichols et al. (2014) similarly measured biomass of a 31-species tallgrass prairie planted in experimental plots on high quality Midwestern agricultural land and found yields of 7.4 Mg ha⁻¹ without fertilizer and 10.4 Mg ha⁻¹ with fertilizer. Other studies have reported higher yields of single perennial grasses on marginal lands. Schmer et al. (2008) reported switchgrass yields as high as 11 Mg ha⁻¹ on low-yielding Midwest cropland, and Brandes et al. (2018) use a maximum switchgrass yield of 10 Mg ha⁻¹ within Iowa agricultural fields. Furthermore, yield variability across time at any given site due to weather and changes in management add considerable uncertainty to any multi-year biomass assessment (Sharma et al., 2020). This suggests that even with fertilizer boosting biomass yields to maximum levels noted in the literature, relatively high biomass prices (e.g., somewhere above \$71 Mg⁻¹) will likely be necessary to generate positive net revenue against variable biomass yield.

Our analysis indicates that, depending on the scenario, at yields of 6.7 Mg ha⁻¹, prices will need to be at least \$88 Mg⁻¹ to generate positive revenue. Recent research points to potentially high break-even farmgate prices. James et al. (2010) found the break-even selling price for mixed perennial grasses grown in southern Michigan and southcentral Wisconsin to be \$130 Mg⁻¹ and others have reported similar selling prices needed for perennial biomass to be profitable (Khanna et al., 2008; Manatt et al., 2013). In our analysis, higher yields would allow for positive revenue at much lower selling prices (i.e., \$55 Mg⁻¹ at 13.5 Mg ha⁻¹ and \$44 Mg⁻¹ at 20.2 Mg ha⁻¹); however, it may not be possible to attain those yields on cropland with relatively low productivity ratings as represented in our study (e.g., <=0.5 NCCPI rating).

Beyond questions about yield capacity and consistency across time and space, there are critical unanswered questions regarding institutional, market, infrastructural, and social-psychological barriers to farmer and/or landowner adoption of grass-based cropping systems in and around their farm systems. The financial viability of a biomass crop depends upon robust

and sustainable bioenergy markets capable of paying unit prices needed to generate consistent net-revenues for landowners while remaining a cost-competitive fuel stock. Historically, broad-scale biofuel biomass markets in the United States have failed to emerge despite legislative mandates (e.g., the 2007 Renewable Fuel Standard; RSF2) and government subsidy programs with regard to research and development (e.g., Ebadian et al., 2020) and local to regional market development (e.g., Miao and Khanna, 2017). There are multiple, interacting factors that have impeded the emergence of regional renewable energy systems centered on biomass fuel. These factors include techno-economic limitations of biofuel industrial processes (Padella et al., 2019), costly supply chain management particularly high handling and transportation costs (Yang et al., 2022), inadequate national and state policy linkages (Miao and Khanna, 2017; McCarty and Sesmero, 2021), and limited landowner and farmer buy-in. Landowners struggle with the legacy of failure regarding bioenergy emergence and continue to question the lack of market development and infrastructure in their regions (Hart et al., 2018). There are additional pragmatic questions regarding the availability of on-farm equipment, compatibility of biomass production to primary cropping systems, land tenure constraints, and available technical support (Khanna et al., 2021). There is however evidence that some landowners would be interested in biomass systems that serve simultaneous production and conservation goals similar to those demonstrated in this study (Hand and Tyndall, 2018).

Along with private economic benefits, converting commodity crops to perennial vegetation can enhance ecosystem services that translate into public economic benefits. We show that in each scenario social benefits are generated from the reduction of water and air pollutants. Water quality values are based on the per unit costs of nitrogen, phosphorus, and sediment related impairments to recreational areas, real estate, and other amenities such as drinking water (more detailed information can be found in the Supplementary material). We valued the reduction of these pollutants to be approximately \$2 million per year in the Buffered scenario and \$22 million per year in the Productivity-based scenario. Others have valued the enhancement of water quality in similar ways and reported high economic benefit as well (Meehan et al., 2013; e.g., Mishra et al. (2019). Meehan et al. (2013) valued a 29% reduction in phosphorus loading from replacing 16% of corn rotations with perennial-grasses in a 400,000-ha watershed in southern Wisconsin at almost \$30 million using a combination of ongoing state-level water quality program payments, estimated avoided phosphorus treatment costs of pollution, and data from a survey that quantified household willingness to pay for reductions in phosphorus loading in surface water.

When calculated per hectare, the Buffered scenario generated more public economic benefits in terms of enhancing

water quality (nitrogen and sediment reductions). Therefore, given our data and methods, riparian buffers might be considered more effective at enhancing water quality for public use than our simulated within-field practice. Zhou et al. (2009) tested the effectiveness and economic benefits (including social costs of sediment erosion) of three conservation practices and tillage systems for sediment reduction in the Major Land Resource Areas (MLRA) of Iowa. Although Zhou et al. (2009) did not specifically test riparian buffers against in-field prairie blocks; they did find that when the costs of sediment erosion were considered, conservation practices could outperform each other depending on MLRA and tillage system. We did not compare results in relation to areas of greater erosion or runoff potential. Doing so could provide further insight into where within the GRB within-field practices are more effective than riparian buffers. In addition, a closer look at social cost implications of the two scenarios together might provide further insight.

Carbon sequestration also produced considerable watershed-level public economic benefit at over \$1.3 million per year in the Buffered scenario and over \$14 million per year in the Productivity-based scenario. These estimates were based on the social cost of carbon given a market discount rate of 7% over a 10-year period (Tol, 2009; Polasky et al., 2011; Sharp et al., 2018). Other studies have calculated potential benefits generated from sequestering carbon using the same method and produced comparable results (Polasky et al., 2011; Johnson et al., 2012; Mishra et al., 2019). For example, Johnson et al. (2012) determined that using perennial vegetation in different amounts within agricultural riparian areas in the Minnesota River Basin generated millions of dollars' worth of climate regulation benefits that varied proportionally with the amount of land converted.

Many scientists have explored the possibility of using perennial vegetation to jointly improve ecosystem services and enhance economic benefits (Meehan et al., 2013; e.g. Blanco-Canqui, 2016; Woodbury et al., 2018). For example, Woodbury et al. (2018) determined that replacing corn with switchgrass in Maryland reduced nitrogen loading and created revenue for growers through ecosystem service enhancement and biomass sales. Conversely, Meehan et al. (2013) determined that although buffering streams with perennial grasses led to higher energy production and ecosystem service enhancement than corn rotations, it did not generate as much income for growers. Meehan et al. (2013) did find that incorporating the societal value of ecosystem service enhancement that is generated from replacing corn with switchgrass outweighs the decrease in grower income; however, markets would need to exist to generate private opportunities associated with biomass production, as well as connect societal benefits to those growers. For example, Mishra et al. (2019) found that planting bioenergy feedstock such as switchgrass on marginal cropland in the Midwest could be better incentivized through both biomass sales

and ecosystem service payments. While markets that facilitate the transfer of these economic benefits back to growers are quickly developing (Salzman et al., 2018), we did not include ecosystem service payments into this analysis. Doing so might alleviate the need for high biomass yields or selling prices. Future analyses should incorporate this financial tradeoff. We did explore the potential multifunctionality of bioenergy grassland to provide income to growers and generate public economic benefits through enhanced ecosystem services. Our findings suggest that incorporating perennial grasses and forbs within agricultural fields will create societal value through reductions in water and air pollutants. These perennial plantings may also be able to provide income to farmers if more robust markets were to develop, given certain biomass yields.

Conclusions

This study was the first to quantify potential changes in revenue and the value of multiple ecosystem services associated with grassland restoration in the GRB. We quantify how replacing annual row crops with perennial grasslands in riparian areas and on lower-yielding portions of the agricultural landscape has the ability to jointly enhance water quality, pollinator abundance, and carbon storage in the GRB. Grassland restoration provides a suite of public economic benefits through the reduction of pollutant-related costs, and could provide private economic benefits through the sale of harvested perennial biomass if robust markets were to develop. Our work informs discussion among watershed stakeholders by quantifying potential tradeoffs among alternative land use decisions at varying spatial scales. Future analyses could test other land use scenarios (e.g., cover crops, pastureland) and more ecosystem services (e.g., habitat quality, recreation) to better understand a fuller suite of ecosystem services associated with grassland cover and the potential for layering economic enhancement in the GRB and similar watersheds.

Data availability statement

The original contributions presented in the study are included in the article/Supplementary material, further inquiries can be directed to the corresponding author/s.

Author contributions

EA, LS, and JT contributed to conception and design of the study. EA collected all the data, performed the spatial modeling and statistical analysis, and wrote the first draft of the manuscript. LS and JT helped write and edit sections of the manuscript. All authors contributed to manuscript revision, read, and approved the submitted version.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

References

- Alexander, R. B., Smith, R. A., Schwarz, G. E., Boyer, E. W., Nolan, J. V., and Brakebill, J. W. (2008). Differences in phosphorus and nitrogen delivery to the Gulf of Mexico from the Mississippi River Basin. *Environ. Sci. Technol.* 42, 822–830. doi: 10.1021/es0716103
- Asbjornsen, H., Hernandez-Santana, V., Liebman, M., Bayala, J., Chen, J., Helmers, M., et al. (2014). Targeting perennial vegetation in agricultural landscapes for enhancing ecosystem services. *Renew. Agric. Food Syst.* 29, 101–125. doi: 10.1017/S1742170512000385
- Austin, A. (2011). *Show Me Energy Gets Grant for Bioenergy Plant Study*. Available online at: <http://biomassmagazine.com/articles/5935/show-me-energy-gets-grant-for-bioenergy-plant-study> (accessed March 12, 2022).
- Benez-Secanho, F. J., and Dwivedi, P. (2019). Does quantification of ecosystem services depend upon scale (Resolution and Extent)? A case study using the invest nutrient delivery ratio model in Georgia, United States. *Environments* 6, 52. doi: 10.3390/environments6050052
- Bennett, A. B., Meehan, T. D., Gratton, C., and Isaacs, R. (2014). Modeling pollinator community response to contrasting bioenergy scenarios. *PLoS ONE* 9, e110676. doi: 10.1371/journal.pone.0110676
- Blanco-Canqui, H. (2016). Growing dedicated energy crops on marginal lands and ecosystem services. *Soil Sci Soc Am J.* 80, 845–858. doi: 10.2136/sssaj2016.03.0080
- Bonham, J. G., Bosch, D. J., and Pease, J. W. (2006). Cost-effectiveness of nutrient management and buffers: comparisons of two spatial scenarios. *J. Agric. Appl. Econ.* 38, 17–32. doi: 10.1017/S1074070800022045
- Bonner, I. J., Cafferty, K. G., Muth, D. J., Tomer, M. D., James, D. E., Porter, S. A., et al. (2014). Opportunities for energy crop production based on subfield scale distribution of profitability. *Energies* 7, 6509–6526. doi: 10.3390/en7106509
- Brandes, E., McNunn, G. S., Schulte, L. A., Bonner, I. J., Muth, D. J., Babcock, B. A., et al. (2016). Subfield profitability analysis reveals an economic case for cropland diversification. *Environ. Res. Lett.* 11, 014009. doi: 10.1088/1748-9326/11/1/014009
- Brandes, E., Plastina, A., and Heaton, E. A. (2018). Where can switchgrass production be more profitable than corn and soybean? An integrated subfield assessment in Iowa, USA. *GCB Bioenerg* 10, 473–488. doi: 10.1111/gcbb.12516
- Bravard, E. E., Zimmerman, E., Tyndall, J. C., and James, D. (2022). The agricultural conservation planning framework financial and nutrient reduction tool: A planning tool for cost effective conservation. *J. Environ. Qual.* 51, 670–682. doi: 10.1002/jeq2.20345
- Butler (2019). *Prairie Power: Roeslein Alternative Energy moves toward Horizon 2*. Available online at: <https://roesleinalternativeenergy.com/prairie-power-roeslein-alternative-energy-moves-toward-horizon-2/> (accessed March 12, 2022).
- Cunningham, C., Tyedmers, P., and Sherren, K. (2018). Primary data in pollination services mapping: potential service provision by honey bees (*Apis mellifera*) in Cumberland and Colchester, Nova Scotia. *Int. J. Biodivers. Sci. Ecosyst. Serv. Manag.* 14, 60–69. doi: 10.1080/21513732.2017.1417331
- Dabney, S. M., Delgado, J. A., and Reeves, D. W. (2001). Using winter cover crops to improve soil and water quality. *Commun. Soil. Sci. Plant Anal.* 32, 1221–1250. doi: 10.1081/CSS-100104110
- Dabney, S. M., Moore, M. T., and Locke, M. A. (2006). Integrated management of in-field, edge-of-field, and after-field buffers. *J. Am. Water Resour. Assoc.* 42, 15–24. doi: 10.1111/j.1752-1688.2006.tb03819.x
- Daigh, A. L., Zhou, X., Helmers, M. J., Pederson, C. H., Horton, R., Jarchow, M., et al. (2015). Subsurface drainage nitrate and total reactive phosphorus losses in bioenergy-based prairies and corn systems. *J. Environ. Qual.* 44, 1638–1646. doi: 10.2134/jeq2015.02.0080
- DeLuca, T. H., and Zabinski, C. A. (2011). Prairie ecosystems and the carbon problem. *Front. Ecol. Environ.* 9, 407–413. doi: 10.1890/100063
- Dobos, R., Sinclair, H. Jr., and Robotham, M. (2012). *User Guide for the National Commodity Crop Productivity Index*. Washington, DC: US Department of Agriculture–National Resources Conservation Service.
- Dobos, R., Sinclair, R., and Hipple, K. (2008). *NCCPI National Crop Productivity Index*. Nebraska: USDA-NRCS National Soil Survey Center Lincoln.
- Dosskey, M. G., Eisenhauer, D. E., and Helmers, M. J. (2005). Establishing conservation buffers using precision information. *J. Soil Water Conserv.* 60, 349–354. Available online at: <https://www.jswnonline.org/content/57/6/336.short>
- Dosskey, M. G., Helmers, M. J., Eisenhauer, D. E., Franti, T. G., and Hoagland, K. D. (2002). Assessment of concentrated flow through riparian buffers. *J. Soil Water Conserv.* 57, 336–343. Available online at: <https://www.jswnonline.org/content/60/6/349.short>
- Ebadian, M., van Dyk, S., McMillan, J. D., and Saddler, J. (2020). Biofuels policies that have encouraged their production and use: An international perspective. *Ener. Policy* 147, 111906. doi: 10.1016/j.enpol.2020.111906
- Erdozain, M., Emilson, C. E., Kreutzweiser, D. P., Kidd, K. A., Mykytczuk, N., and Sibley, P. K. (2020). Forest management influences the effects of streamside wet areas on stream ecosystems. *Ecol. Appl.* 30, e02077. doi: 10.1002/eap.2077
- Ferrarini, A., Serra, P., Almagro, M., Trevisan, M., and Amaducci, S. (2017). Multiple ecosystem services provision and biomass logistics management in bioenergy buffers: a state-of-the-art review. *Renew. Sustain. Energy Rev.* 73, 277–290. doi: 10.1016/j.rser.2017.01.052
- Fleming, D. A. (2014). Slippage effects of land-based policies: Evaluating the Conservation Reserve Program using satellite imagery. *Pap. Reg. Sci.* 93, S167–S178. doi: 10.1111/pirs.12049
- Foley, J. A., DeFries, R., Asner, G. P., Barford, C., Bonan, G., Carpenter, S. R., et al. (2005). Global consequences of land use. *Science* 309, 570–574. doi: 10.1126/science.1111772
- Gelfand, I., Sahajpal, R., Zhang, X., Izaurrealde, R. C., Gross, K. L., and Robertson, G. P. (2013). Sustainable bioenergy production from marginal lands in the US Midwest. *Nature* 493, 514–517. doi: 10.1038/nature11811

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Supplementary material

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- Glover, J. D., Culman, S. W., DuPont, S. T., Broussard, W., Young, L., Mangan, M. E., et al. (2010). Harvested perennial grasslands provide ecological benchmarks for agricultural sustainability. *Agric. Ecosyst. Environ.* 137, 3–12. doi: 10.1016/j.agee.2009.11.001
- gSSURGO Database (2020). *Soil Survey Staff. Gridded Soil Survey Geographic (gSSURGO) Database for Iowa and Missouri*. United States Department of Agriculture, Natural Resources Conservation Service. Available online at: <https://gdg.sc.egov.usda.gov/> (accessed September 21, 2020).
- Gulf Hypoxia Action Plan (2008). *Gulf Hypoxia Action Plan 2008*. Available online at: https://www.epa.gov/sites/production/files/2015-03/documents/2008_8_28_msbasin_ghap2008_update082608.pdf (accessed May 25, 2022).
- Gutierrez-Lopez, J., Asbjornsen, H., Helmers, M., and Isenhardt, T. (2014). Regulation of soil moisture dynamics in agricultural fields using strips of native prairie vegetation. *Geoderma* 226, 238–249. doi: 10.1016/j.geoderma.2014.02.013
- Guzman, J. G., and Al-Kaisi, M. M. (2010). Soil carbon dynamics and carbon budget of newly reconstructed tall-grass prairies in south central Iowa. *J. Environ. Qual.* 39, 136–146. doi: 10.2134/jeq2009.0063
- Ha, M., and Wu, M. (2022). Environmental and cost benefits of multi-purpose buffers in an agricultural watershed for biomass production. *Biofuel Bioprod. Biorefin.* 16, 228–243. doi: 10.1002/bbb.2311
- Hamel, P. (2014). *Uncertainty Analysis of the InVEST 3.0 Nutrient Model: Case Study of the Cape Fear Catchment, NC*. Available online at: https://naturalcapitalproject.stanford.edu/sites/g/files/sbiybj9321/f/publications/uncertainty_analysis_of_the_invest_3.0_nutrient_model.pdf (accessed May 25, 2022).
- Hand, A. M., and Tyndall, J. C. (2018). A qualitative investigation of farmer and rancher perceptions of trees and woody biomass production on marginal agricultural land. *Forests* 9, 724. doi: 10.3390/f9110724
- Hart, N. M., Townsend, P. A., Chowyuk, A., and Gustafson, R. (2018). Stakeholder assessment of the feasibility of poplar as a biomass feedstock and ecosystem services provider in Southwestern Washington, USA. *Forests* 9, 655. doi: 10.3390/f9100655
- Hellerstein, D., Hitaj, C., Smith, D., and Davis, A. (2017). *Land use, Land Cover, and Pollinator Health: A Review and Trend Analysis*. Available online at: <https://www.ers.usda.gov/publications/pub-details/?pubid=84034> (accessed May 25, 2022).
- Helmers, M., Zhou, X., Asbjornsen, H., Kolka, R., and Tomer, M. (2011). "Water quality benefits of perennial filter strips in row-cropped watersheds," in *Proceedings of the 23rd Annual Integrated Crop Management Conference* (Ames, IA: Iowa State University), 139–144. doi: 10.31274/icm-180809-270
- Helmers, M. J., Zhou, X., Asbjornsen, H., Kolka, R., Tomer, M. D., and Cruse, R. M. (2012). Sediment removal by prairie filter strips in row-cropped ephemeral watersheds. *J. Environ. Qual.* 41, 1531–1539. doi: 10.2134/jeq2011.0473
- Hernandez-Santana, V., Zhou, X., Helmers, M. J., Asbjornsen, H., Kolka, R., and Tomer, M. (2013). Native prairie filter strips reduce runoff from hillslopes under annual row-crop systems in Iowa, USA. *J. Hydrol.* 477, 94–103. doi: 10.1016/j.jhydrol.2012.11.013
- Hirsh, S. M., Mabry, C. M., Schulte, L. A., and Liebman, M. (2013). Diversifying agricultural catchments by incorporating tallgrass prairie buffer strips. *Ecol. Restor.* 31, 201–211. doi: 10.3368/er.31.2.201
- Hoekstra, J. M., Boucher, T. M., Ricketts, T. H., and Roberts, C. (2005). Confronting a biome crisis: global disparities of habitat loss and protection. *Ecol. Lett.* 8, 23–29. doi: 10.1111/j.1461-0248.2004.00686.x
- Iowa Department of Natural Resources (2015). *Iowa's Wildlife Action Plan: Securing a Future for Fish and Wildlife*. Available online at: https://www.iowadnr.gov/Portals/1/dnr/uploads/Wildlife%20Stewardship/iwap/iwap_summary.pdf (accessed May 25, 2022).
- Iowa Department of Natural Resources (2020). *2020 Assessment Summary 303(d) List of Impaired Waters*. Available online at: <https://programs.iowadnr.gov/adbnnet/Assessments/Summary/2020> (accessed August 24, 2022).
- Iowa Monarch Conservation Consortium (IMCC) (2018). *Conservation Strategy for the Eastern Monarch Butterfly (Danaus plexippus) in Iowa*. Iowa Monarch Conservation Consortium, Iowa State University, IA, USA. Available online at: <https://monarch.ent.iastate.edu/files/file/iowa-monarch-conservation-strategy.pdf> (accessed May 25, 2022).
- Iowa Nutrient Reduction Strategy (INRS) (2017). *Prepared by the Iowa Department of Agriculture and Land Stewardship, Iowa Department of Natural Resources and Iowa State University College of Agriculture and Life Sciences, Iowa*. Available online at: <https://www.nutrientstrategy.iastate.edu/> (accessed May 25, 2022).
- Iowa State University Extension and Outreach (ISUEO) (2022). *Ag Decision Maker*. Available online at: <https://www.extension.iastate.edu/agdm/> (accessed July 27, 2022).
- James, L. K., Swinton, S. M., and Thelen, K. D. (2010). Profitability analysis of cellulosic energy crops compared with corn. *J. Agron.* 102, 675–687. doi: 10.2134/agronj2009.0289
- John, S., and McIsaac, G. (2017). Multifunctional agriculture: a new paradigm of mixed cropping. *Solut. J.* 8, 66–76. Available online at: <https://thesolutionsjournal.com/2017/01/16/multifunctional-agriculture-new-paradigm-mixed-cropping/>
- Johnson, K. A., Polasky, S., Nelson, E., and Pennington, D. (2012). Uncertainty in ecosystem services valuation and implications for assessing land use tradeoffs: an agricultural case study in the Minnesota River Basin. *Ecol. Econ.* 79, 71–79. doi: 10.1016/j.ecolecon.2012.04.020
- Kennedy, C. M., Lonsdorf, E., Neel, M. C., Williams, N. M., Ricketts, T. H., Winfree, R., et al. (2013). A global quantitative synthesis of local and landscape effects on wild bee pollinators in agroecosystems. *Ecol. Lett.* 16, 584–599. doi: 10.1111/ele.12082
- Khanna, M., Chen, L., Basso, B., Cai, X., Field, J. L., Guan, K., et al. (2021). Redefining marginal land for bioenergy crop production. *GCB Bioenerg.* 13, 1590–1609. doi: 10.1111/gcbb.12877
- Khanna, M., Dhungana, B., and Clifton-Brown, J. (2008). Costs of producing miscanthus and switchgrass for bioenergy in Illinois. *Biomass Bioenerg.* 32, 482–493. doi: 10.1016/j.biombioe.2007.11.003
- Li, X., Tack, J. B., Coble, K. H., and Barnett, B. J. (2016). *Can crop productivity indices improve crop insurance rates?* 333-2016-14429. doi: 10.22004/ag.econ.235750
- Liebman, M. Z., Jarchow, M. E., Dietzel, R. N., and Sundberg, D. N. (2014). *Above-and Below-Ground Biomass Production in Corn and Prairie Bioenergy Cropping Systems*. Available online at: <https://dr.lib.iastate.edu/handle/20.500.12876/35932> (accessed May 25, 2022).
- Liebman, M. Z., and Schulte, L. A. (2015). Enhancing agroecosystem performance and resilience through increased diversification of landscapes and cropping systems. *Elementa* 3, 41. doi: 10.12952/journal.elementa.000041
- Liu, T. T., McConkey, B. G., Ma, Z. Y., Liu, Z. G., Li, X., and Cheng, L. L. (2011). Strengths, weaknessness, opportunities and threats analysis of bioenergy production on marginal land. *Energy Procedia.* 5, 2378–2386. doi: 10.1016/j.egypro.2011.03.409
- Manatt, R. K., Hallam, A., Schulte, L. A., Heaton, E. A., Gunther, T., Hall, R. B., et al. (2013). Farm-scale costs and returns for second generation bioenergy cropping systems in the US Corn Belt. *Environ. Res. Lett.* 8, 035037. doi: 10.1088/1748-9326/8/3/035037
- Martinez-Feria, R. A., Basso, B., and Kim, S. (2022). Boosting climate change mitigation potential of perennial lignocellulosic crops grown on marginal lands. *Environ. Res. Lett.* 17, 044004. doi: 10.1088/1748-9326/ac541b
- Massey, R., and Brown, B. (2021). *Cash rental rates in Missouri. G427. University of Missouri Extension*. Available online at: <https://extension.missouri.edu/publications/g427> (accessed May 25, 2022).
- McCarty, T., and Sesmero, J. (2021). Contracting for perennial energy crops and the cost-effectiveness of the biomass crop assistance program. *Energy Policy.* 149, 112018. doi: 10.1016/j.enpol.2020.112018
- MEA (2005). *Millenium Ecosystem Assessment: Ecosystems and Human Well-Being*. World Resources Institute.
- Meals, D. W., Dressing, S. A., and Davenport, T. E. (2010). Lag time in water quality response to best management practices: a review. *J. Environ. Qual.* 39, 85–96. doi: 10.2134/jeq2009.0108
- Meehan, T. D., Gratton, C., Diehl, E., Hunt, N. D., Mooney, D. F., Ventura, S. J., et al. (2013). Ecosystem-service tradeoffs associated with switching from annual to perennial energy crops in riparian zones of the US Midwest. *PLoS ONE* 8, e80093. doi: 10.1371/journal.pone.0080093
- Miao, R., and Khanna, M. (2017). Costs of meeting a cellulosic biofuel mandate with perennial energy crops: Implications for policy. *Ener. Econom.* 64, 321–334. doi: 10.1016/j.eneco.2017.03.018
- Mishra, S. K., Negri, M. C., Kozak, J., Cacho, J. F., Quinn, J., Secchi, S., et al. (2019). Valuation of ecosystem services in alternative bioenergy landscape scenarios. *GCB Bioenerg.* 11, 748–762. doi: 10.1111/gcbb.12602
- Missouri Department of Conservation (2015). *Missouri State Wildlife Action Plan: conserving healthy fish, forests, and wildlife*. Missouri Department of Conservation, Missouri, USA. Available online at: <https://mdc.mo.gov/sites/default/files/2020-04/SWAP.pdf> (accessed May 25, 2022).
- MoDNR (2014). *Missouri Nutrient Loss Reduction Strategy*. Missouri Department of Natural Resources, Jefferson City, Missouri. Available online at: <https://dnr.mo.gov/document-search/missouri-nutrient-loss-reduction-strategy> (accessed May 25, 2022).

- MoDNR (2020). 2020 EPA Approved Section 303(d) Listed Waters. Available online at: <https://dnr.mo.gov/document/2020-epa-approved-section-303d-listed-waters> (accessed March 12, 2022).
- Multi-Resolution Land Characteristic Consortium (MRLCC) (2018). *National Land Cover Database 2016 (NLCD 2016)*. Available online at: <https://www.mrlc.gov/data/nlcd-2016-land-cover-conus> (accessed May 1, 2019).
- Nair, S. K., Griffel, L. M., Hartley, D. S., McNunn, G. S., and Kunz, M. R. (2018). Investigating the efficacy of integrating energy crops into non-profitable subfields in Iowa. *Bioenergy Res.* 11, 623–637. doi: 10.1007/s12155-018-925-0
- Nair, S. K., Hartley, D. S., Gardner, T. A., McNunn, G., and Searcy, E. M. (2017). An integrated landscape management approach to sustainable bioenergy production. *BioEnergy Res.* 10, 929–948. doi: 10.1007/s12155-017-9854-3
- Natural Capital, Project (2019). *InVEST*. Available online at: <https://naturalcapitalproject.stanford.edu/invest/#what-is-invest> (accessed May 25, 2022).
- Nichols, V. A., Miguez, F. E., Jarchow, M. E., Liebman, M. Z., and Dien, B. S. (2014). Comparison of cellulosic ethanol yields from midwestern maize and reconstructed tallgrass prairie systems managed for bioenergy. *Bioenergy Res.* 7, 1550–1560. doi: 10.1007/s12155-014-9494-9
- NRCS (2006). *Land resource regions and major land resource areas of the United States, the Caribbean, and the Pacific Basin*. Available online at: https://efotg.sc.gov.usda.gov/references/public/IL/Land_Resource_Regions_and_Major_Land_Resource_Areas_of_the_United_States_USDA_NRCS_1996.pdf (accessed May 25, 2022).
- Padella, M., O'Connell, A., and Prussi, M. (2019). What is still limiting the deployment of cellulosic ethanol? Analysis of the current status of the sector. *Appl. Sci.* 9, 4523. doi: 10.3390/app9214523
- Paustian, K. H., and Cole, C. V. (1998). CO₂ mitigation by agriculture – an overview. *Clim. Change* 40, 135–162. doi: 10.1023/A:1005347017157
- Pitchford, G., and Kerns, H. (1999). *Grand River Watershed Inventory And Assessment*. Missouri Department of Conservation, St. Joseph, Missouri. Available online at: https://mdc.mo.gov/sites/default/files/2021-12/140_2021_GrandRiver.pdf (accessed May 25, 2022).
- Plastina, A., Johanns, A., and Welter, C. (2022). *Cash Rental Rates for Iowa 2022 Survey*. Available online at: <https://www.extension.iastate.edu/agdm/wholefarm/html/c2-10.html> (accessed June 21, 2022).
- Polasky, S., Nelson, E., Pennington, D., and Johnson, K. A. (2011). The impact of land-use change on ecosystem services, biodiversity and returns to landowners: a case study in the state of Minnesota. *Environ. Resour. Econ.* 48, 219–242. doi: 10.1007/s10640-010-9407-0
- Potts, S. G., Biesmeijer, J. C., Kremen, C., Neumann, P., Schweiger, O., and Kunin, W. E. (2010). Global pollinator declines: trends, impacts and drivers. *Trends Ecol. Evol.* 25, 345–353. doi: 10.1016/j.tree.2010.01.007
- Powell, L. A., Edwards, R., Powell, K. D., and Nieland, K. (2018). Geography of ecotourism potential in the Great Plains: incentives for conservation. *Great Plains Res.* 28, 15–24. doi: 10.1353/gpr.2018.0001
- Power, A. G. (2010). Ecosystem services and agriculture: tradeoffs and synergies. *Philos. Trans. Royal Soc. B: Biol. Sci.* 365, 2959–2971. doi: 10.1098/rstb.2010.0143
- Prairie Lands (2022). *Prairie Lands Bio-Products Purchases Biomass Project Assets*. Available online at: <https://www.iowaswitchgrass.com/> (accessed March 12, 2022).
- Rao, N. S., Easton, Z. M., Schneiderman, E. M., Zion, M. S., Lee, D. R., and Steenhuis, T. S. (2009). Modeling watershed-scale effectiveness of agricultural best management practices to reduce phosphorus loading. *J. Environ. Manage.* 90, 1385–1395. doi: 10.1016/j.jenvman.2008.08.011
- Redhead, J. W., May, L., Oliver, T. H., Hamel, P., Sharp, R., and Bullock, J. M. (2018). National scale evaluation of the InVEST nutrient retention model in the United Kingdom. *Sci. Total Environ.* 610, 666–677. doi: 10.1016/j.scitotenv.2017.08.092
- Reed, D. (2020). *Ecosystem Services Markets Conceived and Designed for US Agriculture. Soil and Water Conservation: A Celebration of 75*. Available online at: https://www.swcs.org/static/media/cms/75th_Book_Chapter_6_9EED1D79CE996.pdf (accessed July 27, 2022).
- Ritchie, H., and Roser, M. (2019). *Crop Yields*. Available online at: <https://ourworldindata.org/crop-yields#the-trade-off-between-higher-yields-and-land-use> (accessed May 25, 2022).
- Roberts, D. C., Clark, C. D., English, B. C., Park, W. M., and Roberts, R. K. (2009). Estimating annualized riparian buffer costs for the Harpeth River watershed. *Appl. Econom. Perspect. Policy.* 31, 894–913. Available online at: <https://www.jstor.org/stable/40588534>
- Salata, S., Garnero, G., Barbieri, C. A., and Giaimo, C. (2017). The integration of ecosystem services in planning: An evaluation of the nutrient retention model using InVEST software. *Land* 6, 48. doi: 10.3390/land6030048
- Salzman, J., Bennett, G., Carroll, N., Goldstein, A., and Jenkins, M. (2018). The global status and trends of payments for ecosystem services. *Nature Sustain.* 1, 136–144. doi: 10.1038/s41893-018-0033-0
- Schmer, M. R., Vogel, K. P., Mitchell, R. B., and Perrin, R. K. (2008). Net energy of cellulosic ethanol from switchgrass. *PNAS*. 105, 464–469. doi: 10.1073/pnas.0704767105
- Schulte, L. A. (2014). Prairie strips: bringing biodiversity, improved water quality, and soil protection to agriculture. *Missouri Prairie J.* 35, 12. Available online at: <https://www.nrem.iastate.edu/research/STRIPS/files/publication/Schulte%202014%20MoPrairieJournal%20-%20Prairie%20Strips.pdf>
- Schulte, L. A., Asbjornsen, H., Liebman, M., and Crow, T. R. (2006). Agroecosystem restoration through strategic integration of perennials. *J. Soil Water Conserv.* 61, 164A–169A. Available online at: <https://www.jswconline.org/content/61/6/164A.short>
- Schulte, L. A., Niemi, J., Helmers, M. J., Liebman, M., Arbuckle, J. G., James, D. E., et al. (2017). Prairie strips improve biodiversity and the delivery of multiple ecosystem services from corn-soybean croplands. *PNAS*. 114, 11247–11252. doi: 10.1073/pnas.1620229114
- Schulze, J., Frank, K., Priess, J. A., and Meyer, M. A. (2016). Assessing regional-scale impacts of short rotation coppices on ecosystem services by modeling land-use decisions. *PLoS ONE* 11, e0153862. doi: 10.1371/journal.pone.0153862
- Sharma, B. P., Yu, T. E., English, B. C., Boyer, C. N., and Larson, J. A. (2020). Impact of government subsidies on a cellulosic biofuel sector with diverse risk preferences toward feedstock uncertainty. *Energy Policy* 146, 111737. doi: 10.1016/j.enpol.2020.111737
- Sharp, R., Tallis, H. T., Ricketts, T., Guerry, A. D., Wood, S. A., Chaplin-Kramer, R., et al. (2018). *InVEST user's guide. The Natural Capital Project: Stanford, CA, USA*. Available online at: <https://invest-userguide.readthedocs.io/en/latest/> (accessed May 25, 2022).
- Shepherd, P. (2000). *Pioneering energy crops in the Midwest, project update: Chariton Valley. National Renewable Energy Lab. (NREL), Golden, CO (United States)*. <https://www.nrel.gov/docs/fy00osti/28112.pdf> (accessed May 25, 2022).
- Tilman, D., Hill, J., and Lehman, C. (2006). Carbon-negative biofuels from low-input high-diversity grassland biomass. *Science* 314, 1598–1600. doi: 10.1126/science.1133306
- Tol, R. S. (2009). The economic effects of climate change. *JEP* 23, 29–51. doi: 10.1257/jep.23.2.29
- Tomer, M. D., Porter, S. A., James, D. E., Boomer, K. M. B., Kostel, J. A., and McLellan, E. (2013). Combining precision conservation technologies into a flexible framework to facilitate agricultural watershed planning. *J. Soil Water Conserv.* 68, 113A–120A. doi: 10.2489/jswc.68.5.113A
- Tyndall, J., and Roesch, G. E. (2014). Agricultural water quality BMPs: a standardized approach to financial analysis. *J. Ext.* 52, 1.
- Tyndall, J. C., Schulte, L. A., Liebman, M., and Helmers, M. (2013). Field-level financial assessment of contour prairie strips for enhancement of environmental quality. *Environ. Manage.* 52, 736–747. doi: 10.1007/s00267-013-0106-9
- Tyndall, J. C., Valcu-Lisman, A., Bogert, M., and Zobrodsky, A. (2021). The cover crop seed industry: an Indiana case study. *J. Appl. Farm Econ.* 4, 4. doi: 10.7771/2331-9151.1056
- USDA FSA (2020). *Conservation Reserve Program Statistics*. Available online at: <https://www.fsa.usda.gov/programs-and-services/conservation-programs/reports-and-statistics/conservation-reserve-program-statistics/index> (accessed March 12, 2022).
- Vaché, K. B., Eilers, J. M., and Santelmann, M. V. (2002). Water quality modeling of alternative agricultural scenarios in the us cornbelt. *J. Am. Water Resour. Assoc.* 38, 773–787. doi: 10.1111/j.1752-1688.2002.tb00996.x
- Whitmore, A. P., Kirk, G. J. D., and Rawlins, B. G. (2015). Technologies for increasing carbon storage in soil to mitigate climate change. *Soil Use Manag.* 31, 62–71. doi: 10.1111/sum.12115
- Winfree, R., Aguilar, R., Vázquez, D. P., LeBuhn, G., and Aizen, M. A. (2009). A meta-analysis of bees' responses to anthropogenic disturbance. *Ecology* 90, 2068–2076. doi: 10.1890/08-1245.1
- Woodbury, P. B., Kemanian, A. R., Jacobson, M., and Langholtz, M. (2018). Improving water quality in the Chesapeake Bay using payments for ecosystem services for perennial biomass for bioenergy and biofuel production. *Biomass Bioenergy* 114, 132–142. doi: 10.1016/j.biombioe.2017.01.024

- Yang, P., Cai, X., Hu, X., Zhao, Q., Lee, Y., Khanna, M., et al. (2022). An agent-based modeling tool supporting bioenergy and bio-product community communication regarding cellulosic bioeconomy development. *Renew. Sustain. Energy Rev.* 167, 112745. doi: 10.1016/j.rser.2022.112745
- Zhou, X., Al-Kaisi, M., and Helmers, M. J. (2009). Cost effectiveness of conservation practices in controlling water erosion in Iowa. *Soil Tillage Res.* 106, 71–78. doi: 10.1016/j.still.2009.09.015
- Zhou, X., Helmers, M. J., Asbjornsen, H., Kolka, R., Tomer, M. D., and Cruse, R. M. (2014). Nutrient removal by prairie filter strips in agricultural landscapes. *J. Soil Water Conserv.* 69, 54–64. doi: 10.2489/jswc.69.1.54
- Zilverberg, C. J., Johnson, W. C., Owens, V., Boe, A., Schumacher, T., Reitsma, K., et al. (2014). Biomass yield from planted mixtures and monocultures of native prairie vegetation across a heterogeneous farm landscape. *Agric. Ecosyst. Environ.* 186, 148–159. doi: 10.1016/j.agee.2014.01.027
- Zimmerman, E., James, D., Jordahl, J., Magala, R., Schulte Moore, L. A., and Tyndall, J. C. (2022). "Chapter 8. agricultural carbon planning," in *Carbon Science for Carbon Markets: Emerging Opportunities in Iowa*. CROP 3175, eds L. Schulte Moore and J Jordahl. (Iowa: Iowa State University Extension and Outreach).
- Zimmerman, E. K., Tyndall, J. C., and Schulte, L. A. (2019). Using spatially targeted conservation to evaluate nitrogen reduction and economic opportunities for best management practice placement in agricultural landscapes. *Environ. Manag.* 64, 313–328. doi: 10.1007/s00267-019-01190-7



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Johann G. Zaller,
University of Natural Resources and
Life Sciences Vienna, Austria

REVIEWED BY

Alexandra Huddell,
Columbia University, United States
Ardesir Adeli,
United States Department of
Agriculture (USDA), United States
Stephen K. Hamilton,
Michigan State University,
United States

*CORRESPONDENCE

Jacob M. Jungers
junge037@umn.edu

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Reductions in soil water nitrate beneath a perennial grain crop compared to an annual crop rotation on sandy soil

Evelyn C. Reilly¹, Jessica L. Gutknecht², Craig C. Sheaffer¹ and Jacob M. Jungers^{1*}

¹Department of Agronomy and Plant Genetics, University of Minnesota, St. Paul, MN, United States,

²Department of Soil, Water, and Climate, University of Minnesota, St. Paul, MN, United States

Nitrate (NO_3^- -N) leaching into groundwater as a result of high nitrogen (N) fertilizer rates to annual crops presents human health risks and high costs associated with water treatment. Leaching is a particularly serious concern on sandy soils overlying porous bedrock. Intermediate wheatgrass (IWG) [*Thinopyrum intermedium* (Host.) Barkw. & D.R. Dewey], is a perennial grass that is being bred to produce agronomically and economically viable grain, which is commercially available as Kernza[®]. Intermediate wheatgrass is a low-input crop has the potential to produce profitable grain and biomass yields while reducing NO_3^- -N leaching on sandy soils compared with common annual row crop rotations in the Upper Midwest. We compared grain yields, biomass yields, soil solution NO_3^- -N concentration, soil extractable NO_3^- -N, soil water content, and root biomass under IWG and a conventionally managed corn (*Zea mays* L.) and soybean [*Glycine max* (L.) Merr.] rotation for 3 years on a Verndale sandy loam in Central Minnesota. Mean soil solution NO_3^- -N was 77–96% lower under IWG than the annual crop rotation. Soil water content was greater under annuals compared to IWG early in the growing season, suggesting greater water use by IWG during this time. Interactions between crop treatments and depth were observed for soil water content in Year 3. Root biomass from 0 to 60 cm below the soil surface was five times greater beneath IWG compared to soybean, which may explain differences in soil extractable and solution NO_3^- -N among crops. With irrigation on coarse structured soils, IWG grain yields were 854, 434, and 222 kg ha⁻¹ for Years 1–3 and vegetative biomass averaged 4.65 Mg ha⁻¹ yr⁻¹; comparable to other reports on heavier soils in the region. Annual crop grain yields were consistent with local averages. These results confirm that IWG effectively reduces soil solution NO_3^- -N concentrations even on sandy soils, supporting its potential for broader adoption on land vulnerable to NO_3^- -N leaching.

KEYWORDS

intermediate wheatgrass, nitrate, leaching, Kernza, groundwater, perennial grains

Introduction

Water quality in the Upper Midwest is threatened by the intensive management practices used in annual cropping systems, including tillage and fertilizer application that lead to nutrient losses and water contamination through leaching and runoff (Randall and Mulla, 2001; Dinnes et al., 2002; Feyereisen et al., 2006; Erisman et al., 2013). While annual commodity crops like corn provide the potential for high economic return, nutrient losses cause eutrophication and hypoxia in surface waters and contamination of groundwater, posing significant risks to human health (Ward et al., 2010, 2018; Brender et al., 2013). Impacts are often high where shallow aquifers and sandy soils make drinking water sources vulnerable to contamination. This leads to additional water treatment costs of over \$5 million for some counties (Keeler et al., 2016). In Southeast Minnesota, for example, conversion of grassland to agriculture is expected to cause a 45% increase in private wells exceeding 10 ppm NO_3^- -N, resulting in between \$700,000 and \$12,000,000 in associated costs over a 20-year period (Keeler and Polasky, 2014). New alternative cropping systems that provide economic returns comparable to those of annual systems and which effectively reduce nutrient losses will be essential for protecting drinking water sources in the future.

Replacing annual crops with perennials has the potential to help reduce NO_3^- leaching to groundwater and provide other ecosystem services (Asbjornsen et al., 2014; Ferchaud and Mary, 2016). Cropping systems that include perennial grasses for conservation, forage, and biofuel production have lower NO_3^- leaching losses than corn-soybean systems, largely because perennial grasses have greater root biomass that extends deeper into the soil, increasing N recovery and reducing leaching (Culman et al., 2013b; Pugesgaard et al., 2015; Ferchaud and Mary, 2016). Deep roots may be particularly important in reducing NO_3^- leaching since they can expand the total volume of soil from which NO_3^- -N is taken up, and because NO_3^- is highly mobile and more prone to leaching from deep soil horizons (Maeght et al., 2013). NO_3^- losses in the subsurface drainage water for a corn-soybean system were about 37 times higher than from a Conservation Reserve Program (CRP) planting dominated by perennial grasses (Randall et al., 1997). This reduction was attributed to the greater season-long evapotranspiration (ET) that resulted in less drainage and greater uptake and/or immobilization of N. In that study, average NO_3^- concentrations in the water during the flow period were 24 mg/L for the corn-soybean rotation and 2 mg/L for the perennial grass CRP (Randall et al., 1997). Although plantings that include perennial grasses are effective at reducing NO_3^- leaching, a lack of economic return has prevented their large-scale adoption in Midwestern agricultural landscapes.

Intermediate wheatgrass (IWG), [*Thinopyrum intermedium* (Host.) Barkw. & D.R. Dewey] is a perennial cool-season grass being domesticated to produce a grain marketed as Kernza® (DeHaan et al., 2018) with the first commercial variety, “MN-Clearwater,” released in 2020 (Bajgain et al., 2020). The crop has potential to provide economic return for producers (Hunter et al., 2020a,b; Law et al., 2022) while reducing NO_3^- leaching compared to corn (Jungers et al., 2019). Intermediate wheatgrass initiates growth earlier in the season than warm-season forage and bioenergy grasses and is thus better able to reduce NO_3^- -N losses early in the season (Jungers et al., 2019) when losses are typically the highest in the Upper Midwest (Randall and Mulla, 2001; Crews and Peoples, 2005). Vegetative regrowth following IWG grain harvest helps reduce post-harvest nitrate losses and erosion late into the fall.

One potential mechanism by which IWG can reduce NO_3^- leaching compared to annual crops is related to water demand. Although total growing season ET and drainage were similar between IWG and corn, soil water content was lower under IWG compared to corn and switchgrass at 50 and 100 cm depths (Jungers et al., 2019), suggesting that soil moisture may be stored in other regions of the soil profile. Compared to annual wheat (*Triticum aestivum* L.), IWG had lower soil moisture up to a depth of 70–100 cm, which was associated with NO_3^- -N leaching reductions of up to 86% (Culman et al., 2013b). The distribution of IWG root biomass and its effects on soil water content throughout the soil profile are largely unknown.

Reductions in NO_3^- leaching beneath IWG compared to annual crops can also be related to differences in nitrogen fertilization regimes and associated losses of N in the form of soluble NO_3^- -N in the soil water. Soil solution NO_3^- increased from 0.1 to 0.3 mg L⁻¹ when IWG was fertilized with 120 kg N ha⁻¹ compared to an unfertilized control, yet this was still lower than the 24.0 mg L⁻¹ measured beneath corn fertilized at 160 kg ha⁻¹ (Jungers et al., 2019). Integrating legumes such as soybean into annual crop rotations can limit N fertilizing needs, yet the effects of legume crops in rotation on NO_3^- -N leaching compared to IWG are unknown.

Our objective was to assess the potential of IWG grain production to reduce NO_3^- -N leaching compared to an annual soybean-corn-soybean rotation on irrigated sandy soil by measuring soil solution NO_3^- -N concentration and soil water content. We hypothesized that soil water NO_3^- -N concentrations and soil water content would be lower under IWG, and that this would be related to increased root biomass and rooting depth of IWG compared to corn and soybean. Crop yields and vegetative biomass were measured to assess potential profitability.

TABLE 1 Average air temperature, precipitation, irrigation, and 30-year averages for each month of the growing season in Staples, MN.

	Mean monthly air temperature (°C)				Monthly and season total precipitation (P) and irrigation (I) (mm)							
	2018	2019	2020	30-year avg.	2018		2019		2020		30-year avg.	
					P	I	P	I	P	I	P	
April	2	5	3	5	4.6	0	25.7	0	22.4	0	36.8	
May	17	11	12	12	62.8	0	62.5	0	33.8	0	72.9	
June	20	18	21	18	78.3	12.7	68.4	12.7	57.2	25.4	117.3	
July	21	21	22	20	62.5	38.1	103.2	63.5	102.7	38.1	99.1	
Aug.	19	18	20	19	66.6	38.1	93.8	12.7	158.6	38.1	74.4	
Sept.	14	15	14	15	73.7	0	106.3	0	16	0	71.1	
Oct.	4	5	3	7	80.0	0	92.3	0	10.9	0	56.6	
					428.5	88.9	552.2	88.9	401.6	101.6	528.2	

Methods

Site description

Field research was conducted from 2018 to 2020 at the Central Lakes Community College in Staples, MN, USA (lat. 46.38, long. -94.80). The soil type was a Verndale sandy loam (Typic Argiudoll). The soil contains 1–1.7% organic matter, is excessively well-drained, and is considered low fertility potential (USDA-NRCS, 2021). Local climate data are reported in Table 1. Plots had previously been planted to a corn-soybean rotation followed by barley fertilized with 40 kg N ha⁻¹ applied in spring prior to IWG planting in 2017. Baseline soil samples from 0 to 30 cm were collected by block in the fall of 2017. Soil extractable nitrogen was 10.0 mg kg soil⁻¹ for NO₃⁻-N and 3.9 mg kg soil⁻¹ for ammonium (NH₄⁺-N). Soil phosphorus (P) and potassium (K) concentrations were 9.13 and 72.21 ppm, respectively.

Experimental design

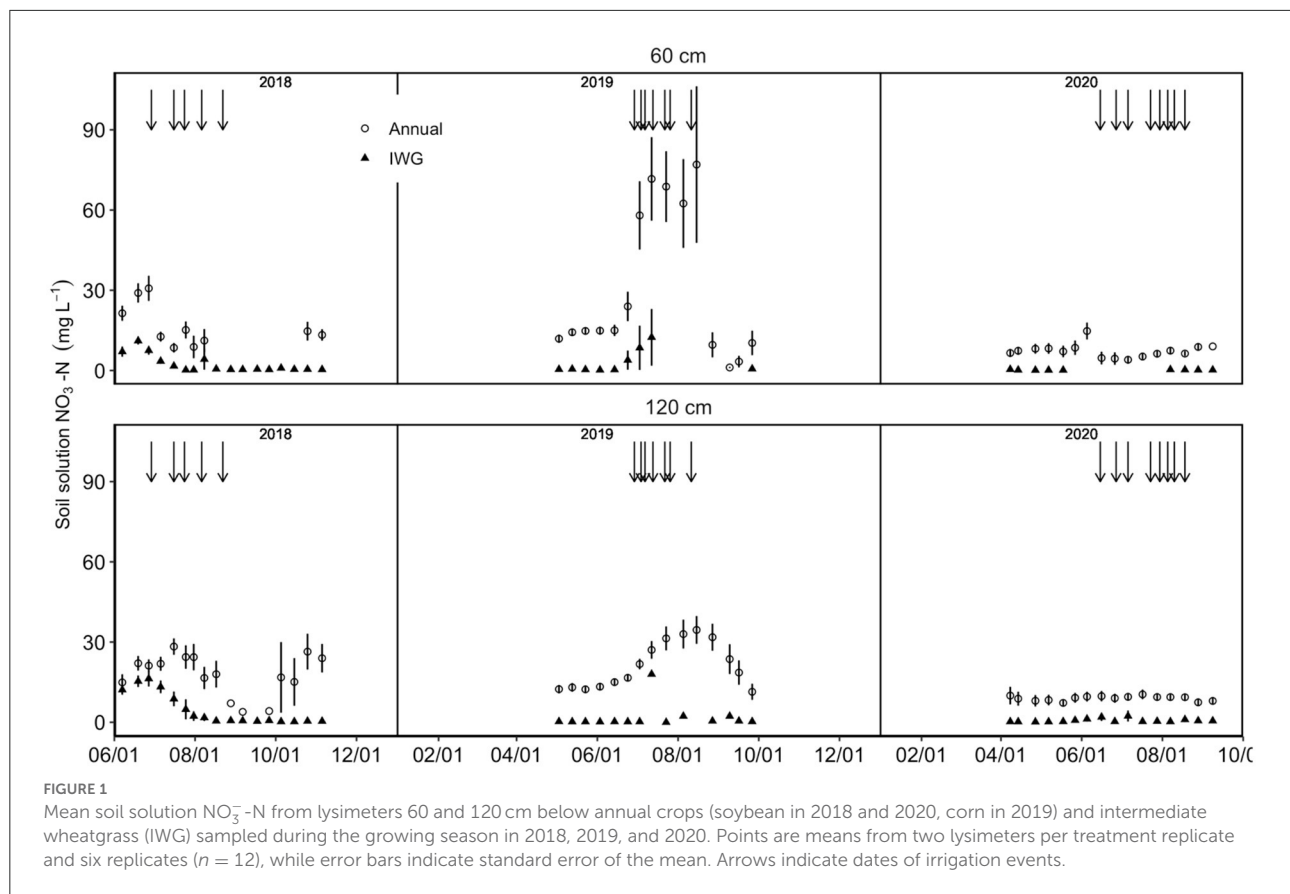
Treatments were applied in a randomized block design with two cropping systems replicated once in each of six blocks for a total of twelve plots. Plots were 4.11 by 9.14 m (13.5 by 30 ft.). The annual cropping system was a soybean-corn-soybean rotation. The perennial system was IWG. Soybeans were planted as the first phase of the soybean-corn rotation in May 2018, followed by corn in May 2019 and soybean again as the third phase in June 2020. Corn and soybeans were seeded in 75 cm rows at rates of 346,000 and 84,000 seeds ha⁻¹, respectively, with four rows per plot. The corn variety was Organic Viking O.84-95UP Seed Corn and the soybean was Organic MN0810CN.

An improved population of IWG bred for increased grain yield was used in this study. The population came from the fourth cycle of selection by Land Institute (Salina, KS) and

was seeded at a rate of 15 kg ha⁻¹. The IWG was seeded in 15-cm rows with 20 rows per plot on 20 August 2017. Intermediate wheatgrass was fertilized with urea at rates of 80, 100, and 100 kg N ha⁻¹ in May 2018, 2019, and 2020, respectively. Urea was split-applied to corn at 140 and 80 kg N ha⁻¹ in May and June 2019. Soybean was not fertilized. Weed pressure was low and when present, weeds were manually removed in all plots. The experiment was irrigated with a linear irrigation system with events based on ET estimates and water demand for the annual crop. The fields received 89 mm of irrigation water over five events in 2018, 89 mm over seven events in 2019, and 102 mm over eight events in 2020. Dates of irrigation events are in Figure 1. Each individual irrigation event resulted in an application of 13 mm of water with the exception of 7/16/2018 and 8/22/2018, which received 25 mm.

Soil fertility and extractable N

Soil was sampled at four depth intervals (0–15, 15–30, 30–45, and 45–60 cm) in June 2019 and October 2020 and analyzed for organic matter, K, P, pH, and extractable NO₃⁻-N and NH₄⁺-N. Samples were taken from eight cores in each plot and aggregated by depth, stored in a cooler when transported, and kept refrigerated until analyzed or processed for shipping. All soil analyses except extractable N were conducted by Agvise Laboratories (Benson, MN; www.agvise.com). Agvise samples were oven-dried prior to shipping. Soil extractable N was determined by extraction with a 2 M KCl solution, where 40 ml solution was added to 10 g fresh soil followed by 1 h shaking (Culman et al., 2013a). Extractions were performed within 48 h of field collection. NO₃⁻-N and NH₄⁺ analyses of the extractions were performed at the UMN Research Analytical Lab. Method details can be found at <http://ral.cfans.umn.edu/tests-analysis/soil-analysis>.



Crop yields

Crop yields were estimated each year from 2018 to 2020. Samples were taken in August of each year when the IWG had reached physiological maturity from two 76 by 76 cm quadrats with a total area of 0.58 m². Seed heads were removed from all IWG plants within the quadrat by cutting approximately 2 cm below the basal spikelet. After seed heads were removed, all remaining IWG biomass was harvested to an 8 cm stubble height. The remaining biomass was mechanically harvested and removed from the plots following quadrat sampling.

Biomass and seed heads were dried at 35°C for 72 h or until constant mass before being weighed. Grain was removed from spikes using a Wintersteiger LD 350 laboratory thresher (Wintersteiger; www.wintersteiger.com/us/Plant-Breeding-and-Research). Grain was separated from the chaff and other debris by hand-sieving and with a fractionating aspirator (Carter-Day International, Inc.; <http://www.carterday.com>).

Corn and soybean yields were determined by harvesting a subsection of the middle two rows of each plot. For corn, two 2-m sections of rows were cut from each corn plot. The number of corn stalks cut was recorded for each plot. All ears

from the cut stalks were collected, dried (35°C for 72 h), shelled, and both cobs and kernels were weighed. Three stalks from each row section were randomly selected, dried, and weighed to estimate stover mass. Soybean yields were determined by harvesting whole plants from two 1-m sections of rows from each plot, followed by drying, threshing, and weighing. Following harvest for yield measurement, the remaining corn and soybean plants were mechanically harvested and removed from the plots.

Root biomass

Root biomass samples were taken in September 2020 with two 5-cm diameter manual push cores per plot at depths of 0–15, 15–30, 30–45, and 45–60 cm. Roots were separated from soil and debris using a hydropneumatic elutriation system (Smucker et al., 1982), then removed manually from sieves using tweezers. Due to the difficulty of distinguishing live from dead roots, no effort was made to separate them. Roots were dried at 35°C for 72 h. Samples were checked after drying for any remaining sand and debris, which was removed before weighing.

Soil solution NO_3^- -N concentration

Soil solution NO_3^- -N concentrations were determined by collecting soil solution samples with suction lysimeters. Lysimeters consisted of a porous ceramic end cap, a PVC tube, and an airtight rubber stopper (Jungers et al., 2019). Two pairs of 60 and 120 cm lysimeters were installed in each plot. Samples were collected every 7–10 days from April to October each year and analyzed by depth for soil solution NO_3^- -N concentration using a colorimetric assay with a HACH DR 6000 spectrophotometer (Hach, <https://www.hach.com>).

Soil water content

Soil water content was measured on four dates in 2019 (June 17, July 19, August 21, October 31) and six dates in 2020 (May 12, June 23, July 15, August 5, September 1, and September 25) at 10, 20, 30, 40, 60, and 100 cm using a Delta-T Devices PR2/6 Probe (Delta-T Devices, 2021).

Statistical analysis

Analysis of variance (ANOVA) was conducted using mixed effects models to explain variation in soil water NO_3^- -N concentration, soil water content, soil extractable NO_3^- -N, root biomass, and crop yields. Predictor variables for the ANOVA were cropping system, depth (for soil variables), and their interaction. Years were analyzed separately because the annual crop varied. Cropping system was treated as a categorical variable; depth was treated as a categorical variable for root biomass and soil extractable NO_3^- -N. Soil solution NO_3^- -N concentrations from the 60 and 120 cm depths were not statistically different, based on preliminary statistics, and thus were averaged for the analysis. The treatment applied to the nearest neighboring plot was included in the model as a covariate to account for possible lateral movement of N applied to the neighboring plot. Data were analyzed with block as a random effect. For the soil solution NO_3^- -N, which included two pairs of lysimeters per plot, plot was nested within block in the random effects structure. An autoregressive 1 correlation structure was fit to the model to account for temporal correlation in sample results within each plot. Analysis of variance was used to explain variation in soil water content for each sampling date, with a model including treatment, depth, and their interaction. Total water content from 0 to 100 cm was calculated for each plot and date using trapezoidal integration (Hupet et al., 2004) and compared among treatments using ANOVA. Mean comparisons using Tukey's adjusted *P*-value were used to generate estimated means for effects. Statistical analysis was carried out using statistical software program R (Version 3.5.2

GUI 1.70) including *emmeans* and *nlme* packages (R Core Team, 2018; Length, 2019; Pinheiro et al., 2019).

Results

Soil solution NO_3^- -N concentration

Annual average soil solution NO_3^- -N concentration differed by cropping system treatment in 2018 ($P < 0.001$), 2019 ($P = 0.004$), and 2020 ($P = 0.003$; Table 2; Figure 1), but did not vary by sampling depth or show an interaction effect in any year ($P > 0.05$). The average soil solution NO_3^- -N concentration was 77%, 96%, and 96% lower in the perennial system than the annual system in Years 1–3, respectively (Table 2).

Throughout the seasons, both intra- and inter-annual variation was observed (Figure 1). Soil solution NO_3^- -N concentrations under IWG initially had mean values between 10 and 20 mg L⁻¹ in Year 1 but declined to nearly zero by the end of July 2018 and remained at those levels for all 3 years except for occasional deviations. In 2018, soil solution NO_3^- -N concentrations under soybean were initially high at levels above 20 mg L⁻¹, declining to near zero in mid-September, but increasing to early season levels after harvest. In 2019, however, soil solution NO_3^- -N concentrations under corn were between 10 and 20 mg L⁻¹ but spiked to levels over twice that between late June and late August. Concentrations slowly declined over the remainder of the year. In 2020, mean soil solution NO_3^- -N concentrations under soybean were consistently around 10 mg L⁻¹.

Soil extractable NO_3^- -N

There was an effect of cropping system treatment, depth, and a depth by treatment interaction (*p*-values < 0.001) on soil extractable NO_3^- -N measured at the end of the study in 2020 (Table 3). Soil NO_3^- -N was greater in the annual cropping system compared to IWG at 0–15, 15–30, and 30–45 cm depths at the end of the study ($P < 0.001$), but extractable NO_3^- -N levels were similar among treatments at the 45–60 cm depth. Soil extractable NO_3^- -N was greatest at the 0–15 cm depth below the annual crops and decreased with each depth interval until 45–60 cm, which was similar to the 30–45 cm depth interval. There was no difference in soil extractable NO_3^- -N across depths beneath the IWG.

Root biomass

Root biomass collected at the end of the study in 2020 was affected by treatment ($P = 0.006$), depth ($P < 0.001$), and a treatment by depth interaction ($P < 0.001$). Root biomass was

TABLE 2 Average soil solution NO_3^- -N, grain, and biomass yields in the annual and IWG systems in 2018, 2019, and 2020.

	2018		2019		2020	
	Annual	IWG	Annual	IWG	Annual	IWG
Soil solution NO_3^- -N (mg L^{-1})	19.0a	4.3b	22.1a	0.8b	7.8a	0.3b
Grain yield (Mg ha^{-1})	3.05a	0.85b	7.33a	0.43b	1.98a	0.22b
Biomass yield (Mg ha^{-1})	2.43b	4.12a	5.85	5.41	2.86b	4.41a

Crops in the annual system were soybean, corn and soybean in 2018, 2019, and 2020, respectively. Soil solution NO_3^- -N were averaged across depths. Lower-case letters denote statistical significance between treatments at $P < 0.05$ within each year.

TABLE 3 Mean root biomass and soil extractable nitrate (mg NO_3^- -N kg soil^{-1}) at four depth intervals from 0 to 60 cm at the end of the study in 2020.

	Root biomass (Mg ha^{-1})		Soil extractable nitrate (mg NO_3^- -N kg soil^{-1})	
	Annual	IWG	Annual	IWG
0–15	1.69b	8.57aA	2.77aA	0.17b
15–30	0.42b	2.82aB	1.38aB	0.00b
30–45	0.25	1.30B	0.48aC	0.00b
45–60	0.17	1.03B	0.25C	0.00

Letters denote statistical significance at $P < 0.05$; lower-case indicates difference between treatments; upper-case indicates difference between depths.

greater under IWG compared to the annual cropping system at all depths (Table 3). Soybean root biomass was 80%, 85%, 81%, and 83% lower than IWG root biomass at 0–15, 15–30, 30–45, and 45–60 cm, respectively. Summed over all the depths, total IWG root biomass was 13.73 Mg ha^{-1} while soybean root biomass was 2.54 Mg ha^{-1} , 82% lower ($P < 0.001$).

Crop yield

Grain yield was higher for the annual crops than for IWG in all years ($P < 0.001$, Table 2). Intermediate wheatgrass vegetative biomass yields (Table 2) were higher than soybean in 2018 ($P = 0.001$) and 2020 ($P = 0.009$) but similar to corn in 2019 ($P = 0.322$).

Soil water content

Of the four dates when soil water content was measured in 2019, there were very few effects of treatment, depth, or an interaction. Dates had a significant treatment by depth interaction. There was a main effect of cropping system treatment on soil water content on July 19 and October 31 ($P < 0.001$), in which soil water content was greater beneath the annual cropping system (0.09 m m^{-3}) compared to the perennial (0.03 m m^{-3}) on July 19 but lower in the annual (0.04 m m^{-3}) compared to the perennial (0.05 m m^{-3}) on October 31, 2019. In 2020, soil water content varied by treatment on June 23 ($P < 0.001$), in which soil water content was greater

beneath the annual compared to the perennial. There was a significant interaction between treatment and depth on three other dates in 2020. Soil water content by treatment and depth is shown in Figure 2 to illustrate the interaction.

Discussion

Soil solution NO_3^- -N concentration

Consistent with previous findings, we observed drastically lower concentrations of soil solution NO_3^- -N beneath IWG compared to the annual cropping system (Figure 1). Concentrations under IWG were initially between 10 and 20 mg L^{-1} during June and July of 2018, the first spring after seeding, but approached zero by August and remained very low for the duration of the experiment. A previous study in Minnesota found that soil solution NO_3^- -N beneath IWG averaged $0.09\text{--}0.3 \text{ mg L}^{-1}$ when fertilized with 80 kg N ha^{-1} (Jungers et al., 2019). Despite only receiving 20 kg N ha^{-1} more fertilizer annually in this study, annual average soil solution NO_3^- -N concentrations ranged from 4.3 mg L^{-1} in the first-year to 0.3 mg L^{-1} at the end of the study. Higher NO_3^- -N concentrations found in this study compared to previous finding in Minnesota could be related to the potentially higher drainage rate associated with coarse structured soil at our study. These relatively higher soil solution NO_3^- -N levels observed in the first-year of our study were also likely attributable to lower root biomass during stand establishment and thus reduced ability to capture and assimilate soil solution NO_3^- -N. In line with this thinking, our results were similar to a study on sandy soil in

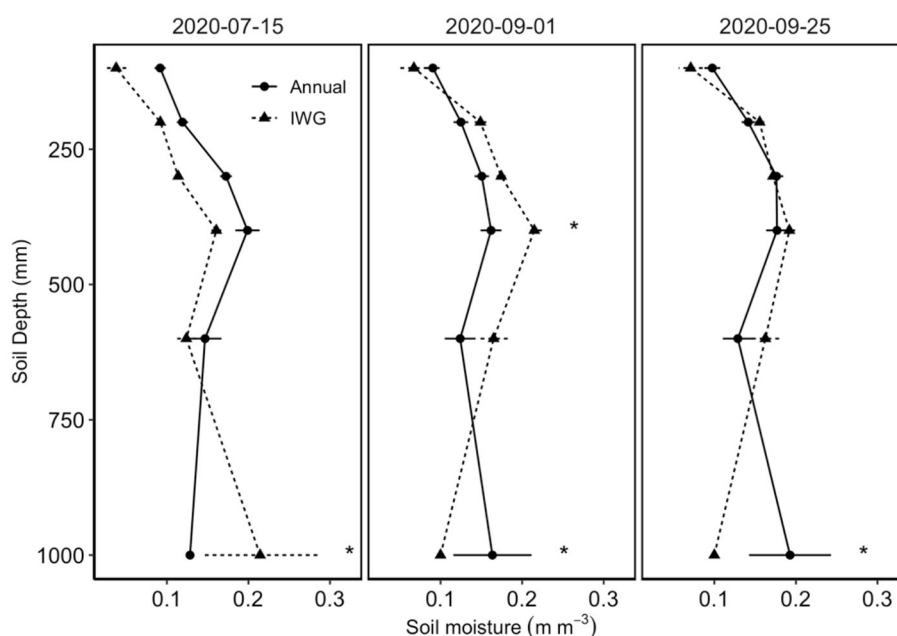


FIGURE 2

Soil water content beneath soybean (annual) and intermediate wheatgrass (IWG) on days when there was a significant interaction between cropping system treatment and depth. Asterisks indicate depths for which soil water content differed significantly between cropping systems.

Michigan during stand establishment (Culman et al., 2013b). Despite the slightly higher soil solution NO_3^- -N concentrations observed in Year 1 here and on other sandy soils, values were comparable to mixtures of perennial grasses and forbs found in CRP plantings (Randall et al., 1997) and consistently below the EPA safe drinking standard of 10 mg L^{-1} .

Average annual soil solution NO_3^- -N concentrations beneath the annual crops were similar to or slightly lower than those reported by other studies in Minnesota. During the corn phase of the annual rotation, our annual soil solution NO_3^- -N of 22.1 mg L^{-1} was similar to findings by Ochsner et al. (2017), who reported an average soil solution NO_3^- -N of 21.2 mg L^{-1} beneath a corn-soybean rotation with corn phases fertilized at 146 kg N ha^{-1} as urea annually. In another study also conducted on coarse-structured soils in Minnesota, Struffert et al. (2016) reported an average annual soil solution NO_3^- -N concentration of 18.8 mg L^{-1} beneath soybean, and determined that soil solution NO_3^- -N during the soybean phase was not affected by N fertilizer rates applied to corn the previous year.

This is also among the first studies to compare soil solution NO_3^- -N levels of fertilized IWG to an unfertilized legume crop. Despite applying 100 kg N ha^{-1} of urea annually to the IWG, lower soil solution NO_3^- -N concentrations were observed in the IWG compared to the unfertilized soybean. Biologically fixed N may have been mineralized after exudation or sloughing of soybean roots, which may have contributed to higher soil solution NO_3^- -N levels compared to IWG. The

elevated soil solution NO_3^- -N in the soybean could also have originated from N fertilizer applied during the previous crops. However, as previously mentioned, N fertilizer rates applied to a previous corn crop did not affect soil solution NO_3^- -N beneath subsequent soybean (Struffert et al., 2016). Significant N demand by IWG may have also contributed to the large difference in soil solution NO_3^- -N.

Soil extractable NO_3^- -N

In addition to lower soil solution NO_3^- -N concentration, we also found less extractable NO_3^- -N in the soil after 3 years of IWG production compared to the annual rotation system. This suggests that the IWG assimilated NO_3^- more thoroughly from the soil than the annual rotation system, especially because the Year 3 crop was unfertilized soybean. Extractable NO_3^- -N remaining in the soil is a major factor determining the concentration of dissolved NO_3^- -N in soil solution, which in turn determines total leaching loads (Randall and Mulla, 2001; Culman et al., 2013a; Jungers et al., 2019).

The low levels of extractable NO_3^- -N under IWG also suggest that the plants may have been N-limited, despite being fertilized at the high end of optimal rates (Jungers et al., 2017). Nitrogen removal during IWG grain and biomass harvest can exceed 150 kg N ha^{-1} in the first-year of production (Crews

et al., 2022; Tautges et al., 2018). Intermediate wheatgrass tissue N concentrations at the time of grain harvest in Minnesota peaked above 10 g N kg^{-1} biomass and declined with stand age (Jungers et al., 2017). If tissue N concentrations were similar to previous studies in MN, removal rates could have been between 46 and $58 \text{ kg N ha}^{-1} \text{ year}^{-1}$, thus less than the N applied as fertilizer (100 kg N ha^{-1}). However, total N demand may have been greater to support root biomass production. If root tissue N was similar to previously reported estimates between 9 and 11 g N kg^{-1} (Dobbratz, 2019), then there would be another pool of nearly 130 kg N ha^{-1} in belowground root tissues. It is not known what fraction of root N is recycled during root death and mineralization of root biomass from year to year in an IWG system, but our results suggest that the N fertilizer applied was needed to support above and belowground IWG biomass and that little N was likely lost *via* leaching or left in the soil.

Root biomass

Root biomass is considered an important trait of perennial crops for providing ecosystem services such as reduced nitrate leaching to groundwater. Intermediate wheatgrass root biomass averaged 13.7 Mg ha^{-1} after the third-year of production, while soybean root biomass was 2.5 Mg ha^{-1} when sampled from 0 to 60 cm. These values are similar to other reported values for these crops. For example, Intermediate wheatgrass fertilized at 80 kg N ha^{-1} had root biomass of 4.10, 7.32, and 9.51 Mg ha^{-1} (0–60 cm depth) in Years 1–3 of a 3-year study, while a soybean-corn-soybean rotation had root biomass of 2.22, 2.93, and 2.30 Mg ha^{-1} in Years 1–3 (Bergquist, 2019). Root biomass accumulation over time allows IWG to more effectively capture NO_3^- -N before it reaches depths below the rooting zone where it is subject to leaching to groundwater. Nearly 63% of the IWG root biomass was found in the top 0–15 cm depth. Previous work has reported IWG belowground biomass to be 3.28 Mg ha^{-1} in the first 10 cm, on average, in Minnesota and Wisconsin (Sakiroglu et al., 2020). In an intra-annual study of root biomass beneath IWG, total root biomass from 0 to 20 cm peaked between 3.5 and 4 Mg ha^{-1} in June and July before declining to 1 Mg ha^{-1} at the end of the growing season (Pugliese et al., 2019). This concentration of root biomass at shallow depths also likely increases NO_3^- -N capture and consequently reduce soil solution NO_3^- -N below the rooting zone.

Soil water content

We found inconsistent differences in soil water content between annual crops and IWG. In the second-year of the study, soil water content was greater beneath the corn compared to the IWG in July, perhaps because IWG biomass would have been approaching peak biomass and thus been demanding more

water than corn. A similar early-season pattern was found in Year 3 when soil water content was greater beneath the soybean compared to the IWG when measured in June. By the end of Year 2 (October), soil water content was greater in IWG compared to corn. Only in Year 3 did we observe any differences in soil water content by depth across treatments (Figure 2). In July, soil water content was greater beneath IWG compared to soybean at the deepest measured depth of 1,000 mm. This treatment effect was opposite at the 1,000 mm depth in September, where soil moisture content was greater for the soybean compared to IWG. Our results do match those from previous studies. In one comparison of perennial and annual systems, soil water content beneath *Miscanthus* and switchgrass was lower than a corn-soybean rotation earlier in the season, but the treatment effect flipped later in the season when switchgrass had higher soil water content (McIsaac et al., 2010). It has also been observed that soil water content tended to be higher under annuals than semi-perennials, and that there was less drainage from semi-perennials and perennials than annuals (Ferchaud and Mary, 2016). In studies with IWG, researchers have reported less in soil water content under IWG compared to annual wheat (Culman et al., 2013b) and corn (Jungers et al., 2019).

Soil water content can be used to make inferences on transpiration and drainage, the latter being an important component of nitrate leaching. The timing and frequency of our soil water content measurements precluded us from determining if both treatments had similar ET and drainage rates. Irrigation at our experiment could also have minimized our ability to detect differences in soil water content from plant ET. It is also established that greater root biomass increases water and nutrient uptake, which could reduce soil water content (Ehdaie et al., 2010; Matsunami et al., 2012; Carvalho et al., 2014). In our study, the similar soil moisture contents observed in the perennial and annual treatments may have been a function of the low water holding capacity of the sandy soil, which may have promoted drainage regardless of root biomass.

Grain and biomass yields

Intermediate wheatgrass grain yields at our sandy site were comparable to previous reports from sites with higher soil fertility levels. Under similar fertilizer treatments, reported first-year values range from 763 kg ha^{-1} (Zimbric et al., 2020) to $1,089 \text{ kg ha}^{-1}$ at sites in Wisconsin (Favre et al., 2019) and from 893 kg ha^{-1} (Jungers et al., 2017) to $1,150 \text{ kg ha}^{-1}$ (Fernandez et al., 2020) in Minnesota. Second- and third-year yields tend to be much lower, typically ranging from 150 kg ha^{-1} (Fernandez et al., 2020) to 630 kg ha^{-1} (Sakiroglu et al., 2020) in Year 2 and from 153 kg ha^{-1} (Jungers et al., 2017) to 371 kg ha^{-1} (Zimbric et al., 2020) in Year 3. Our yields suggest that this soil type and climate is appropriate for IWG grain and biomass production with irrigation.

Forage production is important for profitable IWG systems, since a major challenge of IWG grain production is the substantial yield declines in later years of production (Jungers et al., 2017; Pugliese et al., 2019; Hunter et al., 2020a). Intermediate wheatgrass biomass yields in this study included the stems and leaves that were remaining after grain harvest soon after peak productivity. Biomass harvested at this time, after physiological maturity, is relatively low in terms of forage quality compared to IWG biomass harvested at vegetative stages, but high compared to annual small grain biomass after grain harvest (Hunter et al., 2020b). Intermediate wheatgrass biomass yields were similar to those of other reports in Minnesota, though they were at the lower end of the range. Reported summer aboveground biomass values include 5,130 kg ha⁻¹ in the second-year and 5,850 kg ha⁻¹ in the third-year for IWG fertilized at 90 and 134 kg ha⁻¹ in Wisconsin and 10,600 kg ha⁻¹ for third-year stands in Minnesota (Sakiroglu et al., 2020). Similarly, summer yields of approximately 6,200 kg ha⁻¹ were reported for first-year monocultures fertilized at 100 kg N ha⁻¹ as urea (Favre et al., 2019). Biomass yields averaged 13,400 to 14,320 kg ha⁻¹ for control treatments in a management study fertilized at 56 kg ha⁻¹ the previous year (Pinto et al., 2021). Our results support that understanding that post-grain harvest biomass yields can be high enough for growers to consider harvesting for used as feed or straw on the farm or marketed for an additional revenue stream.

Conclusion

We found that soil solution NO₃⁻-N concentrations were 77–96% lower under IWG than the annual corn-soybean rotation, even in the unfertilized soybean phase of rotation, but soil water content was similar. This suggests that the IWG captured and utilized a greater proportion of soil solution NO₃⁻-N, which is also demonstrated by very low residual soil extractable NO₃⁻-N levels at the end of the experiment relative to the annual crops. The lower NO₃⁻-N concentrations in soil solution would be expected to translate to reductions in total leaching load of a similar magnitude. The increased uptake of N by IWG was likely facilitated by its greater root biomass, which was 5.4 times higher than that under the annual system. Despite the challenges associated with production of IWG on low-fertility sandy soils, grain yields were comparable to other locations and the system would likely be profitable in the first-year for grain alone. Biomass yields would support additional revenue streams in subsequent years to improve economic viability, and together our study provides evidence that IWG could be a good option for coarse textured soils that are prone to nitrate pollution.

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

Author contributions

ER: collected data, analyzed data, wrote the first draft of the manuscript, and contributed to the final draft of the manuscript. JG: oversaw soil sample processing and contributed to the final draft of the manuscript. CS: designed the field experiment and contributed to the final draft of the manuscript. JJ: acquired funding, designed the field experiment, oversaw field sampling and data collection, data visualization, and contributed to the final draft of the manuscript. All authors contributed to data interpretation, manuscript writing, and revision.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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References

- Asbjornsen, H., Hernandez-Santana, V., Liebman, M., Bayala, J., Chen, J., Helmers, M., et al. (2014). Targeting perennial vegetation in agricultural landscapes for enhancing ecosystem services. *Renew. Agric. Food Syst.* 29, 101–125. doi: 10.1017/S1742170512000385
- Bajgain, P., Zhang, X., Jungers, J. M., DeHaan, L. R., Heim, B., Sheaffer, C. C., et al. (2020). 'MN-Clearwater', the first food-grade intermediate wheatgrass (Kernza perennial grain) cultivar. *J. Plant Regist.* 14, 288–297. doi: 10.1002/plr2.20042
- Bergquist, G. (2019). Biomass Yield and Soil Microbial Response to Management of Perennial Intermediate Wheatgrass (*Thinopyrum intermedium*) as Grain Crop and Carbon Sink. Master's Theses, University of Minnesota.
- Brender, J. D., Weyer, P. J., Romitti, P. A., Mohanty, B. P., Shinde, M. U., Vuong, A. M., et al. (2013). Prenatal nitrate intake from drinking water and selected birth defects in offspring of participants in the national birth defects prevention study. *Environ. Health Perspect.* 121, 1083–1089. doi: 10.1289/ehp.1206249
- Carvalho, P., Azam-Ali, S., and Foulkes, M. J. (2014). Quantifying relationships between rooting traits and water uptake under drought in Mediterranean barley and durum wheat: root traits and water uptake. *J. Integr. Plant Biol.* 56, 455–469. doi: 10.1111/jipb.12109
- Crews, T. E., Kemp, L., Bowden, J. H., and Murrell, E. G. (2022). How the nitrogen economy of a perennial cereal-legume intercrop affects productivity: can synchrony be achieved? *Front. Sustain. Food Syst.* 6:755548. doi: 10.3389/fsufs.2022.755548
- Crews, T. E., and Peoples, M. B. (2005). Can the synchrony of nitrogen supply and crop demand be improved in legume and fertilizer-based agroecosystems? A review. *Nutr. Cycl. Agroecosystems* 72, 101–120. doi: 10.1007/s10705-004-6480-1
- Culman, S. W., Snapp, S. S., Green, J. M., and Gentry, L. E. (2013a). Short- and long-term labile soil carbon and nitrogen dynamics reflect management and predict corn agronomic performance. *Agron. J.* 105, 493–502. doi: 10.2134/agronj2012.0382
- Culman, S. W., Snapp, S. S., Ollenburger, M., Basso, B., and DeHaan, L. R. (2013b). Soil and water quality rapidly responds to the perennial grain Kernza wheatgrass. *Agron. J.* 105, 735–744. doi: 10.2134/agronj2012.0273
- DeHaan, L., Christians, M., Crain, J., and Poland, J. (2018). Development and evolution of an intermediate wheatgrass domestication program. *Sustainability* 10:1499. doi: 10.3390/su10051499
- Delta-T Devices (2021). *PR2 Profile Probe - Analogue Version*. Cambridge, UK. Available online at: www.delta-t.co.uk/product/pr2/
- Dinnes, D. L., Karlen, D. L., Jaynes, D. B., Kaspar, T. C., Hatfield, J. L., Colvin, T. S., et al. (2002). Review and interpretation: nitrogen management strategies to reduce nitrate leaching in tile-drained Midwestern soils. *Agron. J.* 94, 153–171. doi: 10.2134/agronj2002.1530
- Dobbratz, M. (2019). *Perennial fuel, feed, and cereal: High diversity perennials for biofuel and intermediate wheatgrass for grain and forage*. PhD Dissertation retrieved from University of Minnesota Digital Conservancy. Available online at: <https://hdl.handle.net/11299/211748>
- Ehdaie, B., Merhaut, D. J., Ahmadian, S., Hoops, A. C., Khuong, T., Layne, A. P., et al. (2010). Root system size influences water-nutrient uptake and nitrate leaching potential in wheat: root system and nutrient uptake in wheat. *J. Agron. Crop Sci.* 196, 455–466. doi: 10.1111/j.1439-037X.2010.00433.x
- Erisman, J. W., Galloway, J. N., Seitzinger, S., Bleeker, A., Dise, N. B., Petrescu, A. M. R., et al. (2013). Consequences of human modification of the global nitrogen cycle. *Philos. Trans. R. Soc. B Biol. Sci.* 368, 20130116. doi: 10.1098/rstb.2013.0116
- Favre, J. R., Castiblanco, T. M., Combs, D. K., Wattiaux, M. A., and Picasso, V. D. (2019). Forage nutritive value and predicted fiber digestibility of Kernza intermediate wheatgrass in monoculture and in mixture with red clover during the first production year. *Anim. Feed Sci. Technol.* 258, 114298. doi: 10.1016/j.anifeeds.2019.114298
- Ferchaud, F., and Mary, B. (2016). Drainage and nitrate leaching assessed during 7 years under perennial and annual bioenergy crops. *Bioenergy Res.* 9, 656–670. doi: 10.1007/s12155-015-9710-2
- Fernandez, C. W., Ehlike, N., Sheaffer, C. C., and Jungers, J. M. (2020). Effects of nitrogen fertilization and planting density on intermediate wheatgrass yield. *Agron. J.* 112, 4159–4170. doi: 10.1002/agj2.20351
- Feyereisen, G. W., Wilson, B. N., Sands, G. R., Strock, J. S., and Porter, P. M. (2006). Potential for a rye cover crop to reduce nitrate loss in Southwestern Minnesota. *Agron. J.* 98, 1416–1426. doi: 10.2134/agronj2005.0134
- Hunter, M. C., Sheaffer, C. C., Culman, S. W., and Jungers, J. M. (2020a). Effects of defoliation and row spacing on intermediate wheatgrass I: Grain production. *Agron. J.* 112, 1748–1763. doi: 10.1002/agj2.20128
- Hunter, M. C., Sheaffer, C. C., Culman, S. W., Lazarus, W. F., and Jungers, J. M. (2020b). Effects of defoliation and row spacing on intermediate wheatgrass II: forage yield and economics. *Agron. J.* 112, 1862–1880. doi: 10.1002/agj2.20124
- Hupet, F., Bogaert, P., and Vanclooster, M. (2004). Quantifying the local-scale uncertainty of estimated actual evapotranspiration. *Hydrol. Process.* 18, 3415–3434. doi: 10.1002/hyp.1504
- Jungers, J. M., DeHaan, L. H., Mulla, D. J., Sheaffer, C. C., and Wyse, D. L. (2019). Reduced nitrate leaching in a perennial grain crop compared to maize in the Upper Midwest, USA. *Agric. Ecosyst. Environ.* 272, 63–73. doi: 10.1016/j.agee.2018.11.007
- Jungers, J. M., DeHaan, L. R., Betts, K. J., Sheaffer, C. C., and Wyse, D. L. (2017). Intermediate wheatgrass grain and forage yield responses to nitrogen fertilization. *Agron. J.* 109, 462–472. doi: 10.2134/agronj2016.07.0438
- Keeler, B. L., Gourevitch, J. D., Polasky, S., Isbell, F., Tessum, C. W., Hill, J. D., et al. (2016). The social costs of nitrogen. *Sci. Adv.* 2, e1600219. doi: 10.1126/sciadv.1600219
- Keeler, B. L., and Polasky, S. (2014). Land-use change and costs to rural households: a case study in groundwater nitrate contamination. *Environ. Res. Lett.* 9, 074002. doi: 10.1088/1748-9326/9/7/074002
- Law, E. P., Wayman, S., Pelzer, C. J., Culman, S. W., Gómez, M. I., DiTommaso, A., et al. (2022). Multi-criteria assessment of the economic and environmental sustainability characteristics of intermediate wheatgrass grown as a dual-purpose grain and forage crop. *Sustainability* 14, 3548. doi: 10.3390/su14063548
- Length, R. (2019). *emmeans: Estimated Marginal Means, aka Least-Squares Means. R package version 1.3.3*. Available online at: <https://CRAN.R-project.org/package=emmeans>
- Maeght, J.-L., Rewald, B., and Pierret, A. (2013). How to study deep roots—and why it matters. *Front. Plant Sci.* 4:299. doi: 10.3389/fpls.2013.00299
- Matsunami, M., Matsunami, T., Ogawa, A., Toyofuku, K., Kodama, I., and Kokubun, M. (2012). Genotypic variation in biomass production at the early vegetative stage among rice cultivars subjected to deficient soil moisture regimes and its association with water uptake capacity. *Plant Prod. Sci.* 15, 82–91. doi: 10.1626/ppls.15.82
- McIsaac, G. F., David, M. B., and Mitchell, C. A. (2010). *Miscanthus* and switchgrass production in Central Illinois: impacts on hydrology and inorganic nitrogen leaching. *J. Environ. Qual.* 39, 1790–1799. doi: 10.2134/jeq2009.0497
- Ochsen, T. E., Schumacher, T. W., Venterea, R. T., Feyereisen, G. W., and Baker, J. M. (2017). Soil water dynamics and nitrate leaching under corn-soybean rotation, continuous corn, and kura clover. *Vadose Zone J.* 17, 1–11. doi: 10.2136/vzj2017.01.0028
- Pinheiro, J., Bates, D., DebRoy, S., Sarkar, D., and and, R., Core Team (2019). *nlme: Linear and Nonlinear Mixed Effects Models. R package version 3.1-139*. Available online at: <https://CRAN.R-project.org/package=nlme>
- Pinto, P., De Haan, L., and Picasso, V. (2021). Post-harvest management practices impact on light penetration and Kernza intermediate wheatgrass yield components. *Agronomy* 11, 442. doi: 10.3390/agronomy11030442
- Pugesgaard, S., Schelde, K., Larsen, S. U., Laerke, P. E., and Jørgensen, U. (2015). Comparing annual and perennial crops for bioenergy production - influence on nitrate leaching and energy balance. *GCB Bioenergy* 7, 1136–1149. doi: 10.1111/gcbb.12215
- Pugliese, J. Y., Culman, S. W., and Sprunger, C. D. (2019). Harvesting forage of the perennial grain crop Kernza (*Thinopyrum intermedium*) increases root biomass and soil nitrogen cycling. *Plant Soil* 437, 241–254. doi: 10.1007/s11104-019-03974-6
- R Core Team (2018). *R: A Language and Environment for Statistical Computing*. Vienna: R Foundation for Statistical Computing Available online at: <https://www.R-project.org/>
- Randall, G. W., Huggins, D. R., Russelle, M. P., Fuchs, D. J., Nelson, W. W., and Anderson, J. L. (1997). Nitrate losses through subsurface tile drainage in Conservation Reserve Program, Alfalfa, and row crop systems. *J. Environ. Qual.* 26, 1240–1247. doi: 10.2134/jeq1997.00472425002600050007x
- Randall, G. W., and Mulla, D. J. (2001). Nitrate nitrogen in surface waters as influenced by climatic conditions and agricultural practices. *J. Environ. Qual.* 30, 337–344. doi: 10.2134/jeq2001.302337x
- Sakiroglu, M., Dong, C., Hall, M. B., Jungers, J., and Picasso, V. (2020). How does nitrogen and forage harvest affect belowground biomass and nonstructural carbohydrates in dual-use Kernza intermediate wheatgrass? *Crop Sci.* 60, 2562–2573. doi: 10.1002/csc2.20239

Smucker, A. J. M., McBurney, S. L., and Srivastava, A. K. (1982). Quantitative separation of roots from compacted soil profiles by the hydropneumatic elutriation system. *Agron. J.* 74, 500–503. doi: 10.2134/agronj1982.00021962007400030023x

Struffert, A. M., Rubin, J. C., Fernández, F. G., and Lamb, J. A. (2016). Nitrogen management for corn and groundwater quality in Upper Midwest irrigated sands. *J. Environ. Qual.* 45.1557–1564. doi: 10.2134/jeq2016.03.0105

Tautges, N. E., Jungers, J. M., DeHaan, L. R., Wyse, D. L., and Sheaffer, C. C. (2018). Maintaining grain yields of the perennial cereal intermediate wheatgrass in monoculture v. bi-culture with alfalfa in the Upper Midwestern USA. *J. Agric. Sci.* 156, 758–773. doi: 10.1017/S0021859618000680

USDA–NRCS (2021). *Web Soil Survey*. USDA – Natural Resources Conservation Service. Available online at: <https://websoilsurvey.sc.egov.usda.gov/App/HomePage.htm>

Ward, M., Jones, R., Brender, J., de Kok, T., Weyer, P., Nolan, B., et al. (2018). Drinking water nitrate and human health: an updated review. *Int. J. Environ. Res. Public Health* 15, 1557. doi: 10.3390/ijerph15071557

Ward, M. H., Kilfoy, B. A., Weyer, P. J., Anderson, K. E., Folsom, A. R., and Cerhan, J. R. (2010). Nitrate intake and the risk of thyroid cancer and thyroid disease. *Epidemiology* 21, 389–395. doi: 10.1097/EDE.0b013e3181d6201d

Zimbric, J. W., Stoltenberg, D. E., and Picasso, V. D. (2020). Effective weed suppression in dual-use intermediate wheatgrass systems. *Agron. J.* 112, 2164–2175. doi: 10.1002/agi2.20194



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EDITED BY

Ashley Conway,
University of Missouri, United States

REVIEWED BY

Tom Bilbo,
Clemson University, United States
Simerjeet Kaur,
Punjab Agricultural University, India

*CORRESPONDENCE

Erin M. Silva
emsilva@wisc.edu

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Suppression of weed and insect populations by living cover crop mulches in organic squash production

Dylan Bruce^{1,2}, Erin M. Silva^{1*} and J. C. Dawson²

¹Department of Plant Pathology, University of Wisconsin-Madison, Madison, WI, United States,

²Department of Horticulture, University of Wisconsin-Madison, Madison, WI, United States

Living mulch systems can provide multiple agronomic and ecosystem benefits, including reducing erosion and decreasing weed and pest pressure. However, inconsistent yields and lack of best practices for weed and pest management have contributed to their lack of adoption by farmers. In 2018 and 2019, living mulch practices for organic zucchini (*Cucurbita pepo* L.) production were assessed in Wisconsin on certified organic land. Living mulches of Dutch white clover (*Trifolium repens*), annual ryegrass (*Lolium multiflorum*), and a mix of Dutch white clover and annual ryegrass were compared with full tillage cultivated ground and straw mulch controls for effect on yield, fruit marketability, weed and pest counts, and weed management time. Mixed species living mulch, cultivated, and straw mulch treatments were consistently higher yielding than clover treatments, while ryegrass had variable results. No differences were observed in the number of squash bug (*Anasa tristis*) egg clusters per plant across mulch treatments, but clover treatments had fewer adult squash bugs, with ryegrass and mixed species living mulches also trending lower. Lower counts of striped cucumber beetles (*Acalymma trivittatum*) were also observed in living mulch treatments. Ryegrass and mixed species living mulches were generally more weed suppressive than clover and cultivated aisles, although living mulch treatments generally had more weeds than straw mulched aisles, apart from comparable suppression of grass weeds for ryegrass in 2019. Time required for weed management was greater for the living mulch treatments than straw, while cultivated treatments took longer to manage than all other treatments in 2019 and longer than ryegrass and straw in 2018. Despite higher weed counts in clover than in cultivated aisles in 2019, all living mulches required less time for weed management than cultivation, indicating that managing living mulches with mowing can be more efficient than hand cultivation, even with higher weed counts. Our results support previous evidence that certain living mulch species may reduce pest and weed pressure, but also reinforces that living mulch systems can negatively impact yield depending on species selection and environment.

KEYWORDS

organic agriculture, cover crops, squash, weed management, continuous living cover

Introduction

Weed management is a critical challenge facing organic farmers and is consistently cited as a priority for further research (Moynihan, 2010; Jenkins and Ory, 2016). To manage weeds in vegetable crops, organic growers rely heavily on both mechanical cultivation and plastic mulches (Jabbour et al., 2016; Brown and Gallandt, 2018). Plastic mulches can be used to prevent weed emergence within the planting row where mechanical and hand weeding may be difficult once the crop establishes. In addition to their weed suppressive benefits, plastic mulches provide other positive aspects to the production systems, including increased soil temperature and moisture retention, which often contributes to higher yield (Kasirajan and Ngouajio, 2012; Steinmetz et al., 2016). However, plastic mulch systems also present management challenges, including exacerbation of erosion due to water runoff into the aisles between beds, which are usually managed as bare soil with cultivation or herbicide (Arnold et al., 2004; Rice et al., 2004).

Environmental impacts of runoff and erosion can be mitigated in plastic mulch systems by planting living cover crops between the plastic-covered beds (Arnold et al., 2004). The use of cover crops between rows can also reduce the long term weed seedbank while providing additional ecological services (Liebman et al., 1997; Baraibar et al., 2018; Wauters et al., 2021). Cover crops can suppress weeds through direct competition (Hiltbrunner et al., 2007; Bezuidenhout et al., 2012; Brust et al., 2014) and by generating residues which can suppress weed emergence through physical (mulch) effects, release of allelochemicals, and changes in nutrient dynamics (Teasdale and Mohler, 2000; Sarrantonio and Gallandt, 2003; Teasdale et al., 2012). Full season cover crops utilized as living mulches may also have benefits unique as compared to terminated cover crop mulches, such as promoting arbuscular mycorrhizal colonization and enhancing nutrient uptake (Deguchi et al., 2012).

While cover crops are used extensively in organic production (USDA-NASS, 2019), they are typically terminated and incorporated prior to planting the cash crop (Magdoff and Van Es, 2000). Shorter growing seasons in temperate climates, coupled with diverse, complex, and high value rotations on vegetable farms, further complicate integration of cover crops into tillage-intensive production systems of cucurbit growers in cooler climates (Sarrantonio, 1992; Snapp et al., 2005). The use of living mulches between plastic-mulched beds provides an opportunity to integrate cover crops into vegetable systems, as the cash crop can be grown concurrently with a full season cover crop while maintaining the benefits of the plastic mulch within a targeted planting zone (Tarrant et al., 2020).

Adoption of living mulch-based reduced tillage vegetable systems has been limited partly because of variable or negative effects on yield (Law et al., 2006; Butler et al., 2013; Reid and Klotzbach, 2013; Warren et al., 2015; Hinds et al., 2016; Pfeiffer et al., 2016), although other studies have shown positive results

(e.g., Sportelli et al., 2022). The unique interactions of each cash crop and cover crop contributes to the variability in observed results, creating challenges in the development of robust best practices for the diversity of crops produced by organic vegetable growers (Walters et al., 2011; Brainard et al., 2013). Living mulch studies focused on cucurbit production have shown inconsistent impacts on yields, with some indicating potential for equivalent or higher yields (Nelson and Gleason, 2018; Kahl et al., 2019) and others showing negative or variable impacts (Nyoike and Liburd, 2010; Hinds et al., 2016).

Choice of living mulch species is important to maximize weed control benefits of living mulches while minimizing risks associated with competition (Tarrant et al., 2020). Clovers are a common choice as their ability to fix atmospheric N provides fertility benefits and reduces the risk of N competition with the cash crop (Hartwig and Ammon, 2002). However, clovers also tend to be slower growing and less competitive against summer annual weeds (MacLaren et al., 2019). Tarrant et al. tested nine living mulch species and combinations and found that all living mulch treatments reduced weed biomass, with weed biomass negatively correlated with living mulch biomass. In addition, Tarrant et al. found that all treatments had the potential to compete with cash crops by lowering soil inorganic nitrogen and moisture levels within the plastic mulched beds (Tarrant et al., 2020). However, specific management such as root pruning, which reduces the depth and biomass of living roots, may reduce potential for competition (Báth et al., 2008). The drastic removal of above ground biomass caused by mowing may be associated with corresponding reductions in root biomass, and thus reduce competition potential (Liu and Huang, 2002). For instance, Hinds et al. (2016) found that zucchini yields were reduced in a living mulch system with sunn hemp (*Crotalaria juncea* L.) grown to a height of 45 cm, but when the sunn hemp was managed to a height of 20 cm, zucchini yields were equivalent or greater in the living mulch treatment than bare ground.

Mulch choice can also affect pest pressure. Two major pests of cucurbits in the upper Midwestern USA include striped cucumber beetle (*Acalymma trivittatum*) and squash bug (*Anasa tristis*). As chemical control options for organic growers are limited, organic growers must integrate cultural and mechanical methods, such as rotation, exclusion, and intercropping, in addition to allowable chemical controls to both effectively manage pest pressure and mitigate the risk of insecticide resistance (Doughty et al., 2016; Haber et al., 2021).

Some studies have shown that living mulch can exacerbate pest issues (Reid and Klotzbach, 2013), while others have shown variable or beneficial effects on pest levels (Amirault and Caldwell, 1998; Nyoike and Liburd, 2010; Grasswitz, 2013; Hinds et al., 2016). For instance, Kahl et al. (2019) found that cucumber (*Cucumis sativus* L.) interplanted with red clover (*Trifolium pratense* L.) had increased counts of natural enemies and lower counts of cucumber beetles and

TABLE 1 Summary of field activities for living mulch management of organic squash, 2018 and 2019.

Date (2018)	Date (2019)	Activity
May 17	May 15	Terminate fall-planted rye cover crop
May 17	May 17	Application of fertilizer
May 17	May 21	Additional Tillage
May 17	May 23	Application of plastic and straw mulches
May 18	May 23	Seed living mulches
June 6	June 7	Transplant
July 18 and 25; August 8 and 20	July 9 16, 23, and 29; August 6 and 13	Insect counts
July 17 and 25; August 8	June 27; July 8, 16 and 28	Weed counts
July 17 and 25; August 8	June 27; July 8, 16 and 28	Timed weed management
-	July 31	Apply pyrethrin pesticide (Pyganic®)
July 5, 17, 18, and 23; August 2nd, 8, 13, 15, 20 and 27.	July 9, 11, 15, 18, 22nd, 24, 26, 29, and 31; August 1, 3, 5 and 9	Harvests

TABLE 2 Weather data collected at UW-Madison Arboretum Weather Station, 2018 and 2019.

Time period	Total precipitation in cm (deviation from 40 year average)	Average daily temperature in °C (deviation from 40 year average)	GDDU 50 (deviation from 40 year average)
October 2017–February 2018	27.89 (+2.02)	−0.7 (+0.49)	182 (+77)
March–May 2018	33.07 (+7.71)	7.41 (−0.13)	463 (+128)
June–Sept 2018	86.11 (+41.28)	20.57 (+0.95)	2286 (+238)
October 2018–February 2019	37.24 (+11.37)	−1.13 (+0.06)	86 (−19)
March–May 2019	24.05 (−3.53)	7.45 (−0.09)	259 (−76)
June–Sept 2019	58.90 (+14.07)	20.28 (+0.66)	2164 (+116)

reduced melon aphid (*Aphis gossypii*) pressure, although spotted cucumber beetle (*Diabrotica undecimpunctata howardi*) had a variable response. Grasswitz (2013) found a similar negative response to interplanting for cucumber beetles, but saw no effect on squash bug presence, while Nyoike and Liburd (2010) also found increased natural predator populations in a buckwheat (*Fagopyrum esculentum* Moench) living mulch.

This study expands on previous research on living mulches in a plasticulture system. We specifically evaluate the effects of cover crop living aisles [Dutch white clover (*Trifolium repens*), annual ryegrass (*Lolium multiflorum*), and a mix of Dutch white clover and annual ryegrass] as compared to control treatments (cultivated management and straw mulch) on organic zucchini fruit yield, weed and pest pressure, and weed management time. We tested the null hypotheses that there would be no effect from aisle mulch in explaining yield, plant survival, or percent cover. We also tested the null hypothesis that there would be no significant effect from or interaction between aisle and date on weed counts, pest counts, or weed management time, which would indicate that the mulch treatments performed the same throughout the season.

Materials and methods

Site and treatment descriptions

Field trials were conducted at the University of Wisconsin West Madison Agricultural Research Station on Batavia and Troxel Silt Loams in 2018 and 2019. Two areas of certified organic land (43.0734, −89.5474, and 43.0744, −89.5465) were used for the experiment (following the termination of a third-year alfalfa stand) and managed in accordance with the United States Department of Agriculture National Organic Program (USDA-NOP) regulations (National Organic Program, 2000). Soil organic matter was 3.3% in 2018 and 2.9% in 2019, and pH was 6.6 in 2018 and 7.2 in 2019. The experiment was established as a randomized complete block design with four replications, one row per replication, additional guard rows in between data rows to separate living mulch treatments (for a total of 9 rows), and 8 plants per plot at 0.61 m in-row and 2.44 m between-row spacing. Thus, each plot was 5.49 m long by 2.44 m wide (Supplementary Figure 1). Aisle mulch treatments included a cultivated control, ground straw mulch at a rate of ~31 T ha^{−1}, Dutch white clover seeded at a rate of 24.64 kg ha^{−1}, and annual ryegrass, seeded at a rate of 101.66 kg ha^{−1}, as

TABLE 3 Cumulative yield, fruit quality, and survival data of organic zucchini by living mulch treatment.

Aisle mulch	Proportion plant survival (SE)	Marketable fruit per m (SE)	Total fruit per m (SE)	Unmark. fruit per m (SE)	Proportion unmark. (SE)	Marketable fruit per plant (SE)	Total fruit per plant (SE)
Cultivated	0.86 (0.061)	15.0 (0.93) ab	22.93 (1.05) ab	7.94 (0.57)	0.26 (0.071)	8.29 (0.51) a	12.8 (0.62)
Straw	0.81 (0.061)	16.0 (0.93) a	23.95 (1.06) a	7.96 (0.57)	0.29 (0.071)	8.13 (0.51) a	12.1 (0.62)
Clover	0.83 (0.061)	11.7 (0.93) b	19.84 (1.06) b	8.18 (0.57)	0.28 (0.071)	5.69 (0.51) b	10.6 (0.62)
Ryegrass	0.84 (0.061)	11.8 (0.93) b	20.06 (1.05) ab	8.26 (0.57)	0.31 (0.071)	6.04 (0.51) b	10.8 (0.62)
Mix	0.92 (0.061)	12.5 (0.95) ab	20.57 (1.07) ab	8.10 (0.58)	0.23 (0.071)	6.58 (0.51) ab	11.1 (0.62)
Treatment effects							
Cov: Stand Ct (numDF/denDF)	NA	$F = 21.13$, $p < 0.0001$ (1/23)	$F = 20.35$, $p < 0.001$ (1/23)	$F = 2.05$, ns (1/23)	NA	NA	NA
Aisle Mulch (numDF/denDF)	$F = 0.48$, ns (4/24)	$F = 3.16$, $p < 0.05$ (4/23)	$F = 3.77$, $p < 0.05$ (4/23)	$F = 0.08$, ns (4/23)	$F = 0.22$, ns	$F = 5.85$, $p < 0.01$ (4/24)	$F = 2.60$, $p < 0.1$ (4/24)
Year (numDF/denDF)	$F = 2.23$, ns (1/6)	$F = 23.67$, $p < 0.01$ (1/6)	$F = 32.47$, $p < 0.01$ (1/6)	$F = 16.32$, $p < 0.01$ (1/6)	$F = 10.55$, $p < 0.05$	$F = 153.24$, $p < 0.0001$ (1/6)	$F = 195.51$, $p < 0.0001$ (1/6)
Aisle \times Year (numDF/denDF)	$F = 0.94$, ns (4/24)	$F = 2.09$, ns (4/23)	$F = 1.01$, ns (4/23)	$F = 0.54$, ns (4/23)	$F = 0.66$, ns	$F = 2.38$, $p < 0.1$ (4/24)	$F = 1.39$, ns (4/24)

Columns with the same letter (or no letter) were not significantly different across mulch treatments and years $p < 0.05$. Lowercase letters indicate significance groupings for the simple main effect of living mulch treatments, with a p -value adjustment using the Tukey method for comparing a family of estimates.

well as a mix of the two seeded at a rate of 15.57 kg ha⁻¹ Dutch white clover and 31.37 kg ha⁻¹ annual ryegrass.

Field activities

Field activities are summarized in Table 1. Cereal rye (*Secale cereale*) rye was seeded throughout the entire study area with a Landoll grain drill (Landoll Corporation, Marysville, KS) at a rate of 127 kg ha⁻¹ on September 25, 2017 and September 27, 2018, 2–3 weeks following the termination of a third-year alfalfa stand with a Brillion Super Soil Builder Disk Chisel (Brillion Iron Works, Brillion, WI). The following spring, rye was terminated through tillage with a Case IH JX65 tractor with 65 horsepower (Case IH, Racine, WI) with a PTO driven Land Pride RTA3576 tiller with a 1.83 m working width (Land Pride, Salinas, KS). One tillage event was adequate to terminate the rye in 2018, but a second tillage event was required in 2019. Fertilizer was broadcast applied according to University of Wisconsin-Madison Division of Extension recommendations (Laboski and Peters, 2019) based on soil test results, and was incorporated with an additional rototilling. Plastic mulch (1.22 m wide) and drip irrigation was applied in planting strips with a Mechanical Transplanter Model 85 mulch layer (Mechanical Transplanter Company, Holland, MI), ground winter wheat straw mulch was applied by hand for check plots and living mulch treatments were seeded by hand

and lightly incorporated by raking. Three-week-old “Dunja F1” zucchini summer squash (*Cucurbita pepo*) transplants grown in 50 cell trays were hand transplanted. Drip irrigation placed under the mulch was applied as needed throughout the season.

Weeds were categorized as broadleaf or grass weeds and counted within four randomly placed 0.25 m² quadrats (two each side of the data row, $n = 16$ per treatment at each date) within 24 h prior to timed manual weeding. Weeds were removed manually within the ground straw treatment and with stirrup hoes supplemented by additional hand weeding on the shoulders of beds to avoid tearing plastic within the cultivated treatment. Living mulch treatments were managed by mowing with a Simplicity 13.5 hp walk-behind brush hog (Simplicity Manufacturing, Port Washington, WI) with a 15 cm blade height, supplemented by additional hand weeding to avoid weeds reaching reproductive maturity. Total weeding time (for a single person) required for weed management after the planting of the cash crops was recorded separately for each treatment at each weeding event ($n = 4$ per treatment at each date). Weeding data was taken either when weed pressure necessitated weeding, as determined by weeds approaching flowering or being above 30 cm, or when ryegrass or mixed species living mulches needed mowing, as determined by ryegrass being above 30 cm. Cucumber beetle, squash bug egg clusters, and adult squash bugs per plant were counted as close to a weekly basis as possible ($n =$

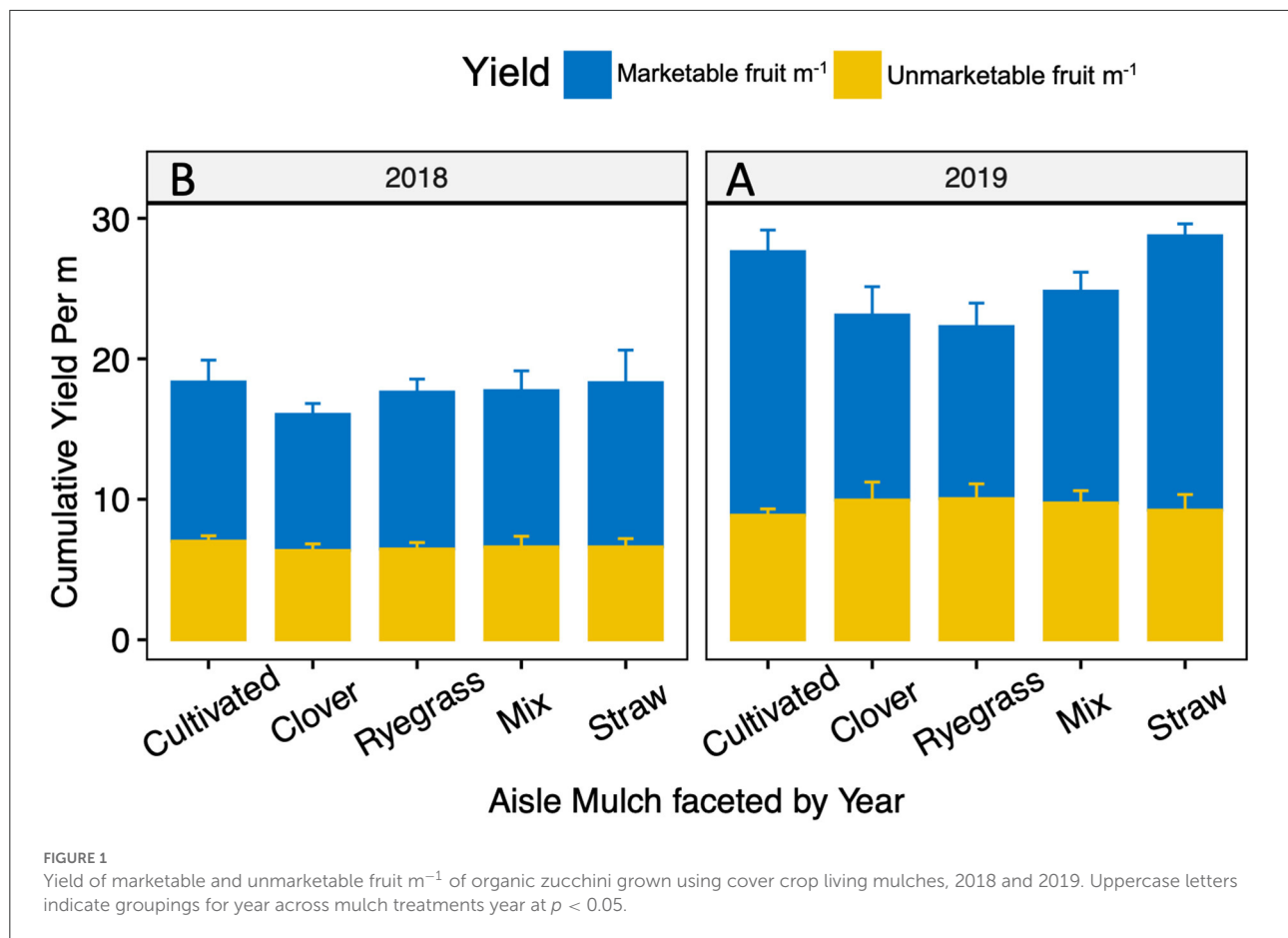


TABLE 4 Final cumulative counts of striped cucumber beetle, squash bugs and egg clusters in 2018 and 2019 on organic zucchini plants by living mulch treatment.

Aisle mulch	Cumulative cucumber beetles per m (SE)	Cumulative egg clusters per m (SE)	Cumulative squash bugs per m (SE)
Cultivated	1.41 (0.20) ab	4.10 (0.51)	1.95 (0.15) a
Straw	1.85 (0.20) a	3.74 (0.51)	1.97 (0.15) a
Clover	0.54 (0.20) c	3.08 (0.51)	0.38 (0.15) b
Ryegrass	0.46 (0.20) c	3.38 (0.51)	0.95 (0.15) ab
Mix	0.62 (0.20) bc	3.15 (0.51)	0.54 (0.15) ab
Treatment effects			
Aisle mulch (numDF/denDF)	$F = 9.93, p < 0.001$ (4/24)	$F = 0.69, ns$ (4/24)	$F = 4.26, p < 0.01$ (4/24)
Year (numDF/denDF)	$F = 30.12, p < 0.01$ (1/6)	$F = 28.40, p < 0.01$ (1/6)	$F = 1.66, ns$ (1/6)
Aisle \times Year (numDF/denDF)	$F = 9.18, p < 0.001$ (4/24)	$F = 0.32, ns$ (4/24)	$F = 0.30, ns$ (4/24)

Columns with the same letter (or no letter) were not significantly different across mulch treatments and years at $p < 0.05$.

Lowercase letters indicate significance groupings for the simple main effect of living mulch treatments, with a p -value adjustment using the Tukey method for comparing a family of estimates.

32 per treatment at each date), but were reported as per m of row for easier translation to field scales, with 1.64 plants per m.

Squash was harvested when fruit had reached marketable maturity at >15 cm, averaging every 6 days in 2018 and every

2.5 days in 2019. In each plot, the plant stand count was recorded and all squash of adequate size were harvested and sorted by quality as marketable or non-marketable. Fruit was counted as unmarketable if it showed visible evidence of rot, insect damage, surface blemishes, or was misshapen. Fruit was

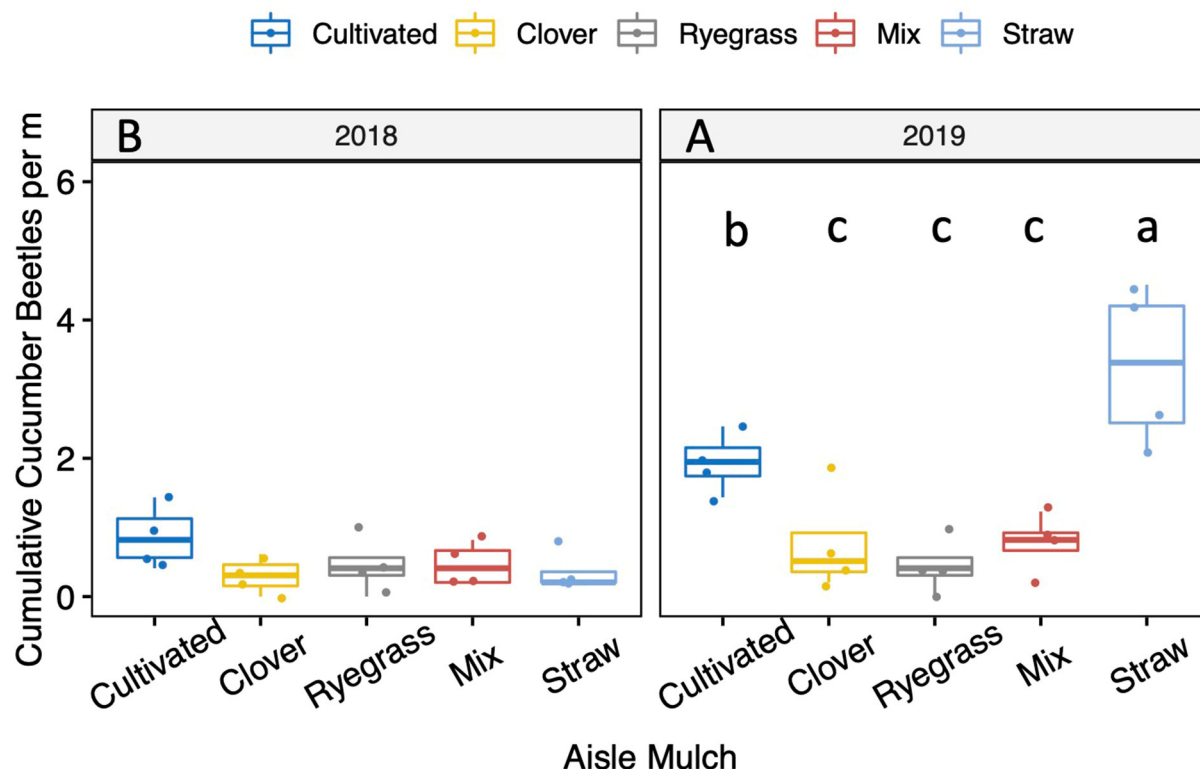


FIGURE 2

Cumulative cucumber beetles m^{-1} on organic zucchini plants grown using cover crop living mulches, averaged across 2018 and 2019. Treatments with the same lowercase letter (or no letter) were not significantly different within the same year at $p < 0.05$, while uppercase letters indicate groupings for year across mulch treatments.

counted as marketable if firm and had smooth, unblemished skins. Due to early season squash bug pressure in 2019, pyrethrin (PyGanic[®], Sumitomo Chemical, Chuo City, Tokyo, Japan) was applied once on July 31.

Data analysis

Data was analyzed in R [Rapp GUI 1.4 “Juliet Rose” (df86b69e, 2021-05-24), © R Foundation for Statistical Computing, 2021]. Analysis of Variance (ANOVA) for data such as yield, marketability, survival, weed management time, and pest counts were done using the `lme()` function in the “nlme” package (Pinheiro et al., 2022) using the following model:

$$Y_{ijk} = \mu + A_i + B_{j(i)} + M_k + \delta_{k(ji)} + SP_l + (AM)_{ik} + \epsilon_{ijk}$$

where Y_{ijk} is the observation for the i^{th} year, j^{th} block, and k^{th} aisle mulch treatment, A_i is the fixed effect of the i^{th} year ($i = 2018, 2019$), $B_{j(i)}$ is the random effect of the j^{th} block nested within the i^{th} year ($j = 1, 2, 3$), M_k is the fixed effect of the k^{th} aisle mulch treatment ($k = \text{cultivated, straw, clover, ryegrass, or mix}$), $(AM)_{ik}$ is the effect of the interaction between the i^{th}

year and k^{th} aisle mulch and ϵ_{ijk} is the residual error associated with the observation for the i^{th} year, j^{th} block, and k^{th} aisle mulch treatment.

Pest counts, harvest counts, and weed management time were transformed to cumulative counts per plot, with only the final cumulative count analyzed to meet assumptions of independent observations and improve assumptions of normality and equality of variance due to the large amount of zeros in the raw data. Use of cumulative counts and time was also chosen because the focus was on the cumulative impact of pests and weed management time, and total harvest potential in relation to mulching strategy, rather than prevalence of pests, weed management time, and harvest over time in relation to informing management decisions during the season. Analysis for weed counts included an additional subsampling error term $\gamma_{m(kjil)}$ which was the random effect of the m^{th} subsample where $m = 1, 2, 3, 4$ subsamples for weed counts. Since survival rate was not associated with aisle mulch treatment yield m^{-1} analyses also included a covariate of stand count, βX_{ijk} where β is the slope of the covariate of stand count X within the i^{th} year, j^{th} block, and k^{th} aisle mulch treatment.

Normality and equality of variances were checked visually with standardized residuals vs. fitted value plots and normal

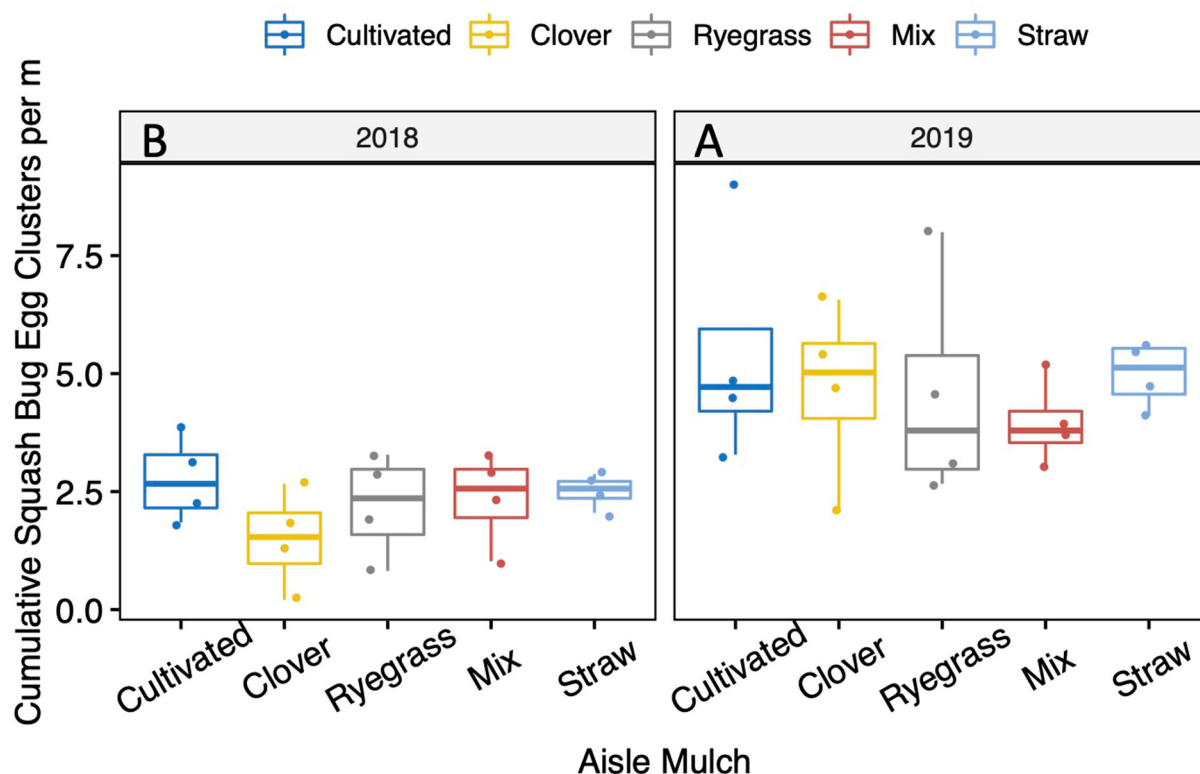


FIGURE 3
Cumulative squash bug egg clusters m^{-1} on organic zucchini plants grown using cover crop living mulches, 2018 and 2019.

QQ plots, respectively, (R Core Team, 2022). Right skewed weed count data for each ANOVA for a given dependent were transformed with $\log(x + 1)$ when necessary to improve assumptions of normality and equality of variances. When ANOVA F -tests were significant, Tukey's Multiple Comparison Procedure was used to compare treatment means and develop significance groupings using the `emmeans()` function in the "emmeans" package (Lenth, 2022), which is also how estimated marginal means for tables were obtained. When two-way interactions between main effects were found, pairwise comparisons for the simple main effect were made for each level of the other factor, again using the `emmeans()` function with a Tukey adjustment. All figures are shown with non-transformed data though significance groupings are based on transformed data when applicable.

Results and discussion

Weather

The winter and spring months leading into the 2018 growing season experienced slightly more precipitation than average and close to average temperatures with the accumulation

of more growing degree day units (GDDU) than normal, providing an environment conducive to greater cereal rye biomass accumulation as compared to 2019, which experienced a particularly cold, wet winter and a cool, dry spring (Table 2). Both 2018 and 2019 experienced more rainfall than average, with a single rain event in late August of 2018 releasing over 25 cm of rain within 24 h at the study site (MRCC, 2021).

Plant survival and fruit yield and quality

Average survival rates across both years ranged between 81% for straw mulch and 92% mixed species living mulch treatments, but was not significantly impacted by aisle mulch treatment, year, or an interaction between the two (Table 3).

Because variation in the proportion of plants that survived was observed which would affect yield m^{-1} , but aisle mulch treatments themselves did not affect this proportion, the proportion of plants surviving was used as a covariate, which had a significant effect on both cumulative marketable fruit and total fruit, but not on unmarketable fruit.

Year influenced all yield response variables. Aisle mulch affected marketable and total fruit m^{-1} but did not affect

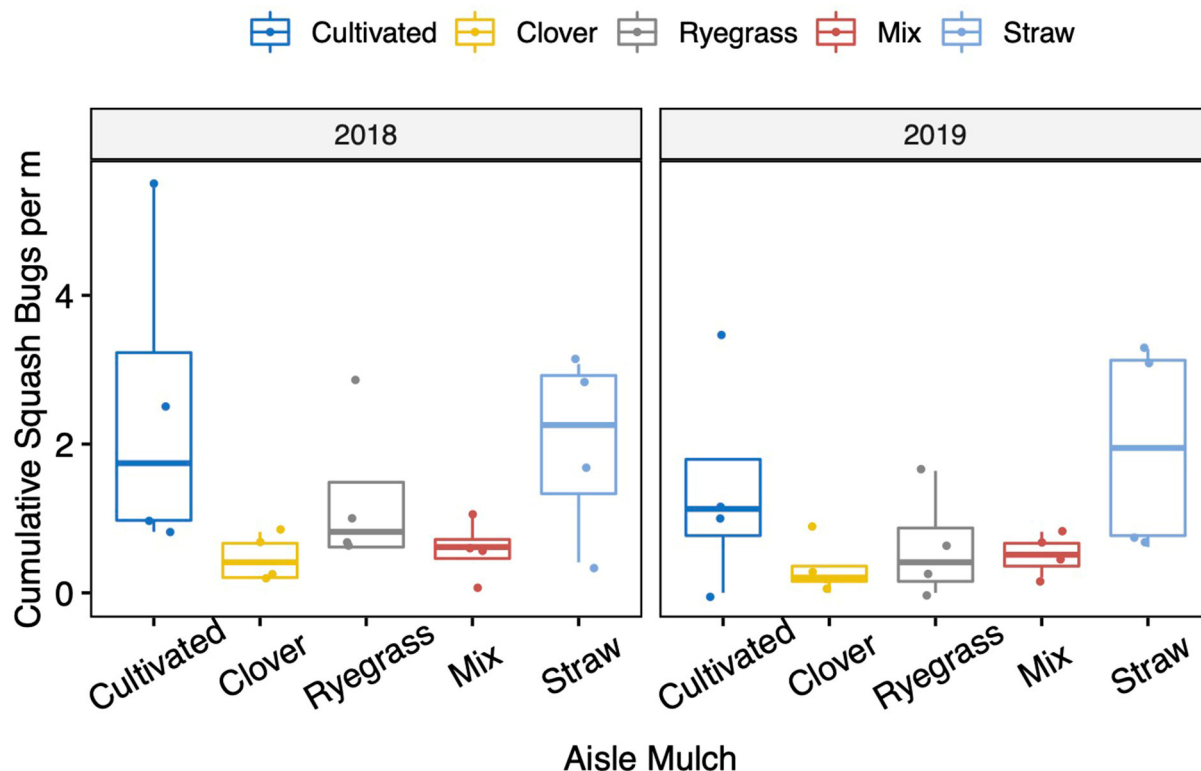


FIGURE 4

Cumulative squash bugs m^{-1} on organic zucchini plants grown using cover crop living mulches, 2018 and 2019. There was no interaction between year and aisle mulch, but across both years clover had significantly fewer squash bugs m^{-1} than cultivated or straw mulched aisles.

unmarketable fruit counts m^{-1} or the proportion of fruit that were unmarketable (Figure 1). Aisle mulch was also significant for marketable fruit plant $^{-1}$ but not total fruit plant $^{-1}$ (Supplementary Figure 2). Both fruit m^{-1} and fruit plant $^{-1}$ were analyzed since the surviving plant population ranged from 50 to 100% survival. On a m^{-1} basis, straw mulch treatments out yielded clover treatments for both marketable and total fruit, and the ryegrass treatment for marketable fruit. On a plant $^{-1}$ basis, both the straw mulch and cultivated treatments yielded more marketable fruit than clover or ryegrass treatments, while the mixed species living mulch treatment was similar to both groups.

Insect pest pressure

There was a significant year \times mulch interaction for cumulative number of cucumber beetles m^{-1} (Table 4). Although cucumber beetle pressure was negligible in 2018 and there were no differences between treatments, clear differences were evident during the 2019 season. Cultivated and straw mulch treatments resulted in higher cucumber

beetle counts m^{-1} than the clover or ryegrass treatments, while the mixed species living mulch treatment resulted in lower cucumber beetle counts as compared with the straw mulch treatment but was not different from other living mulch treatments (Figure 2). The cumulative number of squash bug egg clusters m^{-1} was not affected by aisle mulch, although more egg clusters were observed in 2019 as compared to 2018 (Figure 3). In contrast, the number of adult squash bugs m^{-1} was affected by aisle mulch but not year (Figure 4), with clover having lower counts as compared to straw or cultivated treatments. Both ryegrass and mixed species cover crop treatments were not different from either group.

Living mulch cover

A significant year \times aisle mulch interaction was observed for living mulch percent cover (Figure 5). Whereas, in 2018 ryegrass had lower coverage than both other living mulch treatments and the mixed species living mulch in turn had lower coverage than the clover treatment, in 2019 only the

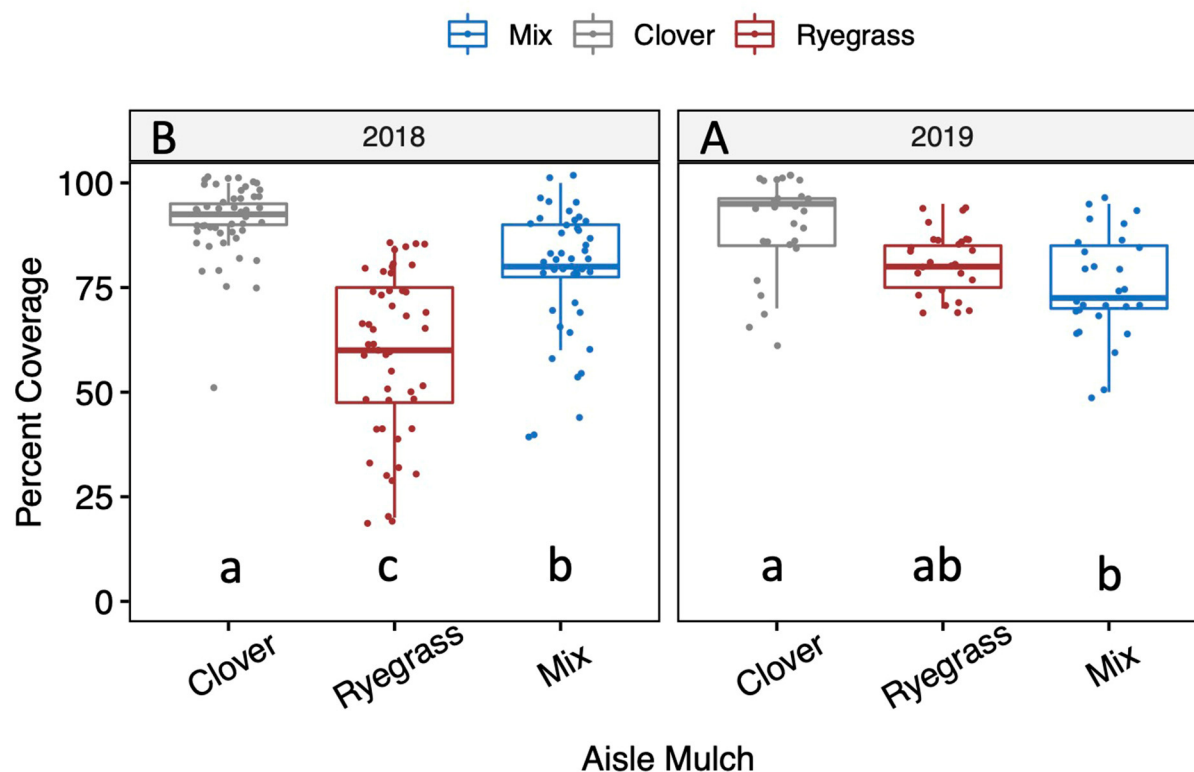


FIGURE 5

Percent ground cover provided by living mulch treatments, 2018 and 2019. Lowercase letters indicate significance groupings for mulch treatments within a given year, while uppercase letters indicate significance groupings for year across mulch treatments. Treatments that share the same letter are not significantly different at $p < 0.05$.

mixed species had less coverage than the clover treatment, and the ryegrass was not different from either group. Overall, percent cover was lower in 2018 than 2019, and across both years clover clearly had the best soil coverage at 90%, while the mixed species living mulch had lower cover at 79% and ryegrass averaged the lowest coverage at 59%.

Weed populations and management time

A significant year \times aisle mulch interaction explained the amount of total, broadleaf, and grass weeds (Figure 6 and Table 5). Across both years, the straw mulch resulted in lower weed counts than other treatments, except for ryegrass for grass weeds, where it performed similarly. For total and grass weeds, the clover resulted in greater weed numbers than all other treatments, and the cultivated treatment resulted in greater weed numbers than ryegrass and mixed species living mulch treatments. Broadleaf weed numbers were similar among all treatments except straw mulch.

Despite its notably higher weed numbers, the clover treatment required less time for weed management than cultivated aisles. The mixed species living mulch required a similar amount of weed management time as compared to the clover treatment, despite having fewer weeds. Straw mulch and ryegrass required less weed management time than all other groups (Figure 7).

Differences in weed management time relative to the quantity of weeds may have been influenced by different field crews in different years, although during a specific weed management event, the same crew member always weeded the entirety of a given block across treatments.

Discussion

Previous research demonstrated variable or negative impacts on yield when cucurbit species were produced using living mulch systems (Nyoike and Liburd, 2010; Hinds et al., 2016), although some limited results demonstrated a mitigation of yield losses when plastic mulch was laid within the planting row (Nelson and Gleason, 2018; Kahl et al., 2019).

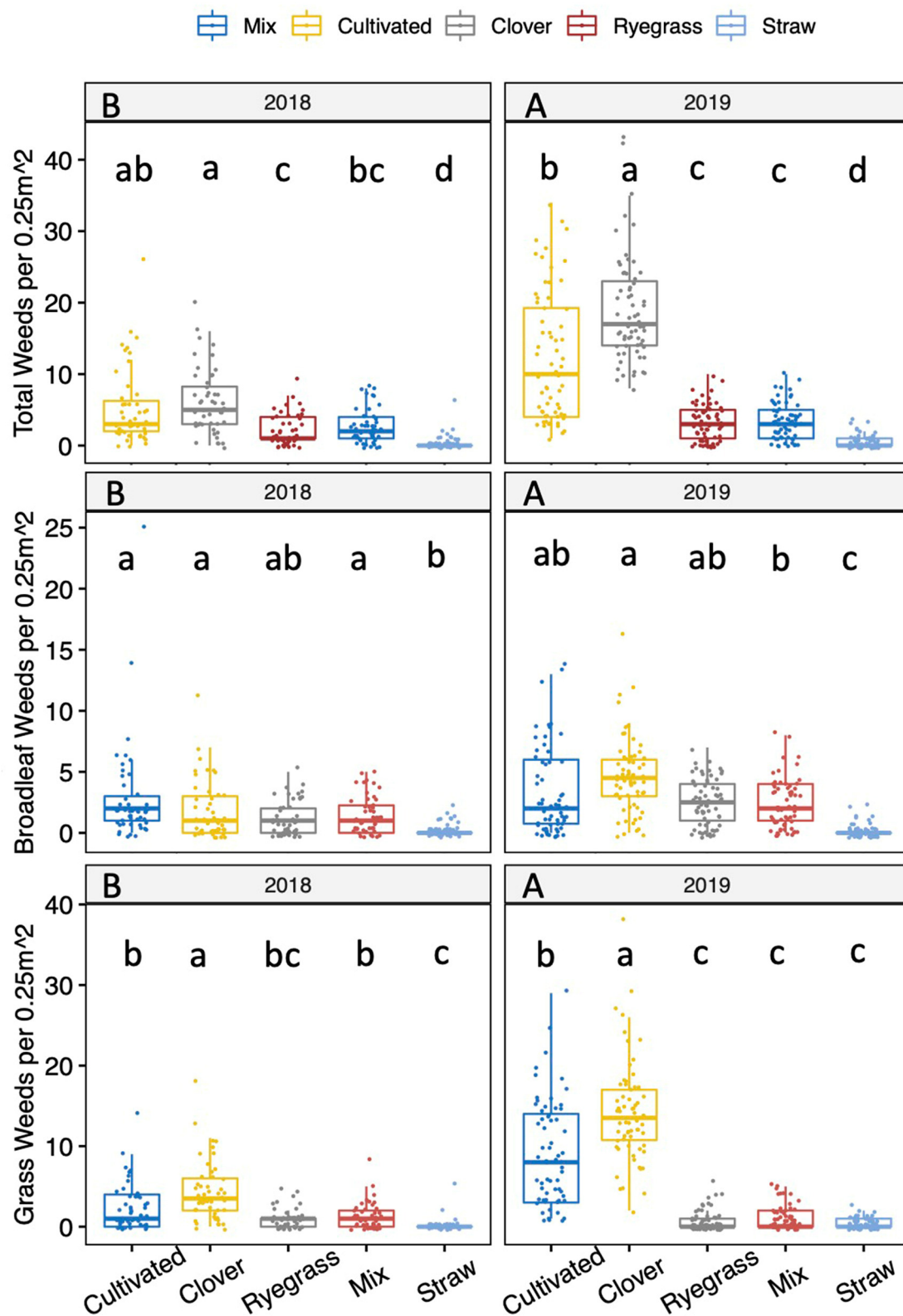


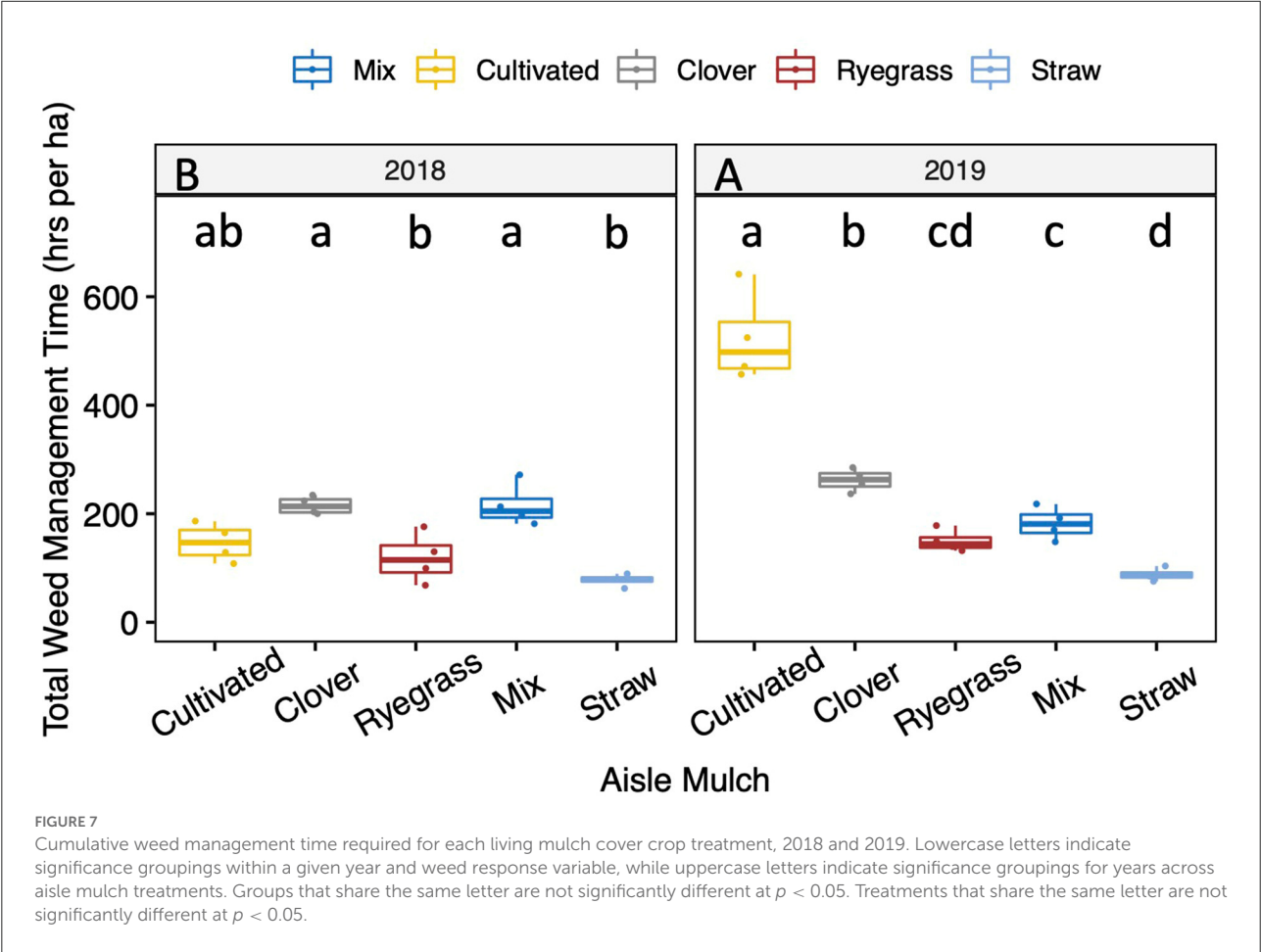
FIGURE 6

Total (top), broadleaf (center), and grass (bottom) weed counts per 0.25 m² across dates and subsamples, 2018 and 2019. Lowercase letters indicate significance groupings within a given year and weed response variable, while uppercase letters indicate significance groupings for years across aisle mulch treatments. Groups that share the same letter are not significantly different at $p < 0.05$.

TABLE 5 Least square means of cumulative weed counts and management required for organic zucchini plots grown using living mulch treatments.

Aisle mulch	Total weed Ct per. 25 m ² (SE)	Grass weed Ct per. 25 m ² (SE)	Broadleaf weed Ct per. 25 m ² (SE)	Total Weeding time, hrs per ha (SE)	Living mulch percent cover (SE)
Cultivated	8.88 (0.083) b	5.82 (0.084) b	3.04 (0.075) a	264.3 (13.3) a	-
Straw	0.48 (0.083) d	0.23 (0.084) d	0.20 (0.075) b	82.8 (13.3) c	-
Clover	12.49 (0.083) a	9.27 (0.084) a	3.22 (0.075) a	238.6 (13.3) b	90.34% (1.62) a
Ryegrass	2.72 (0.083) c	0.85 (0.084) cd	1.88 (0.075) a	134.0 (13.3) c	59.12% (1.62) c
Mix	3.00 (0.083) c	1.13 (0.084) c	1.88 (0.075) a	198.9 (13.3) b	78.96% (1.62) b
Treatment effects					
Aisle Mulch (numDF/denDF)	<i>F</i> = 98.60, <i>p</i> < 0.0001 (4/24)	<i>F</i> = 121.03, <i>p</i> < 0.0001 (4/24)	<i>F</i> = 24.14, <i>p</i> < 0.0001 (4/24)	<i>F</i> = 53.36, <i>p</i> < 0.0001 (4/24)	<i>F</i> = 54.87, <i>p</i> < 0.0001 (2/12)
Year (numDF/denDF)	<i>F</i> = 49.49, <i>p</i> < 0.0001 (1/6)	<i>F</i> = 47.95, <i>p</i> < 0.001 (1/6)	<i>F</i> = 18.55, <i>p</i> < 0.01 (1/6)	<i>F</i> = 52.21, <i>p</i> < 0.0001 (1/6)	<i>F</i> = 9.20, <i>p</i> < 0.05 (1/6)
Aisle × Year (numDF/denDF)	<i>F</i> = 7.93, <i>p</i> < 0.001 (4/24)	<i>F</i> = 20.04, <i>p</i> < 0.0001 (4/24)	<i>F</i> = 4.32, <i>p</i> < 0.01 (4/24)	<i>F</i> = 38.43, <i>p</i> < 0.0001 (4/24)	<i>F</i> = 19.39, <i>p</i> < 0.001 (2/12)

Columns with the same letter were not significantly different across mulch treatments and years at *p* < 0.05. Lowercase letters indicate significance groupings for the simple main effect of aisle mulch, with results averaged across blocks, dates, and samples and a *p*-value adjustment using the Tukey method for comparing a family of estimates.



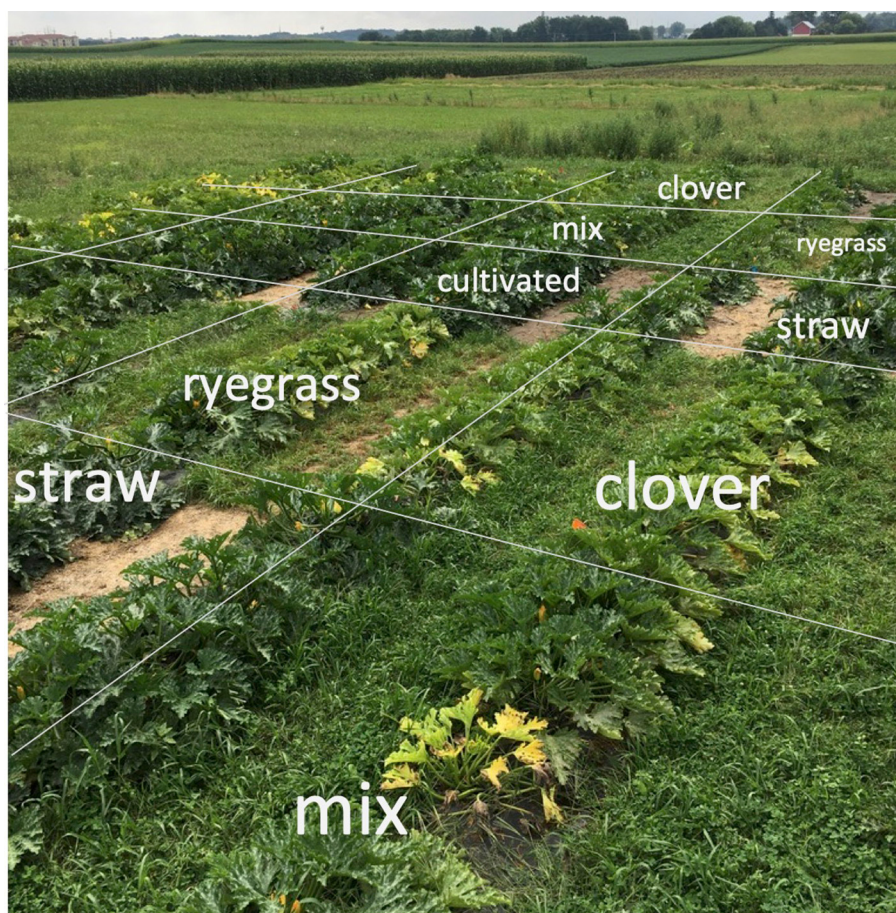


FIGURE 8

Photograph of organic zucchini plants grown in living mulch treatments, 2019. Note that the particularly stunted and yellowed plant in the foreground mixed species living mulch plot was afflicted with fusarium wilt, whereas in many other living mulch plots the plants were yellowing without any particular disease identifiable as the cause.

Our results reinforce the risk of reduced yield in living mulch systems, with lower marketable yield in treatments using clover and annual ryegrass as a living mulch as compared to managing the aisles using cultivation or straw mulch. However, using the mixture of annual ryegrass and clover performed comparably to the more standard management practices of cultivation and straw mulch. Management practices to reduce the potential for competition between cover crops and cash crops, such as the regular mowing of living mulches to a height of 15 cm (Bâth et al., 2008; Hinds et al., 2016), and the management of the planting strip using plastic mulch (Nelson and Gleason, 2018), did not fully mitigate reduced yields in our study. While low mowing has the potential to result in reduced competition or mitigate cash crop yield loss (Liu and Huang, 2002; Hinds et al., 2016), future studies could compare mowing with mechanical root pruning, which has also been suggested as a practice to reduce living mulch competition with cash crops (Bâth et al., 2008).

Our results supported previous studies suggesting potential benefits of living mulches for reducing pest pressure (Nyoike and Liburd, 2010; Grasswitz, 2013; Kahl et al., 2019). However, our results should be interpreted in the context of relatively low overall pest pressure, apart from early season squash bug pressure in 2019, which was high enough that we chose to apply pyrethrin once in order to ensure enough marketable harvest data. In addition, the relatively small plot size may have introduced more noise due to the mobility of the pests evaluated in this study.

In contrast to Grasswitz's observation that living mulch systems resulted in greater squash bug pressure as compared to standard management, our results showed no clear differences between management approaches for the numbers of squash bug eggs. However, the lower numbers of adults resulting from the use of living mulch cover crops observed in our study could be due to increased natural predators in living mulch systems (Nyoike and Liburd, 2010; Grasswitz, 2013; Kahl et al., 2019),

although reduced plant health or N content in living mulch plots could also have caused pests to prefer control treatments (Mauck et al., 2010). Given the prevalence of plastic mulch for producers, future studies could investigate whether the same mechanism of increased predator populations might be responsible for reduced pest pressure in plasticulture systems with living mulch.

Our results also support previous research indicating that clover does not adequately suppress weeds during the establishment year (MacLaren et al., 2019; Tarrant et al., 2020). Soil coverage by the cover crop, a potential indicator of light competition (Place et al., 2011), did not appear to be the significant driver of reduced weed counts in our study, given that clover resulted in a higher percent coverage than ryegrass or the mixed species treatments but still had higher weed counts. The clover treatment also had a consistently lower yield m^{-1} . As compared with the cultivated control, ryegrass reduced weed counts both years, but still yielded fewer marketable fruit. The mixed species performed best out of the living mulch treatments, with weed control comparable to ryegrass and yields equivalent to the cultivated and straw mulch controls.

The use of annual ryegrass and an annual ryegrass/clover mix resulted in better weed suppression as compared to a clover cover crop alone. Results from Tarrant et al. (2020) suggest that both ryegrass and clover have the potential to reduce soil nitrate and moisture within the cash crop row relative to cultivated controls, supporting the negative impacts on yields observed in both treatments in our study. Anecdotally, chlorosis was visible in living mulch treatments in 2019 (Figure 8), suggesting that nutrient competition between cash and cover crops may have contributed to reduced yields. Lower N content in the cash crop grown with living mulches may also have resulted in reduced pest preference for those plants, while on the other hand poor plant health can also reduce tolerance to pests (Magdoff and Van Es, 2000; Altieri and Nicholls, 2003).

Given the equivalent proportions of unmarketable fruit, benefits for pest control, and even comparable weed control in ryegrass as compared to straw mulch, future studies could address the potential of nutrient and water resource competition as a possible driver of reduced marketable fruit yields in living mulch cucurbit systems. Analyzing nutrient and water status of both cash crop and cover crops and testing supplementary fertilizer, such as has been done in other crops (e.g., Fracchiolla et al., 2020 or Warren et al., 2015), may help understand the role of cover crop competition in reducing cash crop yield.

In one of the two years of our study, managing the aisle as bare ground required significantly longer weed management time as compared to managing the aisles using any of the cover crop treatments. Similar to the observation of Butler et al. (2013) that a single mowing event is not adequate to

eliminate some weed species' reproductive capacity, the greatest proportion of the weed management time required for living mulch treatments was in additional hand weeding to remove weeds not terminated completely by the mower. Anecdotally, most of the hand weeding required was found outside of the mower management zone, either below the mower deck, or at the shoulders of the bed underneath the more mature cash crop canopy encroaching into the aisle. However, all treatments were weeded completely clean at each weeding event to create equivalent conditions between the living mulch treatments and the bare cultivated control. In a more practical circumstance, farmers may have a higher tolerance for weed pressure, in which case simply mowing the living mulch treatments may suffice.

Clover had higher weed counts than cultivated aisles in 2019, yet required less management time, indicating the use of mowing as a management tool in living mulch treatments likely hindered weed growth, thus contributing to reduced impact of higher weed counts in 2019 as compared with 2018. Despite significantly higher weed counts in 2019 as compared to 2018 across all mulch treatments, only the cultivated treatment took longer for weed management in the second year, whereas straw and living mulches had equivalent management times between years. Our results suggest that alongside traditional organic and plastic mulches, living mulch managed with mowing has potential to mitigate some of the increased management time associated with very weedy conditions, whereas in less weedy conditions they may take longer to manage than traditional options like straw mulch or cultivation.

Conclusions

This study contributes to our further understanding of effects of living mulch on weed and pest pressure, with the system demonstrating potential for agroecosystem benefits but variable impacts on cash crop yield. Further research over multiple years, across multiple environments and with additional crops will contribute to our understanding of the system's performance across organic vegetable farms. While pest pressure was low during both years of our study, production environments experiencing greater pest pressure may benefit more from the use of living mulches. However, to reduce the risk associated with the adoption of these practices, future research should address potential economic and management considerations such as weed management thresholds, supplementary weed management methods. It is also important to investigate the competition potential between cover crops and cash crops, how nutrient status influences pest preference, specific causes of unmarketability, and yield response to supplementary fertilizer.

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

Author contributions

ES: conceptualization and funding application. ES and DB: research design. DB: data collection. DB, ES, and JD: data analysis/interpretation, writing, and editing manuscript. All authors contributed to the article and approved the submitted version.

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References

- Altieri, M. A., and Nicholls, C. I. (2003). Soil fertility management and insect pests: harmonizing soil and plant health in agroecosystems. *Soil Tillage Res.* 72, 203–211. doi: 10.1016/S0167-1987(03)00089-8
- Amirault, J. P., and Caldwell, J. S. (1998). Living mulch strips as habitats for beneficial insects in the production of cucurbits. *HortScience* 33, 524–525. doi: 10.21273/HORTSCI.33.3.524f
- Arnold, G. L., Luckenbach, M. W., and Unger, M. A. (2004). Runoff from tomato cultivation in the estuarine environment: biological effects of farm management practices. *J. Exp. Mar. Biol. Ecol.* 298, 323–346. doi: 10.1016/S0022-0981(03)00366-6
- Båth, B., Kristensen, H. L., and Thorup-Kristensen, K. (2008). Root pruning reduces root competition and increases crop growth in a living mulch cropping system. *J. Plant Interact.* 3, 211–221. doi: 10.1080/17429140801975161
- Baraibar, B., Hunter, M. C., Schipanski, M. E., Hamilton, E., and Mortensen, D. A. (2018). Weed suppression in cover crop monocultures and mixtures. *Weed Sci.* 66, 121–133. doi: 10.1017/wsc.2017.59
- Bezuidenhout, S. R., Reinhardt, C. F., and Whitwell, M. I. (2012). Cover crops of oats, strolling rye and three annual ryegrass cultivars influence maize and *Cyperus esculentus* growth. *Weed Res.* 52, 153–160. doi: 10.1111/j.1365-3180.2011.00900.x
- Brainard, D. C., Peachey, R. E., Haramoto, E. R., Luna, J. M., and Rangarajan, A. (2013). Weed ecology and nonchemical management under strip-tillage: implications for northern US vegetable cropping systems. *Weed Technol.* 27, 218–230. doi: 10.1614/WT-D-12-00068.1
- Brown, B., and Gallandt, E. R. (2018). A systems comparison of contrasting organic weed management strategies. *Weed Sci.* 66, 109–120. doi: 10.1017/wsc.2017.34
- Brust, J., Claupein, W., and Gerhards, R. (2014). Growth and weed suppression ability of common and new cover crops in Germany. *Crop Prot.* 63, 1–8. doi: 10.1016/j.cropro.2014.04.022
- Butler, R. A., Brouder, S. M., Johnson, W. G., and Gibson, K. D. (2013). Response of four summer annual weed species to mowing frequency and height. *Weed Technol.* 27, 798–802. doi: 10.1614/WT-D-12-00112.1
- Deguchi, S., Uozumi, S., Touno, E., Kaneko, M., and Tawarayama, K. (2012). Arbuscular mycorrhizal colonization increases phosphorus uptake and growth of corn in a white clover living mulch system. *Soil Sci. Plant Nutr.* 58, 169–172. doi: 10.1080/00380768.2012.662697
- Doughty, H. B., Wilson, J. M., Schultz, P. B., and Kuhar, T. P. (2016). Squash bug (*Hemiptera: Coreidae*): biology and management in cucurbitaceous crops. *J. Integr. Pest Manag.* 7, 1. doi: 10.1093/jipm/pmv024
- Fracchiolla, M., Renna, M., D'Imperio, M., Lasorella, C., Santamaria, P., and Cazzato, E. (2020). Living mulch and organic fertilization to improve weed management, yield and quality of broccoli raab in organic farming. *Plants* 9, 177. doi: 10.3390/plants9020177
- Grasswitz, T. R. (2013). Development of an insectary plant mixture for New Mexico and its effect on pests and beneficial insects associated with pumpkins. *Southwest. Entomol.* 38, 417–436. doi: 10.3958/059.038.0306
- Haber, A. I., Wallingford, A. K., Grettenberger, I. M., Ramirez Bonilla, J. P., Vinchesi-Vahl, A. C., Weber, D. C. (2021). Striped cucumber beetle and western

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Supplementary material

The Supplementary Material for this article can be found online at: <https://www.frontiersin.org/articles/10.3389/fsufs.2022.995224/full#supplementary-material>

SUPPLEMENTARY FIGURE 1

RCBD layout with four replications. 2019 layout is shown here, while 2018 was similar but randomized differently.

SUPPLEMENTARY FIGURE 2

Yield of marketable and unmarketable plant⁻¹ of organic zucchini grown using cover crop living mulches, 2018 and 2019. Uppercase letters indicate groupings for year across mulch treatments year at $p < 0.05$.

- striped cucumber beetle (Coleoptera: Chrysomelidae). *J. Integr. Pest Manag.* 12, 1. doi: 10.1093/jipm/pmaa026
- Hartwig, N. L., and Ammon, H. U. (2002). Cover crops and living mulches. *Weed Sci.* 50, 688–699. doi: 10.1614/0043-1745(2002)050[0688:AIACCA]2.0.CO;2
- Hiltbrunner, J., Jeanneret, P., Liedgens, M., Stamp, P., and Streit, B. (2007). Response of weed communities to legume living mulches in winter wheat. *J. Agron. Crop Sci.* 193, 93–102. doi: 10.1111/j.1439-037X.2007.00250.x
- Hinds, J., Wang, K. H., and Hooks, C. R. (2016). Growth and yield of zucchini squash (*Cucurbita pepo* L.) as influenced by a sunn hemp living mulch. *Biol. Agric. Hort.* 32, 21–33. doi: 10.1080/01448765.2015.1017736
- Jabbour, R., Pisani-Gareau, T., Smith, R. G., Mullen, C., and Barbercheck, M. (2016). Cover crop and tillage intensities alter ground-dwelling arthropod communities during the transition to organic production. *Renew. Agric. Food Syst.* 31, 361–374. doi: 10.1017/S1742170515000290
- Jenkins, D., and Ory, J. (2016). *National Organic Research Agenda. Organic Farming Research Foundation*. Santa Cruz CA.
- Kahl, H. M., Leslie, A. W., and Hooks, C. R. (2019). Effects of red clover living mulch on arthropod herbivores and natural enemies, and cucumber yield. *Ann. Entomol. Soc. Am.* 112, 356–364. doi: 10.1093/aesa/say036
- Kasirajan, S., and Ngouajio, M. (2012). Polyethylene and biodegradable mulches for agricultural applications: a review. *Agron. Sustain. Dev.* 32, 501–529. doi: 10.1007/s13593-011-0068-3
- Laboski, C. A. M., and Peters, J. B. (2019). *Nutrient Application Guidelines for Field, Vegetable, and Fruit Crops in WI. A2809*. University of Wisconsin Cooperative Extension.
- Law, D. M., Rowell, A. B., Snyder, J. C., and Williams, M. A. (2006). Weed control efficacy of organic mulches in two organically managed bell pepper production systems. *HortTechnology* 16, 225–232. doi: 10.21273/HORTTECH.16.2.0225
- Lenth, R. V. (2022). *Emmeans: Estimated Marginal Means, aka Least-Squares Means*. R package version 1.7.2. Available online at: <https://CRAN.R-project.org/package=emmeans>
- Liebman, M., Gallandt, E. R., and Jackson, L. E. (1997). Many little hammers: ecological management of crop-weed interactions. *Ecol. Agric.* 1, 291–343. doi: 10.1016/B978-012378260-1/50010-5
- Liu, X., and Huang, B. (2002). Mowing effects on root production, growth, and mortality of creeping bentgrass. *Crop Sci.* 42, 1241–1250. doi: 10.2135/cropsci2002.1241
- MacLaren, C., Swanepoel, P., Bennett, J., Wright, J., and Dehnen-Schmutz, K. (2019). Cover crop biomass production is more important than diversity for weed suppression. *Crop Sci.* 59, 733–748. doi: 10.2135/cropsci2018.05.0329
- Magdoff, F., and Van Es, H. (2000). *Building Soils for Better Crops* (No. 631.584/M188b). Beltsville: Sustainable Agriculture Network.
- Mauck, K. E., De Moraes, C., and Mescher, M. C. (2010). Deceptive chemical signals induced by a plant virus attract insect vectors to inferior hosts. *Proc. Natl. Acad. Sci. U.S.A.* 107, 3600–3605. doi: 10.1073/pnas.0907191107
- Moyinhan, M. (2010). *Status of Organic Agriculture in Minnesota*. Available online at: <https://www.leg.mn.gov/docs/2010/mandated/100851.pdf> (accessed July 4, 2022).
- MRCC (2021). *Daily precipitation and temperatures – UW-Madison Arboretum Station*. Midwestern Regional Climate Center (accessed July 4, 2022).
- National Organic Program (2000). *Final Rule. Federal Register*, Vol. 65, p. 80548–96.
- Nelson, H., and Gleason, M. (2018). Comparing between-row mulches in organic muskmelon and squash-year 2. *2017 Farm Progress Rep.* 19–21. Available online at: <https://dr.lib.iastate.edu/server/api/core/bitstreams/34683d4b-2588-4423-89a3-b69a25cc1034/content>
- Nyoi, T. W., and Liburd, O. E. (2010). Effect of living (buckwheat) and UV reflective mulches with and without imidacloprid on whiteflies, aphids and marketable yields of zucchini squash. *Int. J. Pest Manag.* 56, 31–39. doi: 10.1080/09670870902991815
- Pfeiffer, A., Silva, E., and Colquhoun, J. (2016). Living mulch cover crops for weed control in small-scale applications. *Renew. Agric. Food Syst.* 31, 309–317. doi: 10.1017/S1742170515000253
- Pinheiro, J., Bates, D., and R Core Team. (2022). *nlme: Linear and Nonlinear Mixed Effects Models*. R package version 3.1-157. Available online at: <https://CRAN.R-project.org/package=nlme>
- Place, G. T., Reberg-Horton, S. C., Dickey, D. A., and Carter, T. E. Jr. (2011). Identifying soybean traits of interest for weed competition. *Crop Sci.* 51, 2642–2654. doi: 10.2135/cropsci2010.11.0654
- R Core Team (2022). *R: A Language and Environment for Statistical Computing*. R Foundation for Statistical Computing, Vienna, Austria. Available online at: <https://www.R-project.org/>
- R Foundation for Statistical Computing (2021). *R: A Language and Environment for Statistical Computing*. R Foundation for Statistical Computing, Vienna, Austria. Available online at: <https://www.R-project.org/> (accessed July 4, 2022).
- Reid, J., and Klotzbach, K. (2013). *2013 Final Report for ONE12-171*. Available online at: <https://projects.sare.org/project-reports/one12-171/> (accessed July 4, 2022).
- Rice, P. J., Harman-Fetcho, J. A., Teasdale, J. R., Sadeghi, A. M., McConnell, L. L., Coffman, C. B., et al. (2004). Use of vegetative furrows to mitigate copper loads and soil loss in runoff from polyethylene (plastic) mulch vegetable production systems. *Environ. Toxicol. Chem. Int. J.* 23, 719–725. doi: 10.1897/03-14
- Sarrantonio, M. (1992). Opportunities and challenges for the inclusion of soil-improving crops in vegetable production systems. *HortScience* 27, 754–758. doi: 10.21273/HORTSCI.27.7.754
- Sarrantonio, M., and Gallandt, E. (2003). The role of cover crops in North American cropping systems. *J. Crop Prod.* 8, 53–74. doi: 10.1300/J144v08n01_04
- Snapp, S. S., Swinton, S. M., Labarta, R., Mutch, D., Black, J. R., Leep, R., et al. (2005). Evaluating cover crops for benefits, costs, and performance within cropping system niches. *Agron. J.* 97, 322–332. doi: 10.2134/agronj2005.0322a
- Spertelli, M., Frascioni, C., Fontanelli, M., Pirchio, M., Gagliardi, L., Raffaelli, M., et al. (2022). Innovative living mulch management strategies for organic conservation field vegetables: evaluation of continuous mowing, flaming, and tillage performances. *Agronomy* 12, 622. doi: 10.3390/agronomy12030622
- Steinmetz, Z., Wollmann, C., Schaefer, M., Buchmann, C., David, J., Tröger, J., et al. (2016). Plastic mulching in agriculture. Trading short-term agronomic benefits for long-term soil degradation? *Sci. Total Environ.* 550, 690–705. doi: 10.1016/j.scitotenv.2016.01.153
- Tarrant, A. R., Brainard, D. C., and Hayden, Z. D. (2020). Cover crop performance between plastic-mulched beds: impacts on weeds and soil resources. *HortScience* 55, 1069–1077. doi: 10.21273/HORTSCI.14956-20
- Teasdale, J. R., and Mohler, C. L. (2000). The quantitative relationship between weed emergence and the physical properties of mulches. *Weed Sci.* 48, 385–392. doi: 10.1614/0043-1745(2000)048[0385:TQRBWE]2.0.CO;2
- Teasdale, J. R., Rice, C. P., Cai, G., and Mangum, R. W. (2012). Expression of allelopathy in the soil environment: soil concentration and activity of benzoxazinoid compounds released by rye cover crop residue. *Plant Ecol.* 213, 1893–1905. doi: 10.1007/s11258-012-0057-x
- USDA-NASS (2019). *2019 Organic Survey (2017) Census of Agriculture Special Report. United States Department of Agriculture – National Agricultural Statistics Service*. Available online at: https://www.nass.usda.gov/Surveys/Guide_to_NASS_Surveys/Organic_Production/ (accessed July 4, 2022).
- Walters, S. A., Soloneski, S., and Larramendy, M. L. (2011). *Weed Management Systems for No-Tillage Vegetable Production*. Herbicides, theory and applications.
- Warren, N. D., Smith, R. G., and Sideman, R. G. (2015). Effects of living mulch and fertilizer on the performance of broccoli in plasticulture. *HortScience* 50, 218–224. doi: 10.21273/HORTSCI.50.2.218
- Wauters, V. M., Grossman, J. M., Pfeiffer, A., and Cala, R. (2021). Ecosystem services and cash crop tradeoffs of summer cover crops in northern region organic vegetable rotations. *Front. Sustain. Food Syst.* 5, 635955. doi: 10.3389/fsufs.2021.635955



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EDITED BY

Jacob Jungers,
University of Minnesota Twin Cities,
United States

REVIEWED BY

Jessica L. M. Gutknecht,
University of Minnesota Twin Cities,
United States
Joshua Gamble,
Agricultural Research Service (USDA),
United States

*CORRESPONDENCE

Louise M. Egerton-Warburton
legerton@chicagobotanic.org

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Rapid improvement in soil health following the conversion of abandoned farm fields to annual or perennial agroecosystems

Lucas A. Chamberlain^{1,2}, Teresa Aguayo³, Nyree J. C. Zerega^{1,2},
Ray Dybzinski⁴ and Louise M. Egerton-Warburton^{1,2*}

¹Plant Biology and Conservation, Northwestern University, Evanston, IL, United States, ²Chicago Botanic Garden, Negaunee Institute for Plant Conservation Science and Action, Glencoe, IL, United States, ³Department of Biology, Missouri State University, Springfield, MO, United States, ⁴School of Environmental Sustainability, Loyola University Chicago, Chicago, IL, United States

Incorporating perennial crops into agroecosystems has been shown to mitigate soil degradation and improve soil health by enhancing soil aggregation and soil organic carbon (SOC) accrual. However, our understanding of the ability and timeframe for perennial crop systems to build soil health within the context of conversion from abandoned crop land remains limited. Here, we examined changes in soil health in the first year following the conversion of an abandoned crop field into an agroecosystem planted with various treatments, including: novel perennial grain (intermediate wheatgrass, IWG; *Thinopyrum intermedium*), IWG/ alfalfa biculture, forage grass, tallgrass prairie, or annual wheat. We analyzed factors considered central to the concept of mitigating soil degradation to improve soil health (soil aggregation, aggregate organic carbon (OC), bulk SOC) and their soil biological and physicochemical correlates throughout the first growing season. Comparisons between treatments showed that both annual and perennial treatments rapidly and significantly improved soil health metrics including aggregation, aggregate stability, and OC levels compared to pre-conversion conditions. Such increases were positively correlated with the abundance of arbuscular mycorrhizal fungi (AMF hyphae, root colonization), labile SOC and microbial activity. Notably, IWG/ alfalfa biculture resulted in significantly higher levels of macroaggregate OC in comparison to other treatments, including tallgrass prairie, supporting the potential of perennial grasses to contribute to soil carbon gains. Overall, the conversion of this abandoned land to an agroecosystem produced rapid and substantial increases in soil health in the first year after planting.

KEYWORDS

agroecosystem, intermediate wheatgrass, soil organic carbon, soil aggregation, arbuscular mycorrhizal fungi

Introduction

Agricultural practices have contributed to widespread soil degradation by contributing to nutrient loss, erosion, salinity, and compaction and by reducing carbon storage (Lal, 2015). In response, there has been a push to adopt management practices that minimize soil physical disturbances (e.g., no-till), maximize surface coverage (e.g., cover crops), stimulate biological activity (e.g., organic amendments), and use polycultures or perennial crops to regenerate ecosystem services (Sprunger and Robertson, 2018). Of these, planting perennial crops or polycultures is expected to produce soil health benefits comparable to native ecosystems such as enhanced soil stability and increased water and nutrient cycling (Glover et al., 2010; Syswerda et al., 2012; Pugliese et al., 2019; Ledo et al., 2020), but this is not always the case (Johnson et al., 2021). These benefits are largely driven by soil organic carbon (SOC), a factor considered central to the concept of soil health (Bünemann et al., 2018). Establishing perennial crops on agricultural lands can rebuild SOC stocks by 19–39%, especially in the upper soil layers (Post and Kwon, 2010; Ledo et al., 2020), and on short time-scales (within two years; Sprunger and Robertson, 2018; Peixoto et al., 2020). The deep, extensive root systems of perennial species may also create significant subsoil C pools (>2 m depth). Peixoto et al. (2020) reported that perennial crops allocated substantial amounts of fresh root residues and exudates to subsoils (3.6 m deep). In turn, the substrates were metabolized by soil microbial communities and transformed into microbial necromass, a contributor to C stabilization.

Besides C inputs, soil aggregate dynamics influence SOC accrual and thus soil health (Totsche et al., 2018). For example, macroaggregates (>250 μm diameter) are formed as transient organic binding agents (e.g., fine roots, microbial mucilage, arbuscular mycorrhizal fungal (AMF) hyphae) enmesh and bind soil particles and organic matter (Jastrow et al., 2007). Over time, the plant and microbial residues within macroaggregates become encrusted onto mineral surfaces, humify, and condense to form stable organic C (OC)-rich microaggregates (20–250 μm diameter) that contribute to decadal—to century—scale SOC sequestration (Jastrow et al., 2007). Perennialization is thus expected to increase the abundance of macroaggregates and macroaggregate OC in the short-term (Chivenge et al., 2011), and microaggregate OC and SOC pools in the longer term (Virto et al., 2012; Novelli et al., 2017).

At the same time, there is increasing recognition that above- and below-ground components of ecosystems are closely linked (Kardol and Wardle, 2010). For example, plant species may differentially influence soil physicochemical properties including pH, organic matter content, soil structure and microclimate as well as the quality and quantity of root litter and the supply of C to root symbioses (e.g., mycorrhizas, and root exudates that support rhizosphere microbes) (De Deyn et al., 2008). In addition, root growth differs among species (Bergmann et al., 2016) and may physically alter the soil structure to

create various physical and metabolic microhabitats (Freschet et al., 2021). Plant species identity may thus be an important consideration in regenerating soil health.

Perennial grasses show promise in land regeneration efforts. One emerging perennial grass species, *Thinopyrum intermedium* (Host) Barkworth and D.R. Dewey [intermediate wheatgrass (IWG), Kernza®], has the potential to rapidly enhance soil health. In crop fields, studies have reported that IWG may rapidly improve soil quality (Culman et al., 2010, 2013), SOC gain (Sprunger et al., 2017, 2019; Sprunger and Robertson, 2018), and water quality (Culman et al., 2013) in comparison to annual wheat (*Triticum aestivum* L.), but not always (Syswerda et al., 2011; Sprunger et al., 2018). IWG may have a similarly large potential for regenerating abandoned and/or degraded lands. However, the relative importance of IWG and its interaction with factors at local scales is poorly understood, thereby limiting quantitative predictions of how perennial crops could reverse land degradation.

To address this knowledge gap, we converted an abandoned old field to replicated plots containing IWG (monoculture or biculture with alfalfa—*Medicago sativa* L.), annual wheat, forage grasses, and tallgrass prairie. Over the first growing season, we measured the abundance of water-stable aggregates (WSA), aggregate and bulk SOC content, and soil biological and physicochemical variables that are expected to contribute to variations in WSA and SOC. Our goals were to: (1) investigate the role of crop type in soil aggregate stability and OC accrual in a newly-established agroecosystem on abandoned land, (2) identify the abiotic or biotic factors that enhanced the formation of water-stable (macro) aggregates, and (3) use these data to test the hypothesis that planting IWG in abandoned fields produces (a) increases in soil health factors that are comparable to those of a perennial tallgrass prairie restoration and (b) more rapid and substantial increases in soil health than those produced by annual crops (wheat, forage grasses).

Materials and methods

Study site and experimental design

The study was conducted in Mettawa, IL (42°14' N, 87°55' W; 191 m a.s.l.) in a field that had last been used for agriculture and pasture ~30 years ago and then abandoned (W. Kurtis; personal correspondence). Since then, the site was mowed several times each year and occasionally used to park cars. The area has a temperate, humid mid-continental climate with average annual minimum and maximum daily temperatures of 4.61°C and 15°C respectively (1991–2020, NOAA), and average annual precipitation is 1003.55 mm, the majority of which is deposited throughout the growing season (April–October). The soils belong to the Nappanee and Montgomery series and are described as deep, somewhat poorly drained silty clay loams with moderate shrink-swell potential (USDA-NRCS, 2019). Prior to

conversion, the plant community comprised a mix of native and non-native grasses and forbs, including some aggressive weeds (Supplementary Table S1).

In 2018, the field was treated with two applications of herbicide (Roundup[®], active ingredient glyphosate, $C_3H_6NO_5P^{-2}$). The first application (July) cleared the field of existing plants, while the second (August) reduced emergent weeds from the seedbank. Both applications were applied at a rate of 0.95 liter per acre. During the first week of September prior to planting, the field was rototilled to bare soil to a depth of ~30 cm. The experimental area is ~100 × 30 m and is oriented North-South (Supplementary Figure S1). The relief of the field varies slightly (4 m) and is higher in the south than the north. We initially established 30 plots, each 9 × 9 m, with a 1 m buffer strip around each plot. However, six plots within the northern section of the experimental area were regularly inundated by flooding and therefore removed from the experiment.

We used a randomized block design to account for stochastic effects of slope and spatial heterogeneity across the site (Supplementary Figure S1). We created four blocks, each comprising six plots, that were assigned to one of the following treatments: *Thinopyrum intermedium* (IWG-TLI 801) provided by The Land Institute, IWG- alfalfa biculture (cv “Kansas Common”; The Land Institute), wheat (Organic Soft Red Winter, LCS 3334; Albert Lea Seed, Albert Lea MN), forage grass (Organic Hay Mix; Albert Lea Seed, MN), or tallgrass prairie (detention basin seed mix; Prairie Moon Nursery, Winona MN). Species’ lists for the forage grass and tallgrass prairie treatments are listed in Supplementary Table S2. One plot in each block was left as fallow (5 treatments + 1 fallow = 6 plots per block). For IWG and alfalfa, seeding rates followed The Land Institute’s recommendations. For forage grass, annual wheat, and the prairie mix, seeding rates followed the suppliers’ recommendations. The seeding rates were 16.82 kg/ha for IWG and alfalfa; 28 kg/ha for the forage grass; 22.40 kg/ha annual wheat; and 10.52 kg/ha for the prairie mix.

Seeding of row-crop treatments (IWG, IWG biculture, annual wheat) was initiated in September 2018 using an EarthWay Precision Garden Seeder (EarthWay Products, Bristol IN) at 1.5–2 cm depth. Each plot contained 29 rows with 30-cm of inter-row spacing; rows were oriented North-South. Plots containing non-row-crop treatments (forage, prairie) were hand-broadcast atop snow in November 2018. Grass seed (“Sunny Mix”; Main St. Seed and Supply, Bay City MI) was sown within buffer strips in April 2019 to limit weed emergence. Weeds were managed periodically in row-crop plots using a wheel-hoe and hand weeding, buffer strips were mowed as needed, and none of the plots received supplemental watering, fertilization, or pesticides from the time of seed sowing until harvest.

Soil sampling

In our site, sampling with a soil corer resulted in compressed soil plugs, and the destructive removal of plugs from the corer altered aggregate abundances. To avoid these problems, samples were collected using a modification of the spade method (Fernández-Ugalde et al., 2020). A V-shaped hole was dug to a depth of 15 cm using a clean trowel and a slice of soil (~3-cm thick) was taken parallel to one of the sides of the hole with the trowel. Samples were collected to 15 cm depth to detect the most rapid transitions in soil aggregation and SOC accrual (Matamala et al., 2008).

Pre-treatment soil samples were collected in Spring 2018 from 20 points distributed across the experimental area; these points roughly corresponded to all blocks and most treatment plots. At each point, three soil samples separated by at least 50 cm were collected, pooled and gently mixed in a 4.5-liter Ziploc[™] bag to create one composite bulk sample per point.

In the 2019 growing season, samples were collected three times to coincide with the early plant growth (June), vegetative growth (July), and seed-set in IWG (August). Within each plot, we marked out a central 5 × 5 m area for sampling to reduce edge effects. We collected eight soil samples across two transects within the marked-out area, i.e., one sample every 1.2 m. The eight samples were then pooled in a 4.5-liter Ziploc[™] bag to create one composite sample per plot. After collection, the samples were stored in coolers and transported back to the laboratory. Each soil sample was passed through an 8-mm sieve to remove coarse debris and gently homogenized. A sub-sample of fresh soil was removed from each bag for analysis of labile SOC and microbial activity (see Soil analyses) and gravimetric soil moisture, expressed as percent difference in weight between field moist and oven dried soils (90°C, 48 h). The remaining soil was air-dried and stored at room temperature (23°C) before analysis for soil physical properties (texture, WSA), SOC, pH, nutrient levels (inorganic N, P), and AMF hyphal length and AMF colonized of fine roots.

Soil analyses

Soil texture was determined using the micropipette method (Miller and Miller, 1987). Overall, soils across the experimental site comprised 25% sand, 62% silt, 13% clay, which is texturally-classed as a silt loam (USDA-NRCS, 2019). Soil aggregates were extracted from each sample using slaking and wet sieving on a set of nested sieves: 2000 μm (large macroaggregates; >2000 μm diameter), 250 μm (small macroaggregates; 250–2000 μm), and 53 μm (microaggregates, 53–250 μm; Tisdall and Oades, 1982).

Aggregate fractions on each sieve were dried (80°C) for 48 h and weighed, and aggregate stability calculated as the mean weight diameter (MWD; Kemper and Rosenau, 1986).

Inorganic N and P were extracted in deionized water (1: 10 w/v, soil: water; pH 6.8) by vigorous shaking for 30 min. Extracts were filtered and analyzed colorimetrically on a microplate spectrophotometer (Biotek Epoch, Winooski, VT) using the vanadium reduction method for NO₃ (Doane and Horwath, 2003), phenol-hypochlorite method for NH₄ (Weatherburn, 1967), and the malachite green method for PO₄ (Baykov et al., 1988). Soil pH was measured in 1:5 soil: water (v/v) using a pH probe (Fisher Scientific). SOC was measured on finely ground bulk soil samples and aggregate fractions by combustion using a Leco TruSpecTM CN Elemental Analyzer (Leco Corp., St. Joseph, MI). Bulk SOC values are expressed as % soil dry weight. Aggregate fractional OC was corrected for sand content and expressed as grams OC per kg soil.

Two methods were used to quantify AMF abundance. First, AMF external hyphae were extracted in 5% (w/v) sodium hexametaphosphate and filtered onto gridded membrane (Jakobsen et al., 1992), and viewed and scored using a Leica DMLB LB30T microscope (400× mag). Glomeromycotan hyphae were quantified over 50 fields of view for each sample, and AMF hyphal length was calculated using the method of Newman (1966). We defined AMF hyphae as non-septate or irregularly septate hyphae with characteristic unilateral elbow-like projections; all other hyphae were categorized as non-AMF hyphae. Second, we quantified AMF colonization in fine roots. Fine roots were manually picked from each soil sample, washed to remove adhering soil and then stained using methods described by Koske and Gemma (1989). Stained root samples were mounted in polyvinyl alcohol-lactic acid-glycerol (PVLG), and viewed and scored on a Leica DMLB LB30T fluorescence microscope (400× magnification) for AMF root colonization using the line intersect method (Tennant, 1975). Fifty fields of view were examined in each sample for the presence or absence of fungal structures unique to Glomeromycota (hyphae, vesicles, arbuscules, and coils) as well as saprophytic fungi. Counts were converted to percentage of root length colonized by AMF structures.

Microbial activity was determined from the flush of CO₂ following the addition of labile C source (sucrose) to field moist soils (i.e., substrate induced respiration, SIR; Degens and Harris, 1997) and analyzed by the NaOH trap-titration method (Franzluebbers, 2016). Readily soluble (labile) SOC pools were extracted from moist field soils with 0.5 M K₂SO₄ by shaking for 30 min. Filtered extracts were analyzed using the phenol-sulfuric method in microplate format (Masuko et al., 2005). Soil MBC was measured using the fumigation-extraction method and calculated as the difference between fumigated and non-fumigated samples divided by k_c , the extraction efficiency coefficient ($k_c = 0.45$; Vance et al., 1987).

Plant analyses

Plant tissue was analyzed at two time points: pre-treatment (oldfield) and July 2019, during crop vegetative growth. For the oldfield samples, leaves were clipped from plants adjacent to the soil sampling point. In July 2019, we collected and pooled the four uppermost leaves from >20 individual plants in each pasture grass, IWG, IWG biculture, and annual wheat plot. Leaf samples were dried (60°C, 72 h), finely ground, and analyzed by combustion for C and N content as described for soil and aggregate samples.

Statistical analyses

All statistical analyses were performed using R version 4.1.3 (R Core Team, 2017) assuming an alpha = 0.05 level of statistical significance. Assumptions of normality were assessed via Shapiro-Wilks normality test and residual diagnostic plots; no data transformations were required prior to statistical analysis. First, we used repeated-measures analysis of variance (ANOVA; *lme4* package) to compare aggregate abundance, aggregate OC levels, and abiotic and biotic soil factors between pre-conversion (2018) and first year (2019) samplings.

Next, we used a mixed effect model to test the effect of individual treatments and sampling date (June–August) on the abundance of each aggregate size fraction, bulk SOC, aggregate OC levels, and abiotic and biotic soil factors (2019). Treatment type and sampling date were treated as fixed effects and block was a random effect. We also analyzed the data set by comparing the effects of mono- vs. polyculture crops, and perennial (IWG, IWG biculture, prairie) vs. annual (wheat) or mixed crops (forage) on soil properties. For analyses with significant outcomes, Tukey's Honestly Significant Differences (HSD) for multiple comparisons test was used to determine differences among crop treatments or sampling times (*Hmisc* package).

Finally, we identified the soil abiotic and biotic correlates of aggregate abundance and OC levels in 2019, especially those that had positive effects. Preliminary analyses showed that soil moisture was significantly correlated (Spearman rank, r_s) with many factors including aggregate abundance, AMF hyphal length, and SIR (Supplementary Table S3). As a result, we used a mixed effect model with soil moisture and sampling date as random effects and individual soil factors as fixed effects. A marginal r^2 value was calculated for individual soil factors as a way to quantify their effect(s) on aggregation and aggregate OC that were not explained by soil moisture (*MuMIN* package, *rsquared.glm* function).

To summarize and visualize the overall effect of crop treatment or sampling time on soil properties, we analyzed data using non-metric multidimensional scaling (NMDS) on a Bray-Curtis dissimilarity matrix and used *k*-means clustering to assign samples to clusters. Permutational multivariate

analysis of variance (PERMANOVA, 999 permutations) was used to determine significance differences between clusters, while pairwise differences between clusters were tested using pairwise PERMANOVA (*RVAideMemoire* package). Results were visualized using *corrplot*, *ggpubr*, and *ggplot2* packages.

Results

Effects of land conversion on soil properties

Site conversion resulted in significant shifts in most soil properties (Figure 1; Supplementary Table S4), including WSA ($p < 0.001$), aggregate OC ($p < 0.003$), microbial factors ($p < 0.03$, except AMF root vesicles), and soil nutrient levels ($p < 0.001$). We found large and significant increases in the abundance and OC content in each aggregate fraction ($p < 0.001$; Figures 1A,B), as well as microbial activity including AMF hyphal length ($p < 0.001$), microbial biomass (MBC; $p = 0.03$) and respiration (SIR; $p < 0.001$; Figure 1C). Conversely, crop establishment resulted in significant reductions in soil moisture (39 ± 3 – $18 \pm 0.4\%$, mean \pm se; $p < 0.001$; Supplementary Table S4) and plant-available N and P ($p < 0.001$; Figure 1D), consistent with the uptake of soil resources necessary for plant productivity. Bulk SOC declined significantly following conversion ($p < 0.001$) whereas bulk %N increased significantly ($p < 0.001$; Figure 1E). There were also significant shifts in bulk soil and aggregate C:N ($p < 0.001$; Figure 1F). Pre-treatment bulk soil and aggregate C:N levels were within the range of oldfield leaf C:N levels (mean 19, range 14–41). Crop establishment resulted in a decline in both bulk soil and aggregate C:N (C:N 4–7), especially in comparison to leaf C:N in IWG, wheat, and pasture grass leaves (C:N 17.9 ± 1.4) and alfalfa (C:N 12 ± 0.5).

Effects of crop systems on soil properties

While land conversion significantly affected soil properties across the site, the only soil health properties that were shown to be significantly affected by cropping systems were small macroaggregate OC content (Figure 2) and AMF root colonization (Supplementary Table S5). Small macroaggregate OC levels were higher under biculture plots than other crop systems in July and August ($p < 0.05$), including the prairie treatment. These increases were significantly and positively correlated with soil pH and AMF root colonization ($p < 0.05$; Figure 3). AMF root colonization was highest in IWG ($93.6 \pm 2\%$ root colonized) and forage plots ($93.2 \pm 2\%$) and lowest in the prairie plots ($49.8 \pm 2\%$; $p < 0.002$). There was no significant difference in other soil

factors by cropping system alone (Supplementary Table S5) or cropping system over time ($p > 0.05$). The NMDS analyses also supported similar levels of soil health under the different crop systems (PERMANOVA, $p = 0.897$). All crop treatments were distributed across NMDS space and detected in every cluster (Figure 4A). This pattern persisted even when crops were classed and analyzed as mono- vs. polycultures, or annual vs. perennial crops (Supplementary Figure S2; Supplementary Table S5).

Temporal patterns of WSA abundance and OC

Temporal patterns in the abundance and OC level differed by aggregate size (Figures 5A,B; Supplementary Table S6). Site conversion initially promoted the development of macroaggregates, with the highest abundance of large macroaggregates in June ($p < 0.001$; Figure 5A). In turn, the increased mass of soil within this fraction drove a significant increase in MWD from 0.46 ± 0.08 mm (Pre-conversion) to 2.34 ± 0.19 mm, indicating a substantial gain in soil stability ($p < 0.001$; Supplementary Tables S4, S6). However, these increases were short-lived. In July and August, large macroaggregate abundance declined significantly whereas the abundances of small macro- and microaggregates increased significantly ($p < 0.001$; Figure 5A). The mass loss from the large macroaggregates was equivalent to the mass gain in the smaller aggregates (Supplementary Table S6), meaning that large macroaggregates were disrupted into their constituent small macro- and micro-aggregates.

Similar patterns were detected in aggregate OC levels. Although site conversion resulted in significant increases in aggregate OC across all WSA fractions (Figure 5B), the largest increase was detected within the large macroaggregate fraction in June ($p < 0.001$). Disruption of large macroaggregates in July resulted in the release of OC-rich small macro- and micro-aggregates and depletion of the large macroaggregate OC pool. Even so, the gain in OC in the small macro- and micro-aggregate fractions (~ 10 g C kg⁻¹ soil) more than compensated for the OC loss in the large macroaggregate fraction (6 g C kg⁻¹ soil). However, there was no change in aggregate C:N (Figure 5C).

NMDS analyses confirmed the significant effect of sampling time (PERMANOVA, $p = 0.008$; Figure 4B) and separated the sampling times into two groups: July and August were clustered together (pairwise PERMANOVA, $p = 0.057$), and significantly different from June (pairwise PERMANOVA, $p = 0.001$). These groups were separated along NMDS1, which corresponds to a gradient of increasing soil physical stabilization and protection of organic matter, i.e., increasing large macroaggregate abundance and OC, MWD, AMF hyphal

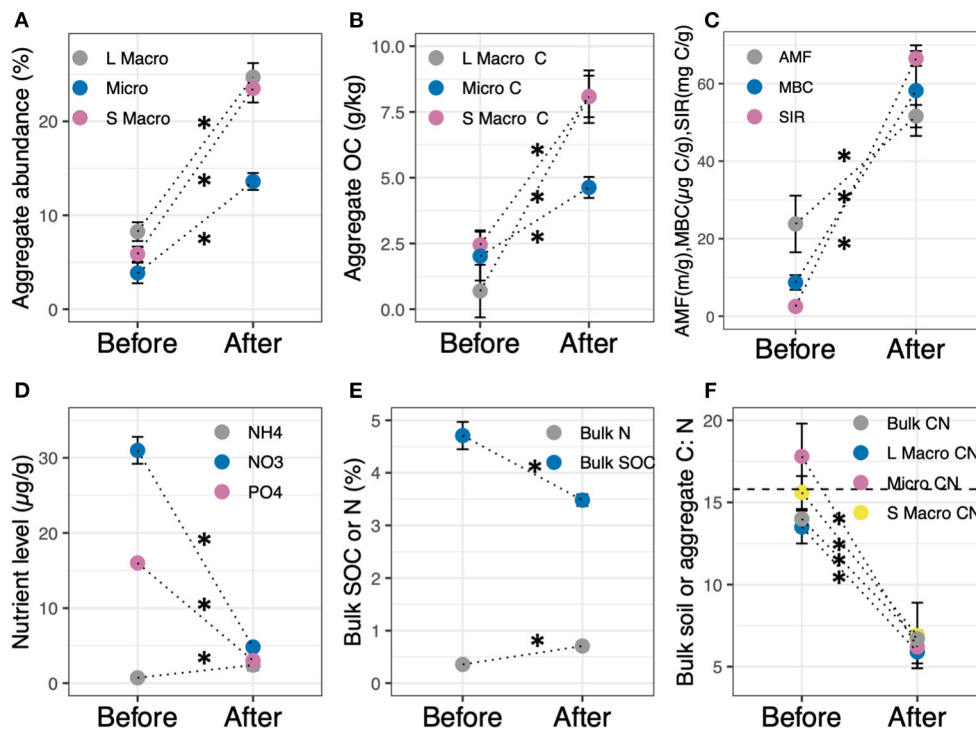


FIGURE 1
Levels of (A) WSA, (B) aggregate OC, (C) microbial factors, (D) plant-available N and P, (E) bulk soil C and N, and (F) bulk soil and aggregate C:N before and after site conversion. Horizontal line in (F) denotes plant C:N. Vertical bars indicate the standard error of the mean. For each soil property, means denoted * differ significantly before and after conversion at $p < 0.05$.

length, and labile C. NMDS2 corresponds to a gradient of increasing macroaggregate abundance within each month.

Covariates of WSA abundance and OC

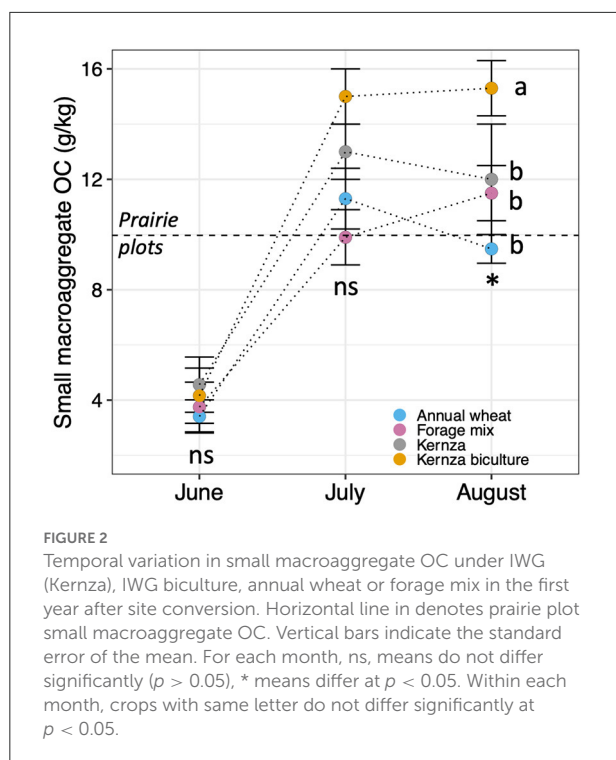
Most soil factors were significantly positively or negatively correlated with soil moisture (Figure 3), including large macroaggregate abundance and OC (positive) and small macro- and micro-aggregate abundance and OC (negative; Supplementary Table S7). After accounting for the effects of soil moisture, mixed models showed that large macroaggregate abundance and OC level were significantly and positively correlated with AMF hyphal length, SIR and labile SOC, with the largest effect size (r_m^2) associated with AMF hyphal length (Table 1; Figure 3; Supplementary Figure S3). These properties were also strongly inter-correlated among themselves (Figure 3) and with MWD (e.g., AMF hyphal length).

Small macro- and micro-aggregate abundance and OC levels were also significantly correlated with AMF hyphal length, labile SOC, and SIR ($p < 0.03$; Table 1; see also Figures 5D,E) but the direction of response was opposite to large macroaggregates (i.e., negative) since small macro- and micro-aggregates were the result of large macroaggregate disruption.

Instead, small macroaggregate abundance and OC levels were positively correlated with AMF root colonization and soil pH (Table 1; Figures 3, 5F; Supplementary Figure S3). However, soil pH was negatively correlated with AMF hyphal length and SIR (Figure 3). WSA abundances and OC levels were not related to soil sand, clay or silt content, plant-available N and P levels, or bulk SOC and %N (Supplementary Table S7).

Discussion

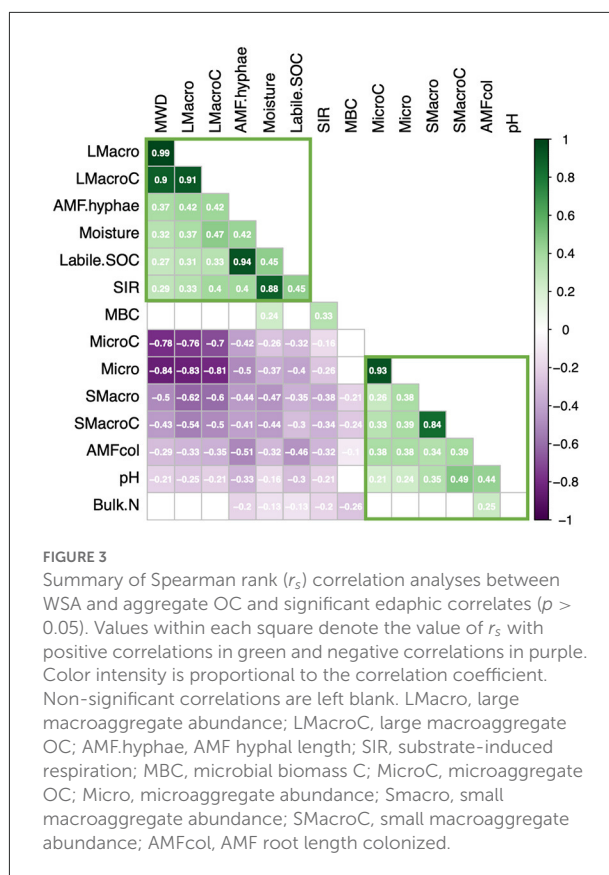
In the context of calls to transition annual grain agriculture to perennial grain agriculture (Crews et al., 2018), we initially hypothesized that converting an old field to perennial IWG would produce increases in soil health comparable to prairie habitat and greater than annual crops (wheat, forage grasses). In support of this hypothesis, we found a large positive effect of IWG biculture and, to a lesser extent IWG alone, on small macroaggregate OC levels. Recent studies have also found promising soil health effects of IWG compared with annuals (e.g., Audu et al., 2022; Martin and Sprunger, 2022). However, our findings must be tempered by the observation that both annual and perennial crops encouraged other early improvements in soil health. Both annual and perennial systems



resulted in rapid and substantial increases in macro- and micro-aggregate abundance, stability (MWD) and OC relative to pre-conversion levels and belowground allocation that replenished AMF abundance and microbial activity.

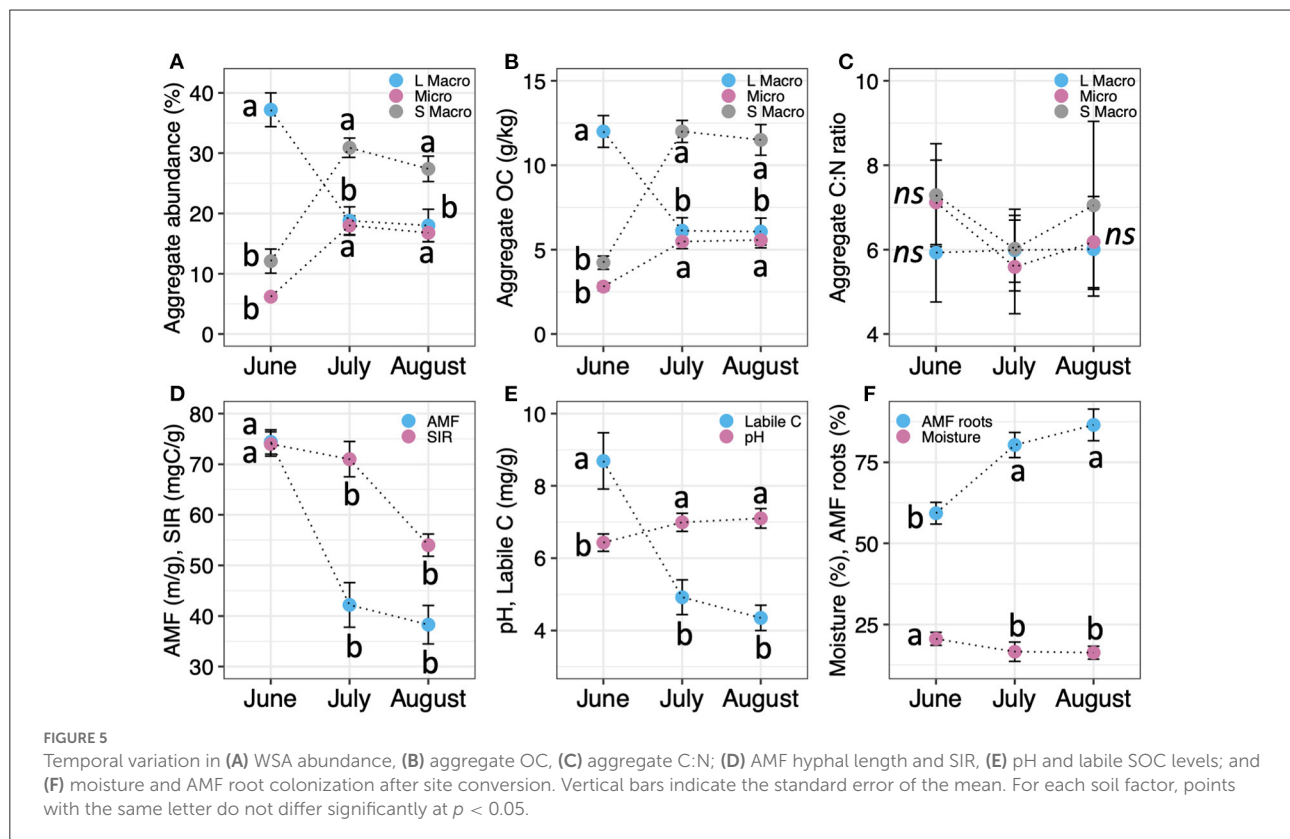
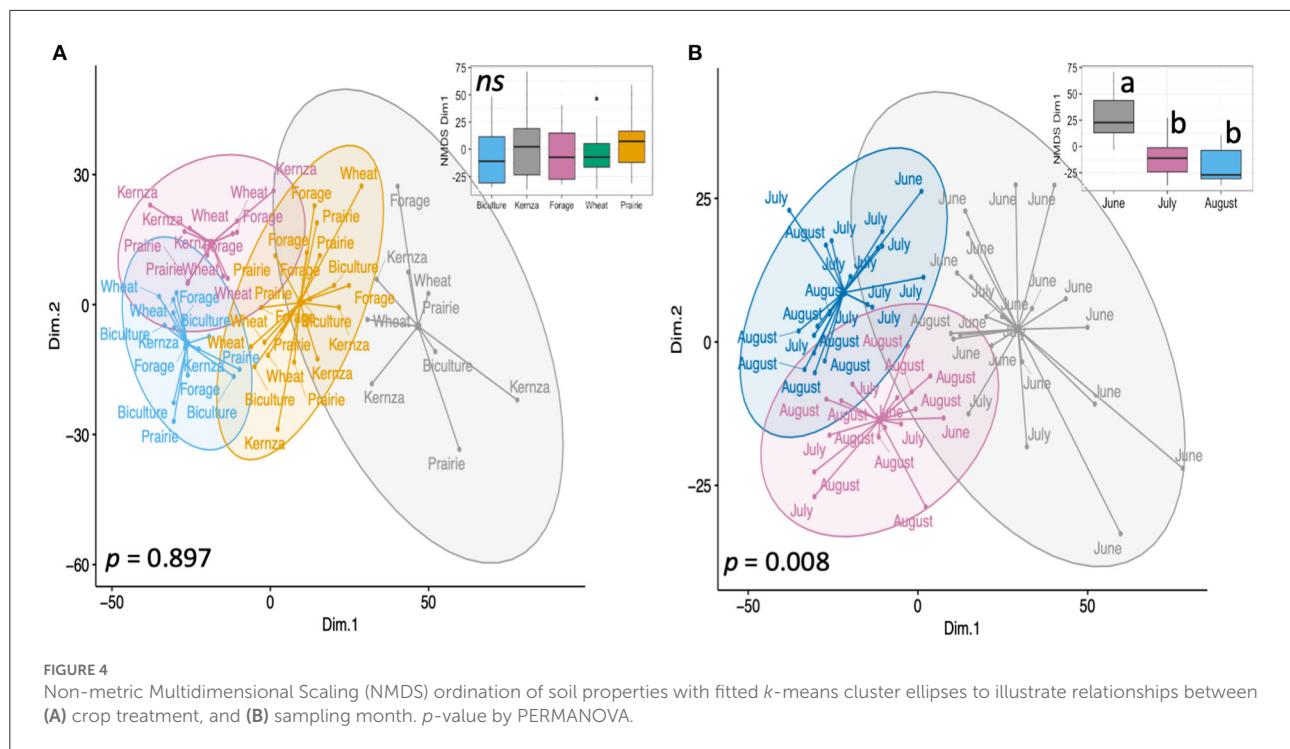
Establishment of both annual and perennial systems resulted in soils dominated by large (>2 mm), relatively unstable macroaggregates. The underlying factor(s) responsible for large macroaggregates were likely AMF hyphal abundance and labile SOC, factors that have been extensively discussed as key drivers of aggregation (Rillig et al., 2015). For example, AMF hyphae physically enmesh soil particles to promote macroaggregate formation (Six et al., 2006). Labile SOC, which includes water-soluble polysaccharides exuded by roots or released by microbial decomposition of green residues, provides the glue or binding agents that stabilize macroaggregates and induces fungal growth (Baumert et al., 2018). Nevertheless, these are transient binding agents and prone to degradation by environmental conditions (Cambardella and Elliot, 1993).

Indeed, the initial increase in large macroaggregate abundance was followed by disruption as evidenced by the loss in large macroaggregate abundance, reduced MWD, and a concomitant increase in the abundance of smaller aggregate subunits. Larger aggregates are generally considered more susceptible to disruption than smaller aggregate fractions (Tisdall and Oades, 1982; Cambardella and Elliot, 1993). In our study, large macroaggregate abundance was strongly and positively correlated with soil moisture. One possibility is that macroaggregate failure may have occurred naturally upon



soil drying or with wet-dry cycles (Amézketa, 1999; Denef et al., 2001). The large biomass of the new crops may have intensified wetting-drying cycles owing to water uptake by roots (Amézketa, 1999) or amplified mechanical stresses *via* root penetration of micro-scale structures (Angers and Caron, 1998). Soil moisture may also control aggregation *via* its influence on biotic mechanisms, including the microbial decomposition of plant residues (Baumert et al., 2018) and/or production of polysaccharides required for aggregate stabilization (Regelink et al., 2015); the correlations between soil moisture and microbial factors in our study support this possibility. However, further experiments are needed to partition the relative contributions of plant roots, local soil hydrologic functions, and microbes in WSA stability.

Small macroaggregate abundance and OC were positively correlated with soil pH, independent of any pH effects on large macroaggregates. In part, this result suggests a direct and abiotic cause for the formation of smaller macroaggregates, possibly through chemical or electrostatic interactions (Denef et al., 2002). Another possibility is that interactions between soil pH and microbial properties (AMF, SIR) influenced aggregate OC pools indirectly by altering the contribution of microbial necromass in aggregates (Yang et al., 2022) or pH-induced changes in microbial community composition (Bååth and Anderson, 2003).



Even allowing for large macroaggregate disruption, land use change achieved substantial increases in aggregate OC, implying a positive scenario for soil health restoration as noted by

Springer and Robertson (2018). We detected two AMF factors that explained the increases in large and small macroaggregate OC. Large macroaggregate OC was best correlated with AMF

TABLE 1 Significant correlates of aggregate abundance and aggregate OC level expressed as marginal r_m^2 and slope of the line.

Fraction	AMF hyphal length (m g ⁻¹ soil)			Labile SOC (mg C g ⁻¹ soil)			SIR (mg C g ⁻¹ soil day)			pH			AMF colonization (% root length colonized)		
	r_m^2	p-value	slope	r_m^2	p-value	slope	r_m^2	p-value	slope	r_m^2	p-value	slope	r_m^2	p-value	slope
Aggregate abundance(%)															
Large macro	0.174	0.001	0.269	0.096	0.017	1.500	0.107	0.012	0.321	-	-	ns	0.106	0.012	-0.241
Small macro	0.189	<0.001	-0.230	0.189	0.009	-1.386	0.143	0.004	-0.310	0.093	<0.001	6.180	0.115	0.010	0.207
Micro	0.244	<0.001	-0.170	0.160	0.001	-0.998	-	-	ns	-	-	ns	0.141	0.003	0.143
Aggregate OC(g kg⁻¹ soil)															
Large macro	0.172	0.001	0.087	0.105	0.013	0.512	0.161	0.002	0.128	-	-	ns	0.119	0.007	-0.083
Small macro	0.168	0.003	-0.090	0.082	0.030	-0.458	0.112	0.010	-0.108	0.199	<0.001	3.489	0.146	0.003	0.091
Micro	0.172	0.001	-0.040	0.099	0.016	-0.211	-	-	ns	-	-	ns	0.139	0.004	0.038

ns, Not significant.

hyphal abundance. AMF hyphae play substantial roles in SOC sequestration by forming a surface for mineral and SOM adsorption (Totsche et al., 2018) and the transfer of plant root-derived OC into aggregates (Frey et al., 2003; Kallenbach et al., 2016). In addition, AMF-colonized fine roots were the main mechanism for improving small macroaggregate OC levels. Fine roots can physically entangle soil particles meaning that sloughed cell walls, root hairs, mucilages, and cell debris, including AMF structures, may have been incorporated into aggregates (von Lützow et al., 2006). In our study, the contribution of AMF root fragments to aggregate OC likely reflects the greater root biomass and associated increases in AMF biomass and activity following site conversion. Other microbial factors were also important.

Materials of microbial origin, including polysaccharides, may have contributed to aggregate OC levels. These inputs can be deduced from WSA C:N levels, all of which appear to be close to the C:N stoichiometry of the soil microbial biomass. In pre-conversion soils, WSA C:N (C:N 16) largely mirrored the old-field plant community indicating plant debris to be the major source of C. Following site conversion, however, the C:N ratio of WSA fractions declined to ~4–7, which is consistent with a substantial contribution from root exudates or labile, microbially-derived substrates to aggregate OC (bacteria C:N 6, fungi C:N 5–17; Cleveland and Liptzin, 2007) and that SOC is largely dependent on inputs of microbial origin (Yang et al., 2022). This is reasonable given that the addition of organic matter to soils stimulated microbial activity (SIR), and macroaggregates form around fresh residues and become enriched in labile (low C:N) substrates derived from the microbial decomposition of residues (Jastrow et al., 2007). The narrow C:N coupled with increasing total %N over time also indicates a substantial contribution of N-containing materials including extracellular polymeric substances (EPS, glycoproteins) that are major agents for aggregating mineral particles and binding OM onto mineral surfaces (Kleber et al., 2007). Taken together, these observations suggest that all treatments resulted in high C inputs with relatively fast decomposition rates (Vesterdal et al., 2002).

Despite the large increases in aggregate-held OC, plant productivity, and organic matter inputs across all treatments, site conversion resulted in a loss of bulk SOC levels. In part, tillage likely forced the physical destruction of existing soil aggregates and the rapid turnover of SOC by soil microbes (Six et al., 2006; Sprunger and Robertson, 2018). Another possibility is that increased availability of easily degradable OC, e.g., labile C or microbial residues, may have initiated the “priming effect” (Blagodatskaya and Kuzyakov, 2008; Kuzyakov, 2010). Because soil microbes are generally C limited (Cleveland and Liptzin, 2007), large inputs of fresh labile C from the new crops may have stimulated the microbial decomposition of SOM, as indicated by the increased SIR. Under these conditions, macroaggregates may also be more prone to enhanced degradation, which may explain

the tendency of large macroaggregates to disaggregate into smaller aggregate fractions in this study site. While observations of substantial increases in labile C, microbial biomass and soil respiration (SIR) after planting may be consistent with priming, more detailed studies are needed to address this possibility.

Implications

Our findings show that the first year of annual or perennial plant establishment substantially and rapidly improved soil aggregation and aggregate-associated OC, two key components of soil health (Golchin et al., 1994; Six et al., 2004; Chivenge et al., 2011; Lal, 2015; Sprunger and Robertson, 2018). The most effective and sensitive integrators of these processes were AMF and labile C. Specifically, AMF hyphae in concert with labile C were the main factors correlated with improved aggregate abundance and stability, while AMF hyphae and AMF-colonized root fragments were of paramount importance in increasing macroaggregate OC over time. This result confirms the importance of AMF in soil health, as has been noted elsewhere (e.g., Rillig et al., 2015). The link between labile C and aggregation, however, brings focus to the importance of the active soil C pool in soil health. This pool fuels microbial metabolism and the subsequent production of aggregate-binding agents and necromass (Peixoto et al., 2020), interactions that are key to stabilizing C. Recently, Martin and Sprunger (2022) noted that soil health indicators reflecting labile C pools were sensitive to temporal fluctuations in soil health under annual vs. perennial crops. Measurements of AMF and labile C should therefore be considered for future assessments of soil health.

On the other hand, it was difficult to detect changes in soil health induced by the different plant systems. Only small macroaggregate OC emerged as a signal of management effects (Sprunger et al., 2018) despite the documented sensitivity of other indicators, such as labile C, to management effects (Xia and Wander, 2022). The contradictory findings in this and other studies (e.g., Sprunger et al., 2018) may reflect high spatial variability levels of soil properties across the plots that masked the effects of crop systems (De et al., 2020) or it may reflect differences in establishment methods (especially since this is a one-year study). There is also growing evidence that soil enzyme activities related to nutrient cycling (i.e., N, P), as well as analyses of readily decomposable pools of SOM, such as permanganate-oxidizable C (reactive carbon), may have potential as early and more sensitive indicators of soil ecological restoration (Martin and Sprunger, 2022; Xia and Wander, 2022). Although these factors comprise a relatively small fraction of SOM, they have turnover rates of weeks to months and may be more sensitive to soil health changes with management and land use practices.

Detecting active soil health recovery may also depend on the depth of measurement. We analyzed soils within the top 15 cm

of soil profile, where the expansive shallow root systems of plants are expected to readily replenish SOC (Matamala et al., 2008; Syswerda et al., 2011; Jaikumar et al., 2012). Future analyses may need to consider the depth distribution of soil health attributes, such as SOC, associated with the deep roots of perennial plants and their capacity to deliver organic matter inputs at depth (DuPont et al., 2014; Peixoto et al., 2020).

Further, we compared crops established by seeding (annual, perennial crops) vs. broadcasting (forage, prairie). While we used cultivation methods typical of current practices, our approach may have inadvertently constrained the net positive effects of kernza mono- and bicultures on soil health. For example, the mean root biomass in kernza mono- and bicultures ($73 \pm 17 \text{ g/m}^2$) was lower than in forage ($143 \pm 91 \text{ g/m}^2$) and prairie plots ($112 \pm 12 \text{ g/m}^2$; E. Kilbane and R. Dybziński, unpublished data). Even so, kernza mono- and bicultures and forage grasses showed similar levels of aggregation. Thus, integrating plant and soil properties by scaling the levels of aggregate OC per unit root biomass (or other plant functional trait) may provide a more nuanced indicator of land-use change on soil health than soil properties alone.

Finally, it has yet to be seen whether perennial agroecosystems can retain as much or greater C stocks as restored prairies. In our study, the IWG and biculture treatments showed promise for soil C accrual. This is consistent with soil resilience, or the capacity of soil to recover after disturbance. However, our study only reports the results of the first growing season, and it is yet to be seen whether IWG (or other crops) can continue to improve SOC levels in the second and subsequent years after crop establishment in our site; work is in progress to test this possibility. Like tallgrass prairie restorations, perennial crops may require time to fully develop differences in aggregation and OC accrual following conversion to an agroecosystem (Virto et al., 2012; Anderson-Teixeira et al., 2013; Novelli et al., 2017; De et al., 2020). Both experimental studies (Steinbeiss et al., 2008) and meta-analyses (Deng et al., 2014) demonstrated that restoration age was the most important factor influencing soil C stocks whereby C accrual increases with site age. This is consistent with the observations that perennial taxa require more time to develop an extensive root system that re-establishes nutrient and water cycling, microbial community succession and metabolism in tandem with the absence of physical disruptions such as tilling (Matamala et al., 2008; DuPont et al., 2014). Further, plant species diversity may be more important in increasing soil C stocks than rooting depth (Steinbeiss et al., 2008) owing to differences in plant root traits (Freschet et al., 2021). If this is the case in our study system, SOC accrual may accelerate over time in tallgrass prairie plots (vs. crops) with the build-up of new OC pools. Taken together, we suggest that determining which factors drive soil health after site conversion to perennial agroecosystems requires long-term monitoring, a consideration of crop species

biomass and diversity, and a more nuanced approach to soil property measurements.

Data availability statement

The original contributions presented in the study are included in the article/[Supplementary material](#), further inquiries can be directed to the corresponding author.

Author contributions

LC, RD, NZ, and LE-W conceived, designed, and installed the experimental plots and procured funding for the research. LC, TA, RD, and LE-W analyzed samples, and analyzed and interpreted the data. LC wrote the first draft of the manuscript. All authors contributed to editing the manuscript and approved the final submission.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

References

- Amézqueta, E. (1999). Soil aggregate stability: a review. *J. Sustain. Agric.* 14, 83–151. doi: 10.1300/J064v14n02_08
- Anderson-Teixeira, K. J., Masters, M. D., Black, C. K., Zeri, M., Hussain, M. Z., Bernacchi, C. J., et al. (2013). Altered belowground carbon cycling following land-use change to perennial bioenergy crops. *Ecosystems* 16, 508–520. doi: 10.1007/s10021-012-9628-x
- Angers, D. A., and Caron, J. (1998). Plant-induced changes in soil structure: processes and feedbacks. *Biogeochemistry* 42, 55–72. doi: 10.1023/A:1005944025343
- Audu, V., Rasche, F., Dimitrova Mártensson, L.-M., and Emmerling, C. (2022). Perennial cereal grain cultivation: implication on soil organic matter and related soil microbial parameters. *Appl. Soil Ecol.* 174:104414. doi: 10.1016/j.apsoil.2022.104414
- Bååth, E., and Anderson, T. H. (2003). Comparison of soil fungal/ bacterial ratios in a pH gradient using physiological and PLFA-based techniques. *Soil Biol. Biochem.* 35, 955–963. doi: 10.1016/S0038-0717(03)00154-8
- Baumert, V. L., Vasilyeva, N. A., Vladimirov, A. A., Meier, I. C., Kögel-Knabner, I., and Mueller, C. W. (2018). Root exudates induce soil

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Supplementary material

The Supplementary Material for this article can be found online at: <https://www.frontiersin.org/articles/10.3389/fsufs.2022.1010298/full#supplementary-material>

SUPPLEMENTARY FIGURE S1

(A) Location of study plots in northeastern Illinois; (B) aerial view of study plots (courtesy Google Earth), each plot measures 9 × 9 m with a 1 m wide aisle between plots; (C) plot view north in spring; (D) plot view north in late summer.

SUPPLEMENTARY FIGURE S2

NMDS output based: (A) crop treatment as annual, perennial or mixed; (B) mono- or poly-culture status; (C) crop treatment; (D) month. Ellipses based on *k*-means clustering, *p*-value by PERMANOVA.

SUPPLEMENTARY FIGURE S3

Significant correlations between large macroaggregate abundances and (A) AMF hyphal length, (B) labile C, and (C) SIR, and (D) small macroaggregates and soil pH.

SUPPLEMENTARY TABLE S1

Pre-treatment beta-diversity.

SUPPLEMENTARY TABLE S2

Seed mix used for tall grass prairie treatment.

SUPPLEMENTARY TABLE S3

Correlations between soil moisture and soil properties.

SUPPLEMENTARY TABLE S4

Summary of soil properties before and after site conversion.

SUPPLEMENTARY TABLE S5

Summary of soil properties by crop treatment.

SUPPLEMENTARY TABLE S6

Summary of soil properties between months.

SUPPLEMENTARY TABLE S7

Summary of correlation analyses between aggregate abundance and OC level and soil properties.

macroaggregation facilitated by fungi in subsoil. *Front. Environ. Sci.* 6:140. doi: 10.3389/fenvs.2018.00140

Baykov, A. A., Evtushenko, O. A., and Avaeva, S. M. (1988). A malachite green procedure for orthophosphate determination and its use in alkaline phosphate-based enzyme immunoassay. *Anal. Biochem.* 171, 266–270. doi: 10.1016/0003-2697(88)90484-8

Bergmann, J., Verbruggen, E., Heinze, J., Xiang, D., Chen, B., Joshi, J., et al. (2016). The interplay between soil structure, roots, and microbiota as a determinant of plant–soil feedback. *Ecol. Evol.* 6, 7633–7644. doi: 10.1002/ece3.2456

Blagodatskaya, E., and Kuzyakov, Y. (2008). Mechanisms of real and apparent priming effects and their dependence on soil microbial biomass and community structure: critical review. *Biol. Fertil. Soils* 45, 115–131. doi: 10.1007/s00374-008-0334-y

Bünemann, E. K., Bongiorno, G., Bai, Z., Creamer, R. E., De Deyn, G., de Goede, R., et al. (2018). Soil quality—a critical review. *Soil Biol. Biochem.* 120, 105–125. doi: 10.1016/j.soilbio.2018.01.030

Cambardella, C. A., and Elliott, E. T. (1993). Carbon and nitrogen distribution in aggregates from cultivated and native grassland soils. *Soil Sci. Soc. Am. J.* 57, 1071–1076. doi: 10.2136/sssaj1993.03615995005700040032x

Chivenge, P., Vanlauwe, B., Gentile, R., and Six, J. (2011). Organic resource quality influences short-term aggregate dynamics and soil organic carbon and nitrogen accumulation. *Soil Biol. Biochem.* 43, 657–666. doi: 10.1016/j.soilbio.2010.12.002

Cleveland, C. C., and Liptzin, D. (2007). C:N:P stoichiometry in soil: is there a “Redfield ratio” for the microbial biomass? *Biogeochemistry* 85, 235–252. doi: 10.1007/s10533-007-9132-0

Crews, T. E., Carton, W., and Olsson, L. (2018). Is the future of agriculture perennial? Imperatives and opportunities to reinvent agriculture by shifting from annual monocultures to perennial polycultures. *Glob. Sustain.* 1:e11. doi: 10.1017/sus.2018.11

Culman, S. W., Snapp, S. S., Ollenburger, M., Basso, B., and DeHann, L. R. (2013). Soil and water quality rapidly responds to the perennial grain kernza wheatgrass. *Agron. J.* 105, 735–744. doi: 10.2134/agronj2012.0273

Culman, S. W., Young-Mathews, A., Hollander, A. D., Ferris, H., Sánchez-Moreno, S., O’Geen, A. T., et al. (2010). Biodiversity is associated with indicators of soil ecosystem functions over a landscape gradient of agricultural intensification. *Landsc. Ecol.* 25, 1333–1348. doi: 10.1007/s10980-010-9511-0

De Deyn, G. B., Cornelissen, J. H., and Bardgett, R. D. (2008). Plant functional traits and soil carbon sequestration in contrasting biomes. *Ecol. Lett.* 11, 516–531. doi: 10.1111/j.1461-0248.2008.01164.x

De, M., Riopel, J. A., Cihacek, L. J., Lawrinenko, M., Baldwin-Kordick, R., Hall, S. J., et al. (2020). Soil health recovery after grassland reestablishment on cropland: the effects of time and topographic position. *Soil Sci. Soc. Am. J.* 84, 568–586. doi: 10.1002/saj2.20007

Degens, B. P., and Harris, J. A. (1997). Development of a physiological approach to measuring the catabolic diversity of soil microbial communities. *Soil Biol. Biochem.* 29, 1309–1320. doi: 10.1016/S0038-0717(97)00076-X

Denef, K., Six, J., Bossuyt, H., Frey, S. D., Elliott, E. T., Merckx, R., et al. (2001). Influence of dry–wet cycles on the interrelationship between aggregate, particulate organic matter, and microbial community dynamics. *Soil Biol. Biochem.* 33, 1599–1611. doi: 10.1016/S0038-0717(01)00076-1

Denef, K., Six, J., Merckx, R., and Paustian, K. (2002). Short-term effects of biological and physical forces on aggregate formation in soils with different clay mineralogy. *Plant Soil* 246, 185–200. doi: 10.1023/A:1020668013524

Deng, L., Liu, G.-B., and Shangguan, Z.-P. (2014). Land-use conversion and changing soil carbon stocks in China’s ‘Grain-for-Green’ Program: a synthesis. *Glob. Change Biol.* 20, 3544–3556. doi: 10.1111/gcb.12508

Doane, T. A., and Horwath, W. R. (2003). Spectrophotometric determination of nitrate with a single reagent. *Anal. Lett.* 36, 2713–2722. doi: 10.1081/AL-120024647

DuPont, S. T., Beniston, J., Glover, J. D., Hodson, A., Culman, S. W., Lal, R., et al. (2014). Root traits and soil properties in harvested perennial grassland, annual wheat and never-tilled annual wheat. *Plant Soil* 381, 405–420. doi: 10.1007/s11104-014-2145-2

Fernández-Ugalde, O., Jones, A., and Meuli, R. G. (2020). Comparison of sampling with a spade and gouge auger for topsoil monitoring at the continental scale. *Eur. J. Soil Sci.* 7, 137–150. doi: 10.1111/ejss.12862

Franzuebbers, A. J. (2016). Should soil testing services measure soil biological activity? *Agric. Environ. Lett.* 1:150009. doi: 10.2134/acl2015.11.0009

Freschet, G. T., Roumet, C., Comas, L. H., Weemstra, M., Bengough, A. G., Rewald, B., et al. (2021). Root traits as drivers of plant and ecosystem functioning: current understanding, pitfalls and future research needs. *New Phytol.* 232, 1123–1158. doi: 10.1111/nph.17072

Frey, S. D., Six, J., and Elliott, E. T. (2003). Reciprocal transfer of carbon and nitrogen by decomposer fungi at the soil–litter interface. *Soil Biol. Biochem.* 35, 1001–1004. doi: 10.1016/S0038-0717(03)00155-X

Glover, J. D., Culman, S. W., DuPont, S. T., Broussard, W., Young, L., Mangan, M. E., et al. (2010). Harvested perennial grasslands provide ecological benchmarks for agricultural sustainability. *Agric. Ecosyst. Environ.* 137, 3–12. doi: 10.1016/j.agee.2009.11.001

Golchin, A., Oades, J. M., Skjemstad, J. O., and Clarke, P. (1994). Soil structure and carbon cycling. *Soil Res.* 32, 1043–1068. doi: 10.1071/sr9941043

Jaikumar, N. S., Snapp, S. S., Murphy, K., and Jones, S. S. (2012). Agronomic assessment of perennial wheat and perennial rye as cereal crops. *Agron. J.* 104, 1716–1726. doi: 10.2134/agronj2012.0291

Jakobsen, I., Abbott, L. K., and Robson, A. D. (1992). External hyphae of vesicular-arbuscular mycorrhizal fungi associated with *Trifolium subterraneum* L. 1. Spread of hyphae and phosphorus inflow into roots. *New Phytol.* 120, 371–380. doi: 10.1111/j.1469-8137.1992.tb01077.x

Jastrow, J. D., Amonette, J. E., and Bailey, V. L. (2007). Mechanisms controlling soil carbon turnover and their potential application for enhancing carbon sequestration. *Clim. Change* 80, 5–23. doi: 10.1007/s10584-006-9178-3

Johnson, A. M., Gamble, A. V., Balkcom, K. S., and Hull, N. R. (2021). Influence of cover crop mixtures on soil health in southeastern crop production systems. *Agrosyst. Geosci. Environ.* 4:e20202. doi: 10.1002/agg2.20202

Kallenbach, C. M., Frey, S. D., and Grandy, A. S. (2016). Direct evidence for microbial-derived soil organic matter formation and its ecophysiological controls. *Nat. Commun.* 7:13630. doi: 10.1038/ncomms13630

Kardol, P., and Wardle, D. A. (2010). How understanding aboveground–belowground linkages can assist restoration ecology. *Trends Ecol. Evol.* 25, 670–679. doi: 10.1016/j.tree.2010.09.001

Kemper, W. D., and Rosenau, R. (1986). Soil cohesion as affected by time and water content. *Soil Sci. Soc. Am. J.* 48, 1001–1006. doi: 10.2136/sssaj1984.03615995004800050009x

Kleber, M., Sollins, P., and Sutton, R. (2007). A conceptual model of organo-mineral interactions in soils: self-assembly of organic molecular fragments into zonal structures on mineral surfaces. *Biogeochemistry* 85, 9–24. doi: 10.1007/s10533-007-9103-5

Koske, R. E., and Gemma, J. N. (1989). A modified procedure for staining roots to detect VA mycorrhizas. *Mycol. Res.* 92, 486–505. doi: 10.1016/S0953-7562(89)80195-9

Kuzyakov, Y. (2010). Priming effects: interactions between living and dead organic matter. *Soil Biol. Biochem.* 42, 1363–1371. doi: 10.1016/j.soilbio.2010.04.003

Lal, R. (2015). Restoring soil quality to mitigate soil degradation. *Sustainability* 7, 5875–5895. doi: 10.3390/su7055875

Ledo, A., Smith, P., Zerihun, A., Whitaker, J., Vicente-Vicente, J. L., Qin, Z., et al. (2020). Changes in soil organic carbon under perennial crops. *Glob. Change Biol.* 26, 4158–4168. doi: 10.1111/gcb.15120

Martin, T., and Sprunger, C. D. (2022). Sensitive measures of soil health reveal carbon stability across a management intensity and plant biodiversity gradient. *Front. Soil Sci.* 2:917885. doi: 10.3389/fsoil.2022.917885

Masuko, T., Minami, A., Iwasaki, N., Majima, T., Nishimura, S. I., and Lee, Y. C. (2005). Carbohydrate analysis by a phenol-sulfuric acid method in microplate format. *Anal. Biochem.* 339, 69–72. doi: 10.1016/j.ab.2004.12.001

Matamala, R., Jastrow, J. D., Miller, R. M., and Garten, C. T. (2008). Temporal changes in C and N stocks of restored prairie: implications for C sequestration strategies. *Ecol. Appl.* 18, 1470–1488. doi: 10.1890/07-1609.1

Miller, W. P., and Miller, D. M. (1987). A micro-pipette method for soil mechanical analysis. *Commun. Soil Sci. Plant Anal.* 18, 1–15. doi: 10.1080/00103628709367799

Newman, E. I. (1966). A method of estimating the total length of root in a sample. *J. Appl. Ecol.* 3, 139–145. doi: 10.2307/2401670

Novelli, L. E., Caviglia, O. P., and Piñeiro, G. (2017). Increased cropping intensity improves crop residue inputs to the soil and aggregate-associated soil organic carbon stocks. *Soil Tillage Res.* 165, 128–136. doi: 10.1016/j.still.2016.08.008

Peixoto, L., Elsgaard, L., Rasmussen, J., Kuzyakov, Y., Banfield, C. C., Dippold, M. A., et al. (2020). Decreased rhizodeposition, but increased microbial carbon stabilization with soil depth down to 3.6 m. *Soil Biol. Biochem.* 150:108008. doi: 10.1016/j.soilbio.2020.108008

- Post, W. M., and Kwon, K. C. (2010). Soil carbon sequestration and land-use change: processes and potential. *Glob. Change Biol.* 6, 317–327. doi: 10.1046/j.1365-2486.2000.00308.x
- Pugliese, J. Y., Culman, S. W., and Sprunger, C. D. (2019). Harvesting forage of the perennial grain crop kernza (*Thinopyrum intermedium*) increases root biomass and soil nitrogen. *Plant Soil.* 437, 241–254. doi: 10.1007/s11104-019-03974-6
- R Core Team (2017). *R: A Language and Environment for Statistical Computing*. R Foundation for Statistical Computing. Available online at: <http://www.R-project.org/> (accessed December 1, 2018).
- Regelink, I. C., Stoof, C. R., Rousseva, S., Weng, L., Lair, G. J., Kram, P., et al. (2015). Linkages between aggregate formation, porosity and soil chemical properties. *Geoderma.* 247–248, 24–37. doi: 10.1016/j.geoderma.2015.01.022
- Rillig, M. C., Aguilar-Trigueros, C. A., Bergmann, J., Verbruggen, E., Veresoglou, S. D., and Lehmann, A. (2015). Plant root and mycorrhizal fungal traits for understanding soil aggregation. *New Phytol.* 205, 1385–1388. doi: 10.1111/nph.13045
- Six, J., Bossuyt, H., Degryze, S., and Denef, K. (2004). A history of research on the link between (micro)aggregates, soil biota, and soil organic matter dynamics. *Soil Tillage Res.* 79, 7–31. doi: 10.1016/j.still.2004.03.008
- Six, J., Frey, S. D., Thiet, R. K., and Batten, K. M. (2006). Bacterial and fungal contributions to carbon sequestration in agroecosystems. *Soil Sci. Soc. Am. J.* 70, 555–569. doi: 10.2136/sssaj2004.0347
- Sprunger, C., Culman, S., Robertson, G., and Snapp, S. (2018). Perennial grain on a midwest alfisol shows no sign of early soil carbon gain. *Renew. Agric. Food Syst.* 33, 360–372. doi: 10.1017/S1742170517000138
- Sprunger, C. D., Culman, S. W., Peralta, A. L., DuPont, S. T., Lennon, J. T., and Snapp, S. S. (2019). Perennial grain crop roots and nitrogen management shape soil food webs and soil carbon dynamics. *Soil Biol. Biochem.* 137, 107573. doi: 10.1016/j.soilbio.2019.107573
- Sprunger, C. D., Culman, S. W., Robertson, G. P., and Snapp, S. S. (2017). Perennial grain on a midwest alfisol shows no sign of early soil carbon gain. *Renew. Agric. Food Syst.* 33, 360–372. doi: 10.1017/s1742170517000138
- Sprunger, C. D., and Robertson, R. P. (2018). Early accumulation of active fraction soil carbon in newly established cellulosic biofuel systems. *Geoderma* 318, 42–51. doi: 10.1016/j.geoderma.2017.11.040
- Steinbeiss, S., Besler, H., Engels, C., Temperton, V. M., Buchmann, N., Roscher, C., et al. (2008). Plant diversity positively affects short-term soil carbon storage in experimental grasslands. *Glob. Change Biol.* 14, 2937–2949. doi: 10.1111/j.1365-2486.2008.01697.x
- Syswerda, S. P., Basso, B., Hamilton, S. K., Tausig, J. B., and Robertson, G. P. (2012). Long-term nitrate loss along an agricultural intensity gradient in the Upper Midwest USA. *Agric. Ecosyst. Environ.* 149, 10–19. doi: 10.1016/j.agee.2011.12.007
- Syswerda, S. P., Corbin, A. T., Mokma, D. L., Kravchenko, A. N., and Robertson, G. P. (2011). Agricultural management and soil carbon storage in surface vs. deep layers. *Soil Sci. Soc. Am. J.* 75, 92–101. doi: 10.2136/sssaj2009.0414
- Tennant, D. (1975). A test of a modified line intersect method of estimating root length. *J. Ecol.* 63, 995–1001. doi: 10.2307/2258617
- Tisdall, J. M., and Oades, J. M. (1982). Organic matter and water stable aggregates in soils. *J. Soil Sci.* 33, 141–163. doi: 10.1111/j.1365-2389.1982.tb01755.x
- Totsche, K. U., Amelung, W., Gerzabek, M. H., Guggenberger, G., Klumpp, E., Knief, C., et al. (2018). Microaggregates in soils. *J. Plant Nutr. Soil Sci.* 181, 104–136. doi: 10.1002/jpln.201600451
- USDA-NRCS, Soil Science Division Staff (2019). *Web Soil Survey, Soil Data Explorer Tool*. Available online at: <https://websoilsurvey.sc.egov.usda.gov/App/HomePage> (accessed July 21, 2022).
- Vance, E. D., Brookes, P. C., and Jenkinson, D. S. (1987). An extraction method for measuring soil microbial biomass C. *Soil Biol. Biochem.* 19, 703–707. doi: 10.1016/0038-0717(87)90052-6
- Vesterdal, L., Ritter, E., and Gundersen, P. (2002). Change in soil organic carbon following afforestation of former arable land. *For. Ecol. Manage.* 169, 137–147. doi: 10.1016/S0378-1127(02)00304-3
- Virto, I., Barré, P., Burlot, A., and Chenu, C. (2012). Carbon input differences as the main factor explaining the variability in soil organic C storage in no-tilled compared to inversion tilled agrosystems. *Biogeochemistry* 108, 17–26. doi: 10.1007/s10533-011-9600-4
- von Lütow, M., Kögel-Knabner, I., Ekschmitt, K., Matzner, E., Guggenberger, G., et al. (2006). Stabilization of organic matter in temperate soils: mechanisms and their relevance under different soil conditions—a review. *Eur. J. Soil Sci.* 57, 426–445. doi: 10.1111/j.1365-2389.2006.00809.x
- Weatherburn, M. W. (1967). Phenol-hypochlorite reaction for determination of ammonia. *Anal. Chem.* 39, 971–974. doi: 10.1021/ac60252a045
- Xia, Y., and Wander, M. (2022). Evaluation of indirect and direct scoring methods to relate biochemical soil quality indicators to ecosystem services. *Soil Sci. Soc. Am. J.* 86, 678–702. doi: 10.1002/saj2.20370
- Yang, Y., Xie, H., Mao, Z., Bao, X., He, H., Zhang, X., et al. (2022). Fungi determine increased soil organic carbon more than bacteria through their necromass inputs in conservation tillage croplands. *Soil Biol. Biochem.* 167, 108587. doi: 10.1016/j.soilbio.2022.108587



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EDITED BY

Jose G. Franco,
Agricultural Research Service (USDA),
United States

REVIEWED BY

Patrick Hatfield,
Retired, Bozeman, MT, United States
Emerson Nafziger,
University of Illinois at
Urbana-Champaign, United States

*CORRESPONDENCE

Heather D. Karsten
hdk3@psu.edu

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Integrated weed management with strategic tillage can maintain soil quality in continuous living cover systems

Devyn McPheeters^{1,2}, Mary Ann Bruns^{1,2}, Heather D. Karsten^{3*}
and Curtis J. Dell⁴

¹Intercollege Graduate Degree Program in Ecology, The Pennsylvania State University, University Park, PA, United States, ²Department of Ecosystem Science and Management, College of Agricultural Sciences, The Pennsylvania State University, University Park, PA, United States,

³Department of Plant Science, The Pennsylvania State University, University Park, PA, United States,

⁴USDA-ARS Pasture Systems and Watershed Management Research Unit, University Park, PA, United States

Maximizing living cover and minimizing soil disturbance with no-till are key strategies in regenerative row-crop production. Although living cover and no-till can increase beneficial soil carbon and water stable aggregates (WSA), annual crops in rotation with perennials often rely on herbicides to control weeds and terminate perennials. Integrated weed management (IWM) reduces reliance on herbicides by employing multiple weed control strategies including tillage and/or cultivation. However, many no-till growers are reluctant to implement some soil disturbance due to concerns about negative impacts on soil health. For that reason, we hypothesized that compared to continuous no-till and standard herbicides (NT-SH), a strategic inversion tillage in IWM (ST-IWM) would result in lower soil carbon and WSA in the year following the tillage event. We also hypothesized that soil carbon and WSA would not differ between the two systems when sampled after cover cropping and 2 years of perennials. We tested these hypotheses within a 6-year, diverse, dairy crop rotation initiated in 2010 in central Pennsylvania in a channery silt loam soil. The systems were compared in split-plots in a full crop entry experiment, where the six phases of the crop rotation were planted every year in a randomized complete block design, replicated four times. We compared the soil health indicators in spring 2010 prior to the start of the experiment and in 2013 and 2019 following inversion tillage (ST-IWM) or herbicide termination (NT-SH) of the perennial forage in the first year of the rotation. We also compared these indicators in the sixth year of the rotation after 3 years of annual and cover crops and 2 years of perennial forage. We sampled at two depths: 0–5 and 5–15 cm for total carbon and bulk density, 0–5 cm for labile carbon and 0–15 cm for WSA. Results indicate that despite initial smaller soil health values in the ST-IWM system following inversion tillage, all properties except labile carbon were similar to the NT-SH system in the sixth year of the rotation.

KEYWORDS

tillage, no-till, soil health, perennial, alfalfa, soil aggregate, soil carbon, bulk density

Introduction

Annual crop production that fulfills the regenerative agriculture goals of enhancing soil carbon without tillage but with continuous plant cover often relies on herbicides to terminate cover and perennial crops and to control weeds. No-till equipment places crop seeds into soil without plowing or disking, reducing soil disturbance and leaving previous crop residues on the surface, thus reducing soil erosion and maintaining soil structure (Jarecki and Lal, 2003; Baker et al., 2007). Besides saving fuel and protecting soil from erosion, no-till farming conserves soil carbon near the soil surface by slowing the decomposition of residues on or near the surface relative to soils that are mixed through tillage (Stubbs et al., 2004; Kan et al., 2021) and is therefore considered an important management tool for conserving soil and improving key components of soil health. When cover crops and perennial crops are integrated into no-till cropping systems, farmers typically apply herbicides to terminate the crops that provide continuous cover and control weeds. This reliance on herbicides for crop and weed termination can contribute to the evolution of herbicide resistant weeds (Quincke et al., 2007; Green and Siehl, 2021; Heap, 2022). Concerns that herbicide use can adversely affect humans (Sanborn et al., 2007; Zhang et al., 2019; Stradtman and Freeman, 2021), aquatic ecosystems (Hunt et al., 2017), wildlife (Freemark and Boutin, 1995), soil organisms (Gaupp-Berghausen et al., 2015) and soil health, present a conundrum for no-till farmers. No-till also increases nutrient stratification in the soil because it allows fertilizers, lime and residues from terminated crops to accumulate near the surface (Scheiner and Lavado, 1998). This stratification can be detrimental to crop production and the environment (Baker et al., 2017; Norton, 2020). These issues have led to the consideration of using strategic disturbance events, such as occasional inversion tillage and shallow cultivation to terminate perennials or cover crops, reducing the frequency and rate of herbicide applications and incorporating nutrients into the soil profile (Kettler et al., 2000; Dang et al., 2015; Summers et al., 2021).

In a 2017 survey by the USDA, 67% of crop acreage in Pennsylvania was managed with no-till and 24% with cover crops [National Agricultural Statistics Service (NASS) USDA, 2017]. Organizations such as the Pennsylvania No-Till Alliance have played an important role in advocating for these conservation practices and are often averse to using any type of tillage that may destroy the soil health benefits gained from continuous no-till (PA No-Till Alliance, 2022). Multiple studies have documented the negative effects of tillage on soil health, with soil health indicators tending to decline with increased tillage intensity (Jarecki and Lal, 2003; Bhardwaj et al., 2011; Cates et al., 2016; Nunes et al., 2020; Sprunger et al., 2021).

A nationwide meta-analysis compared the effects of moldboard plowing, chisel plowing, no-till, and perennial systems on soil organic carbon, permanganate oxidizable C

(POXC or active C), soil respiration, microbial biomass C and N, soil protein, and beta-glucosidase activity (Nunes et al., 2020). The authors found that converting from moldboard to chisel plowing improved soil organic carbon, microbial biomass carbon and soil respiration in the first 0–15 cm of soil, however converting from moldboard plowing to no-till improved all seven soil health indicators at 0–15 cm. Additionally, compared to moldboard plowing, perennial systems had improved soil health indicators at all depths sampled (0–40 cm). The authors concluded that combining cover crops and minimizing crop residue removal along with no-till improved soil health indicators more than switching to no-till alone (Nunes et al., 2020).

The addition of cover crops, perennials and increased crop diversity in cropping systems also enhances soil health indicators, such as SOC and water stable aggregates (Angers and Caron, 1998; Salvo et al., 2014; Congreves et al., 2015; King and Blesh, 2017; Basche and DeLonge, 2019; Sprunger and Martin, 2020). In a Wisconsin study, particulate organic matter (POM) and aggregate C and N were the soil health indicators assessed across six cropping systems ranging from continuous maize with yearly chisel plowing to more diverse rotations with less frequent chisel plowing and perennial forages, to never-tilled perennial pasture (Cates et al., 2016). Although the authors hypothesized that soil health would be reduced in proportion to tillage intensity they found that the systems that were tilled every year had POM and aggregate C levels similar to the crop rotation tilled every 3 years that included significant crop residues from corn stover and perennials (Cates et al., 2016). The integration of crop residues and perennials also diversifies weed control strategies and can reduce reliance on herbicides with diverse crop lifecycles that interrupt weed lifecycles and mechanical weed control *via* frequent harvests of perennial forages (Cavigelli et al., 2008; Davis et al., 2012; Summers et al., 2021).

One-time tillage, also referred to as strategic tillage, occasional tillage, and single inversion tillage, etc., is a potential alternative to continuous no-till; however, disagreement exists as to the efficacy of one-time tillage on otherwise no-till land. Some studies report little to no effect of one-time tillage on soil health (Salvo et al., 2014; Dang et al., 2015; Blanco-Canqui and Wortmann, 2020), whereas others report a persistent decrease in soil health following tillage (Wortmann et al., 2010; Stavi et al., 2011). For instance, one-time tillage was effective at reducing herbicide dependence and controlling downy brome (*Bromus tectorum* L.), an annual grassy weed, in a 20 year NT winter wheat (*Triticum aestivum* L.)-fallow system in Nebraska (Kettler et al., 2000). The authors compared a one-time tillage using a moldboard plow with secondary tillage (disking, chisel) and rod weeding to a no-till control treatment. Tillage reduced the downy brome densities and wheat yield increased; although 5 years after the tillage event SOC

at 0–7.5 cm depth was still 20% less than in continuous no-till. However, in the 7.5–15 cm depth, SOC was 15% greater compared to the continuous no-till treatment, suggesting that carbon was redistributed through tillage rather than lost from the soil profile (Kettler et al., 2000). In grain-crop systems in Northeastern Australia, Dang et al. (2015) also evaluated strategic or occasional shallow tillage in 14 sites and found weed populations were reduced the year following tillage, without reducing soil TOC. Although bulk density and soil water availability decreased for the first 12 months at some sites, most appeared to recover after 24 months except in high clay soils (Dang et al., 2015).

By contrast, integrating perennials into an annual crop rotation with some tillage has been reported to maintain SOC that was similar to an annual crop rotation with no-till and some tillage (Cates et al., 2016) or 100% no-till management of the same crop rotation of perennials and annuals (Salvo et al., 2014). For instance, Salvo et al. (2014) found that at the end of 9 years of integrating a few years of perennial pasture into annual crop rotations with tillage maintained total SOC at levels similar to no-tillage with the same crop rotation. A review by Blanco-Canqui and Wortmann (2020) examined the impacts of occasional tillage on SOC and physical properties. Although results varied from study to study, they concluded that occasional tillage generally does not reduce overall SOC content but can affect its vertical distribution, with effects lasting up to 2 years following tillage. Additionally, occasional tillage was effective at reducing nutrient stratification and suppressing weed populations for several years. Blanco-Canqui and Ruis (2020) concluded that the benefits of occasional tillage depend on the type, timing, depth, and frequency of the tillage and that the ideal type of one-time tillage will vary. These findings explain the variability in the results of other studies but are encouraging for the use of one-time tillage when occasionally integrated. Though many studies include one-time tillage or cover crops and perennials as a factor, a research gap exists concerning the interaction between one-time tillage and continuous cover with cover and perennial crops (Osterholz et al., 2021).

We undertook this study because we wanted to assess the effects of strategic tillage on soil health indicators. Although farmer cooperators frequently point out that increased water infiltration is a major benefit of no-till, a recent meta-analysis of 89 field trial studies reported that no-till had no significant effect ($5.7 \pm 9.7\%$) on infiltration rates even with residue retention (Basche and DeLonge, 2019). Instead, the use of either perennials or cover crops across those studies was found to increase mean infiltration rates by 59.2 ± 20.9 and $34.8 \pm 7.7\%$, respectively. Moreover, recognition is growing that no-till management by itself is not a consistently effective means to increase soil carbon, because soil carbon accrual can vary, due to differences in climate, soil texture, organic mineralization rates, and carbon saturation (Ogle et al., 2012;

Powlson et al., 2014; Daryanto et al., 2020). Thus, farmers are more likely to recognize the utility of weed control provided by strategic tillage combined with the integration of perennials and cover crops that can be more effective for sequestering soil carbon than no-till alone (Mary et al., 2020).

In this study, we assessed the effects of one-time tillage in a 6-year annual and perennial crop rotation with continuous cover by measuring soils across crop rotation years for total organic carbon (TOC), permanganate oxidizable carbon (POXC), and water-stable soil aggregates (WSA). We chose POXC as an indicator of soil carbon dynamics because it is a good proxy for labile C (i.e., readily available to soil microorganisms) and has been reported to be more responsive to soil management than TOC (Culman et al., 2012; Hurisso et al., 2016). We also chose WSA as a sensitive indicator of changes in soil structure due to management (Haynes and Swift, 1990). To those ends, we hypothesized that (1) in the spring following an inversion tillage event, the three soil health indicators would be smaller compared to those for continuous no-till soils; and (2) indicator values would return to similar levels observed in no-till soils after returning to no-till with cover crops and 2 years of perennial crops in the sixth year of the rotation.

Materials and methods

The experiment was conducted as part of the Dairy Cropping Systems (DCS) project established in 2010 at the Pennsylvania State University Russell E. Larson Agronomy Research Farm near Pennsylvania Furnace, PA (40.72°N, –77.92°W). The project aimed to simulate a confinement 97-ha dairy farm at 1/20th the scale (4.86 ha) that could produce all forage and grain needed for a simulated 65-cow dairy herd while minimizing off-farm inputs and environmental impacts. We sampled soils from a 6-year crop rotation of annual and perennial crops comparing two weed control systems: (i) continuous no-till with standard herbicides (NT-SH) and (ii) strategic tillage and integrated weed management that reduced herbicide applications (ST-IWM). The crop sequence (Figure 1) consisted of: (1) winter canola (*Brassica napus* L.) or canola plus oats followed by a rye cover crop (2) soybean [*Glycine max* (L.) Merr.] followed by a rye (*Secale cereale* L.) cover crop (3) corn grain or corn silage (*Zea mays* L.) followed by (4–6) 3 years of perennial forage. The perennial forage in the ST-IWM system was alfalfa (*Medicago sativa* L.) and orchard grass (*Dactylis glomerata* L.) planted with a companion small grain (for species over the 9 years, see Summers et al., 2021) (Table 1), while the NT-ST system was alfalfa as the perennial crop until 2016 when orchardgrass was added so that the perennial forage systems would have the same species that were harvested for hay and silage. The practice of harvesting perennial forages grown in rotation with annual crops to feed a total mixed ration to cattle in

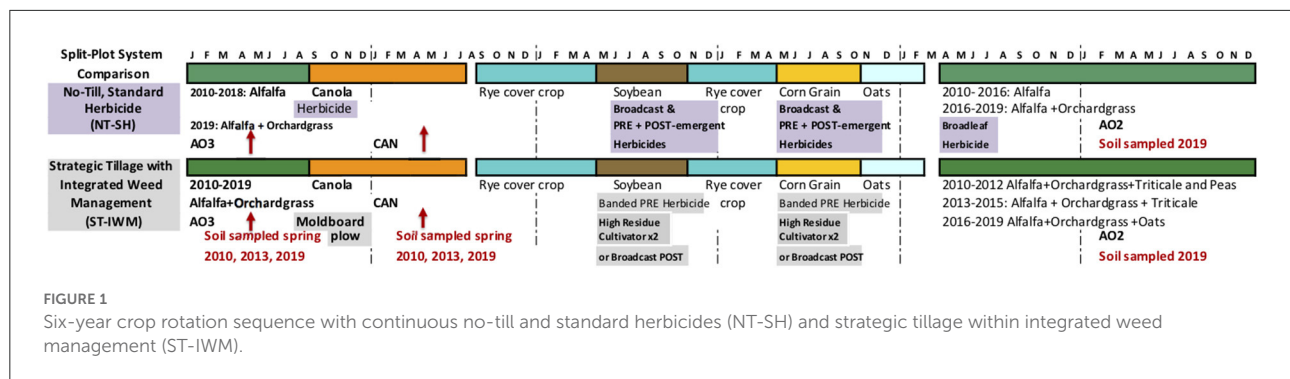


TABLE 1 The main crops in the rotation sequence, the abbreviation used within the paper, and which soil analyses was conducted in each crop.

Main crop name	Abbreviation	Annual/perennial	Indicators analyzed
Canola	Can	Annual	All
Soybean	n/a	Annual	None
Corn	n/a	Annual	None
grain/silage			
Alfalfa-orchardgrass	AO (2 or 3)	Perennial	Second-year: TOC and bulk density by volume Third year: all

confinement is typical for dairy farms of this size in Pennsylvania (Holly et al., 2019).

Experimental design

Every crop phase of the 6-year rotation was present every year in randomized main plots (37 by 27 m) replicated four times, and the two weed control systems were split-plots (18 × 27 m) within each crop entry. A winter rye cover crop was present on the entire experiment when the experimental plots were established in spring 2010. Agronomic production details (ex. crop varieties, seeding rates, planting dates, herbicides, etc.) are described in Summers et al. (2021). Most relevant to this study, the NT-SH system received standard herbicide application rates to control weeds and terminate perennial crops without any tillage. To terminate the perennial alfalfa or alfalfa-orchardgrass in fall 2018 prior to planting winter canola, 0.9 kg ae ha⁻¹ of glyphosate (N-[phosphonomethyl]glycine), 0.5 kg ae ha⁻¹ 2-4-D LV4, and 0.3 kg ae ha⁻¹ dicamba was applied in late August. Pre- and post-emergence herbicides were broadcast as part of the NT-SH system weed management strategy (see Summers et al., 2021).

By contrast, in ST-IWM prior to fall planting or winter canola, the perennial crops (alfalfa-orchardgrass) were terminated with a moldboard plow followed by secondary tillage (disk, a S-tine chisel and cultimulcher) in late August

followed by planting winter canola. For the corn and soybean row crops, pre-emergent herbicide was banded over only corn and soybean rows, a high residue shallow-disk cultivator was used to control weeds between the row crops twice early in the season, instead of postemergence herbicide from 2010 to 2012 (Summers et al., 2021). From 2013 to 2018 the corn and soybean crop plots were split into two nested split-split (9 by 27 m) plots, one nested split-plot received the shallow-disk cultivation, the other nested split plot received a post-herbicide application instead of the high residue cultivation as described in Summers et al. (2021). Soil samples from both of these nested split-split plots were combined in 2019 for the ST-IWM. Herbicide reduction in the ST-IWM varied over the 9 years and by crop entries, with the exception of when the perennial forage was terminated with tillage and herbicide was eliminated or reduced 100%. For the other crops, burndown herbicide applications were the same in both systems, and the STM-IWM system herbicide reductions occurred in the pre-emergent and post-emergent herbicide applications and were calculated as kilograms per hectare of active ingredient. Compared to the NT-SH system, over the 9 years the STM-IWM herbicide reduction averaged 18% in soybean, 37% in corn and 37% in the establishment year of the perennials. Herbicides were not applied to either treatment in the second and third years of the perennial forages or when the winter canola was planted (Summers et al., 2021).

The predominant soil series at the experimental site is Murrill (Fine-loamy, mixed, semiactive, mesic Typic Hapludults). Manure management practices were chosen to reflect best on-farm practices. When soil was managed without tillage (NT-SH), manure was injected following perennial termination prior to planting winter canola, and before planting a rye cover crop (Figure 1). Manure was broadcast and incorporated by the tillage in the ST-IWM system prior to planting canola.

Soil sampling

Prior to any experimental field operations, soils were sampled in spring 2010 to establish baseline values. In spring

2013 and 2019, soils were sampled from plots containing winter canola (planted the previous fall) and the third year of the perennial crop. In the ST-IWM system, the canola plots followed the full fall tillage event, while the third year perennial plots were in the sixth year after tillage. In the NT-SH system, neither the canola nor perennial crops had received any tillage. In 2013, the NT-SH system had been in only alfalfa, while the ST-IWM had alfalfa and orchardgrass. In 2019, both the NT-SH and ST-IWM plots were planted in alfalfa and orchardgrass in 2016. For simplicity, we therefore refer to this crop as alfalfa + orchardgrass (AO3). In 2019, we also sampled soil from the second-year alfalfa orchardgrass (AO2) crop (5 years after tillage) from both the NT-SH and ST-IWM systems for TOC and bulk density measurements.

Ten to 15 soil cores were randomly collected between 4 and 20 of April 2010, on 9 and 10 of April 2013 and 19 March, 2019, when winter canola and perennials had begun greening-up after winter dormancy. Soil cores were split into 0–5 and 5–15 cm depths for separate composite samples. After being air dried and passed through a 2 mm sieve, these samples were measured for TOC and POXC.

Additional soil cores were sampled to 15 cm depth and composited on 20 and 21 May of 2010; on 17, 20 and 21 May, 2013; and on 3 June, 2019, when the canola and the perennials were flowering or beginning to flower. These samples were measured for WSA after storage in cool, airtight containers to minimize microbial activity.

Bulk density

Soil bulk density was sampled on 17 November, 2019, which also allowed the calculation of TOC on a volumetric basis in 2019. Ten to 15 bulk density samples were collected from canola, and the second and third years of alfalfa and orchardgrass (AO2 and AO3, respectively) using a tractor mounted Giddings soil probe (7.5 cm diameter). Following a modified bulk density procedure described by Blake and Hartge (1986), cores were collected in a thin plastic sheath, cut open in the laboratory, and soil was separated into 0–5 and 5–15 cm depths. Soils were air dried, weighed, and passed through a 2 mm sieve. Material collected on the sieve was weighed, washed, and volume determined by water displacement to allow correction of bulk density values for stone content.

Total organic carbon

TOC concentrations at 0–5 and 5–15 cm depths were measured by combustion with a 2,400 CHNS/O Series II Analyzer (Perkin Elmer, Waltham, MA, USA) in 2010 and 2013 and a Vario Max elemental analyzer (Elementar, Langensfeld, Hesse, Germany) in 2019.

Labile carbon (POXC) and water-stable aggregates

Samples (0–5 cm) were tested for POXC using the Weil et al. (2003) method. Briefly, 2.5 g of air-dried soils were mixed with 2 mL stock solution (0.2 M KMnO₄ in 1 M CaCl₂, pH 7.2) and brought to 20 mL volume with water. Cleared soil suspensions were diluted 1/10 before measuring absorbances at 550 nm in a colorimeter (Hach, Loveland, CO). Standard curves of absorbance values for known KMnO₄ concentrations provided the y-intercept (a) and slope (b) in the following equation to calculate soil POXC concentration:

$$\text{POXC (mg kg}^{-1}\text{)} = (0.02 \text{ Mol/L} - (a + b * \text{absorbance})) * (9000 \text{ mg C/Mol}) * (0.02 \text{ L soln./0.0025 kg soil})$$

For water-stable aggregates, field moist soils sampled from 0 to 15 cm were sieved through 2 and 1-mm sieves, with material remaining on top of the 1-mm sieve retained and air-dried. A modified version (Grover, 2008) of the Kemper and Rosenau (1986) method was used with a dispersing solution (2 g of sodium hexametaphosphate in 1 L of deionized water). Four grams of air-dried soils were added to sieve-bottom cups in an 8-cup wet-sieving apparatus (Eijkelpamp Soil & Water Giesbeek, Netherlands). Tins containing deionized water were placed under the sieve-bottom cups, which were lowered to completely submerge the soils. After submersion without disturbance for 5 min, the cupholder was raised and lowered at a rate of 33 times per minute for another 5 min. Tins containing water were replaced with tins containing dispersing solution, and the process was repeated.

Once raised out of the solution, samples in the sieve-bottom cups were gently rubbed with a rubber-tipped rod for 20 s each. Samples were lowered again into dispersing solution and raised and lowered for a final 5 min. The two sets of tins, one containing water and unstable soil fraction and the other containing dispersing solution and the stable fraction, were removed and placed in a drying oven at 110°C for 2 days. Any sand, rock, or particulate organic matter remaining in the sieve bottom cup was discarded.

WSA percentage was measured using the equation:

$$\left[\frac{\text{Stable Aggregate}}{\text{Stable Aggregate} + \text{Unstable Aggregate}} \right] * 100$$

where stable aggregate refers to the fraction of the soil slaked off in dispersing solution and unstable aggregate refers to the fraction of the soil that slaked off in water.

Two replicates were performed for each plot and a percent difference between replicates was determined using the equation:

$$[(\text{Rep 1} - \text{Rep 2}) / \text{Average}(\text{Rep1, Rep2})] * 100$$

If the percent difference between replicates was greater than 29%, a third replicate was performed. This was only necessary for one sample.

Statistical analysis

Because the WSA procedure was carried out by different individuals in 2010, 2013, and 2019, potential variation in individuals' techniques called for separate analyses of 2010, 2013 and 2019 results, and standardized scores were generated for water stable aggregate data within each year. Standardized scores were calculated by subtracting the population mean (all blocks and both systems) from each WSA percentage of each sampled plot and dividing this value by the population standard deviation.

Data were analyzed with PROC MIXED for a split-plot design in JMP Pro 15 by SAS (SAS Institute, Inc. Cary, North Carolina). For TOC, labile carbon, WSA, and standardized WSA, fixed effects were year, system, crop, and the two and three-way interactions between these terms with block, block \times crop, and block \times year as random effects, with fixed effects separated by the Personality Standard Least Squares test, equivalent to the Satterthwaite approximation. Bulk density and total organic carbon by volume was analyzed for 2019, which was the only year with sufficient bulk density data. Crop and system and the interaction between them were fixed effects and block and the interaction between block and crop were random. The SLICE test, which analyzes simple effects to separate LSmeans within an interaction, was used to test the pre-planned hypotheses, comparing the system within the same crop and year. Means were considered significantly different at $p < 0.05$. We also conducted the SLICE test when there was a significant interaction, with the exception of testing the Year effect. We only conducted the SLICE test to compare 2013 and 2019, the years when crops were planted in the same plots or soil and appropriate to compare between those years.

Results and discussion

Total organic carbon concentration

For TOC at the 0–5 cm depth, “system” showed the only significant effect ($p = 0.00089$), where ST-IWM averaged 14% lower TOC than the NT-SH average. Canola TOC concentration in ST-IWM was 21% less in 2013 ($p = 0.0197$) and 29% less in 2019 ($p = 0.0013$), compared to the NT-SH system (Table 2). Others have also reported a reduction in soil TOC in the upper soil layer following inversion tillage compared to no-till (Kettler et al., 2000; Jarecki and Lal, 2003; Mary et al., 2020).

By contrast, there were no significant differences between systems in the AO3 in any year (Table 2). In 2013 the ST-IWM

AO3 had not yet been exposed to tillage since the start of the experiment, but in 2019 the ST-IWM AO3 had experienced inversion tillage 6 years earlier. The similar TOC concentrations indicated that levels in the ST-IWM system had recovered to those observed in the NT-SH system (Table 2). In Uruguay, Salvo et al. (2014) also found that when rotated to perennial pastures following tillage, SOC was not reduced relative to no-till. In addition, in 14 grain crop locations that were occasionally tilled in Australia, Dang et al. (2015) found that soil organic carbon in the top 0–10 cm did not differ from paired no-till systems three to 24 months after tillage.

For TOC concentration at 5–15 cm depth, none of the fixed effects were significant. And the hypothesis that the systems would differ after tillage in Canola was not true in any year (Table 2), indicating tillage did not have an effect at this depth. It is possible that some TOC was redistributed to a deeper depth than what was sampled in our study, as Kettler et al. (2000) noted a redistribution of carbon to deeper depths following tillage. Because the plow depth is closer to 30 cm, increased TOC from buried residue could be buried in the 15–30 cm layer. Due to channery soils, sampling past 15 cm was not feasible on a wide scale and this possibility was not investigated. The potential for redistribution of a portion of the carbon, rather than loss of carbon due to tillage, would support the idea that occasional tillage does not reverse the benefits of no-till.

Bulk density and calculation of total organic C by volume

In the 2019 bulk density analysis, only crop showed a significant effect in the 0–5 cm depth ($p = 0.00418$), where AO3 had 9% lower average bulk density than the canola and AO2 was not significantly different from either of the other crops (Table 3). Overall, bulk density values at both depths were lower than 1.55 g cm^{-3} , which is the bulk density considered to be root-restrictive for silt loam soils (Kaufmann et al., 2010). Nevertheless, lowered bulk densities are an indication of higher organic matter content, greater porosity, and improved soil hydraulic function (Kaufmann et al., 2010; Bagnal et al., 2022). The lower bulk density after 2 years of perennial forage, regardless of tillage, thus suggested that soils had better structure in support of root growth. As with the TOC concentration results, there were no significant effects observed in the 5–15 cm depth for bulk density (Table 3).

Decreased bulk density following perennial crops was expected as the perennial roots are likely to add organic matter to the soil, as well as improving fungal hyphae and biological activity that promotes soil porosity, making the soil less dense (Angers and Caron, 1998; Blanco-Canqui and Ruis, 2020) and reducing the potential for compaction. This result supports our hypothesis that the lack of disturbance during perennial

TABLE 2 Total organic carbon (g C kg soil⁻¹) at 0–5 cm and 5–15 cm, labile carbon by POXC (mg C kg soil⁻¹) at 0–5 cm, water stable aggregates at 0–15 cm and water stable aggregate standard scores, calculated using the means and standard deviation of within-year score compared in 2010, 2013, and 2019.

	TOC (0–5 cm) g C kg ⁻¹	TOC (5–15 cm) g C kg ⁻¹	POXC (0–5 cm) mg C kg ⁻¹	WSA %	WSA standard score
Crop					
Average of Can	1.64a ^a	1.32a	458a	41.3b	−0.43b
Average of AO3	1.62a	1.25a	478a	46.8a	0.46a
System					
Average of ST-IWM	1.51b	1.31a	424b	42.9a	−0.15a
Average of NT-SH	1.75a	1.27a	512a	45.3a	0.19a
Year					
Average of 2010	1.61a	1.23a	495a	35.8b	5.39E − 07a
Average of 2013	1.64a	1.37a	446a	40.7b	−7.03E − 08a
Average of 2019	1.64a	1.27a	462a	55.7a	0.05a
Three-way effect combination					
Can, ST-IWM, 2010	1.57A ^b	1.20A	500A	32.7A	−0.33A
Can, NT-SH, 2010	1.62A	1.28A	473A	32.4A	−0.37A
AO3, ST-IWM, 2010	1.57A	1.22A	515A	36.8A	0.12A
AO3, NT-SH, 2010	1.67A	1.22A	494A	41.1A	0.59A
Can, ST-IWM, 2013	1.47Ba ^c	1.52Aa	353Bb	38.0Aab	−0.65Aa
Can, NT-SH, 2013	1.87Aa	1.35Aa	506Aa	38.8Aab	−0.46Aa
AO3, ST-IWM, 2013	1.59Aa	1.34Aa	440Aa	43.5Ab	0.68Aa
AO3, NT-SH, 2013	1.65Aa	1.25Aa	486Aa	42.5Ab	0.43Aa
Can, ST-IWM, 2019	1.37Ba	1.35Aa	342Bb	48.0Ba	−1.17Ba
Can, NT-SH, 2019	1.94Ab	1.26Aa	574Aa	58.1Aa	0.40Aa
AO3, ST-IWM, 2019	1.47Aa	1.23Aa	395Ba	58.2Aa	0.43Aa
AO3, NT-SH, 2019	1.76Aa	1.25Aa	537Aa	58.7Aa	0.53Aa
Main effect					
	<i>p-value</i>	<i>p-value</i>	<i>p-value</i>	<i>p-value</i>	<i>p-value</i>
Crop	0.69561	0.14599	0.48898	0.03776	0.02489
System	0.00089	0.40978	0.00284	0.19563	0.19915
Year	0.85648	0.11073	0.12520	0.00067	0.98440
System × crop	0.15841	0.68003	0.23882	0.52971	0.37678
System × year	0.09678	0.35064	0.01134	0.48342	0.40040
Crop × year	0.88381	0.60267	0.92488	0.91313	0.80845
System × crop × year	0.42148	0.70549	0.64071	0.29173	0.32388
SLICE tests					
Effect of system					
	<i>p-value</i>	<i>p-value</i>	<i>p-value</i>	<i>p-value</i>	<i>p-value</i>
Can, ST-IWM, 2010 vs. Can, NT-SH, 2010	0.7770	0.5058	0.6741	0.9383	0.9531
Can, ST-IWM, 2013 vs. Can, NT-SH, 2013	0.0197	0.1474	0.0264	0.8512	0.7530
Can, ST-IWM, 2019 vs. Can, NT-SH, 2019	0.0013	0.4560	0.0014	0.0278	0.0194
AO3, ST-IWM, 2010 vs. AO3, NT-SH, 2010	0.5305	0.9667	0.7505	0.3285	0.4726
AO3, ST-IWM, 2013 vs. AO3, NT-SH, 2013	0.6828	0.4672	0.4842	0.8119	0.6899
AO3, ST-IWM, 2019 vs. AO3, NT-SH, 2019	0.0827	0.8868	0.0374	0.9228	0.8795
Effect of year					
Can, ST-IWM, 2013 vs. Can, ST-IWM, 2019	0.4963	0.1479	0.8500	0.0351	0.4189
AO3, ST-IWM, 2013 vs. AO3, ST-IWM, 2019	0.4732	0.3560	0.4584	0.0028	0.6893
Can, NT-SH, 2013 vs. Can, NT-SH, 2019	0.6275	0.4688	0.2566	0.0002	0.1802
AO3, NT-SH, 2013 vs. AO3, NT-SH, 2019	0.4811	0.9541	0.3942	0.0027	0.8793

^a Lowercase letters (a, b) denote differences due to main effects at $p < .05$.

^b Uppercase letters (A, B) denote systems that differ at $p < .05$ within the same crop and year via the “SLICE” procedure.

^c Italicized lowercase letters (a, b) denote years (2013 vs. 2019) that differ at $p < .05$ within the same system and crop via the “SLICE” procedure.

Significant differences of main system and year effects comparing 2013 and 2019 (System and Year) were determined by the SLICE statement in JMP to perform a partitioned F test of least square means (LSMeans) of their interaction. p -values < 0.05 are in bold.

TABLE 3 Bulk density and TOC by volume averages at 0–5 and 5–15 cm depths for strategic tillage within integrated weed management (ST-IWM) and continuous no-till with standard herbicide (NT-SH) in canola (Can), second-year alfalfa orchardgrass (AO2) and third-year alfalfa orchardgrass (AO3) crops sampled in 2019.

	BD (0–5 cm) g cm ⁻³	BD (5–15 cm) g cm ⁻³	TOC (0–5 cm) Mg ha ⁻¹	TOC (5–15 cm) Mg ha ⁻¹
System average				
ST-IWM	1.13A	1.26A	8.35B	7.92A
NT-SH	1.09A	1.26A	10.41A	7.84A
Crop average				
Can	1.16a	1.26a	9.58a	8.18a
AO2	1.11ab	1.26a	10.03a	8.00a
AO3	1.06b	1.27a	8.53a	7.47a
System × crop				
ST-IWM, Can	1.17a ^a A ^b	1.23aA	7.97aB	8.30aA
ST-IWM, AO2	1.13abA	1.26aA	9.09aB	8.01aA
ST-IWM, AO3	1.08bA	1.28aA	8.00aA	7.46aA
NT-SH, Can	1.15aA	1.28aA	11.18aA	8.06aA
NT-SH, AO2	1.09bA	1.25aA	10.97aA	7.98aA
NT-SH, AO3	1.03bA	1.27aA	9.07bA	7.48aA
Factor	<i>p-value</i>	<i>p-value</i>	<i>p-value</i>	<i>p-value</i>
Crop	0.00418	0.859	0.07343	0.22671
System	0.07235	0.80057	0.00012	0.84223
System × crop	0.64693	0.43808	0.06324	0.96056
SLICE tests	<i>p-value</i>	<i>p-value</i>	<i>p-value</i>	<i>p-value</i>
Effect of system				
Can, ST-IWM vs.	0.6596	0.2078	0.0003	0.7152
Can, NT-SH				
AO2, ST-IWM vs.	0.2353	0.7452	0.0078	0.9659
AO2, NT-SH				
AO3, ST-IWM vs.	0.1058	0.7459	0.0844	0.9820
AO3, NT-SH				

^a Different lowercase letters (a, b) indicate that crops differ at $p < .05$.

^b Different uppercase letters (A, B) indicate that systems differ at $p < .05$.

Significant differences of the main effects of system (system) and crop (Crop) were determined by the SLICE function in JMP to conduct a partitioned F-test of LSMeans of the interaction of System × Crop. p -values < 0.05 are in bold.

growth would lead to improvement in soil health indicators. Interestingly, the bulk density did not increase significantly following tillage as there was no difference between the two systems in the canola crop. This did not support our first hypothesis that strategic tillage would cause an initial decrease in soil quality. Tillage has been shown to decrease bulk density in the short term but increase it in the long-term (Logsdon and Karlen, 2004; Dang et al., 2015). However, we did not detect a difference in systems or following tillage in the canola.

The TOC by volume (Mg/ha) showed similar trends to the TOC concentration in both sampled depths. System had a significant effect in the 0–5 cm depth ($p = 0.00012$), with TOC by volume 20% smaller in ST-IWM compared to the NT-SH. The canola had 29% less ($p = 0.0003$) and AO2 had 17% less TOC by

volume ($p = 0.0078$) in ST-IWM compared to NT-SH (Table 3). There were no significant differences however, between the two systems in AO3 (Table 3), indicating that following strategic tillage, 2 years of the perennial forage were required for TOC to increase to the same level as the no-till system. At the 5–15 cm depth, there were no significant differences in TOC by volume between systems or crops (Table 3). In summary, TOC by volume decreased significantly following tillage, but was similar between systems following 3 years of annuals and cover crops and 2 years of perennials.

Labile carbon

In the analysis of POXC, system showed a significant effect ($p = 0.00079$), where ST-IWM averaged 17% lower than NT-SH. Compared to NT-SH, canola POXC in ST-IWM was 30% less in 2013 ($p = 0.0264$) and 40% less in 2019 ($p = 0.0014$, Table 2). However in 2019, POXC in AO3 of the ST-IWM system was 26% less than in the NT-SH system ($p = 0.0374$), indicating that POXC had not increased to similar levels in ST-IWM 6 years after tillage. In 2013, AO3 in the ST-IWM system hadn't yet experienced tillage, while in 2019, the AO3 in that system had been tilled 6 years before. Quincke et al. (2007) also noted a significant decrease in labile carbon following tillage in the top few centimeters of soil, but found increased labile carbon in deeper layers that essentially offset the carbon lost from the more superficial layer, suggesting that labile carbon had been redistributed by moldboard plow rather than lost. In our study, POXC was only analyzed at the 0–5 cm depth, and it may have been possible that a similar redistribution of carbon occurred at deeper soil depth. Kettler et al. (2000) and Mary et al. (2020) also noted a redistribution of carbon to deeper depths following tillage.

Water stable aggregates

In the analysis of the WSA and standardized WSA scores, only year showed a significant effect on WSA ($p = 0.00067$) while crop had significant effects on both scores ($p = 0.03776$ and $p = 0.02489$, respectively, Table 2). Compared to 2010, the average WSA was 37% smaller than the averages for 2013 and 2019. The larger average WSA values in 2013 and 2019 may be attributed to the combination of cover crops, perennials and lack of disturbance in most years of both systems. The role of different people conducting the WSA analysis also likely explains some of the WSA differences among years. Year-to-year variation was why we used a standardization procedure for raw WSA scores. Compared to the canola, WSA and the standardized score in AO3 averaged 11.8% greater and 1.9 fold greater, respectively (Table 2). Others have also reported that perennial roots and their associated fungal hyphae and soil

microorganisms stabilized soil aggregates regardless of tillage (Angers and Caron, 1998).

Following tillage in 2019, WSA of canola in the ST-IWM system was 17% smaller than in NT-SH ($p = 0.0278$), and the standardized WSA score was also significantly smaller in the ST-IWM system ($p = 0.0194$). However, WSA scores for canola in 2013 did not differ between the two systems when the NT-SH system had not included orchardgrass in the perennial forage prior to herbicide termination. In 2019, by contrast, AO3 in the systems did not differ for either WSA or the standardized scores, indicating that less disturbance with cover crops and 2 years of perennial alfalfa and orchardgrass significantly enhanced the physical soil stability in the ST-IWM. The lack of significant difference following tillage in the 2013 canola, however, may also be explained by the presence of orchardgrass in addition to alfalfa in the ST-IWM system when the NT-SH was planted to only alfalfa. The presence of orchardgrass in the ST-IWM system may have promoted and protected WSA enough to counter the tillage impact. Because perennial grasses have more fibrous roots, it has been suggested that they promote greater aggregate stability than other types of perennials (Miller and Jastrow, 1990; Angers and Caron, 1998; Rachman et al., 2003). In 2019, the NT-SH system also had orchardgrass planted with alfalfa, and the benefits of both no-till and perennial grass roots may have assisted in the formation of stable aggregates.

Other studies that compared occasional tillage to continuous no-till have also reported that occasional tillage often did not reduce soil aggregation, or that it recovered within a year with return to no-till (Dang et al., 2015; Blanco-Canqui and Wortmann, 2020). A 2010 study by Wortmann et al. found no significant differences in water stable aggregates across several tillage treatments, including no-till and moldboard plow tillage, 5 years following tillage. It is possible that ST-IWM might have returned to levels similar to NT-SH after several years without the aid of perennial roots. However, Dougherty et al. (2022) have suggested that only slight differences in soil health indicators can be expected within short timeframes, which may explain the lack of significant differences in the Wortmann et al. (2010) study. Additionally, the Wortmann et al. (2010) study used the water stable aggregate sampling method by Cambardella and Elliott (1994), which involves wet sieving soil and combining weights of three aggregate size classes, which could have accounted for different sensitivities.

Conclusion

Herbicides are typically used to terminate cover and perennial crops and control weeds during no-till management of annual and perennial crop rotations. Reliance on herbicides can lead to herbicide-resistant weeds and negative environmental impacts that conflict with the goals of regenerative agriculture. Integrated weed management employs multiple weed control practices but the use of occasional tillage is often dismissed in

minimal tillage systems because of its association with reduced soil health. In this study, we found that most of the negative effects of strategic tillage on soil health indicators (soil carbon at 0–5 cm, water stable aggregates and bulk density) were mitigated in no-till annual and perennial cropping systems after 3 years' growth of annuals and cover crops and 2 years of perennial forages. We also found evidence that integrating perennial forage grasses with legumes is beneficial for promoting water stable aggregates. When compared to the continuous NT system, labile carbon was the only soil indicator that was lower in the ST-IWM system when measured in the third year of alfalfa-orchardgrass (AO3). Further research on the significance of labile carbon for total soil carbon accumulation and the long-term impacts of strategic tillage in these systems would elucidate the implications of this small reduction in labile carbon. Additionally, this study found no effects of the strategic tillage on soil carbon and bulk density at the 5–15 cm depth, implying that tillage did not impact soil health below the 0–5 cm range in this study, although an investigation of soil carbon at lower depths would be worth pursuing.

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

Author contributions

DM, HK, and MB contributed to the conception and design of the study. CD advised on data collection. DM generated and organized data. DM and HK performed statistical analysis. DM wrote the first draft of the manuscript. All authors contributed to manuscript revision, read, and approved the submitted version.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

References

- Angers, D. A., and Caron, J. (1998). Plant-induced changes in soil structure: processes and feedbacks. *Biogeochemistry* 42, 55–72. doi: 10.1007/978-94-017-2691-7_3
- Bagnal, D.K., Morgan, C.L.S., Bean, G.M., Liptzin, D., Cappellazzi, S.B., et al. (2022). Selecting soil hydraulic properties as indicators of soil health: measurement responses to management and site characteristics. *Soil Sci. Soc. Am. J.* 86, 1206–1226. doi: 10.1002/saj2.20428
- Baker, C. J., Justice, S., Saxton, K., Hobbs, P., Ritchie, W., Charmen, W., et al. (2007). *No-Tillage Seeding in Conservation Agriculture*, 2nd ed. CABI Publishing, FAO. Available online at: <http://www.fao.org/docrep/012/A1298e/al298e.pdf> doi: 10.1079/9781845931162.0000
- Baker, D. B., Johnson, L. T., Confesor, R. B. and Crumrine J. P., (2017). Vertical stratification of soil phosphorus as a concern for dissolved phosphorus runoff in the Lake Erie Basin. *J. Environ. Qual.* 46, 1287–1295 doi: 10.2134/jeq2016.09.0337
- Basche, A. D., and DeLonge, M. S. (2019). Comparing infiltration rates in soils managed with conventional and alternative farming methods: A meta-analysis. *PLoS ONE* 14, e0215702. doi: 10.1371/journal.pone.0215702
- Bhardwaj, A. K., Jasrotia, P., Hamilton, S. K., and Robertson, G. P. (2011). Ecological management of intensively cropped agro-ecosystems improves soil quality with sustained productivity. *Agric. Ecosys. Environ.* 140, 419–429. doi: 10.1016/j.agee.2011.01.005
- Blake, G. R. and Hartge, K. H. (1986) "Bulk density," in *Methods of Soil Analysis, Part 1—Physical and Mineralogical Methods, 2nd Edition* ed. Klute, A., Agronomy Monograph 9, American Society of Agronomy—Soil Science Society of America, Madison, 363–382. doi: 10.2136/sssabookser5.1.2ed.c13
- Blanco-Canqui, H., and Ruis, S. J. (2020). Cover crop impacts on soil physical properties: a review. *Soil Sci. Soc. Am. J.* 84, 1527–1576. doi: 10.1002/saj2.20129
- Blanco-Canqui, H., and Wortmann, C. S. (2020). Does occasional tillage undo the ecosystem services gained with no-till? A review. *Soil Tillage Res.* 198:104534. doi: 10.1016/j.still.2019.104534
- Cambardella, C. A., and Elliott E. T. (1994). Carbon and nitrogen dynamics of soil organic matter fractions from cultivated grassland soils. *Soil Sci. Soc. Am. J.* 58:123–130. doi: 10.2136/sssaj1994.03615995005800010017x
- Cates, A. M., Ruark, M. D., Hedtcke, J. L., and Posner, J. L. (2016). Long-term tillage, rotation and perennialization effects on particulate and aggregate soil organic matter. *Soil Tillage Res.* 155, 371–380. doi: 10.1016/j.still.2015.09.008
- Cavigelli, M. A., Teasdale, J. R., and Conklin, A. E. (2008). Long-term agronomic performance of organic and conventional field crops in the mid-atlantic region. *Agron. J.* 100, 785–94. doi: 10.2134/agronj2006.0373
- Congreves, K. A., Hayes, K., Verhallen, E. A., and Van Eerd, L. L. (2015). Long-term impact of tillage and crop rotation on soil health at four temperate agroecosystems. *Soil Tillage Res.* 152, 17–28. doi: 10.1016/j.still.2015.03.012
- Culman, S. W., Snapp, S. S., Freeman, M. A., Schipanski, M. E. Beniston, J., and Lal, R., et al. (2012). Permanganate oxidizable carbon reflects a processed soil fraction that is sensitive to management. *Soil Sci. Soc. Am. J.* 76, 494–504. doi: 10.2136/sssaj2011.0286
- Dang, Y. P., Moody P. W., Bell, M. J., Seymour, N. P., Dalal, R. C., and Freebairn, D. M. (2015). Strategic tillage in no-till farming systems in Australia's northern grains-growing regions: II. Implications for Agronomy, Soil and Environment. *Soil and Tillage Research. Elsevier* 152, 115–123 doi: 10.1016/j.still.2014.12.013
- Daryanto, S., Wang, L., and Jacinthe, P.A. (2020). No-till is challenged: complementary management is crucial to improve its environmental benefits under a changing climate. *Geogr. Sustain.* 1:229–232. doi: 10.1016/j.geosus.2020.09.003
- Davis, A. S., Hill, J. D., Chase, C. A., Johanns, A. M., and Liebman, M. (2012). Increasing cropping system diversity balances productivity, profitability and environmental health. *PLOS ONE*, 7: 1–13. doi: 10.1371/journal.pone.0047149
- Dougherty, B. W., Andersen, D. S., and Helmers, M. J. (2022). Evaluating the impact of midwestern cropping systems on soil health and soil carbon dynamics. *J. Soil Water Conserv.* 77, 78–87. doi: 10.2489/jswc.2022.00056
- Freemark, K., and Boutin, C. (1995). Impacts of agricultural herbicide use on terrestrial wildlife in temperate landscapes: a review with special reference to North America. *Agric. Ecosys. Environ.* 52, 7–91. doi: 10.1016/0167-8809(94)00534-L
- Gaupp-Berghausen, M., Hofer, M., Rewald, B., and Zaller, J. G. (2015). Glyphosate-Based herbicides reduce the activity and reproduction of earthworms and lead to increased soil nutrient concentrations. *Sci. Rep.* 5, 12886. doi: 10.1038/srep12886
- Green, J. M., and Siehl, D. L. (2021). History and outlook for glyphosate-resistant crops. *Reviews of Environmental Contamination and Toxicology Volume* 255: 67–91. Cham: Springer International Publishing. doi: 10.1007/398_2020_54
- Grover, K. K. (2008). *Long-Term Cropping Systems Effects on Soil Aggregate Stability, Corn Grain Yields, and Yield Stability*. Ph. D. diss. Dep. of Crop and Soil Sci. The Pennsylvania State Univ. University Park.
- Haynes, R. J., and Swift, R. S. (1990). Stability of soil aggregates in relation to organic constituents and soil water content. *J. Soil Sci.* 41, 73–83. doi: 10.1111/j.1365-2389.1990.tb00046.x
- Heap, I. (2022). The International survey of herbicide resistant weeds. Online. Internet. Available online at: www.weedscience.org (accessed July 26, 2022).
- Holly, M. A., Gunn, K. M., Rotz, C. A., Kleinman, P. J. A. (2019). Management characteristics of Pennsylvania dairy farms. *Appl. Anim. Sci.* 35, 325–338. doi: 10.15232/aas.2018-01833
- Hunt, N. D., Hill, J. D., and Liebman, M. (2017). Reducing freshwater toxicity while maintaining weed control, profits, and productivity: effects of increased crop rotation diversity and reduced herbicide usage. *Environ. Sci. Technol.* 51, 1707–17. doi: 10.1021/acs.est.6b04086
- Hurisso, T. T., Culman, S. W., Horwath, W. R., Wade, J., Cass, D., Beniston, J. W., Bowles, T. M., Grandy, A. S., Franzluebbers, A. J., Schipanski, M. E., Lucas, S. T., Ugarte, C. M. (2016). Comparison of permanganate-oxidizable carbon and mineralizable carbon for assessment of organic matter stabilization and mineralization. *Soil Sci. Soc. Am. J.* 80, 1352–1364. doi: 10.2136/sssaj2016.04.0106
- Jarecki, M. K., and Lal, R. (2003). Crop management for soil carbon sequestration. *Crit. Rev. Plant Sci.* 22, 471–502. doi: 10.1080/713608318
- Kan, Z. R., Liu, Q. Y., Virk, A. L., He, C., Qi, J. Y., and Dang, Y. P., et al. (2021). Effects of experiment duration on carbon mineralization and accumulation under no-till. *Soil Tillage Res.* 209, 104939. doi: 10.1016/j.still.2021.104939
- Kaufmann, M., Tobias, S., Schulin, R. (2010). Comparison of critical limits for crop plant growth based on different indicators for the state of soil compaction. *J. Plant Nutr. Soil.* 173,573–583. doi: 10.1002/jpln.200900129
- Kemper, W. and Rosenau, R. (1986). "Aggregate stability and size distribution", in *Methods of Soil Analysis, Part 1. Physical and Mineralogical Methods, Agronomy Monograph, American Society of Agronomy and Soil Science Society of America, Madison*, Klute, A., ed. 425–442. doi: 10.2136/sssabookser5.1.2ed.c17

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- Kettler, T. A., Lyon, D. J., Doran, J. W., Powers, W. L., and Stroup, W. W. (2000). Soil quality assessment after weed-control tillage in a no-till wheat-fallow cropping system. *Soil Sci. Soc. Am. J.* 64, 339–346. doi: 10.2136/sssaj2000.641339x
- King, A. E., and Blesh, J. (2017). Crop rotations for increased soil carbon: perennality as a guiding principle. *Ecol. Appl.* 28, 249–261. doi: 10.1002/eap.1648
- Logsdon, S. D., and Karlen, D. L. (2004). Bulk density as a soil quality indicator during conversion to no-tillage. *Soil Tillage Res.* 78, 143–149. doi: 10.1016/j.still.2004.02.003
- Mary, B., Clivot, H., Blaszczyk, N., Labreuche, J., and Ferchaud, F. (2020). Soil carbon storage and mineralization rates are affected by carbon inputs rather than physical disturbance: evidence from a 47-year tillage experiment. *Agric. Ecosyst. Environ.*, 299, 106972. doi: 10.1016/j.agee.2020.106972
- Miller, R. M., and Jastrow, J. D. (1990). Hierarchy of root and mycorrhizal fungal interactions with soil aggregation. *Soil Biol. Biochem.*, 22, 579–584. doi: 10.1016/0038-0717(90)90001-G
- National Agricultural Statistics Service (NASS) USDA (2017). Census of Agriculture, Pennsylvania. Chpt. 1 Table 47. Available online at: https://www.nass.usda.gov/Publications/AgCensus/2017/Full_Report/Volume_1,_Chapter_1_State_Level/Pennsylvania/st42_1_0047_0047.pdf
- Norton, R. M. (2020). “Challenges and opportunities in fertilizer placement in no-till farming systems”, in *No-till Farming Systems for Sustainable Agriculture*, Dang, Y., Dalal, R., Menzies, N. (eds.). Springer, Cham. doi: 10.1007/978-3-030-46409-7_5
- Nunes, M. R., Karlen, D. L., Veum, K. S., Moorman, T. B., and Cambardella, C. A. (2020). Biological soil health indicators respond to tillage intensity: a US meta-analysis. *Geoderma*, 369:114335. doi: 10.1016/j.geoderma.2020.114335
- Ogle, S. M., Swan, A., and Paustian, K. (2012). No-till management impacts on crop productivity, carbon input and soil carbon sequestration. *Agric. Ecosyst. Environ.* 149, 37–49. doi: 10.1016/j.agee.2011.12.010
- Osterholz, W. R., Culman, S. W., Herms, C., Joaquim De Oliveira, F., Robinson, A., and Doohan, D., et al. (2021). Knowledge gaps in organic research: understanding interactions of cover crops and tillage for weed control and soil health. *Org. Agr.* 11, 13–25. doi: 10.1007/s13165-020-00313-3
- PA No-Till Alliance. 4, September, (2022). <https://panotill.org/about/>
- Powlson, D. S., Stirling, C. M., Jat, M. L., Gerard, B. G., Palm, C. A., and Sanchez, P. A., et al. (2014). Limited potential of no-till agriculture for climate change mitigation. *Nat. Clim. Chang.*, 4, 678–683. doi: 10.1038/nclimate2292
- Quincke, J. A., Wortmann, C. S., Mamo, M., Franti, T. G., Drijber, R. A., and Garcia, J. (2007). One-time tillage of no-till systems: soil physical properties, phosphorus runoff, and crop yield. *Agron. J.*, 99, 1104–1110. doi: 10.2134/agronj2006.0321
- Rachman, A., Anderson, S. H., Gantzer, C. J., and Thompson, A. L. (2003). Influence of Long-term Cropping Systems on Soil Physical Properties Related to Soil Erodibility. *Soil Sci. Soc. Am. J.*, 67: 637–644. doi: 10.2136/sssaj2003.6370
- Salvo, L., Hernandez, J., Ernst, O. (2014). Soil organic carbon dynamics under different tillage systems in rotations with perennial pastures. *Soil Tillage Res.*, 135, 41–48. doi: 10.1016/j.still.2013.08.014
- Sanborn, M., Kerr, K. J., Sanin, L. H., Cole, D. C., Bassil, K. L., and Vakil, C. (2007). Non-cancer health effects of pesticides: systematic review and implications for family doctors. *Can. Fam. Physician.* 53, 1712–1720.
- Scheiner, J. D., and Lavado, R. S. (1998). The role of fertilization on phosphorus stratification in no-till soils. *Commun. Soil Sci. Plant Anal.* 29, 2705–2711.
- Sprunger, C. D., Culman, S. W., Deiss, L., Brock, C., and Jackson-Smith, D. (2021). Which management practices influence soil health in midwest organic corn systems? *Agron. J.* 113, 4201–4219. doi: 10.1002/agj2.20786
- Sprunger, C. D., Martin, T. and Mann, M. (2020). Systems with greater perennality and crop diversity enhance soil biological health. *Agric. Environ. Lett.*, 5, 1. doi: 10.1002/acl2.20030
- Stavi, I., Lal, R., and Owens, L. B. (2011). On-farm effects of no-till vs. occasional tillage on soil quality and crop yields in eastern Ohio. *Agron. Sustain. Develop.*, 31, 475–482. doi: 10.1007/s13593-011-0006-4
- Stradtman, S. C., and Freeman, J. L. (2021). Mechanisms of neurotoxicity associated with exposure to the herbicide atrazine. *Toxics*, 9, 207. doi: 10.3390/toxics9090207
- Stubbs, T. L., Kennedy, A. C., and Schillinger, W. F. (2004). Soil ecosystem changes during the transition to no-till cropping. *J. Crop. Improv.* 11, 105–135. doi: 10.1300/J411v11n01_06
- Summers, H., Karsten, H. D., Curran, W., and Malcolm, G. M. (2021). Integrated weed management with reduced herbicides in a no-till dairy rotation. *Agron. J.*, 113, 3418–3433. doi: 10.1002/agj2.20757
- Weil, R. R., Islam, K. R., Stine, M. A., Gruver, J. B., and Samson-Liebig, S. E. (2003). Estimating active carbon for soil quality assessment: a simplified method for laboratory and field use. *Am. J. Altern. Agric.* 18:3–17. doi: 10.1079/AJAA2003003
- Wortmann, C. S., Drijber, R. A., and Franti, T. G. (2010). One-time tillage of no-till crop land five years post-tillage. *Agron. J.* 102, 1302–1307. doi: 10.2134/agronj2010.0051
- Zhang, L., Rana, I., Shaffer, R. M., Taioli, E., and Sheppard, L. (2019). Exposure to glyphosate-based herbicides and risk for Non-Hodgkin lymphoma: A Meta-analysis and supporting evidence. *Mutation Res. Rev. Mut. Res.*, 781, 186–206. doi: 10.1016/j.mrrev.2019.02.001



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E. Britt Moore,
University of North Carolina
Wilmington, United States

REVIEWED BY

Vesna Radovanović,
Educons University, Serbia
Shah Fahad,
The University of Haripur, Pakistan

*CORRESPONDENCE

Erin M. Silva
emsilva@wisc.edu

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Cover crop-based reduced tillage management impacts organic squash yield, pest pressure, and management time

Dylan Bruce^{1,2}, Erin M. Silva^{1*} and Julie C. Dawson²

¹Department of Plant Pathology, University of Wisconsin-Madison, Madison, WI, United States,

²Department of Horticulture, University of Wisconsin-Madison, Madison, WI, United States

Cover crop-based reduced tillage (CCBRT) systems can provide multiple benefits in cucurbit cropping systems, including potential to reduce spread of soil-borne pathogens, minimize erosion, and decrease weed pressure. Despite benefits and farmer interest, adoption has been limited, in part due to inconsistent weed suppression and potential for reduced yields. Prior studies have suggested that N competition, allelopathy, and lower temperature may be factors in reducing vegetable yield in CCBRT systems. A strip tillage approach has been suggested as one strategy that could mitigate those issues, but cucurbit yields using these systems have shown mixed results in prior studies, some of which did not include other important considerations for growers such as the impact on weed and pest pressure. In 2018 and 2019, CCBRT strip till practices for organic acorn winter squash (*Cucurbita pepo* L.) production were assessed in Wisconsin on certified organic land. Combinations of different between-row (aisle) and in-row mulches were compared to attempt to identify reduced tillage combinations that effectively manage weeds while resulting in yields comparable to full tillage production, testing our hypothesis that no differences between production systems would be observed due to strip tillage and plastic mulch warming soil and minimizing competition while promoting cash crop growth. Aisle treatments included roller-crimped cereal rye (*Secale cereale* L.) mulch, straw mulch and cultivated bare ground, and in-row treatments included plastic mulch, ground straw mulch, and cultivated ground. Weed and pest counts, weed management time, and yields were compared between treatments. Plots managed with rye and straw in the aisles had significantly less weed pressure as compared to cultivated aisle treatments, although rye required more weed management time than ground straw mulch. In addition, rye resulted in lower marketable yield due to higher proportion unmarketable fruit in 2018, likely related to a 25 cm rain event 2 weeks prior to harvest. A significant row mulch × aisle mulch interaction was observed for marketable fruit m^{-1} , showing that yield was not significantly affected by the type of in-row mulch in plots with crimped rye mulch in the aisle. Pressure from squash bugs (*Anasa tristis*) was also higher in treatments with organic or synthetic mulches (straw in aisles or rows, rye in aisles, and plastic in rows). Our results support previous evidence that crimped rye can be an effective mulching strategy to reduce weed pressure, with more efficient management than traditional straw mulch. However, crimped rye

systems may have negative implications for yield and pest pressure regardless of the use of a strip-tillage approach, indicating that more research is needed to refine the production system.

KEYWORDS

organic agriculture, cover crop-based reduced tillage, acorn squash, cover crops, crimped rye, no till vegetables

Introduction

Weed management is consistently cited as a significant obstacle for organic farmers (Moynihan, 2010; Jenkins and Ory, 2016). To manage weeds in cucurbit crops, most organic growers rely heavily on either mechanical cultivation or black plastic mulches, both of which bring considerable economic and biological costs. For example, a recent survey of 105 organic farmers in Michigan revealed that the mean number of tillage passes in winter squash production was 6.5 per season, with some growers tilling as many as 15 times (Lowry and Brainard, 2019). Cover crops have been recognized as a valuable tool in the “many little hammers” approach to creating long-term organic production strategies that lower the weed seedbank while providing additional ecological benefits (Liebman et al., 1997; Baraibar et al., 2018; Wauters et al., 2021). Cover crops can support weed management through direct competition, the creation a physical barrier through crop residues, the release of allelochemicals, and the alteration of soil nutrient dynamics (Teasdale and Mohler, 2000; Sarrantonio and Gallandt, 2003; Hiltbrunner et al., 2007; Bezuidenhout et al., 2012; Teasdale et al., 2012; Brust et al., 2014). Beyond their weed suppressive benefits, cover crops also improve soil health and water quality by reducing erosion and increasing organic matter (Reicosky and Forcella, 1998; Sarrantonio and Gallandt, 2003; Ryder and Fares, 2008; Luo et al., 2010; De Baets et al., 2011; Kaspar et al., 2011).

Although cover crops are used extensively in organic production (USDA-NASS, 2019), adoption as a full-season weed control strategy has been limited, and cover crops are usually terminated and incorporated prior to planting the cash crop (Magdoff and Van Es, 2000), precluding potentially unique benefits afforded by full season cover crops (Deguchi et al., 2012). Shorter growing seasons in temperate climates (Snapp et al., 2005), and diverse, complex, and high value rotations on vegetable farms complicate integration of cover crops (Sarrantonio, 1992) into tillage-intensive production systems of northern cucurbit growers. Cover crop-based reduced tillage (CCBRT) encompasses a suite of practices which strategically integrate cover crops into a cash crop rotation with the goal of suppressing weeds while reducing soil disturbance (Vincent-Caboud et al., 2019). These practices frequently integrate the

use of a roller-crimper to create an *in-situ* mulch of killed cover crop residue into which the cash crop can be planted, providing a thick layer of biomass allowing for season-long weed suppression without the need for tillage and cultivation (Smith et al., 2011; Delate et al., 2012; Mirsky et al., 2012; Silva, 2014; Silva and Delate, 2017). Such full season applications extend the environmental benefit of cover crops typically limited by cash crop seasonality, while also limiting the number of tillage passes required and thus reducing production costs.

While much of the research regarding CCBRT has been conducted with grain crops, an increasing number of studies have evaluated this system for organic vegetable production. The performance of CCBRT in organic vegetable systems has varied widely depending on the vegetable crop, cover crop, and environment (Forcella et al., 2015; Chehade et al., 2019; Lounsbury et al., 2020). In certain circumstances, the practice has resulted in equivalent or greater vegetable yields than those obtained from more typical organic systems using mechanical weed management (e.g., Creamer et al., 1996; Campiglia et al., 2010; Vollmer et al., 2010; Lounsbury and Weil, 2015; Jokela and Nair, 2016; Sportelli et al., 2022), while in other studies, the system resulted in reduced yields (e.g., Leavitt et al., 2011; Delate et al., 2012; Bietila et al., 2016; Jokela and Nair, 2016).

Reduced yields under CCBRT management can often be attributed to several factors, including insufficient weed suppression and competition of the cover crop with the cash crop, such as for nitrogen (Vincent-Caboud et al., 2019). Slow nitrogen mineralization rates associated with lower soil temperatures can limit available nitrogen at key phases of crop growth within CCBRT systems (Leavitt et al., 2011). Some of the most used cover crops found in CCBRT management, such as cereal rye (*Secale cereale* L.), are characterized by high carbon to nitrogen (N) ratios at maturity which can lead to N immobilization, especially if cover crop residue remains on the soil surface rather than incorporated into the soil (Clark et al., 1994; Van Den Bossche et al., 2009; Salon, 2012; Chehade et al., 2019). The effects of these phenomena can be observed in the results of several CCBRT vegetable studies. For example, in Iowa, organic bell pepper yields under CCBRT management were comparable in one season, but lower during the second year, with the differences being attributed to differences in temperature and nutrient availability in soil under

no-till management (Jokela and Nair, 2016). This phenomenon may have also been a factor in the performance of CCBRT systems in the Northeastern US, where organic cabbage yields were reduced 21% and temperatures under rye mulch were 2–3°C lower than bare soil, although other factors such as stunting due to rye allelopathy may have also impacted final yields of the crop (Mochizuki et al., 2008). Leavitt et al. (2011) also suggested that lower temperatures in CCBRT treatments led to lower yields for organic tomato, pepper, and zucchini in Minnesota.

Strip tillage has been presented as an alternative management approach to mitigate the potential yield losses related to the adoption of CCBRT practices, including in organic vegetable systems (Delate et al., 2003, 2012; Mochizuki et al., 2007; Leavitt et al., 2011; Luna et al., 2012; Bietila et al., 2016; Ginakes and Grossman, 2021). With strip tillage management, primary tillage and associated cover crop incorporation is restricted to the in-row planting zone, with the aisles between the rows remaining undisturbed. Strip tillage systems have the potential to combine the weed management benefits of intensive cover cropping practices with soil-building and reduced soil disturbance, while reducing risk of yield loss compared to full NT systems (Thomas et al., 2001; Brainard et al., 2013). Strip tillage systems can promote plant growth and yields through quicker warming of soil temperatures comparable to conventional tillage systems but not as great as with the use of plastic mulch (Licht and Al-Kaisi, 2005; Tillman et al., 2015).

Further, strip tillage management allows for the incorporation of high-carbon crop residues with the planting zone, which supports microbial populations and promotes N mineralization (Brainard et al., 2013). In one case in a conventional pumpkin production system, the use of strip tillage in combination with a crimped rye/hairy vetch mixture increased the number of marketable fruits by reducing pathogen incidence (Everts, 2002).

CCBRT systems in vegetable production have shown clear promise for conserving soil health and playing a role in long term weed management. Yet results for pest pressure and, most critically, yield have been variable across different reduced tillage systems and vegetable crops (e.g., Delate et al., 2003; Snyder, 2015; Jabbour et al., 2016; Jokela and Nair, 2016; Skidmore et al., 2019), including for cucurbits specifically. For instance, Forcella et al. compared conventionally cultivated and crimped rye systems in Western Minnesota and found that cucumber (*Cucumis sativus* L.) yields were comparable, pumpkin (*Cucurbita pepo* L.) yields were 25% lower in rye, and watermelon (*Citrullus lanatus*) yields were 75% lower in rye (Forcella et al., 2015). However, few if any studies have assessed the feasibility of crimped rye as a management strategy for organic winter squash production while assessing critical management considerations such as yields, insect pest and weed pressure, and labor, all of which need to be understood before growers can be confident adapting CCBRT practices.

This study expands on previous research on organic CCBRT management for cucurbit systems by integrating strip tillage strategies, evaluating both in-row management of the tilled strips and between row (aisle) management strategies. Specific objectives included: (1) comparison of pest (weed, disease, and insect pressure) throughout the cucurbit production season, through the visual assessment of disease incidence and physical counts of insect and weed pressure; (2) comparison of yields through the fruit counts and weights of both marketable and unmarketable fruits; and (3) comparison of labor required for weed management throughout the cucurbit production season through tracking of hours needed for plot maintenance. Whole plot row mulch treatments representing possible strip tillage options included plastic mulch, straw mulch and bare cultivated ground, while split plot aisle mulch treatments included full tillage cultivated ground, straw mulch and crimped rye. Data collected included vegetable yield, plant survival rate, weed counts and management time, and cucumber beetle and squash bug counts.

Materials and methods

Site and treatment descriptions

Field trials were conducted at the University of Wisconsin's West Madison Agricultural Research Station (Verona, WI, USA) from September 2017 to September 2019. Two adjacent areas of certified organic land (43.0734, −89.5474 and 43.0744, −89.5465) were used for the experiment, both of which had been previously planted with a 3-year old alfalfa stand and managed in accordance with the United States Department of Agriculture National Organic Program (USDA-NOP) regulations (National Organic Program, 2000). Soil types were Batavia and Troxel silt loams, with organic matter content of 3.3% in 2018 and 2.9% in 2019, and pH 6.6 in 2018 and 7.2 in 2019. The experiment was established as a split-plot randomized complete block design with three replications, with row mulch as the whole-plot factor and aisle mulch as the strip-plot factor (Supplementary Figure 1). Each subplot had 10 plants. Whole plot, row mulch factors included a cultivated control, black plastic mulch, and ground straw mulch applied at a rate of 33,625 kg ha^{−1}. Strip plot, aisle mulch treatments included cereal rye crimped at anthesis with a roller-crimper (I&J Manufacturing, Gap, PA), ground winter wheat straw mulch applied at a rate of 33,625 kg ha^{−1}, and a cultivated control.

Field activities

Field activities are summarized in Table 1. Cereal rye was seeded in the entire study area with a Landoll grain drill (Landoll Corporation, Marysville, KS) at a rate of 250.96 kg

TABLE 1 Summary of field activities for CCBRT management of organic squash, 2018 and 2019.

Date (2017)	Date (2018)	Date (2019)	Activity
September 25	September 27		“Aroostock” rye seeding (4 bu/acre)
	May 17, June 14	May 15, May 30, June 11	Tilling planting strips and control plots to terminate cover crop or incorporate fertilizer
	June 6	June 6	Rye biomass
	June 7	June 7	Termination of rye plots by crimping
	June 14	June 11	Application of fertilizer
	June 14	June 12	Application of straw and plastic mulches
	June 14	June 14	Winter squash transplanting
	July 18 and 25; August 8, 20 and 31st; September 7	July 16, 23, and 28; August 6, 13, 20, and 27	Insect counts
	July 17 and 25; August 8 and 20	July 3, 12, and 23; August 7 and 28	Weed counts
	July 17; August 8 and 20	July 3, 12, and 23; August 7 and 28	Timed weed management
	September 13	September 3	Harvest

ha⁻¹ on September 25, 2017 and September 27, 2018, 2–3 weeks following the termination of a 3-year alfalfa stand with a Brillion Super Soil Builder Disk Chisel (Brillion Iron Works, Brillion, WI). The following spring, cultivated and straw mulched treatments were terminated when the cereal rye reached 0.25 m in height. Planting rows were strip tilled on 2.74 m centers within roller-crimped treatment plots using 900DRT Husqvarna walk-behind rototiller (Husqvarna Group, Stockholm, Sweden) to a 1.22 m width. In all treatment plots containing ground straw or cultivation, the cereal rye cover crop was mowed using a rotary mower followed by tillage using a Case IH JX65 tractor with 65 horsepower (Case IH, Racine, WI) with a PTO driven Land Pride RTA3576 tiller with a 1.83 m working width (Land Pride, Salinas, KS). One tillage event was adequate to terminate the rye in 2018, but a second tilling was required in 2019. Rye biomass was measured at anthesis immediately prior to crimping by clipping above ground growth in two 0.25 m² sections, immediately adjacent to each rye plot but outside of the study area, so as not to affect weed pressure within plot. Biomass samples were then placed in a heated air dryer (54°C) at WMARS for 14 days and weighed. Remaining cereal rye within the rye aisle treatments was terminated by roller-crimping at anthesis, with the 4.57 m roller-crimper (I&J Manufacturing, Gap, PA).

Fertilizer was applied by hand within planting strips according to University of Wisconsin-Extension recommendations (Laboski and Peters, 2019) based on soil test results, including 134.5 kg ha⁻¹ of N, followed by an additional shallow pass with the rototiller to incorporate fertilizer. Drip irrigation, plastic and straw mulches were applied by hand following final rye termination. Three-week old “Honey Bear F1” acorn squash (*Cucurbita*

pepo) transplants grown in 50 cell trays were hand transplanted at 0.61 m in-row and 2.74 m between-row spacing 1 week after crimping, in both years. Drip irrigation placed under mulch was applied as needed throughout the season.

In both rows and aisles, weeds were categorized as broadleaf or grass weeds and counted within two randomly placed 0.25 m² quadrats within 24 h prior to timed manual weeding ($n = 18$ per treatment at each date). Straw and plastic mulch treatments were weeded by hand, and cultivated treatments were managed with stirrup hoes supplemented by additional hand weeding close to plants. Total weeding time (for a single person) required for weed management after the planting of the cash crops was recorded separately for each row and aisle treatment at each weeding event ($n = 9$ per treatment at each date). Cucumber beetle, squash bug egg clusters, and adult squash bugs per plant were counted as close to a weekly basis as possible ($n = 90$ per treatment at each date). Squash was harvested at maturity, assessed visually by the condition of fruit peduncles and plant senescence in combination with projected days to maturity. In each plot, the final plant count was recorded, and all mature squash of marketable size were harvested and sorted as marketable or non-marketable as determined by visible evidence of rot, insect damage, surface blemishes, or being misshapen. Immature fruit (as assessed by very small size and green peduncles) were not counted.

Data analysis

Data were analyzed in R (Rapp GUI 1.73 (7892 Catalina build), S. Urbanek & H.-J. Bibiko, © R Foundation for Statistical

TABLE 2 Weather data collected at UW-Madison Arboretum Weather Station, 2018 and 2019.

Time period	Total precipitation in cm (deviation from 40 yr average)	Average daily temperature in °C (deviation from 40 yr average)	GDDU 50 (deviation from 40 yr average)
October 2017 to February 2018	27.89 (+2.02)	−0.7 (+0.49)	182 (+77)
March to May 2018	33.07 (+7.71)	7.41 (−0.13)	463 (+128)
June to Sept 2018	86.11 (+41.28)	20.57 (+0.95)	2,286 (+238)
October 2018 to February 2019	37.24 (+11.37)	−1.13 (+0.06)	86 (−19)
March to May 2019	24.05 (−3.53)	7.45 (−0.09)	259 (−76)
June to Sept 2019	58.90 (+14.07)	20.28 (+0.66)	2,164 (+116)

Computing, 2020). ANOVAs were done using the `lme()` function in the “nlme” package (Pinheiro and Bates, 2022) using the following model:

$$Y_{ijkl} = \mu + A_i + B_{j(i)} + WP_k + \delta_{k(ji)} + SP_l + (AWP)_{ik} + (ASP)_{il} + (AWPSP)_{ikl} + \epsilon_{ijkl}$$

where Y_{ijkl} is the observation for the i th year, j th block, k th row mulch (whole plot) treatment, and l th aisle mulch (subplot) treatment, A_i is the fixed effect of the i th year ($i = 2018, 2019$), $B_{j(i)}$ is the random effect of the j th block nested within the i th year ($j = 1, 2, 3$), WP_k is the fixed effect of the k th whole plot row mulch treatment ($k = \text{cultivated, straw, plastic}$), $\delta_{k(ji)}$ is the random effect of the whole plot error term nested within the j th block within the i th year, SP_l is the fixed effect of the l th subplot aisle mulch treatment ($l = \text{cultivated, straw, rye}$), $(AWP)_{ik}$ is the effect of the interaction between the i th year and k th aisle mulch, $(ASP)_{il}$ is the effect of the interaction between the i th year and l th row mulch, $(AWPSP)_{ikl}$ is the effect of the interaction between the i th year and k th aisle mulch and l th row mulch, and ϵ_{ijkl} is the residual error associated with the observation for the i th year, j th block, k th row mulch (whole plot) treatment, and l th aisle mulch (subplot) treatment.

Pest data, weed management time, weed counts, and survival data were analyzed following the same procedure. However, pest counts and weed management time were transformed to cumulative counts, with only the final cumulative count analyzed. Weed counts and weed management time were analyzed with either the whole plot or subplot terms as appropriate, not both, and thus did not include the whole plot error term or associated interactions, so in-row weeding data was only associated with row mulch effects, and aisle weeding data was only associated with aisle mulch treatments. Pest and weed counts also included an additional subsampling error term $\gamma_{m(k(ji))}$ which was the random effect of the m th subsample ($m = 1 \dots 10$ where 10 is the number of plants per plot checked for pests, or where $m = 1, 2$ subsamples for weed counts).

Normality and equality of variances were checked visually with standardized residuals vs. fitted value plots and normal QQ plots, respectively (R Core Team, 2021). Right skewed count data for individual models (i.e., an entire given variable for a single model) were transformed with $\log(x + 1)$ when necessary to improve assumptions of normality and equality of variances. Pest count data could not be fully transformed to meet assumptions, but due to relative robustness of the F-test to deviations from normality and equal variances F-tests were performed anyway. Left skewed plant survival data was transformed with an $\arcsin(\sqrt{x})$ transformation. When ANOVA F-tests were significant, Tukey's Multiple Comparison Procedure was used to compare treatment means and develop significance groupings using the `emmeans()` function in the “emmeans” package, which is also how estimated marginal means for tables were obtained (Lenth, 2022). When two-way interactions between main effects were found, pairwise comparisons for the simple main effect were made for each level of the other factor, again using the `emmeans()` function with a Tukey adjustment. All figures are shown with non-transformed data though significance groupings are based on transformed data when applicable.

Results and discussion

Weather

Winter and spring precipitation leading into the 2018 season was slightly greater than average, with close to average temperatures and the accumulation of more growing degree day units (GDDU) than normal (Table 2). In contrast, winter conditions prior to the 2019 production season were colder and wetter than average, with a cooler and drier than average spring. Both 2018 and 2019 saw greater summer rainfall than average, with a single rain event in late August of 2018 releasing over 25 cm of rain within 24 h at the study site. Weed data was ended after that extreme rainfall event (MRCC, 2021).

Yield and plant survival

While the rye treatment yielded equivalent total fruit m^{-1} to the cultivated treatment, rye produced lower yields with respect to marketable fruit m^{-1} and a higher proportion and count of unmarketable fruit than cultivated aisles, regardless of row mulch (Table 3, Figure 1). The amount of marketable fruit plant^{-1} produced by cultivated aisles was similar to straw-mulched aisles; however, yield in terms of total fruit plant^{-1} was lower. Across aisle mulch treatments, plastic rows produced fewer total fruit m^{-1} than rows mulched with straw or cultivated rows, likely due to the low survival rate observed in plastic rows (Figure 2). With a lower number of unmarketable fruit, plastic rows produced yields of marketable fruit comparable to that of the rows mulched with straw despite the reduced number of total fruit, although the trend was toward lower marketable yields. Treatments utilizing straw produced greater total fruit yield in rows on a m^{-1} basis, and greater yields in both rows and aisles on a plant^{-1} basis but did not result in better marketable fruit yields due to producing more unmarketable fruit than cultivated treatments.

A significant row mulch \times aisle mulch interaction was observed for marketable fruit m^{-1} (Figure 3). Where rows were cultivated, yield was similar regardless of the combination with straw, rye or cultivation in the aisle. Similarly, whenever aisles were cultivated, equivalent yields were observed regardless of row mulch treatment. The use of straw mulch within the row resulted in higher yields when coupled with cultivated aisles as compared to rye aisles (Figure 3A). Within rows with the plastic mulch treatment, higher yields were observed for plots with cultivated aisles as compared to straw or rye in the aisle (Figure 3A). No significant differences were observed for row mulch treatments utilizing cultivated or rye aisles. However, within straw-mulched treatments, the marketable fruit yield utilizing cultivated rows was double that of treatments utilizing plastic rows (Figure 3B).

Differences in marketability were driven by a significant year \times aisle mulch interaction, with clearly higher proportions of unmarketable fruits for both straw mulched aisles and crimped rye in 2018, the year the field flooded prior to harvest, and no significant differences in 2019 (Figure 1; Supplementary Tables 1, 2). Similarly, a year \times aisle mulch interaction was observed for marketable fruit plant^{-1} (Supplementary Figure 2). In both years, rot was the most common cause of fruit being deemed unmarketable, followed by rodent damage.

While the primary yield declines in this study appeared to be caused by the 2018 rain event and subsequent fruit rot, the crimped rye treatments also produced fewer total fruit plant^{-1} than treatments with straw mulch in the aisle, suggesting there may be other mechanisms impacting yield. One such mechanism may be N immobilization with rye cover crops (e.g.,

Delate et al., 2008; Van Den Bossche et al., 2009; Chehade et al., 2019) or reduced N mineralization thanks to lower temperatures (as suggested by Leavitt et al., 2011).

Although previous research suggests that supplementary fertilization could improve vegetable yields in reduced tillage systems, studies largely focus on either fertigation or sidedressing (e.g., Delate et al., 2008; Schellenberg et al., 2009; Jokela and Nair, 2016). Future studies assessing the benefits of supplementary fertilizers should compare approaches, timing, and rates within a single study. Choosing cover crop species or mixes that include the benefit of nitrogen fixation from legumes and optimizing management to maximize nitrogen cycling may also be an option for reducing the potential for yield declines (Ginakes and Grossman, 2021).

In general, a stronger effect from row mulch than aisle mulch on total yield m^{-1} was observed in our study. Cultivated rows had higher yields than mulched rows, again pointing to the sensitivity of these systems to environmental conditions. These system \times environment interactions indicate the need for further study of disease and pest dynamics within CCBRT systems as driven by different environmental conditions. Despite the potential for reduced yields, all treatments generally produced well relative to the advertised marketable yield plant^{-1} for the variety used (All American Selections, 2009).

CCBRT systems provide the notable benefit of resilience in the face of extreme rainfall events through protecting the soil and reducing erosion. However, while soil is protected under wet conditions, our study indicated that trade-offs may exist with respect to the system exacerbating disease pressure. While some research has investigated disease dynamics in CCBRT systems for cucurbits (e.g., Maglione et al., 2022), it is crucial that such research also simultaneously integrates the assessment other agronomic impacts such as yield quantity and weed management in order to form a more holistic picture of system performance.

Insect pest pressure

Striped cucumber beetle

Striped cucumber beetle counts were very low overall, especially in 2018, and the only clear effect was from year (Table 4; Supplementary Table 3).

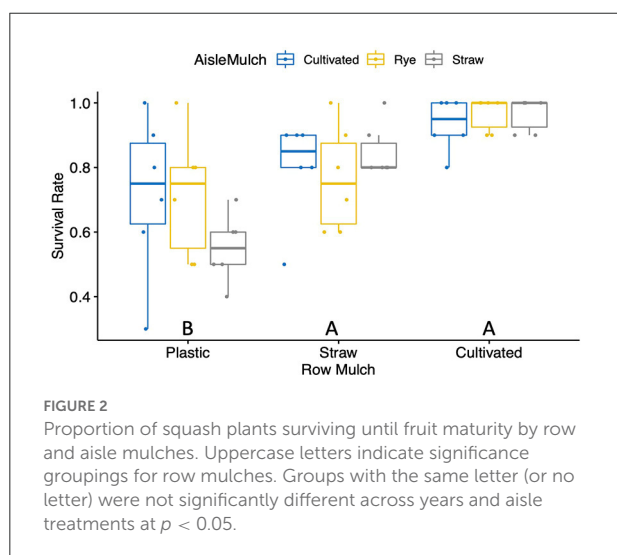
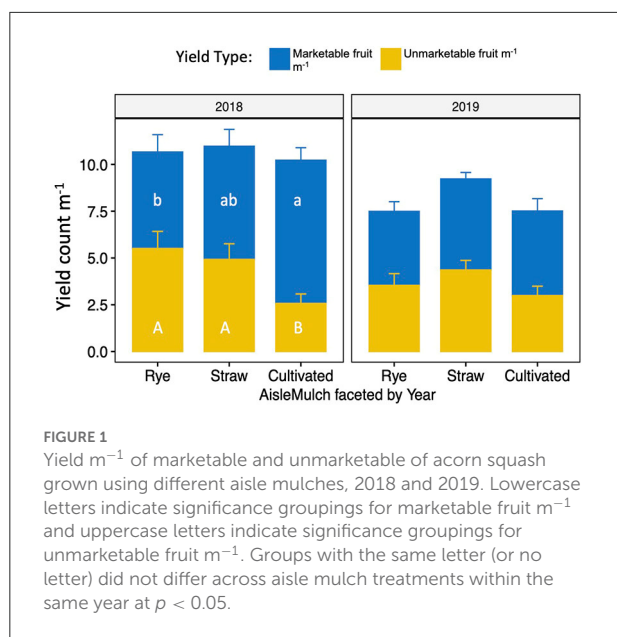
Squash bug adults

Significant aisle mulch \times year and row mulch \times year interactions were observed in explaining squash bug pressure due to lower counts in 2019, with overall effects driven primarily by 2018 (Figure 4). The simple main effect of year was also significant due to the low counts in 2019. Both aisle and row mulches were significant in 2018. Rows with straw and plastic mulch had higher numbers than cultivated rows across

TABLE 3 Yield, quality, and survival data for organic squash managed with CCBRT by mulch treatment.

Aisle mulch	Row mulch	Total fruit m^{-1}	Marketable fruit m^{-1}	Proportion unmarketable fruit	Unmarketable fruit m^{-1}	Total fruit plant $^{-1}$	Marketable fruit plant $^{-1}$	Proportion plant survival
Cultivated	Straw	10.29	6.48	0.36	3.66	8.92	5.11	0.80
	Black plastic	8.50	5.85	0.32	2.11	7.48	4.58	0.72
	Cultivated	9.25	5.93	0.32	2.57	6.82	3.93	0.93
Simple main effect across row mulch	Cultivated aisle average	8.87	6.09 A	0.33 B	2.78 B	6.70 B	4.54 A	0.82
Roller-crimped rye	Straw	10.59	3.83	0.58	5.71	9.17	3.01	0.77
	Black plastic	6.71	3.91	0.45	3.01	6.20	3.05	0.72
	Cultivated	11.78	5.93	0.44	4.84	8.14	3.78	0.97
Simple main effect across row mulch	Rye aisle average	9.08	4.56 B	0.46 A	4.52 A	6.77 B	3.28 B	0.82
Straw	Straw	11.43	5.30 ab	0.53	6.12	9.58	3.84	0.85
	Black plastic	6.76	3.77 b	0.47	3.47	8.58	4.34	0.55
	Cultivated	11.73	7.30 a	0.38	4.32	7.87	4.57	0.97
Simple main effect across row mulch	Straw aisle average	10.10	5.46 AB	0.49 A	4.64 A	7.93 A	4.25 A	0.79
Row mulch Simple main effect across aisle mulch	Straw	10.37 A	5.20	0.49	5.17 A	7.96 A	3.98	0.81 B
	Black plastic	7.37 B	4.51	0.41	2.86 B	6.85 AB	3.99	0.66 B
	Cultivated	10.30 A	6.39	0.37	3.91 AB	6.58 B	4.09	0.96 A
Treatment effects	Row mulch	F = 9.28, $p < 0.01$	F = 2.54, ns	F = 1.79, ns	F = 9.70, $p < 0.01$	F = 5.04, $p < 0.05$	F = 0.03, ns	F = 11.54, $p < 0.01$
	Aisle mulch	F = 2.10, ns	F = 10.54, $p < 0.001$	F = 7.71, $p < 0.01$	F = 8.65, $p < 0.01$	F = 5.62, $p < 0.05$	F = 10.54, $p < 0.001$	F = 0.19, ns
	Row \times aisle	F = 2.03, ns	F = 2.98, $p < 0.05$	F = 0.53, ns	F = 0.17, ns	F = 1.56, ns	F = 2.69, $p < 0.1$	F = 1.38, ns
	Year	F = 12.59, $p < 0.05$	F = 5.22, $p < 0.1$	F = 0.09, ns	F = 0.21, ns	F = 13.97, $p < 0.05$	F = 2.62, ns	F = 0.15, ns
	Year \times aisle	F = 0.64, ns	F = 2.87, $p < 0.1$	F = 3.83, $p < 0.05$	F = 2.61, $p < 0.1$	F = 0.33, ns	F = 6.47, $p < 0.01$	F = 0.26, ns
	Year \times row	F = 0.89, ns	F = 1.58, ns	F = 1.67, ns	F = 0.91, ns	F = 1.32, ns	F = 2.81, ns	F = 0.24, ns
	Year \times row \times aisle	F = 0.46, ns	F = 0.14, ns	F = 0.09, ns	F = 0.32, ns	F = 0.27, ns	F = 0.34 ns	F = 0.71, ns

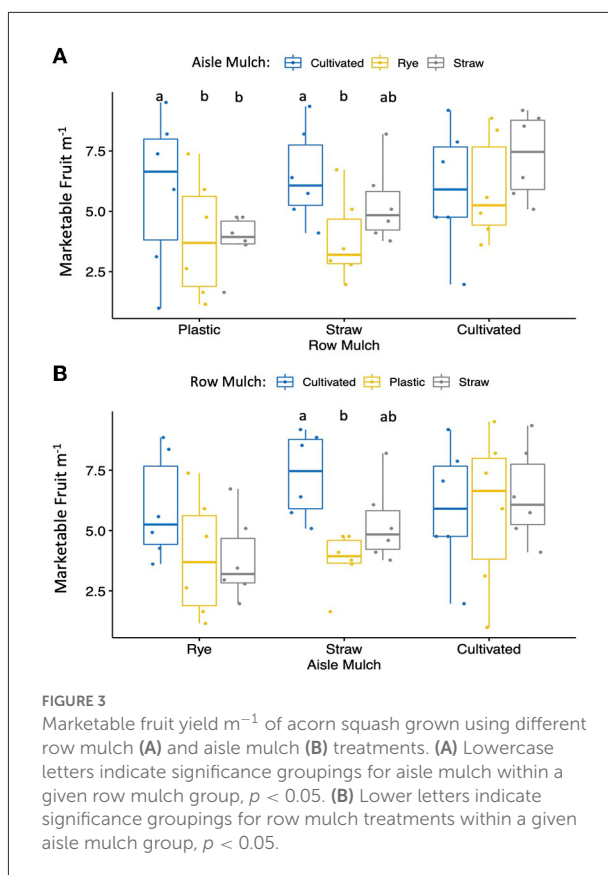
Estimated marginal means in 2018 and 2019 averaged across the level of block and year are shown. Untransformed data is shown in the table but significance groupings according to a p -value adjustment for pairwise comparisons following the Tukey method are based on transformed data where applicable. Columns with the same letter (or no letter) were not significantly different across mulch treatments within the same year at $p < 0.05$. Lowercase letters indicate significance groupings for the whole plot effect of row mulch treatments within one aisle mulch treatment, and uppercase letters indicate significance groupings for the whole plot effect of row mulch across aisle mulch treatments, or the sub plot effect of aisle mulch across row mulch treatments. Significance groupings for the simple main effects of aisle mulch within row mulch treatments are not shown. Cultivated aisles yielded significantly higher than rye aisles when paired with hay mulch in rows and cultivated also yielded higher than hay aisle mulch when paired with plastic in rows.



aisle mulch levels, while cultivated aisles also resulted in lower numbers than the mulched treatments of ground straw or rye aisles across row mulch levels. Overall, cultivated treatments resulted in lower populations compared to other mulches, and across all aisle mulch treatments rows with plastic mulch consistently resulted in the highest counts.

Squash bug egg clusters

A significant aisle mulch \times year interaction explained cumulative squash bug egg cluster counts per plant (Figure 5). The simple main effects of row mulch and aisle mulch were also significant across years. Similar to results for squash bug adults, cultivated treatments had lower egg cluster counts. For



row mulches, ground straw performed similarly to cultivation, with lower counts than plastic. In aisles, rye resulted in higher egg cluster counts as compared to cultivation.

Results regarding both squash bug adults and their egg clusters are consistent with observations reported by Doughty et al. (2016) who suggested that squash bugs will often be found in the planting holes of plastic mulches, a behavior that could make it difficult for a grower to effectively apply pesticide when needed. While the effect of row mulches was clear, the results of our 2-year study showed inconsistent effects of aisle mulching (either as crimped rye or ground straw) on squash bugs, with 2018 demonstrating greater squash bug pressure with aisle mulching, and 2019 showing no clear effect, when counts were lower across treatments. Habitat provided by mulches may benefit cash crops by promoting within-field natural enemy activity and biological control (Tonhasca and Byrne, 1994; Langellotto and Denno, 2004; Bryant et al., 2013; Hinds and Cerruti, 2013), but our results indicated that the habitat could also benefit pests. Cranshaw et al. (2001) also showed increased damage to pumpkin by squash bugs when using straw or plastic mulches.

The strongest effect was seen from plastic mulch in rows, so applying hay mulch within the tilled planting strip may be a better option than black plastic for growers adopting CCBRT

TABLE 4 Average (2018 and 2019) cumulative cucumber beetle, squash bugs and egg cluster counts by aisle mulch treatment.

Aisle Mulch	Row mulch	Cumulative cucumber beetles per plant	Cumulative squash bugs per plant	Cumulative egg clusters per plant
Cultivated control	Ground straw	0.67	1.18	0.22
	Black plastic	0.53	3.33	1.07
	Cultivated	0.80	0.37	0.28
	Cultivated aisle average	0.67	1.63	0.52 B
Roller-crimped rye	Ground straw	1.03	2.50	0.62
	Black plastic	0.77	3.58	1.62
	Cultivated	0.78	0.95	0.40
	Rye aisle average	0.86	2.34	0.88 A
Ground straw	Ground straw	102	2.25	0.52
	Black plastic	0.73	3.43	1.32
	Cultivated	0.70	0.87	0.43
	Straw aisle average	0.82	2.18	0.76 AB
Row type	Ground straw rows	0.91	1.98 B	0.47 B
	Black plastic rows	0.68	3.45 A	1.33 A
	Cultivated rows	0.76	0.73 B	0.35 B
Treatment effects	Row mulch	F = 1.93, ns	F = 16.01, $p < 0.01$	F = 19.48, $p < 0.001$
	Aisle mulch	F = 1.34, ns	F = 2.06, ns	F = 4.85 $p < 0.05$
	Year	F = 264.19, $p < 0.0001$	F = 60.39, $p < 0.01$	F = 1.86, ns
	Row \times aisle	F = 1.03, ns	F = 0.34, ns	F = 0.25, ns
	Aisle \times year	F = 2.58, $p < 0.1$	F = 6.72, $p < 0.001$	F = 12.39, $p < 0.001$
	Row \times year	F = 0.61, ns	F = 5.10, $p < 0.05$	F = 0.87, ns
	Aisle \times row \times year	F = 0.61, ns	F = 0.41, ns	F = 1.78, ns

Untransformed data is shown, but significance groupings are based on transformed data where applicable. Columns with the same letter (or no letter) were not significantly different across mulch treatments within the same year at $p < 0.05$. Uppercase letters indicate significance groupings for the simple main effect of aisle mulch across row mulch treatments or row mulch across aisle mulch treatments.

practices for cucurbit production with the potential for high pest pressure. In general, pest abundance on the squash was relatively low in our experimental field during the study period, which may have contributed to the variable response between years.

Weed populations and management time

Aisle weed counts and management time

Cultivated aisles resulted in the highest total, broadleaf, and grass weed counts and required the greatest weed management time inputs (Table 5). Rye aisles resulted in fewer weeds and required less weed management time as compared to cultivated treatments but had significantly more weeds and took longer to manage than straw mulch (Figure 6). There was a significant aisle mulch \times year interaction for all weed related data points due primarily to changes in significance level in pairwise comparisons between aisle mulches because of generally higher weed counts in 2019 than 2018, except for higher broadleaf weed counts in 2019 than 2018, except for higher broadleaf weed counts in 2018 (Supplementary Table 4). There were no significant crossover interactions, except for rye and straw aisle

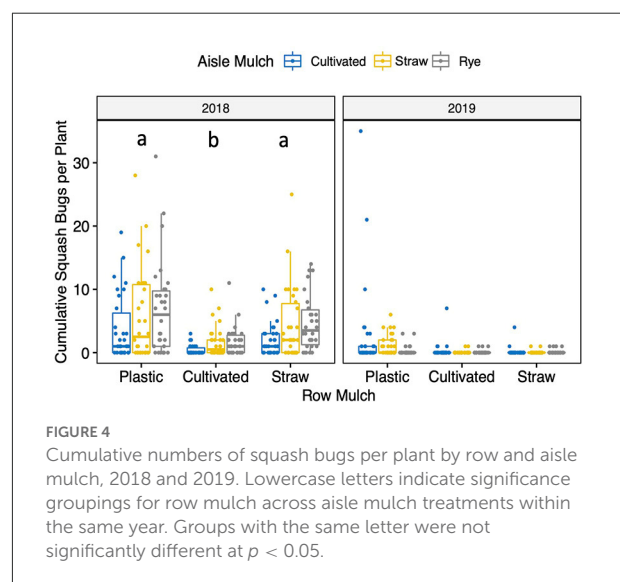


FIGURE 4 Cumulative numbers of squash bugs per plant by row and aisle mulch, 2018 and 2019. Lowercase letters indicate significance groupings for row mulch across aisle mulch treatments within the same year. Groups with the same letter were not significantly different at $p < 0.05$.

present in 2018 and more grass weeds in 2019. Overall, year was significant for both broadleaf and grass weed counts because of the higher counts in 2018 and 2019, respectively.

The effectiveness of the cereal rye treatment with respect to weed suppression was likely influenced by heavy mulch residue created by the rye cover crop. One key factor affecting successful weed suppression of CCBRT systems is the cover crop biomass at termination; cover crop biomass on the soil surface should reach 8–9 Mg ha⁻¹ to obtain satisfactory weed suppression without additional weed control methods, which can include time-consuming and labor-intensive hand-weeding to rescue the vegetable crop from excessive yield loss (Smith et al., 2011;

Mirsky et al., 2012; Bietila et al., 2016). In the 2 years of the study, the biomass of cover crop produced reached or nearly reached the threshold needed for adequate weed suppression (mean biomass of 11,756 kg ha⁻¹ in 2018 and 7,866 kg ha⁻¹ in 2019). Lower biomass in 2019 may have contributed to that year's higher weed counts.

While the use of CCBRT techniques in this study did result in fewer weeds as compared to management with cultivation, a small number of weeds were still present in the field throughout the production season. In organic production, crop canopy cover is another important tool for continued weed suppression (Hoad et al., 2012). Variety trials conducted within CCBRT management systems could further optimize the system toward complete elimination of weed seed production; for example, the cultivar in this trial was a semi-bush type, and vining cucurbit cultivars providing greater ground cover which could further contribute to weed suppression, especially during years where cover crop biomass might be lower than the ideal range.

The weed suppression provided by the CCBRT approach translated into fewer weeding hours required for crop management as compared to cultivation. Despite the decreased yields observed in 2018 using the metric of marketable fruit, this approach could still be considered advantageous to farmers, as labor needs across the entire farm during the peak production times of mid-summer can be limiting, and the opportunity costs of not having the ability to use that labor elsewhere on the farm (e.g., harvesting crops or attending a market), as well as the actual costs of the labor, may justify the tolerance of the lower yields.

A notable benefit of CCBRT systems is the potential to reduce erosion during extreme rainfall events, such as the

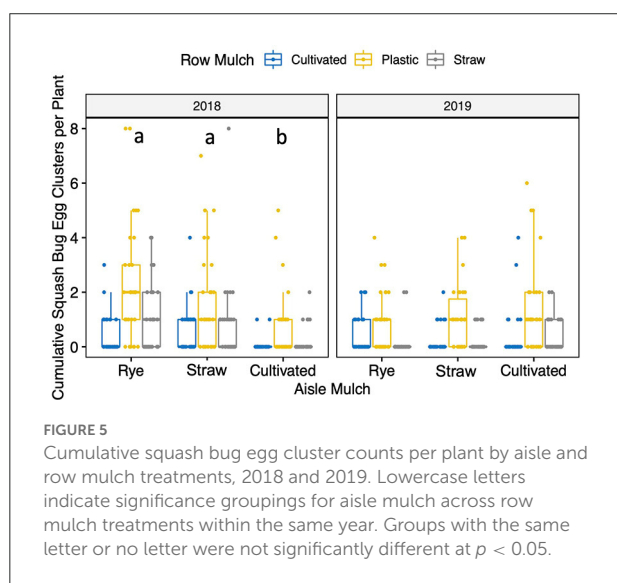
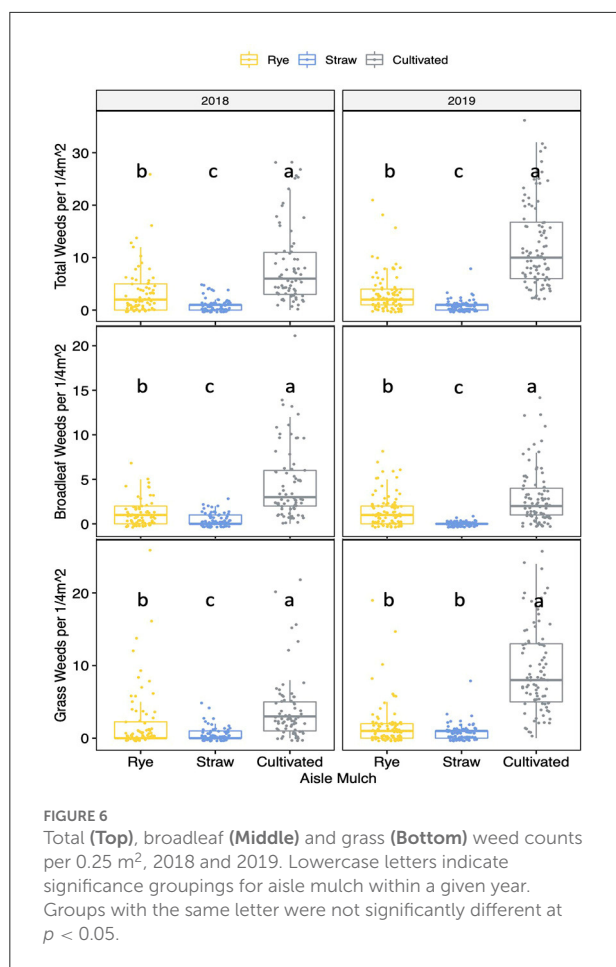


FIGURE 5
Cumulative squash bug egg cluster counts per plant by aisle and row mulch treatments, 2018 and 2019. Lowercase letters indicate significance groupings for aisle mulch across row mulch treatments within the same year. Groups with the same letter or no letter were not significantly different at $p < 0.05$.

TABLE 5 Weed counts and management time in 2018 and 2019 relative to row and aisle mulch treatments.

Mulch type	Weeding time (h/ha)	Total weed ct per $\frac{1}{4}$ m ²	Broadleaf ct per $\frac{1}{4}$ m ²	Grass ct per $\frac{1}{4}$ m ²
Cultivated aisle	841 a	10.43 a	3.73 a	6.69 a
Rye aisle	523 b	3.28 b	1.31 b	1.97 b
Straw	206 c	0.86 c	0.23 c	0.64 c
Aisle treatment effects				
Aisle mulch	F = 95.39, $p < 0.0001$	F = 155.12, $p < 0.0001$	F = 133.05, $p < 0.0001$	F = 127.36, $p < 0.0001$
Year	F = 70.31, $p < 0.01$	F = 1.14, ns	F = 5.54, $p < 0.1$	F = 7.71, $p < 0.05$
Aisle \times year	F = 15.11, $p < 0.001$	F = 3.73, $p < 0.05$	F = 8.01, $p < 0.01$	F = 13.86, $p < 0.0001$
Straw row	119 b	0.28 b	0.12 b	0.16 b
Plastic row	140 b	0.72 b	0.20 b	0.52 b
Cultivated row	704 a	8.57 a	3.64 a	4.94 a
Row treatment effects				
Row mulch	F = 108.53, $p < 0.0001$	F = 370.90, $p < 0.0001$	F = 135.55, $p < 0.0001$	F = 192.29, $p < 0.0001$
Year	ns	ns	F = 11.55, $p < 0.05$	F = 12.03, $p < 0.05$
Row \times year	ns	F = 6.03, $p < 0.01$	F = 20.52, $p < 0.0001$	F = 14.72, $p < 0.0001$

Untransformed data is shown in the table but significance groupings are based on transformed data where applicable. Columns with the same letter were not significantly different across mulch treatments and years at $p < 0.05$ in either aisles or rows.



one in late August 2018 at this study site, but if the CCBRT systems are at risk of increasing disease pressure after such events then growers need more information to be able to adequately assess tradeoffs. Previous research suggests that supplementary fertilization could improve vegetable yields in reduced tillage systems, but studies largely focus on either fertigation or sidedressing, rather than comparing approaches and rates within a single study (e.g., Schellenberg et al., 2009; Jokela and Nair, 2016). In addition, the large Rodale-style chevron blade roller-crimpers, such as the one used in this study, rely on weight to effectively crimp, and thus require relatively large tractors with adequate horsepower to operate; testing the efficacy of smaller crimper types, such as those that mount on a walk-behind tractor or do not rely solely on weight as a crimping mechanism (Kornecki and Reyes, 2020) would further elucidate the adaptability of the system to small scale vegetable production.

Row weed counts and management time

Overall, weed counts and management time were higher in cultivated rows than in those mulched with either straw or

plastic. Similar to aisle weed counts, a significant effect of year was observed with respect to broadleaf and grass weed counts due to higher counts in 2018 and 2019, respectively, and a significant row mulch \times year interaction for cultivated rows was observed due to those higher counts. A crossover interaction for row weed counts was also observed; straw and plastic mulches were equivalent for total and grass weed counts in 2018, but plastic had higher counts than straw in 2019. This interaction was likely due to the overall increased prevalence of grass weeds in 2019, exacerbated by the difficulty of managing weeds at the shoulders of the beds with plastic mulch where exposed soil was present, whereas the in-row straw mulch extended to the rye or straw mulches in aisles.

Mulching with straw resulted in adequate weed suppression and increased the total fruit yield, while avoiding the problems of plastic mulch with respect to increased squash bug pest pressure. Thus, applying straw mulch within the tilled planting strip may be a better option than black plastic (which also resulted in higher pest pressure) for growers adopting CCBRT practices for cucurbit production. Anecdotally, the straw mulch was also easier to apply in combination with rye than it was to dig the plastic mulch in by hand since conventional mulch-layers could not deal with the heavy residue at the edge of the tilled strip.

Conclusions

The primary goal of this study was to evaluate the impact of strip tillage management with CCBRT practices for organic squash production. The data derived from this work demonstrated that the use of CCBRT practices with strip tillage techniques in organic cucurbit systems has the potential to produce overall yields comparable to that of standard organic cucurbit production practices using cultivation, with total fruit m⁻¹ equivalent between approaches in both years and marketable fruit comparable in 2019. This supports the suggestions of previous research that strip tillage in CCBRT systems can be a viable alternative to full tillage systems (Forcella et al., 2015; Tillman et al., 2015; Jokela and Nair, 2016). However, reduced marketable fruits plant⁻¹ and m⁻¹ were observed in 2018 as a result of increased rates of unmarketable fruit in that year, likely influenced by the record-breaking rain event that released 25 cm of precipitation in <24 h two weeks prior to harvest.

Overall, rolled-crimped management strategies for organic cucurbit management were demonstrated to be a valuable tool for organic vegetable farmers in the upper Midwestern US. However, our research did highlight questions related to the interaction between specific management choices and environmental conditions and the resulting agronomic impacts; providing answers to these questions will reduce risk for growers and drive further adoption of this practice. Thus, future research

should focus on understanding the more nuanced management aspects of the system, including the identification of cultivars adapted to reduced tillage systems, supplementary fertilization methods that might result in more reliable yields, and longer term studies that explore disease and pest dynamics (such as the potential for cover crop species to provide alternate hosts for diseases, residue to increase fruit rot incidence by maintaining higher soil moisture, and predator populations and predation of common pests).

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors upon request, without undue reservation.

Author contributions

ES: conceptualization and funding application. ES and DB: research design. DB: data collection. DB, ES, and JD: data analysis and interpretation, writing manuscript, and editing manuscript. All authors contributed to the article and approved the submitted version.

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References

- All American Selections. (2009). *Squash Honey Bear F1*. Available online at: <https://all-americanselections.org/product/squash-honey-bear/> (accessed July 04, 2022).
- Baraibar, B., Hunter, M. C., Schipanski, M. E., Hamilton, A., and Mortensen, D. A. (2018). Weed suppression in cover crop monocultures and mixtures. *Weed Sci.* 66, 121–133. doi: 10.1017/wsc.2017.59
- Bezuidenhout, S. R., Reinhardt, C. F., and Whitwell, M. I. (2012). Cover crops of oats, strolling rye and three annual ryegrass cultivars influence maize and *Cyperus esculentus* growth. *Weed Res.* 52, 153–160. doi: 10.1111/j.1365-3180.2011.00900.x
- Bietila, E., Silva, E. M., Pfeiffer, A. C., and Colquhoun, J. B. (2016). Fall-sown cover crops as mulches for weed suppression in organic small-scale diversified vegetable production. *Renew. Agric. Food Syst.* 32, 349. doi: 10.1017/S1742170516000259
- Brainard, D. C., Peachey, R. E., Haramoto, E. R., Luna, J. M., and Rangarajan, A. (2013). Weed ecology and nonchemical management under strip-tillage: implications for northern US vegetable cropping systems. *Weed Technol.* 27, 18–230. doi: 10.1614/WT-D-12-00068.1
- Burst, J., Claupein, W., and Gerhards, R. (2014). Growth and weed suppression ability of common and new cover crops in Germany. *Crop Prot.* 63, 1–8. doi: 10.1016/j.cropro.2014.04.022
- Bryant, A., Brainard, D. C., Haramoto, E. R., and Szendrei, Z. (2013). Cover crop mulch and weed management influence arthropod communities in strip-tilled cabbage. *Environ. Entomol.* 42, 293–306. doi: 10.1603/EN12192
- Campiglia, E., Caporali, F., Radicetti, E., and Mancinelli, R. (2010). Hairy vetch (*Vicia villosa* Roth.) cover crop residue management for improving weed control and yield in no-tillage tomato (*Lycopersicon esculentum* Mill.) production. *Eur. J. Agron.* 33, 94–102. doi: 10.1016/j.eja.2010.04.001
- Chehade, L. A., Antichi, D., Martelloni, L., Frascioni, C., Sbrana, M., Mazzoncini, M., et al. (2019). Evaluation of the agronomic performance of organic processing tomato as affected by different cover crop residues management. *Agronomy* 9, 504. doi: 10.3390/agronomy9090504
- Clark, A. J., Decker, M., and Meisinger, J. J. (1994). Seeding rate and kill date effects on hairy vetch-cereal rye cover crop mixtures for corn production. *Agron. J.* 86, 1065–1070. doi: 10.2134/agronj1994.00021962008600060025x
- Cranshaw, W., Bartolo, M., and Schweissing, F. (2001). Control of squash bug (Hemiptera: Coreidae) injury: Management manipulations at the base of pumpkin. *Southwestern Entomol.* 26, 147–150.
- Creamer, N. G., Bennett, M. A., Stinner, B. R., and Cardina, J. (1996). A comparison of four processing tomato production systems differing in cover crop and chemical inputs. *J. Am. Soc. Hortic. Sci.* 121, 559–568. doi: 10.21273/JASHS.121.3.559

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

The Supplementary Material for this article can be found online at: <https://www.frontiersin.org/articles/10.3389/fsufs.2022.991463/full#supplementary-material>

- De Baets, S., Poesen, J., Meersmans, J., and Serlet, L. (2011). Cover crops and their erosion-reducing effects during concentrated flow erosion. *Catena* 85, 237–244. doi: 10.1016/j.catena.2011.01.009
- Deguchi, S., Uozumi, S., Touno, E., Kaneko, M. and Tawarayama, K. (2012). Arbuscular mycorrhizal colonization increases phosphorus uptake and growth of corn in a white clover living mulch system. *Soil Sci. Plant Nutr.* 58, 169–172. doi: 10.1080/00380768.2012.662697
- Delate, K., Cambardella, C., and McKern, A. (2008). Effects of organic fertilization and cover crops on an organic pepper system. *Horttechnology* 18, 15–226. doi: 10.21273/HORTTECH.18.2.215
- Delate, K., Cwach, D., and Chase, C. (2012). Organic no-tillage system effects on soybean, corn and irrigated tomato production and economic performance in Iowa, USA. *Renew. Agric. Food Syst.* 27, 49–59. doi: 10.1017/S1742170511000524
- Delate, K., Friedrich, H., and Lawson, V. (2003). Organic pepper production systems using compost and cover crops. *Biol. Agric. Horticult.* 21, 131–150. doi: 10.1080/01448765.2003.9755258
- Doughty, H. B., Wilson, J. M., Schultz, P. B., and Kuhar, T. P. (2016). Squash bug (Hemiptera: Coreidae): biology and management in cucurbitaceous crops. *J. Integrated Pest Manag.* 7:1–6. doi: 10.1093/jipm/pmv024
- Everts, K. L. (2002). Reduced fungicide applications and host resistance for managing three diseases in pumpkin grown on a no-till cover crop. *Plant Dis.* 86, 1134–1141. doi: 10.1094/PDIS.2002.86.10.1134
- Forcella, F., Eklund, J., and Peterson, D. (2015). Rolled-crimped winter rye cover effects on hand-weeding times and fruit yield and quality of cucurbits. *Int. J. Veg. Sci.* 21, 386–396.
- Ginakes, P., and Grossman, J. M. (2021). Extending cover crop benefits with zone till management in northern organic summer squash production. *Agron.* 11, 983.
- Hiltbrunner, J., Jeanneret, P., Liedgens, M., Stamp, P., and Streit, B. (2007). Response of weed communities to legume living mulches in winter wheat. *J. Agron. Crop Sci.* 193, 93–102. doi: 10.1111/j.1439-037X.2007.00250.x
- Hinds, J., and Cerruti, R. R. H. (2013). Population dynamics of arthropods in a sunn-hemp zucchini interplanting system. *Crop Prot.* 53, 6–12. doi: 10.1016/j.cropro.2013.06.003
- Hoad, S. P., Bertholdsson, N.-Ø., Neuhoof, D., and Köpke, U. (2012). “Approaches to breed for improved weed suppression in organically grown cereals,” in *Organic Crop Breeding*, 61–76.
- Jabbour, R., Pisani-Gareau, T., Smith, R. G., Mullen, C., and Barbercheck, M. (2016). Cover crop and tillage intensities alter ground-dwelling arthropod communities during the transition to organic production. *Renew. Agric. Food Syst.* 31, 361–374. doi: 10.1017/S1742170515000290
- Jenkins, D., and Ory, J. (2016). *2016 National Organic Research Agenda*. Organic Farming Research Foundation, Santa Cruz, CA.
- Jokela, D., and Nair, A. (2016). Effects of reduced tillage and fertilizer application method on plant growth, yield, and soil health in organic bell pepper production. *Soil Tillage Res.* 163, 243–254. doi: 10.1016/j.still.2016.06.010
- Kaspar, T. C., Singer, J. W., Hatfield, J. L., and Sauer, T. J. (2011). *The Use of Cover Crops to Manage Soil*, Vol. 409. Madison, WI: American Society of Agronomy and Soil Science Society of America.
- Kornecki, T. S., and Reyes, M. R. (2020). Equipment development for small and urban conservation farming systems. *Agriculture* 10, 595. doi: 10.3390/agriculture10120595
- Laboski, C. A. M., and Peters, J. B. (2019). Nutrient Application Guidelines for Field, Vegetable, and Fruit Crops in WI. A2809. University of Wisconsin Cooperative Extension.
- Langelotto, G. A., and Denno, R. F. (2004). Responses of invertebrate natural enemies to complex-structured habitats: a meta-analytical synthesis. *Oecologia* 139, 1–10. doi: 10.1007/s00442-004-1497-3
- Leavitt, M. J., Sheaffer, C. C., Wyse, D. L., and Allan, D. L. (2011). Rolled winter rye and hairy vetch cover crops lower weed density but reduce vegetable yields in no-tillage organic production. *HortScience* 46, 387–395. doi: 10.21273/HORTSCI.46.3.387
- Lenth, R. V. (2022). *emmeans: Estimated Marginal Means, aka Least-Squares Means*. R package version 1.7.2. Available online at: <https://CRAN.R-project.org/package=emmeans> (accessed July 04, 2022).
- Licht, M. A., and Al-Kaisi, M. (2005). Strip-tillage effect on seedbed soil temperature and other soil physical properties. *Soil Tillage Res.* 80, 233–249. doi: 10.1016/j.still.2004.03.017
- Liebman, M., Gallandt, E. R., and Jackson, L. E. (1997). Many little hammers: ecological management of crop-weed interactions. *Ecol. Agric.* 1, 291–343. doi: 10.1016/B978-012378260-1/50010-5
- Lounsbury, N. P., Warren, N. D., Wolfe, S. D., and Smith, R. G. (2020). Investigating tarps to facilitate organic no-till cabbage production with high-residue cover crops. *Renew. Agric. Food Syst.* 35, 227–233.
- Lounsbury, N. P., and Weil, R. R. (2015). No-till seeded spinach after winterkilled cover crops in an organic production system. *Renew. Agric. Food Syst.* 30, 473–485. doi: 10.1017/S1742170514000301
- Lowry, C. J., and Brainard, D. C. (2019). Organic farmer perceptions of reduced tillage: A Michigan farmer survey. *Renew. Agric. Food Syst.* 34, 103–115.
- Luna, J. M., Mitchell, J. P., and Shrestha, A. (2012). Conservation tillage for organic agriculture: evolution toward hybrid systems in the western USA. *Renew. Agric. Food Syst.* 27, 21–30. doi: 10.1017/S1742170511000494
- Luo, Z., Wang, E., and Sun, O. (2010). Can no-tillage stimulate carbon sequestration in agricultural soils? A meta-analysis of paired experiments. *Agric. Ecosyst. Environ.* 2010, 224–231. doi: 10.1016/j.agee.2010.08.006
- Magdoff, F., and Van Es, H. (2000). *Building Soils for Better Crops* (No. 631.584/M188b). Beltsville: Sustainable Agriculture Network.
- Maglione, R., Ciotola, M., Cadieux, M., Toussaint, V., Laforest, M., and Kembel, S. W. (2022). Winter rye cover cropping changes squash (*Cucurbita pepo*) phyllosphere microbiota and reduces *Pseudomonas syringae* symptoms. *Phytobiomes J.* 6, 3–12. doi: 10.1094/PBIOMES-04-21-0029-R
- Mirsky, S. B., Ryan, M. R., Curran, W. S., Teasdale, J. R., Maul, J., Spargo, J. T., et al. (2012). Conservation tillage issues: cover crop-based organic rotational no-till grain production in the mid-Atlantic region, USA. *Renew. Agric. Food Syst.* 27, 31–40. doi: 10.1017/S1742170511000457
- Mochizuki, M. J., Rangarajan, A., Bellinder, R. R., Björkman, T., and van Es, H. M. (2007). Overcoming compaction limitations on cabbage growth and yield in the transition to reduced tillage. *HortScience* 42, 1690–1694. doi: 10.21273/HORTSCI.42.7.1690
- Mochizuki, M. J., Rangarajan, A., Bellinder, R. R., van Es, H. M., and Björkman, T. (2008). Rye mulch management affects short-term indicators of soil quality in the transition to conservation tillage for cabbage. *HortScience* 43, 862–867. doi: 10.21273/HORTSCI.43.3.862
- Moynihan, M. (2010). *Status of Organic Agriculture in Minnesota*. Available online at: <https://www.leg.mn.gov/docs/2010/mandated/100851.pdf> (accessed July 04, 2022).
- MRCC (2021). (1978-2019) Daily precipitation and temperatures - UW-Madison Arboretum Station, Midwestern Regional Climate Center.
- National Organic Program (2000). Final Rule. *Federal Register*. 65, 80548–96.
- Pinheiro, J., and Bates, D. R. (2022). *nlme: Linear and Nonlinear Mixed Effects Models*. R package version 3.1-157. Available online at: <https://CRAN.R-project.org/package=nlme> (accessed July 4, 2022).
- R Core Team. (2021). *Rapp GUI 1.4 “Juliet Rose”* (df86b69e, 2021-05-24). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. Available online at: <https://www.R-project.org/> (accessed July 04, 2022)
- Reicosky, D. C., and Forcella, F. (1998). Cover crop and soil quality interactions in agroecosystems. *J. Soil Water Conserv.* 53, 224–229.
- Ryder, M. H., and Fares, A. (2008). Evaluating cover crops (Sudex, Sunn Hemp, Oats) for use as vegetative filters to control sediment and nutrient loading from agricultural runoff in a Hawaiian watershed. *J. Am. Water Resour. Assoc.* 44, 640–653. doi: 10.1111/j.1752-1688.2008.00189.x
- Salon, P. R. (2012). *Diverse cover Crop Mixes for Good Soil Health*. Corning, NY: USDA-NRCS.
- Sarrantonio, M. (1992). Opportunities and challenges for the inclusion of soil-improving crops in vegetable production systems. *HortScience*, 27, 754–758.
- Sarrantonio, M., and Gallandt, E. (2003). The role of cover crops in North American cropping systems. *J. Crop Prod.* 8, 53–74. doi: 10.1300/J144v08n01_04
- Schellenberg, D. L., Morse, R. D., and Welbaum, G. E. (2009). Organic broccoli production on transition soils: comparing cover crops, tillage and sidedress N. *Renew. Agric. Food Syst.* 2, 85–91. doi: 10.1017/S1742170508002470
- Silva, E. M. (2014). Screening five fall-sown cover crops for use in organic no-till crop production in the Upper Midwest. *Agroecol. Sust. Food Syst.* 38, 748–763. doi: 10.1080/21683565.2014.901275
- Silva, E. M., and Delate, K. (2017). A decade of progress in organic cover crop-based reduced tillage practices in the upper midwestern USA. *Agriculture* 7, 44. doi: 10.3390/agriculture7050044

- Skidmore, A., Wilson, N., Williams, M., and Bessin, R. (2019). The impact of tillage regime and row cover use on insect pests and yield in organic cucurbit production. *Renew. Agric. Food Syst.* 34, 338–348.
- Smith, A. N., Reberg-Horton, S. C., Place, G. T., Meijer, A. D., Arellano, C., and Mueller, J. P. (2011). Rolled rye mulch for weed suppression in organic no-tillage soybeans. *Weed Sci.* 59, 224–231. doi: 10.1614/WS-D-10-00112.1
- Snapp, S.S., Swinton, S.M., Labarta, R., Mutch, D., Black, J.R., Leep, R., et al. (2005). Evaluating cover crops for benefits, costs and performance within cropping system niches. *Agron. J.* 97, 322–332.
- Snyder, W. E. (2015). *Managing Cucumber Beetles in Organic Farming Systems*. Department of Entomology, Washington State University Pullman, WA, United States. Available online at: <https://eorganic.org/node/5307/>
- Sportelli, M., Frascioni, C., Fontanelli, M., Pirchio, M., Gagliardi, L., Raffaelli, M., et al. (2022). Innovative living mulch management strategies for organic conservation field vegetables: evaluation of continuous mowing, flaming, and tillage performances. *Agronomy* 12, 622. doi: 10.3390/agronomy12030622
- Teasdale, J. R., and Mohler, C. L. (2000). The quantitative relationship between weed emergence and the physical properties of mulches. *Weed Sci.* 48, 385–392. doi: 10.1614/0043-1745(2000)048[0385:TQRBWE]2.0.CO;2
- Teasdale, J. R., Rice, C. P., Cai, G., and Mangum, R. W. (2012). Expression of allelopathy in the soil environment: soil concentration and activity of benzoxazinoid compounds released by rye cover crop residue. *Plant Ecol.* 213, 893–1905. doi: 10.1007/s11258-012-0057-x
- Thomas, R., O'Sullivan, J., Hamill, A., and Swanton, C. J. (2001). Conservation tillage systems for processing tomato production. *HortScience* 36, 1264–1268. doi: 10.21273/HORTSCI.36.7.1264
- Tillman, J., Nair, A., Gleason, M., and Batzer, J. (2015). Evaluating strip tillage and rowcover use in organic and conventional muskmelon production. *Horttechnology* 25, 487–495. doi: 10.21273/HORTTECH.25.4.487
- Tonhasca, A. Jr., and Byrne, D. N. (1994). The effects of crop diversification on herbivorous insects: a meta-analysis approach. *Ecol. Entomol.* 19, 239–244. doi: 10.1111/j.1365-2311.1994.tb00415.x,
- USDA-NASS (2019). *2019 Organic Survey (2017 Census of Agriculture Special Report)*. United States Department of Agriculture – National Agricultural Statistics Service. Available online at: https://www.nass.usda.gov/Surveys/Guide_to_NASS_Surveys/Organic_Production/ (accessed on July 04, 2022).
- Van Den Bossche, A., De Bolle, S., De Neve, S., and Hofman, G. (2009). Effect of tillage intensity on N mineralization of different crop residues in a temperate climate. *Soil Tillage Res.* 103, 316–324. doi: 10.1016/j.still.2008.10.019
- Vincent-Caboud, L., Casagrande, M., David, C., Ryan, M. R., Silva, E. M., and Peigne, J. (2019). Using mulch from cover crops to facilitate organic no-till soybean and maize production. A review. *Agron. Sust. Dev.* 39, 1–15. doi: 10.1007/s13593-019-0590-2
- Vollmer, E. R., Creamer, N., Reberg-Horton, C., and Hoyt, G. (2010). Evaluating cover crop mulches for no-till organic production of onions. *HortScience Horts.* 45, 61–70. Available online at: <https://journals.ashs.org/hortsci/view/journals/hortsci/45/1/article-p61.xml>
- Wauters, V. M., Grossman, J. M., Pfeiffer, A., and Cala, R. (2021). Ecosystem services and cash crop tradeoffs of summer cover crops in northern region organic vegetable rotations. *Front. Sust. Food Syst.* 5, 39. doi: 10.3389/fsufs.2021.635955



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EDITED BY

E. Britt Moore,
University of North Carolina
Wilmington, United States

REVIEWED BY

Eugene Law,
University of Delaware, United States
K. Ann Bybee-Finley,
Cornell University, United States

*CORRESPONDENCE

Priscila Pinto
ppinto@wisc.edu

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Intercropping legumes and intermediate wheatgrass increases forage yield, nutritive value, and profitability without reducing grain yields

Priscila Pinto^{1*}, Stefania Cartoni-Casamitjana¹,
Colin Cureton², Andrew W. Stevens³, David E. Stoltenberg¹,
Joseph Zimbric¹ and Valentin D. Picasso¹

¹Department of Agronomy, University of Wisconsin–Madison, Madison, WI, United States, ²Supply Chain Development Specialist, Forever Green Initiatives, University of Minnesota, St. Paul, MN, United States, ³Department of Agricultural and Applied Economics, University of Wisconsin–Madison, Madison, WI, United States

Introduction: Kernza intermediate wheatgrass (IWG) is a perennial grain and forage crop. Intercropping IWG with legumes may increase the forage yields and nutritive value but may compromise Kernza grain yields. The interaction between IWG and legumes depends on planting season, row spacing, and legume species. Our aim was to evaluate the effects of those management practices on Kernza grain yield, summer and fall forage yield and nutritive value, weed biomass and, the profitability of the cropping system in Wisconsin, USA.

Methods: In the spring and fall of 2017, we planted eight cropping systems at 38 and 57 cm of row spacing: four IWG monocultures [control without N fertilization or weed removal (IWG), hand weed removal (hand weeded), IWG fertilized with urea at rates of 45 or 90 kg ha⁻¹], and four IWG-legume intercrops (IWG with alfalfa, Berseem clover, Kura clover, or red clover).

Results and discussion: Most of the intercropping systems were similar to IWG monoculture in grain (ranging from 652 to 1,160 kg ha⁻¹) and forage yield (ranging from 2,740 to 5,190 kg ha⁻¹) and improved the forage quality. However, for spring planted IWG, intercropped with red clover or alfalfa, the grain and forage yields were lower than the IWG monoculture (~80 and 450 kg ha⁻¹, respectively). The best performing intercrops in the first year were Kura clover in the spring planting (652 kg Kernza grain ha⁻¹, 4,920 kg IWG forage ha⁻¹ and 825 kg legume forage ha⁻¹) and red clover in the fall planting (857 kg Kernza grain ha⁻¹, 3,800 kg IWG forage ha⁻¹, and 450 kg legume forage ha⁻¹). In the second year, grain yield decreased 84% on average. Overall, the profitability of the IWG legume intercropping was high, encouraging the adoption of dual-purpose perennial crops.

KEYWORDS

polycultures, row spacing, seeding season, dual-purpose crop, net economic return, Kernza®, forage quality

Introduction

Ecosystem disservices from modern agriculture challenge the ability of society to meet current and future needs (Tilman et al., 2002; Power, 2010). The frequent tillage of soils and a lack of vegetation cover for prolonged periods have led to extensive soil erosion, soil carbon loss, and nutrient runoff into groundwater, among other problems, which demands the rethinking of the way humans produce food. Some novel approaches seek to diversify and perennialize cropping systems by reducing soil tillage (Crews and Rumsey, 2017), replacing fallow periods with service crops (Schipanski et al., 2014; Pinto et al., 2017), integrating crop and livestock systems (de Faccio Carvalho et al., 2021; Franco et al., 2021; Picasso et al., 2022), intercropping multifunctional species (Malézieux et al., 2009; Gaba et al., 2015) or including dual-purpose perennial crops in the agricultural rotations (Hunter et al., 2020b; Franco et al., 2021). Recent advances in domestication and breeding of perennial cereals for seed yield offer the opportunity to reintroduce perennial polycultures and regenerate components and processes of natural ecosystems to agroecosystems (Glover et al., 2010; Pimentel et al., 2012; Ryan et al., 2018). Since perennial crops last beyond one season, the disturbance needed for establishment can be compensated throughout multiple production years (Crews et al., 2016). Through their continuous productivity, perennial crops protect soil from erosion (Ryan et al., 2018), compete with weeds (Zimbric et al., 2020), catch nutrients preventing leaching (Culman et al., 2013; Jungers et al., 2019) and improve soil health (Culman et al., 2010; de Oliveira et al., 2020).

Intermediate wheatgrass [IWG, *Thinopyrum intermedium* (Host) Barkworth and D.R. Dewey)] is among the most promising perennial cereal crops to date (Ryan et al., 2018), due to its synchronous seed maturity, edible grain, moderate shattering, and moderate threshability (Wagoner, 1990). The current grain yield is relatively low relative to annual wheat [i.e., up to $\sim 1,660 \text{ kg ha}^{-1}$ in experimental fields (Franco et al., 2021) and averaging 460 kg ha^{-1} in the primary production areas (Skelly and Peters, 2021)] but breeders expect IWG to achieve comparable yields in the near future (DeHaan et al., 2018; Bajgain et al., 2020). The grain of IWG is sold as Kernza® to restaurants, bakeries, and other food-related businesses in the United States for use in value-added products (Lubofsky, 2016; Ryan et al., 2018). The forage can be harvested in summer, removing the crop residue or straw, and mixed with higher value forage (e.g., alfalfa hay) to feed beef or dry dairy cows. The forage harvested in spring or fall, as other cool-season grasses commonly grown in the humid climate of the Upper Midwestern US, is suitable for lactating beef cows, dairy cows, and growing heifers (Favre et al., 2019).

Growing legumes with perennial grasses can provide multiple benefits, including providing N inputs by biological fixation (Pinto et al., 2021b), increasing soil organic matter

(Lehmann et al., 2020), suppressing weeds (Law et al., 2021), and increasing the total forage harvested and its nutritive value (Favre et al., 2019). Nevertheless, little is known about the agronomic management of IWG-legume intercropping. Furthermore, different legume species could be better or worse companions of IWG to maximize benefits. Some experiences with perennial legumes have shown lower Kernza grain yields in intercropping with alfalfa (*Medicago sativa* sp.) or red clover (*Trifolium pratense* L.) than in the IWG monocultures (Tautges et al., 2018; Favre et al., 2019; Mårtensson et al., 2022). However, others showed similar Kernza grain yields in alfalfa, sweet clover, and white clover intercropping (Dick et al., 2018; Reilly et al., 2022). Slow establishing perennial legumes like Kura clover (*Trifolium ambiguum* M. Bieb) (Sleugh et al., 2000) or annual legumes like Berseem clover (*Trifolium alexandrinum* L.) could reduce competition and avoid the observed Kernza grain reductions. The interaction between different species involves the co-occurrence of both complementary and competitive relationships (Picasso et al., 2011; Duchene et al., 2017). Usually when grass-legume intercropping systems are compared with grass monocultures, negative and positive effects are confounded (e.g., competition for radiation and soil resources or facilitation processes through the symbiotic association between legumes and N-fixing bacteria). In order to separate the effects of competition for resources from N facilitation, weed removal and N fertilization treatments can be added to intercropping experiments.

Additionally, effective stand establishment is critical for IWG's long-term productivity, and in intercropping systems, it can be influenced by both the planting date and the row spacing. For IWG monocultures of the USA Midwest, late summer and early fall typically achieve successful establishment of Kernza grain production systems (Jungers et al., 2022). Intermediate wheatgrass requires a two-stage induction period with vernalization for flowering (Duchene et al., 2021; Locatelli et al., 2022), thus spring seedlings will not produce grain during the first year (Olugbenle et al., 2021; Jungers et al., 2022). For IWG-legume intercropping systems, limited information is available. When IWG was seeded in the fall, the highest intercropped Kernza grain yields were observed when red clover was frost seeded in the spring season (Olugbenle et al., 2021). When red clover was planted in the fall at the same time as IWG, lower Kernza grain yields were likely due to more competition during IWG establishment. The optimal planting season for IWG-legumes should be carefully studied because there is a trade-off between reducing the competition during IWG establishment and promoting the growth of the legumes to optimize benefits related to N fixation (Pinto et al., 2021b). In addition, IWG-legume interactions can be influenced by the distance between IWG rows (i.e., row spacing). In IWG monocultures, wider row spacing has been associated with higher Kernza grain yields than narrower row spacing (Hunter et al., 2020a). However, changes in the available resources such

as light, water, and nutrients due to row spacing are likely to vary among different legume species.

The profitability of the IWG cropping systems depends on both grain and forage incomes (Hunter et al., 2020b). Therefore, the lower Kernza grain yields harvested in alfalfa and red clover intercrops (Tautges et al., 2018; Favre et al., 2019; Mårtensson et al., 2022) could be compensated by positive effects on the increased forage yield and nutritive value. Intercropping IWG with red clover has consistently increased the nutritive value of the summer and fall forage and tripled the amount of available forage in the fall (Favre et al., 2019), positively affecting the revenue perceived by the farmers. In fact, it has been seen that higher forage yields achieved by IWG-legume intercropping systems reduce the Kernza grain price required to be profitable (Law et al., 2022). Although Kernza® grain markets are in a price discovery phase, estimating potential net returns could be useful to compare different cropping systems.

Our objectives were to evaluate the effects of IWG planting season, row spacing, and legume species in intercropping on (a) Kernza grain yield, (b) summer and fall forage yield and nutritive value, and (c) the potential profitability of the cropping system in Wisconsin, USA.

Materials and methods

The experiment was conducted at the University of Wisconsin-Madison Arlington Agricultural Research Station, WI (43°18'6.97" N, 89°21'9.98" W) on a Plano silt loam soil (fine-silty, mixed, superactive, mesic Typic Argiudoll; NRCS-USDA, 2022a). The mean annual temperature is 6.7°C, and the mean annual rainfall is 863 mm (Arguez et al., 2010). A large grain IWG germplasm, a product of four successive breeding cycles at The Land Institute (Salina, KS) was seeded at the rate of 11.2 kg ha⁻¹ in a field previously harvested for soybean grain. At the beginning of the experiment, thirty-two composite soil samples taken at 0–15 cm in the experimental area averaged 3.5% of soil organic matter, 56.5 ppm of P, 244 ppm of K, 5.1 ppm of NO₃-N and 21.1 ppm of NH₄-N. The experiment was established in 2017 in two different planting seasons (Figure 1): spring (April 12, 2017) and fall (September 21, 2017). In the spring planting season, the plot size was 3 m by 4.8 m, and in fall it was 3 m by 1.5 m. The first Kernza grain harvest for both planting seasons was in 2018 due to IWG vernalization requirements (Locatelli et al., 2022). During the establishment (year 2017), the growing degree days (GDD) and the precipitation accumulated until the first frost were 3,265 GDD and 646 mm, respectively, in the spring planting and, 534 GDD and 114 mm in the fall planting. Precipitation during the growing season until harvest (April to July) was 512 mm in 2018 and 560 mm in 2019, higher than normal (University of Wisconsin, Division of Extension, 2022). Data

were collected during two consecutive grain production years (Figure 1).

We installed a full factorial experiment of three factors: planting season, row spacing, and cropping systems. The planting season factor had two levels: IWG planted in the spring or in the fall of the year 2017. In the IWG spring planting, all forage legumes were sown drilling the inter-row 1 week after sowing IWG. In the IWG fall planting, forage legumes were sown frost in March 2018, hand seeded in the inter-row, pushing the IWG biomass by hand to improve the seed-soil contact. The IWG row spacing factor had two levels: wide (57 cm) or narrow (38 cm) spacing. The seeding rate (11.2 kg ha⁻¹) was the same for both row spacings, so the wide row spacing had ~50% more seeds per row than the narrow row spacing. The cropping system factor had 8 levels: four IWG monocultures [control without N fertilization or weed removal (IWG), hand weed removal (hand weeded), IWG fertilized with urea at rates of 45 or 90 kg ha⁻¹], and four IWG-legume intercrops (IWG with alfalfa, Berseem clover, Kura clover, or red clover). The hand weed removal was bi-weekly in the years 2017 and 2018, and only in May and June in the year 2019. The urea for the fertilized monocultures was broadcasted in a split application during the spring, half of the rate was applied at green up and the other half at IWG stem elongation. The legume seeding rates for fall were higher than the spring ones following the recommendation from forages in Wisconsin according to the planting method (Table 1). The annual legume (Berseem clover) was re-sown every spring. None of the intercrops were fertilized or hand weeded. The experimental design for each planting season was randomized complete blocks with five replications. The column of plots was also included as a source of variation in the model.

Data collection

Kernza grain yield and aboveground biomass were sampled approximately at physiological maturity on August 7, 2018, and August 1, 2019, from a 0.25 m² quadrat randomly placed in each plot. Aboveground biomass was also sampled in fall, on October 27, 2017, and October 24, 2018. Grain yield was determined by cutting the spikes from all tillers within the quadrat. Spikes were dried at 35°C for at least 2 days, threshed with a mechanical seed thresher, and weighed. Aboveground biomass was cut by hand, separated into IWG forage, legume forage, and weeds, dried at 60°C for at least 5 days, and weighed. The quadrat was placed so that one row of IWG would fit inside the quadrat for the wide row spacing, and two rows for the narrow row spacing. Both in wide and narrow row spacing, one legume inter-row of the quadrat was sampled. Kernza grain yield, IWG forage, legume forage, and weed biomass data were adjusted proportionally to the number of rows within the sampled quadrat, to obtain yields in kilograms per hectare. After sampling, grain was harvested

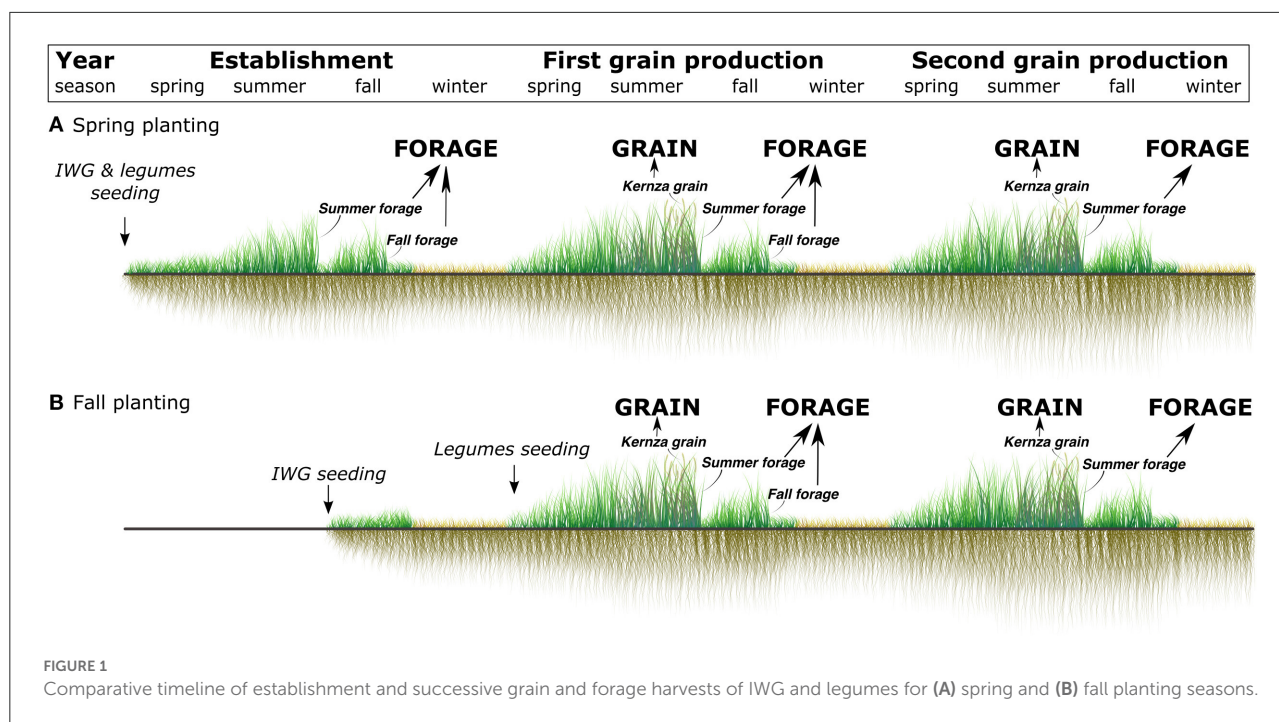


TABLE 1 Seeding rates and planting methods for legumes intercropped with IWG established in spring and fall at UW Arlington Research Station, Wisconsin, USA.

	Seeding rate (kg ha ⁻¹)		Planting method		Planting date	
	Spring	Fall	Spring	Fall	Spring	Fall
Alfalfa	6.7	19.2	Drilled	Hand seeded	April 2017	March 2018
Red clover	9.0	17.8	Drilled	Hand seeded	April 2017	March 2018
Kura clover	11.2	16.8	Drilled	Hand seeded	April 2017	March 2018
Berseem clover	11.2	16.8	Drilled	Hand seeded	April 2017, 2018, 2019	March 2018, 2019

with a plot combine and aboveground biomass was removed from the experiment using a mechanical forage harvester (Almaco, FH-88) leaving a stubble height of 10 cm. In the years 2017 and 2019, yield data was collected in all five replications. In the year 2018, yield data was collected in three replications due to labor availability limitations.

Forage nutritive value

Forage samples of IWG, alfalfa, Berseem clover, Kura clover, and red clover were analyzed to characterize the different species and be able to estimate the forage nutritive value in the intercropping systems. Samples of IWG forage were harvested from the different cropping systems in the summer and fall of the establishment and first grain production years. Samples of the different legume species were harvested in the summer and fall of the establishment year. We assumed that all the

species had the same forage quality in the first and second production years (Pinto et al., 2021c). Crude protein (CP), neutral detergent fiber (NDF), and acid detergent fibers (ADF) of the forage were analyzed using wet chemistry procedures and reported as a proportion of the dry matter. The selected samples were first ground with a Christy hammer mill (Christy-Turner Ltd, Ipswich, England) to pass a 1-mm screen. Total N was determined according to the Dumas combustion method (Method 990.03-AOAC, 2000) and the analysis was conducted in a LECO FP-528 (LECO Corporation, St-Joseph, MI). Crude protein was calculated as $N \times 6.25$. Neutral detergent fiber and ADF were analyzed sequentially in an Ankom 2,000 Fiber Analyser (Ankom technology, Macedon, NY) according to the procedure of Robertson and Van Soest (1981) and modified by Hintz et al. (1996) to include sodium sulfite during refluxing. For the IWG-legume intercrops, CP, NDF, and ADF concentration of the mixture forage was calculated as the weighted average of intermediate wheatgrass and legumes based on their respective

biomass proportion of the total forage accumulation. Relative Feed Value (RFV), an index that relativizes the nutritive value of forages to the fresh full-bloom alfalfa nutritive value, was calculated based on the following equations (Jeranyama and Garcia, 2004): Digestible Dry Matter = $88.9 - (0.779 \times \% \text{ acid detergent fiber})$; Dry Matter Intake (% of body weight) = $120 / (\% \text{ neutral detergent fiber})$; Relative Feed Value = $(\text{Digestible Dry Matter} \times \text{Dry Matter Intake}) / 1.29$.

Economic analysis

The potential profitability to grain and forage production was calculated from current market rates and the estimated cost of production in Wisconsin, USA. Variable incomes of each cropping system were estimated from Kernza grain harvested in 2018 and 2019, and summer and fall forage harvested in 2017 (spring season planting only), 2018, and 2019. Fall forage was not evaluated in the second year (2019), so it was assumed to equal 90% of fall forage in the first year (Hunter et al., 2020b) to complete the potential total incomes per year. Kernza grain yield losses (41%) usually observed in early commercial harvesting were estimated from the difference between average grain observed in our experiment and the last harvest data report from The Land Institute (Skelly and Peters, 2021). Kernza grain prices before cleaning or dehulling were \$3.30 kg⁻¹ (Tessa Peters, 2022, *pers comm*). Forage price was assigned by comparing the RFV (of IWG forage or IWG + legume forage in the intercrops) with the Upper Midwest hay price by quality grade. For each species, the same RFV was used for the first and second grain production year. The prices for hay grade Prime (>151 RFV), Grade 1 (125 to 150 RFV), Grade 2 (103 to 124 RFV), and Grade 3 (87 to 102) were \$0.23 kg⁻¹, \$0.18 kg⁻¹, \$0.13 kg⁻¹, and \$0.12 kg⁻¹, respectively (Halopka, 2022), and for forage with <87 RFV was \$0.10 kg⁻¹. In addition, a payment for actively managing and expanding conservation activities, offered to farmers in Wisconsin by the Conservation Stewardship Program (CSP) from Natural Resources Conservation Service (NRCS) from the US Department of Agriculture (USDA), was included in the establishment year (\$391 ha⁻¹, NRCS-USDA, 2022b). Variable costs were estimated considering the different inputs applied for each crop system. Berseem clover and red clover seed price was \$7.50 kg⁻¹ (Albert Lea Seed, 2022a,b), alfalfa seed price was \$9.92 kg⁻¹ (Albert Lea Seed, 2022c), Kura clover seed price was \$26.50 kg⁻¹ (Welter Seed Honey Co., 2022). Licensing and fees were \$12.4 ha⁻¹ and 3% of the income. Fixed costs included IWG seed (\$123 ha⁻¹), crop establishment (seeding including labor, \$137 ha⁻¹), Kernza grain harvest (\$64 ha⁻¹), and forage harvest (\$54 ha⁻¹) (Tessa Peters, 2022, *pers comm*). The land cost rent was \$329 ha⁻¹ (Wisconsin Agricultural Statistics, 2022). In order to account for the opportunity cost of not using the land for another crop when Kernza is planted in the spring season, we estimated the value of forage harvestable

of a 3-years-old Alfalfa pasture as \$319 ha⁻¹ and it was included as an income in the fall planting season (Extension Wisconsin, 2022).

Statistical analyses

All variables (i.e., Kernza grain yield, IWG and legumes forage yield, and weed biomass) were tested for normality and homogeneity of variances and transformed using square root to satisfy the assumptions of ANOVA. Different models were used to test specific hypotheses. First, in order to test the effects of the intercropping on the yields, we conducted an analysis of variance on Kernza grain yield and, IWG and legume forage with year (Y) as a repeated measure (covariance structure of compound symmetry); planting season (PS), cropping system (CS, including only the IWG monoculture control and the four intercropping treatments), row spacing (RS), and their interaction as fixed effects and block and column as random effects. Since these analyses showed a RS*PS*Y interaction effect on grain and forage yields, we performed a follow-up analyses of variance by year.

Second, in order to test the effects of the intercropping on the forage quality, we conducted an analysis of variance on the nutritive value metrics (percent CP, NDF, ADF, and RFV) with species, harvest season, and their interaction as fixed effects. Usually, the nutritive value metrics are rather constant over the years (Pinto et al., 2021c) but as IWG is in a vegetative state in the summer of the establishment year and in a reproductive state in the summer of the following years we considered both phenological states. Then, %CP and RFV of IWG-legume intercropping, estimated as the weighted average of IWG and legume forage and their nutritive values, were compared with %CP and RFV of IWG monoculture.

Third, in order to test the effects of the intercropping on the economic results, we conducted an analysis of variance on the annual profit (\$ ha⁻¹ year⁻¹) with PS, CS, and their interaction as fixed effect. In this analysis, row spacing was not included because different row spacings have the same costs (i.e., no changes in seeding rate) and similar incomes (i.e., little grain and forage variation) in our experiment. Since the Kernza® grain price is in a discovery phase, we performed analyses of variance by the % of the current Kernza grain yield utilized in the calculation (i.e., 100, 75, or 50% of the current Kernza grain price).

Fourth, in order to test the effect of management practices on IWG monocultures yields only, we conducted an analysis of variance on Kernza grain yield, IWG forage, and weed biomass by year, considering the effects of management (fertilization and weed removal), planting season (PS), row spacing (RS), and their interactions as fixed and block and column as random. Finally, since we found differences in the weed biomass, we compared the IWG monoculture with different weed management and

IWG intercrops with different legumes. We conducted an analysis of variance on weed biomass by year, with PS, CS, RS and their interaction as fixed effects and block and column as random effects. All analyses were performed using PROC MIXED procedure in SAS (SAS on Demand, SAS Institute, Cary, North Carolina, USA). Means were compared using the Tukey-Kramer honest significant difference test at $\alpha = 0.05$. Graphs were created using the ggplot2 (Wickham, 2009) package in RStudio Team. (2020).

Results

IWG-legume intercropping

The IWG cropping systems (IWG monoculture control and intercrops) had a high variability in grain and forage yields explained by row spacing, planting season, year, and their interactions (Supplementary Table 1). In the first grain production year, IWG planted at 38 cm of row spacing in the fall planting season yielded more than in the spring planting season (867 and 447 kg ha⁻¹, respectively, $p < 0.01$, Figure 2A). However, when IWG was planted at wider row spacing (57 cm), there was no difference between planting seasons (800 kg ha⁻¹, $p = 0.35$, Figure 2A). Overall grain yields decreased 85% in the second year regardless of the planting season or the row spacing (Figure 2A). The IWG forage remained relatively stable between years while that of legumes increased (Figures 2B,C). In the first year IWG forage was higher when it was planted in the fall at 57 cm of row spacing than when it was planted in the spring at 38 cm of row spacing (4,180 and 2,160 kg ha⁻¹, respectively, $p < 0.01$, Figure 2B). The IWG forage remained rather constant between the first and second grain production year for most cases except for IWG planted in fall at 57 cm where IWG decreased 41% in the second year (Figure 2B). The legume forage was rather constant between different planting seasons or row spacing but consistently increased in the second grain production year (Figure 2C). As a result, whereas the legume forage was 12% of the total summer forage in the summer of the first year, it increased to 47% in the second grain production year (Figures 2B,C).

The planting season and the legume species intercropped with IWG affected the grain and forage yields (Supplementary Table 1). When IWG was planted in the fall, the intercropping systems had similar grain yields to IWG monoculture, regardless of the legume species (Figure 3A). However, when IWG was planted in the spring, grain yields for intercrops with Berseem clover (1,160 kg ha⁻¹) or Kura clover (652 kg ha⁻¹) were similar to IWG monoculture, whereas red clover and alfalfa intercrops had lower grain yields (24 and 136 kg ha⁻¹, respectively, Figure 3A). In the second grain production year, all intercropping systems had similar grain yields to the monoculture in both planting seasons, except for

red clover intercrop in the fall, which had lower grain yields (Figure 3B). The row spacing effect was also different depending on the cropping system (p -value row spacing * cropping system * planting season interaction = 0.01). IWG-Kura clover intercrop planted in spring was the only cropping system with higher Kernza grain yield at wider row spacing (1,050 kg ha⁻¹ at wide vs. 256 kg ha⁻¹ at narrow). The rest of the cropping systems had similar Kernza grain at different row spacing independently of the planting season.

The forage yields had a similar response to the grain yield response: they were lower in the IWG planted in the spring intercropped with red clover (Supplementary Table 1). In the establishment year (2017), when forage can be harvested only for IWG planted in the spring, the intercropping with red clover had lower IWG forage yield (422 kg ha⁻¹) than the monoculture and the rest of intercrops (1,300 kg ha⁻¹ on average, Figure 4A). In the first grain production year (2018), the IWG-legume intercrops had similar IWG summer forage yield to the IWG monoculture planted in the fall (3,730 kg ha⁻¹ on average, Figure 4B). However, when planted in the spring, IWG intercropped with red clover or alfalfa had lower IWG summer forage yield (152 and 744 kg ha⁻¹, respectively) than the IWG monoculture or the rest of the intercrops (4,610 kg ha⁻¹ on average). Only red clover and alfalfa had differences on IWG summer forage yield between the spring and fall planting (Figure 4B). Usually, the legume summer forage yield did not compensate for low IWG summer forage yield. Red clover and alfalfa had the lowest total summer forage yield although they produced ~1,100 kg ha⁻¹ of legume summer forage (Figure 4B). In contrast, legumes tend to increase total fall forage yield in the first grain year production, although only the Kura clover intercrop had higher total fall forage yield than the IWG monoculture when both were planted in the spring (2,420 vs. 1,140 kg ha⁻¹, respectively, Figure 4C). Finally, in the second grain production year, most of the intercrops had similar IWG summer forage yield to the IWG monoculture except for the red clover intercrop, which had lower yield. The intercropping systems with the least legume biomass accumulation were IWG with Berseem clover and alfalfa in both planting seasons. Their biomass were lower than Kura clover legume biomass in the spring planting and lower than that of red clover in the fall planting (Figure 4D). The summer Kura clover forage yield was more than half of the total forage yield in the second year, resulting in higher total summer forage yield than the IWG monocrop (7,160 vs. 2,930 kg ha⁻¹, respectively when both were planted in the spring, Figure 4D).

Overall, the forage yield in the establishment year and in the fall of the first production year was lower than the forage yield in the summer of the first production year but had higher nutritive value. In the summer of the establishment year, IWG and legumes had similar percentages of CP, NDF, ADF, and RFV. In the fall of the establishment year, alfalfa forage had higher CP and RFV than IWG forage, whereas other legumes

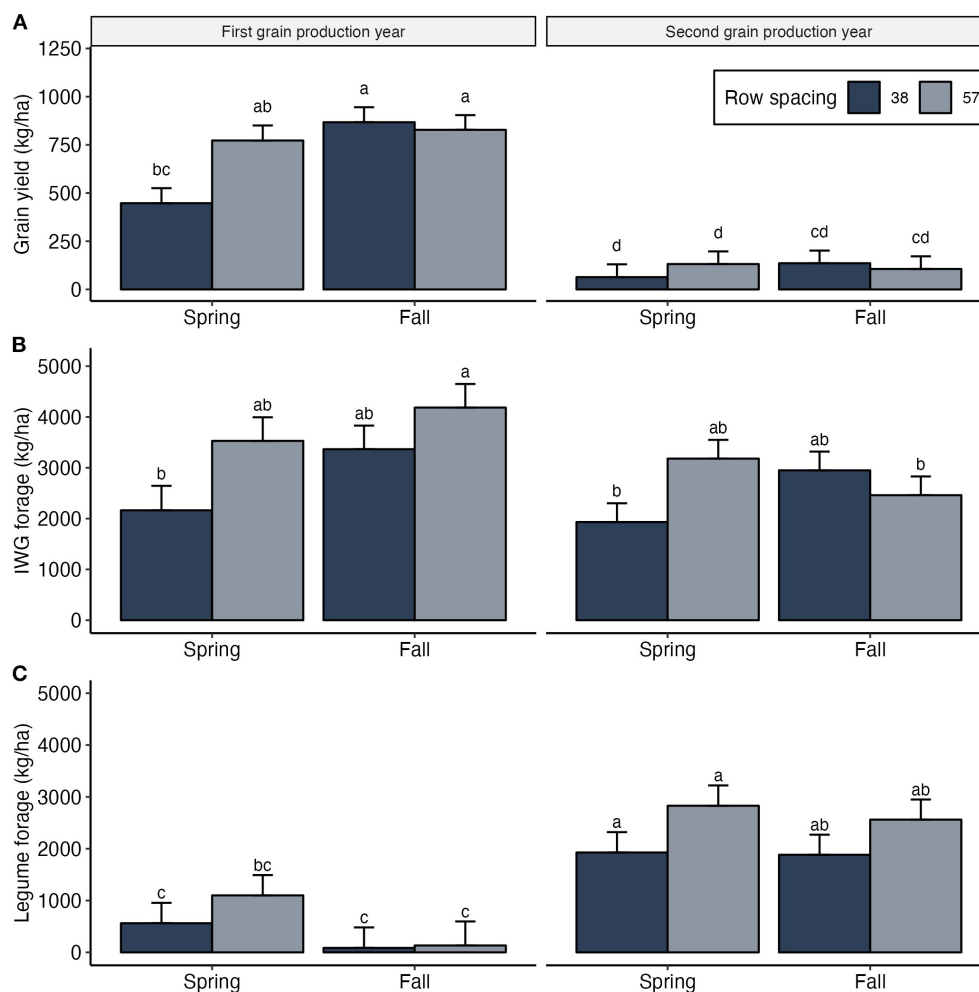


FIGURE 2

Grain (A), and IWG (B) and legume (C) forage yields (kg ha^{-1}) of the first (2018) and second (2019) grain production year for five IWG cropping systems sown at two row spacings (38 or 57 cm) in two planting seasons (spring or fall 2017), at Arlington, Wisconsin, USA. The cropping systems are IWG monoculture control without N fertilization or weed removal, and four IWG-legume intercrops (IWG with Berseem clover, Kura clover, red clover, or alfalfa). Same letters for each variable indicate no differences at $\alpha = 0.05$.

had intermediate values (Table 2). In the first grain production year, red clover forage harvested both in summer and fall, had similar CP, NDF, ADF, and RFV to legumes in the establishment year. Instead, in the summer of the first grain production year IWG had the lowest percent CP (Table 2). Considering the legume proportion of the total forage harvested in each cropping system and the nutritive values of the IWG and the legumes, we estimated that some intercrops had better nutritive value than IWG monoculture. On the one hand, the fall harvested forage of IWG planted in the spring and intercropped with Kura clover, red clover or alfalfa had higher CP and the RFV than that of IWG monoculture (Figure 5). The fall forage of IWG planted in the fall intercropped with red clover also had higher nutritive value than the IWG monoculture, but the intercrop with other legumes did not. All these CP and RFV increases meant positive

changes on the hay quality designation. On the other hand, the IWG monoculture summer forage was classified as “fair” while the intercrop with red clover and alfalfa reached “premium” or “grade 3–4” when IWG and the legumes were planted together in the spring (Figure 5).

Economic analysis

Mean potential profitability to grain and forage production varied between $\$260$ and $\$961 \text{ ha}^{-1} \text{ year}^{-1}$ (Table 3). Cost and incomes were highly variable among cropping systems between years (Supplementary Table 2). In the establishment year, costs varied between $\$589$ and $\$994 \text{ ha}^{-1}$ whereas incomes varied between $\$682$ and $\$976 \text{ ha}^{-1}$ among cropping systems.

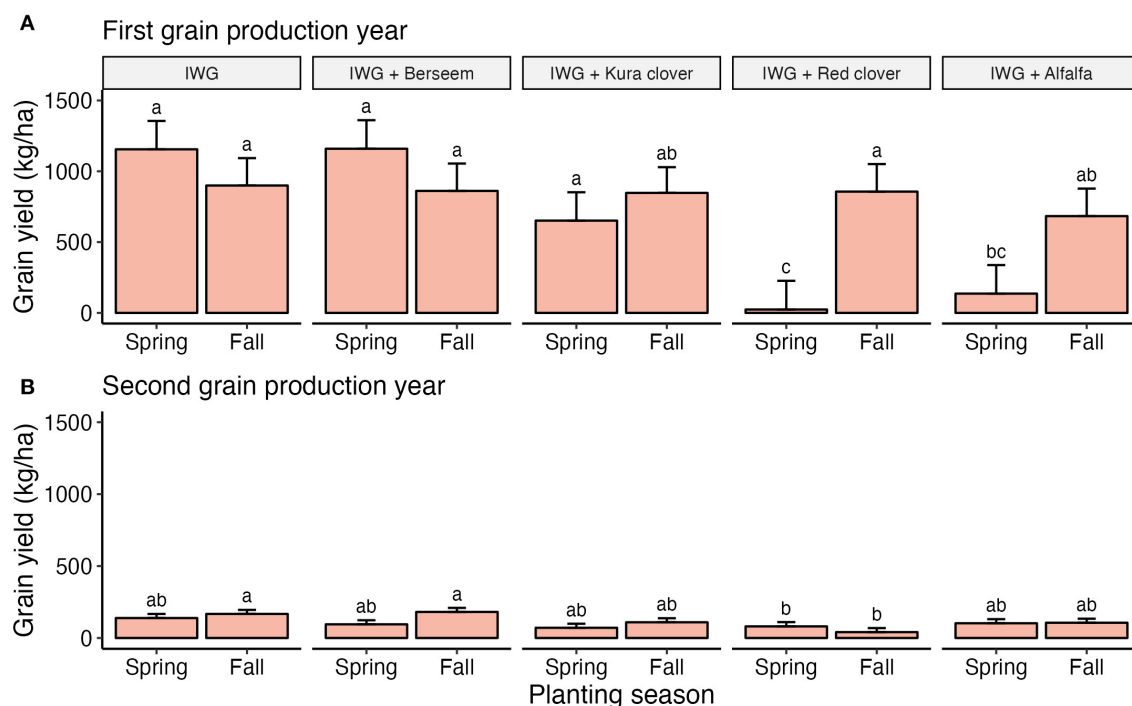


FIGURE 3

Grain yields (kg ha^{-1}) of the first (2018, **A**) and second (2019, **B**) grain production years for IWG monoculture without N fertilization or weed removal (IWG) and four IWG intercrops with annual (Berseem clover) or perennial legumes (Kura clover, red clover, alfalfa) sown in two planting seasons (spring or fall 2017), at Arlington, Wisconsin, USA. Same letters indicate no differences at $\alpha = 0.05$.

The subsidy for planting perennial crops represented from 40 to 57% of the income in the establishment year and the net returns were lower than in the grain production years (Supplementary Table 2). The first grain production year had higher profitability than the other years, except when IWG was planted in the spring intercropped with red clover (Supplementary Table 2). The sale of the Kernza grain represented 60% of the total income $\text{ha}^{-1} \text{ year}^{-1}$ in the IWG monoculture cropping system and varied between 8 and 55% in the intercropping systems (Table 3). The highest profitability per year was $\$898 \text{ ha}^{-1} \text{ year}^{-1}$ in the IWG intercropped with Kura clover planted in the spring and $\$961 \text{ ha}^{-1} \text{ year}^{-1}$ in the IWG intercropped with red clover planted in the fall. Most of the cropping systems had similar annual profit, except for IWG intercropped with alfalfa planted in the spring ($\$260 \text{ ha}^{-1} \text{ year}^{-1}$, Table 3). The sensitivity analysis indicated that a change in the price of Kernza grain has a little impact in which cropping systems are most profitable, as the pattern of significant differences is largely the same at all three Kernza grain price considered (Table 3). The influence of the Kernza grain price on the annual profit was high but variable among the cropping systems. Assuming a Kernza grain price reduction of 25 and 50%, the annual profit had similar reductions in the IWG-control cropping system (26 and 52%, respectively, Table 3). In contrast, the Kernza grain price reduction tended to impact

less on the red clover intercropping system's annual profit (reductions of 10 and 20%, respectively, Table 3) indicating that this cropping system is more highly dependent on forage production than grain.

IWG management practices on yields and weeds

The IWG monocultures had high variability in grain and forage yields but this was more explained by year than by row spacing, management, or planting season effect. The Kernza monoculture grain yields were $945 \pm 73 \text{ kg ha}^{-1}$ in the first grain production year and decreased to $147 \pm 10 \text{ kg ha}^{-1}$ ($p < 0.01$) in the second year. The IWG summer forage (straw) yields were $4,370 \pm 308 \text{ kg ha}^{-1}$ in the first grain production year, and $3,490 \pm 235 \text{ kg ha}^{-1}$ in the second year ($p < 0.01$). No response to different N fertilization rates or weed removal (i.e., management practices) was found in grain and IWG forage yields in the first grain production year (Supplementary Table 3). In the second grain production year, spring planting at wide row spacing and fall planting at narrow row spacing had higher grain yields than spring planting at narrow row spacing (173 kg ha^{-1} on average and 91 kg ha^{-1} ,

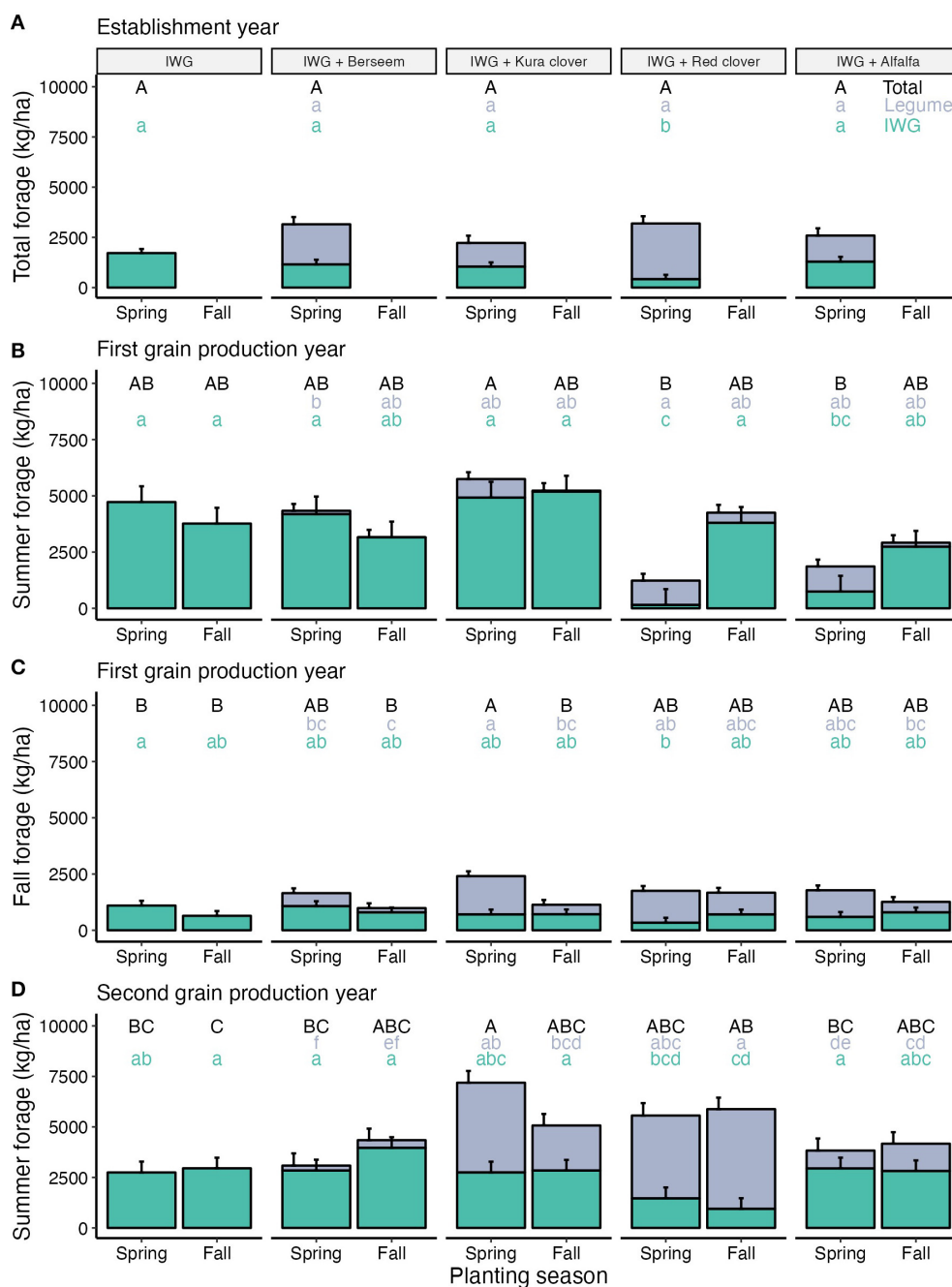


FIGURE 4

IWG and legume forage (kg ha^{-1}) harvested in the establishment year (A), in the summer (B) or fall (C) of the first grain production year, and in the summer of the second grain production year (D) for IWG monoculture (IWG) and four IWG intercrops with annual (Berseem clover) or perennial legumes (Kura clover, red clover, alfalfa) sown in two planting seasons (spring or fall 2017), at Arlington, Wisconsin, USA. Same letters indicate no differences at $\alpha = 0.05$ in IWG forage (green lowercase letters), legume forage (purple lowercase letters) or total forage (black capital letters).

respectively, [Supplementary Table 3](#)). The IWG forage yield was higher in both spring and fall planting at wide row spacing than in spring at narrow row spacing ($3,740 \text{ kg ha}^{-1}$ on average, and $2,750 \text{ kg ha}^{-1}$, respectively, [Supplementary Table 3](#)). In contrast to the first year, the management practices affected the IWG

forage yield ([Supplementary Table 3](#)): the IWG fertilized with urea at 45 kg ha^{-1} had higher IWG forage than IWG unfertilized hand weeded ($4,280 \text{ kg ha}^{-1}$ and $2,950 \text{ kg ha}^{-1}$, respectively).

As expected, weed summer biomass was lower in the hand weeded plots (203 kg ha^{-1}) than the control ($1,900 \text{ kg ha}^{-1}$) in

TABLE 2 Means (standard errors) for crude protein (CP), neutral detergent fiber (NDF), acid detergent fiber (ADF), and the relative feed value (RFV) of intermediate wheatgrass (IWG), alfalfa, Berseem clover, Kura clover and red clover forage harvested in the summer and fall of the establishment and the first Kernza grain production years (IWG-vegetative was harvested in the summer of the establishment year and IWG-reproductive, in the summer of the first grain production year).

Harvest season	Species	CP%			NDF%			ADF%			RFV		
Summer	IWG-vegetative	17.3	(0.5)	b	55.1	(1.3)	b	30.0	(0.8)	bc	111	(5)	de
	IWG-reproductive	5.6	(1.1)	c	69.6	(2.6)	a	42.6	(1.6)	a	75	(11)	e
	Berseem	17.0	(2.5)	ab	45.7	(5.7)	bcdef	31.2	(3.6)	abcde	131	(24)	abcde
	Kura clover	-			47.2	(3.3)	bcde	32.9	(2.1)	bc	127	(14)	bcde
	Red clover	18.8	(1.3)	ab	46.2	(1.9)	cd	31.7	(1.2)	b	135	(8)	cd
Fall	IWG	17.1	(0.4)	b	50.6	(0.9)	bc	27.5	(0.5)	bcd	125	(4)	d
	Alfalfa	23.6	(1.0)	a	33.5	(2.3)	f	22.0	(1.5)	e	201	(10)	a
	Berseem	16.7	(1.1)	b	39.8	(2.2)	def	27.1	(1.4)	bcde	161	(9)	abc
	Kura clover	20.5	(1.0)	ab	37.5	(1.7)	ef	26.1	(1.0)	cde	178	(7)	a
	Red clover	20.0	(1.0)	ab	39.5	(1.7)	def	24.4	(1.1)	de	175	(7)	ab

Same letters within each parameter indicate no differences at $\alpha = 0.05$.

the first year, when the weeds were weekly removed. However, it was similar in the second year when weeds were only removed twice ($1,310 \text{ kg ha}^{-1}$, mean of both, [Figure 6](#)). In the second year, summer weed biomass was lower at narrow row spacing ($1,010 \text{ kg ha}^{-1}$) than at wide row spacing ($1,760 \text{ kg ha}^{-1}$) in both spring and fall plantings ([Supplementary Table 3](#)). Kura clover and red clover were effective in reducing weed biomass. In the first year, both intercropping systems had similar weed biomass to the hand weeded treatment ([Figure 6](#)). In the second year, weed biomass decreased 39% on average among the different cropping systems (from $1,400 \text{ kg ha}^{-1}$ in the first year to 851 kg ha^{-1} in the second). Weed biomass was lower in the intercrops with red clover and Kura clover than in the hand weeded treatment (87, 317, and $1,166 \text{ kg ha}^{-1}$, respectively, in the second year).

Discussion

IWG management practices

The N fertilization and the weed removal management in IWG monocultures are useful to understand the potential limitations by N or weed competition in our experiment. The lack of Kernza grain yield response when weeds were removed by hand, suggest that interspecific competition may not be a problem as IWG is well established and accumulated $\sim 4,200 \text{ kg ha}^{-1}$ of IWG aboveground biomass in the summer. However, most of the weeds present at the beginning of the experiment were annuals, and therefore, different results are likely found in fields dominated by perennial weeds ([Zimbric et al., 2020](#)). On the other hand, N fertilization usually mitigates the decline in grain yield with stand age ([Jungers et al., 2017](#); [Tautges et al., 2018](#); [Fernandez et al., 2020](#)) but no effect was found in grain or IWG forage in our experiment. Likely, both the high initial

N soil content and the fact that the previous crop was a legume avoided N limitation in the IWG control cropping system. In the following years, N is likely to become limited because of the N exportation with the harvest of Kernza grain and IWG forage. Without any other N fertilization, the legumes intercropped would need to accumulate approximately $4,500 \text{ kg}$ of biomass ha^{-1} to provide enough N to meet IWG demands considering that 50% of the N uptake by the legumes comes from biological fixation ([Pinto et al., 2021b](#)).

The positive response of planting at wider row spacing was lower than we expected. Based on previous experiences, we hypothesized that increasing the distance between rows would help maintain grain allocation over time ([Canode, 1968](#); [Hunter et al., 2020a](#)). However, large declines in the allocation to grain were consistently observed in our experiment since grain yield declined whereas forage remained rather constant over time. A possible reason for the limited effect of row spacing is that the different row spacing treatments had the same seeding density per area in our experiment. This means that the wide row spacing treatment has a higher seeding density per row than the narrow, so higher row competition could likely have confounded the effects. Future research should maintain density per row in wide row spacing (i.e., reduce density per hectare) to see if increasing resources per plant in wide row spacing allows yields to be maintained in older stands.

In general, planting IWG and the legumes together in the spring was no better than planting in the fall. Although it is widely known that IWG does not produce grain in the summer of the establishment year when it is planted in the spring due to lack of vernalization induction ([Duchene et al., 2021](#); [Olugbenle et al., 2021](#); [Jungers et al., 2022](#); [Locatelli et al., 2022](#)), we hypothesized that some advantages could be manifested in the first grain production year. However, growing IWG for a longer establishment period (i.e., with more

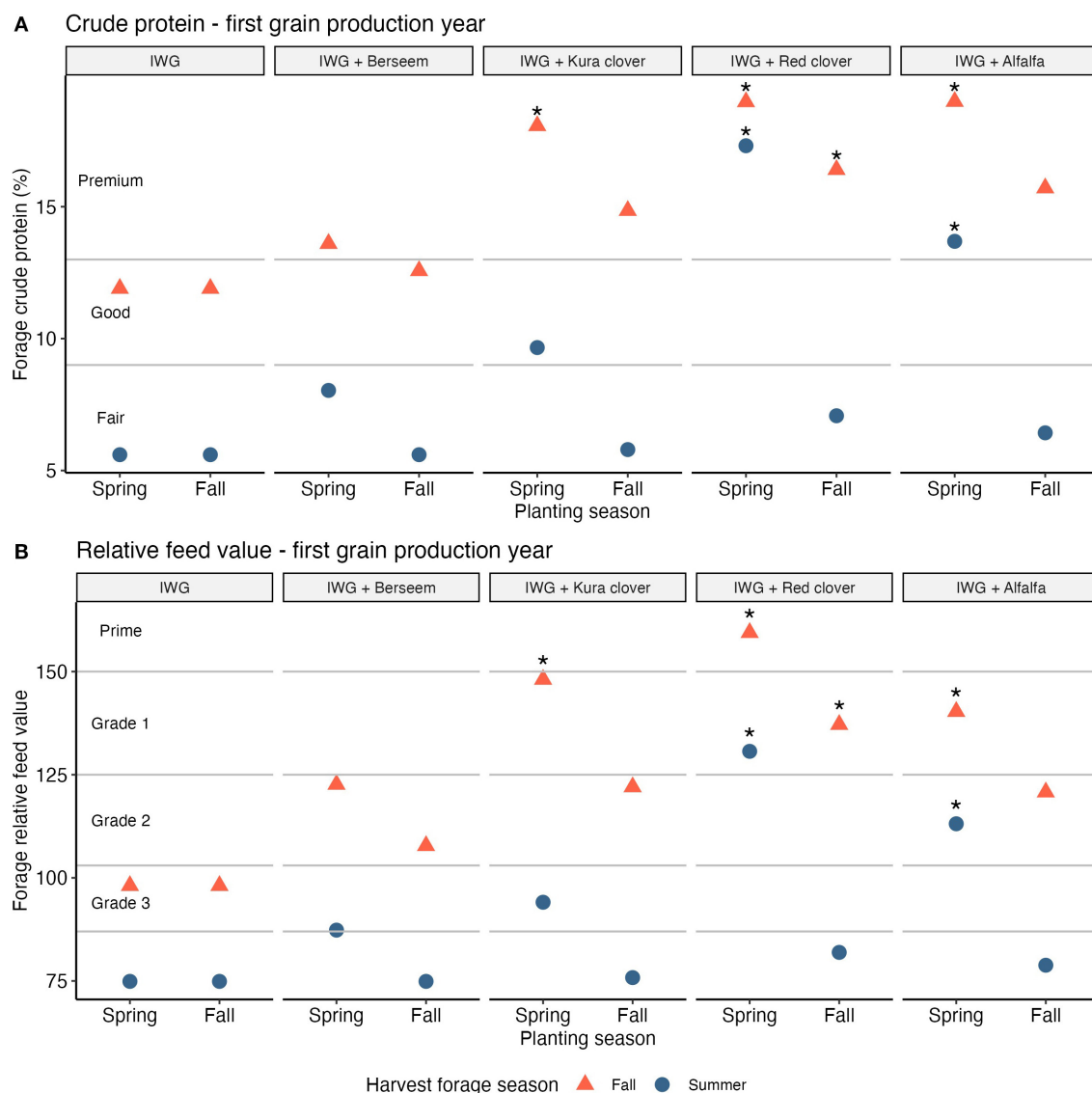


FIGURE 5

Percent crude protein (A) and relative feed value (B) of the total forage (IWG + legumes) harvested in the summer or fall of the first grain production year (2018), for the control IWG monoculture (i.e., without N fertilization or weed removal), and four IWG intercrops with legumes (Berseem clover, Kura clover, red clover, alfalfa) sown at two planting seasons (spring, fall), at Arlington, Wisconsin, USA. Gray lines show the limits of the forage quality grade according to Halopka (2022) and USDA (2022). The asterisks (*) indicate differences with the minimum value (control IWG monoculture) at $\alpha = 0.05$.

rainfall and GDD accumulated when IWG is planted in the spring) did not lead to higher grain or IWG forage in the first grain production year than planting in the fall. In the IWG-legume intercropping systems, planting IWG and legumes together in the spring tended to favor the growth of legumes. Higher legume biomass accumulation usually implies a higher N contribution by biological N fixation (Pinto et al., 2021b) but in our experiment this potential N contribution was not relevant since no response to N fertilization was seen in the IWG monocultures. In contrast, high biomass accumulated by red

clover and alfalfa, which are legumes well adapted to Wisconsin (Sheaffer et al., 2020), compromised the establishment of IWG and its grain and forage yields.

IWG-legume intercropping

Most of the legume species intercropped with Kernza were good companions of IWG since they did not compromise the Kernza grain and IWG forage yields (Figure 7). Overall,

TABLE 3 Mean costs, income, and annual profit (\$ ha⁻¹year⁻¹) of five different cropping systems planted in spring or fall 2017, at both narrow and wide row spacing: IWG monoculture without N fertilization or weed removal (control), and IWG intercropped with Berseem clover, Kura clover, red clover, or alfalfa, for three Kernza grain prices.

Cropping system	IWG						IWG + Berseem clover						IWG + Kura clover						IWG + Red clover						IWG + Alfalfa								
Planting season	Spring			Fall			Spring			Fall			Spring			Fall			Spring			Fall			Spring			Fall					
Costs (\$ ha ⁻¹ year ⁻¹)																																	
Land rent	329			329			329			329			329			329			329			329			329			329			329		
Crop establishment	46			46			46			46			46			46			46			46			46			46			46		
Grain harvest	42			42			42			42			42			42			42			42			42			42			42		
Forage harvest	108			72			108			72			108			72			108			72			108			72			108		
Seed	41			41			125			125			140			189			64			86			63			105					
Licensing and fees	45			38			45			39			48			43			29			48			27			35					
Income (\$ ha ⁻¹ year ⁻¹)																																	
Alfalfa forage	0			106			0			106			0			106			0			106			0			106			106		
NRCS payment	130			130			130			130			130			130			130			130			130			130			130		
Kernza grain	872			711			832			702			478			646			82			621			147			548					
Summer forage	305			219			321			251			591			391			451			527			328			242					
Fall forage	141			46			229			86			397			106			344			190			259			117					
Annual profit (SE)																																	
100% of Kernza price	841	(166)	ab	644	(157)	ab	825	(160)	ab	626	(151)	ab	898	(217)	a	664	(206)	ab	401	(147)	ab	961	(175)	a	260	(53)	b	520	(162)	ab			
75% of Kernza price	630	(131)	ab	472	(123)	ab	623	(128)	ab	456	(123)	ab	783	(184)	a	507	(166)	ab	381	(135)	ab	811	(156)	a	224	(43)	b	387	(135)	ab			
50% of Kernza price	418	(97)	abc	300	(89)	abc	421	(97)	abc	286	(96)	abc	667	(152)	ab	351	(125)	abc	361	(123)	abc	660	(137)	a	188	(37)	c	254	(108)	bc			

Same letters indicate that there are no differences at alpha = 0.05 among annual profit of the cropping systems means (SE), for each % of the current Kernza grain price. See costs and incomes for the establishment and the two first grain production years for all cropping systems in [Supplementary Table 2](#).

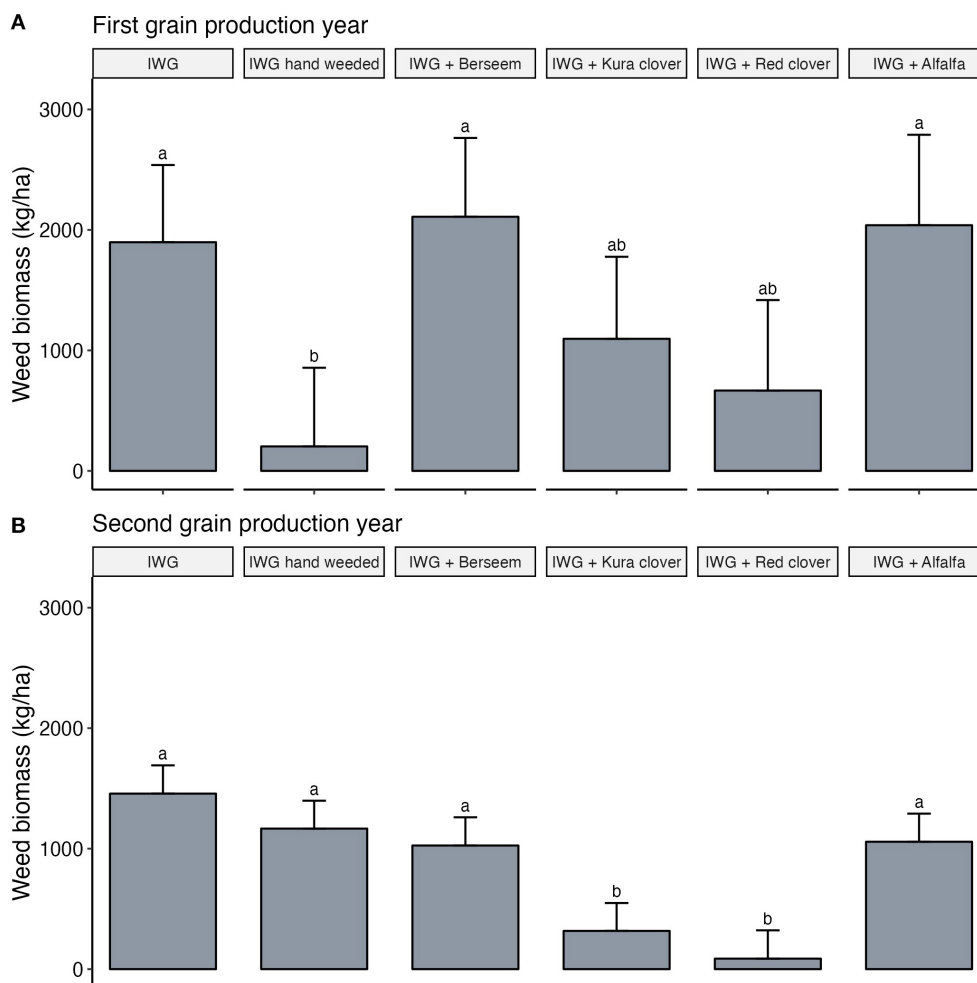


FIGURE 6

Weed biomass (kg ha^{-1}) of the first (2018, **A**) and second (2019, **B**) grain production years for two IWG monocultures (IWG: without N fertilization or weed removal, IWG hand weeded: with hand weed removal) and four IWG intercrops with annual (Berseem clover) or perennial legumes (Kura clover, red clover, alfalfa) sown in two planting seasons (spring or fall 2017), at Arlington, Wisconsin, USA. Same letters indicate no differences at $\alpha = 0.05$.

Kernza grain yields were highly variable in the first year with differences found only when IWG was not well established. The low Kernza yields achieved by IWG intercropped with red clover or alfalfa in the spring planting suggest that the early IWG biomass accumulation is key. As these legumes' establishment is aggressive (Tautges et al., 2018), legume frost seeded in the spring on IWG planted in the previous fall has been recommended (Law et al., 2021; Olugbenle et al., 2021). The other cropping systems had a wide grain yield range consistent with previously reported yields for 1-year-old stands (Franco et al., 2021). Summer forage yields had a similar response to treatments as Kernza grain yields. Except for IWG intercropped with red clover and alfalfa planted in the spring, the cropping systems had summer forage yields within the range previously reported (Franco et al., 2021). In the second year, summer

forage yields decreased but at a slower rate than grain yields. This reduction in grain allocation found in older IWG stands is consistent with previous results (Tautges et al., 2018; Fernandez et al., 2020; Law et al., 2021) although in most of them this was associated with summer forage yield increases.

The benefits of intercropping IWG with legumes were more related to an improvement on nutritive value than on the amount of total forage harvested. Although previous studies had shown increases in the total forage harvested in IWG intercropped with red clover (Favre et al., 2019; Law et al., 2022); in our experiment red clover biomass production did not compensate for the decrease in IWG biomass, resulting in similar total forage yields. The most promising legume to increase the total forage was Kura clover when it was planted with IWG together in the spring even though the greatest

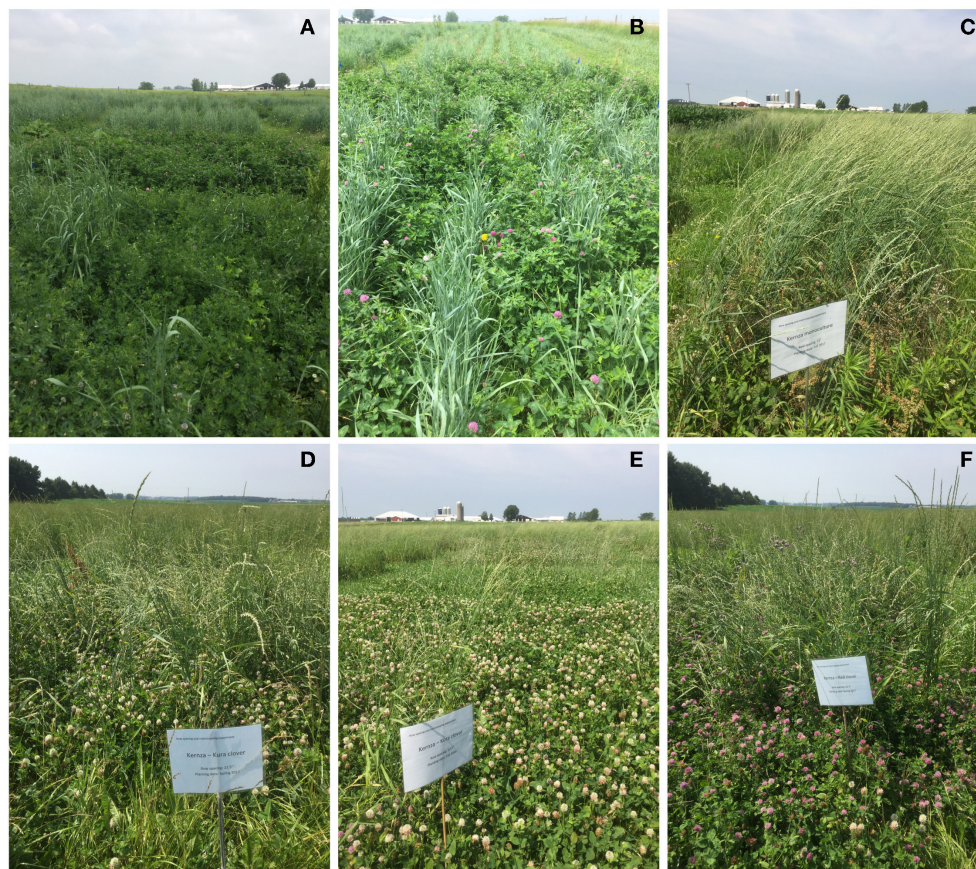


FIGURE 7

Pictures of intermediate wheatgrass cropping systems in Arlington, Wisconsin, USA: (A) IWG-alfalfa intercropping, spring planting (date picture taken: June 12, 2018); (B) IWG-red clover intercropping, spring planting (June 12, 2018); (C) IWG monoculture control, fall planting (July 9, 2019); (D) IWG-Kura clover intercropping, spring planting (July 9, 2019); (E) IWG-Kura clover intercropping, fall planting (July 9, 2019); (F) IWG-red clover intercropping, spring planting (July 9, 2019).

differences were seen after the first grain harvest (i.e., fall of the first grain production year and summer of the second grain production year). In contrast, all the perennial legumes improved the forage nutritive value, as seen previously in IWG-red clover intercropping (Favre et al., 2019). Intercropping IWG with red clover, Kura clover, or alfalfa was associated with increases in the hay quality designation, increasing the price per kg of forage (Halopka, 2022; USDA, 2022).

We hypothesized that annual legumes would be better companions than perennial legumes, but scarce biomass was accumulated by annuals in our experiment. Annual legumes could be good alternative companions in the long term since the biomass accumulation by perennial legumes tends to increase as stands get older (e.g., Figure 2, Tautges et al., 2018) and competition with IWG could become limiting. However, in our experiment the biomass accumulated by the annual legume Berseem clover was insufficient to provide increases in the total forage harvested or its nutritive value. Initially we had incorporated soybean (*Glycine max* (L.) Merr.) as another annual legume companion, but we decided to exclude this

treatment from our analysis because the soybean biomass was negligible (data not shown). This treatment tended to yield more Kernza grain than the other intercrops, especially when IWG was planted in the fall. The positive effect was unlikely to be due to changes in N levels but probably because the soil in the inter-row was tilled before soybean planting potentially creating beneficial effects such as those observed under mechanical thinning (Law et al., 2020; Pinto et al., 2021a). Proper management of annual legumes could lead to better intercropping results but the need to plant them every year limits its benefits. Therefore, it seems more promising to learn to regulate the competition between IWG and perennial legumes, for example with forage cuttings, than to intercrop with annual legumes.

Economic analysis

Most of the cropping systems had higher profitability than a 3-years-old Alfalfa pasture (\$319 ha⁻¹, Wisconsin Agricultural

Statistics., 2022), soybean (\$421 ha⁻¹, Economic Research Service., 2022a) or corn (\$637 ha⁻¹, Economic Research Service., 2022b) except when IWG was poorly established in the red clover and alfalfa intercrops planted in the spring. However, this high profitability is explained by the high Kernza[®] grain price and the subsidy for its ecosystem services provision. A 50% reduction in Kernza grain price would make only the best-performing IWG systems competitive with corn and soybean. Kernza[®] grain markets are in a price discovery phase with prices varying significantly by the management system, year, region, and grain quality. The price included in this paper falls in the middle of the range of observed 2021 farmgate prices for non-organic Kernza grain. Current target and received grain prices reflect significant risks in early commercial Kernza production and marketing. The multiple uncertainties involved with Kernza represent the main disadvantage perceived by the farmers who decide to plant this crop (Lanker et al., 2020). Although Kernza production can be quite profitable where it succeeds, several growers still fail at IWG establishment, experience major weed pressure, do not meet food-grade specifications, and have limited market access. These relatively high risks also explain Kernza's relatively high price (Tessa Peters, 2022, *pers comm*). Progress in the genetics and management of Kernza IWG will likely lead to less risky scenarios but with lower and more stable prices. That means, the projected net returns given by our relatively strong prices and yields, should be considered optimistic and do not fully encompass the risks of commercial Kernza production and marketing at this time (Tessa Peters, 2022, *pers comm*).

Our results suggest that red clover and Kura clover are good companions of IWG, given their high profitability at any Kernza grain price considered. Kura clover and IWG can be planted together in the spring, but IWG must be planted in the fall and red clover in the next spring to see the benefits. The legume forage contribution seems to be key to buffer the potential impact of kernza grain price volatility on the overall profitability. Both higher quantity and quality of the forage harvested in the intercropping systems than in the IWG monoculture lead to increases in the income from forages. As a result, the proportion of revenue coming from Kernza grain decreases as well as the impact of Kernza grain price volatility. This means a great potential for intercropping in the future. If Kernza grain yields increase from breeding advances or agronomic management innovations without sacrificing forage yield or quality in these best intercropping systems, it would be a win-win from a production standpoint and increase the crop's economic viability even if grain prices are reduced.

The current ecosystem services value to growers, communities, or society is reflected in the subsidy provided by the Conservation Stewardship Program (NRCS-USDA, 2022b). However, the IWG's key role in preventing nutrient leaching (Culman et al., 2013; Jungers et al., 2019) and improving soil health (Culman et al., 2010; de Oliveira et al., 2020)

can be also reflected in the access to new markets in the future, such as water and carbon credits. Ecosystem services, such as water quality, soil health, carbon sequestration, and biodiversity, are appreciated by people, but the incentives for the provision that comes with prices are incipient (Swinton et al., 2007). Understanding how humans perceive and value ecosystem services is key, but a lack of low-cost measurability and valuation currently precludes efficient allocation of many ecosystem services through market-based approaches (Kroegeer and Casey, 2007). How to rigorously incorporate these benefits into economic analyses of cropping system performance warrants more research conducted with a transdisciplinary approach.

Limitations and future perspectives

Interpretation of how planting season determined our results is limited because our experiment was not replicated in time or space. This means that the effect of stand age and weather on grain and forage yields cannot be separated. Our experiment was installed in spring and fall of a year wetter than normal (year 2017). In normal or drier years, the interaction between Kernza IWG and legumes could be different and therefore, it should be studied in other environments. For example, in our experiment the intercropping with red clover or alfalfa planted in the spring season seems to be risky because it limited the IWG establishment. However, in other environments or using different management practices (e.g., different seed rate, different cutting regiment, spring forage harvest), legume may have less of a competitive advantage over IWG, resulting in a viable intercrop. Future experiments should consider repeated plantings in consecutive years as recommended to evaluate perennial forage grasses (Casler, 1999). Besides, considering other environments or trying different management strategies for intercropping will help to have more tools to design more diverse cropping systems.

On the other hand, the lack of some measurements led us to rely on assumptions to interpret our results. In the second grain production year, the fall forage production was not evaluated and we assumed a 10% reduction of the first year forage according to the annual averages published by Hunter et al. (2020b) (3,000 kg/ha in 2015 and 2,700 kg/ha in 2016). This allowed us to estimate the total potential incomes of each year and calculate the annual profit but different intercropping could differently affect forage yields in the second year. In addition, weed biomass was not measured in the fall forage harvest nor considered to determine the forage nutritive value. Both in the summer and fall forage, the nutritive value was determined considering the %CP and RFV of IWG and legumes and their proportion in the mixture. However, the analysis of composite samples of all the forage harvested in the plot could differ due to the impact of the weed biomass. Lastly, taking measurement

in the five blocks in the first year would have helped to better characterize the first grain and forage productions.

The design of dual-purpose intercrops is a promising practice that should be carefully evaluated considering multiple dimensions (Duchene et al., 2017; Crews et al., 2018; Law et al., 2021). The possibility to harvest forage twice a year provides an additional source of income and is often beneficial for the Kernza grain yield maintenance. Usually, the high presence of straw and biomass residues in older Kernza IWG stands reduces the resource allocation to grains because reproductive tiller initiation is reduced by shade (Ensign et al., 1983; Chastain, 2003). Therefore, harvesting forage could help to maintain a high harvest index to avoid the grain declines commonly observed (Pugliese et al., 2019; Hunter et al., 2020b; Pinto et al., 2021a). In dual-purpose crops where the forage represents an important proportion of the total income, the mixture with legume helps to improve the forage value (Favre et al., 2019; Halopka, 2022). These advantages make it feasible to include perennial and diverse cropping systems in agricultural rotations to improve their sustainability.

Conclusion

Dual-purpose IWG-legume intercropping systems are promising alternative production systems but both legume species and intercrop management techniques should be carefully chosen to favor the benefits. Most of the intercropping systems achieved similar Kernza grain and forage to IWG monoculture and improve the forage quality. However, our results suggest that when IWG is planted in the spring, intercropped with red clover or alfalfa, the Kernza grain and the IWG summer forage can be reduced by an early high competition. The intercropping with Kura clover or red clover was as effective in weed suppressing as the hand-removal management in the IWG monoculture. In the second year, Kernza grain yields decreased consistently in all cropping systems. In our experiment, planting in a wide row spacing did not prevent the grain yield decline but reducing the seeding rate per hectare in the wide row spacing could lead to different results. Overall, the profitability of the IWG legume intercropping was high mainly due to the current high Kernza grain price and the subsidies provided to farmers in Wisconsin to encourage the adoption of dual-purpose perennial crops.

Data availability statement

The original contributions presented in the study are included in the article/Supplementary material, further inquiries can be directed to the corresponding author.

Author contributions

VP, DS, and PP contributed to conception and design of the study. SC-C and JZ collected and organized the database. PP and SC-C performed the statistical analysis. PP, SC-C, CC, and AS wrote sections of the manuscript. PP wrote the first draft of the manuscript. All authors contributed to manuscript, revision, read, and approved the submitted version.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

The Supplementary Material for this article can be found online at: <https://www.frontiersin.org/articles/10.3389/fsufs.2022.977841/full#supplementary-material>

References

- Albert Lea Seed (2022a). *Berseem clover*. Available online at: <https://alseed.com/product/frosty-berseem-clover/> (accessed March, 2022).
- Albert Lea Seed (2022b). *Red clover*. Available online at: <https://alseed.com/product/medium-red-clover/> (accessed March, 2022).
- Albert Lea Seed (2022c). *Alfalfa*. Available online at: <https://alseed.com/product/viking-394ap-brand-alfalfa/> (accessed March, 2022).
- Arguez, A., Durre, I., Applequist, S., Squires, M., Vose, R., Yin, X., et al. (2010). NOAA's U.S. Climate Normals (1981–2010). NOAA National Centers for Environmental Information.
- Bajgain, P., Zhang, X., Jungers, J. M., DeHaan, L. R., Heim, B., Sheaffer, C. C., et al. (2020). 'MN-Clearwater', the first food-grade intermediate wheatgrass (Kernza perennial grain) cultivar. *J. Plant Regist.* 14, 288–297. doi: 10.1002/plr2.20042
- Canode, C. L. (1968). Influence of row spacing and nitrogen fertilization on grass seed production. *Agron. J.* 60, 263–267. doi: 10.2134/agronj1968.00021962006000030006x
- Casler, M. D. (1999). Repeated measures vs. repeated plantings in perennial forage grass trials: an empirical analysis of precision and accuracy. *Euphytica*. 105, 33–42. doi: 10.1023/A:1003476313826
- Chastain, T. G. (2003). Biological principles of seed production. *The art and science of seed production in the Pacific Northwest*. p. 12–34.
- Crews, T., and Rumsey, B. (2017). What agriculture can learn from native ecosystems in building soil organic matter: a review. *Sustainability*. 9, 578. doi: 10.3390/su9040578
- Crews, T. E., Blesh, J., Culman, S. W., Hayes, R. C., Jensen, E. S., Mack, M. C., et al. (2016). Going where no grains have gone before: from early to mid-succession. *Agric. Ecosyst. Environ.* 223, 223–238. doi: 10.1016/j.agee.2016.03.012
- Crews, T. E., Carton, W., and Olsson, L. (2018). Is the future of agriculture perennial? Imperatives and opportunities to reinvent agriculture by shifting from annual monocultures to perennial polycultures. *Glob. Sustain.* 1, 1–18. doi: 10.1017/sus.2018.11
- Culman, S. W., DuPont, S. T., Glover, J. D., Buckley, D. H., Fick, G. W., Ferris, H., et al. (2010). Long-term impacts of high-input annual cropping and unfertilized perennial grass production on soil properties and belowground food webs in Kansas, USA. *Agric. Ecosyst. Environ.* 137, 223–238. doi: 10.1016/j.agee.2009.11.008
- Culman, S. W., Snapp, S., Ollenburger, M., and Basso, B. (2013). Soil and water quality rapidly responds to the perennial grain Kernza wheatgrass. *Agron. J.* 105, 735–744. doi: 10.2134/agronj2012.0273
- de Faccio Carvalho, C. P., Savian, J. V., Della Chiesa, T., de Souza, W., Terra, J. A., Pinto, P., et al. (2021). Land-use intensification trends in the Rio de la Plata region of South America: Toward specialization or recoupling crop and livestock production. *Front. Agric. Sci. Eng.* 8, 97–110. doi: 10.15302/J-FASE-2020380
- de Oliveira, G., Brunsell, N. A., Crews, T. E., DeHaan, L. R., and Vico, G. (2020). Carbon and water relations in perennial Kernza (*Thinopyrum intermedium*): an overview. *Plant Sci.* 295, 107747. doi: 10.1016/j.plantsci.2019.110279
- DeHaan, L., Christians, M., Crain, J., and Poland, J. (2018). Development and evolution of an intermediate wheatgrass domestication program. *Sustainability*. 10, 1499. doi: 10.3390/su10051499
- Dick, C., Cattani, D., and Entz, M. H. (2018). Kernza intermediate wheatgrass (*Thinopyrum intermedium*) grain production as influenced by legume intercropping and residue management. *Can. J. Plant Sci.* 98, 1376–1379. doi: 10.1139/cjps-2018-0146
- Duchene, O., Dumont, B., Cattani, D. J., Fagnant, L., Schlautman, B., DeHaan, L. R., et al. (2021). Process-based analysis of *Thinopyrum intermedium* phenological development highlights the importance of dual induction for reproductive growth and agronomic performance. *Agric. For Meteorol.* 301–302, 108341. doi: 10.1016/j.agrformet.2021.108341
- Duchene, O., Vian, J. F., and Celette, F. (2017). Intercropping with legume for agroecological cropping systems: complementarity and facilitation processes and the importance of soil microorganisms. a review. *Agric. Ecosyst. Environ.* 240, 148–161. doi: 10.1016/j.agee.2017.02.019
- Economic Research Service. (2022a). USDA-Recent Cost and Return: Soybean (Northern Crescent). Available online at: <https://www.ers.usda.gov/data-products/commodity-costs-and-returns/> (accessed September, 2022).
- Economic Research Service. (2022b). USDA-Recent Cost and Return: Corn (Northern Crescent). Available online at: <https://www.ers.usda.gov/data-products/commodity-costs-and-returns/> (accessed September, 2022).
- Ensign, R. D., Hickey, V. G., and Bernardo, M. D. (1983). Effects of sunlight reduction and post-harvest residue accumulations on seed yields of Kentucky bluegrass. *Agron. J.* 75, 549–551. doi: 10.2134/agronj1983.00021962007500030030x
- Extension Wisconsin. (2022). Alfalfa profitability by year. Available online at: <https://fyi.extension.wisc.edu/forage/alfalfa/#econ> (accessed March, 2022).
- Favre, J. R., Munoz, T., Combs, D. K., Wattiaux, M. A., and Picasso, V. D. (2019). Forage nutritive value and predicted fiber digestibility of Kernza intermediate wheatgrass in monoculture and in mixture with red clover during the first production year. *Anim. Feed Sci. Technol.* 258, 114298. doi: 10.1016/j.anifeedsci.2019.114298
- Fernandez, C. W., Ehlke, N., Sheaffer, C. C., and Jungers, J. M. (2020). Effects of nitrogen fertilization and planting density on intermediate wheatgrass yield. *Agron. J.* 112, 4159–4170. doi: 10.1002/agi2.20351
- Franco, J. G., Berti, M. T., Grabber, J. H., Hendrickson, J. R., Nieman, C. C., Pinto, P., et al. (2021). Ecological intensification of food production by integrating forages. *Agronomy*. 11, 1–26. doi: 10.3390/agronomy11122580
- Gaba, S., Lescouret, F., Boudsocq, S., Enjalbert, J., Hinsinger, P., Journet, E. P., et al. (2015). Multiple cropping systems as drivers for providing multiple ecosystem services: from concepts to design. *Agron. Sustain. Dev.* 35, 607–623. doi: 10.1007/s13593-014-0272-z
- Glover, J. D., Reganold, J. P., Bell, L. W., Borevitz, J., Brummer, E. C., Buckler, E. S., et al. (2010). Increased food and ecosystem security via perennial grains. *Science*. 328, 1638–1639. doi: 10.1126/science.1188761
- Halopka, R. (2022). Hay Market Demand and Price Report for the Upper Midwest—for August 29, 2022. University of Wisconsin-Madison, Crop and Soil Division of Extension. Available online at: <https://cropsandsoils.extension.wisc.edu/hay-market-demand-and-price-report-for-the-upper-midwest-for-august-29-2022/>
- Hintz, R. W., Mertens, D. R., and Albrecht, K. A. (1996). Effects of sodium sulfite on recovery and composition of detergent fiber and lignin. *J. AOAC Int.* 79, 16–22. doi: 10.1093/jaoac/79.1.16
- Hunter, M. C., Sheaffer, C. C., Culman, S. W., and Jungers, J. M. (2020a). Effects of defoliation and row spacing on intermediate wheatgrass I: grain production. *Agron. J.* 112, 1748–1763. doi: 10.1002/agi2.20128
- Hunter, M. C., Sheaffer, C. C., Culman, S. W., Lazarus, W. F., and Jungers, J. M. (2020b). Effects of defoliation and row spacing on intermediate wheatgrass II: forage yield and economics. *Agron. J.* 112, 1862–1880. doi: 10.1002/agi2.20124
- Jeranyama, P., and Garcia, A. D. (2004). *Understanding relative feed value (RFV) and relative forage quality (RFQ)*. SDSU Extension Extra Archives, 352. Available online at: https://openprairie.sdstate.edu/extension_extra/352
- Jungers, J. M., DeHaan, L. H., Mulla, D. J., Sheaffer, C. C., and Wyse, D. L. (2019). Reduced nitrate leaching in a perennial grain crop compared to maize in the Upper Midwest, USA. *Agric. Ecosyst. Environ.* 272, 63–73. doi: 10.1016/j.agee.2018.11.007
- Jungers, J. M., Schiffrer, S., Sheaffer, C. C., Ehlke, N. J., Dehaan, L. R., Torrior, J., et al. (2022). Effect of seeding date on grain and biomass yield of Intermediate wheatgrass. *Agron. J.* 114, 2342–51. doi: 10.1002/agi2.21083
- Jungers, J. M., DeHaan, L. R., Betts, K. J., Sheaffer, C. C., and Wyse, D. L. (2017). Intermediate wheatgrass grain and forage yield responses to nitrogen fertilization. *Agron. J.* 109, 462–472. doi: 10.2134/agronj2016.07.0438
- Kroeger, T., and Casey, F. (2007). An assessment of market-based approaches to providing ecosystem services on agricultural lands. *Ecol. Econ.* 64, 321–332. doi: 10.1016/j.ecolecon.2007.07.021
- Lanker, M., Bell, M., and Picasso, V. D. (2020). Farmer perspectives and experiences introducing the novel perennial grain Kernza intermediate wheatgrass in the US Midwest. *Renew. Agric. Food Syst.* 35, 653–62. doi: 10.1017/S1742170519000310
- Law, E. P., Pelzer, C. J., Wayman, S., Ditommaso, A., and Ryan, M. R. (2020). Strip-tillage renovation of intermediate wheatgrass (*Thinopyrum intermedium*) for maintaining grain yield in mature stands. *Renew. Agric. Food Syst.* 1–7. doi: 10.1017/S1742170520000368
- Law, E. P., Wayman, S., Pelzer, C. J., Culman, S. W., Gómez, M. I., DiTommaso, A., et al. (2022). Multi-criteria assessment of the economic and environmental sustainability characteristics of intermediate wheatgrass grown as a dual-purpose grain and forage crop. *Sustainability*. 14, 3548. doi: 10.3390/su14063548
- Law, E. P., Wayman, S., Pelzer, C. J., DiTommaso, A., and Ryan, M. R. (2021). Intercropping red clover with intermediate wheatgrass suppresses weeds without reducing grain yield. *Agron. J.* 114, 700–716. doi: 10.1002/agi2.20914

- Lehmann, J., Hansel, C. M., Kaiser, C., Kleber, M., Maher, K., Maher, K., et al. (2020). Persistence of soil organic carbon caused by functional complexity. *Nat. Geosci.* 13, 529–534. doi: 10.1038/s41561-020-0612-3
- Locatelli, A., Gutierrez, L., and Picasso, V. D. (2022). Vernalization requirements of Kernza intermediate wheatgrass. *Crop Sci.* 62, 524–535. doi: 10.1002/csc2.20667
- Lubofsky, E. (2016). The promise of perennials: working through the challenges of perennial grain crop development. *CSA News.* 61, 4–7. doi: 10.2134/csa2016-61-11-1
- Mårtensson, L. M. D., Barreiro, A., Li, S., and Jensen, E. S. (2022). Agronomic performance, nitrogen acquisition and water-use efficiency of the perennial grain crop *Thinopyrum intermedium* in a monoculture and intercropped with alfalfa in Scandinavia. *Agron. Sustain. Dev.* 21, 1–10. doi: 10.1007/s13593-022-00752-0
- Malézieux, E., Crozat, Y., Dupraz, C., Laurans, M., Makowski, D., Ozier-Lafontaine, H., et al. (2009). Mixing plant species in cropping systems: concepts, tools and models. A review. *Agron. Sustain. Dev.* 29, 43–62. doi: 10.1051/agro:2007057
- NRCS-USA. (2022a). Natural Resources Conservation Service, United States Department of Agriculture. Official Soil Series Descriptions. Available online at: https://www.nrcs.usda.gov/wps/portal/nrcs/detail/soils/scientists/?cid=nracs142p2_053587 (accessed May, 2022).
- NRCS-USA. (2022b). Conservation Stewardship Program-Wisconsin-E3280. Available online at: https://www.nrcs.usda.gov/wps/PA_NRCSConsumption/download?cid=NRCSPEPRD1855304&ext=.pdf (accessed September, 2022).
- Olugbenle, O., Pinto, P., and Picasso, V. D. (2021). Optimal planting date of Kernza intermediate wheatgrass intercropped with red clover. *Agronomy.* 11, 2227. doi: 10.3390/agronomy11112227
- Picasso, D., Brummer, E. C., Liebman, M., and Dixon, P. M. (2011). Diverse perennial crop mixtures sustain higher productivity over time based on ecological complementarity. *Renew. Agric. Food Syst.* 26, 317–327. doi: 10.1017/S1742170511000135
- Picasso, V. D., Berti, M., Cassida, K., Collier, S., Fang, D., Finan, A., et al. (2022). Diverse perennial circular forage systems are needed to foster resilience, ecosystem services, and socioeconomic benefits in agricultural landscapes. *Grasslands Research.* 1, 123–130. doi: 10.1002/qlr2.12020
- Pimentel, D., Cerasale, D., Stanley, R. C., Perlman, R., Newman, E. M., Brent, L. C., et al. (2012). Annual vs. perennial grain production. *Agric. Ecosyst. Environ.* 161, 1–9. doi: 10.1016/j.agee.2012.05.025
- Pinto, P., Culman, S., Crews, T. E., DeHaan, L., Jungers, J., Larsen, J., et al. (2021c). “Dual-use Kernza Intermediate wheatgrass seasonal forage yield and nutritional value across North America.” In *American Society of Agronomy (ASA), Crop Science Society of America (CSSA), and Soil Science Society of America (SSSA) International Annual Meetings* (Salt Lake City, UT), 7–10.
- Pinto, P., DeHaan, L., and Picasso, V. (2021a). Post-harvest management practices impact on light penetration and Kernza intermediate wheatgrass yield components. *Agronomy.* 11, 442. doi: 10.3390/agronomy11030442
- Pinto, P., Fernández-Long, M. E., and Piñeiro, G. (2017). Including Cover Crops during fallow periods for increasing ecosystem services: Is it possible in croplands of Southern South America? *Agric. Ecosyst. Environ.* 248, 48–57. doi: 10.1016/j.agee.2017.07.028
- Pinto, P., Rubio, G., Gutiérrez, F., Sawchik, J., Arana, S., Piñeiro, G., et al. (2021b). Variable root : shoot ratios and plant nitrogen concentrations discourage using just aboveground biomass to select legume service crops. *Plant Soil.* 463, 347–358. doi: 10.1007/s11104-021-04916-x
- Power, A. G. (2010). Ecosystem services and agriculture: tradeoffs and synergies. *Philosophical transactions of the Royal Society B.* 365, 2959–2971. doi: 10.1098/rstb.2010.0143
- Pugliese, J. Y., Culman, S. W., and Sprunger, C. D. (2019). Harvesting forage of the perennial grain crop kernza (*Thinopyrum intermedium*) increases root biomass and soil nitrogen cycling. *Plant Soil.* 437, 241–254. doi: 10.1007/s11104-019-03974-6
- Reilly, E. C., Gutknecht, J. L., Tautges, N. E., Sheaffer, C. C., and Jungers, J. M. (2022). Nitrogen transfer and yield effects of legumes intercropped with the perennial grain crop intermediate wheatgrass. *Field Crops Res.* 286, 108627. doi: 10.1016/j.fcr.2022.108627
- Robertson, J. B., and Van Soest, P. J. (1981). “The detergent system of analysis and its application to human foods.” in *The Analysis of Dietary Fiber in Food*. Marcel Dekker. Eds W.P. James, O. Theander (New York), 123–158.
- RStudio Team. (2020). RStudio: Integrated Development for R. RStudio, PBC, Boston, MA. Available online at: <http://www.rstudio.com/>
- Ryan, M. R., Crews, T. E., Culman, S. W., Dehaan, L. R., Hayes, R. C., Jungers, J. M., et al. (2018). Managing for multifunctionality in perennial grain crops. *BioScience.* 68, 294–304. doi: 10.1093/biosci/biy014
- Schipanski, M. E., Barbercheck, M., Douglas, M. R., Finney, D. M., Haider, K., Kaye, J. P., et al. (2014). A framework for evaluating ecosystem services provided by cover crops in agroecosystems. *Agric. Syst.* 125, 12–22. doi: 10.1016/j.agsy.2013.11.004
- Sheaffer, C. C., Evers, G. W., and Jungers, J. M. (2020). Cool-season legumes for humid areas. In: *Forages, Volume 2: The Science of Grassland Agriculture*, 7th Edition, Eds K. J. Moore, M. Collins, C. J. Nelson, D. D. Redfearn (Wiley, Chichester, West Sussex, UK), 968.
- Skelly, S., and Peters, T. (2021). Kernza® Perennial Grain: 2021. Planting and Harvest data. Salina, Kansas. Available online at: <https://kernza.org/wp-content/uploads/Kernza-Growers-2021-Data.pdf>
- Sleugh, B., Moore, K. J., George, J. R., and Brummer, E. C. (2000). Binary legume-grass mixtures improve forage yield, quality, and seasonal distribution. *Agron. J.* 92, 24–29. doi: 10.2134/agronj2000.92124x
- Swinton, S. M., Lupi, F., Robertson, G. P., and Hamilton, S. K. (2007). Ecosystem services and agriculture: cultivating agricultural ecosystems for diverse benefits. *Ecol. Econ.* 64, 245–252. doi: 10.1016/j.ecolecon.2007.09.020
- Tautges, N. E., Jungers, J. M., Dehaan, L. R., Wyse, D. L., and Sheaffer, C. C. (2018). Maintaining grain yields of the perennial cereal intermediate wheatgrass in monoculture v. bi-culture with alfalfa in the Upper Midwestern USA. *J. Agric. Sci.* 156, 758–773. doi: 10.1017/S0021859618000680
- Tilman, D., Cassman, K. G., Matson, P. A., Naylor, R., and Polasky, S. (2002). Agricultural sustainability and intensive production practices. *Nature.* 418, 671–677. doi: 10.1038/nature01014
- University of Wisconsin, Division of Extension. (2022). Current Season Updates-Wisconsin Corn Agronomy. Available online at: <http://corn.agronomy.wisc.edu/Season/Default.aspx> (accessed May, 2022).
- USDA. (2022). Agricultural Marketing Service. Livestock, poultry, and grain market news. Hay quality designation guidelines. Available online at: <https://www.ams.usda.gov/sites/default/files/media/HayQualityGuidelines.pdf> (accessed September, 2022).
- Wagoner, P. (1990). New use for intermediate wheatgrass. *J. Soil Water Conserv.* 45, 81–82.
- Welter Seed and Honey Co. (2022). Kura clover. Available online at: <https://welterseed.com/inventory/kura-clover-coated/> (accessed March, 2022).
- Wickham, H. (2009). *Ggplot2: Elegant Graphics for Data Analysis*. 2nd Edition, Springer, New York.
- Wisconsin Agricultural Statistics. (2022). USDA Statistic by state-Wisconsin. Available online at: https://www.nass.usda.gov/Statistics_by_State/Wisconsin/Publications/Annual_Statistical_Bulletin/2021AgStats-WI.pdf (accessed March, 2022).
- Zimbric, J. W., Stoltenberg, D. E., and Picasso, V. D. (2020). Effective weed suppression in dual-use intermediate wheatgrass systems. *Agron. J.* 112, 2164–2175. doi: 10.1002/agi2.20194



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EDITED BY

Jacob Jungers,
University of Minnesota Twin Cities,
United States

REVIEWED BY

Ignacio Macedo,
University of California, Davis,
United States
Joel Tallaksen,
University of Minnesota Twin Cities,
United States

*CORRESPONDENCE

Carl Wepking
cwepking@wisc.edu

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Perennial grassland agriculture restores critical ecosystem functions in the U.S. Upper Midwest

Carl Wepking ^{1*}, Hunter C. Mackin ¹, Zach Raff ²,
Debendra Shrestha ¹, Anna Orfanou ¹, Eric G. Booth ¹,
Christopher J. Kucharik ¹, Claudio Gratton ³ and
Randall D. Jackson ¹

¹Department of Agronomy, University of Wisconsin-Madison, Madison, WI, United States, ²Social Science Department, University of Wisconsin-Stout, Menomonie, WI, United States, ³Department of Entomology, University of Wisconsin-Madison, Madison, WI, United States

Dominant forms of agricultural production in the U.S. Upper Midwest are undermining human health and well being. Restoring critical ecosystem functions to agriculture is key to stabilizing climate, reducing flooding, cleaning water, and enhancing biodiversity. We used simulation models to compare ecosystem functions (food-energy production, nutrient retention, and water infiltration) provided by vegetation associated with continuous corn, corn-soybean rotation, and perennial grassland producing feed for dairy livestock. Compared to continuous corn, most ecosystem functions dramatically improved in the perennial grassland system (nitrate leaching reduced ~90%, phosphorus loss reduced ~88%, drainage increased ~25%, evapotranspiration reduced ~29%), which will translate to improved ecosystem services. Our results emphasize the need to incentivize multiple ecosystem services when managing agricultural landscapes.

KEYWORDS

agroecology, ecosystem services, livestock, grazing, multifunctionality

Introduction

Agriculture is central to the fundamental challenges facing human society (Godfray et al., 2010; Foley et al., 2011; Wheeler and von Braun, 2013; Amundson et al., 2015; Kremen and Merenlender, 2018). We must develop and grow agricultural systems that provide for our well being while building the capacity of future generations to do the same. These agricultural systems must be resilient in the face of drought, flooding, and extreme weather, as well as socio-economic shocks such as pandemics and market failures (Lesk et al., 2016; Lioutas and Charatsari, 2021; Ortiz-Bobea, 2021).

Critical ecosystem functions of agricultural systems include plant and animal productivity, soil carbon storage (Rowntree et al., 2020; Guillaume et al., 2022; Rui et al., 2022), soil stabilization (Montgomery, 2007; Palm et al., 2014; Schulte et al., 2017), nutrient retention (Schulte et al., 2017; Hussain et al., 2019; Jackson, 2020), water infiltration and storage (Basche and DeLonge, 2019; Baker et al., 2022), and wildlife habitat (Kimoto et al., 2012; Tsiafouli et al., 2015; Schulte et al., 2017). While these factors range in their scale of influence (e.g., soil carbon storage influences greenhouse gas concentrations globally while habitat for soil arthropods is quite local), each of the functions have practical relevance for those living in the U.S. Upper Midwest where surface and groundwater pollution, flooding, soil erosion, and plummeting biodiversity undermine human welfare and well being (Werling et al., 2014; Hussain et al., 2019; Antolini et al., 2020; Bendorf et al., 2021; Borchardt et al., 2021; Burch et al., 2021; Raff and Meyer, 2022; Wisconsin Groundwater Coordinating Council Report to the Legislature, 2022).

Solutions to this multifaceted dilemma require holistic approaches that consider land management effects on ecosystem functions that underpin ecosystem services provided by farms and, more broadly, the landscapes or regions in which they are nested (Strauser et al., 2022). Holistic solutions are required because of the complexity and connectedness of these landscapes, where focusing on a single dimension typically exacerbates problems in others. We must understand and develop systems that solve for multiple variables simultaneously. While the currently dominant form of agriculture is immensely productive, it is also the world's leading driver of environmental change (Foley et al., 2005, 2011). Fortunately, agricultural approaches exist that have the potential to help stabilize global change as we move further into the Anthropocene (Campbell et al., 2017; Rockström et al., 2017).

The dominant agricultural system of the U.S. Upper Midwest is based on monocultures of corn and soybeans grown to feed mostly confined livestock. This system is incentivized by rewarding farmers almost exclusively for more production (Jordan et al., 2018), which comes at the expense of other functions critical to ecological and societal well being [e.g., purification of water, stabilization of soil, regulation of infectious disease, provisioning of wildlife habitat; Alexander et al. (2008), Wepking et al. (2017), Christianson et al. (2018)]. Currently, critical ecosystem functions and the services that they underpin are not properly valued, so their costs are externalized; borne by society as a whole (Suparak Gibson, 2022). An alternative agricultural system, based on perennial grassland, is possible (Jackson, 2022) but requires society to pivot away from the status quo toward a system that rewards a range of ecosystem services. Currently, farmers' individual decisions to participate in the corn and soybean dominated agricultural system are driven by constructed narratives around productivism and maximizing food production (Burton, 2004; McGuire et al., 2013), aesthetic preferences about the 'neatness'

and perceived care of the landscape (Nassauer, 1988), and definitions of *place* at regional scales (Strauser et al., 2022). To incentivize agricultural systems that simultaneously provide multiple ecosystem services to farmers and society, we must understand tradeoffs and synergies in ecosystem functions provided by alternative cropping systems.

To further this understanding we used Agro-IBIS [Integrated Biosphere Simulator; Kucharik et al. (2000), Kucharik (2003), Kucharik and Brye (2003)] to represent a variety of biophysical and biogeochemical processes. These processes included nitrate leaching and phosphorus loss as indicators of nutrient retention and drainage and evapotranspiration as indicators of water retention. These indicators were then simulated across vegetation types associated with three dairy cropping systems (see Methods). In addition to controlling the type of vegetation grown in the model, different simulated land-use decisions can be made regarding fertilizer and manure applications and crop rotations.

Recent work with Agro-IBIS has focused on meeting targeted policy goals with increasing grassland cover. In particular, Campbell et al. (2022) estimated the amount of perennial grassland cover needed to meet water quality goals within the Yahara River Watershed in southern Wisconsin to the year 2070. Similarly, water quality outcomes were assessed with simulations designed to achieve the goals of the Renewable Fuel Standard providing insight into the beneficial water quality effects of improved miscanthus and switchgrass cover (Ferin et al., 2021). Other models, such as DairyMod, APSIM, and DayCent have been used to simulate soil N mineralization and pasture growth (Bilotto et al., 2021), and DairyMod in particular has been instrumental in simulating ammonia volatilization in pastures (Smith et al., 2020), but with a specific focus on Australia and New Zealand where DairyMod was calibrated (Johnson et al., 2008).

Within this stream of the literature, there is no work addressing regional variation in a broad suite of ecosystem services across the U.S. Upper Midwest. In particular, we contribute to the literature by including water quantity in addition to water quality, and matching regional variation within these environmental outcomes to food-energy production outcomes. By simulating these ecosystem functions across three common land cover-land use scenarios (described below), we are better able to anticipate how a wider range of ecosystem services might vary with management.

We examined ecosystem functions under three types of land cover associated with cropping systems typical of the U.S. Upper Midwest. We gathered site-specific data (previous cropping practices, soil type, slope, aspect) from five Wisconsin farms—two in the “Ridge & Valley” region of southwest Wisconsin (Vernon County) and three in the “Cloverbelt” region of central Wisconsin (Marathon County). While both of these regions have a strong agricultural focus, they vary in their topography as well as their edaphic and environmental characteristics.

TABLE 1 General descriptions of the five farms used in this analysis.

Farm	Location	Size (acres)	Primary soil type	Soil P	Soil pH	Slope	Slope length	Primary land use
1	Vernon County, WI, USA	145	Pepin	20	6.70	17.6	110	Grazing
2*	Vernon County, WI, USA	40	Arenzville	na	na	2	250	Grazing
3	Marathon County, WI, USA	483	Loyal	28.0	6.59	3.97	284	Corn-soy with some no till
4	Marathon County, WI, USA	407	Loyal	20.4	6.60	3.84	294	Grazing with some corn-soy
5	Marathon County, WI, USA	280	Loyal	30.9	7.05	3.67	292	Dairy rotation

Characteristics represent the acres that the operator of each farm uses for agricultural production. *Despite the listed slope, Farm 2 and its productive acreage are located at the base of a steep (>50°) ridge and is located in a highly flood-prone area. All soil types listed are silt loam.

We used simulation models and literature estimates to predict outcomes of ecosystem functions under three land cover-land use scenarios—continuous corn, corn-soy rotation, and grassland. Ecosystem functions included estimates of food-energy production as well as water and nutrient dynamics. We expected that increasing perennial cover would improve a range of ecosystem services with potential tradeoffs in food-energy (meat and dairy) production.

Methods

Study region

The Cloverbelt and Ridge & Valley regions of Wisconsin are both known for dairy, beef, and crop production and each region has a strong identity and ethos associated with agriculture and the environment (Supplementary Figure S1, Supplementary Tables S2–S4). In each region annual grain crops (mainly corn and soybeans) are grown on most agricultural land to feed confined livestock whose genetic improvements and concentration in space continue to increase production and efficiency when the latter is assessed as calories produced per input.

While similar in many ways, these regions are quite different. The Cloverbelt is relatively flat with moderate- to poorly-drained soils where local climate and edaphic conditions are favorable to clover production in pastures, giving the agriculture of the region a distinctive *Dairyland* signature. The Ridge & Valley region is characterized by silty, erodible soils on highly dissected topography that make the region flood prone. Annual average precipitation and temperature for the last 10 years (2010 through 2020) were 107.2 cm and 7.1°C for Vernon County (Ridge & Valley) and 93.7 cm and 6.4°C for Marathon County [Cloverbelt; PRISM Climate Group, Oregon State University (2022)]. This variation between farms and regions provides a representation of a significant part of farming in the U.S. Upper Midwest (see Tables S2–S4 for additional details).

Data collection

We gathered crop histories and soil tests from reports submitted by the operators of each of the five farms (Table 1). To protect the privacy of these farm operators, we retain the confidentiality of each farm and report only overall summaries of each. Data included farm size, individual field delineation, soil types, soil phosphorus, slope, slope length, and previous land use.

The physical characteristics of these farms are highly variable (Table 1) and they currently use a mix of row-crop rotations, tillage and no till, and managed grazing. Importantly, especially for water quality, soil P levels varied considerably among farms. Also, the slope and slope length of each farm likely affect water quality in different ways. Consistent with regional descriptions above, Cloverbelt farms were larger, flatter, and participated primarily in more row-crop agriculture for dairy production while Ridge & Valley farms were smaller, on steeper slopes, and used more pasture. We reported land in agricultural production only, not including some forested land, which for one farm was steep. Therefore, while the slope was relatively shallow for Farm 2 the adjacent forested land was steep and listed as highly flood prone.

Simulation models

Agro-IBIS is a spatially explicit agroecosystem and land surface model that simulates the movement of water, energy, momentum, carbon, nitrogen, and phosphorus, in both natural and managed ecosystems. The structure of Agro-IBIS has been described in detail (Kucharik et al., 2000; Kucharik, 2003; Kucharik and Brye, 2003; Motew et al., 2017) and many components and output variables of the model (e.g., crop yield, net primary productivity (NPP), net ecosystem exchange (NEE), evapotranspiration and drainage, nitrate leaching, soil temperature and moisture) have been validated across a range of ecosystems at various spatial and temporal scales (Kucharik et al., 2000, 2006; Kucharik, 2003; Kucharik and Brye, 2003; Kucharik and Twine, 2007; Motew and Kucharik, 2013;

Soylu et al., 2014; Zipper et al., 2015; Motew et al., 2017). Agro-IBIS was integrated with the variably saturated soil water flow model HYDRUS-1D to enable simulation of groundwater-vegetation interactions (Soylu et al., 2014), and P cycling and dynamics were recently added based on SurPhos, a state-of-the-art dissolved P loss model for agricultural systems receiving manure (Vadas et al., 2004, 2005, 2007; Motew et al., 2017). The P module features P application, transformation, and loss of dissolved P to runoff; in-soil cycling of organic and inorganic forms of P; and loss of particulate-bound P with erosion (Motew et al., 2017).

Before running the scenarios of land cover-land use, a long-term model spin-up run was executed from 1650 to 1961 to achieve a steady-state equilibrium in soil biogeochemical cycling that reflects changes in land use and build-up of soil organic C and N pools (Donner and Kucharik, 2003). Agro-IBIS model simulations were executed using a 60-min time-step on a 1 x 1-km regularly spaced grid; the model uses SSURGO soil textural data to delineate dominant soil texture and soil physical properties for each grid cell and soil layer, and daily weather data (air temperature, precipitation, relative humidity, solar radiation, and wind speed) from the gridMET (gridMET, 2013) that was interpolated from 4- to 1-km spatial resolution. Agro-IBIS uses statistical models to interpolate daily weather variables to the hourly time-step (Kucharik et al., 2000). During the model simulation period from 1650 through 1978, a random draw of weather years was taken from the actual data time-series of 1979 through 2016; simulation years from 1979 through 2016 represent the actual weather time-series from gridMET. Nutrient inputs (inorganic fertilizer and manure) originate from a spatiotemporal database of linked agricultural, environmental, and economic data (Lark et al., 2022).

We simulated three different agricultural scenarios or vegetation types: continuous corn, corn-soybean rotation, and generalized C4-dominant perennial grassland for five locations described in Table 1. Continuous corn and corn-years in the corn-soybean rotation received between 91.8 and 180 kg N ha⁻¹ yr⁻¹ and 9 and 22 kg P ha⁻¹ yr⁻¹ based on historical fertilization for that location, which varied by year; soybean and grass did not receive any N and P fertilizer because neither receive N and P fertilization as part of typical grass or soy production. Soil was tilled in continuous corn and corn-soybean rotation was tilled before planting while the grass was never tilled.

For the corn-soy rotation, we ran two scenarios starting with both corn and soybean and then aggregated the output. Annual estimates of nitrate leaching, phosphorus loss (including both sediment and dissolved phosphorus), evapotranspiration, and groundwater recharge (drainage) were gathered after running those different vegetation scenarios from 1961 through 2016. We then filtered out the first 18 years of data (keeping the 38 years from 1979 through 2016), as the model output took approximately 10 years after a restart simulation (the restart year

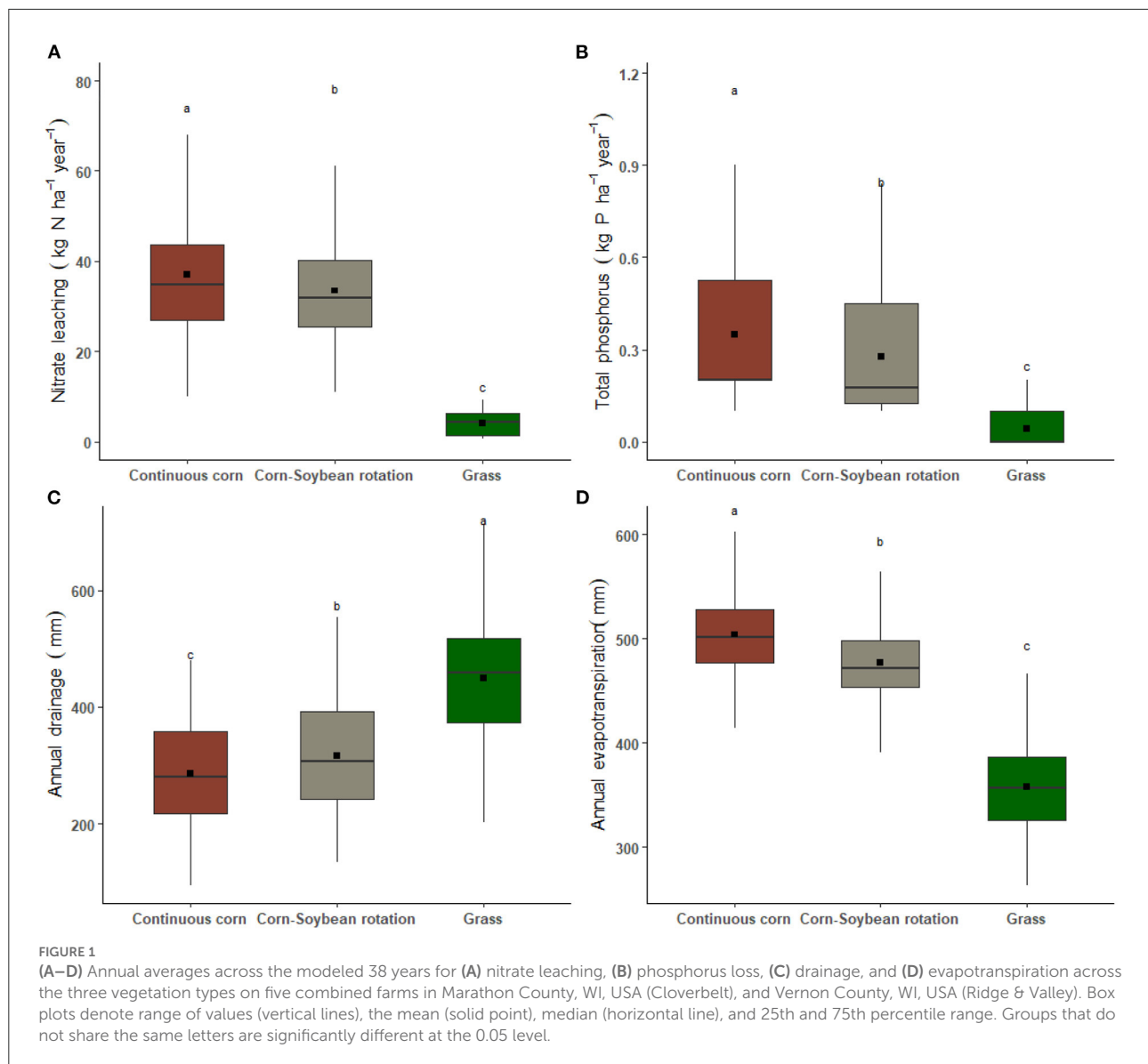
was 1961 for each scenario) to reach equilibrium, and because the time-series of actual weather begins in 1979.

After gathering the output data, we used linear mixed effects models (one for each of the above dependent variables) using the *nlme* package in R (Pinheiro et al., 2022) to limit the effects of vegetation type and year (fixed effects) and farm nested within region (random effects) to account for site/geographical variation.

Food-energy calculations

We examined another important ecosystem function, agricultural production, for each of the three scenarios by calculating their food-energy output from harvested biomass. We used the historical cropping data for each farm to generate average farm-level output (bushels ha⁻¹). We then used this past output and created a “target” output to use in the counterfactual simulations. As a specific example, the average farm-level production at Farm 3 during the pre-simulation period was 354.1 bushels ha⁻¹ yr⁻¹ (9.6 Mg ha⁻¹ yr⁻¹) of corn grain and 49.8 bushels ha⁻¹ yr⁻¹ (3.3 Mg ha⁻¹ yr⁻¹) of soybeans. We therefore simulated food-energy production for Farm 3 using these values as output, while averaging the output over two years for the corn-soy rotation. We did not have farm-specific yield data for farms that did not practice a cropping system during the years for which we obtained observational data, so for these farms we used USDA Census of Agriculture average yield maps for that county and set the target yield at the value given in the map (USDA—National Agricultural Statistics Service, 2022). For pastures without yield data, we assumed an average height of the grass of 63.5 cm and multiplied that by an estimated harvest of 326 kg DM ha⁻¹ (Barnhart, 1998), which is equivalent to 8.4 Mg ha⁻¹ yr⁻¹ of harvested dry matter. We then converted these target yields for each farm to food-energy using the process described in Sanford et al. (2021). Like Sanford et al. (2021), our representative farms and cropping systems represent agricultural production in the U.S. Upper Midwest (WI), which is a major producer of dairy products. We therefore examined food-energy in the form of milk and dairy beef (Gcal ha⁻¹ yr⁻¹). Briefly, Sanford et al. (2021) make their conversion from harvested yield to food-energy using the following steps:

1. Convert volume yield (bushels ac⁻¹) to mass yield (Mg ha⁻¹), while assuming that 79.2% of soybean grain results in soybean meal and 18.7% results in soybean oil. We used national data from 1980 to 2016 to estimate these percentages (USDA-ERS, 2018).
2. Convert dry matter yield to total digestible nutrients (TDN) using mean nutrient content values from Dairy One Cooperative (Feed Composition Library | Dairy One, 2022): 88% for corn, 80% for soybeans, and 60% for grass.



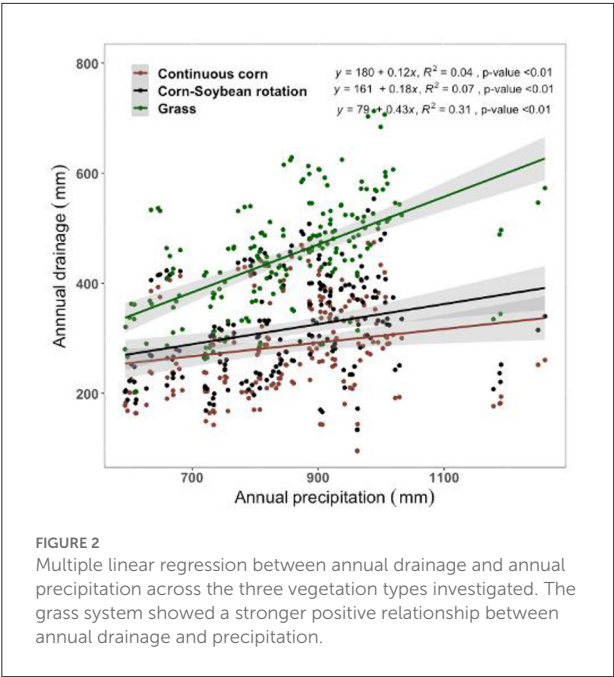
3. Convert TDN yield to milk and dairy beef food-energy using conversions from Peters et al. (2014) and USDA (USDA—FoodData Central, 2022). We used the same process as Sanford et al. (2021) to calculate a conversion factor of 1.04 Gcal Mg⁻¹ for milk (1,042 kcal kg⁻¹ TDN) and 0.37 Gcal Mg⁻¹ for dairy beef (366 kcal kg⁻¹ TDN).

Finally, we compared observable farm factors and ecosystem functions by assessing correlations among these variates.

Results

Across the five farms and two regions, vegetation type significantly affected nitrate leaching (Figure 1A, Supplementary Table S1, Supplementary Figure S2). This

effect was driven by the low average level of nitrate leaching over the 38 years analyzed in the grass system ($4.1 \pm 0.2 \text{ kg ha}^{-1}$) compared to continuous corn ($39.9 \pm 1.0 \text{ kg ha}^{-1}$) and the corn-soy rotation ($33.5 \pm 0.8 \text{ kg ha}^{-1}$). Both continuous corn and the corn-soy rotation leached significantly more than the grass system in a pairwise comparison ($P < 0.001$ and $P < 0.001$, respectively), but continuous corn also leached significantly more than corn-soy ($P < 0.001$). Across the three vegetation types nitrate leaching generally increased over time, on average increasing $0.4 \text{ kg ha}^{-1} \text{ yr}^{-1}$ [$P < 0.001$ (Supplementary Figure S2)]. Annual variation in nitrate leaching was driven by precipitation and management differences (Supplementary Figure S3). When considered across the five farms and two regions separately within the grass vegetation, nitrate leaching appeared to vary by region with Marathon County ($6.0 \pm 0.1 \text{ kg ha}^{-1}$) exhibiting



greater leaching than Vernon County [$1.3 \pm 0.04 \text{ kg ha}^{-1}$ (Supplementary Figure S4)].

Continuous corn was found to have the highest level of phosphorus loss ($0.35 \pm 0.02 \text{ kg ha}^{-1}$) followed by the corn-soy rotation ($0.28 \pm 0.02 \text{ kg ha}^{-1}$); grass had the lowest level of phosphorus loss ($0.042 \pm 0.004 \text{ kg ha}^{-1}$) across the five farms and two counties investigated (Figure 1B, Supplementary Table S1). Phosphorus runoff with continuous corn vegetation was found to be significantly greater than both the corn-soy rotation ($P < 0.001$) and the grass vegetation types ($P < 0.001$). The grass vegetation type was found to exhibit significantly lower phosphorus than the corn-soy rotation vegetation type ($P < 0.001$). Phosphorus tended to decrease over time as well [$P < 0.001$ (Supplementary Figure S2)]. However, this decline over time appears to be driven by the row-crop vegetation types; the grass vegetation type held relatively steady over the course of the 38 years analyzed (Supplementary Figure S2).

The amount of annual drainage, or recharge to groundwater, was shown to vary by vegetation type (Figure 1C, Supplementary Table S1). The grass system showed the highest annual drainage ($449 \pm 7.4 \text{ mm yr}^{-1}$), significantly greater than both the continuous corn ($286 \pm 6.3 \text{ mm yr}^{-1}$; $P < 0.001$), and corn-soy rotation ($317 \pm 6.4 \text{ mm yr}^{-1}$; $P < 0.001$). The corn-soy rotation was shown to have significantly greater drainage than the continuous corn vegetation ($P < 0.001$). Over the course of the 38 years analyzed, annual drainage shows a general decrease over time, on average 2.23-mm lower annually ($P < 0.001$). In addition, grass vegetation was shown to have a stronger positive relationship than the other two vegetation types between annual drainage and annual precipitation (Figure 2).

TABLE 2 Average \pm standard errors for food-energy production across three vegetation types.

Vegetation	Milk (Gcal ha ⁻¹ yr ⁻¹)	Dairy beef (Gcal ha ⁻¹ yr ⁻¹)
Continuous corn	8.45 \pm 0.32	2.97 \pm 0.11
Corn-Soybean rotation	5.36 \pm 0.19	1.88 \pm 0.07
Grass	5.27 \pm 0.11	1.85 \pm 0.04

Values are calculated using the methodology of Sanford et al. (2021) and represent means and standard errors for the two counterfactual years given past production on each farm.

Evapotranspiration trends were opposite of drainage (Figure 1D, Supplementary Table S1). Grass systems had the lowest evapotranspiration ($358 \pm 3.1 \text{ mm yr}^{-1}$; $P < 0.001$), corn-soy systems had the second-highest evapotranspiration ($477 \pm 2.5 \text{ mm yr}^{-1}$; $P < 0.001$), and continuous corn had the highest ($504 \pm 2.7 \text{ mm yr}^{-1}$; $P < 0.001$). Generally, evapotranspiration was higher in the Ridge & Valley than the Cloverbelt (Supplementary Figure S5). In addition, across all vegetation types, the annual average ET trend was 0.91 mm yr^{-1} ($P < 0.001$).

Finally, the vegetation types differed in their levels of food-energy production. Across all cropping systems, the same level of harvested dry matter produced higher amounts of food-energy in the form of milk compared to dairy beef (Table 2). Continuous corn had the highest food-energy output, producing over 3 Gcal ha⁻¹ yr⁻¹ more milk energy and over 1 Gcal ha⁻¹ yr⁻¹ more beef energy ($\sim 60\%$ for each) than the corn-soy rotation and grass ($P < 0.001$). However, no significant difference in food-energy production was observed between corn-soy rotation and grass for both milk ($P = 0.70$) and dairy beef ($P = 0.71$) output.

Discussion

Nitrate leaching was greatly reduced under grass vegetation compared to both continuous corn and corn-soy rotation because no manure or fertilizer was added to grass vegetation, which aligns with empirical field studies. Under most perennial grass bioenergy and grazed systems, nitrate leaching is much lower than corn-based cropping systems (Hussain et al., 2019; Jackson, 2020), differences that can lead to significant disparities in water quality for rural regions, where nitrate leaching contributes to impaired health and infant mortality (Knobeloch et al., 2013). Further, nitrate leaching from common corn-based systems contributes to eutrophication and consequently impaired rivers, lakes, and oceans (Orth et al., 2006; Liu et al., 2022). These waterways arguably are more impaired from phosphorus runoff, with nitrogen potentially working in concert with phosphorus to induce further eutrophication (Dodds and Smith, 2016; Schindler et al., 2016). As mentioned in the introduction, there is promise for reducing eutrophication and

meeting established water quality goals by reducing nutrients in watersheds and increasing perennial cover as shown by [Campbell et al. \(2022\)](#).

From a water quantity perspective, grassland promoted higher water drainage (i.e., more infiltration and less runoff) and lower evapotranspiration than the other systems. Increasing evapotranspiration reduces local temperatures through increasing latent heat flux, and has been shown to mitigate increased temperatures, e.g., the urban heat island effect ([Qiu et al., 2013](#)). In an agricultural context, however, higher evapotranspiration (e.g., continuous corn, corn-soy rotation) is linked to higher water demand and consequently higher irrigation rates or water demand for crops. Systems that promote higher evapotranspiration and lower recharge have a direct impact on the volume of surface water bodies. In the case of irrigation, this has been shown to deplete groundwater levels, especially as climate change increases evapotranspiration rates over time ([Condon et al., 2020](#)). While only 1.5 and 0.4% of agricultural land in Marathon County and Vernon County are irrigated, this is an important consideration in drier areas. While we found lower evapotranspiration rates in grasses than continuous corn or the corn-soy rotation, some grasslands can be comparable to corn-based systems ([Abraha et al., 2020](#)). Agricultural impacts on groundwater depend in large part on irrigation (which was zero in our modeled grassland) and drainage (which was highest in our modeled grassland) back into groundwater systems. While beyond the scope of our modeling study, many studies have also shown the potential of perennial grassland to reduce runoff and flood risk downstream because of its ability to enhance infiltration ([Jackson and Keeney, 2010](#); [Schilling et al., 2014](#)).

Balancing our current emphasis on agricultural production with other ecosystem services is critical. The continuous corn system produced more food-energy than the corn-soy and grass systems, which is consistent with previous work ([Peters et al., 2014](#)). However, while the grass and corn-soy systems produced similar output from a food-energy perspective, the grass system outperformed the corn-soy rotation on all other ecosystem metrics. While recent research has shown that the current amount of beef raised within the U.S. could be raised entirely on grass—and without adding acreage not already in some form of agricultural production ([Jackson, 2022](#))—more work is needed to better understand the ramifications of transformative changes to our agricultural landscape. Spatially explicit research that can show where various forms of agriculture can either do the least damage, or conversely, can promote the most beneficial ecosystem services, is a clear need in improving our understanding and decision making around agricultural production.

From a dairy perspective, milk yields dropped when cows were fed from grassland exclusively ([Jackson, 2022](#)). However, this drop in milk production with the grass-based system can be countered by a dramatic drop in production costs ([Dartt et al., 1999](#); [Kriegel, 2005](#); [Hanson et al., 2013](#)) making grass-based

dairies economically competitive with confinement dairies; work that shows that grass-based systems can outperform others from a multifunctionality perspective. Other work shows that from a *true-cost accounting* perspective, grass-based farms provide much more value to society than what are considered conventional farms, and are dramatically undervalued ([Suparak Gibson, 2022](#)). Instances such as this require a framework to reward farmers for the societal good produced, whether it be from a policy perspective or some other structure ([Rissman et al., in this volume](#)).

While we focused on agricultural production from an energetics perspective to broadly compare the vegetation types in question, there is more to food than energy. Nutritional profiles are an important consideration to include in future analysis of tradeoffs among ecosystem services. Research on this topic shows that grassfed livestock production improves both animal health and the nutritional profile of livestock products compared to conventionally raised livestock ([van Vliet et al., 2021b](#)). A key driver of this improvement in nutritional profile was the biodiversity of the plants consumed by grassfed livestock, suggesting that the promotion of biodiversity is strongly linked with human health ([Provenza et al., 2021](#); [van Vliet et al., 2021a](#)).

Our model did not include a grazing module that mimicked disturbance-plant growth dynamics, nutrient uptake and NPP, which would likely be stimulated to an even greater degree under well-managed grazing resulting in improvement in most ecosystem functions. Current work is adding grazing and cover crop modules to further explore continuous living cover in agroecosystems. These types of modeling advances, integrated with the other capabilities of Agro-IBIS, will allow scientists to develop advanced decision support tools (DSTs) that contain model output data from many scenarios representing the potential impacts of a changing climate and land management on ecosystem services. The goal is to have crop consultants, land managers, farmers and other end users use DSTs to guide future agroecosystem management decision-making to meet sustainable development goals for humanity.

Conclusions

With the exception of food-energy yield, all the ecosystem functions we explored were improved under the grassland-based system compared to annual grain crops. If agricultural policy continues to reward yield exclusively, it will be difficult to transition to more multifunctional agricultural systems. Continuous corn yielded more food-energy than corn-soy rotation and perennial grassland, but a significant tradeoff was observed: this system had the poorest performance across all other ecosystem functions – nitrate leaching, phosphorus loss, drainage, and evapotranspiration. While the corn-soy rotation provided slightly better outcomes than continuous corn (except for yield), it was inferior to perennial grassland

for most outcomes and not significantly different in food calorie yield. A more balanced delivery of ecosystem functions underpinning critical ecosystem services will require more reliance on perennial grassland for livestock production.

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

Author contributions

CW, DS, HM, AO, ZR, EB, CK, CG, and RJ designed the research. DS, HM, AO, and ZR performed the research and analyzed the data. CW wrote the manuscript. CW, DS, HM, AO, ZR, EB, CG, and RJ edited the manuscript. All authors contributed to the article and approved the submitted version.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

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References

- Abraham, M., Chen, J., Hamilton, S. K., and Robertson, G. P. (2020). Long term evapotranspiration rates for rainfed corn versus perennial bioenergy crops in a mesic landscape. *Hydrol. Process.* 34, 810–822. doi: 10.1002/hyp.13630
- Alexander, R. B., Smith, R. A., Schwarz, G. E., Boyer, E. W., Nolan, J. V., and Brakebill, J. W. (2008). Differences in phosphorus and nitrogen delivery to The Gulf of Mexico from the Mississippi River Basin. *Environ. Sci. Technol.* 42, 822–830. doi: 10.1021/es0716103
- Amundson, R., Berhe, A. A., Hopmans, J. W., Olson, C., Szein, A. E., and Sparks, D. L. (2015). Soil and human security in the twenty-first century. *Science* 348, 1261071. doi: 10.1126/science.1261071
- Antolini, F., Tate, E., Dalzell, B., Young, N., Johnson, K., and Hawthorne, P. L. (2020). Flood risk reduction from agricultural best management practices. *JAWRA J. Am. Water Resour. Assoc.* 56, 161–179. doi: 10.1111/1752-1688.12812
- Baker, J. M., Albrecht, K. A., Feyereisen, G. W., and Gamble, J. D. (2022). A perennial living mulch substantially increases infiltration in row crop systems. *J. Soil Water Conserv.* 77, 212–220. doi: 10.2489/jswc.2022.00080
- Barnhart, S. K. (1998). *Estimating Available Pasture Forage*. Available online at: <http://www.extension.iastate.edu/Publications/PM1758.pdf>.
- Basche, A. D., and DeLonge, M. S. (2019). Comparing infiltration rates in soils managed with conventional and alternative farming methods: a meta-analysis. *PLoS ONE* 14, e0215702. doi: 10.1371/journal.pone.0215702
- Bendorf, J., Hubbard, S., Kucharik, C. J., and VanLoocke, A. (2021). Rapid changes in agricultural land use and hydrology in the driftless region. *Agrosyst. Geosci. Environ.* 4, e20214. doi: 10.1002/agg2.20214
- Bilotto, F., Harrison, M. T., Migliorati, M. D. A., Christie, K. M., Rowlings, D. W., Grace, P. R., et al. (2021). Can seasonal soil N mineralisation trends be leveraged to enhance pasture growth? *Sci. Total Environ.* 772, 145031. doi: 10.1016/j.scitotenv.2021.145031
- Borchardt, M. A., Stokdyk, J. P., Kieke, B. A., Muldoon, M. A., Spencer, S. K., Firnstahl, A. D., et al. (2021). Sources and risk factors for nitrate and microbial contamination of private household wells in the fractured dolomite aquifer of Northeastern Wisconsin. *Environ. Health Perspect.* 129, 067004. doi: 10.1289/EHP7813
- Burch, T. R., Stokdyk, J. P., Spencer, S. K., Kieke, B. A., Firnstahl, A. D., Muldoon, M. A., et al. (2021). Quantitative microbial risk assessment for contaminated private wells in the fractured dolomite aquifer of Kewaunee County, Wisconsin. *Environ. Health Perspect.* 129, 067003. doi: 10.1289/EHP7815
- Burton, R. J. F. (2004). Seeing through the “good farmer’s” eyes: towards developing an understanding of the social symbolic value of “productivist” behaviour. *Sociol. Rural.* 44, 195–215. doi: 10.1111/j.1467-9523.2004.00270.x
- Campbell, B. M., Beare, D. J., Bennett, E. M., Hall-Spencer, J. M., Ingram, J. S. I., Jaramillo, F., et al. (2017). Agriculture production as a major driver of the Earth system exceeding planetary boundaries. *Ecol. Soc.* 22, art8. doi: 10.5751/ES-09595-220408
- Campbell, T. A., Booth, E. G., Gratton, C., Jackson, R. D., and Kucharik, C. J. (2022). Agricultural landscape transformation needed to meet water quality goals in the Yahara River watershed of Southern Wisconsin. *Ecosystems* 25, 507–525. doi: 10.1007/s10021-021-00668-y
- Christianson, R., Christianson, L., Wong, C., Helmers, M., McIsaac, G., Mulla, D., et al. (2018). Beyond the nutrient strategies: common ground to accelerate agricultural water quality improvement in the upper Midwest. *J. Environ. Manag.* 206, 1072–1080. doi: 10.1016/j.jenvman.2017.11.051
- Condon, L. E., Atchley, A. L., and Maxwell, R. M. (2020). Evapotranspiration depletes groundwater under warming over the contiguous United States. *Nat. Commun.* 11, 873. doi: 10.1038/s41467-020-14688-0

- Dartt, B. A., Lloyd, J. W., Radke, B. R., Black, J. R., and Kaneene, J. B. (1999). A comparison of profitability and economic efficiencies between management-intensive grazing and conventionally managed dairies in Michigan. *J. Dairy Sci.* 82, 2412–2420. doi: 10.3168/jds.S0022-0302(99)75492-5
- Dodds, W., and Smith, V. (2016). Nitrogen, phosphorus, and eutrophication in streams. *Inland Waters* 6, 155–164. doi: 10.5268/IW-6.2.909
- Donner, S. D., and Kucharik, C. J. (2003). Evaluating the impacts of land management and climate variability on crop production and nitrate export across the Upper Mississippi Basin. *Glob. Biogeochem. Cycles* 17, n/a–n/a. doi: 10.1029/2001GB001808
- Feed Composition Library | Dairy One (2022). *Dairy One Feed Compos. Libr.* Available online at: <https://dairystone.com/services/forage-laboratory-services/feed-composition-library/> (accessed June 24, 2022).
- Ferin, K. M., Chen, L., Zhong, J., Acquah, S., Heaton, E. A., Khanna, M., et al. (2021). Water quality effects of economically viable land use change in the Mississippi River Basin under the renewable fuel standard. *Environ. Sci. Technol.* 55, 1566–1575. doi: 10.1021/acs.est.0c04358
- Foley, J. A., DeFries, R., Asner, G. P., Barford, C., Bonan, G., Carpenter, S. R., et al. (2005). Global consequences of land use. *Science* 309, 570–574. doi: 10.1126/science.1111772
- Foley, J. A., Ramankutty, N., Brauman, K. A., Cassidy, E. S., Gerber, J. S., Johnston, M., et al. (2011). Solutions for a cultivated planet. *Nature* 478, 337–342. doi: 10.1038/nature10452
- Godfray, H. C. J., Beddington, J. R., Crute, I. R., Haddad, L., Lawrence, D., Muir, J. F., et al. (2010). Food security: the challenge of feeding 9 billion people. *Science* 327, 812–818. doi: 10.1126/science.1185383
- gridMET (2013). *Climatol. Lab.* Available online at: <https://www.climatologylab.org/gridmet.html> (accessed June 30, 2022).
- Guillaume, T., Makowski, D., Libohova, Z., Elfouki, S., Fontana, M., Leifeld, J., et al. (2022). Carbon storage in agricultural topsoils and subsoils is promoted by including temporary grasslands into the crop rotation. *Geoderma* 422, 115937. doi: 10.1016/j.geoderma.2022.115937
- Hanson, J. C., Johnson, D. M., Lichtenberg, E., and Minegishi, K. (2013). Competitiveness of management-intensive grazing dairies in the mid-Atlantic region from 1995 to 2009. *J. Dairy Sci.* 96, 1894–1904. doi: 10.3168/jds.2011-5234
- Hussain, M. Z., Bhardwaj, A. K., Basso, B., Robertson, G. P., and Hamilton, S. K. (2019). Nitrate leaching from continuous corn, Perennial Grasses, and poplar in the US Midwest. *J. Environ. Qual.* 48, 1849–1855. doi: 10.2134/jeq2019.04.0156
- Jackson, L., and Keeney, D. (2010). “Perennial farming systems that resist flooding,” in *A Watershed Year: Anatomy of the Iowa Floods of 2008 A Bur Oak Book*. (Iowa City: University of Iowa Press).
- Jackson, R. D. (2020). Soil nitrate leaching under grazed cool-season grass pastures of the North Central US. *J. Sci. Food Agric.* 100, 5307–5312. doi: 10.1002/jsfa.10571
- Jackson, R. D. (2022). Grazed perennial grasslands can match current beef production while contributing to climate mitigation and adaptation. *Agric. Environ. Lett.* 7. doi: 10.1002/acl2.20059
- Johnson, I. R., Chapman, D. F., Snow, V. O., Eckard, R. J., Parsons, A. J., Lambert, M. G., et al. (2008). DairyMod and EcoMod: biophysical pasture-simulation models for Australia and New Zealand. *Aust. J. Exp. Agric.* 48, 621. doi: 10.1071/EA07133
- Jordan, N. R., Mulla, D. J., Slotterback, C., Runck, B., and Hays, C. (2018). Multifunctional agricultural watersheds for climate adaptation in Midwest USA: commentary. *Renew. Agric. Food Syst.* 33, 292–296. doi: 10.1017/S1742170517000655
- Kimoto, C., DeBano, S. J., Thorp, R. W., Taylor, R. V., Schmalz, H., DelCurto, T., et al. (2012). Short-term responses of native bees to livestock and implications for managing ecosystem services in grasslands. *Ecosphere* 3, art88. doi: 10.1890/ES12-00118.1
- Knobeloch, L., Gorski, P., Christenson, M., and Anderson, H. (2013). Private drinking water quality in rural Wisconsin. *J. Environ. Health* 75, 7.
- Kremen, C., and Merenlender, A. M. (2018). Landscapes that work for biodiversity and people. *Science* 362, eaau6020. doi: 10.1126/science.aau6020
- Kriegl, T. (2005). *Pastures of Plenty: Financial Performance of Wisconsin Grazing Dairy Farms*. University of Wisconsin-Madison.
- Kucharik, C. J. (2003). Evaluation of a process-based agro-ecosystem model (Agro-IBIS) across the U.S. Corn belt: simulations of the interannual variability in Maize Yield. *Earth Interact.* 7, 1–33. doi: 10.1175/1087-3562(2003)007<0001:EOAPAM>2.0.CO;2
- Kucharik, C. J., Barford, C. C., Maayar, M. E., Wofsy, S. C., Monson, R. K., and Baldocchi, D. D. (2006). A multiyear evaluation of a dynamic global vegetation model at three AmeriFlux forest sites: vegetation structure, phenology, soil temperature, and CO₂ and H₂O vapor exchange. *Ecol. Model.* 196, 1–31. doi: 10.1016/j.ecolmodel.2005.11.031
- Kucharik, C. J., and Brye, K. R. (2003). Integrated biosphere simulator (IBIS) yield and nitrate loss predictions for wisconsin maize receiving varied amounts of nitrogen fertilizer. *J. Environ. Qual.* 32, 247–268. doi: 10.2134/jeq2003.2470
- Kucharik, C. J., Foley, J. A., Delire, C., Fisher, V. A., Coe, M. T., Lenters, J. D., et al. (2000). Testing the performance of a dynamic global ecosystem model: water balance, carbon balance, and vegetation structure. *Glob. Biogeochem. Cycles* 14, 795–825. doi: 10.1029/1999GB001138
- Kucharik, C. J., and Twine, T. E. (2007). Residue, respiration, and residuals: evaluation of a dynamic agroecosystem model using eddy flux measurements and biometric data. *Agric. For. Meteorol.* 146, 134–158. doi: 10.1016/j.agrformet.2007.05.011
- Lark, T. J., Hendricks, N. P., Smith, A., Pates, N., Spawn-Lee, S. A., Bougie, M., et al. (2022). Environmental outcomes of the US renewable fuel standard. *Proc. Natl. Acad. Sci.* 119, e2101084119. doi: 10.1073/pnas.2101084119
- Lesk, C., Rowhani, P., and Ramankutty, N. (2016). Influence of extreme weather disasters on global crop production. *Nature* 529, 84–87. doi: 10.1038/nature16467
- Lioutas, E. D., and Charatsari, C. (2021). Enhancing the ability of agriculture to cope with major crises or disasters: what the experience of COVID-19 teaches us. *Agric. Syst.* 187, 103023. doi: 10.1016/j.agry.2020.103023
- Liu, J., Bowling, L., Kucharik, C., Jame, S., Baldos, U., Jarvis, L., et al. (2022). Multi-scale analysis of nitrogen loss mitigation in the US Corn Belt. *arXiv* doi: 10.48550/ARXIV.2206.07596
- McGuire, J., Morton, L. W., and Cast, A. D. (2013). Reconstructing the good farmer identity: shifts in farmer identities and farm management practices to improve water quality. *Agric. Hum. Values* 30, 57–69. doi: 10.1007/s10460-012-9381-y
- Montgomery, D. R. (2007). Soil erosion and agricultural sustainability. *Proc. Natl. Acad. Sci.* 104, 13268–13272. doi: 10.1073/pnas.0611508104
- Motew, M., Chen, X., Booth, E. G., Carpenter, S. R., Pinkas, P., Zipper, S. C., et al. (2017). The Influence of legacy P on lake water quality in a midwestern agricultural watershed. *Ecosystems* 20, 1468–1482. doi: 10.1007/s10021-017-0125-0
- Motew, M. M., and Kucharik, C. J. (2013). Climate-induced changes in biome distribution, NPP, and hydrology in the Upper Midwest U.S.: a case study for potential vegetation: climate impacts in the upper Midwest U.S. *J. Geophys. Res. Biogeosci.* 118, 248–264. doi: 10.1002/jgrg.20025
- Nassauer, J. I. (1988). The aesthetics of horticulture: neatness as a form of care. *HortScience* 23, 973–977. doi: 10.21273/HORTSCI.23.6.973
- Orth, R. J., Carruthers, T. J. B., Dennison, W. C., Duarte, C. M., Fourqurean, J. W., Heck, K. L., et al. (2006). A global crisis for seagrass ecosystems. *BioScience* 56, 987. doi: 10.1641/0006-3568(2006)56[987:AGCFSE]2.0.CO;2
- Ortiz-Bobea, A. (2021). Anthropogenic climate change has slowed global agricultural productivity growth. *Nat. Clim. Chang.* 11, 28. doi: 10.1038/s41558-021-01000-1
- Palm, C., Blanco-Canqui, H., DeClerck, F., Gater, L., and Grace, P. (2014). Conservation agriculture and ecosystem services: an overview. *Agric. Ecosyst. Environ.* 187, 87–105. doi: 10.1016/j.agee.2013.10.010
- Peters, C. J., Picardy, J. A., Darrouzet-Nardi, A., and Griffin, T. S. (2014). Feed conversions, ration compositions, and land use efficiencies of major livestock products in U.S. agricultural systems. *Agric. Syst.* 130, 35–43. doi: 10.1016/j.agry.2014.06.005
- Pinheiro, J., Bates, D., DebRoy, S., Sarkar, D., Heisterkamp, S., Van Willigen, B., et al. (2022). Package “nlme.” Available online at: <https://svn.r-project.org/R-packages/trunk/nlme/> (accessed June 13, 2022).
- PRISM Climate Group, Oregon State University (2022). Available online at: <https://prism.oregonstate.edu/> (accessed July 27, 2022).
- Provenza, F. D., Anderson, C., and Gregorini, P. (2021). We are the Earth and the Earth Is Us: How palates link foodscapes, landscapes, heartscapes, and thoughts. *Front. Sustain. Food Syst.* 5, 547822. doi: 10.3389/fsufs.2021.547822
- Qiu, G., Li, H., Zhang, Q., Chen, W., Liang, X., and Li, X. (2013). Effects of evapotranspiration on mitigation of urban temperature by vegetation and urban agriculture. *J. Integr. Agric.* 12, 1307–1315. doi: 10.1016/S2095-3119(13)60543-2
- Raff, Z., and Meyer, A. (2022). CAFOs and surface water quality: evidence from Wisconsin. *Am. J. Agric. Econ.* 104, 161–189. doi: 10.1111/ajae.12222

- Rissman, A. R., Fochesatto, A., Lowe, E. B., Lu, Y., Hirsch, R. M., and Jackson, R. D. (in this volume). Grasslands and managed grazing policy review: trends and options from the Midwestern United States. *Front. Sustain. Food Syst.* (in this volume).
- Rockström, J., Williams, J., Daily, G., Noble, A., Matthews, N., Gordon, L., et al. (2017). Sustainable intensification of agriculture for human prosperity and global sustainability. *Ambio* 46, 4–17. doi: 10.1007/s13280-016-0793-6
- Rowntree, J. E., Stanley, P. L., Maciel, I. C. F., Thorbecke, M., Rosenzweig, S. T., Hancock, D. W., et al. (2020). Ecosystem impacts and productive capacity of a multi-species pastured livestock system. *Front. Sustain. Food Syst.* 4. doi: 10.3389/fsufs.2020.544984
- Rui, Y., Jackson, R. D., Cotrufo, M. F., Sanford, G. R., Spiesman, B. J., Deiss, L., et al. (2022). Persistent soil carbon enhanced in Mollisols by well-managed grasslands but not annual grain or dairy forage cropping systems. *Proc. Natl. Acad. Sci.* 119, e2118931119. doi: 10.1073/pnas.2118931119
- Sanford, G. R., Jackson, R. D., Booth, E. G., Hedtcke, J. L., and Picasso, V. (2021). Perenniality and diversity drive output stability and resilience in a 26-year cropping systems experiment. *Field Crops Res.* 263, 108071. doi: 10.1016/j.fcr.2021.108071
- Schilling, K. E., Gassman, P. W., Kling, C. L., Campbell, T., Jha, M. K., Wolter, C. F., et al. (2014). The potential for agricultural land use change to reduce flood risk in a large watershed: land use change and flooding. *Hydrol. Process.* 28, 3314–3325. doi: 10.1002/hyp.9865
- Schindler, D. W., Carpenter, S. R., Chapra, S. C., Hecky, R. E., and Orihel, D. M. (2016). Reducing phosphorus to curb lake eutrophication is a success. *Environ. Sci. Technol.* 50, 8923–8929. doi: 10.1021/acs.est.6b02204
- Schulte, L. A., Niemi, J., Helmers, M. J., Liebman, M., Arbuckle, J. G., James, D. E., et al. (2017). Prairie strips improve biodiversity and the delivery of multiple ecosystem services from corn–soybean croplands. *Proc. Natl. Acad. Sci.* 114, 11247–11252. doi: 10.1073/pnas.1620229114
- Smith, A. P., Johnson, I. R., Schwenke, G., Lam, S. K., Suter, H. C., and Eckard, R. J. (2020). Predicting ammonia volatilization from fertilized pastures used for grazing. *Agric. For. Meteorol.* 287, 107952. doi: 10.1016/j.agrformet.2020.107952
- Soylu, M. E., Kucharik, C. J., and Loheide, S. P. (2014). Influence of groundwater on plant water use and productivity: development of an integrated ecosystem—variably saturated soil water flow model. *Agric. For. Meteorol.* 189–190, 198–210. doi: 10.1016/j.agrformet.2014.01.019
- Strauser, J., Stewart, W. P., and Leitschuh, B. (2022). Producing regions: connecting place-making with farming practices. *Soc. Nat. Resour.* 35, 1–9. doi: 10.1080/08941920.2022.2101080
- Suparak Gibson, A. (2022). *The Underground Economy: Regenerative Farming's Hidden Economic, Ecological, and Social Value*. Master's Thesis. Harvard University Division of Continuing Education.
- Tsiafouli, M. A., Thébault, E., Sgardelis, S. P., de Ruiter, P. C., van der Putten, W. H., Birkhofer, K., et al. (2015). Intensive agriculture reduces soil biodiversity across Europe. *Glob. Chang. Biol.* 21, 973–985. doi: 10.1111/gcb.12752
- USDA—Economic Research Service. (2018). *Soybean U.S. stocks*. Available online at: <https://www.ers.usda.gov/> (accessed October 31, 2020).
- USDA—FoodData Central USDA Agricultural Research Service (2022). *FoodData Cent.* Available online at: <https://fdc.nal.usda.gov/> (accessed June 24, 2022).
- USDA-NASS—National Agricultural Statistics Service (2022). *Charts and Maps - County Maps*. Available online at: https://www.nass.usda.gov/Charts_and_Maps/Crops_County/index.php (accessed June 15, 2022).
- Vadas, P. A., Gburek, W. J., Sharpley, A. N., Kleinman, P. J. A., Moore, P. A., Cabrera, M. L., et al. (2007). A model for phosphorus transformation and runoff loss for surface-applied manures. *J. Environ. Qual.* 36, 324–332. doi: 10.2134/jeq2006.0213
- Vadas, P. A., Haggard, B. E., and Gburek, W. J. (2005). Predicting dissolved phosphorus in runoff from manured field plots. *J. Environ. Qual.* 34, 1347–1353. doi: 10.2134/jeq2004.0424
- Vadas, P. A., Kleinman, P. J. A., and Sharpley, A. N. (2004). A simple method to predict dissolved phosphorus in runoff from surface-applied manures. *J. Environ. Qual.* 33, 749–756. doi: 10.2134/jeq2004.7490
- van Vliet, S., Bain, J. R., Muehlbauer, M. J., Provenza, F. D., Kronberg, S. L., Pieper, C. F., et al. (2021a). A metabolomics comparison of plant-based meat and grass-fed meat indicates large nutritional differences despite comparable nutrition facts panels. *Sci. Rep.* 11, 13828. doi: 10.1038/s41598-021-93100-3
- van Vliet, S., Provenza, F. D., and Kronberg, S. L. (2021b). Health-promoting phytonutrients are higher in grass-fed meat and milk. *Front. Sustain. Food Syst.* 4, 555426. doi: 10.3389/fsufs.2020.555426
- Wepking, C., Avera, B., Badgley, B., Barrett, J. E., Franklin, J., Knowlton, K. F., et al. (2017). Exposure to dairy manure leads to greater antibiotic resistance and increased mass-specific respiration in soil microbial communities. *Proc. R. Soc. B Biol. Sci.* 284, 20162233. doi: 10.1098/rspb.2016.2233
- Werling, B. P., Dickson, T. L., Isaacs, R., Gaines, H., Gratton, C., Gross, K. L., et al. (2014). Perennial grasslands enhance biodiversity and multiple ecosystem services in bioenergy landscapes. *Proc. Natl. Acad. Sci.* 111, 1652–1657. doi: 10.1073/pnas.1309492111
- Wheeler, T., and von Braun, J. (2013). Climate change impacts on global food security. *Science* 341, 508–513. doi: 10.1126/science.1239402
- Wisconsin Groundwater Coordinating Council Report to the Legislature (2022). *Wisconsin Groundwater Coordinating Council*.
- Zipper, S. C., Soylu, M. E., Booth, E. G., and Loheide, S. P. (2015). Untangling the effects of shallow groundwater and soil texture as drivers of subfield-scale yield variability: yield, groundwater, soil. *Water Resour. Res.* 51, 6338–6358. doi: 10.1002/2015WR017522



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EDITED BY

Carol Williams,
University of Wisconsin–Madison,
United States

REVIEWED BY

Raghavendra Singh,
Indian Institute of Pulses Research
(ICAR), India
Tim Delbridge,
Oregon State University, United States

*CORRESPONDENCE

Andrew Smith
✉ andrew.smith@rodaleinstitute.org
Yichao Rui
✉ ruiy@purdue.edu

†PRESENT ADDRESSES

Kirsten A. Pearsons,
University of California Cooperative
Extension, Salinas, CA, United States

Yichao Rui,
Department of Agronomy, Purdue
University, West Lafayette, IN,
United States

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Reducing tillage does not affect the long-term profitability of organic or conventional field crop systems

Kirsten A. Pearsons^{1†}, Craig Chase², Emmanuel C. Omondi³,
Gladis Zinati¹, Andrew Smith^{1*} and Yichao Rui^{1*†}

¹Rodale Institute, Kutztown, PA, United States, ²Iowa State University Extension and Outreach, Ames, IA, United States, ³Department of Agricultural and Environmental Sciences, Tennessee State University, Nashville, TN, United States

Reducing tillage and supporting continuous living cover (CLC) can improve agroecosystem sustainability under both organic and conventional field crop production. What is less clear, however, is how reducing tillage affects the economic sustainability of organic field crop systems with CLC as compared to conventional field crop systems. To address this knowledge gap, we conducted a comprehensive economic analysis based on field records and crop yields from the long-term Farming Systems Trial (FST) at Rodale Institute in Kutztown, Pennsylvania. The FST (established in 1981) comprises three farming systems (conventional, low-input organic, and manure-based organic) which were split into tilled and reduced-till treatments in 2008. FST field activities, inputs, and crop yields from 2008 to 2020 were used to construct enterprise budgets to assess cumulative labor, costs, returns, and economic risk of six replicated theoretical farms. Reducing tillage on the conventional farms led to lower gross revenues (–10%), but lower annual costs (–5%) helped maintain similar net returns but increased economic risk as compared to tilled conventional farms. Reducing tillage on the low-input organic farms also led to lower gross revenues (–13%) and lower annual costs (–6%), which maintained net returns and increased risk relative to the tilled, low-input organic farms. For the more diverse manure-based organic farms that include periods of mixed perennial cover, reducing tillage had a smaller effect on overall costs (–2%) and no effect on gross revenues, net returns, or economic risk. Overall, reducing tillage did not affect the long-term profitability of any of the three FST farming systems. Regardless of tillage practices or organic price premiums, the manure-based organic system supported higher net returns than the conventional system. This finding suggests that continuous living cover and manure inputs may have a greater influence on system profitability than tillage practices.

KEYWORDS

conservation tillage, no-till, continuous living cover, organic agriculture, economics, profitability

1. Introduction

Innovative agricultural practices developed during the 20th century helped double grain yields since the 1960's (Ramankutty et al., 2018), but the widespread adoption of synthetic pesticides, inorganic fertilizers, expansive monocultures, and intensive tillage has come at great human health and environmental costs (Tilman et al., 2002; LaCanne and Lundgren, 2018; Sanaullah et al., 2020). To continue feeding, fueling, and clothing a growing population, these once-innovative, now conventional agricultural practices may need to be phased out in favor of alternative, conservation-based practices. Conservation-based practices can improve soil health and environmental quality, and include strategies such as reducing pesticide use, diversifying crop rotations, aiming for continuous living cover (CLC), and applying organic fertilizers (Palm et al., 2014). It is no coincidence that these practices are fundamental to organic crop production, which overall has been shown to reduce the negative effects of agriculture on environmental health (Gomiero et al., 2011). In addition to these environmental benefits, organic production is often more profitable because of the price premiums that consumers are willing to pay for organic products (Reganold and Wachter, 2016).

Although organic farmers have outpaced conventional farmers in the overall adoption of conservation-based practices (Gomiero et al., 2011), one specific practice—reducing tillage—has been more readily adopted under conventional management (Mirsky et al., 2012; Claassen et al., 2018). Reduced-till management has been adopted by over 65% of conventional farmers in the United States in part because most have access to low-cost herbicides that provide an effective way to manage weeds without regular tillage (Pittelkow et al., 2015; Benbrook, 2016; NASS, 2019; White et al., 2019). Conventional reduced-till strategies can significantly lower operating costs and, depending on climate and other cropping conditions, can support high crop yields (Archer and Reicosky, 2009; Chavas et al., 2009; Toliver, 2010; Deines et al., 2019). Lower crop yields in response to reducing tillage are often attributed to high soil compaction, nutrient deficiencies, and/or high weed pressure (Pittelkow et al., 2015)—all factors that could be mitigated by supporting continuous living cover. While many conventional farmers have adopted reduced-till strategies, far fewer have adopted practices that support continuous living cover (e.g., only 7.5% of farmers plant cover crops; NASS, 2019).

In contrast, one of the most popular strategies to reduce tillage in organic systems is cover crop-based, rotational no-till, where cover crops are mechanically terminated to form a weed-barrier mulch (Ashford and Reeves, 2003; Wallace et al., 2017; Frascioni et al., 2019). Under this strategy, mechanical termination of cover crops usually occurs at the same time as planting, eliminating any period of bare soil between cover crop termination and planting cash crops. Despite the potential cost savings and soil conservation benefits of reducing tillage

and maintaining continuous living cover, uncertainty regarding yields and profitability could limit the adoption of reduced-till adoption in organic systems.

Reducing tillage under organic production has been hypothesized to have similar economic benefits as under conventional production (Peigné et al., 2007; Mirsky et al., 2012), but few studies have tested this hypothesis (Delate et al., 2012; Wittwer et al., 2021). Moreover, existing studies were either based on short-term trials (Delate et al., 2012), which may not capture year-to-year variability in crop yields (Delbridge et al., 2011) or excluded genetically modified crops and pesticide seed coatings (Wittwer et al., 2021) that currently dominate U.S. field crop production (Douglas and Tooker, 2015; Donley, 2019) and significantly affect management costs and profitability of conventional systems (Finger et al., 2011; Alford and Krupke, 2018). Although these existing studies support the hypothesis that reduced-till organic production is profitable, there is a clear knowledge gap regarding the long-term economic impacts of reducing tillage in organic farming systems compared to conventional systems. A long-term economic comparison of such systems could help address this knowledge gap and provide critical information for farmers and policymakers interested in organic reduced-till production.

In this paper, we conducted a comprehensive economic analysis of the long-term Farming Systems Trial (FST) at the Rodale Institute in Kutztown, Pennsylvania, which provided a unique opportunity to perform a side-by-side economic comparison of reducing tillage in organic and conventional farming systems with different fertility inputs and different degrees of continuous living cover. We hypothesized that reducing tillage would lower crop yields and gross revenue in both organic and conventional field crop systems, but net returns would be higher due to the lower production costs associated with reducing tillage. The results of this economic analysis will serve as a valuable resource for extension agents, farmers, and policymakers to assess the economic advantages and disadvantages of reducing tillage in organic and conventional farming systems.

2. Materials and methods

2.1. Design of the farming systems trial

This economic analysis was based on field operation and input records of the Farming Systems Trial (FST) at the Rodale Institute in Kutztown, Pennsylvania (Berk County, 40° 33' 5"–75° 43' 47"). The FST was originally established in 1981 to study soil health, agronomy, and economics during a transition to organic grain production. The FST initially comprised three conventionally-tilled cropping systems: (1) a conventional system with inorganic fertilizer inputs (CNV); (2) a low-input organic system that relies on leguminous cover crops

to supply nitrogen inputs (LEG); and (3) an organic system with cover crops, periodic manure inputs, and a perennial hay phase of 2–3 years during each crop rotation (MNR). Cropping systems were replicated eight times (18×92 -m plots), with each replicate divided into three subplots (6×92 -m) planted in different phases of the crop rotations. In 2008, reduced-till treatments were introduced to the study by reducing tillage in half of the system replicates (RT-CNV, RT-LEG, and RT-MNR) while conventional, full tillage (FT) continued in the other four replicates (FT-CNV, FT-LEG, and FT-MNR). Herbicide application helped to achieve complete no-till production in the RT-CNV treatment, lowering average Soil Tillage Intensity Ratings (STIR) ratings to 4.5 ± 0.4 compared to 142.7 ± 8.4 in the FT-CNV treatment (Pearsons et al., 2023). Rotational no-till was achieved in the organic systems by no-till planting organic maize (*Zea mays* L.) and soybeans (*Glycine max* L.) into cover crops that were mechanically terminated by use of a roller-crimper (Ashford and Reeves, 2003; Moyer, 2021). Depending on the sub-plot, no-till organic maize and soybeans accounted for 15–24% of planting events in the RT-MNR treatment and 15–30% of planting events in the RT-LEG treatment (Supplementary Figures S1, S2). Moldboard plowing and disking preceded all other crops and cover crops in the organic treatments while chisel plowing and disking preceded all crops and cover crops in the FT-CNV treatment. Average STIR ratings for the FT-LEG and FT-MNR treatments were 263.7 ± 29.2 and 196.2 ± 37.9 respectively, while STIR ratings for the RT-LEG and RT-MNR treatments were 178.5 ± 15.5 and 126.3 ± 18.8 respectively (Pearsons et al., 2023).

Between 2008 and 2020, crop rotations differed among the three systems (Supplementary Figures S1, S2). The CNV system followed 2- and 3-year rotations of maize, soybean and occasionally wheat (*Triticum aestivum* L.). In some years, hairy vetch (*Vicia villosa*) or cereal rye (*Secale cereale*) cover crops were planted in CNV sub-plots. The LEG system followed a 4-year maize–oats (*Avena sativa* L.)–soybeans–wheat rotation, with hairy vetch preceding maize and rye cover preceding oats and soybeans. As an additional source of nitrogen, clover (*Trifolium spp.*) was planted concurrently with oats. From 2008–2014, a barley crop (*Hordeum vulgare* L.) was grown in the FT-LEG plots prior to soybeans and in place of the rye cover crop. The MNR system followed the LEG rotation but with the addition of 2–3 years of alfalfa-orchardgrass hay (7:4 w:w *Medicago sativa* L. and *Dactylis glomerata* L.), 1 year of maize silage, and one additional year of wheat (Supplementary Figures S1, S2). In an average year, all four organic treatments had living cover for over 10 months (FT-LEG = 10.3 months, RT-LEG = 10.6 months, FT-MNR = 10.4 months, RT-MNR = 10.2 months). Even with occasional cover crops, the CNV treatments had living cover for fewer months (FT-CNV = 6.3 months; RT-CNV = 8.4 months) compared to the organic treatments.

Typical field operations for each crop within each treatment are summarized in Table 1. For each year of the study, fertility inputs in the CNV system were based on soil tests and recommendations from the Penn State Agricultural Analytical Services Laboratory (University Park, PA, U.S.A.) and herbicide applications (timing, mixtures, and rates) were based on recommendations from Weed Extension Specialists from the Pennsylvania State University. Between 2008 and 2020, the only external fertility input to the LEG system was potassium sulfate, applied at a rate of 170 kg K ha^{-1} in 2008 and 2012. For the MNR system, composted manure was applied at a target rate of $89.7 \text{ kg N ha}^{-1}$ prior to planting corn silage and oats. Potassium sulfate was also applied to the MNR system in 2008 and 2012, at the same 170 kg K ha^{-1} rate as in the LEG system. No pest management strategies were deployed in the MNR system, but parasitoid wasps (*Trichogramma ostriniae*; IPM Labs, Locke, NY, USA) were deployed in 2012 to help control European corn borer (*Ostrinia nubilalis*) in the LEG system based on recommendations from Pennsylvania State University Extension.

2.2. Relative crop yields

Relative yields (Y_R) were analyzed to account for the different frequencies for which specific crops were grown within each treatment. Y_R was calculated as the ratio of the experiment yield compared to county average yields:

$$Y_R = Y_E/Y_A$$

Where Y_R = relative yield (a unitless value), Y_E = experimental yield at the subplot level, and Y_A = county average yield for a given crop in a given year. Average county yields were obtained from the USDA NASS – Quick Stats database (NASS, 2021). In years where average crop yields were unavailable for Berks County, values from nearby Lehigh County (maize: 2009, 2010, 2013; oats: 2010, 2013, 2019) or all of Pennsylvania (wheat: all years; barley: 2009, 2010; hay: 2016, 2017, 2019, 2020; silage corn: 2013, 2018) were substituted. Student's t-tests were used to test if average crop yields were significantly different than county averages ($H_0: Y_R = 1$) for each crop in each treatment. Raw yield data from the FST for this period (2008–2020) were analyzed and discussed as part of an assessment of grain quality from the FST (Pearsons et al., 2022).

2.3. Enterprise budgets

Records of field activities, inputs, and crop yields from the FST were used to construct enterprise budgets (Chase et al., 2019; Chase, 2020) for each subplot for each year from 2008 through 2013, as well as from 2016 through

TABLE 1 Typical field operations for each crop grown in each treatment.

Cover crop, cash crop	Field operation(s)	CNV		LEG		MNR	
		FT	RT	FT	RT	FT	RT
<i>Hairy vetch, maize</i>	Plow	-	-	MB	MB	MB	MB
	Disk and pack	-	-	DP	DP	DP	DP
	Plant hairy vetch cover, kg ha ⁻¹	-	-	P, 34	P, 34	P, 34	P, 34
	Herbicide, burn-down	-	H1	-	-	-	-
	Plow	CP	-	MB	-	MB	-
	Disk and pack	DP	-	DP	-	DP	-
	Plant maize, 1,000 seeds ha ⁻¹	P, 82	NT, 82	P, 89	RC+NT, 89	P, 89	RC+NT, 89
	Fertilize	NPK	NPK	-	-	-	-
	Herbicide, pre-emergent	H3		-	-	-	-
	Herbicide, post-emergent	H4	H4	-	-	-	-
	Fertilize	N1	N1	-	-	-	-
	Cultivate	-	-	TW 2× RC 2×	-	TW 2× RC 2×	-
	Harvest maize	✓	✓	✓	✓	✓	✓
<i>Rye, oats</i>	Plow	-	-	MB	MB	MB	MB
	Disk and pack	-	-	DP	DP	DP	DP
	Plant rye cover, kg ha ⁻¹	-	-	P, 202	P, 202	P, 202	P, 202
	Apply compost, 1,000 kg ha ⁻¹	-	-	-	-	LM, 28–38	LM, 28–38
	Plow	-	-	MB	MB	MB	MB
	Disk and pack	-	-	DP	DP	DP	DP
	Plant oats, kg ha ⁻¹	-	-	P, 202	P, 202	P, 202	P, 202
	Plant clover, kg ha ⁻¹	-	-	-	-	BC, 10–15	BC, 10–15
	Harvest oats	-	-	✓	✓	✓	✓
<i>Rye or barley, soybeans</i>	Plow	-	-	MB	MB	MB	MB
	Disk and pack	-	-	DP	DP	DP	DP
	Plant rye cover, kg ha ⁻¹	-	-	-	P, 202	P, 202	P, 202
	Plant barley, kg ha ⁻¹	-	-	P, 202	-	-	-
	Harvest barley, rake, ted, and bale straw	-	-	✓	-	-	-
	Plow	CP	-	MB	-	MB	-
	Disk and pack	DP	-	DP	-	DP	-
	Herbicide, burn-down	-	H1	-	-	-	-
	Plant soybeans, 1,000 seeds ha ⁻¹	P, 445–494	NT, 445–494	P, 495–544	RC+NT, 495–544	P, 495–544	RC+NT, 495–544
	Herbicide, post-emergent	H5	H5	-	-	-	-

(Continued)

TABLE 1 (Continued)

Cover crop, cash crop	Field operation(s)	CNV		LEG		MNR	
		FT	RT	FT	RT	FT	RT
	Cultivate	-	-	TW 2× RC 2×	HRC	TW 2× RC 2×	HRC
	Harvest soybeans	✓	✓	✓	✓	✓	✓
Wheat	Plow	CP	-	MB	MB	MB	MB
	Disk and pack	DP	-	DP	DP	DP	DP
	Herbicide, burndown	-	H2	-	-	-	-
	Plant wheat, kg ha ⁻¹	P, 202	NT, 202	P, 202	P, 202	P, 202	P, 202
	Fertilize	N2	N2	-	-	-	-
	Herbicide, spring	H6	H6	-	-	-	-
	Harvest wheat/ rake, ted, bale straw	✓	✓	✓	✓	✓	✓
Hay 2–3 seasons	Plow	-	-	-	-	MB	MB
	Disk and pack	-	-	-	-	DP	DP
	Plant orchardgrass, kg ha ⁻¹	-	-	-	-	P, 16	P, 16
	Plant alfalfa, kg ha ⁻¹	-	-	-	-	BC, 9	BC, 9
	Cut/rake/ted/bale hay	-	-	-	-	8–10 ×	8–10 ×
Maize silage	Apply compost, 1,000 kg ha ⁻¹	-	-	-	-	LM, 28–62	LM, 28–62
	Plow	-	-	-	-	MB	MB
	Disk and pack	-	-	-	-	DP	DP
	Plant maize	-	-	-	-	P, 89	P, 89
	Cultivate	-	-	-	-	TW 2× RC 2×	TW 2× RC 2×
	Harvest silage	-	-	-	-	✓	✓

Field operations deviated where crop rotations changed (e.g., when cover crops were included in CNV rotations) and in response to planting issues, high weed pressure, or weather events. These atypical operations (e.g., re-planting cash crops, additional cultivation, and additional herbicide applications) were included in enterprise budgets when they were performed. Planting and fertilizer rates are listed as kg ha⁻¹ unless otherwise noted. Abbreviations are listed alphabetically and provide additional details regarding specific field operations, including herbicide rates and mixtures applied in the CNV treatments.

BC, broadcast; CP, chisel plow; DP, disk and pack; H1, glyphosate (0.84 kg ha⁻¹) + 2,4-D (0.56 kg ha⁻¹) + ammonium sulfate (AMS; 2.24 kg ha⁻¹); H2, glyphosate (0.84 kg ha⁻¹) + 2,4-D (0.56 kg ha⁻¹) + AMS (3.36 kg ha⁻¹); H3, Degree Xtra (acetochlor + atrazine, 7.0 L ha⁻¹) + Balance Flex (isoxaflutole, 219 mL ha⁻¹); H4, Callisto (mesotrione, 219 mL ha⁻¹) + atrazine (0.56 kg ha⁻¹) + 1% v/v COC + 2% v/v UAN; H5, glyphosate (0.84 kg ha⁻¹); H6, Harmony Extra 50 SG (48 mL ha⁻¹) + 2,4-D (1.4 L ha⁻¹); HRC, high residue cultivate; LM, composted leaf and dairy manure; MB, moldboard plow; N1, 135 kg nitrogen ha⁻¹; N2, 67 kg nitrogen ha⁻¹; NPK, 34 kg nitrogen ha⁻¹ + 34 kg phosphorus ha⁻¹ + 11 kg potassium ha⁻¹; NT, no-till drill; P, conventional drill; RC, row cultivate; RC+NT, roller-crimper with no-till drill; TW, time weed.

2020 (780 budgets). The typical FST crop rotations were interrupted in 2014, so 2014 and 2015 were excluded from this analysis.

For each year of the study, input costs were estimated using management records, recent prices from vendors, government databases, agricultural extension documents, and communication with extension specialists. Estimated costs of crop production were not available for Pennsylvania after 2016, so machinery operation costs, per hour labor costs, fertilizer costs, cash rent equivalent of land, and most conventional seed costs (maize, soybeans, oats, alfalfa, and orchardgrass) were derived from annual Estimated Costs of Crop Production in

Iowa documents (Duffy and Smith, 2008a,b; Duffy, 2009, 2011, 2012, 2013; Plastina, 2016, 2017, 2018, 2019, 2020). Hourly labor requirements were derived from Estimating the Field Capacity of Farm Machines (Hanna, 2016). Other seed costs were based on actual purchase prices for seeds planted in the FST or were estimated based on the relative price of seeds purchased for the FST in 2021. Based on these relative prices, organic maize and soybean seeds were priced as 80% the cost of conventional maize and soybean seeds; organic oat, orchardgrass, and alfalfa seeds were priced as 120% conventional seeds; rye seeds were priced the same as oat seeds; wheat and barley seeds were priced at 2 × oats; hairy vetch was priced as 40% the cost of alfalfa

seeds; and clover was priced as 70% the cost of alfalfa seeds. All seed costs and estimates used in this study are listed in [Supplementary Table S1](#).

Herbicide cost estimates were based on annual average prices set by dealers throughout Pennsylvania; these estimates were compiled and confidentially provided to the authors by Penn State Extension. The cost of deploying parasitoid wasps in 2012 was provided by IPM Labs (Locke, NY, USA), with labor costs estimates derived from [Gagnon et al. \(2016\)](#). Year-adjusted costs to apply composted manure were based on scaled-up labor, fuel, and machinery inputs required to produce, haul, and spread compost at the Rodale Institute in 2008. Annual cost estimates for labor, fertilizers, manure, parasitoid wasps, and seeds are included in the [Supplementary Table S1](#) (confidential herbicide costs not included).

Average annual market crop prices for Pennsylvania were obtained from USDA NASS Quick stats database ([NASS, 2021](#)). For years where organic market prices for specific crops were unavailable for Pennsylvania (noted in [Supplementary Table S2](#)), the organic price premium was either estimated based on the organic price premium for grains in Iowa ([AMS, 2021](#); [NASS, 2021](#)) or interpolated based on averaging the price premium in adjacent years. Missing organic maize silage prices were estimated as $0.33 \times$ the value of organic hay based on the method used by Oregon State University ([Downing et al., 2013](#)), while all straw prices were estimated as $0.75 \times$ the average value of conventional hay ([Chase et al., 2019](#)). Market prices are listed in [Supplementary Table S2](#), with annotations to note estimated or adjusted values.

2.4. Modeling representative farms

To better reflect how the management practices applied in the FST would affect labor, costs, returns, and risk for a representative farm, we used enterprise budgets to model six replicated theoretical farms, each based on one FST plot (3 systems \times 2 tillage treatments, each replicated 4 times). Each farm comprised three, 18-ha fields which corresponded to the three subplots within each FST plot. Field size was chosen based on the average farm size in Pennsylvania during 2008–2020 (54-ha; [NASS, 2019](#)). Statistical analyses were performed on the labor, costs, and returns from these 24 theoretical farms.

2.5. Risk and sensitivity analyses

We assessed system risk using (1) a simple assessment of year-to-year variability of net returns for each farm (standard deviation) and (2) a safety-first model which additionally accounts for average net returns ([Musser et al., 1981](#); [Hanson et al., 1990](#); [White et al., 2019](#)). The output of the safety-first model (lower confidence limit for net returns, L) can give

farmers an idea of how often they can expect net returns to exceed a certain value. The most common lower confidence limit for net returns is L_{75} , which represents the lowest net returns a farmer can expect in 3 out of every 4 years, where:

$$L_{75} = E - (0.674 \times S)$$

with E equal to average annual net returns and S equal to the standard deviation of average annual net returns. Larger L_{75} values indicate less risky systems, as they are expected to produce higher net returns in most years (3 out of 4 years). Additionally, we used linear interpolation to assess the sensitivity of the two organic systems to variation in price premiums ([White et al., 2019](#)).

2.6. Statistical analyses

All statistical analyses were performed in R (v.4.0.3). Relative crop yields and returns (with and without organic premiums) were compared across systems and tillage treatments using linear mixed effect models (LMER from the “lme4” package), with system and tillage as fixed effects and year, farm replicate, and the interaction of year and farm replicate as random effects ([Bates et al., 2015](#)). As the goal of this analysis was to test the effect of reducing tillage within each system, tillage treatment was nested within system. For all models, model residuals were tested for homogenous variance and normality. Pairwise mean comparisons were generated using the EMMEANS function from the “emmeans” package, with “mvt” P -value adjustments to account for multiple comparisons ([Lenth, 2021](#)). Raw yield data from the same period (2008–2020) for this experiment have been previously analyzed and discussed as part of an assessment of grain quality from the FST ([Pearsons et al., 2022](#)).

3. Results

3.1. Relative crop yields

Depending on the crop, relative yields differed across farming systems and between tillage treatments ([Figure 1](#), crop-specific results are presented and discussed in the [Supplementary material](#)). Contrary to our hypothesis, overall relative yields (averaged across all crops) did not significantly differ between tillage treatments [Tillage (System): $\chi^2_2 = 3.4$, $P = 0.34$]. Overall relative yields significantly differed across farming systems (System: $\chi^2_2 = 118.9$, $P < 0.0001$), with higher yields in the MNR system compared to county averages and significantly lower yields in the LEG system compared to county averages ($MNR_{relative} = 1.23 \pm 0.06$, $LEG_{relative} = 0.78 \pm 0.07$).

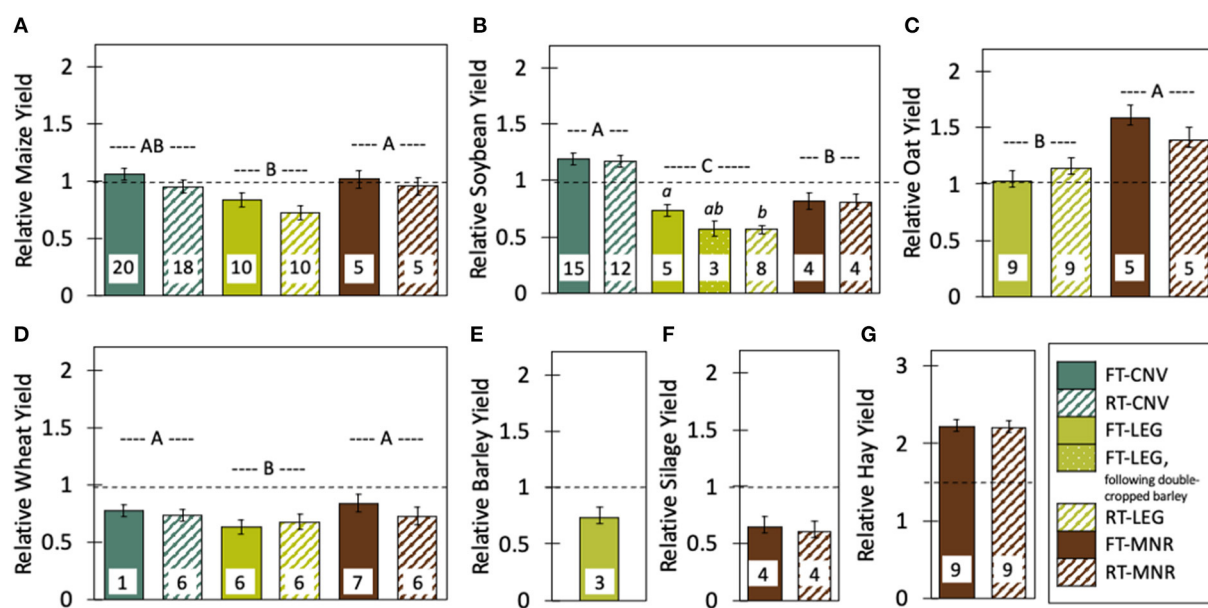


FIGURE 1
Relative yields across all tillage \times system treatments in the FST from 2008–2020 for (A) maize [System: $\chi^2_2 = 31.8$, $P < 0.0001$, Tillage (System): $\chi^2_3 = 4.7$, $P = 0.20$], (B) soybeans [System: $\chi^2_2 = 279.2$, $P < 0.0001$, Tillage (System): $\chi^2_3 = 4.9$, $P = 0.18$], (C) oats [System: $\chi^2_2 = 29.3$, $P < 0.0001$, Tillage (System): $\chi^2_3 = 6.2$, $P = 0.05$], (D) wheat [System: $\chi^2_2 = 18.5$, $P < 0.0001$, Tillage (System): $\chi^2_3 = 11.2$, $P = 0.01$], (E) barley, (F) maize silage (Tillage: $\chi^2_3 = 0.7$, $P = 0.42$), and (G) hay (Tillage: $\chi^2_3 = 0.02$, $P = 0.90$). Means and error bars are estimated marginal means (EMM) and standard errors from mixed models. Different uppercase letters above bars indicate significant differences across systems (Tukey HSD, $P < 0.05$). Different lowercase letters above bars in (B) indicate significant differences in soybean yields across tillage treatments, with FT yields split between soybeans that were planted following rye cover crops (solid green bar) or double-cropped following barley (green bar with white dots).

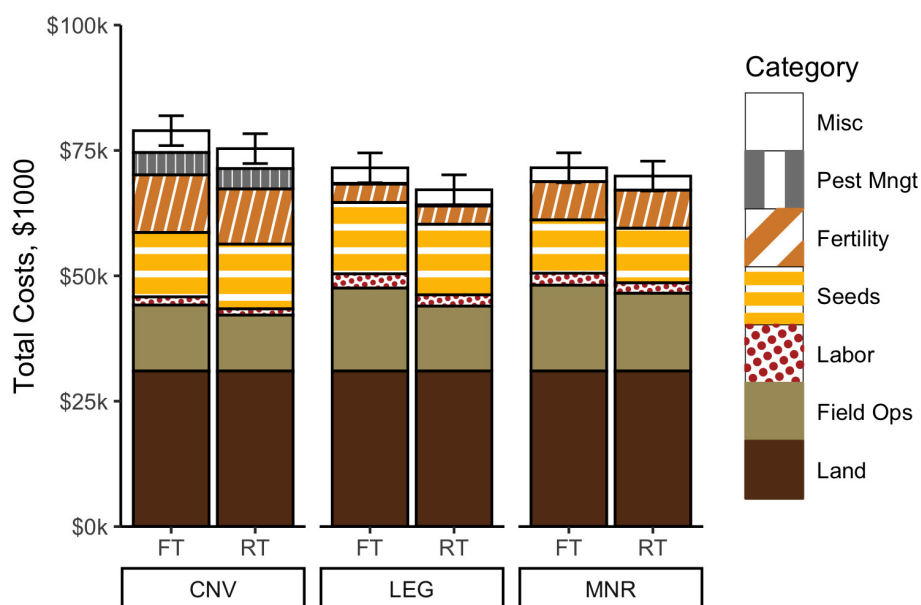


FIGURE 2
Average annual costs for each representative farm with different tillage \times system treatments. Total costs include land (brown), field operations (variable and fixed preharvest and harvest activities; tan), labor (dotted red), seeds (horizontal yellow lines), fertility inputs (diagonal gold lines), pest management inputs (vertical gray lines), and all other costs (interest, insurance, and miscellaneous expenses; white). Error bars are standard errors for total costs within each tillage \times system treatment.

3.2. Costs

3.2.1. Field operations and labor costs

Annual costs associated with field operation were lowest under conventional management (Figure 2; Table 2), averaging 21% higher in the LEG system and 34% higher in the MNR system. Reducing tillage reduced field operation costs across all three systems by an average of 16%. Low field operation costs in the CNV system corresponded with the lowest labor costs, which were over twice as high in both organic systems. Reducing tillage reduced labor costs across all three systems by an average of 20%.

3.2.2. Input costs (seeds, fertility, pest management)

Fertility, seed, and pest management costs all differed across farming systems, but these inputs did not vary much between tillage treatments (Figure 2, Table 2). Although manure applications were more expensive than mineral fertilizer application (an average of \$452 ha⁻¹ for manure, \$318 ha⁻¹ for mineral fertilizers applied to conventional maize), more frequent fertilizer applications in the CNV rotation meant average annual fertility costs were nearly 50% higher for the CNV system compared to the MNR system. Regional differences in manure costs could further increase or decrease the difference in fertility costs between the CNV and MNR systems (Delate et al., 2003; Carr et al., 2020). Minimal fertility inputs in the low-input LEG rotation led to notably lower fertility costs in the LEG system compared to the CNV (−67%) and MNR systems (−52%).

Seed costs were the lowest in the MNR system (Table 2). Despite the higher cost of GM maize and soybean seeds, limited cover crop use meant seed costs were 10% lower in the CNV system compared to the LEG system. The long hay phase of the MNR system kept seed costs 16% lower than in the CNV system.

Input costs associated with pest management averaged around \$4,272 in the CNV treatments, largely as herbicide inputs for weed control. Pest management inputs were negligible in the two organic systems. Including the labor and field operations associated with weed management (cultivation or herbicide applications), overall weed management costs were nearly four times higher in the CNV system (FT-CNV = \$5,332, RT-CNV = \$5,256) compared to the FT organic treatments (FT-LEG = \$1,465, FT-MNR = \$1,195) and over eleven times higher than the RT organic treatments (RT-LEG = \$455, RT-MNR = \$478). Reducing tillage did decrease weed management costs in the LEG system by 18% but did not appreciably affect weed management costs in the CNV or MNR systems.

3.2.3. Total costs

Excluding land (rent) costs (average of \$567 ha⁻¹ year⁻¹), annual field operations and seed costs were the most substantial cost categories. Overall, total annual costs (sum of land, labor,

TABLE 2 Means and standard deviations (sd) of annual costs for representative farms with different tillage x system treatments.

System	Tillage	Total costs		Field operations		Labor		Seeds		Fertility inputs		Pest management inputs	
		Mean	sd	Mean	sd	Mean	sd	Mean	sd	Mean	sd	Mean	sd
CNV	FT	\$78,948	\$11,805	\$13,129	\$1,881	\$1,658	\$302	\$12,834	\$2,244	\$11,470	\$7,555	\$4,473	\$2,236
	RT	\$75,365	\$10,174	\$11,147	\$1,406	\$1,237	\$132	\$12,921	\$1,994	\$10,995	\$6,784	\$4,072	\$1,633
LEG	FT	\$71,533	\$12,084	\$16,532	\$2,876	\$2,829	\$467	\$14,229	\$2,436	\$3,651	\$7,133	\$175	\$579
	RT	\$67,159	\$12,163	\$12,947	\$2,148	\$2,235	\$428	\$14,073	\$2,509	\$3,651	\$7,133	\$196	\$652
MNR	FT	\$71,552	\$9,376	\$17,124	\$2,515	\$2,361	\$281	\$10,636	\$3,240	\$7,613	\$7,582	\$-	\$-
	RT	\$69,892	\$8,649	\$15,513	\$2,484	\$2,069	\$205	\$10,888	\$3,700	\$7,613	\$7,582	\$-	\$-

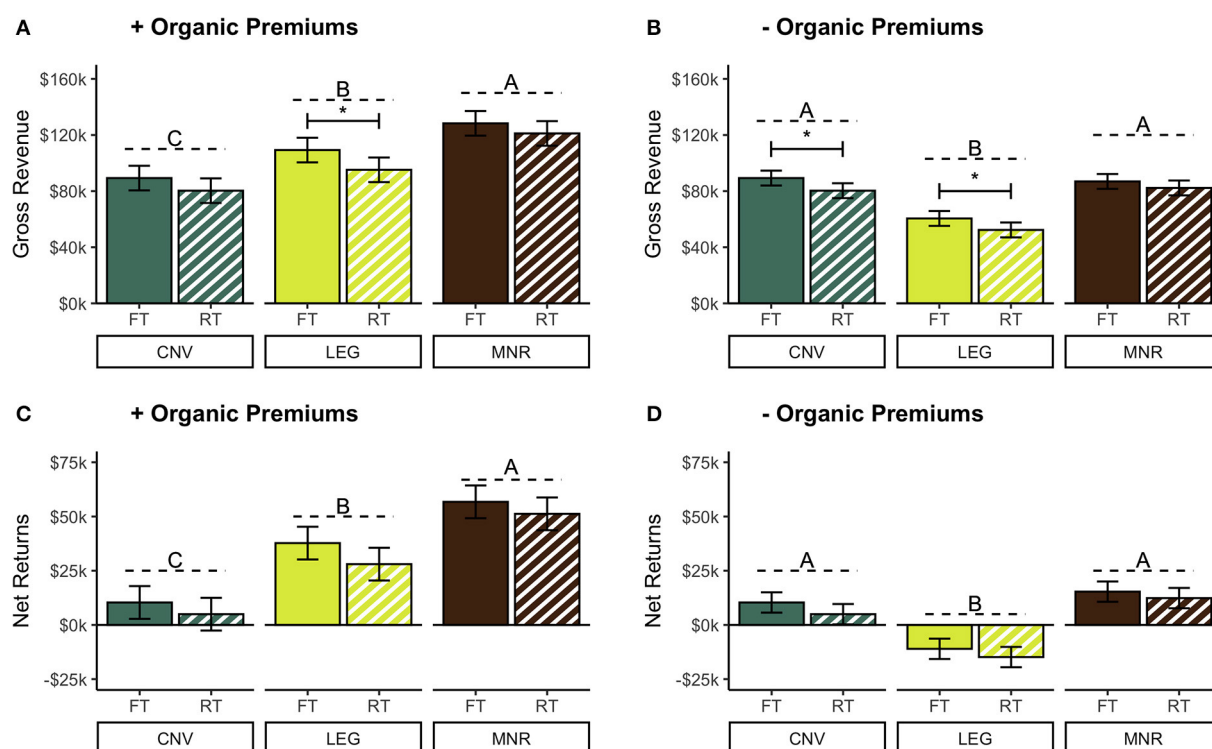


FIGURE 3

Average annual gross revenue and net returns for each representative farm with different tillage \times system treatments (from 2008 to 2013, 2016 to 2020). Gross revenue (A) with and (B) without organic premiums, net returns (gross revenue – total costs) (C) with and (D) without organic premiums. Error bars are standard errors, different letters indicate significant differences across systems (Tukey HSD, $P < 0.05$), while asterisks indicate significant differences between tillage treatments within each system (at $P < 0.05$).

field operations, inputs [seeds, fertility, pest management], and other costs [insurance, interest, and miscellaneous costs]) were notably different across the three farming systems and between tillage treatments within each system (Figure 2, Table 2). Even when accounting for very different organic-management strategies (i.e., different tillage practices, crop rotations, and fertility sources), total costs were comparable between the two organic systems and only 9% lower compared to conventional management. Although no-till management reduced total costs for the representative NT-CNV farms compared to FT-CNV (–5%), annual costs were still around 7% higher under NT-CNV management compared to organic management (Figure 2). Reducing tillage also lowered total costs under LEG management (–6%) but only decreased total costs under MNR management by 2%.

3.3. Gross revenues and net returns

Both with and without organic premiums, economic returns differed across the representative farms that employed different management and tillage strategies (Figure 3; Table 3). When organic price premiums were applied to the grain and forage

produced in the organic systems, both organic farms had higher gross and net returns than the CNV farm (Figures 3A, C; Table 3; gross returns: CNV = \$84,807, LEG = \$102,228, MNR = \$124,716; net returns: CNV = \$7,651, LEG = \$32,882, MNR = \$53,994); on average, LEG and MNR management supported 21 and 47% greater gross revenues than CNV management, respectively. Organic premiums increased gross revenue in the LEG and MNR systems by 81 and 47%, respectively (Figures 3A, B), and with lower costs under organic management, cumulative net returns under MNR management were over 7 times higher than under CNV management. If the organic grain and forage were sold at conventional prices without organic premiums, the LEG system would have generated net losses (–\$12,924). As overall relative yields were higher in the MNR system compared to the CNV system (largely driven by high-value hay) the MNR system would have generated 45% higher net returns than the CNV system even if the organic grains and forages were sold at conventional prices (Figure 3D; MNR = \$13,859, CNV = \$7,651; t-ratio = –10.4; $P = 0.07$). Reducing tillage led to lower gross revenue under both CNV and LEG management but did not significantly affect gross revenue under MNR management (Figure 3, Table 3; –10% in CNV, –13% in LEG). Net returns,

TABLE 3 ANOVA tables and estimated marginal mean values (EMM) for differences in annual gross revenue and net returns for representative farms with different tillage \times system treatments, with tillage nested in system and denoted as tillage (system).

Factor		Returns with organic premiums					Returns without organic premiums				
		Gross revenue, \$			Net returns, \$		Gross revenue, \$			Net returns, \$	
		χ^2	P		χ^2	P	χ^2	P		χ^2	P
System		118.3	***		171.9	***	143.4	***		118.3	***
Tillage (System)		12.2	0.007		6.2	0.10	11.4	0.01		4.0	0.27
System	Tillage	EMM	<i>system</i>	P , <i>tillage</i>	EMM	<i>system</i>	EMM	<i>system</i>	P , <i>tillage</i>	EMM	<i>system</i>
CNV	FT	\$89,299	C	0.10	\$10,351	C	\$89,299	A	0.03	\$10,351	A
	RT	\$80,315			\$4,950		\$80,315			\$4,950	
LEG	FT	\$109,266	B	0.01	\$37,733	B	\$60,513	B	0.048	- \$11,020	B
	RT	\$95,190			\$28,030		\$52,332			- \$14,828	
MNR	FT	\$128,317	A	0.18	\$56,766	A	\$86,901	A	0.24	\$15,350	A
	RT	\$121,114			\$51,222		\$82,262			\$12,369	

Degrees of freedom for system = 2, tillage (system) = 3. Net Returns = gross revenue–total costs. In the EMM table, letters indicate significant differences between systems (Tukey HSD mean comparisons; $P < 0.05$) and *P*-values indicate significance differences between tillage treatments within each system. The highest revenue and returns are shown in bold.

TABLE 4 Year-to-year variability (SD, standard deviation) and 75% lower confidence limits (L) for net returns in each tillage \times system treatments.

System	Tillage	SD	L
CNV	FT	\$18,025	–\$1,797
	RT	\$18,047	–\$7,214
LEG	FT	\$30,474	\$17,194
	RT	\$32,195	\$6,331
MNR	FT	\$40,457	\$29,497
	RT	\$40,372	\$24,011

however, did not significantly differ between tillage treatments (Figures 3C, D).

3.4. Risk and sensitivity analysis

Year-to-year variability (SD) and risk (L) differed for the three farming systems, but reducing tillage had a much smaller effect on year-to-year variability of net returns. Regardless of tillage practices, CNV management was the most stable, as year-to-year variability in net returns was 41–55% lower than that of the organically managed farms (Table 4). Despite high year-to-year variability in net returns, however, organic farms were lower-risk options. All four organic farms had positive values for 75% lower risk limits while the two conventional farms had negative values (Table 4). Lower risk limits were higher under organic management because high cumulative net returns compensated for high year-to-year variability. Values for 75% lower risk limits were consistently lower where tillage was

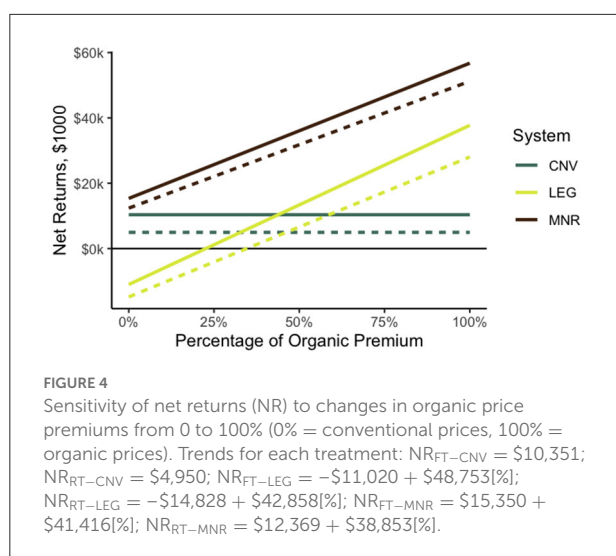
reduced in all three systems, so reducing tillage may increase system risk.

The four organic systems demonstrated different sensitivities to organic price premiums (Figure 4). Regardless of organic price premiums, the FT-MNR and RT-MNR treatments generated higher net returns than the FT-CNV and RT-CNV treatments. For the FT-LEG and RT-LEG treatments, organic price premiums would have to drop over 56 or 41%, respectively, for net returns to drop below the returns of the FT-CNV treatment.

4. Discussion

We expected lower crop yields under reduced-till management but the corresponding reduction in management costs would more-than-compensate for decreased revenues on the representative farms. Although field operation and labor costs were lower under reduced-till management, these cost savings did not over-compensate for decreased revenue and did not lead to significantly higher net returns. The MNR system supported the most days of continuous living cover and was the overall most profitable system, but reducing tillage did not appreciably affect management costs, gross revenues, or net returns for the representative RT-MNR farms. For the representative CNV and LEG farms, reducing tillage did negatively affect overall gross revenues (–10% in CNV, –13% in LEG), which drove down gross revenues compared to the FT-CNV and FT-LEG treatments.

For the CNV system, reducing tillage marginally reduced maize yields. RT systems often require co-adoption of high residue retention (Pittelkow et al., 2015) and/or continuous



living cover provided by winter cover crops (Marcillo and Miguez, 2017) to maintain high maize yields. As noted in the methods section, the RT-CNV treatment only had living cover for an average of 8.4 months of the year. Compared to the RT organic systems, this left an additional 2 months without living cover—a higher risk for erosion, nutrient loss, and missed opportunities to build soil health (Dabney et al., 2001). The LEG system, despite supporting continuous living cover for over 10 months of the year, did not support robust crop growth and high yields (Pearsons et al., 2022). The low-input design of the LEG system did help minimize fertility costs, but such low inputs significantly limited crop productivity. In past assessments of the FST, the LEG system was able to support comparable yields to the CNV system (Hanson et al., 1997), but decades of relying on biological N fixation may have limited long-term N, P, and K availability in this low-input system (Reimer et al., 2020).

Like with gross revenues, overall costs were notably lower in the RT-CNV and RT-LEG treatments compared to their FT counterparts. These cost savings, however, were smaller than hypothesized, sitting around 5%. In the CNV system, input costs accounted for a greater proportion of total costs (37%) than field operations and labor (18% of total costs). Eliminating tillage operations in the RT-CNV treatment reduced field operation and labor costs by 16.3% but did not have a substantial effect on herbicide use, seed inputs, or inorganic fertilizer inputs. Reducing tillage may have even small effects on conventional management costs in future years, as herbicide-resistant weeds (Reddy and Norsworthy, 2010) and emerging nutrient deficiencies (Elkin et al., 2016) drive up input costs relative to field operations and labor, but only if these costs outpace rising labor and fuel costs (White et al., 2019).

For the LEG system, input costs accounted for a smaller proportion of total costs (23%) than field operations and labor (25%), which presented a greater opportunity for reducing

tillage to affect total management costs. Indeed, despite fewer opportunities to reduce tillage compared to the CNV system (15–30% of planting events), the associated field operation and labor cost savings of reducing tillage (−22%) had a slightly greater effect on total costs (−6%) under LEG management.

Due to the multiyear hay phase of the MNR crop rotation, tillage intensity in the FT-MNR treatment was more similar to the RT-LEG treatment than the FT-LEG treatment. This meant there were fewer opportunities to further reduce tillage in the RT-MNR treatment, and therefore fewer opportunities for costs to differ between the MNR treatments.

Despite lower overall costs for the two organic systems, field operation and labor costs were notably higher compared to the conventional system. Most of these additional field operation and labor costs were associated with cover crop establishment. Although cover crops can provide numerous agronomic and environmental benefits, the added costs to establish cover crops may be a barrier to adoption (Roesch-McNally et al., 2018). Unlike cover crops, adding perennial crops to a cropping system (e.g., hay in the MNR system) can increase continuous living cover and inherently reduces tillage activity, without the added cost of cover crop seeds or no-till specific equipment (e.g., no-till planters, roller-crimpers, and high residue cultivators). Perennial hay production also reduced labor costs and crop seed costs across the MNR rotation. High hay yields and value helped drive the MNR system's profitability, and years of continuous living cover likely supported the high maize, soybean, and small grain yields in the MNR system. As long as farmers have the equipment, time, and interest in hay production (Olmstead and Brummer, 2008), perennial hay production can be a valuable addition to organic and conventional farming systems.

This perennial hay production, however, meant the MNR crop rotation presented fewer opportunities to further reduce tillage compared to the rotations used in the CNV and LEG systems. Overall, net returns in the MNR treatments were more reflective of overall lower costs associated with organic production, manure fertilization, high organic premiums, and high hay yields rather than tillage practices. Strategies to reduce tillage may be best applied to crop rotations which present more opportunities to reduce tillage, such as simple maize-soybean rotations. Efforts to reduce tillage in more diverse organic rotations (i.e., no-till planting cover crops, small grains, and hay) would help to further reduce costs and increase benefits of reducing tillage under organic management.

Regardless of tillage practices, the effect of organic management on costs and net returns was largely consistent with past economic analyses of the FST (Hanson et al., 1990, 1997), other long-term trials (Delate et al., 2003; Chavas et al., 2009; White et al., 2019), and meta-analyses (Crowder and Reganold, 2015; McBride et al., 2015). Generally, field operations and labor costs are higher under organic management, but lower inputs and high organic premiums offset these added costs.

From 1982 to 1984, seed costs, field operation, and labor costs were higher for the LEG system compared to the CNV system (by 23, 4, and 20%, respectively; [Hanson et al., 1997](#)), while fertilizer and pest management inputs were only a factor for the CNV system (The MNR system was not included in this initial analysis; [Hanson et al., 1997](#)). While the magnitude of these differences changed over four decades, seed costs, field operation, and labor costs were still higher for the LEG system (by 10, 21, and 75%, respectively) while fertilizer and pest management inputs remained higher for the CNV system (by 67 and 96%, respectively). At the start of the FST overall costs were about 18% lower for the LEG system compared to the CNV system ([Hanson et al., 1997](#)). Decades later, overall costs were still lower for the LEG system, but this cost advantage had dropped to just over 10%. This decrease can partly be attributed to the fertility and pest management inputs in the LEG system but is mostly reflective of how rising fuel and labor costs have a stronger impact on organic systems. Other long-term trials have similarly observed how fertility inputs and pest management drive up overall costs under conventional management, despite higher field operation and labor costs under organic management ([Delate et al., 2003](#); [Crowder and Reganold, 2015](#); [White et al., 2019](#)).

Also consistent with the first economic assessment of the FST (covering the first 9 years after establishment, 1981–1989; [Hanson et al., 1990](#); [Roberts and Swinton, 1995](#)), the low-input LEG system continued to be a less risky option than CNV management. In the initial analysis, the LEG system was considered less risky because annual net returns were more stable in the LEG system compared to the CNV system ([Hanson et al., 1990](#); [Roberts and Swinton, 1995](#)). After three decades, net returns destabilized in the LEG system (greater year-to-year variability), but higher prices for organic grain meant the more variable, low-input, and lower-yielding LEG system was still a lower-risk option compared to the CNV system. With the highest annual net returns and highest lower limit of net returns, the system with the most diverse crop rotation—the MNR system—would be considered the least risky option based on the safety-first method ([Musser et al., 1981](#)). Other economic analyses have similarly concluded that organic systems with long, diverse crop rotations can be lower risk than conventionally managed systems ([White et al., 2019](#)). Although high crop diversity will usually increase the year-to-year variability of costs and gross revenues, high diversity can help buffer against fluctuating crop prices and crop failures.

The most notable difference between the first economic assessment of the FST and this current one is how price premiums affect organic system returns. In the first economic assessment of the FST, the LEG system generated similar net profits as the CNV system, without organic price premiums ([Hanson et al., 1997](#)). With the decreased cost advantage (from 18 to 10%) and lower yields compared to the CNV system, the LEG system became reliant on organic price premiums. As demonstrated by the sensitivity analysis, however, both the

FT-LEG and RT-LEG treatments would remain competitive unless price premiums were reduced by over 56 or 41%, respectively. Additionally, the MNR system generated similar or higher net returns as the CNV system, regardless of price premiums. Similar studies and meta-analyses have found that organic systems can be profitable at lower price premiums as long as crop yields are supported by sufficient fertility inputs and timely pest management ([Delate et al., 2003](#); [Crowder and Reganold, 2015](#); [White et al., 2019](#)).

4.1. Implications for farmer decision-making

For reducing tillage to positively affect net returns, reduced-till strategies must substantially decrease input costs or increase gross revenue. High input costs (seeds, fertility, and pest management) can overshadow the economic benefits of reducing tillage in conventional systems, whereas limited opportunities to reduce tillage can overshadow the economic benefits of reducing tillage in diverse organic systems. With no significant negative effect on overall crop yields, net returns, nor overall economic risk, the decision to reduce tillage will likely depend on farmers' capital, labor availability, or non-economic factors such as soil conservation.

Grain farmers are often limited by capital ([Baumgart-Getz et al., 2012](#); [Roesch-McNally et al., 2018](#)), which can incentivize practices that reduce overall input and operation costs, even if those practices have limited effects on yields or profits. As reducing tillage did not affect net returns or overall yields, the lower operation and labor costs of the RT-CNV and RT-LEG treatments could be an incentive for both conventional and organic producers to reduce tillage. Other studies have found similar cost-related benefits of reducing tillage in conventional systems, especially as rising fuel costs have led tillage and cultivation operations to account for a growing proportion of overall production costs ([Chavas et al., 2009](#); [White et al., 2019](#)). Limited capital, however, could also be a significant barrier for farmers to lease or purchase the equipment needed to reduce tillage (e.g., no-till planter, roller crimper, and high residue cultivator).

Although labor accounted for <5% of total costs, labor can play a disproportionate role for farmers that rely on off-farm income ([Lee and McCann, 2019](#)). For farms of similar size to the theoretical farms in this economic analysis (54 ha), most labor is provided by farmers who have primary occupations beyond farming (for farms between 40 and 56 ha, <25% used hired labor and 61% of operators spent more than half of their time earning off-farm income; [NASS, 2019](#)). Farmers that rely on off-farm income may be inclined to adopt practices that reduce labor inputs, such as reduced-till production. Reduced-till organic management could provide a less labor-intensive option for conventional farmers interested in transitioning to organic

production, which is consistently shown to have higher labor requirements compared to conventional production (Hanson et al., 1990, 1997; Delate et al., 2003; Crowder and Reganold, 2015; White et al., 2019). Additionally, conventional farmers that are interested in organic production but do not want to move away from RT practices (Smith et al., 2011) may be encouraged by the overall lower costs and similar net returns between the RT and FT organic systems. The high profitability (net returns) of the MNR system especially indicates that manure inputs in combination with continuous living cover (perennial hay) into long and diverse crop rotations may be the most profitable production strategy from the FST. The similarity between yields in the CNV system and county averages ($CNV_{relative} = 1.04 \pm 0.06$) indicate that the conditions of the FST experiment are representative of how these systems would perform at a larger scale in the mid-Atlantic region, so it follows that similar yields could be achieved on farms modeled after the RT-MNR system.

4.2. Conclusion

Overall, this updated economic analysis of the FST reinforces findings that organic field crop production can be economically favorable compared to conventional field crop production. Although reducing tillage does not appear to affect the profitability of organic field crop production, there are clear long-term economic benefits for grain farmers to transition to organic production. The decision to also reduce tillage, however, will depend on the specific goals and resources of an individual farmer and may be driven more by prospects of improved soil health rather than economics (Zikeli and Gruber, 2017). As price premiums were not the only driver of high profits in the MNR system, conventional farmers and organic farmers alike may be encouraged to adopt some of the conservation-based practices employed in this organic system—long and diverse crop rotations, continuous living cover, and organic fertility inputs—in pursuit of more environmentally and economically sustainable agriculture.

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

Author contributions

AS, EO, and YR: conceptualization and project administration. CC and KP: data curation. KP: formal analysis and visualization. AS, EO, GZ, and YR: funding acquisition. EO, GZ, KP, and YR: investigation. AS, CC, KP, and YR: writing—original draft. AS, EO, CC, GZ, KP, and YR: writing—review and editing. All authors have read and agreed to the published version of the manuscript.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

The Supplementary Material for this article can be found online at: <https://www.frontiersin.org/articles/10.3389/fsufs.2022.1004256/full#supplementary-material>

References

- Alford, A. M., and Krupke, C. H. (2018). A meta-analysis and economic evaluation of neonicotinoid seed treatments and other prophylactic insecticides in Indiana maize from 2000–2015 with IPM recommendations. *J. Econ. Entomol.* 111, 689–699. doi: 10.1093/jee/tox379
- AMS (2021). *Market News: Custom Reports Portal*. Available online at: <https://marketnews.usda.gov/mnp/lr-report-config> (accessed August 6, 2021).
- Archer, D. W., and Reicosky, D. C. (2009). Economic performance of alternative tillage systems in the northern corn belt. *Agron. J.* 101, 296–304. doi: 10.2134/agronj2008.0090x
- Ashford, D. L., and Reeves, D. W. (2003). Use of a mechanical roller-crimper as an alternative kill method for cover crops. *Am. J. Altern. Agric.* 18, 37–45. doi: 10.1079/AJAA2003037
- Bates, D., Mächler, M., Bolker, B., and Walker, S. (2015). Fitting linear mixed-effects models using lme4. *J. Stat. Softw.* 67, 1–48. doi: 10.18637/jss.v067.i01
- Baumgart-Getz, A., Prokopy, L. S., and Floress, K. (2012). Why farmers adopt best management practice in the United States: A meta-analysis of the adoption literature. *J. Environ. Manage.* 96, 17–25. doi: 10.1016/j.jenvman.2011.10.006
- Benbrook, C. M. (2016). Trends in glyphosate herbicide use in the United States and globally. *Environ. Sci. Eur.* 28, 3. doi: 10.1186/s12302-016-0070-0
- Carr, P. M., Cavigelli, M. A., Darby, H., Delate, K., Eberly, J. O., Fryer, H. K., et al. (2020). Green and animal manure use in organic field crop systems. *Agron. J.* 112, 648–674. doi: 10.1002/agj2.20082
- Chase, C. (2020). Using whole farm and enterprise records to make decisions. *IOWA State Univ. Exten. Outreach*. FFED33, 1–6.
- Chase, C., Delate, K., Hanlon, O., and Topaloff, A. (2019). Organic crop production enterprise budgets. *IOWA State Univ. Exten. Outreach*. FFED0027, 1–7.
- Chavas, J. P., Posner, J. L., and Hedtcke, J. L. (2009). Organic and conventional production systems in the Wisconsin Integrated Cropping Systems Trial: II. economic and risk analysis 1993–2006. *Agron. J.* 101, 288–295. doi: 10.2134/agronj2008.0055x
- Claassen, R., Bowman, M., McFadden, J., Smith, D., and Wallander, S. (2018). *Tillage Intensity and Conservation Cropping in the United States*. USDA, Economic Research Service Economic Information Bulletin Number 197.
- Crowder, D. W., and Reganold, J. P. (2015). Financial competitiveness of organic agriculture on a global scale. *Proc. Natl. Acad. Sci. U. S. A.* 112, 7611–7616. doi: 10.1073/pnas.1423674112
- Dabney, S. M., Delgado, J. A., and Reeves, D. W. (2001). Using winter cover crops to improve soil and water quality. *Commun. Soil Sci. Plant Anal.* 32, 1221–1250. doi: 10.1081/CSS-10010410
- Deines, J. M., Wang, S., and Lobell, D. B. (2019). Satellites reveal a small positive yield effect from conservation tillage across the US Corn Belt. *Environ. Res. Lett.* 14, 124038. doi: 10.1088/1748-9326/ab503b
- Delate, K., Cwach, D., and Chase, C. (2012). Organic no-tillage system effects on soybean, corn and irrigated tomato production and economic performance in Iowa, USA. *Renew. Agric. Food Syst.* 27, 49–59. doi: 10.1017/S174217051100524
- Delate, K., Duffy, M., Chase, C., Holste, A., Friedrich, H., and Wantane, N. (2003). An economic comparison of organic and conventional grain crops in a long-term agroecological research (LTAR) site in Iowa. *Am. J. Altern. Agric.* 18, 59–69. doi: 10.1079/AJAA200235
- Delbridge, T. A., Coulter, J. A., King, R. P., Sheaffer, C. C., and Wyse, D. L. (2011). Economic performance of long-term organic and conventional cropping systems in Minnesota. *Agron. J.* 103, 1372–1382. doi: 10.2134/agronj2011.0371
- Donley, N. (2019). The USA lags behind other agricultural nations in banning harmful pesticides. *Environ. Health* 18, 44. doi: 10.1186/s12940-019-0488-0
- Douglas, M. R., and Tooker, J. F. (2015). Large-scale deployment of seed treatments has driven rapid increase in use of neonicotinoid insecticides and preemptive pest management in U.S. field crops. *Environ. Sci. Technol.* 49, 5088–5097. doi: 10.1021/es506141g
- Downing, T., Chamberlain, A.-M., Gamroth, M., and Peters, A. (2013). *What Are Your Forages Worth?* Oregon State University. PNW 259, 1–4.
- Duffy, M. (2009). *Estimated Costs of Crop Production in Iowa - 2010*. Iowa State University Extension and Outreach. Ag Decision Maker. File A1-20, 1–12.
- Duffy, M. (2011). *Estimated Costs of Crop Production in Iowa - 2011*. Iowa State University Extension and Outreach. Ag Decision Maker. File A1-20, 1–13.
- Duffy, M. (2012). *Estimated Costs of Crop Production in Iowa - 2012*. Iowa State University Extension and Outreach. Ag Decision Maker. File A1-20, 1–13.
- Duffy, M. (2013). *Estimated Costs of Crop Production in Iowa - 2013*. Iowa State University Extension and Outreach. Ag Decision Maker. File A1-20, 1–13.
- Duffy, M., and Smith, D. (2008a). *Estimated Costs of Crop Production in Iowa - 2008*. Iowa State University Extension and Outreach. Ag Decision Maker. File A1-20, 1–12.
- Duffy, M., and Smith, D. (2008b). *Estimated Costs of Crop Production in Iowa - 2009*. Iowa State University Extension and Outreach. Ag Decision Maker. File A1-20, 1–12.
- Elkin, K. R., Veith, T. L., Lu, H., Goslee, S. C., Buda, A. R., Collick, A. S., et al. (2016). Declining atmospheric sulfate deposition in an agricultural watershed in central Pennsylvania, USA. *Agric. Environ. Lett.* 1, 160039. doi: 10.2134/aer2016.09.0039
- Finger, R., El Benni, N., Kaphengst, T., Evans, C., Herbert, S., Lehmann, B., et al. (2011). A Meta analysis on farm-level costs and benefits of GM Crops. *Sustainability* 3, 743–762. doi: 10.3390/su3050743
- Frasconi, C., Martelloni, L., Antichi, D., Raffaelli, M., Fontanelli, M., Peruzzi, A., et al. (2019). Combining roller crimpers and flaming for the termination of cover crops in herbicide-free no-till cropping systems. *PLoS ONE* 14, e0211573. doi: 10.1371/journal.pone.0211573
- Gagnon, A.-È., Audette, C., Duval, B., and Boisclair, J. (2016). Can the use of trichogramma ostrinae (hymenoptera: trichogrammatidae) to control ostrinia nubilalis (lepidoptera: crambidae) be economically sustainable for processing sweet corn? *J. Econ. Entomol.* 110, 59–66. doi: 10.1093/jee/tow293
- Gomiero, T., Pimentel, D., and Paoletti, M. G. (2011). Environmental impact of different agricultural management practices: conventional vs. organic agriculture. *CRC. Crit. Rev. Plant Sci.* 30, 95–124. doi: 10.1080/07352689.2011.554355
- Hanna, M. (2016). *Estimating the Field Capacity of Farm Machines*. Iowa State University Extension and Outreach. Ag Decision Maker. File A3-24, 1–5.
- Hanson, J. C., Johnson, D. M., Peters, S. E., and Janke, R. R. (1990). The profitability of sustainable agriculture on a representative grain farm in the Mid-Atlantic Region, 1981–89. *Northeast. J. Agric. Resour. Econ.* 19, 90–98. doi: 10.1017/S0899367X00002154
- Hanson, J. C., Lichtenberg, E., and Peters, S. E. (1997). Organic vs. conventional grain production in the mid-Atlantic: An economic and farming system overview. *Am. J. Altern. Agric.* 12, 2–9. doi: 10.1017/S0889189300007104
- LaCanne, C. E., and Lundgren, J. G. (2018). Regenerative agriculture: merging farming and natural resource conservation profitably. *PeerJ.* 6, e4428. doi: 10.7717/peerj.4428
- Lee, S., and McCann, L. (2019). Adoption of cover crops by U.S. Soybean Producers. *J. Agric. Appl. Econ.* 51, 527–544. doi: 10.1017/aae.2019.20
- Lenth, R. V. (2021). *emmeans: Estimated Marginal Means, aka Least-Squares Means*. Available online at: <https://CRAN.R-project.org/package=emmeans> (accessed October 06, 2021).
- Marcillo, G. S., and Míguez, F. E. (2017). Corn yield response to winter cover crops: an updated meta-analysis. *J. Soil Water Conserv.* 72, 226–239. doi: 10.2489/jswc.72.3.226
- McBride, W. D., Greene, C., Foreman, L., and Ali, M. (2015). *The Profit Potential of Certified Organic Field Crop Production*. USDA, Economic Research Service Economic Research Report Number 188. doi: 10.2139/ssrn.2981672
- Mirsky, S. B., Ryan, M. R., Curran, W. S., Teasdale, J. R., Maul, J., Spargo, J. T., et al. (2012). Conservation tillage issues: Cover crop-based organic rotational no-till grain production in the mid-Atlantic region, USA. *Renew. Agric. Food Syst.* 27, 31–40. doi: 10.1017/S1742170511000457
- Moyer, J. (2021). *Roller/Crimper No-Till: Advancing No-Till Agriculture – Crops, Soil, Equipment*. Greenly, CO: Acres U.S.A.
- Musser, W. N., Ohannesian, J., and Benson, F. J. (1981). A safety first model of risk management for use in extension programs. *North Cent. J. Agric. Econ.* 3, 41. doi: 10.2307/1349407
- NASS (2019). *2017 Census of Agriculture: Pennsylvania State and County Data*. United States Department of Agriculture, National Agricultural Statistics Service.
- NASS (2021). *Quick stats Database*. Available online at: <https://quickstats.nass.usda.gov/> (accessed August 6, 2021).
- Olmstead, J., and Brummer, E. C. (2008). Benefits and barriers to perennial forage crops in Iowa corn and soybean rotations. *Renew. Agric. Food Syst.* 23, 97–107. doi: 10.1017/S1742170507001937

- Palm, C., Blanco-Canqui, H., DeClerck, F., Gatere, L., and Grace, P. (2014). Conservation agriculture and ecosystem services: an overview. *Agric. Ecosyst. Environ.* 187, 87–105. doi: 10.1016/j.agee.2013.10.010
- Pearsons, K. A., Omondi, E. C., Heins, B. J., Zinati, G., Smith, A., and Rui, Y. (2022). Reducing tillage affects long-term yields but not grain quality of maize, soybeans, oats, and wheat produced in three contrasting farming systems. *Sustainability* 14, 631. doi: 10.3390/su14020631
- Pearsons, K. A., Omondi, E. C., Zinati, G., Smith, A., and Rui, Y. (2023). A tale of two systems: Does reducing tillage affect soil health differently in long-term, side-by-side conventional and organic agricultural systems? *Soil Tillage Res.* 226, 105562. doi: 10.1016/j.still.2022.105562
- Peigné, J., Ball, B. C., Roger-Estrade, J., and David, C. (2007). Is conservation tillage suitable for organic farming? A review. *Soil Use Manag.* 23, 129–144. doi: 10.1111/j.1475-2743.2006.00082.x
- Pittellkow, C. M., Liang, X., Linquist, B. A., van Groenigen, K. J., Lee, J., Lundy, M. E., et al. (2015). Productivity limits and potentials of the principles of conservation agriculture. *Nature* 517, 365–368. doi: 10.1038/nature13809
- Plastina, A. (2016). *Estimated Costs of Crop Production in Iowa - 2016*. Iowa State University Extension and Outreach. Ag Decision Maker, File A1-20, 1–13.
- Plastina, A. (2017). *Estimated Costs of Crop Production in Iowa - 2017*. Iowa State University Extension and Outreach. Ag Decision Maker, File A1-20, 1–13.
- Plastina, A. (2018). *Estimated Costs of Crop Production in Iowa - 2018*. Iowa State University Extension and Outreach. Ag Decision Maker, File A1-20, 1–13.
- Plastina, A. (2019). *Estimated Costs of Crop Production in Iowa - 2019*. Iowa State University Extension and Outreach. Ag Decision Maker, File A1-20, 1–13.
- Plastina, A. (2020). *Estimated Costs of Crop Production in Iowa - 2020*. Iowa State University Extension and Outreach. Ag Decision Maker, File A1-20, 1–13.
- Ramankutty, N., Mehrabi, Z., Waha, K., Jarvis, L., Kremen, C., Herrero, M., et al. (2018). Trends in global agricultural land use: Implications for environmental health and food security. *Annu. Rev. Plant Biol.* 69, 789–815. doi: 10.1146/annurev-arplant-042817-040256
- Reddy, K. N., and Norsworthy, J. K. (2010). "Managing glyphosate-resistant weeds and population shifts in midwestern US cropping systems," in: *Glyphosate-resistant Crop Production Systems: Impact on Weed Species Shifts*. eds. Nandula, V.K. (Hoboken, United States: Wiley).
- Reganold, J. P., and Wachter, J. M. (2016). Organic agriculture in the twenty-first century. *Nat. Plants* 2, 15221. doi: 10.1038/nplants.2015.221
- Reimer, M., Hartmann, T. E., Oelofse, M., Magid, J., Bünemann, E. K., and Möller, K. (2020). Reliance on biological nitrogen fixation depletes soil phosphorus and potassium reserves. *Nutr. Cycl. Agroecosyst.* 118, 273–291. doi: 10.1007/s10705-020-10101-w
- Roberts, W. S., and Swinton, S. M. (1995). The profitability of sustainable agriculture on a representative grain farm in the Mid-Atlantic Region, 1981–89: comment. *Agric. Resour. Econ. Rev.* 24, 136–137. doi: 10.1017/S1068280500003695
- Roesch-McNally, G. E., Basche, A. D., Arbuckle, J. G., Tyndall, J. C., Miguez, F. E., Bowman, T., et al. (2018). The trouble with cover crops: Farmers' experiences with overcoming barriers to adoption. *Renew. Agric. Food Syst.* 33, 322–333. doi: 10.1017/S1742170517000096
- Sanaullah, M., Usman, M., Wakeel, A., Cheema, S. A., Ashraf, I., and Farooq, M. (2020). Terrestrial ecosystem functioning affected by agricultural management systems: a review. *Soil Tillage Res.* 196, 104464. doi: 10.1016/j.still.2019.104464
- Smith, R. G., Barbercheck, M. E., Mortensen, D. A., Hyde, J., and Hulting, A. G. (2011). Yield and net returns during the transition to organic feed grain production. *Agron. J.* 103, 51–59. doi: 10.2134/agronj2010.0290
- Tilman, D., Cassman, K. G., Matson, P. A., Naylor, R., and Polasky, S. (2002). Agricultural sustainability and intensive production practices. *Nature* 418, 671–677. doi: 10.1038/nature01014
- Toliver, D. K. (2010). *Effects of no-tillage on crop yields and net returns across the United States*. (Master's thesis). University of Tennessee, Knoxville, TN, United States.
- Wallace, J., Williams, A., Liebert, J., Ackroyd, V., Vann, R., Curran, W., et al. (2017). Cover crop-based, organic rotational no-till corn and soybean production systems in the mid-Atlantic United States. *Agriculture* 7, 34. doi: 10.3390/agriculture7040034
- White, K. E., Cavigelli, M. A., Conklin, A. E., and Rasmann, C. (2019). Economic performance of long-term organic and conventional crop rotations in the Mid-Atlantic. *Agron. J.* 111, 1358–1370. doi: 10.2134/agronj2018.09.0604
- Wittwer, R. A., Bender, S. F., Hartman, K., Hydbom, S., Lima, R. A. A., Loaiza, V., et al. (2021). Organic and conservation agriculture promote ecosystem multifunctionality. *Sci. Adv.* 7, eabg6995. doi: 10.1126/sciadv.abg6995
- Zikeli, S., and Gruber, S. (2017). Reduced tillage and no-till in organic farming systems, Germany—Status Quo, Potentials and Challenges. *Agriculture* 7, 35. doi: 10.3390/agriculture7040035



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EDITED BY

Carol Williams,
University of Wisconsin-Madison,
United States

REVIEWED BY

Miguel Alfonso,
Spanish National Research Council
(CSIC), Spain
Nicholas R. Jordan,
Independent Researcher, St. Paul, MN,
United States

*CORRESPONDENCE

Russ W. Gesch
✉ russ.gesch@usda.gov

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Evaluation of soybean selection and sowing date in a continuous cover relay-cropping system with pennycress

Russ W. Gesch^{1*}, Yesuf Assen Mohammed¹ and
Heather L. Matthees²

¹USDA-ARS-NCSCL, Morris, MN, United States, ²WinField United, Land O'Lakes Inc., Arden Hills, MN, United States

Pennycress (*Thlaspi arvense* L.) is a new winter annual oilseed crop that can be integrated as a "cash cover crop" in Midwestern USA cropping systems. Relay-cropping pennycress with soybean [*Glycine max* (L.) Merr.] has been shown as an effective way to produce three crops over 2 years while providing living cover on the agricultural landscape nearly year-round. However, management improvements are needed to optimize pennycress and soybean production in this new system. A 2-year field study was conducted to evaluate three soybean interseeding dates (based on pennycress growth stage) and three soybean cultivars (varied in maturity date) on the overall productivity of this relay system. Interseeding dates were SD1 (rosette stage), SD2 (bolting stage), and SD3 (initial flowering), and soybean cultivars were MG0.2 (early), MG1.1 (standard), and MG1.7 (late). In the second season, relaying soybean reduced pennycress seed yield compared with its monocrop counterpart, but the reduction was lowest (23%) at SD2. Cultivar maturity group impacted soybean seed yields in the relay system, which for MG0.2, MG1.1, and MG1.7 averaged 2,589, 3,196, and 3,445 kg ha⁻¹, respectively. Although there was soybean yield drag associated with relay cropping, the seed yield of the MG1.7 cultivar relay interseeded at SD2 was not significantly different from a monocropped MG1.1 soybean using conventional practice (CP; winter fallow, no pennycress). The results indicate that relay interseeding of longer maturity (MG1.7) soybean for the region at the bolting stage (SD2) of pennycress optimized overall system productivity while keeping the continuous living cover on the agricultural landscape. More research will likely be needed to improve soybean selection and management regionally for this unique relay system.

KEYWORDS

cover crops, pennycress, relay-cropping, soybean, continuous cover, land use productivity

Introduction

The agricultural landscape of the Midwest Corn Belt region of the USA is dominated by summer annual cropping systems that rely heavily on corn (*Zea mays* L.) and soybean production (Sindelar et al., 2017). Because of simplified crop rotations, agricultural diversity has suffered (Aguilar et al., 2015) and, consequently, so has agronomic and environmental sustainability. Intense management (e.g., tillage, fertilizers, and pesticides) of only a few crops to maintain high yields has resulted in unintended negative consequences including reduced water quality (Kladivko et al., 2014), soil erosion (Reicosky, 2015), increase in herbicide-resistant weeds (Mortensen et al., 2012), and declining pollinator diversity and abundance (Eberle et al., 2015; Thom et al., 2018). Furthermore, there is a growing societal trend among consumers of being more concerned about where and how their food is produced, influencing large food and beverage companies to source ingredients from more sustainable systems (Ringquist et al., 2016). A potential strategy to mitigate some of these issues is to employ perennial or annual cropping systems that provide diversity and keep living cover on the agricultural landscape as long as possible throughout the year (Heaton et al., 2013; Ryan et al., 2018). An obvious choice to do this in annual cropping-based systems is the use of cover crops during the fallow season. However, cover crop adoption in the Midwest Corn Belt of the USA has been slow, and farmers often cite the cost of establishing covers and little or no near-term economic return as reasons for this (Myers and Watts, 2015).

Pennycress is a member of the Brassicaceae family and is a new oilseed crop that has gained considerable attention as a potential cash cover crop that can be grown between summer annual commodity crops (Sindelar et al., 2017; Cubins et al., 2019). Several studies, mostly conducted in the upper Midwest USA, have demonstrated that soybean can be successfully double-cropped or relay-cropped with pennycress (Phippen and Phippen, 2012; Johnson et al., 2017; Bishop and Nelson, 2019; Ott et al., 2019; Hoerning et al., 2020). The impact of pennycress on soybean yield, when used in a double-cropping or relay-cropping scenario, has been mixed. For example, in the southern regions of the Corn Belt, pennycress had little or no effect on double-cropped soybean yields as compared with conventional monocrop soybean (Phippen and Phippen, 2012; Bishop and Nelson, 2019). However, in the northern Corn Belt, soybean yield reductions of 18–30% have been reported to be associated with double-cropping and relay-cropping with pennycress as compared with monocrop soybean (Johnson et al., 2017; Ott et al., 2019). Differences between the regions are likely due to a longer growing season in the southern Corn Belt. Regardless

of the potential soybean yield drag associated with double-cropping and relay-cropping with pennycress, one thing that remains consistent is that total seed and oil yield per land area (i.e., pennycress + soybean) are generally greater than conventionally producing a single soybean crop (Cubins et al., 2019).

In the Corn Belt region, pennycress is primarily being targeted for integration into corn and soybean systems (Sindelar et al., 2017; Bishop and Nelson, 2019). However, full-season grain corn due to its long growing season presents challenges for establishing pennycress, especially in the northern Corn Belt. Generally, pennycress seed yield and oil content are maximized by planting in early- to mid-September (Dose et al., 2017). Because grain corn is typically harvested in late autumn, there is often little time to directly plant and establish pennycress before the soil freezes. A few studies have evaluated interseeding pennycress into a standing corn crop at various stages of growth (Nolan et al., 2018; Bishop and Nelson, 2019; Mohammed et al., 2020) with mixed results. For instance, Mohammed et al. (2020) demonstrated that good pennycress establishment was achievable by interseeding with a highboy device at the late stages of corn development (R4 to R6), but this did not translate into high seed yields (Patel et al., 2021), likely due to suppressed growth caused by the high amount of corn residue following harvest. More consistent establishment and seed yield results have been achieved by direct planting pennycress in early September following short-season summer crops such as spring wheat (*Triticum aestivum* L.) (Dose et al., 2017; Ott et al., 2019) and corn harvested for silage (Hoerning et al., 2020) where there is minimal crop residue. In the northern Corn Belt, there is a significant hectareage of small grain cereals such as spring wheat grown followed by fallow soil until the next spring. Therefore, introducing pennycress as a cash cover crop to keep the soil covered in wheat–soybean systems is needed, but the information is limited to optimize pennycress and soybean production.

The relay-cropping system, which involves interseeding soybean into standing pennycress such that their lifecycles overlap during the growing season, effectively keeps living cover on the field year-round. The environmental benefits of using this system are manifold. Weyers et al. (2019) demonstrated that autumn-sown pennycress and winter camelina (*Camelina sativa* L.) grown in a relay system with soybean reduced nitrate N in soil water by as much as 89% in spring as compared with conventional practices of keeping the soil fallow between summer annual crops. This has a significant implication for water quality given that soils in the Midwest Corn Belt region are most prone to N loss by leaching and runoff in the spring (Strock et al., 2004). Moreover, pennycress can reduce total suspended solids in spring runoff from snow melt and rains by as much as 75% compared with soil left fallow over the winter (Weyers et al., 2021), indicating its ability to prevent soil erosion. Pennycress also suppresses spring and early summer weeds (Johnson et al.,

Abbreviations: CP, conventional practice; MG, maturity group; SSB, sole soybean; RSB, relayed soybean; SD, seeding date.

2015) by as much as nearly 100% (Hoerning et al., 2020) when used in a relay system with soybean, resulting in less herbicide use in the subsequent soybean crop. An abundance of pollinating insects visit pennycress when it is flowering (Eberle et al., 2015), and its flowers provide pollen and nectar resources for pollinators (Thom et al., 2016, 2018).

While growing pennycress has many positive environmental effects, pennycress seed oil and meal, like that of rape seed (*Brassica napus* L.), is presently high in erucic acid (C22:1) and glucosinolates, which are antinutritional and therefore not desirable for food and feed use. However, extensive work is underway to develop commercially viable pennycress genotypes that are low in glucosinolates and possess seed oil profiles conducive to food and feed uses (Chopra et al., 2020). While low glucosinolate pennycress genotypes are being developed, the near-term markets for currently available pennycress seed oil will likely be for biofuels. Pennycress seed oil has been demonstrated to be a good feedstock for making biodiesel (Moser et al., 2009), renewable aviation fuel (Fan et al., 2013), and biolubricants (Cermak et al., 2015).

Although research shows that pennycress can successfully be double-cropped and relay-cropped with soybean and other short-season summer annual crops (Cubins et al., 2019; Moore et al., 2020), little work has focused on improving agronomic management practices for such systems. The present study was designed to address optimizing the timing for relay sowing of soybean into pennycress and explored the effect of soybean maturity. We hypothesized that relaying soybean into pennycress as late as possible during its development (e.g., during bolting or initiation of reproduction) may improve soybean yield by reducing the amount of time the two crops overlap. We also hypothesized that using a longer maturing soybean than normally used for the region might improve productivity by allowing soybean to remain in vegetative growth longer during and after the overlap period. The overall goal of the study was to improve relay-crop soybean yields and minimize yield drag while simultaneously maintaining high pennycress seed yields. The objectives of the study were to determine the effects of soybean interseeding (relay seeding) date and maturity group on pennycress and soybean growth performance, seed yield, and seed qualities.

Materials and methods

Experimental location and cultural practices

The study was done over two growing seasons (2015–2016 and 2016–2017) at the USDA-ARS Swan Lake Research Farm near Morris, MN, USA, located at 45°40'N, 95°48'W, and 345 m a.s.l. The soil at the experiment site was predominantly a Barnes loam (fine-loamy, mixed, superactive, and frigid Calcic

Hapludolls). The long-term (over the last 30 years) average annual air temperature at the location is 5.7°C, and the long-term average yearly precipitation is 670 mm.

The experimental design was a randomized complete block with a split-plot arrangement. The main plots (9.1 m by 9.1 m) consisted of three soybean sowing dates based on the growth stage of pennycress, which were SD1 (rosette stage), SD2 (bolting or stem elongation stage), and SD3 (initial flowering stage). The subplots, which were 3 m by 9.1 m in size, consisted of three different soybean maturity groups representing early (MG0.2), standard (MG1.1), and late (MG1.7) cultivars for the region. The three soybean cultivars used in both years were from CROPLAN Genetics and were R2T0200 (early), R2C1100 (standard), and R2C1750 (late). The experimental design included three monocrop soybean check treatments that involved no-till sowing of MG0.2, MG1.1, and MG1.7 all at SD2 after winter fallow (no pennycress). Among these check treatments, soybean MG1.1 sown at SD2 was designated as conventional practice (CP), and this is the standard soybean maturity and seeding date for the region considered as normal (i.e., conventional). The CP was used in a planned orthogonal contrast analysis with the pennycress-relayed soybean treatments. The same CP monocrop soybean check treatment was used for contrast analysis in a companion study (Mohammed et al., 2022) for relay-cropping soybean with winter camelina.

Pennycress accession MN106 used for the study originated from a collection made of a natural wild population near Coates, Minnesota, USA. Pennycress was no-till sown with an InterSeeder drill (InterSeeder Technologies, Woodward, PA) into spring wheat stubble (i.e., previous crop) at a seeding rate of 9 kg ha⁻¹ on 19 cm spaced rows leaving every fourth row unseeded (i.e., “skip row”) for relay seeding soybean the following spring. The day before sowing pennycress, 1.12 kg a.i. ha⁻¹ of trifluralin (α, α, α -trifluoro-2,6-dinitro-N,N, -dipropyl-p-toluidine) was applied and lightly incorporated with one pass of a no-till drill for weed control. The pennycress was sown on 10 September 2015 (1st season) and 13 September 2016 (2nd season). The following spring, pennycress plots were broadcast fertilized at a rate of 78–34–34 kg N-P-K ha⁻¹ on 4 April 2016 and 11 April 2017 using urea, diammonium phosphate, and potassium chloride.

Relayed and monocrop soybeans were all sown at a rate of 432,000 seeds ha⁻¹ on 76 cm row spacing using a John Deere MaxEmerge seeder (Model 1730, Moline, IL). The relayed soybean was sown in the skip rows (76 cm row spacing), and all plots contained four rows of soybean. A diagram of the row spacing scheme used is shown by Mohammed et al. (2022). The first soybean sowing date (SD1) was 18 April 2016 and 23 April 2017, SD2 was sown on 5 May in both years, and SD3 was sown on 15 May in both years. No fertilizer was applied to either relayed or monocrop soybean. Weeds were controlled by applying glyphosate [N-(phosphonomethyl)glycine] at 1.3 kg a.i. ha⁻¹ to all plots containing soybean on 28 June 2016 (following

the pennycress harvest), and another application at the same rate was made on 1 August. In 2017, the same rate of glyphosate was applied to monocrop soybean on 7 June and applied on 7 July to all relayed soybean for weed control.

Plant measurements

At each relay soybean sowing date (SD1, 2, and 3), the height of pennycress was measured from three randomly chosen plants from each plot and averaged. At pennycress harvest, the heights of both pennycress and soybean were measured on six randomly chosen plants in each of the relay and monocrop treatment plots. Pennycress was harvested when at least 90% of its silicles were yellowish-brown in color and seeds were black indicating full maturity (Cubins et al., 2022). Pennycress seeds were harvested with a plot combine (Hege 160, Waldenburg, Germany) on 20 June 2016 and 19 June 2017 from the center of the plot (1.5 m wide), and the plot length was measured to calculate the net plot area. For the relay treatments, this consisted of straddling two rows of soybean to harvest six rows of pennycress, similar to what (Mohammed et al., 2022) have described for winter camelina relay-cropped with soybean. Seeds were dried at 65°C to constant weight before screen cleaning to remove debris. Pennycress seed yields were adjusted to 100 g kg⁻¹ moisture.

At the R7 growth stage prior to full maturity, six soybean plants were randomly sampled from all plots containing soybean (controls and relay treatments) and brought back to the lab where height, node, and pod numbers per plant were measured. At soybean harvest, the number of plants was measured in 1 m of the row from either of the two center rows of each plot to determine plant density. Soybean was combined-harvested (Hege 160, Waldenburg, Germany) for grain at full maturity (R8) by taking the center two rows, and the exact plot length was measured to determine the harvest area for both monocrop and relay-crop treatments. Soybean harvest date varied by year and treatment. In 2016, all (monocrop and relay-crop) MG0.2 soybeans were harvested on 21 September, all MG1.1 soybeans were harvested on 29 September, and all MG1.7 soybeans were harvested on 13 October. In 2017, all monocrop and relay-crop soybeans were harvested on 11 October except for SD3 relayed MG1.7 soybean, which was harvested on 18 October. The grain was dried to constant weight at 65°C and screen cleaned for yield determination. Soybean grain yields were adjusted to 130 g kg⁻¹ moisture.

Weather variables were measured and recorded at an automated weather station located at the experiment site. Daily average air temperature (2-m height) and daily precipitation were used for determining mean monthly temperature and accumulated precipitation (Figure 1). The long-term average (LTA) temperature and precipitation were based on data recorded between 1987 and 2017 (Figure 1).

Seed oil and protein analysis

The seed oil content of pennycress and soybean was measured by pulsed nuclear magnetic resonance (pNMR) using a Minispec mq10 (Bruker, The Woodlands, TX). Harvested seed from each plot, 5 g for pennycress and 6 g for soybean, was measured by pNMR as previously described (Gesch et al., 2014) after calibrating the instrument independently with pure pennycress and soybean oil, respectively. In brief, clean seed samples were dried for 4 h at 130°C and cooled in a desiccator for 15 min before measuring oil content. After measuring oil, the seed samples were ground to a fine powder in a Wiley Mill, and the total percent N was measured by combustion analysis using a LECO CN828 (LECO Corp., St. Joseph, MI). Crude protein content was estimated by multiplying percent N by 6.25 (Mariotti et al., 2008).

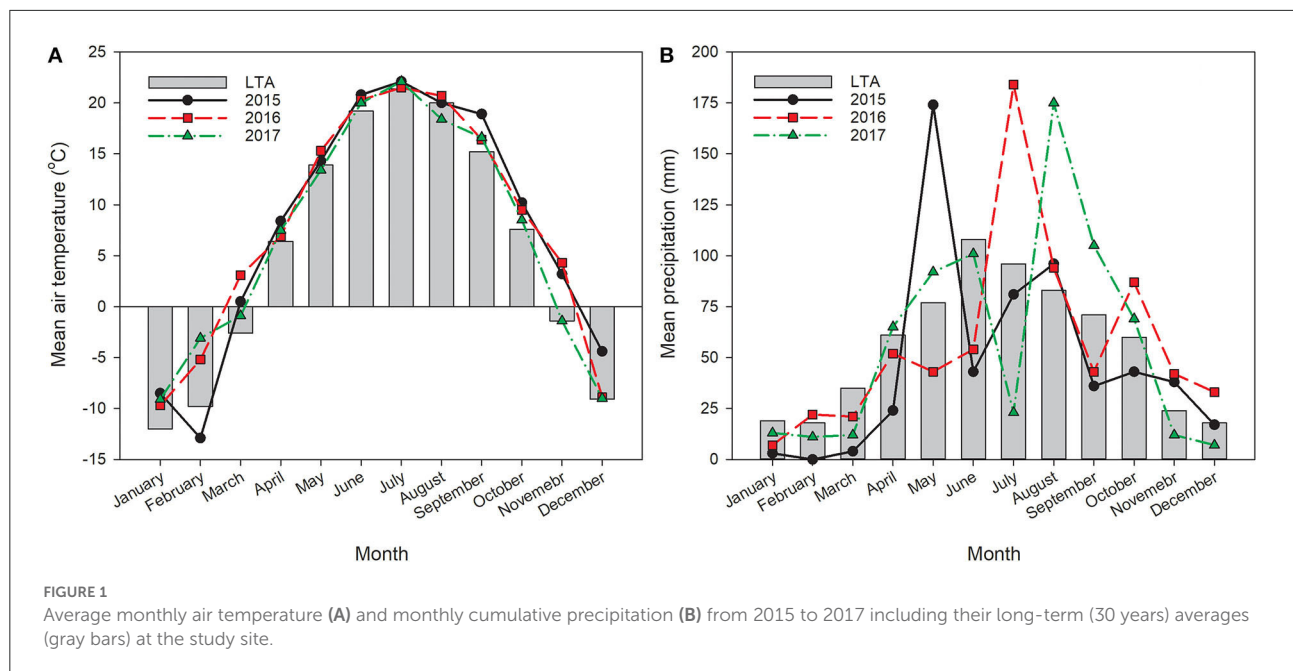
Statistical analysis

The MIXED procedure of SAS 9.4 (SAS Institute, Cary, NC) was used for data analysis (SAS Institute, 2014). The effect of year on pennycress seed yield was significant; thus, data were analyzed by year using replication as a random effect and sowing date (SD), soybean maturity group (MG), and their interaction (SD × MG) as fixed effects. However, pennycress and soybean plant heights at pennycress harvest did not differ by year and were therefore combined across years with year and replication as random effects and sowing date (SD), soybean maturity group (MG), and their interaction (SD × MG) as fixed effects. Soybean seed yields and plant attributes also did not differ by year, and data were combined across years using year and replication as random effects and sowing date (SD), soybean maturity group (MG), and their interaction (SD × MG) as fixed effects. When ANOVA showed significant treatment effects ($P \leq 0.05$), LSD at $\alpha = 0.05$ was used to differentiate treatment means. Planned orthogonal contrast analysis was performed with SAS to compare the monocrop conventional practice (CP) with the relay treatments, and the results were declared significant when P -values were <0.05 .

Results

Weather

From 2015 to 2017, monthly average air temperatures tended to be greater between September and March than the LTA except for February in 2015, which was lower, and November and December of 2017, which were on par with the LTA (Figure 1). Between April and August of all 3 years, air temperatures were generally similar to the LTA. Precipitation distribution varied widely among years with considerably



greater amounts of rainfall than normal in May 2015, July 2016, and August 2017 (Figure 1). The total accumulated precipitation in 2016 and 2017, however, was only 12 and 15 mm greater than the LTA (670 mm total), while 2015 was drier, with a total accumulation of 111 mm less than the LTA. It is also important to note that in September and October of 2015, during pennycress sowing and establishment, precipitation was low with greater temperatures than normal, making for quite dry conditions.

Pennycress seed yield, oil content, and plant height

Pennycress seed yield across relayed treatments differed considerably between years, and there was a relay sowing date by year interaction (Table 1). Across all relay treatments, pennycress seed yield averaged 352 kg ha^{-1} in 2016 and 823 kg ha^{-1} in 2017. In 2017, pennycress seed yield was greater for SD2 than SD3, but this difference was not observed in 2016 (Table 2).

In 2016, pennycress seed yields in the relay treatments were statistically the same as the monocrop control (i.e., no soybean relayed into it) (Table 1). However, in 2017, monocrop pennycress yielded $1,185 \text{ kg ha}^{-1}$, which was generally greater than the seed yield of pennycress from relayed treatments (823 kg ha^{-1}). The only exception in 2017 was for the SD2 MG1.7 relay treatment where the average pennycress yield was $1,071 \text{ kg ha}^{-1}$ and not significantly different than the monocrop control.

TABLE 1 Analysis of variance table showing seeding date (SD), soybean maturity group (MG), year, and their interactions on pennycress seed yield and oil content in the relay-cropped treatments, and contrast analysis of monocrop pennycress (MPC) with pennycress relay-sown with soybean (RPC) in 2016 and 2017.

Effect	Seed yield (kg ha^{-1})	Oil content (g kg^{-1})
F value and significance		
SD	0.95	0.11
MG	1.10	0.92
Year	154.18***†	0.37
SD × MG	0.58	0.40
SD × Year	5.29**	0.19
MG × Year	1.47	1.34
SD × MG × Year	0.49	0.41
Contrast	Seed yield (kg ha^{-1})	Oil content (g kg^{-1})
P > F		
MPC vs. RPC 2016	0.275	0.024
MPC vs. RPC 2017	0.0004	0.022

† Denotes level of significance ** < 0.001 and *** < 0.0001.

Pennycress oil content was quite stable across the years and was not impacted across relayed soybean treatments (Table 1). In 2016, oil content across all relay treatments averaged 337 g kg^{-1} , while in 2017, it was 336 g kg^{-1} . However, in both years of the study, the oil content of monocrop pennycress was slightly

TABLE 2 Pennycress seed yield as affected by the interaction of sowing date (SD) and year.

Year	Sowing date	Seed yield (kg ha ⁻¹)
2016	SD1	358 c
	SD2	303 c
	SD3	394 c
2017	SD1	844 ab
	SD2	917 a
	SD3	707 b

Means followed by a different letter are significantly different at the *P*-value of < 0.05 level using LSD.

TABLE 3 Pennycress and soybean plant heights in the relay treatments of sowing date (SD) and soybean maturity group (MG) at the time of pennycress harvest for data combined over years (2016 and 2017).

Main factor	Level	Pennycress height (cm)	Soybean height (cm)
SD	SD1	67.9 a	21.1 a
	SD2	64.6 a	21.9 a
	SD3	67.5 a	15.8 b
MG	MG0.2	65.4 b	19.6 a
	MG1.1	66.2 ab	20.2 a
	MG1.7	68.3 a	19.2 a

For a given factor, means within a column followed by a different letter are significantly different at the *P*-value of < 0.05 level using LSD.

less but significantly different than that of plants in the relay treatments (Table 1), averaging 325 g kg⁻¹ in 2016 and 330 g kg⁻¹ in 2017. The average oil content between relayed and monocrop treatments over both years was relatively small (0.9%) and was not practically significant.

Over both years of the study, the height of pennycress plants, at the time soybean was relay-sown, averaged 11 ± 3.0 cm StdDev at SD1, 19 ± 8.6 cm at SD2, and 46 ± 14.8 cm at SD3. At pennycress harvest in the relay treatments, soybean MG but not SD affected pennycress height (*P* < 0.05), and there was no interaction. Pennycress plants were on average 2.9 cm taller in plots relayed with the MG1.7 soybean than those relay-sown with the MG0.2 cultivar (Table 3). For the relayed soybean plants, height at pennycress harvest was affected by SD (*P* < 0.05) but not MG. For soybean in SD3, plants were 5.3 cm and 6.1 cm shorter than the soybean in SD1 and SD2, respectively, at the time pennycress was harvested (Table 3). For comparison, at the time of pennycress harvest, the average height of monocrop soybean planted at SD2 was 20.2, 17.6, and 16.7 cm for the MG0.2, MG1.1, and MG1.7 cultivars, respectively. Except for the MG0.2 cultivar, the relayed MG1.1 and 1.7 soybeans were slightly taller (about 3 cm) than their monocrop counterpart at the pennycress harvest.

Soybean seed yield, yield components, and quality

The seed yield of relayed soybean was affected by SD and MG, while seed oil and protein content were only influenced by MG (Table 4). Seed yield was lowest for SD1 and did not differ between SD2 and SD3 (Table 5). Both MG1.1 and 1.7 relayed soybeans yielded greater than MG0.2, but MG1.7 and MG1.1 had statistically similar yield. Seed oil content was greater for the MG1.1 and 1.7 soybeans than the MG0.2 cultivar, but the earlier maturing MG0.2 had greater protein content (Table 5).

Late season relayed soybean plant height (taken at R7) only differed by MG (Table 4), with the MG1.7 being the tallest and the MG0.2 being the shortest cultivar (Table 5). Relayed soybean node, branch, and pod numbers per plant differed by SD and MG (Table 4). All these yield components were greatest in SD1, and branches per plant continued to decline between SD1 and SD3 (Table 5). Similar to height, biomass per plant and the yield components of node, branch, and pod numbers were all lowest for the MG0.2 cultivar (Table 5). Although node, pod numbers, and biomass per plant did not differ between the MG1.1 and 1.7 soybeans, the branch number was slightly greater for the MG1.1 cultivar. Biomass per plant of the relayed MG0.2 soybean was 58% lower than the average of the relayed MG1.1 and 1.7 soybeans (Table 5).

Relay sowing date (SD) and soybean MG affected soybean plant population density (Table 4). Relay sowing at SD3 favored greater plant density (Table 5). Across cultivars, plant density at SD1 was 13% lower than for SD3 at soybean harvest. Both the MG0.2 and 1.1 relayed soybeans gave similar plant densities at harvest, but the MG1.7 cultivar was on average 8% greater.

Compared with the conventional practice (CP) of winter fallow (no pennycress) followed by monocrop MG1.1 soybean sown at an average time of early May (SD2), the relay-cropped soybean yields were generally lower (Table 6). The yield reduction for relayed soybean ranged from as high as 38% (SD1 MG0.2) to as low as 10% (SD2 MG1.7). As shown by the contrast analysis (Table 6), the seed yield of the relayed SD2 MG1.7 was not statistically different from the CP treatment. Seed oil and protein content did not differ between the CP and relayed soybean. However, generally, the oil content was lower and protein content greater for the relayed MG0.2 cultivar than the CP soybean. Soybean plant height and biomass greatly differed between CP and relayed soybean with CP plants always taller and almost always heavier than the relayed soybeans, except for the SD1 MG1.1 treatment (Table 6). Node number per plant differed between CP and relayed soybean, but primarily because node numbers were consistently less for the relayed MG0.2 soybean but were not different from the CP treatment for the relayed MG1.1 and 1.7 cultivars. Branch and pod numbers varied among the treatments but overall were not found to differ between CP and relayed soybean (Table 6).

TABLE 4 Analysis of variance table showing the effects of sowing date (SD), soybean maturity group (MG), and their interaction (SD × MG) on relayed soybean plant and seed attributes for data combined over years (2016 and 2017).

Effect	Seed yield (kg ha ⁻¹)	Oil content (g kg ⁻¹)	Protein content (g kg ⁻¹)	Height at R7 (cm)	Node	Branch	Pod	Biomass (g plant ⁻¹)	Plant density (plants ha ⁻¹)
					Number plant ⁻¹				
SD	4.90* [†]	0.54	0.31	2.91	4.49*	13.85***	5.42**	1.77	12.84***
MG	21.67***	25.18***	33.23***	103.19***	119.70***	42.49***	24.73***	24.80***	5.96**
SD × MG	0.53	0.32	0.49	0.14	0.64	1.23	0.90	1.38	1.51

Shown are F values followed by the level of significance. [†] Denotes level of significance * < 0.05, ** < 0.001, and *** < 0.0001.

TABLE 5 Mean plant and seed attributes for the soybean maturity group (MG) cultivars at different sowing dates (SD) in a relay system with pennycress for data combined over years (2016 and 2017).

Main factor	Treatment	Seed yield (kg ha ⁻¹)	Oil content (g kg ⁻¹)	Protein content (g kg ⁻¹)	Height at R7 (cm)	Node	Branch	Pod	Biomass (g plant ⁻¹)	Plant density (plant ha ⁻¹)
						Number plant ⁻¹				
SD	SD1	2,858 b	205.8	392.5	58.48	14.85 a	17.02 a	41.63 a	20.18	281,605 c
	SD2	3,271 a	206.2	389.2	63.34	14.10 b	15.35 b	35.13 b	18.04	304,024 b
	SD3	3,101 ab	205.0	391.9	63.08	13.94 b	14.13 c	33.85 b	17.49	322,616 a
MG	MG0.2	2,589 b	201.1 b	402.0 a	49.77 c	11.40 b	12.73 c	26.60 b	12.48 b	293,088 b
	MG1.1	3,196 a	207.3 a	392.8 b	64.41 b	15.61 a	17.74 a	41.46 a	22.26 a	296,369 b
	MG1.7	3,445 a	208.6 a	378.9 c	70.73 a	15.89 a	16.03 b	42.55 a	20.98 a	318,788 a

Means within a column for a given factor followed by different letters are significantly different at the *P*-value of <0.05 level using LSD.

A planned contrast analysis was also done between the relayed soybean cultivars sown at SD2 and their monocrop counterparts sown on the same date (Table 7). Compared with their monocrop counterparts, relayed soybean generally had lower seed yields, except for the MG1.7, which did not significantly differ from its monocrop control. Seed quality, both oil and protein contents, did not differ between monocrop and relay soybeans (Table 7). Contrast analysis showed plant height was taller and biomass larger for monocrop soybean compared with relayed soybean. Node, branch, and pod numbers were lower for the relayed MG0.2 soybean compared with its monocrop control. However, there was generally no difference in these yield components when comparing the relayed MG1.1 and 1.7 soybeans with their monocrop counterpart (Table 7).

Discussion

Pennycress

The overarching goal of this study was to determine whether relayed soybean yield could be improved by adjusting the sowing date and maturity of soybean without reducing pennycress yield. Regardless of relay or monocrop treatments, pennycress seed yields were much lower in 2016 than in 2017. Although pennycress plant density was not measured, pennycress stands were noticeably less dense (field observation) in 2016. It is likely that the pennycress stand was reduced due to the lack of precipitation and dry conditions during early autumn (September and October) of 2015, followed by a dry spring (March to June) in 2016, which could have resulted in poor emergence and early growth of plants. Pennycress seed germination and emergence are highly dependent on adequate soil moisture (Hazebrook and Metzger, 1990), and low seedling emergence in west central Minnesota of the USA has previously been demonstrated to be linked to low precipitation and dry soil in September and October (Royo-Esnal et al., 2015). Early to mid-September has been shown to be a near optimal time to sow pennycress in the northern Corn Belt of the USA (Dose et al., 2017). Furthermore, when sowing in early autumn, Johnson et al. (2015) have shown that the amount of precipitation during the pennycress growing season is closely associated with seed yield, increasing with increased precipitation.

Pennycress yields were not affected by relay-sowing soybean in 2016, but were in 2017, where generally yields were lower in the relayed treatments as compared with monocrop pennycress. The only exception was the SD2 MG1.7 treatment where seed yield was not different from the monocrop control. The difference was most pronounced in the SD3 treatments where the pennycress plants were initially flowering and averaged 46 cm tall at the time soybean was relay-sown. A similar response was noted in a companion study with winter camelina (Mohammed et al., 2022). However, in that study, when

compared with the monocrop check, camelina seed yields were only reduced when relay sowing soybean at the initial flowering of camelina (SD3) and not at the earlier rosette or bolting stages. The decline in pennycress yield associated with relay-cropping was most likely due to damage caused by wheel traffic of equipment used for sowing soybean. In 2017, pennycress plant density was measured in the spring prior to interseeding soybean and was found to average 288 ± 96 StdDev and 261 ± 73 plants m^{-2} for the monocrop and relay-cropped treatments, respectively. This small difference in plant density was unlikely the reason for yield differences between the two systems. Nevertheless, the yield reduction in 2017 was the least for SD2, which averaged across all three MGs was 23% lower than the monocrop control. Previous research on relay-cropping pennycress and soybean indicated that the less time soybean remained under the pennycress canopy (i.e., lifecycles overlap), the greater its yield (Ott et al., 2019; Hoerning et al., 2020). The results of the present study indicate that soybean can be sown into pennycress at the bolting stage (SD2) to minimize pennycress yield reduction while allowing less time for the two crops to overlap than relaying at the rosette stage (SD1).

In both years of the study, relay-cropping slightly, but consistently, increased pennycress seed oil content by an average of about 9 g kg^{-1} compared with monocrop pennycress. Although this difference ($\sim 1\%$) was statistically significant, it was not enough to be of practical agronomic significance. A similar response was reported by Mohammed et al. (2022) for winter camelina when it was relayed-cropped with soybean and was most likely due to less available soil N in the relay system caused by competition between the two crops for N uptake during their overlap period. Lower available N for oilseed crops is often associated with greater seed oil synthesis (Gehring et al., 2006).

Soybean

A vital aspect of the relay system is being able to harvest pennycress without damaging soybean. When pennycress was harvested, the height difference between pennycress and soybean in the relay treatments was large enough to keep the cutting bar of the combine above the soybean without severing the soybean and causing damage.

Relayed soybean yield was impacted by the sowing date. Across cultivars, soybean relayed at SD1 yielded 13 and 8% less than those sown at SD2 and SD3, respectively. However, the number of nodes, branches, and pods was greater for plants sown at SD1 than at SD2 and SD3. Generally, higher soybean yields are correlated with a greater number of yield components per plant (Akhter and Sneller, 1996). Moreover, an increase in yield components is often associated with lower soybean plant density, which can compensate for yield in lower populations (Carpenter and Board, 1997). However, in the present study, the

TABLE 6 Contrast analysis for CP vs. the different relay treatments and mean soybean agronomic parameters for data combined over years (2016 and 2017).

Contrast	Seed yield (kg ha ⁻¹)	Oil (g kg ⁻¹)	Protein (g kg ⁻¹)	Plant height (cm)	Node	Branch	Pod	Biomass (g plant ⁻¹)
					Number plant ⁻¹			
	P > F							
CP [†] vs. all relayed	<0.0001	0.3897	0.4530	<0.0001	0.0028	0.5529	0.1205	<0.0001
CP vs. relayed SD1 MG0.2	<0.0001	0.0084	0.0072	<0.0001	<0.0001	0.1429	0.0043	<0.0001
CP vs. relayed SD1 MG1.1	<0.0001	0.8558	0.5367	<0.0001	0.7891	<0.0001	0.1011	0.1914
CP vs. relayed SD1 MG1.7	0.0031	0.4236	0.2813	<0.0001	0.2514	0.0142	0.2911	0.0012
CP vs. relayed SD2 MG0.2	<0.0001	0.0467	0.0959	<0.0001	<0.0001	0.0132	0.0007	<0.0001
CP vs. relayed SD2 MG1.1	0.0235	0.7624	0.4852	<0.0001	0.9672	0.0051	0.6044	0.0011
CP vs. relayed SD2 MG1.7	0.1335	0.6764	0.1014	<0.0001	0.9017	0.6198	0.5161	0.0004
CP vs. relayed SD3 MG0.2	<0.0001	0.0070	0.0167	<0.0001	<0.0001	0.0014	0.0001	<0.0001
CP vs. relayed SD3 MG1.1	0.0068	0.6285	0.2630	<0.0001	0.4473	0.6952	0.1410	0.0002
CP vs. relayed SD3 MG1.7	0.0222	0.6260	0.1240	<0.0001	0.9344	0.9791	0.9954	0.0029
Treatments	Means							
Conventional practice (CP)	4,072	207	388	86	16	15	42	30.0
Means for all relayed	3,077	206	391	62	14	16	37	19.0
Relayed SD1 MG0.2	2,505	201	405	47	12	14	29	13.1
Relayed SD1 MG1.1	2,830	208	392	60	16	20	49	26.5
Relayed SD1 MG1.7	3,238	209	381	68	16	17	47	21.0
Relayed SD2 MG0.2	2,711	202	398	51	11	13	27	13.0
Relayed SD2 MG1.1	3,443	208	392	66	16	18	40	20.9
Relayed SD2 MG1.7	3,660	208	377	73	16	16	39	20.2
Relayed SD3 MG0.2	2,551	200	403	51	11	12	24	11.3
Relayed SD3 MG1.1	3,314	206	395	67	15	15	35	19.4
Relayed SD3 MG1.7	3,437	208	378	72	16	15	42	21.8

[†] CP, conventional practice (monocrop SD2 MG1.1); SD1, SD2, and SD3 are soybean seeding dates at rosette, bolting, and initial flowering growth stages of pennycress, respectively. MG0.2, MG1.1, and MG1.7 are soybean maturity groups represented by three soybean genotypes.

small but significant increase in nodes, branches, and pods in the SD1 treatment did not compensate in yield for the lower plant density. Although seed size, which was not measured, cannot be ruled out, the most likely reason for the lower yield of SD1 soybean was because of lower plant density at harvest.

Soil moisture availability and light interception are critical factors for the survival and development of interseeded soybean in a relay system (Duncan and Schapaugh, 1997; Gesch and Johnson, 2015; Ott et al., 2019). The reduction in plant density and lower yields of SD1 relayed soybean most likely resulted from extended competition between the two crops (i.e., pennycress and soybean) for soil moisture, nutrients, and light, but especially moisture. In 2016, the precipitation was below normal for May and June (Figure 1) when relayed soybean was emerging and vegetatively developing under the pennycress canopy, which likely intensified competition for available water. Furthermore, it has been demonstrated that the longer soybean remains under a canopy in a relay system, the more intensified competition is for resources, which often leads to reduced soybean plant stands and yields (Wallace et al., 1992; Duncan and Schapaugh, 1997). In the present study, from sowing to pennycress harvest, relayed soybean was under the pennycress canopy for 57 to 63 d in SD1, 45 to 46 d for SD2, and 35 to 36 d for SD3. In a related but independent study where soybean was relayed into pennycress, Hoerning et al. (2020) reported that soybean plant population density and seed yield were greatly reduced compared with monocrop soybean at Morris, Minnesota, in 2016. However, in the same study, plant stands were unaffected by relaying at two other Minnesota sites in the same year (Lamberton and Rosemount). Hoerning et al. (2020) concluded that early season drought was the main cause of the soybean population density and yield reductions. However, in that study, soybean was relayed into broadcast solid seeded pennycress rather than using a direct-drilled skip-row pattern like in the present study. Previous research on relay intercropping of soybean with small grain cereals indicates that skip-row patterns tend to reduce interplant competition and consistently result in greater soybean yields (Duncan et al., 1990; Duncan and Schapaugh, 1997).

Whether relayed or grown as a monocrop, both the MG1.1 and MG1.7 soybeans yielded the MG0.2 cultivar (Tables 5, 7). Regardless of the relay sowing date, the MG0.2 soybean had fewer yield components and plants tended to be shorter and have less biomass than either of the longer maturity soybean cultivars. This result is not surprising given that generally earlier maturing soybean for a region tend to be lower yielding if all other management factors (e.g., plant population and row spacing) are equal (Edwards and Purcell, 2005). However, our study is one of the first to explore the effect of soybean maturity in a relay system with pennycress. In a companion study where the same soybean cultivars were relayed into winter camelina, Mohammed et al. (2022) showed that the longer maturing MG1.7 gave a clear advantage over the commonly

TABLE 7 Contrast analysis for the monocrop controls at SD2 vs. the relay treatments at SD2 and mean soybean agronomic parameters for data combined over years (2016 and 2017).

Contrast	Seed yield (kg ha ⁻¹)	Oil content (g kg ⁻¹)	Protein content (g kg ⁻¹)	Plant height (cm)	Node	Branch	Pod	Biomass (g plant ⁻¹)
Pr > F								
Monocrop SD2 MG0.2 vs. relay SD2 MG0.2	0.0122	0.9561	0.7862	<0.0001	<0.0001	0.0077	0.0088	<0.0001
Monocrop SD2 MG1.1 vs. relay SD2 MG1.1	0.0116	0.8089	0.5200	<0.0001	0.9770	0.0147	0.6535	0.0063
Monocrop SD2 MG1.7 vs. relay SD2 MG1.7	0.0708	0.7903	0.9304	<0.0001	0.5461	0.1598	0.4416	0.0013
Means								
Monocrop SD2 MG0.2	3,336	202	400	84	16	16	40	33.6
Monocrop SD2 MG1.1 (CP)	4,072	207	388	86	16	15	42	30.0
Monocrop SD2 MG1.7	4,101	209	377	90	16	14	43	31.0
Relayed SD2 MG0.2	2,711	202	398	51	11	13	27	13.0
Relayed SD2 MG1.1	3,443	208	392	66	16	18	40	20.9
Relayed SD2 MG1.7	3,660	208	377	73	16	16	39	20.2

used MG1.1 cultivar for the study region. In that study, it was postulated that the advantage of the longer maturity soybean was because of its extended vegetative growth prior to and after removal (i.e., harvest) of the camelina. In the present study, however, there was no clear advantage to relaying the MG1.1 or 1.7 cultivar in pennycress as their yields did not significantly differ when averaged over sowing dates. The difference in soybean cultivar response between the two different winter oilseed relay systems might be related to plant architecture. Ott et al. (2019) demonstrated that when relayed with soybean, pennycress allowed less light penetration to soybean than winter camelina. Therefore, this might have influenced the result of no difference in productivity between the MG1.1 and 1.7 cultivars in the present study with pennycress, whereas Mohammed et al. (2022) reported a difference when relayed into winter camelina. Another potential explanation is that pennycress was harvested about a week earlier than the study of Mohammed et al. (2022). Therefore, the overlap of pennycress and soybean was less than with winter camelina, and the period of soybean vegetative growth during that time may have been less of a factor than with the camelina relay system.

Compared with monocrop soybean, relayed soybean did not differ in seed oil and protein contents, which has important implications given that soybean is the most important vegetable protein source in the world. Soybean yield drag in a relay system is common and mainly due to interplant competition during the growth overlap of the crops (Wallace et al., 1992), which becomes a greater factor the longer they overlap (McBroom et al., 1981). However, relay-cropping, especially in the northern and central regions of the USA, has the advantage over double-cropping in that soybean is seeded earlier at a more normal time, thus allowing it a longer growing season (Nelson et al., 2011). Gesch et al. (2014) demonstrated that in a winter oilseed relay system, earlier planting of soybean greatly reduced soybean yield loss associated with late sowing in double-crop systems.

In the present study, relayed soybean yields were generally lower than that of the conventional practice (CP), but yield loss was considerably less and not significantly different than CP when using the longer maturity soybean (MG1.7) sown in early May (SD2) when pennycress was at the bolting stage. Studies have shown that although there is a yield drag of soybean in the winter oilseed relay system, the combined seed and oil yield of the oilseed and soybean of the relay system are often greater than growing a sole crop of soybean (Gesch et al., 2014; Johnson et al., 2017; Ott et al., 2019; Mohammed et al., 2022). More research is needed to select soybean cultivars with better tolerance to shading that may perform better in the relay system. Nevertheless, there are several ecosystem service benefits to consider when using a continuous cover cropping system such as relay-cropping pennycress and soybean. These services include reducing soil erosion and sequestering soil N (Weyers et al., 2019, 2021), suppressing herbicide-resistant

weeds (Hoerning et al., 2020), provisioning pollinators (Eberle et al., 2015), reducing global warming potential (Berti et al., 2017; Cecchin et al., 2021), and greatly increasing agricultural land use efficiency (Mohammed et al., 2022) as compared with conventional corn and soybean systems in the upper Midwest USA. The economics of the pennycress-soybean relay system remain to be addressed and will highly depend on the development of robust markets for pennycress seed oil and meal by-product.

Conclusion

The newly developed system for relay-cropping soybean with pennycress used as a cash cover crop offers a way to keep living cover on the landscape for nearly the entire year. The incorporation of pennycress into agricultural systems will offer new economic opportunities and environmental benefits.

This study demonstrates the importance of managing soybean cultivar selection and sowing date for a given region to optimize the productivity of this unique oilseed relay-cropping system. Our first hypothesis that relay interseeding soybean as late as possible into pennycress to reduce their lifecycle overlap was partially correct in that the best time to relay soybean was around the time pennycress was at its bolting stage. Although relaying soybean reduced pennycress yields 1 out of 2 years during the study, the reduction was least at the bolting stage (SD2). Our second hypothesis that the longer maturity soybean (MG1.7) would be most productive in the relay system was not fully correct. Both the common maturity soybean (MG1.1) and the longer maturing cultivar (MG1.7) across SDs gave similar results in the relay system but were greater yielding than the early soybean (MG0.2). Nevertheless, the seed yield of the MG1.7 soybean relay interseeded at pennycress bolting was not significantly different from the conventional soybean practice (CP). Importantly, seed oil and protein contents did not differ between relayed soybean and their monocrop counterparts. Furthermore, as compared with CP soybean, the relayed MG1.1 and MG1.7 cultivars did not differ in the number of yield components per plant, although relayed soybean tended to be shorter with less biomass. Additional research is needed to further identify soybean genotypes best suited for the system that reduces yield drag. Also, further research is needed to improve other management factors such as row spacing and plant populations (i.e., sowing geometry) of pennycress and soybean in the relay system.

Data availability statement

The original contributions presented in the study are included in the article/supplementary material, further inquiries can be directed to the corresponding author.

Author contributions

RG: conceptualization, methodology, investigation, formal analysis, writing original draft, supervision, and project administration. YM: formal analysis and writing-review and editing. HM: methodology, investigation, and writing-review and editing. All authors contributed to the article and approved the submitted version.

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Conflict of interest

HM was employed by WinField United, Land O'Lakes Inc.

The remaining authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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References

- Aguilar, J., Gramig, G. G., Hendrickson, J. R., Archer, D. W., Forcella, F., and Liebig, M. A. (2015). Crop species diversity changes in the United States: 1978–2012. *PLoS ONE* 10, e0136580. doi: 10.1371/journal.pone.0136580
- Akhter, M., and Sneller, C. H. (1996). Yield and yield components of early maturing soybean genotypes in the Mid-South. *Crop Sci.* 36, 877–882. doi: 10.2135/cropsci1996.0011183X0036000400010x
- Berti, M. B., Johnson, B. L., Ripplinger, D., Gesch, R. W., and Aponte, A. (2017). Environmental impact assessment of double- and relay-cropping with winter camelina in the northern Great Plains, USA. *Agric. Syst.* 156, 1–12. doi: 10.1016/j.agsy.2017.05.012
- Bishop, L., and Nelson, K. A. (2019). Field pennycress seeding date and corn herbicide management effects on corn, pennycress, and soybean production. *Agron. J.* 111, 257–263. doi: 10.2134/agronj2018.03.0156
- Carpenter, A. C., and Board, J. E. (1997). Branch yield components controlling soybean yield stability across plant populations. *Crop Sci.* 37, 885–891. doi: 10.2135/cropsci1997.0011183X003700030031x
- Cecchin, A., Pourhashem, G., Gesch, R. W., Lenssen, A. W., Mohammed, Y. A., Patel, S., et al. (2021). Environmental trade-offs of relay-cropping winter cover crops with soybean in the maize-soybean rotation. *Agric. Syst.* 189, 103062. doi: 10.1016/j.agsy.2021.103062
- Cermak, S. C., Durham, A. L., Isbell, T. A., Evangelista, R. L., and Murray, R. E. (2015). Synthesis and physical properties of pennycress estolides and esters. *Ind. Crops Prod.* 67, 179–184. doi: 10.1016/j.indcrop.2015.01.050
- Chopra, R., Johnson, E. B., Emenecker, R., Cahoon, E. B., et al. (2020). Identification and stacking of crucial traits required for the domestication of pennycress. *Nat. Food* 1, 84–91. doi: 10.1038/s43016-019-0007-z
- Cubins, J. A., Wells, M. S., Frels, K., Ott, M. A., Forcella, F., Johnson, G. A., et al. (2019). Management of pennycress as a winter annual cash cover crop: A review. *Agron. Sustain. Dev.* 39, 1–11. doi: 10.1007/s13593-019-0592-0
- Cubins, J. A., Wells, S. S., Walia, M. K., Wyse, D. L., Becker, R., Forcella, F., et al. (2022). Harvest attributes and seed quality predict physiological maturity of pennycress. *Ind. Crop. Prod.* 176, 114355 doi: 10.1016/j.indcrop.2021.114355
- Dose, H. L., Eberle, C. A., Forcella, F., and Gesch, R. W. (2017). Early planting dates maximize winter annual field pennycress (*Thlaspi arvense* L.) yield and oil content. *Ind. Crop. Prod.* 97, 477–483. doi: 10.1016/j.indcrop.2016.12.039
- Duncan, S. R., and Schapaugh, W. T. (1997). Relay-intercropping soybean in different water regimes, planting patterns, and winter wheat cultivars. *J. Prod. Agric.* 10, 123–129. doi: 10.2134/jpa1997.0123
- Duncan, S. R., Schapaugh, W. T., and Shroyer, J. P. (1990). Relay intercropping soybeans into wheat in Kansas. *J. Prod. Agric.* 3, 576–581. doi: 10.2134/jpa1990.0576
- Eberle, C. A., Thom, M. D., Nemec, K. T., Forcella, F., Lundgren, J. G., Gesch, R. W., et al. (2015). Using pennycress, camelina, and canola cash cover crops to provision pollinators. *Ind. Crops Prod.* 75, 20–25. doi: 10.1016/j.indcrop.2015.06.026
- Edwards, J. T., and Purcell, L. C. (2005). Soybean yield and biomass responses to increasing plant population among diverse maturity groups: I. Agronomic characteristics. *Crop Sci.* 45, 1770–1777. doi: 10.2135/cropsci2004.0564
- Fan, J., Shonnard, D. R., Kalnes, T. N., Johnsen, P. B., and Rao, S. (2013). A life cycle assessment of pennycress (*Thlaspi arvense* L.) -derived jet fuel and diesel. *Biomass Bioenerg.* 55, 87–100. doi: 10.1016/j.biombioe.2012.12.040
- Gehringer, A., Friedt, W., Lühs, W., and Snowdon, R. J. (2006). Genetic mapping of agronomic traits in false flax (*Camelina sativa* subsp. *sativa*). *Genome* 49, 1555–1563. doi: 10.1139/g06-117
- Gesch, R. W., Archer, D. W., and Berti, M. T. (2014). Dual cropping winter camelina with soybean in the Northern Corn Belt. *Agron. J.* 106, 1735–1745. doi: 10.2134/agronj14.0215
- Gesch, R. W., and Johnson, J. M.-F. (2015). Water use in camelina-soybean dual cropping systems. *Agron. J.* 107, 1098–1104. doi: 10.2134/agronj14.0626
- Hazebroek, J. P., and Metzger, J. D. (1990). Environmental control of seed germination in *Thlaspi arvense* (Cruciferae). *Amer. J. Bot.* 77, 945–953. doi: 10.1002/j.1537-2197.1990.tb15189.x
- Heaton, E. A., Schulte, L. A., Berti, M., Langeveld, H., Zegada-Lizarazu, W., Parrish, D., et al. (2013). Managing a second-generation crop portfolio through

sustainable intensification: Examples from the USA and the EU. *Biofr.* 7, 702–714. doi: 10.1002/bbb.1429

Hoerning, C., Wells, M. S., Gesch, R., Forcella, F., and Wyse, D. (2020). Yield tradeoffs and weed suppression in a winter annual oilseed relay-cropping system. *Agron. J.* 112, 2485–2495. doi: 10.1002/agj2.20160

Johnson, G. A., Kantar, M. B., Betts, K. J., and Wyse, D. L. (2015). Field pennycress production and weed control in a double crop system with soybean in Minnesota. *Agron. J.* 107, 532–540. doi: 10.2134/agronj14.0292

Johnson, G. A., Wells, M. S., Anderson, K., Gesch, R. W., Forcella, F., and Wyse, D. L. (2017). Yield tradeoffs and nitrogen between pennycress, camelina, and soybean in relay- and double-crop systems. *Agron. J.* 109, 2128–2135. doi: 10.2134/agronj2017.02.0065

Kladivko, E. J., Kaspar, T. C., Jaynes, D. B., Malone, R. W., and Singer, J., Morin, X.K., et al. (2014). Cover crops in the Upper Midwestern United States: Potential adoption and reduction of nitrate leaching in the Mississippi River Basin. *J. Soil Water Conserv.* 69, 279–291. doi: 10.2489/jswc.69.4.279

Mariotti, F., Tome, D., and Mirand, P. P. (2008). Converting nitrogen into protein – beyond 6.25 and Jones' factors. *Crit. Rev. Food Sci.* 48, 177–184. doi: 10.1080/10408390701279749

McBroom, R., Hadley, H., Brown, C., and Johnson, R. (1981). Evaluation of soybean cultivars in monoculture and relay intercropping systems. *Crop Sci.* 21, 673–676. doi: 10.2135/cropsci1981.0011183X002100050010x

Mohammed, Y. A., Gesch, R. W., Matthees, H. L., and Wells, S. S. (2022). Maturity selection but not sowing date enhances soybean productivity and land use in a winter camelina-soybean relay system. *Food Energy Secur.* 11, e346. doi: 10.1002/fes3.346

Mohammed, Y. A., Matthees, H. L., Gesch, R. W., Patel, S., Forcella, F., Aasand, K., et al. (2020). Establishing winter annual cover crops by interseeding into maize and soybean. *Agron. J.* 112, 719–732. doi: 10.1002/agj2.20062

Moore, S. A., Wells, M. S., Gesch, R. W., Becker, R. L., Rosen, C. J., and Wilson, M. L. (2020). Pennycress as a cash cover-crop: Improving the sustainability of sweet corn production systems. *Agron.* 10, 614. doi: 10.3390/agronomy10050614

Mortensen, D. A., Egan, J. F., Maxwell, B. D., Ryan, M. R., and Smith, R. G. (2012). Navigating a critical juncture for sustainable weed management. *BioSci.* 62, 75–84. doi: 10.1525/bio.2012.62.1.12

Moser, B. R., Knothe, G., Vaughn, S. F., and Isbell, T. A. (2009). Production and evaluation of biodiesel from Field Pennycress (*Thlaspi arvense* L.) oil. *Energy Fuels* 23, 4149–4155. doi: 10.1021/ef900337g

Myers, R., and Watts, C. (2015). Progress and perspectives with cover crops: Interpreting three years of farmer surveys on cover crops. *J. Soil Water Conserv.* 70, 125A–129A. doi: 10.2489/jswc.70.6.125A

Nelson, K. A., Massey, R. E., and Burdick, B. A. (2011). Harvest aid application timing affects wheat and relay intercropped soybean yield. *Agron. J.* 103, 851–855. doi: 10.2134/agronj2010.0384

Nolan, R. L., Wells, M. S., Sheaffer, C. C., Baker, J. M., Martinson, K. L., and Coulter, J. A. (2018). Establishment and function of cover crops

interseeded into corn. *Crop. Sci.* 58, 863–873. doi: 10.2135/cropsci2017.06.0375

Ott, M. A., Eberle, C. A., Thom, M. D., Archer, D. W., Forcella, F., Gesch, R. W., et al. (2019). Economics and agronomics of relay-cropping pennycress and camelina with soybean in Minnesota. *Agron. J.* 111, 1281–1292. doi: 10.2134/agronj2018.04.0277

Patel, S., Lenssen, A. W., Moore, K. J., Mohammed, Y. A., Gesch, R. W., Wells, M. S., et al. (2021). Interseeded pennycress and camelina yield and influence on row crops. *Agron. J.* 113, 2629–2647. doi: 10.1002/agj2.20655

Phippen, W. B., and Phippen, M. E. (2012). Soybean seed yield and quality as a response to field pennycress residue. *Crop Sci.* 52, 2767–2773. doi: 10.2135/cropsci2012.03.0192

Reicosky, D. C. (2015). Conservation tillage is not conservation agriculture. *J. Soil Water Conserv.* 70, 103A–108A. doi: 10.2489/jswc.70.5.103A

Ringquist, J., Phillips, T., Renner, B., Slides, R., Stuart, K., Baum, M., et al. (2016). *Capitalizing on the Shifting Consumer Food Value Equation*. London, UK: Deloitte Development LLC. 32.

Royo-Ensal, A., Necajeva, J., Torra, J., Recasens, J., and Gesch, R. W. (2015). Emergence of field pennycress (*Thlaspi arvense* L.): Comparison of two accessions and modelling. *Ind. Crops Prod.* 66, 161–169. doi: 10.1016/j.indcrop.2014.12.010

Ryan, M. R., Crew, T. E., Culman, S. W., Dehaan, L. R., Hayes, R. C., Jungers, J. M., et al. (2018). Managing for multifunctionality in perennial grain crops. *BioSci.* 68, 294–304. doi: 10.1093/biosci/biy014

SAS Institute (2014). *SAS/STAT® 13.2 User's Guide*. Cary, NC: SAS Inst.

Sindelar, A. J., Schmer, M. R., Gesch, R. W., Forcella, F., Eberle, C. A., Thom, M. D., et al. (2017). Winter oilseed production for biofuel in the US Corn Belt: opportunities and limitations. *GCB Bioenergy* 9, 508–524. doi: 10.1111/gcbb.12297

Strock, J. S., Porter, P. M., and Russelle, M. P. (2004). Cover cropping to reduce nitrate loss through subsurface drainage in the northern U.S. *Corn Belt. J. Environ. Qual.* 33, 1010–1016. doi: 10.2134/jeq2004.1010

Thom, M., Eberle, C. A., Forcella, F., Gesch, R., and Weyers, S. (2018). Specialty oilseed crops provide an abundant source of pollen for pollinators and beneficial insects. *J. Appl. Entomol.* 142, 211–222. doi: 10.1111/jen.12401

Thom, M. D., Eberle, C., Forcella, F., Gesch, R. W., Nemecek, K. T., Lundgren, J. G., et al. (2016). Nectar production in oilseeds: food for pollinators in an agricultural landscape. *Crop Sci.* 56, 727–739. doi: 10.2135/cropsci2015.05.0322

Wallace, S., Whitwell, T., Palmer, J., Hood, C., and Hull, S. (1992). Growth of relay intercropped soybean. *Agron. J.* 84, 968–973. doi: 10.2134/agronj1992.00021962008400060012x

Weyers, S., Thom, M., Forcella, F., Eberle, C., Matthees, H., Gesch, R., et al. (2019). Reduced potential for nitrogen loss in cover crop-soybean relay systems in a cold climate. *J. Environ. Qual.* 48, 660–669. doi: 10.2134/jeq2018.09.0350

Weyers, S. L., Gesch, R. W., Forcella, F., Eberle, C. A., Thom, M. D., Matthees, H. L., et al. (2021). Surface runoff and nutrient dynamics in cover crop-soybean systems in the Upper Midwest. *J. Environ. Qual.* 50, 158–171. doi: 10.1002/jeq2.20135



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EDITED BY
Carol Williams,
University of Wisconsin–Madison, United States

REVIEWED BY
Gina Nichols,
University of California, Davis, United States
Mitchell Hunter,
University of Minnesota Twin Cities,
United States

*CORRESPONDENCE
Jennifer B. Thompson
✉ jennifer.thompson@zalf.de

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Remote sensing of hedgerows, windbreaks, and winter cover crops in California's Central Coast reveals low adoption but hotspots of use

Jennifer B. Thompson^{1,2*}, Jennifer Symonds¹, Liz Carlisle³,
Alastair Iles¹, Daniel S. Karp⁴, Joanna Ory¹ and Timothy M. Bowles¹

¹Department of Environmental Science, Policy, and Management, University of California, Berkeley, Berkeley, CA, United States, ²Leibniz Centre for Agricultural Landscape Research, Müncheberg, Germany,

³Environmental Studies Program, University of California, Santa Barbara, Santa Barbara, CA, United States,

⁴Department of Wildlife, Fish, and Conservation Biology, University of California, Davis, Davis, CA, United States

Non-crop vegetation, such as hedgerows and cover crops, are important on-farm diversification practices that support biodiversity and ecosystem services; however, information about their rates and patterns of adoption are scarce. We used satellite and aerial imagery coupled with machine learning classification to map the use of hedgerows/windbreaks and winter cover crops in California's Central Coast, a globally important agricultural area of intensive fresh produce production. We expected that adoption of both practices would be relatively low and unevenly distributed across the landscape, with higher levels of adoption found in marginal farmland and in less intensively cultivated areas where the pressure to remove non-crop vegetation may be lower. Our remote sensing classification revealed that only ~6% of farmland had winter cover crops in 2021 and 0.26% of farmland had hedgerows or windbreaks in 2018. Thirty-seven percent of ranch parcels had cover crops on at least 5% of the ranch while 22% of ranches had at least one hedgerow/windbreak. Nearly 16% of farmland had other annual winter crops, some of which could provide services similar to cover crops; however, 60% of farmland had bare soil over the winter study period, with the remainder of farmland classified as perennial crops or strawberries. Hotspot analysis showed significant areas of adoption of both practices in the hillier regions of all counties. Finally, qualitative interviews revealed that adoption patterns were likely driven by interrelated effects of topography, land values, and farming models, with organic, diversified farms implementing these practices in less ideal, lower-value farmland. This study demonstrates how remote sensing coupled with qualitative research can be used to map and interpret patterns of important diversification practices, with implications for tracking policy interventions and targeting resources to assist farmers motivated to expand adoption.

KEYWORDS

diversified farming systems, non-crop vegetation, remote sensing, cover crops, hedgerows, random forest, windbreaks

Introduction

Non-crop vegetation plays important roles on farms. Non-crop vegetation includes any non-harvested plants on the farm including, but not limited to, isolated trees, hedgerows, windbreaks, cover crops, floral strips, and riparian buffers. Planting non-crop vegetation is an example of a diversification practice, or a practice that brings biodiversity to an agroecosystem and helps support ecosystem services. Planned, non-crop vegetation like winter cover crops and hedgerows supports associated biodiversity including soil microbes, pollinators, birds, and other taxa (Verboom and Huitema, 1997; Pereira and Rodríguez, 2010; Morandin et al., 2011; Lecq et al., 2017). In turn, both planted non-crop species and the biodiversity that they support can provide critical ecosystem services (although disservices can also result; Zhang et al., 2007). Such ecosystem services benefit both farms and the surrounding environment (Kremen and Miles, 2012; Tamburini et al., 2020); for instance, by supporting bees, other pollinators, pest predators, and parasitoids, hedgerows bolster crop pollination and pest control services and, by doing so, may allow for high yields with fewer agrochemical inputs (Cranmer et al., 2012; Morandin and Kremen, 2013; Long et al., 2017; Castle et al., 2019; Ponisio et al., 2019; Albrecht et al., 2020). Similarly, cover crops increase nutrient cycling and retention services and maintain healthier soils, which may support yields with fewer inputs while reducing harmful nutrient losses (Brennan and Smith, 2005; Heinrich et al., 2014; Büchi et al., 2018; Lugato et al., 2020). As such, non-crop vegetation can help enhance farm viability by securing livelihoods for farmers while reducing the negative environmental externalities of agriculture (Kremen and Miles, 2012; Kremen et al., 2012).

Given the potential benefits of such diversification practices, it is important to understand rates and patterns of farmer adoption in order to target investments of research, technical assistance, and policy interventions as well as track their impacts over time. Yet such information is rarely available in the United States, including in California, where multiple recent policies make baseline knowledge of the extent of their usage particularly important. For example, California's Healthy Soils Program provides incentives to producers for adopting practices that sequester carbon or reduce greenhouse gas emissions (CDFA California Department of Food Agriculture, 2022), including planting non-crop vegetation. Similarly, California's Climate Scoping Plan (CARB California Air Resources Board, 2022) highlights non-crop vegetation as a strategy for meeting goals related to climate change mitigation in working landscapes. Regional implementation of water quality regulations recognize cover crops for their ability to scavenge nitrogen and reduce nitrate leaching to groundwater (California Regional Water Quality Control Board: Central Coast Region, 2021). Finally, recent state bans on the pesticide chlorpyrifos make natural pest control services—like those that may be provided by hedgerows—all the more critical (Alternatives to Chlorpyrifos Work Group, 2020).

Even with these emerging policies advocating for non-crop vegetation, available estimates from government surveys or expert opinion suggest it is rarely planted on California farms. Cover crops have been grown on only ~5% of farmland in recent years (Brennan, 2017; USDA United States Department of Agriculture, 2019), while statewide estimates for other non-crop vegetation practices, like hedgerows and windbreaks, do not even exist. Qualitative studies have documented significant barriers to adoption for hedgerows and

cover crops, especially in California's Central Coast (Esquivel et al., 2021; Carlisle et al., 2022). This region produces mainly vegetables and fruits that are often consumed raw (e.g., lettuce and strawberries), and food safety concerns are paramount. Following a 2006 outbreak of pathogenic *E. coli* on bagged spinach in which over 200 people became ill and 3 died (Jay et al., 2007), leafy greens buyers required growers to implement comprehensive new food-safety protocols on farmers, intending to minimize the risk of crop contamination with foodborne pathogens from wildlife vectors (Karp et al., 2015). At least 32% of leafy green growers in California's Central Coast reported removing non-crop vegetation on their farms in a survey following the 2006 *E. coli* outbreak (Beretti and Stuart, 2008). Nearly a decade after the incident, ~45% of California produce growers still reported clearing vegetation to create or expand bare-ground buffers around their fields (Baur et al., 2016), despite evidence suggesting the practice is ineffective at mitigating food-safety risks (Karp et al., 2015; Sellers et al., 2018; Glaize et al., 2021; Weller et al., 2022). Thus, growers often perceive that hedgerows and windbreaks pose food safety risks in attracting and harboring wildlife, and additional supply chain requirements from processors or retailers may actually prohibit hedgerows in close proximity to crops like leafy greens (Carlisle et al., 2022). Other barriers to hedgerow adoption include high costs of initial installation and maintenance, and the relatively long time to mature and provide pest control or pollination benefits, unlike herbaceous non-crop vegetation like insectary strips (Long et al., 2017), not to mention the cost of taking land out of production for non-crop vegetation. Long-term gains are unlikely to motivate adoption for the many growers in the Central Coast with shorter-term land leases (Calo and De Master, 2016; Chapman et al., 2022).

Barriers to cover crop adoption on the Central Coast similarly involve a combination of economic constraints, perceptions of risk, technical challenges, and problems with policy programs and/or incentives (Stuart, 2009). The main obstacle to growing cover crops in the Central Coast is the high cost of land; Monterey and Santa Cruz counties have the 4th and 5th highest agricultural land rents of counties in California (NASS U.S. National Agricultural Statistics Service, 2020). Growing cover crops that could interfere with cash crop production is a major perceived opportunity cost (Carlisle et al., 2022; Chapman et al., 2022). Farms often grow multiple cash crops per year, which requires careful planning to stay on schedule, especially with highly variable weather (Brennan, 2017). In the warm-summer Mediterranean climate (Beck et al., 2020), which has highly variable interannual precipitation, low rainfall reduces cover crop germination and/or growth and discourages growers from planting a cover crop in the first place, especially if additional irrigation could be needed. Unpredictable rain patterns can delay the clearing and incorporation of cover crop residue when soils are heavily saturated late in the winter season (Hartz and Johnstone, 2006). In turn, this delays cash crop planting as growers wait for soils to dry before doing the significant soil and bed preparation operations often used in vegetable and berry production. Residue management is another obstacle to implementing cover crops, since large pieces of plant residue can impede cultivation and planting of small-seeded vegetables (Brennan, 2017).

While social science research has identified barriers to using hedgerows, windbreaks, and cover crops, fine scale information on the extent and patterns of adoption at regional or local landscape scales is unavailable. Such information could identify hotspots and

coldspots of hedgerow and cover crop adoption, which would complement social science research and also identify where to concentrate resources to support greater adoption.

Remote sensing offers an opportunity for detecting and quantifying farming practices across wide areas, over time, and at fine scales by using readily available satellite or aerial imagery. Remote sensing of cover crop use to understand adoption patterns and benefits has been successful in the U.S. Midwest (Hively et al., 2009, 2015; Seifert et al., 2018; Kushal et al., 2021). Several studies have also classified non-crop vegetation such as hedgerows and trees via remote sensing (Ghimire et al., 2014; O'Connell et al., 2015). However, previous studies have often focused on regions, such as much of the Midwest, where just one or two crops dominate vast areas of large fields with little natural habitat remaining. Areas like the Central Coast of California pose additional challenges for remote sensing of non-crop vegetation. As one of the most intensively-cropped and productive agricultural regions in the U.S, the Central Coast produces dozens of crops, including strawberries, leafy greens, grapes, and other specialty crops (CDFA California Department of Food Agriculture, 2022). Such crops can be grown nearly year-round in the region's climate, often on small, irregular fields surrounded by varying levels of natural habitat. Remote sensing of non-crop vegetation in agricultural regions like this requires dealing with the many challenges of differentiating between crops, practices, and features on such a biologically diverse and spatially varied landscape.

In this study, we used remote sensing and machine learning to classify and quantify the extent of hedgerows, windbreaks, and cover crops across the Central Coast of California. We chose these diversification practices as they are important for supporting biodiversity and ecosystem services on farms, both within (cover crops) and around (hedgerows/windbreaks) areas of crop production. They are also readily visible from aerial and satellite imagery compared to other diversification practices like compost use or very narrow floral insectary strips. We focused specifically on winter cover crops as cover crops are most commonly used over the winter in this region (Brennan, 2017), when they play a particularly important role reducing nitrate leaching (Jackson et al., 1993). We also coupled remote sensing observations with qualitative interviews with 20 growers and 8 technical advisors to gain insight into patterns of adoption. Our objectives were to provide baseline understanding of practice adoption across a three-county region and to assess spatial patterns of adoption. We expected that adoption of both practices would be relatively low and unevenly distributed across the landscape with higher levels of adoption found in marginal farmland and in less intensively cultivated areas where the pressure to remove non-crop vegetation may be lower. More broadly, our analysis represents a first step toward tracking adoption of key diversification practices in one of the most intensive agricultural regions in the world.

Materials and methods

Study area and data collection

Our study focused on farmland in San Benito, Santa Cruz, and Monterey counties, encompassing 7,787 km² of California's Central Coast (USDA United States Department of Agriculture, 2019). Farmland within each county was determined with ranch boundary shapefiles provided by each county's agricultural commissioner's

office. Here, ranch refers to an agricultural operation and can include rangeland with livestock as well as orchards, annual cropping operations, and mixed operations. The ranch boundaries represent the entire property boundaries, not individual field boundaries. Ranches composed entirely of rangeland, determined by land use classifications provided by the county agricultural commissioners' offices as well as visual inspection, were excluded from the study areas, as many were covered in naturally growing shrubs that could be mistaken for hedgerows in the analysis.

To collect training data for classification algorithms, we conducted "windshield surveys" for hedgerows in July 2019 and for cover crops in January 2021. For each survey, we drove systematically through farmland and recorded GPS points corresponding to several types of crop and non-crop vegetation. Hedgerows were defined as linear strips of shrubs or small trees at least 5 m in length and longer than it was wide. Windbreaks were defined similarly but instead consisted of taller trees. As there was ambiguity between which strips would be classified as hedgerows and others windbreaks, we combined both into a single "hedgerow/windbreak" category. Based on the windshield survey and additional expert image analysis, we identified 98 hedgerows/windbreaks. For cover crops, we identified bare fields (i.e., no plant cover, 171 points), cover crops (76 points), and various winter cash crops (178 points, for full list of crops see [Supplementary material](#)). Cover crops were predominantly grasses or grain/vetch/radish/legume mixes with some single-species vetch, mustard, legume, and radish cover crops also recorded. While some crops (e.g., radishes or brassicas) can be both cash and cover crops, we distinguished between them to the best of our ability by noting bed and row formation as cash crops are planted in rows while cover crops are broadcast seeded.

Hedgerow/windbreak classification

Object based image analysis (OBIA) (Blaschke, 2010) was used to classify hedgerows/windbreaks. In OBIA, similar pixels are grouped together as objects and are grown until the algorithm determines it has reached a dissimilar pixel. The spectral, geometric, or textural qualities of the object can be used for land-use classification. Previous studies have had success in employing OBIA to identify small farm elements, like hedgerows, as it eliminates error found in pixel-based classification which could prevent small objects from being properly classified (Sheeren et al., 2009; Tansey et al., 2009; Ghimire et al., 2014; O'Connell et al., 2015).

We utilized pre-processed digital 4-band National Agriculture Imagery Program (NAIP) imagery (NAIP, Aerial Photography Field Office (AFPO), 2018) from 2018 with a spatial resolution of 60 cm². We used a multiresolution segmentation algorithm in the eCognition software (Trimble Geospatial Imaging, Munich, Germany) to create image objects. For more detailed methods see [Supplementary material](#).

Once images were segmented, we used rule based classification and 1,010 model training image objects from the windshield survey and expert image analysis to classify the objects as one of the following seven land-use: agriculture (row crops), hedgerows/windbreaks, vineyards, non-linear (non-hedgerow) shrubs or trees, bare soil and/or urban, orchards, and water. We used 98 hedgerow/windbreak image objects, 116 orchard objects, 189 shrub/tree objects, 294

vineyard objects, and 311 row crop objects. Objects with a normalized difference vegetation index (NDVI) of <0.1 were classified as bare/urban and excluded from the subsequent classification. While 0.2 is a common threshold for vegetative vs. bare or urban land (Sobrino et al., 2001), we found that the OBIA sometimes included pixels of the hedgerow/windbreak shadow on bare ground, which subsequently lowered the NDVI. Thus, a more conservative threshold was chosen to avoid excluding any possible hedgerows/windbreaks. Objects with a normalized difference water index (NDWI) >0.3 were classified as water. For each remaining unclassified training object, we exported image object information for 35 variables representing object spectral (e.g., NDVI), geometric (e.g., length to width ratio), and textural (e.g., spatial patterns between pixels in a single object) information that could be used to classify each vegetative class (Supplementary Table S1). Highly correlated variables ($R^2 > 0.75$) were excluded from the classifier. The remaining 23 image object variables of each training object were used to train a random forest classifier model using R statistical software v4.1.2 (R Core Team, 2022) using the package *randomforest* (Liaw and Wiener, 2002). The out-of-bag (OOB) error of the random forest model, a measure of prediction error for machine learning models, was 5.82%. Once the ideal classifier parameters were established in R, we ran a random forest model in eCognition using these parameters to classify all the segmented images. We also distinguished between hedgerows/windbreaks and riparian vegetation in our post-classification analysis. Riparian vegetation can consist of shrubs, trees, and other plants with similar spectral properties and shapes as hedgerows. Thus, to avoid over-classification of riparian vegetation as hedgerows/windbreaks, we reclassified all shrub and hedgerow/windbreak objects bordering water as riparian vegetation and it was excluded from subsequent analysis.

After classification, 500 accuracy assessment points in the classified images were randomly created in ArcGIS by equal stratified random sampling. Each point was assigned one of the seven land covers land cover classes based on expert image interpretation or data from the windshield survey, if available, and compared to the model's predicted classification. The overall accuracy was 90.0% and the kappa, another measure of model accuracy, was 0.89. As for our class of interest, hedgerows/windbreaks, there was a producer's accuracy of 97% and a user's accuracy of 63% indicating that the model almost always classified a hedgerow/windbreak if it was there but also tended to over classify other linear elements, such as drainage ditches filled with vegetation, as hedgerows. Thus, to have the most accurate classification possible, all objects classified as hedgerows/windbreaks were manually inspected and reclassified if needed.

Cover crop classification

We used Sentinel-2 satellite imagery for cover crop classifications in Google Earth Engine (Gorelick et al., 2017). We chose Sentinel-2 imagery for its high spatial (10 m) and temporal resolution (5 day), ideal for capturing small fields and multiple dates of imagery. We used temporal aggregation to create composite images by combining multiple days of imagery useful in differentiating crops. We pre-processed satellite imagery to remove cloud cover and create clean images for analysis and added an NDVI band to allow for

classifications. For the land use classification, we utilized a two-step classification: threshold and random forest.

To first differentiate perennial vs. annual vegetation, we utilized a rule-based threshold classification. Any pixels that fell below 0.2 NDVI at some point between June 15th, 2020 and January 15th, 2021 were classified as non-perennial, while those that did not fall below 0.2 were classified as perennial as an NDVI below 0.2 indicated a bare field and thus the harvest of an annual crop. We used 0.2 NDVI as it is a common cutoff for differentiating between bare soil and green vegetation (Sobrino et al., 2001). If the NDVI of the pixel never dropped below 0.2, then the soil was likely never bare and thus likely contained a perennial crop. We clipped threshold classification to ranch boundaries. The threshold classification was found to have an overall accuracy of 85% based on an accuracy assessment. The non-perennial class had a user's accuracy of 79% and producer's accuracy of 100%, indicating that the model almost always correctly classified non-perennial land, but included some perennial land in the non-perennial class. This means that while it may include some perennial land, the non-perennial boundaries used for subsequent classification likely did not incorrectly exclude non-perennial land.

For the second part of the classification, we classified all the remaining non-perennial vegetated land into the following classes: cover crops, annual crops, strawberries, and bare-field classes from the median pixel values of December 15, 2020 through February 28, 2021. We selected these dates as they are a common time for winter cover crops in the region before the beds are prepared for the spring crop and they also resulted in the highest model accuracy. We used NDVI, the blue band, and green band as classification variables in a random forest classifier of 100 classification trees. We clipped this classification to the boundaries from the previous threshold classification of non-perennial land, i.e., we only consider these classification results in land determined to be non-perennial agriculture. We used 80% of ground truth data points per class for classifier training, while the remaining 20% were used for accuracy testing (Shelestov et al., 2017). The cover crop classifier was found to be 87% accurate with a kappa of 0.82. Here, we report on our class of interest, cover crops, which had both a user's accuracy of 87% and a producer's accuracy of 87%.

Statistical analysis

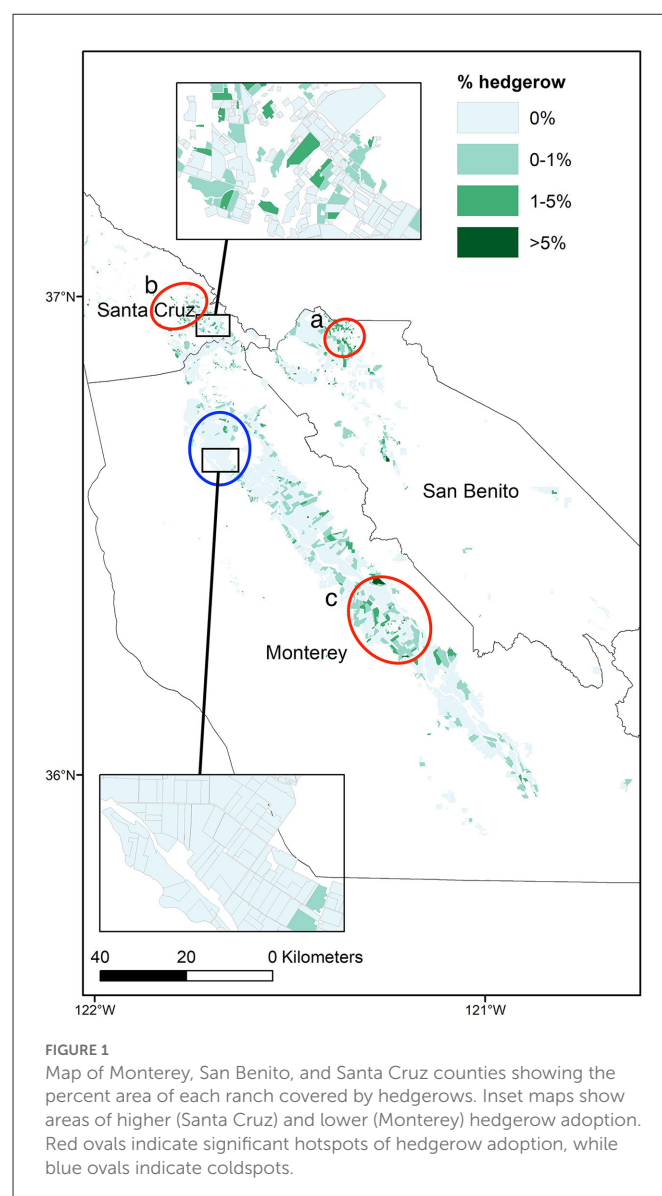
Hedgerow/windbreak and cover crop usage, calculated as the percent of a ranch's total area occupied by hedgerows/windbreaks or cover crops, was calculated for every ranch within the boundary shapefile provided. To determine the role of soil type/land quality on diversification practice usage, farmland classification maps were taken from the Natural Resource Conservation Services and the presence of the practice per farmland type (i.e., local importance, statewide importance, grazing land, other land, prime farmland, and unique farmland) was calculated (California Department of Conservation, [n.d.]). Spatial autocorrelation of hedgerows/windbreaks and cover crops was determined using Moran's i (Moran, 1950) and hot spot analysis of hedgerow/windbreak usage was analyzed using Getis-Ord* in ArcGIS (Ord and Getis, 1995). Getis-Ord* takes a feature's value, here a ranch's hedgerow/windbreak or cover crop usage, and compares it to neighboring features; significant clustering of high

values indicates a hot spot. Due to a large variation of ranch sizes, as well as distance between ranches in each county, each county was analyzed for hotspots separately to ensure that the appropriate scale was used for the distance matrix needed to calculate the Getis-Ord* statistic. When calculating spatial statistics, a key step is to determine the distance or number of neighbors to compare each feature to that is both appropriate for spatial statistics but also relevant to the spatial context of the study. The distance band, defined as how far away a ranch can be located to be considered another ranch's neighbor, for each county was calculated with the incremental spatial autocorrelation tool in ArcGIS. This tool calculates the global Moran's i at increasing distances to determine a distance where one can find peak clustering for the dataset. The distance band chosen for each county for the hedgerow/windbreak hotspots was 8.37, 9.58, and 4.43 km and for the cover crop hotspot analysis 8.73, 9.91, and 4.43 km (Monterey, Santa Cruz, and San Benito, respectively). The False Discovery Rate was applied which decreases the critical p -value thresholds needed to indicate a hot spot in order to address issues with spatial autocorrelation in the dataset which could inflate the number or significance of hot or coldspots.

Qualitative interviews

In February 2019, we conducted semi-structured, in-depth interviews with 20 farmers in the California Central Coast region who grow organic lettuce as either their primary cash crop or part of a diverse array of crops. We focused on lettuce because it is the most economically valuable vegetable crop grown in the region (CDEA California Department of Food Agriculture, 2022). Within our interview sample, farms ranged in size from 4 acres to over 10,000 acres (mean: 1,935 acres; median: 100 acres) and spanned four counties: Monterey (5 interviews), San Benito (4), Santa Cruz (5), and Santa Clara (1), with 5 additional farmers spanning multiple of these counties. Details of participant recruitment and interview procedures can be found in Esquivel et al. (2021) and Carlisle et al. (2022). Briefly, we selected a stratified sample of all organic farms in these counties that listed organic lettuce as a crop and contacted farmers that reflected ecological diversity (e.g., crop diversity) and a diversity of farm scales (i.e., sizes), geographical locations within the study region, and cultural backgrounds/first languages. Twenty farmers agreed to participate and completed an interview. Because we deliberately included farm types that are less common (highly diversified, medium-sized, direct-market), our sample represents a higher-than-average adoption of cover cropping and hedgerows. In 2020, we also interviewed five additional conventional wholesale farmers in order to include the perspectives of larger, less-diversified farmers who are more representative of the average farm type in our study area.

To complement interviews with growers, in May 2019 we conducted semi-structured, in-depth interviews with 8 technical assistance providers whose names came up repeatedly in interviews with growers. While this was not a systematically representative sample of technical assistance providers in the region, interviewing these individuals allowed us to verify and build on what we learned from grower interviews about patterns of adoption of cover crops and hedgerows. Because these technical assistance providers spoke from their knowledge of the sector as a whole, they could both generalize



across multiple operations and speak candidly about sensitive issues that might not be comfortable topics to investigate in the context of a specific operation. These interviews thus provided an opportunity for us to test hypotheses about patterns of adoption that were implied in our grower interviews.

Interview questions posed to both groups (Supplementary material) focused on diversification practices, crop and non-crop diversity, and how farm-level decisions were shaped by various market and policy factors. We began by asking open-ended questions (e.g., what practices do you currently use to maintain or improve soil health on your farm?), and followed with more specific questions (e.g., do you grow any non-crop plants on your farm, such as hedgerows, buffers, or habitat for beneficial insects?). Interviews were digitally recorded and transcribed verbatim. We analyzed interview transcripts in NVivo 12, using an iterative coding method following an open, axial, and selective coding procedure (Corbin and Strauss, 1990). To identify key factors influencing farmer adoption of cover crops and hedgerows, data

TABLE 1 Moran's i spatial autocorrelation assessment; significant values indicate spatial autocorrelation of the practice within each county.

	Moran's i	Expected i	Z score	P value
Hedgerow/windbreaks				
Monterey	0.009	−0.0005	4.952	<0.00001
San Benito	0.0283	−0.0008	9.984	<0.00001
Santa Cruz	0.0065	−0.0009	6.2814	<0.00001
Cover crops				
Monterey	0.069	−0.0005	36.89	<0.00001
San Benito	0.049	−0.0083	16.96	<0.00001
Santa Cruz	0.027	−0.0009	24.39	<0.00001

were coded into thematic categories, such as “Land Costs,” “Pressures from Buyers,” and “Peer Learning and Influence.”

Results

Hedgerows/windbreaks

Our study area included 4,371 ranches across 1,260 km². Average ranch size in each county was 0.5, 0.18, and 0.1 km² (or 124, 44, and 24 acres) for Monterey, San Benito, and Santa Cruz counties, respectively. Hedgerows/windbreaks were detected on 22% of the ranches across all 3 counties (Figure 1), with 18% of ranches in San Benito, 27% in Monterey, and 21% in Santa Cruz County having hedgerows. The average length of a hedgerow/windbreak was 116 m. The total area covered in hedgerow/windbreaks across all counties was 3.27 km² or 0.26% of the study area. Ranches with hedgerows/windbreaks showed significant spatial autocorrelation (Table 1), indicating that ranches with hedgerows/windbreaks were near other ranches with hedgerows.

The Getis-Ord G^* indicated that there are several significant hotspots of hedgerow/windbreak usage. Most notably, these include the most northeast section of San Benito County between (a) the CA-156 highway and nearby hills, (b) the hillier part of the primary Santa Cruz County agricultural region near the census-designated places of Amesti and Freedom, and (c) a hotspot near the cities of Soledad and Greenfield in Monterey County (Figure 1). There were no hotspots detected in the large swath of prime agricultural land between and directly surrounding the cities of Watsonville and Salinas and, notably, a strong coldspot near Salinas in Monterey County.

Cover crops

We found that 74.9 km² of farmland across the three counties in our study was planted with cover crops over the winter 2020–2021 season. This represents only 5.9% of total farmland area with 37% of ranches having least 5% of their fields cover cropped at this time. Santa Cruz County had the largest percentage of cover cropped farmland at 15.4%, whereas 5.1% of land was cover cropped in Monterey and 5.8% in San Benito (Table 2). In contrast, fields with

bare soil constituted the majority of Central Coast farmland (59.9% of land; Supplementary Table S4).

Just as for hedgerows/windbreaks, significant spatial autocorrelation was found between ranches that adopted cover crops, indicating clustering of cover crop adoption (Table 1). The Getis-Ord G^* analysis indicated significant hotspots of cover crop usage within each county. The hotspots in (a) Santa Cruz and (b) San Benito were located in nearly the same areas as the hedgerow/windbreak hotspots (Figure 2) while the cover crop hotspot in (c) Monterey County was found in areas near the Pajaro River and the census-designated place, Las Lomas. No other hotspots were found in Monterey County but significant cold spots existed near the cities of (d) Salinas, (e) Gonzales, and (f) Greenfield.

Adoption and farmland type

Nearly 60% of our study area was classified as “Prime Farmland” but less than half of the total area classified as hedgerows/windbreaks or cover crops was located in this prime farmland area (Table 3). Prime farmland is defined as very important in meeting U.S. food, feed, forage, and fiber needs due to ideal physical and chemical characteristics such as water availability, soil type, and climate. Conversely 16.8% of the area classified as hedgerows/windbreaks and 16.2% of cover crops were located on “Unique Farmland,” which made up just 8.5% of the study area (Table 3). Unique farmland, like prime farmland, has characteristics that make it valuable for growing crops but specifically for more specialized and regional high-value crops such as almonds, citrus, grapes, etc.

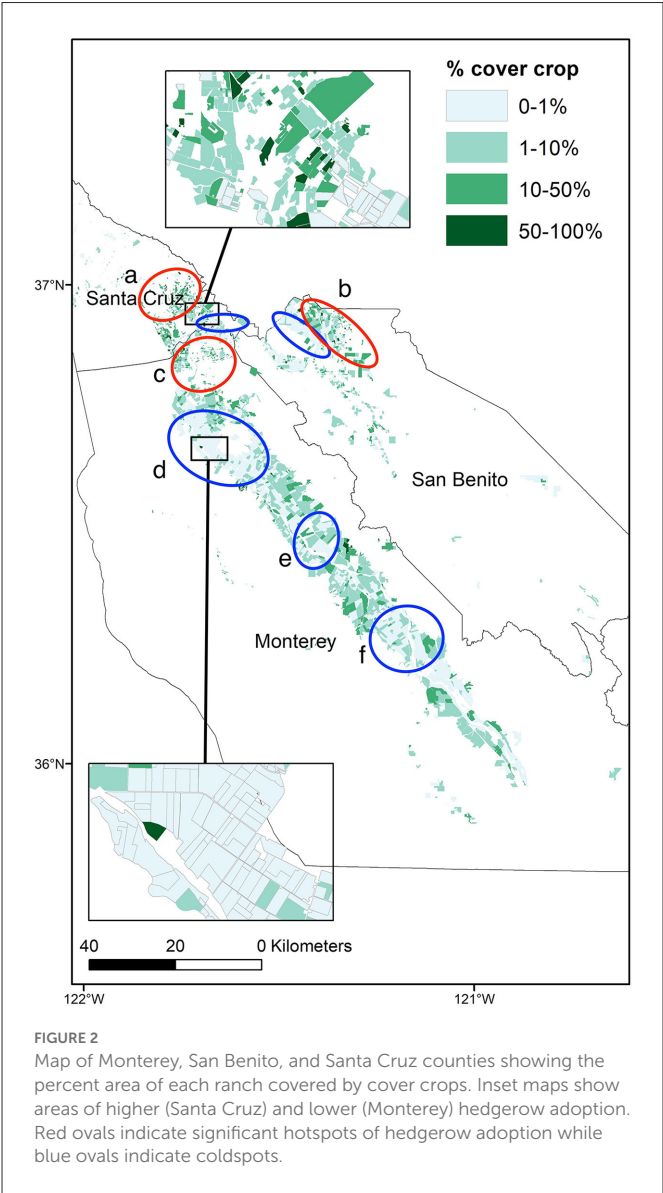
Patterns of adoption

The significant spatial autocorrelation in the locations of hedgerows/windbreaks and cover crops suggests possible biophysical and social mechanisms influencing their adoption, some of which we were able to investigate through our interviews. Hotspots of adoption for hedgerows/windbreaks and cover crops overlapped in Santa Cruz and San Benito counties, coinciding with some of the hillier and less attractive farmland. The only hedgerow/windbreak hotspot in Monterey County was found at the southern end of the valley where there are many vineyards that use windbreaks around and within their fields as observed during the windshield surveys. Different management considerations and supply chain pressures may make it easier to establish and maintain hedgerows/windbreaks in vineyards vs. intensive vegetable and berry production systems.

In our interviews with organic farmers and technical assistance providers, we deliberately explored two hypotheses for the pattern of lower cover crop and hedgerow/windbreak adoption on high-value farmland in the flat areas of Monterey County. The first hypothesis was that higher farmland rent discouraged producers from taking any land out of cash crop production, due to financial pressures to bring in enough revenue to cover these rents (Guthman, 2004). We reasoned that these pressures connected to land rent would bear more heavily on those farming high value lands in Monterey County, where irrigated farmland rent averages US\$2,050/acre, as compared with those farming in San Benito County with rents of US\$725/acre and in

TABLE 2 Land use cover (in km²) in each county from the cover crop classifier.

County	Annual crops	Bare	Strawberry	Cover crops	Perennial crops	Total
San Benito	31.02	140.01	21.63	12.74	15.89	221.29
Santa Cruz	26.40	20.29	9.26	13.90	20.36	90.21
Monterey	144.05	597.02	102.86	48.24	61.64	953.81
Total	201.47	757.32	133.75	74.88	97.89	1,265.31



hillier areas of Santa Cruz County (NASS U.S. National Agricultural Statistics Service, 2020).

This hypothesis was widely confirmed in our interviews, with 11 of 20 growers and all 8 technical assistance providers citing higher land rent costs as a significant discouragement to take any land out of production by planting cover crops or hedgerows/windbreaks (Table 4).

TABLE 3 Percent of total study area and percent of mapped hedgerow/windbreaks and cover crops found in each of the NRCS important farmland classifications.

Farmland type	Study area	Hedgerow/windbreaks	Cover crops
Prime farmland	59.4	44.8	44.7
Statewide importance	14.9	12.4	18.1
Grazing land	10.4	12.1	12.2
Unique farmland	8.5	16.8	16.2
Other land	4.6	13.0	7.5
Local importance	2.2	1.0	1.4

We also explored a related hypothesis, which speaks to the complexity of social and ecological relationships in this agricultural region. Growers on high-value farmland, we hypothesized, not only faced pressures to maximize land in production; these same financial pressures pushed these growers to scale their operations into the hundreds of acres, which in turn forced them to work with wholesale buyers. These wholesale buyers imposed stringent food safety requirements, discouraging farmers from planting any non-crop vegetation, and in some cases asking them to remove existing hedgerows. At the same time, the rigid harvesting schedules required by these buyers discouraged growers from planting cover crops, as they were unwilling to take any risk of getting delayed with spring planting. Buyers at this scale could also penalize growers for having any “foreign material” in the field at the time of harvesting, with cover crop residue counting as one such source of “foreign material.” This hypothesis was also widely confirmed in our interviews, with 7 of 20 growers and all 8 technical assistance providers citing discouragement from large scale buyers as a factor in decisions to avoid cover crops and hedgerows/windbreaks (Table 4).

Additionally, two other patterns of adoption emerged from our interviews, even though we did not design our questions to deliberately explore these hypotheses. For one, a number of farmers (n=6, Table 4) reported that they prioritize adoption of cover crops and hedgerows/windbreaks in hillier areas prone to erosion, as well as other marginal lands. In explaining why they did this, farmers typically cited their own stewardship values, although we suspect some farmers may also be motivated to avoid county penalties when sediment accumulates on roads. Additionally, some remnant native vegetation in hillier regions may be counted as hedgerows/windbreaks

TABLE 4 Key themes and quotes helping to explain patterns of adoption of cover crops and hedgerow/windbreaks from interviews with 20 growers and 8 technical assistance providers in California's Central Coast.

Theme	# of farmers discussing (N = 20)	# of TA providers discussing (N = 8)	Illustrative quotes
Farmers prioritize adoption of cover crops and hedgerow/windbreaks in hillier and more marginal areas	6	0	"Erosion control is another thing. If we have steep slopes, then we'll intentionally plant those with cover crops in the wintertime." "A lot of the cover cropping does happen on some of the poorer soils." "The flat land is mostly in crop production, and then anything that is more on a sloping portion of the land is mostly in native oak woodland ... and we have put in hedgerows along the borders of the fields So hedgerows, I think, play a really important role to kind of buffer those zones off and protect the native habitat that we really like." "When we first arrived, the low spot on the property which receives most of the drainage from the front half of the farm had been badly eroded We took that out of production and planted cuttings from different riparian plants."
Higher land rents on prime farmland discourage adoption of hedgerow/windbreaks and cover crops	11	8	"We do some cover cropping, but it's challenging with our rent structure. Can I tell the landlord, hey, don't charge me this year because I'm going to grow a cover crop?" "One of the tough things to balance with cover crops is because our rent is so high here, that it's hard to take the land out of production." "The rental costs along the coasts are high and people don't think they can afford to cover crop as much as they should or rotate as much as they should."
The marketing relationships tied to prime farmland discourage hedgerow/windbreaks and cover crops due to stringent food safety protocols and rigid supply chain requirements	7	8	"I know from my time talking to bigger farmers ... that cover crops have the potential to delay planting, and the big firms are on really tight planting schedules, right? So that's why they don't do that." "We have to be very careful. Like I said, we've never experimented with hedgerows and stuff like that we don't have any type of hedgerows or anything like [smaller scale neighboring farmer with direct markets] has out there That is tough because, in the eyes of fresh produce, food safety sometimes trumps some of the ecosystem, right? Do this, or don't grow this stuff for us anymore. What do you do? It's super challenging."
Early adopters of cover crops and hedgerow/windbreaks provide models for neighbors who learn about the practices and observe benefits	5	4	"I think what becomes common practice does so by sort of personal diffusion of information and experiences, whether that's from technical assistance advisors or their peers. So another barrier then would be if you're in a region where people aren't using those kinds of practices, then you don't necessarily have what you need in order to make the changes." "For farmers to be able to go to places and see and hear from others and to be able to see the results, I mean that's probably the single most important thing that could persuade a farmer to try something out. So however that happens, whether it be demonstration farms or farmer-to-farmer learning networks, things like that can be super helpful." "I've definitely seen farmers that, oh, they saw this thing at their neighbor's place or on this field day that they managed to get to and they want to try it. That can be huge." "Over the years, there's been a few people that I've really valued their thinking on and have been able to interact with and share ideas and I've gotten ideas. Whenever I go to somebody else's farm, it doesn't really matter what they're doing or what their specific crops are, something can pique your interest that you can think about, "Yeah. Something like that might work on our farm."

when in proximity to cropped areas—and several of the farmers we spoke to mentioned deliberately maintaining this native vegetation as part of their approach to farming adjacent to wildlands.

Secondly, a number of farmers ($n = 5$, Table 4) mentioned learning about practices like cover cropping and hedgerows/windbreaks from watching fellow farmers, often their neighbors. Half of the technical assistance providers we spoke to mentioned this form of peer learning as a key factor in adoption of these practices. This may partially explain the observed spatial autocorrelation (and existence of hotspots) in hedgerow/cover crop adoption.

In contrast, conventional growers ($n = 5$) discussed a trend of moving away from cover cropping and hedgerows, practices that were more widely used in the past. For these growers, like the large-scale, wholesale organic growers, the quick turnover necessitated by intensive planting schedules made cover cropping prohibitive. One grower stated he plants cover crops only on 1% of winter acres because of the non-stop planting of new crops. For conventional farms, cover crops are used in special circumstances where they can

help solve a problem in the field. For example, one conventional farmer uses cover cropping on fields next to a river that is prone to flooding. In this case, cover crops can help dry the area and prevent flooding, which reduces the delay in accessing the field for planting. Additionally, farmers who produce both organic and conventional produce (split operations) reported using cover crops only on their organic land.

Discussion

Our study shows that adoption of two key diversification practices, hedgerows/windbreaks and winter cover crops, is low and patchily distributed throughout the Central Coast agricultural region of California. This remote sensing analysis provides the first spatially-detailed information on the extent and pattern of hedgerow/windbreak and cover crop presence in California. While most ranches did not have any hedgerows/windbreaks (78%), the identification of several hotspots of adoption suggests that particular landscape and/or social factors and policies may encourage use of

hedgerows. For winter cover crops, our finding of 5.9% are consistent with literature estimates of about 5% of Central Coast total farmland area cover cropped (Brennan, 2017), and also similar to the statewide average of 4.8% of “available” farmland cover cropped (USDA United States Department of Agriculture, 2019). While this figure does not account for winter cash crops, some of which provide similar ecosystem services as cover crops, ~60% of farmland in the Central Coast was mapped as having bare soil between mid-December and the end of February (Supplementary Table S4). Despite their many benefits, adoption of non-crop vegetation like cover crops and hedgerows/windbreaks is limited in California’s Central Coast due to persistent structural and technical barriers.

Remote sensing of non-crop vegetation in complex agricultural landscapes

Our study successfully identified hedgerows/windbreaks in a heterogeneous agricultural landscape using easily accessible NAIP imagery with 90.0% accuracy. Other studies have also used OBIA to identify hedgerows and related vegetation in agricultural settings with varying levels of accuracy (Vannier and Hubert-Moy, 2008; Sheeren et al., 2009; Ghimire et al., 2014; O’Connell et al., 2015). The high accuracy of our model is likely due to the high resolution of recent NAIP imagery. Given that some hedgerows are quite narrow, fine-scale imagery is necessary to distinguish hedgerows from other linear elements in heterogeneous agricultural landscapes.

Our study also successfully mapped cover crops in a highly complex agricultural landscape with dozens of cash crop types—including overwintering crops like broccoli—as well as irregular field sizes and the presence of different land uses (e.g., rangeland and riparian areas) with 87% accuracy. Previous studies that have leveraged the increasing availability of satellite data (e.g., Landsat and Sentinel) alongside cloud computing resources (e.g., Google Earth Engine) to map cover crops in agricultural landscapes have been conducted in much more simplified agricultural landscapes (Howard et al., 2012; Ok et al., 2012; Shelestov et al., 2017; Phan et al., 2020). Higher accuracy has been reported when remote detection of winter cover crops is based solely on vegetation presence (Seifert et al., 2018) rather than needing to distinguish between cover crops and other overwintering cash crops as in our study area.

There were several limitations to our study. Other papers reported similar difficulties to those we encountered; most noticeably, model confusion between hedgerows/windbreaks and other small farm elements such as drainage ditches and shrubs. In our case, this led to the over classification of other linear elements as hedgerows, as noted by the lower user’s accuracy of the model, which had to be manually checked, reducing some of the time saving benefits of remote sensing. Additional ground-truth data of hedgerows/windbreaks and other linear elements would likely improve model accuracy in the future. Additional data for the cover crop classification model would also be useful as certain crops like carrots and fennel were not found in enough fields to generate the recommended 40–120 training points needed per class (Mather and Koch, 2011). Due to the limited number of ground truth points for many of the cash crops, winter cash crops were grouped into a single class with a high variation of spectral properties. This made distinguishing the cash crop class from the cover crop class

more difficult, though some cash crops grown over winter provide similar ecosystem services as cover crops. For example, broccoli, cauliflower, cabbage scavenge high amounts of nitrogen during the winter that could reduce nitrate leaching (Smith et al., 2013). Broccoli also leaves a significant amount of high-quality residue as a nitrogen source for the subsequent crop, whereas other common crops in rotation, like lettuce and spinach, have shallower root systems and produce much less biomass and fewer residues. We estimated annual cash crops (other than strawberries) covered 16% of farmland (Supplementary Table S4), vs. 60% with bare soil, but we could not distinguish crop species or varieties. Remote sensing that could distinguish cash crops and map their cover would allow for a more complete assessment of risks and opportunities for at least some important services of winter plant cover.

In addition, typical crop classification distinguishes crops based on unique NDVI values (Foerster et al., 2012; Howard et al., 2012). Many cover crops found in the region were a mixture rather than a sole crop, often comprised of species also grown as cash crops (e.g., brassicas), which makes distinguishing by NDVI values more difficult. Finally, since cover crop classifications have pixel noise, field-based or OBIA classification could provide higher classification accuracy (Ok et al., 2012; Li et al., 2015), but a dataset with accurate field boundaries is not available for this region where fields often contain several different crops planted in blocks or are managed in distinct sections.

Patterns of adoption

Integrating qualitative interviews with remote sensing allowed for interpreting patterns of adoption and provided insight into biophysical and socio-economic drivers of adoption patterns. Our remote sensing of hedgerows/windbreaks found significant clustering of the practice, with strong hotspots of use in the hillier, less intensively farmed areas of San Benito and Santa Cruz counties. Similar patterns were found for cover crops with a hotspot in the more marginal farmland of Monterey County. A study of farms across 20 counties in Indiana found that cover cropped fields were significantly steeper than non-cover cropped fields, likely for erosion control, and that farms that cover cropped were often smaller (Lira and Tyner, 2018), much like the mid-sized farms in our region that have reported using similar diversification practices (Esquivel et al., 2021). This was also supported in our interviews where many farmers reported maintaining hedgerows/windbreaks and cover crops in hilly or less prime farmland to prevent erosion. The paucity of cover crops and hedgerows/windbreaks in areas like the prime flatland area of Monterey County likely also stems from high rents and pressures growers in intensively managed farms face to keep as much land in production as possible as well as maintain “clean” fields for buyers. While Prime Farmland made up about 60% of our study area, <45% of both cover crops and hedgerows/windbreaks were found on Prime Farmland. Conversely, both practices were overrepresented in Unique Farmland and “Other” farmland, which can be marginal land. Landscape elements (hedgerows, tree clusters, riparian buffers) have been found to be inversely correlated with the presence of intensively farmed land (often with livestock) (Klimek et al., 2014). We also noted many hedgerows/windbreaks and windbreaks in vineyards during our windshield survey. Vineyards are commonly

found on land classified as Unique Farmland, especially in the south of Monterey County, which could explain the hedgerow/windbreak hotspot in this area. Moreover, farmers we interviewed who were raising crops on “Unique Farmland” tended to be small or mid-sized and more likely to sell their produce through regional grocery stores, community supported agriculture, farmers markets, or regional aggregators—alternative agri-food networks that could help offset the costs of using these practices while also imposing less pressure to meet supply chain requirements (Esquivel et al., 2021).

Social mechanisms can influence the formation of hotspots. Early adopters could provide models for neighbors who learn about the practices and observe benefits. Peer influence and localized farming norms may also support diffusion. In California’s Sacramento Valley, farmers reported that other farmers were the most important source of knowledge regarding edge plantings on their farms (Garbach and Long, 2017), while several farmers in our interviews also mentioned learning about such practices from their neighbors.

In general, organic farmers are more likely to use hedgerows/windbreaks and cover crops than conventional growers, given their greater reliance on the ecosystem services, rather than synthetic inputs, as well as organic certification guidelines, which encourage these practices. In this region, these practices are more common among smaller to mid-scale organic farms (Esquivel et al., 2021). Santa Cruz County had the highest percentage of cover cropped fields, has a high percentage of organic production (12% of acres), and it also has the smallest average farm size (102 acres; USDA United States Department of Agriculture, 2019) compared to San Benito County and Monterey County (853 and 1,214 acres, respectively). This indicates a greater prevalence of large-scale agricultural production in the latter counties, which may explain the lower levels of cover cropping. This is consistent with our interviews that large-scale farms working with wholesale buyers are disincentivized to plant cover crops and hedgerows. Cold spots in the counties may represent large areas of intensive, commercial conventional farming.

Conclusions: Remote sensing of agricultural practices to track and support policy goals

A number of federal and state policies depend on adoption of agricultural practices like hedgerows/windbreaks and cover cropping to help meet goals related to climate change mitigation, water quality, pollination, and more. For instance, California’s Healthy Soils Program pays farmers and ranchers to adopt agricultural practices, including cover crops and hedgerows, known to reduce greenhouse gas emissions or increase soil carbon. It also funds demonstration projects meant to enhance adoption through regionally specific practice implementation. Yet the state currently has no means of tracking the efficacy of the Healthy Soils Program, especially whether adoption is maintained following the three-year grants or if demonstration projects spur adoption beyond direct grantees. At the state and national level, there is also no mechanism in place to track impacts of federal programs like the Environmental Quality Incentives Program and the Conservation Stewardship Program, both of which provide support for farmers to adopt these and other practices and have been recently expanded with passage of the Inflation Reduction Act (Inflation Reduction Act, 2022).

Our regional analysis reveals that aerial and satellite imagery can be used to map adoption of hedgerows/windbreaks and cover crops with a high degree of accuracy even in complex agricultural landscapes. The overall percentage of cover crops (~5%) matches past expert estimates for the study region and statewide adoption rates. Future work should distinguish and map winter cash crops that could provide similar services. We also provide new data on hedgerow/windbreak crop adoption in the Central Coast. Relative to other methods of tracking adoption like surveys, this approach is also able to identify spatial patterns, including the existence of hot and coldspots of adoption. Coupling remote sensing with qualitative interviews provided insights into the drivers behind these patterns, including interrelated factors related to topography, land values, and farming model that either enabled or hindered adoption. In turn, this understanding could inform creation of enabling policies while using remote sensing tools to evaluate progress.

Data availability statement

Publicly available datasets were analyzed in this study. This data can be found here: <https://www.usgs.gov/centers/eros/science/usgs-eros-archive-aerial-photography-national-agriculture-imagery-program-naip> and <https://scihub.copernicus.eu/>.

Ethics statement

The studies involving human participants were reviewed and approved by Committee for Protection of Human Subjects. Written informed consent for participation was not required for this study in accordance with the national legislation and the institutional requirements.

Author contributions

JT, JS, and TB conceived the idea. JT and JS designed the methodology, collected, and analyzed data. LC and JO conducted and analyzed interviews. TB, DK, and AI acquired funding. JT, JS, LC, and TB co-wrote the manuscript. All authors contributed critically to the drafts and gave final approval for publication.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

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References

- Albrecht, M., Kleijn, D., Williams, N. M., Tschumi, M., Blaauw, B. R., Bommarco, R., et al. (2020). The effectiveness of flower strips and hedgerows on pest control, pollination services and crop yield: a quantitative synthesis. *Ecol. Lett.* 23, 1488–1498. doi: 10.1111/ele.13576
- Alternatives to Chlorpyrifos Work Group. (2020). *Towards Safer and More Sustainable Alternatives to Chlorpyrifos: An Action Plan for California* (p.152). Available online at: https://www.cdpr.ca.gov/docs/chlorpyrifos/pdf/chlorpyrifos_action_plan.pdf (accessed September 22, 2022).
- Baur, P., Driscoll, L., Gennet, S., and Karp, D. (2016). Inconsistent food safety pressures complicate environmental conservation for California produce growers. *Calif. Agric.* 70, 142–151. doi: 10.3733/ca.2016a0006
- Beck, H. E., Zimmermann, N. E., McVicar, T. R., Vergopolan, N., Berg, A., and Wood, E. F. (2020). Publisher correction: present and future Köppen-Geiger climate classification maps at 1-km resolution. *Sci. Data* 7, 1–2. doi: 10.1038/s41597-020-00616-w
- Beretti, M., and Stuart, D. (2008). Food safety and environmental quality impose conflicting demands on central Coast growers. *Calif. Agric.* 62, 68–73. doi: 10.3733/ca.v062n02p68
- Blaschke, T. (2010). Object based image analysis for remote sensing. *ISPRS J. Photogramm. Remote Sens.* 65, 2–16. doi: 10.1016/j.isprsjprs.2009.06.004
- Brennan, E. B. (2017). Can we grow organic or conventional vegetables sustainably without cover crops? *Horttechnology* 27, 151–161. doi: 10.21273/HORTTECH03358-16
- Brennan, E. B., and Smith, R. F. (2005). Winter cover crop growth and weed suppression on the central coast of California. *Weed Technol.* 19, 1017–1024. doi: 10.1614/WT-04-246R1.1
- Büchi, L., Wendling, M., Amossé, C., Necpalova, M., and Charles, R. (2018). Importance of cover crops in alleviating negative effects of reduced soil tillage and promoting soil fertility in a winter wheat cropping system. *Agric. Ecosyst. Environ.* 256, 92–104. doi: 10.1016/j.agee.2018.01.005
- California Department of Conservation. (n.d.). *Important Farmland Categories*. Available online at: <https://www.conservation.ca.gov/dlrp/fimmp/Pages/Important-Farmland-Categories.aspx> (accessed December 15, 2022).
- California Regional Water Quality Control Board: Central Coast Region (2021). *Proposed General Waste Discharge Requirements for Discharges from Irrigated Lands. Order No. R3-2021-0040*. State of California Central Coast Regional Water Quality Control Board.
- Calo, A., and De Master, K. (2016). After the incubator: factors impeding land access along the path from farmworker to proprietor. *J. Agric. Food Syst. Commun. Dev.* 6, 111–127. doi: 10.5304/jafscd.2016.062.018
- CARB California Air Resources Board (2022). *Draft 2022 Scoping Plan Update*. Available online at: <https://ww2.arb.ca.gov/sites/default/files/2022-05/2022-draft-sp.pdf> (accessed September 9, 2022).
- Carlisle, L., Esquivel, K., Baur, P., Ichikawa, N. F., Olimp, E. M., Ory, J., et al. (2022). Organic farmers face persistent barriers to adopting diversification practices in California's Central Coast. *Agroecol. Sustain. Food Syst.* 46, 1145–1172. doi: 10.1080/21683565.2022.2104420
- Castle, D., Grass, I., and Westphal, C. (2019). Fruit quantity and quality of strawberries benefit from enhanced pollinator abundance at hedgerows in agricultural landscapes. *Agric. Ecosyst. Environ.* 275, 14–22. doi: 10.1016/j.agee.2019.01.003
- CDA California Department of Food and Agriculture (2022). *California Agricultural Statistics Review 2020–2021*. California Department of Food and Agriculture. Available online at: https://www.cdaf.ca.gov/Statistics/PDFs/2021_Ag_Stats_Review.pdf (accessed December 15, 2022).
- Chapman, M., Wiltshire, S., Baur, P., Bowles, T., Carlisle, L., Castillo, F., et al. (2022). Social-ecological feedbacks drive tipping points in farming system diversification. *One Earth* 5, 283–292. doi: 10.1016/j.oneear.2022.02.007
- Corbin, J. M., and Strauss, A. (1990). Grounded theory research: Procedures, canons, and evaluative criteria. *Qual. Sociol.* 13, 3–21.
- Cranmer, L., McCollin, D., and Ollerton, J. (2012). Landscape structure influences pollinator movements and directly affects plant reproductive success. *Oikos* 121, 562–568. doi: 10.1111/j.1600-0706.2011.19704.x
- Esquivel, K., Carlisle, L., Ke, A., Olimp, E., Baur, P., Ory, J., et al. (2021). The “Sweet Spot” in the Middle: Why Do Mid-Scale Farms Adopt Diversification Practices at Higher Rates? *Front. Sustain. Food Syst.* 5. doi: 10.3389/fsufs.2021.734088
- Foerster, S., Kaden, K., Foerster, M., and Itzerott, S. (2012). Crop type mapping using spectral-temporal profiles and phenological information. *Comput. Electron. Agric.* 89, 30–40. doi: 10.1016/j.compag.2012.07.015
- Garbach, K., and Long, R. F. (2017). Determinants of field edge habitat restoration on farms in California's Sacramento Valley. *J. Environ. Manag.* 189, 134–141. doi: 10.1016/j.jenvman.2016.12.036
- Ghimire, K., Dulin, M. W., Atchison, R. L., Goodin, D. G., and Shawn Hutchinson, J. M. (2014). Identification of windbreaks in Kansas using object-based image analysis, GIS techniques and field survey. *Agroforest. Syst.* 88, 865–875. doi: 10.1007/s10457-014-9731-4
- Glaize, A., Young, M., Harden, L., Gutierrez-Rodriguez, E., and Thakur, S. (2021). The effect of vegetation barriers at reducing the transmission of Salmonella and Escherichia coli from animal operations to fresh produce. *Int. J. Food Microbiol.* 347, 109196. doi: 10.1016/j.ijfoodmicro.2021.109196
- Gorelick, N., Hancher, M., Dixon, M., Ilyushchenko, S., Thau, D., and Moore, R. (2017). Google earth engine: planetary-scale geospatial analysis for everyone. *Remote Sens. Environ.* 202, 18–27. doi: 10.1016/j.rse.2017.06.031
- Guthman, J. (2004). “Agrarian dreams,” in *Agrarian Dreams* (Oakland, CA: University of California Press).
- Hartz, T. K., and Johnstone, P. R. (2006). Nitrogen availability from high-nitrogen-containing organic fertilizers. *Horttechnology* 16, 39–42. doi: 10.21273/HORTTECH.16.1.0039
- Heinrich, A., Smith, R., and Cahn, M. (2014). Winter-killed cereal rye cover crop influence on nitrate leaching in intensive vegetable production systems. *Horttechnology* 24, 502–511. doi: 10.21273/HORTTECH.24.5.502
- Hively, W. D., Duiker, S., McCarty, G., and Prabhakara, K. (2015). Remote sensing to monitor cover crop adoption in southeastern Pennsylvania. *J. Soil Water Conserv.* 70, 340–352. doi: 10.2489/jswc.70.6.340
- Hively, W. D., Lang, M., McCarty, G. W., Keppler, J., Sadeghi, A., and McConnell, L. L. (2009). Using satellite remote sensing to estimate winter cover crop nutrient uptake efficiency. *J. Soil Water Conserv.* 64, 303–313. doi: 10.2489/jswc.64.5.303
- Howard, D. M., Wylie, B. K., and Tieszen, L. L. (2012). Crop classification modelling using remote sensing and environmental data in the Greater Platte River Basin, USA. *Int. J. Remote Sens.* 33, 6094–6108. doi: 10.1080/01431161.2012.680617
- Inflation Reduction Act. (2022). *H.R.5376, 117th Cong. §2*. Available online at: <https://www.congress.gov/bills/117/congress-house-bill/5376/text> (accessed December 15, 2022).
- Jackson, L., Wyland, L., Klein, J., Smith, R., and Koike, S. (1993). In lettuce production, winter cover crops can decrease soil nitrate, leaching potential. *Calif. Agric.* 47, 12–15. doi: 10.3733/ca.v047n05p12
- Jay, M. T., Cooley, M., Carychao, D., Wiscomb, G. W., Sweitzer, R. A., Crawford-Miksza, L., et al. (2007). Escherichia coli O157:H7 in feral swine near spinach fields and cattle, Central California Coast. *Emerg. Infect. Dis.* 13, 1908–1911. doi: 10.3201/eid1312.070763
- Karp, D. S., Gennet, S., Kilonzo, C., Partyka, M., Chaumont, N., Atwill, E. R., et al. (2015). Comanaging fresh produce for nature conservation and food safety. *Proc. Nat. Acad. Sci.* 112, 11126–11131. doi: 10.1073/pnas.1508435112

- Klimek, S., Lohss, G., and Gabriel, D. (2014). Modelling the spatial distribution of species-rich farmland to identify priority areas for conservation actions. *Biol. Conserv.* 174, 65–74. doi: 10.1016/j.biocon.2014.03.019
- Kremen, C., Iles, A., and Bacon, C. (2012). Diversified farming systems: an agroecological, systems-based alternative to modern industrial agriculture. *Ecol. Soc.* 17. doi: 10.5751/ES-05103-170444
- Kremen, C., and Miles, A. (2012). Ecosystem services in biologically diversified versus conventional farming systems: benefits, externalities, and trade-offs. *Ecol. Soc.* 17. doi: 10.5751/ES-05035-170440
- Kushal, K., Zhao, K., Romanko, M., and Khanal, S. (2021). Assessment of the spatial and temporal patterns of cover crops using remote sensing. *Remote Sens.* 13, 2689. doi: 10.3390/rs13142689
- Lecq, S., Loisel, A., Brischoux, F., Mullin, S. J., and Bonnet, X. (2017). Importance of ground refuges for the biodiversity in agricultural hedgerows. *Ecol. Indic.* 72, 615–626. doi: 10.1016/j.ecolind.2016.08.032
- Li, Q., Wang, C., Zhang, B., and Lu, L. (2015). Object-based crop classification with Landsat-MODIS enhanced time-series data. *Remote Sens.* 7, 16091–16107. doi: 10.3390/rs71215820
- Liaw, A., and Wiener, M. (2002). “Classification and Regression by randomForest.” *R News*, 2, 18–22. Available online at: <https://CRAN.R-project.org/doc/Rnews/> (accessed September 22, 2022).
- Lira, S. M., and Tyner, W. E. (2018). Patterns of cover crop use, adoption, and impacts among Indiana farmers. *J. Crop Improv.* 32, 373–386. doi: 10.1080/15427528.2018.1432515
- Long, R. F., Garbach, K., and Morandin, L. A. (2017). Hedgerow benefits align with food production and sustainability goals. *Calif. Agric.* 71, 117–119. doi: 10.3733/ca.2017a0020
- Lugato, E., Cescatti, A., Jones, A., Ceccherini, G., and Duveiller, G. (2020). Maximising climate mitigation potential by carbon and radiative agricultural land management with cover crops. *Environ. Res. Lett.* 15, 094075. doi: 10.1088/1748-9326/aba137
- Mather, P. M., and Koch, M. (2011). *Computer Processing of Remotely-sensed Images: An Introduction*. Hoboken, NJ: John Wiley and Sons.
- Moran, P. A. (1950). Notes on continuous stochastic phenomena. *Biometrika* 37, 17–23. doi: 10.1093/biomet/37.1-2.17
- Morandin, L., Long, R., Pease, C., and Kremen, C. (2011). Hedgerows enhance beneficial insects on farms in California's Central Valley. *Calif. Agric.* 65, 197–201. doi: 10.3733/ca.v065n04p197
- Morandin, L. A., and Kremen, C. (2013). Hedgerow restoration promotes pollinator populations and exports native bees to adjacent fields. *Ecol. Appl.* 23, 829–839. doi: 10.1890/12-1051.1
- NASS U.S. National Agricultural Statistics Service (2020). *Cash Rents Survey*. United States Department of Agriculture. Available online at: https://www.nass.usda.gov/Surveys/Guide_to_NASS_Surveys/Cash_Rents_by_County/
- O'Connell, J., Bradter, U., and Benton, T. G. (2015). Wide-area mapping of small-scale features in agricultural landscapes using airborne remote sensing. *ISPRS J. Photogramm. Remote Sens.* 109, 165–177. doi: 10.1016/j.isprsjprs.2015.09.007
- Ok, A. O., Akar, O., and Gungor, O. (2012). Evaluation of random forest method for agricultural crop classification. *Eur. J. Remote Sens.* 45, 421–432. doi: 10.5721/EuJRS20124535
- Ord, J. K., and Getis, A. (1995). Local spatial autocorrelation statistics: distributional issues and an application. *Geogr. Anal.* 27, 286–306. doi: 10.1111/j.1538-4632.1995.tb00912.x
- Pereira, M., and Rodríguez, A. (2010). Conservation value of linear woody remnants for two forest carnivores in a Mediterranean agricultural landscape. *J. Appl. Ecol.* 47, 611–620. doi: 10.1111/j.1365-2664.2010.01804.x
- Phan, T. N., Kuch, V., and Lehnert, L. W. (2020). Land cover classification using Google Earth Engine and random forest classifier—The role of image composition. *Remote Sens.* 12, 2411. doi: 10.3390/rs12152411
- Ponisio, L. C., Valpine, P., de, M'Gonigle, L. K., and Kremen, C. (2019). Proximity of restored hedgerows interacts with local floral diversity and species' traits to shape long-term pollinator metacommunity dynamics. *Ecol. Lett.* 22, 1048–1060. doi: 10.1111/ele.13257
- R Core Team (2022). *R: A Language and Environment for Statistical Computing*. R Foundation for Statistical Computing, Vienna, Austria. Available online at: <https://www.R-project.org/>.
- Seifert, C. A., Azzari, G., and Lobell, D. B. (2018). Satellite detection of cover crops and their effects on crop yield in the Midwestern United States. *Environ. Res. Lett.* 13, 064033. doi: 10.1088/1748-9326/aac4c8
- Sellers, L. A., Long, R. F., Jay-Russell, M. T., Li, X., Atwill, E. R., Engeman, R. M., et al. (2018). Impact of field-edge habitat on mammalian wildlife abundance, distribution, and vectored foodborne pathogens in adjacent crops. *Crop Prot.* 108, 1–11. doi: 10.1016/j.cropro.2018.02.005
- Sheeren, D., Bastin, N., Ouin, A., Ladet, S., Balent, G., and Lacombe, J.-P. (2009). Discriminating small wooded elements in rural landscape from aerial photography: a hybrid pixel/object-based analysis approach. *Int. J. Remote Sens.* 30, 4979–4990. doi: 10.1080/01431160903022928
- Shelestov, A., Lavreniuk, M., Kussul, N., Novikov, A., and Skakun, S. (2017). “Large scale crop classification using Google earth engine platform,” in *2017 IEEE International Geoscience and Remote Sensing Symposium (IGARSS)* (Fort Worth, TX), 3696–3699.
- Smith, R., Cahn, M., and Hartz, T. K. (2013). Survey of nitrogen uptake and applied irrigation water in broccoli, cauliflower and cabbage production in the Salinas Valley. *CDA FREP Proc.* 89, 117–119.
- Sobrinho, J. A., Raissouni, N., and Li, Z.-L. (2001). A comparative study of land surface emissivity retrieval from NOAA data. *Remote Sens. Environ.* 75, 256–266. doi: 10.1016/S0034-4257(00)00171-1
- Stuart, D. (2009). Constrained choice and ethical dilemmas in land management: Environmental quality and food safety in California agriculture. *J. Agric. Environ. Ethics* 22, 53–71. doi: 10.1007/s10806-008-9129-2
- Tamburini, G., Bommarco, R., Wanger, T. C., Kremen, C., van der Heijden, M. G. A., Liebman, M., et al. (2020). Agricultural diversification promotes multiple ecosystem services without compromising yield. *Sci. Adv.* 6, eaba1715. doi: 10.1126/sciadv.aba1715
- Tansey, K., Chambers, I., Anstee, A., Denniss, A., and Lamb, A. (2009). Object-oriented classification of very high resolution airborne imagery for the extraction of hedgerows and field margin cover in agricultural areas. *Appl. Geogr.* 29, 145–157. doi: 10.1016/j.apgeog.2008.08.004
- USDA United States Department of Agriculture (2019). *2017 Census of Agriculture (Report no. AC-17-A-51)*. United States Department of Agriculture. Available online at: <https://www.nass.usda.gov/Publications/AgCensus/2017> (accessed December 15, 2022).
- Vannier, C., and Hubert-Moy, L. (2008). “Detection of wooded hedgerows in high resolution satellite images using an object-oriented method,” in *IGARSS 2008-2008 IEEE International Geoscience and Remote Sensing Symposium*, Vol. 4 (Boston, MA: IEEE), 4–731.
- Verboom, B., and Huitema, H. (1997). The importance of linear landscape elements for the pipistrelle *Pipistrellus pipistrellus* and the serotine bat *Eptesicus serotinus*. *Landscape Ecol.* 12, 117–125. doi: 10.1007/BF02698211
- Weller, D. L., Love, T. M., Weller, D. E., Murphy, C. M., Rahm, B. G., and Wiedmann, M. (2022). Structural equation models suggest that on-farm noncrop vegetation removal is not associated with improved food safety outcomes but is linked to impaired water quality. *Appl. Environ. Microbiol.* 88, e01600-22. doi: 10.1128/aem.01600-22
- Zhang, W., Ricketts, T. H., Kremen, C., Carney, K., and Swinton, S. M. (2007). Ecosystem services and dis-services to agriculture. *Ecol. Econ.* 64, 253–260. doi: 10.1016/j.ecolecon.2007.02.024



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EDITED BY

Jacob Jungers,
University of Minnesota Twin Cities,
United States

REVIEWED BY

James DeDecker,
Michigan State University, United States
Michael Gold,
University of Missouri, United States

*CORRESPONDENCE

Diane Mayerfeld
✉ dbmayerfeld@wisc.edu

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Evolving conceptions of silvopasture among farmers and natural resource professionals in Wisconsin, USA

Diane Mayerfeld^{1*}, Keefe O. Keeley^{2,3}, Mark Rickenbach⁴,
Adena Rissman⁴ and Stephen J. Ventura⁵

¹Division of Extension and Center for Integrated Agricultural Systems, Nelson Institute for Environmental Studies, University of Wisconsin-Madison, Madison, WI, United States, ²Savanna Institute, Madison, WI, United States, ³Nelson Institute for Environmental Studies, University of Wisconsin-Madison, Madison, WI, United States, ⁴Department of Forest and Wildlife Ecology, University of Wisconsin-Madison, Madison, WI, United States, ⁵Emeritus, Nelson Institute for Environmental Studies, University of Wisconsin-Madison, Madison, WI, United States

Silvopasture has gained attention as an agroecological practice that may simultaneously meet farmer goals and provide environmental benefits, including climate change mitigation. At the same time there are significant concerns about the potential for livestock to damage trees and forest soils. Like other innovative agroecological systems, silvopasture combines management complexity with limited research knowledge. Unlike annual crops, the effects of silvopasture management can take decades to assess and require forestry as well as agronomic expertise. We conducted mixed-methods research on silvopasture attitudes and knowledge among farmers, agricultural advisors, and foresters in Wisconsin between 2014 and 2019. We asked: (1) How do farmers who practice grazing, agricultural advisors, and foresters perceive silvopasture? and (2) How did coverage of silvopasture change between 2009 and 2019 in a popular grazing publication? Perceptions of silvopasture were influenced by recent weather history, markets for forest and agricultural products, existing land uses, and other contextual factors. Some farmers and agricultural advisors were committed to silvopasture despite significant obstacles to implementing the practice. Over the course of the study period agricultural advisors increased their willingness to provide silvopasture advice to farmers and professional colleagues, and coverage of silvopasture increased in a popular grazing publication. Finally, a multi-county supportive community of practice was associated with greater enthusiasm for the practice. The greater acceptance of silvopasture among resource professionals follows an increase in silvopasture research and outreach in the region. This interest in silvopasture suggests both a need for, and openness to, greater collaboration among forestry and agricultural professionals and farmers to develop sustainable silvopasture standards.

KEYWORDS

adoption, agroforestry, Midwest, human dimensions, silvopasture, agroecology

1. Introduction

The predominant agricultural model of annual row crop monocultures and bare ground seasonal fallow pollutes surface and groundwaters and causes a host of other environmental and social problems (Porter and Voskuil, 2022). In contrast, strategies for providing continuous living cover aim to significantly improve water quality, habitat, aesthetics, and other environmental and social outcomes, while continuing to provide the food, fiber, and fuel society demands (Green Lands Blue Waters, n.d.). One such continuous living cover strategy is silvopasture, an agroforestry practice that intentionally integrates livestock, forage production, and trees.

Shifting annual row crop systems to continuous living cover first requires people to change their ideas about, and goals for, agriculture. Some continuous living cover strategies, like growing cover crops to replace the seasonal fallow, require relatively modest changes to existing annual cropping systems and keep the principal crops, equipment, and planning timelines in place. Even modest agricultural system changes are challenging, though, and as a result the acreage managed by farmers who have shown interest in cover crops far outstrips the amount of land actually planted in cover crops. At the other end of the spectrum of continuous living cover strategies, agroforestry practices such as silvopasture involve major systems changes, including very different crop types (trees and shrubs) and planning timelines of decades. Making these major changes calls for a profound shift in thinking and action on the part of farmers, resource professionals, and policy-makers. This study examined perceptions of silvopasture in Wisconsin from 2014 to 2019.

Within the US, silvopasture systems integrating beef cattle with fast-growing southern pine plantations have been most widely adopted and most studied (Clason, 1998; Ares et al., 2003; Grado and Husak, 2004; Shrestha et al., 2004; Nair et al., 2007; Cubbage et al., 2012). Garrett et al. (2004) proposed silvopasture as a practice that can improve water quality and other environmental outcomes and profits compared to the widespread practice of unmanaged grazing of woodlands in the upper Midwest, while maintaining or increasing meat or milk production (see also Ford et al., 2019). Silvopasture is also seen as an approach to increase carbon storage and reduce the net climate change impacts of agriculture, as well as increase resilience to weather extremes (Montagnini and Nair, 2004; Howlett et al., 2011; Baah-Acheamfour et al., 2014, 2016; Hawken, 2017; Patel-Weynand et al., 2017).

At the same time, there is a long history of natural resource professionals opposing the integration of livestock with trees, especially in western Europe and the US (Dambach, 1944; Ahlgren et al., 1946; Guise, 1950; Abbott, 1954). This opposition stems in part from situations where livestock damage forests, but it also coincided with the professionalization of forest management and the associated assumption that the best use of a forest is to produce timber (Dana and Fairfax, 1980; Rubino, 1996). Forestry professionals continue to be more skeptical of and less knowledgeable about silvopasture than agricultural advisors and farmers. The latter two groups are more likely to support silvopasture, while acknowledging that livestock can compact soil and create erosion (Arbuckle, 2009; Mayerfeld et al., 2016; Stutzman et al., 2019).

Most of the social science research on silvopasture in temperate regions has focused on economic analysis, silvopasture knowledge of resource professionals, and stakeholder perceptions of benefits and costs (Shrestha et al., 2004; Frey et al., 2012; Mayerfeld et al., 2016; Orefice et al., 2017a; Blanco et al., 2019; Wilkens et al., 2022). Stakeholders usually perceive shade and shelter for livestock as key benefits of including trees in the grazing system. Increased income is another widely cited benefit, although in some cases the income benefits are expressed indirectly, for example as “increased utilization of farm woodland” (Orefice et al., 2017a). Reports of silvopasture challenges or disadvantages are less consistent, but problems with maintaining fences and lack of knowledge about silvopasture management are key concerns. Frey et al. (2012) addressed changes in perceptions over time; they reported that farmers in Argentina perceived more benefits and had fewer concerns

about silvopasture after they had several years of experience than when they were first considering the practice.

Following the suggestion of Garrett et al. (2004) that silvopasture may improve environmental and economic outcomes in woodlands degraded by poor management, researchers in the Midwest and Northeastern US began to study silvopasture establishment in existing woodlands (Demchik et al., 2005; Orefice et al., 2017b, 2019; Ford et al., 2019). Many of the farm woodlands in these regions are or were grazed, and much of the existing pasture is in woodlands.

Agroforestry proponents distinguish silvopasture (in which trees, forages and livestock are actively managed for economic and environmental outcomes) from woodland grazing by noting that the latter involves little or no deliberate management of the forage layer, the trees, or the timing and intensity of livestock use (Brantly, 2014). The limited information available indicates that management of pastured woodland (the term used by the Agricultural Census) varies, but that in most cases it is not managed intensively enough to be characterized as silvopasture. In Wisconsin and most surrounding states, the number of farms with pastured woodland exceeds the number of farms practicing rotational grazing, and greatly exceeds the number of farms using agroforestry practices including silvopasture as well as forest farming, windbreaks, alley cropping, and riparian buffers (Figure 1). Across the US, 326,279 farms had pastured woodland, 265,162 farms practiced rotational grazing, and only 30,853 farms practiced agroforestry in 2017 (USDA-NASS, 2019a,b).

Although some farmers practice silvopasture without knowing the technical term, farmers and natural resource professionals in Wisconsin report that most cases of woodland grazing do not include active management of the forage or trees (Keeley, 2014; Mayerfeld et al., 2016; Galleguillos et al., 2018). Only 23% of Wisconsin farms with pasture practice rotational stocking, a necessary component of silvopasture management in this region, and likely only a subset of those farms manage their rotation intensively (USDA-NASS, 2019b; Whitt and Wallander, 2022).

In this context of complexity, controversy, emerging research, and extensive woodland grazing where silvopasture could potentially be practiced, we examined attitudes toward and knowledge about silvopasture during the 6 years following the initiation of silvopasture research and outreach in and around Wisconsin. Specifically, we asked two research questions:

1. How do farmers who practice grazing, agricultural advisors, and foresters perceive silvopasture?
2. How did the amount and type of coverage of silvopasture change between 2009 and 2019 in a popular grazing publication?

2. Methods

This is a descriptive, exploratory mixed-methods study (Byrne and Ragin, 2009; Yin, 2009). To assemble our case, we used (1) focus group and individual interviews clustered in two regions, (2) end of program evaluations, (3) content analysis of a popular grazing publication, and (4) participatory observation. This approach allowed us to examine silvopasture attitudes and knowledge in context, examine interactions among factors, and in some cases observe changes over time. Research with human subjects was approved by the UW-Madison Institutional Review Board (# 2015-1521).

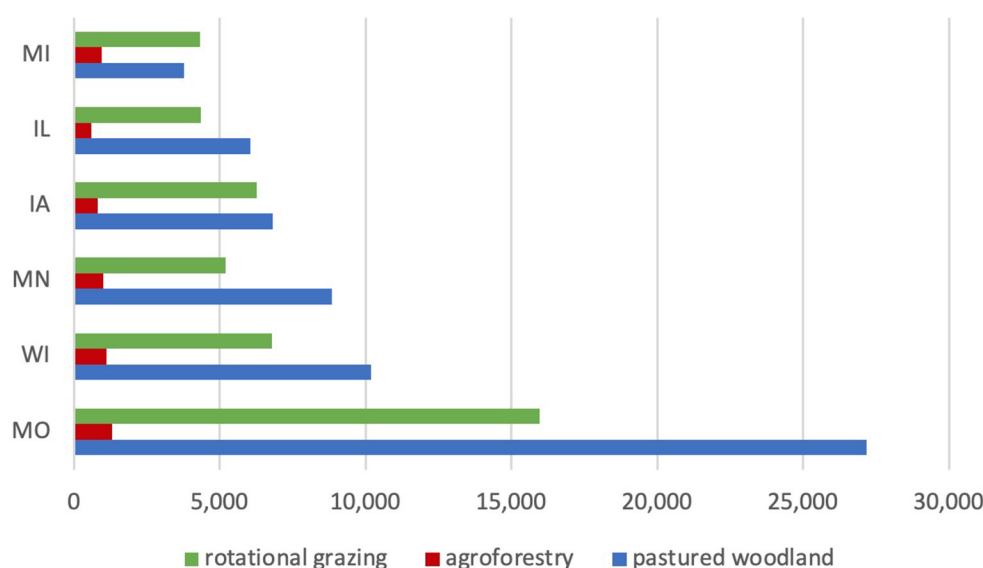


FIGURE 1

Number of farms in Wisconsin and nearby states engaged in the following three practices in 2017: rotational grazing, agroforestry (including but not limited to silvopasture), and pastured woodland (which may also include some silvopasture) (USDA-NASS, 2019a,b).

The subjects of our study were three categories of silvopasture stakeholders in Wisconsin: *farmers*, *agricultural advisors*, and *foresters*, with the latter two categories referred to collectively as *resource professionals*. We focused our study on southwestern and northwestern Wisconsin, but also included stakeholders throughout the state.

In 2014, we began interviewing farmers, agriculture advisors, and foresters about their views on integrating livestock grazing with trees. In their capacities as educators, two of the authors (one with University of Wisconsin Extension and one with the Savanna Institute, an NGO focused on agroforestry research and education) also began conducting educational outreach about silvopasture in 2014. In 2015, we initiated two silvopasture research trials: one on a university research station and the other on two commercial farms. Our work occurred in the context of other agroforestry outreach and research in the region and nationwide. For most farmers and resource professionals in Wisconsin, the workshops, conference presentations, and pasture walks we helped organize were a major source of silvopasture exposure.

2.1. Interviews

We conducted 12 focus group interviews with farmers, agricultural advisors, and foresters between 2014 and 2019 (Table 1). We also conducted individual interviews with two agricultural advisors, a forester, and five farmers who could not participate in the focus groups but were interested in contributing to the project. The focus group interviews form the foundation for our case study. The individual interviews supplemented the focus group interviews and provided a check that there were not issues and questions that participants hesitated to bring up in a group setting.

To some extent, the results of the six focus group interviews in 2014, 2016, and September 2017 serve as a baseline of silvopasture knowledge and attitudes early in the study period. In these initial interviews we asked the participants for their thoughts about integrating grazing livestock with trees and about silvopasture. Although the September 2017 interview took place more than 3 years after the start of the project, the participants were all foresters with whom we had no previous interactions, and for whom our questions about silvopasture were novel.

The four focus group interviews conducted in 2018 and 2019 included 12 individuals who had participated in earlier interviews, as well as at least six individuals who had participated in one or more silvopasture events, such as a pasture walk or presentation. In these later interviews we added prompts asking participants where they had first heard about silvopasture and asking them to reflect on changes in silvopasture knowledge, attitudes, and practices in the past 5 years.

The focus group interviews conducted in January and March 2017 were intermediate in nature. We had not interviewed the participants before, but they were aware of our work, and some had attended a silvopasture event before the interview. Like the individual interviews, they supported the findings of the early focus groups.

Our interviews were clustered in two regions, northwestern and southwestern Wisconsin. The northwestern region is a relatively level landscape shaped by glaciation, with agricultural systems limited by a short growing season and low natural soil fertility. In contrast, southwestern Wisconsin is located in the unglaciated Driftless Area, which has steep topography, making it marginal for large-scale row crop production. Both areas contain substantial woodland, primarily mixed hardwoods and a few small red or white pine plantations. In southwestern and northwestern Wisconsin counties woodland accounts for 15–36% of total farmland (USDA-NASS, 2019a). Roughly 30% of farms have beef cattle, and 6–21% of farms have dairy cows.

TABLE 1 Focus groups dates and participants.

Year	Month	# people*	Male	Female	Farmer	Ag. advisor	Forester	Other
2014	Feb	8	4	4	2	2	5	1
2014	March	2	1	1		2		
2014	May	7	7	0	7			
2016	March	12	8	4	12	1		
2016	Oct	5	3	2		5		
2017	Jan	3	2	1	3			
2017	March	2	2	0	1		1	1
2017	Sept	9	9	0	1		9	
2018	Sept	8	8	0	8**			
2018	Nov	12	9	3	12			2
2019	Jan	7	5	2	2	4	2	1
2019	Feb	6	6	0	5	2		

*The sum of farmers, foresters, and agricultural advisors exceeds the total number of interviewees because several of the natural resource professionals also farm.

**This focus group took place outdoors after a pasture walk, and participants did not fill out a demographic form, but all described themselves as farmers in introductions.

2.1.1. Interviewees

Participants in the five focus groups conducted in 2014 and 2019 were invited based on their experience operating grass-based farms or as resource professionals. The other seven focus groups took place in the context of conferences or pasture walks and were open to any event attendees who chose to participate.

Participant ages ranged from under 30 to over 70, and length of time in their current position (including farming) ranged from <2 years to more than 50 years. The amount of land farmers had in woodland was highly variable, from no woods on the farm to the majority of land in woods, with many respondents having between 10 and 50% of their land in woods. Thirty-three participants managed beef or dairy cattle; five managed sheep, goats, poultry, bison, or pigs. We recruited farmer participants through grazing networks, so the farmers we spoke with practiced rotational stocking (also known as rotational grazing, managed grazing, or adaptive multi-paddock grazing). Because rotational stocking is a requirement for silvopasture management in this region, farmers who practice grazing are the most likely group to try silvopasture. Education levels ranged from high school (10th grade) to graduate degrees in the farmer focus groups.

The farmers participating in the focus groups had a range of experience with and attitudes toward silvopasture. Each farmer focus group had at least one farmer who had no trees in their pastures, as well as at least one farmer who was managing pasture with trees.

Agricultural advisors included university extension, public agency [e.g., Natural Resources Conservation Service (NRCS)], and non-governmental organization (NGO) staff, and grazing consultants or technical service providers (TSPs). Foresters included university extension and Wisconsin Department of Natural Resources staff and private foresters. All resource professional respondents had a 4-year college degree or higher.

2.1.2. Interview structure and analysis

For the interviews, we used guiding questions but also allowed the conversation to flow naturally and encouraged respondents to interact with each other as well as the interviewer(s). All focus

group interviewees consented to having the session recorded, but the recorder malfunctioned at one focus group.

Transcripts from the 2014 to 2017 focus groups were coded manually using a grounded theory approach (Morgan et al., 2008). Focus groups in 2018 and 2019 were coded manually according to the categories that emerged from the initial coding, as well as their responses to a new prompt about changes in knowledge and attitude. Our interview analysis focused on qualitative identification of issues, attitudes, and connections rather than attempting to assess the relative importance of themes through number of mentions or other quantitative measures.

2.2. Evaluation

During the study period we conducted numerous educational programs on silvopasture in Wisconsin, including seven statewide conference presentations, four pasture walks in southwestern Wisconsin, and three 2-day workshops (one in northwestern and two in southwestern Wisconsin and southeastern Minnesota), as well as media interviews and other events. We used end of program evaluation forms at all the workshops, three pasture walks, and two conferences to collect information from participants about their perceptions of silvopasture, as well as their silvopasture information sources and needs. These evaluation results supplement the interview findings.

2.3. Content analysis

Graze magazine focuses on grazing advice, and both farmers and agricultural professionals use it as an information source. The magazine is headquartered in Wisconsin and has been reaching an audience of farmers using managed grazing since 2000. It has ~2,000 paid subscribers across the US, Canada, and overseas, with high concentrations of readers in the Upper Midwest and Northeast states. We conducted a summative content analysis of *Graze* from January

2009 to May 2019 for several terms that we thought would appear in any discussion of silvopasture or integration of livestock with trees (Hsieh and Shannon, 2005). The search terms we used were “shade,” “silv”¹, “tree,” “wood,” “heat,” “brush,” “forest,” and “shrub.” We only counted instances of the term that related to the integration of livestock with trees. In addition to noting when and how often the topic of trees in grazing systems came up, we assessed how trees were discussed. This analysis provided an additional window on attitudes toward silvopasture, as well as the availability of silvopasture information in the farming community. In contrast to the interview analysis, this content analysis includes a quantitative component.

2.4. Note on author engagement

During the study period authors DM and KK also conducted silvopasture field trials in southwestern Wisconsin, and we organized and presented at a variety of silvopasture outreach events. Thus, we were actively engaged in discussions around silvopasture in the state at the same time that we were conducting this study. Our roles as researchers and educators likely influenced who was willing to be interviewed and may have affected what interviewees said. Our active participation in silvopasture research and outreach allowed us to observe conversation around silvopasture beyond the formal methods of interviews and written evaluation responses.

3. Results

3.1. How do farmers who practice grazing, agricultural advisors, and foresters perceive silvopasture?

3.1.1. Farmer perceptions and knowledge

Throughout the study period farmers expressed a range of attitudes toward silvopasture, from uncertainty about its environmental and economic sustainability on their farms to strong enthusiasm for the practice. We did not observe an overall shift to more positive or more negative perceptions among farmers, but we did see differences in how farmers discussed silvopasture at different times, depending on individual farm experience and wider contextual factors.

In all the focus groups, farmers who had been managing silvopasture on their land demonstrated their knowledge by talking about specific management practices and observations based on their experience. In the group interview setting, farmers who did not have personal silvopasture experience did not portray themselves as having silvopasture knowledge, even though some of them mentioned having read or heard about the practice. Often farmers in the focus groups avoided using technical language, including the term silvopasture, even when they were familiar with the terminology.

Several topics appeared in all the interviews: the potential impact of silvopasture on animal welfare, farm profitability, soil and water quality, biodiversity, and the presence of shrubs. However, at the later focus groups there were some shifts in emphasis that reflected changes in the broader farm economy and recent weather patterns

TABLE 2 Overview of silvopasture knowledge and attitudes in Wisconsin USA and surrounding states from 2014 to 2019 interviews with farmers, agricultural advisors, and foresters; evaluations following educational events; and content analysis of a popular grazing publication.

Finding	Patterns and trends
Attitude: A relatively small but dedicated set of farmers is interested in exploring silvopasture (3.11, 3.12)	<ul style="list-style-type: none"> • Farmers' confidence with silvopasture management depended on their goals and own farm experience. • Farmers' and resource professionals' attitudes toward silvopasture were influenced by local context, such as timber markets and recent weather, and by participation in communities of practice.
Attitude: The taboo around silvopasture is weakening, and some agricultural advisors began to provide silvopasture advice (3.12, 3.2)	<ul style="list-style-type: none"> • Early in the study period resource professionals did not address silvopasture in their work. Late in the study period some agricultural advisors gave silvopasture advice, and some foresters were open to considering silvopasture applications. • Coverage of the benefits of trees in pasture systems increased during the study period in a popular grazing publication.
Knowledge: Silvopasture management is more complex, and site- and goal-specific than the dominant grain and livestock systems in the region (3.11, 3.13)	<ul style="list-style-type: none"> • Throughout the study period silvopasture variability and uncertainty continued to challenge resource professionals. • Farmers and agricultural advisors are experimenting with silvopasture to meet goals such as shade and shelter for livestock, brush management, and increased forage. • There is demand for locally-relevant information about silvopasture management, economics, and environmental impacts.

and increased knowledge about silvopasture on the part of both farmers and resource professionals. Key research findings from interviews, as well as from written evaluations following educational events and content analysis of a popular grazing publication, are summarized in Table 2.

Most of the discussion in our farmer interviews centered on conversion of existing farm woodlands to silvopasture, although at least three of the farmers interviewed had planted trees in their pastures. None of the focus group participants expressed direct opposition to silvopasture.

3.1.1.1. Farmer perceptions of benefits and concerns with silvopasture

Key benefits interviewees associated with silvopasture were shade and shelter for livestock; the potential for increased income because of additional pasture, harvest of forest products, and/or lower property taxes associated with converting woodland to silvopasture²; and reduction of brush (i.e., understory shrubs that obstruct herbaceous forage growth, passage and visibility). Concerns included the potential for damage to trees and soils, as well as increased

¹ We used “silv” to capture alternative spellings, e.g., silvopasture or silvopasture or silvopasturing or silvopasture.

² Unlike most states, Wisconsin property tax law assigns the lowest tax rate to “wooded pasture” (Wisconsin Department of Revenue, 2022).

labor to maintain fences and manage the forage layer when trees are present.

These benefits and concerns reveal interactions and some tensions among shared norms and individual values, constraints, and experience. Take these comments from a farmer in a focus group in 2016. Early in the focus group we asked all the farmers to comment on whether they were currently integrating their grazing with trees or considering it. One farmer explained

I have pigs and am interested in feeding the pigs acorns. I've been bringing the pigs acorns because I know that the pigs can really tear up an environment. I have a lot of closed woods with really nice trees and wouldn't dare let the pigs go there. But this little segment that was logged. It has some nice scattered oaks, ... but what's filling in between them is popple, little tiny popple [*Populus sp.*]. Four inches apart – you can't even walk through it. ... I suppose if you're a woodcock it's wonderful. If I were going to move a hog under an oak tree it would be on that piece right there. And then with the hopes that ... I could turn this stand into silvopasture with these sparse oaks if I can get rid of the popple, which I'm sure a hog can do. ... It seems like a good idea, but I'm not sure. ... Most people would say you're not ruining a great field or anything. But there could be something wonderful in there – I don't know.

After an hour of discussion among the 11 farmers in the group, ranging from the animal welfare and tax benefits of silvopasture to its potential impacts on forest soils and trees, this same farmer was still struggling to reconcile the norms and values of providing animal welfare, running a profitable farm, and caring for the environment:

You're rich in direct proportion to the things you can afford to leave alone. And I'm very cautious. When I talk about doing this with hogs – soil science guy says watch out for damage – well leaf cover looks like soil cover to me—things look pretty healthy [as they are now] ... should I even mess with it? That [good woods] is off limits to me; I only toy with the idea of the popple growth. But then woodcock would love that popple.

Hogs embody the conflicting norms around silvopasture particularly strongly because they are highly sensitive to heat stress and thus can benefit from shade, but are also very likely to cause severe soil disturbance because of their rooting behavior. Farmers in all the focus groups spoke about the differences between livestock types, as well as other factors that could affect silvopasture success on a specific farm:

"Question for those using trees at the edge [of fields]: are those trees dying? Ours haven't. Oak, maple, little bit of silver popple."

"Where my trees are, they're tamarack, and [the livestock] rubbed the bark all off, and they're dying."

"If you don't have enough trees and you leave them [the livestock] in long enough, yes, they will [kill trees]. The trick is don't leave them in there very long.... I notice my oak trees grow really fast now that there are animals in there. ... Less competition, more sunlight. Clover, meadow fescue, orchardgrass, some red clover in the open areas. It's my best pasture in the summertime, during the drought."

3.1.1.2. Knowledge-exchange networks, farmer experience, and perceptions of silvopasture in socio-ecological context

The practice of silvopasture is of potential interest to livestock farmers who use grazing as a management practice, and the farmers we interviewed were active in networks that promote rotational stocking. We did not collect information on the details of their grazing management, such as frequency of moves, stocking density, and length of rest periods. In Wisconsin a typical rotation schedule for most grass-based lactating dairy cows involves daily moves over an approximately 30 day rotation. For rotationally grazed beef cows, dairy heifers, dry cows, and small ruminants time in a paddock varies depending on a variety of factors, but is often determined by forage residual height goals. Farmers are advised to size paddocks so the animals will be moved every few days and at least weekly to avoid overgrazing (Cavadini, 2022).

In all the focus groups, farmers emphasized careful management of grazing timing, intensity, and duration as important to mitigating negative impacts on the soil and plants, as well as maintaining the performance of their livestock. Because the timing and duration of grazing is a critical component of silvopasture management, farmers who practice rotational or adaptive multi-paddock grazing are well-positioned to implement silvopasture. Within this group of potential adopters, a subgroup is actively interested in learning about and implementing silvopasture. Although the practice remains poorly understood and adds significant management complexity, that subgroup of interested farmers remained engaged with silvopasture throughout the study period, as evidenced by participation in silvopasture events and by comments in our interviews.

Some farmers showed increasing confidence in silvopasture over the study period, while others expressed more concern about the labor and management needed. For example, in northwestern Wisconsin in 2015 a farmer who had recently converted some woods to silvopasture spoke primarily about the challenges of converting and expressed concerns about how the trees would hold up to livestock impact. In the focus group conducted 32 months later, that farmer was confident about his ability to manage silvopasture (which he often referred to as savanna) and enthusiastic about its benefits for his livestock:

...my [open] pastures always go into dormancy July and August, pretty much. And the savanna pastures do not because of the trees. And while it's not great tonnage, it's of great value because they still have grass when they normally wouldn't.... And now that I've done that, what I value even more is it creates a tremendous amount of diversity in the animal's diet. And I'm absolutely convinced my animals do better than others, not because of genetics, but because of that diversity in their diet. And I really value my savannas because of that. The trees grow faster. We have a lot more game than you normally would, if you're into hunting and that kind of thing.... And if I had to sell land, I'd sell my pastures before I'd sell my savannas.

The grazing network in northwestern Wisconsin included two agricultural advisors who actively supported silvopasture, one of whom had worked with this farmer throughout the process of establishing his silvopasture. In 2014 this network included two presentations and a panel discussion about silvopasture in its spring conference. Farmers learned they could talk about silvopasture with their grazing consultant, and during our study period several of

the pasture walks hosted by the network featured silvopasture. In November 2018 the network's conference again featured a silvopasture presentation.

In contrast, in southwestern Wisconsin agricultural advisors who helped coordinate the grazing networks did not promote silvopasture. Farmers in the initial southwestern focus group identified brush management as a major benefit of silvopasture. While they continued to express interest in managing brush, the 2019 farmer focus group in southwestern Wisconsin placed greater emphasis on the limits of using livestock as a site management tool and on the limits of current silvopasture knowledge. For example, one farmer in the 2019 focus group had cleared an area of woodland for silvopasture. He spoke about how nice it was to regain access to the old oak savanna that had become impassable due to dense understory growth during the years when livestock were excluded. But later in the conversation he added:

We have problems with black locust, and seeing all those runners pop up, it's just a carpet. ... I think [the cattle] get some of those initial sprouts, but it's more of a supplement. With the kind of management system [we use], they're not going wild on it. I do notice they'll get those young, tender sprouts. But if it gets beyond that maybe they'll take a nip of a couple leaves. That's typically what I observe with cattle.

This statement reflected a broader discussion about the challenge of getting sufficient livestock browsing and physical impact to control weedy shrubs and trees without damaging soils or desirable trees. In this same focus group, the farmers discussed the superior ability of goats to browse shrubs but also noted that, like all livestock, goats do not spare the species that a land manager might want to keep. The group also discussed the additional labor required to manage and market multiple livestock species. Similarly, farmers in the 2014 focus group in southwestern Wisconsin spoke of silvopasture as a tool to restore savanna habitat, while farmers in the 2019 focus group in the same region discussed the difficulties and limitations of using livestock for ecological site management such as savanna restoration.

Still, although there was much discussion of the challenges of using grazing to manage the shrub understory, most of the focus group participants felt that livestock could help in some situations. The site with black locust referenced above was part of a silvopasture establishment trial, and in areas planted with improved forages, it was noted that black locust resprouting was much less of a problem compared both to areas that weren't planted and areas that were planted but not grazed. Another farmer, who was quite skeptical of silvopasture, commented.

We had a watershed meeting here last month and one of the members ... fenced off his woods. ... Now it's five years [later] and it's grown up with all this stuff he doesn't want. So he's kind of, 'what do you do, how do you win, or do you have to just be patient and you have to wait fifty, a hundred years for nature to kindly kill this stuff off on its own' or what.

As another respondent said of silvopasture as a strategy to manage brush, "It's not a silver bullet by any means, but it's certainly I think moving in the right direction."

One concern mentioned in a 2018 northwestern focus group was the worry that the growing acceptance of silvopasture could be set back by one bad example:

And then also I'm beginning to wonder about we can make all this progress and ... we're bound to find somebody who's going to do this all wrong. And it's going to be on a major highway and everybody's going to see it where there are 5,000 animals on 10 acres and the hillside comes down and all the trees die. So we need some research to say, "Well, based upon the research, you should never have been doing that or been allowed to do it. And that's why this all happened." It's not the concept. It's the execution of it that was wrong.

This quote illustrates the sense that this loose group of farmers and resource professionals is making progress by working together, as well as their awareness that the approach of integrating grazing with trees still needs to develop clearer guidance, and that research will play an important role in developing that guidance.

3.1.1.3. Contextual factors and economic viability of silvopasture

Farmer comments indicated some regional differences in the economics of converting woodland to silvopasture between the southwestern and northwestern focus groups. In Wisconsin, property tax assessment categories result in lower tax levies on wooded pasture than on ungrazed forest land (not enrolled in state forestry tax incentive programs), and in both regions property taxes were cited as an economic incentive to let livestock graze woodlands. Farmers in both regions saw silvopasture as a way to access those tax benefits without causing the environmental damage associated with unmanaged livestock access to woodlands.

However, in northwestern Wisconsin, where paper mills provide a market for trees that are not timber quality, several farmers mentioned income from commercial thinnings of their woodlands to establish silvopasture. In southwestern Wisconsin the market for wood is limited to high quality sawtimber, and none of the farmers in that area spoke about income from thinning their woods to establish silvopasture.

3.1.2. Resource professionals' perceptions and knowledge

Among resource professionals (i.e., agricultural advisors and foresters) we observed some individuals whose support for silvopasture increased over the course of the study period; we did not observe any individuals who decreased their support. In earlier interviews the agricultural advisors were all open to the idea that silvopasture could play a positive role in Wisconsin grazing farms, and several mentioned examples of farmers who were already experimenting with silvopasture. However, except for one professional in northwestern Wisconsin, they did not talk about providing silvopasture advice in the course of their work. In contrast, in the later interviews several agricultural advisors spoke about incorporating management of paddocks with trees in grazing plans or other advice to farmers:

I usually look at the trees and the cover, see if it's a heavy cover, that might be something we maybe stay out of or just go into during the hot periods for just shade. And if it's a mixed cover with quite a bit of open area, then that might be a separate area for late summer grazing when it's hot.

In 2014, that advisor had said “We were asked to do a presentation on grazing in the woods And we denied it. We didn't want to get into that” (Mayerfeld et al., 2016). In 2019, when asked if incorporating areas with trees was standard practice for grazing specialists, the advisor said, “Right now we're working on that because they usually just see woods, and they just line them out [of the grazing plan].” While this statement shows that many agricultural advisors still were not comfortable providing silvopasture advice, it also indicates that it had become acceptable to promote silvopasture as an agricultural practice to professional colleagues, which was not the case 5 years earlier.

Foresters did not report giving silvopasture management advice but indicated that the opposition to any integration of livestock with trees was softening over time. In a 2019 interview a forester commented that forestry guidance to farmers with woodlands used to be “Don't burn, don't graze and just let it go.” He went on to say

And now what do we do? We tell people, ‘Burning's not so bad. And actually it's fantastic,’ and, ‘Oh, you might want to think about grazing.’ So it's like, okay. We've come a long ways on that.

We also found that foresters in our focus groups varied widely in their attitudes toward silvopasture. At the beginning of this project, we were warned that most foresters were likely to strongly oppose any integration of livestock and trees. In our direct interactions we found that foresters were indeed strongly critical of poorly managed woodland grazing, but most were open to considering how silvopasture management might improve environmental outcomes, at least in some settings. As one forester commented,

Certain agricultural producers out there are going to graze the woodlands, and that's just economics. It's going to happen. So, we should look for those opportunities that we can decrease the environmental impact based on that.

Tentative acceptance of silvopasture was evident both in mixed focus groups that included agricultural advisors as well as foresters, and in a focus group with all foresters. Several expressed particular interest in the potential for goats to manage invasive species.

Like the interviews, workshop, pasture walk, and conference evaluation results suggest that foresters' attitudes toward silvopasture are variable (Figure 2). Nearly half of respondents did not know what their local foresters' attitudes were, but the other respondents reported that forester attitudes toward silvopasture were roughly evenly split between supportive and unsupportive, with many perceived as neutral. Evaluation respondents were primarily farmers but also included a few resource professionals.

Resource professionals' comments about silvopasture were influenced by changes in broader contexts impacting farms and surrounding communities. Two years before our initial focus

group the region had experienced severe drought and extreme heat, while the summers of 2017 and 2018 were relatively cool and wet, and 2018 included extreme precipitation events and flooding. The later focus groups placed less emphasis on the value of trees for shade and woodlands for emergency source of forage, and more emphasis on how silvopasture might handle extreme precipitation. Similarly, shifts in the farm economy were reflected in the discussion. In 2014, when commodity crop prices were high, resource professionals thought silvopasture management might improve environmental outcomes when conversion of pasture to row crop cultivation led to more woodland being converted to pasture. In 2019 resource professionals discussed the increased interest in alternative crops and land management systems such as silvopasture, given depressed crop prices.

The agriculture economy right now, it's especially bad for dairy farmers, but nobody is making very much money right now. This is the first time I've ever heard discussion among dairy farmers about diversifying. ... They're thinking they need to reduce their risk by adding other crops and other sources of income, and trees might be [one of those alternatives].

Although the specific issues changed over time, the discussion among farmers, as well as resource professionals, often highlighted how attitudes toward silvopasture interacted with regional resource and socio-economic issues.

3.1.3. Complexity and uncertainty in perceptions of silvopasture

3.1.3.1. Knowledge limitations

Even though the taboo around discussing the integration of livestock and trees has weakened in our study area, the nature of silvopasture raises challenges for resource professionals who want to offer clear, research-based, financially-sound advice. Silvopasture entails a complex set of principles and practices drawn from both forestry and agricultural science, with context-dependent applications, making universal management prescriptions difficult to develop and deliver. As one forester commented when a focus group was discussing the potential for silvopasture to help with oak regeneration,

I think there are so many variances that could go about this. The type of cattle. If it's beef, dairy cattle, sheep, goats, whatever. There's so many variances in that. The tree species you're wanting to regenerate. The time of year. It seems like a whirlwind of a headache that you're trying to put together.

Furthermore, there are substantial limitations in the fundamental knowledge base, including a lack of regional research. Both natural resource professionals and farmers questioned the applicability of silvopasture research on southern pine plantations to the mixed hardwoods of the upper Midwest:

I'd like to see some controlled experiments in the northern forest rather than just from the southern United States where we could show an impact on the accumulation of forest product.

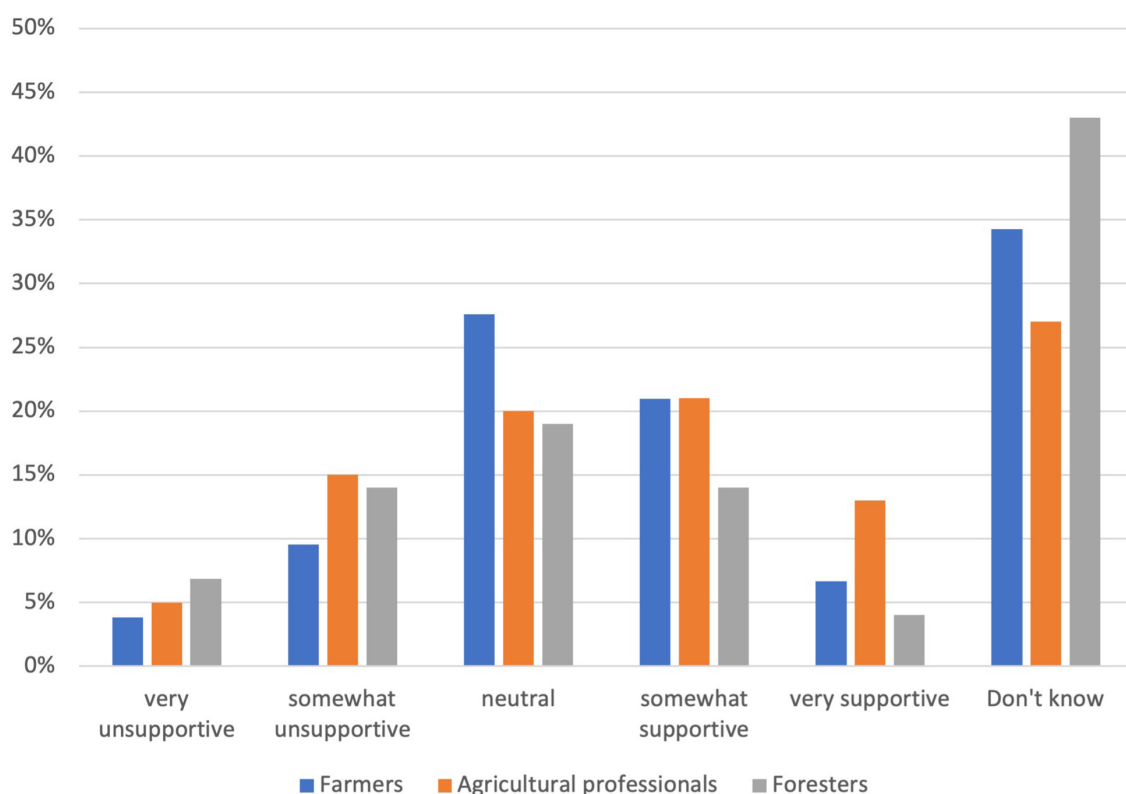


FIGURE 2

Aggregated end of program workshop evaluation responses, 2014–2019, in Wisconsin and Minnesota, USA, to the question “Thinking about the past year, how supportive or unsupportive are professionals and farmers in your county toward silvopasture?” $N = 107$.

This comment was followed by a discussion of the differences between southern pine plantations and diverse northern hardwoods, including slower growth of northern trees, and concluded with the observation that “it might take two generations of scientists to get an answer.”

3.1.3.2. Uncertainty about silvopasture policies and financial support

In the early focus groups, there was only one mention of the possibility of financial support from public agencies for silvopasture. Most of the later interviews, however, included discussion of the possibility of Natural Resources Conservation Service (NRCS) support for silvopasture. This type of financial assistance (provided through NRCS from the US Department of Agriculture) can be an important aspect of silvopasture economics since establishment costs can be substantial, but the interviews reflected considerable uncertainty. NRCS provides financial assistance for rotational grazing but traditionally has strongly discouraged grazing of woodlands. There were some efforts to have NRCS in Wisconsin and some surrounding states provide financial assistance for silvopasture establishment by planting trees, but the reimbursement rates were low, farmers often did not realize they could ask for this assistance, and most agricultural advisors were uncertain about the current policies for silvopasture assistance, as this exchange among resource professionals illustrates:

“And, if we start thinking about silvopasture agroforestry, is that a cost share practice at all? For NRCS?”

“Not right now.”

“So now it would be 100% on the landowner to, again, how long before they start generating revenue or income from that?”

“Well, wait a minute – for planting we don’t call it silvopasture, we call it tree planting. So if you want to plant trees in the pasture, we do cost share that.... There’s also biological brush management... So there’s other practices. We don’t call them silvopasture.”

Another agricultural advisor (and farmer) in a different 2019 interview commented:

And then, with the new EQIP which I work with for cost-sharing with fence or for fencing and watering [for managed grazing], it’s like they are more into promoting converting tillable ground or work ground that can be pasture. When I first started, if you could prove or show there was history of grazing at one time then they would cost share to put the fencing in. But now, if it’s got trees on it, they won’t cost share at all. So that’s actually going to probably blow up your silvopasture part of it to some extent, too.

Over the past 2 years Wisconsin’s NRCS has been working with the Savanna Institute to add financial and technical assistance for planting trees to establish silvopasture, but most farmers and

agricultural professionals, including county NRCS staff, are still uncertain about these policies.

In 2014, resource professionals spoke in general terms about the need for more information on the economics of silvopasture. In the 2019 interview, resource professionals in the southwestern part of the state devoted considerable discussion to the need for better markets for a variety of tree products, from lower quality wood to nuts, in order to increase the economic viability of silvopasture. This focus on markets and financial assistance in later interviews reflects a shift to thinking about silvopasture implementation and advice in concrete rather than abstract terms.

Finally, throughout the study period, farmers and resource professionals stressed that Wisconsin property tax policy is an important economic consideration for silvopasture. Resource professionals were frustrated by the fact that the current law provides a tax break for any pastured woodland, regardless of management and environmental outcomes, and farmers spoke about considerable variation in how local tax assessors interpret the rules. In the November 2018 focus group, one farmer described discussing silvopasture with the assessor:

“We pay much more real-estate taxes on woodland than on cropland, and so last spring, I invited our assessor to come out to the farm. And he was knowledgeable of silvopasture but hadn’t seen any of it, and he didn’t want to go out with me. We sat down and looked at our maps, and he wanted me to show him where I had hardware.... He lowered our valuation—I don’t remember how much—quite a bit on those acres.”

“So he accepted your explanation?”

“Yup.”

“And seemed to be knowledgeable enough to adjust for that?”

“Yup. He’s heard about it, but he just...”

“You were the first person he’d talked to specifically about it.”

“Yeah, well, we’re probably the only rotational grazers in our area.”

3.2. How did coverage of silvopasture change between 2009 and 2019 in a popular grazing publication?

To supplement the interviews we searched all issues of a long-established grazing periodical to understand how perceptions of silvopasture were evolving over time. This analysis revealed an increase in attention to silvopasture over the past decade, as well as a growing appreciation generally of trees as assets to pasture-based livestock systems. Figure 3 summarizes the number of times our search terms appeared in *Graze* in a grazing management context in articles and announcements.

From 2009 until late in 2013 the term “silv[opasture]” was never used in the publication. In November 2013 the term appeared for the first time in an announcement of a combined silvopasture and grazing conference. Then in 2014 *Graze* featured three articles about silvopasture by farmer and writer Tracy Frisch, and the word appeared more than 60 times. In 2015 and 2016 there were no silvopasture articles, and the word only appeared once each year, but in 2017 the word appeared 31 times. In 2018 the word silvopasture

appeared 86 times, with articles about silvopasture by forester and farmer Bret Chedzoy and agroforestry researcher Joe Orefice in five different issues. In the first 6 months of 2019 the word appeared eight times – four times in articles that were not explicitly about silvopasture and the other four times in an article about living barns by Brett Chedzoy, a silvopasture advocate from New York state. However, although the word “silvopasture” does not appear until 2013, many articles both before and after that date refer to the use of trees in pasture systems.

In 2009, 2013, and 2017, *Graze* included a feature where five experienced graziers from different states responded to the question “How do you manage heat stress?” In each of those years use of shade from trees was one of the most common strategies cited in the answers, but there is a progression over that time from barely mentioning shade to discussing shade management in some detail.

For example, in the 2009 *Graze* feature on managing heat stress only one of the farmer columnists listed use of shade as a main strategy, and all mentions of shade were quite brief, like this quote from a Minnesota farmer:

If the heat gets real bad, we use our few shaded paddocks, putting the cows there for a few hours in the middle of the day. We try to use these paddocks sparingly to avoid creating mud pits (Mroczenski et al., 2009).

In 2013, when *Graze* next ran the heat stress feature, three of the five farmers discussed shade management as a primary strategy for dealing with heat stress in their columns, and two of those responses devoted several paragraphs to describing how they manage the use of their shaded paddocks. Here is the final paragraph from one of those responses:

We re-fenced a few of the milk cow areas last year to get more trees in some paddocks. We use those paddocks in the day and then go to the shadeless paddocks at night. There are times when if we see a hot spell being forecast, we’ll alter the rotation if we can to make sure the cows have the shade paddocks in the day. If the timing for that doesn’t work and it’s too hot for the cows, we’ll bring them in the barn in the afternoon until they can go back out. We have been thinking of planting some trees in all the paddocks so that in the future everyone can just stay in their paddocks (O’Neill et al., 2013).

In the 2017 *Graze* heat stress feature, all five farmer columnists discussed using shaded paddocks to manage heat stress, and four of those responses listed access to tree shade as a primary strategy. Those four farmer-advisors each devoted several paragraphs to describing how they manage the use of their shade paddocks, including reserving shaded paddocks for hot weather, timing access to shade for daytime and access to unshaded pastures at night, and need for frequent rotation (Sheffer et al., 2017).

After using trees for shade, the most common positive mention of trees in grazing systems was to provide shelter in winter. Often, articles also mentioned trees and/or shrubs as causing problems (e.g., excess manure accumulation, shelter for predators, or damage to fences) or as something to remove in order to create new pasture. Figure 4 groups search term appearances from 2009 to 2013 and from 2014 to mid-2019, not including the articles about managing heat stress or the articles about silvopasture. Even excluding the articles

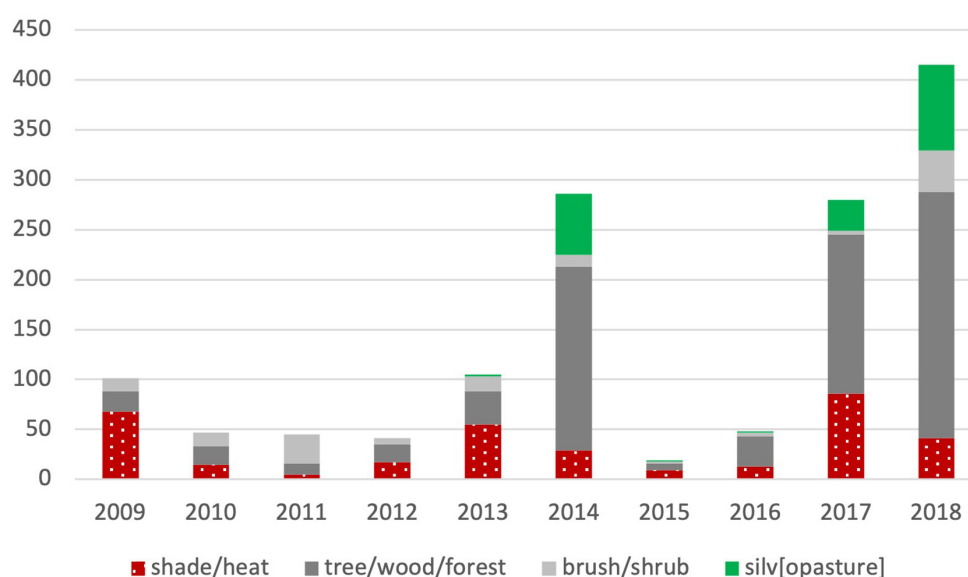


FIGURE 3

Occurrence of search terms in *Graze* related to integrating management of trees, pasture, and livestock. 2009, 2013, and 2017 had special features on managing heat stress.

on silvopasture, trees are more often characterized as an asset to the grazing system after 2013 than before. The reporting on silvopasture and the role of trees in grazing systems reflects increased interest in the practice at the same time that it transmits knowledge.

4. Discussion

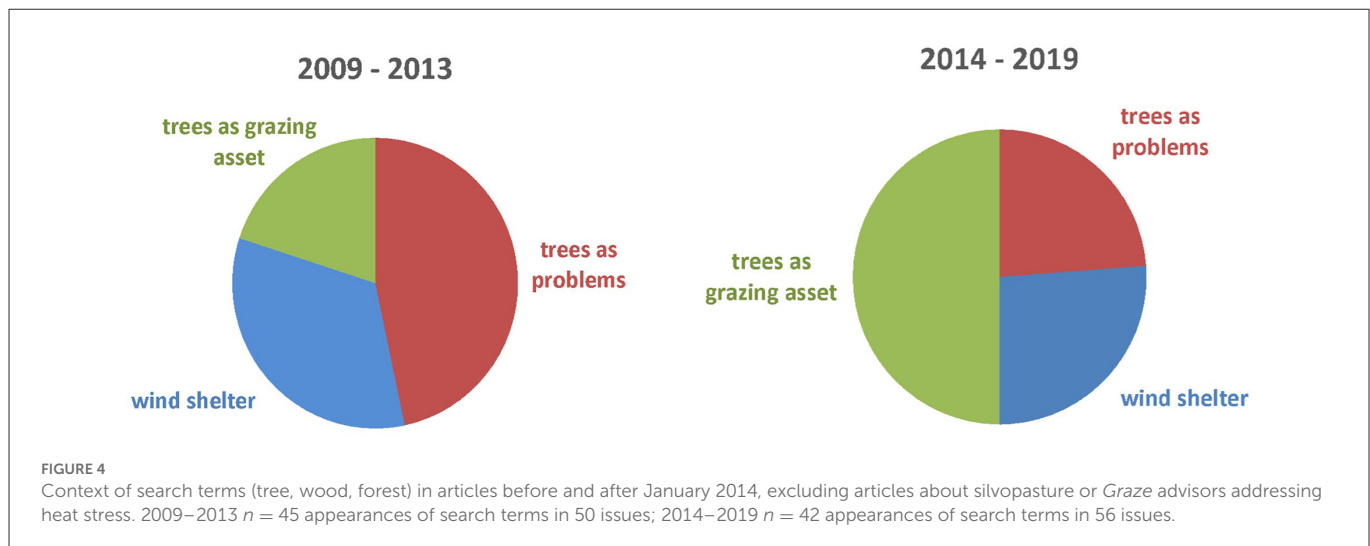
The resource professionals interviewed for this study agreed that conversion of grazed woodlands and some annual row crop fields to silvopasture would likely improve environmental outcomes for those sites (see also Brantly, 2014). One important barrier to adoption of silvopasture is that the majority of livestock farms do not practice rotational stocking, a necessary management tool for silvopasture in the Midwest. As Figure 1 shows, in 2017 only 6,786 farms (20% of the 34,400 farms with pasture) reported practicing rotational grazing in Wisconsin, and only 1,120 farms reported practicing any kind of agroforestry (including but not limited to silvopasture). Nationally 265,162 farms (21% of the 1,236,980 farms with pasture) reported practicing rotational grazing, and only 30,853 practiced any kind of agroforestry. Both in Wisconsin and regionally, farms that already practice rotational stocking constitute the likely pool of potential silvopasture adopters in the near term. Our findings describe how silvopasture is perceived by these potential adopters and identify some of the contexts fostering and limiting the application of silvopasture in this region.

In the absence of a robust history of silvopasture research in this region, those farmers who want to implement silvopasture must act simultaneously as managers and informal researchers, observing conditions on the farm and results of previous management and adjusting their actions accordingly. To support farmers in silvopasture adoption, agricultural researchers and advisors can facilitate farmer-to-farmer knowledge exchange and help identify underlying agroecological principles that guide, but do not dictate,

management (Röling and Jiggins, 1998; Poncet et al., 2010; Lyon et al., 2011). In northwestern Wisconsin, where several agricultural advisors embraced that role and explicitly invited knowledge exchange about silvopasture, we observed good communication among farmers about their experience and a clear increase in support for the practice in the grazing community. In southwestern Wisconsin, which also had an active grazing network but lacked an embedded facilitator of silvopasture knowledge exchange, farmers remained interested in silvopasture but cautious about its challenges and feasibility.

Our interviews reflect the inherent complexity of practicing silvopasture, as well as a dearth of regional research. Many researchers have observed that complex agroecological innovations require a shift from a technology-transfer paradigm of advisors delivering prescriptive direction to a systems-based paradigm of advisors facilitating farmer-led innovation and knowledge exchange (Röling, 2009; Lyon et al., 2011; Provenza et al., 2013; Blesh and Wolf, 2014; Ingram, 2015). When farmers and resource professionals in our study emphasized the need for local research and demonstration, they were implicitly recognizing limits to geographic scalability and the reality that a practice that is sustainable in one location may have different impacts when transferred to other biophysical and socioeconomic settings (Wigboldus et al., 2016).

Individual knowledge and social support (e.g., an active community of practice) are important, but contextual factors (e.g., a local market for pulp-grade wood) also factor crucially into the viability of the innovation (Loorbach et al., 2017). This dynamic, wherein grassroots-level actors' knowledge, agency, and coordination are constrained or supported by contextual factors, is often analyzed in sustainability literature with what is called a multilevel perspective (Geels, 2002, 2011; Klerkx et al., 2010; Elzen et al., 2011; Ingram, 2015; Wigboldus et al., 2016). In our case, a multilevel perspective offers a heuristic for how contextual factors (including markets, research and extension practices, tax policy and agency support,



cost and availability of labor, and other land uses), interact with individual knowledge and social support to influence the viability of silvopasture. For instance, in northwestern Wisconsin, the grazing network and its embedded facilitators of silvopasture knowledge-exchange, as well as the pulp market, were important factors in how the viability of silvopasture was perceived compared to southwestern Wisconsin.

Silvopasture, like all agroforestry practices, brings an added temporal challenge. Farmers managing forages and livestock on a 1 to 3-year basis for short term revenue must simultaneously manage for trees with a growth period from multiple decades to over a century. The uncertainty of long-term outcomes in silvopasture poses challenges for farmers and researchers (Arbuckle, 2009). We suspect that this uncertainty helps explain why most agricultural advisors still do not promote silvopasture, even though the taboo around integrating livestock with trees weakened over the course of the study. Methodologies to manage under conditions of uncertainty in long-lived complex systems, such as adaptive resource management, are well developed in forestry, grazing, and conservation literatures (Gregory et al., 2006; Teague et al., 2013). Despite its limitations (Gregory et al., 2006; Doremus, 2011; Rissman and Wardropper, 2021), adaptive management may offer a useful framework for resource professionals and farmers to develop working silvopasture systems in novel environments such as the mixed hardwoods of Wisconsin. Participatory research approaches offer additional models for combining place-based and long-term farmer insights with academic research to address complex agroecosystem management challenges (Hoffmann et al., 2007; Cerf, 2011; Snapp et al., 2019). Grazing networks, with their history of peer-to-peer knowledge exchange and their promotion of adaptive rather than prescriptive management, offer an appropriate starting point for co-creation of silvopasture knowledge in this context of complexity and limited local research (Paine et al., 2000; Lyon et al., 2011; Nelson et al., 2014; DeDecker et al., 2022).

Confusion around financial assistance and property tax policy added another barrier to silvopasture adoption during our study. At the end of our study period, the Natural Resources Conservation Service in both Wisconsin and Minnesota began working on clarifying state standards for financial assistance for silvopasture

establishment and management, and this work continues as of this writing (Hart, 2019; Braun, 2022). These policy efforts represent a significant step forward in making silvopasture accessible for farmers, and also reflect the change in attitudes toward silvopasture that has occurred in the region.

5. Conclusion

Silvopasture in the US Midwest remains an uncertain proposition for most farmers and natural resource professionals, due in part to the history of woodland degradation by poor livestock management, and in part to the inherent complexity of the practice. Whereas, prior to 2014 there was little research and education about silvopasture in the Midwest, more marked interest in silvopasture emerged and persisted in and around Wisconsin from 2014 to 2019. Of the two regions we studied, the enthusiasm, knowledge, and practice of silvopasture grew in northwest Wisconsin, which coincided with the development of a community of practice that included farmers and agricultural advisors cooperating in a favorable set of landscape and market circumstances. In contrast, farmers remained more cautious about the practicality of silvopasture in southwest Wisconsin where markets were less favorable and farmer adopters and professional advocates did not coalesce into a silvopasture community of practice.

We also observed changes in attitudes among agricultural advisors and foresters: early in the study period most of these resource professionals did not discuss silvopasture in public, but later in the study period some agricultural advisors gave silvopasture advice, and some foresters' attitudes reflected increasing openness to silvopasture in certain situations. Overall, the findings from this study suggest that (1) contextual factors such as climate, landscape attributes, markets, and existing land uses influence stakeholders' attitudes about silvopasture, and (2) positive attitudes and knowledge about silvopasture can be cultivated in local communities of practice that exchange information about management strategies appropriate to the complex, long-term, and context-dependent nature of the practice.

The diversity of potential silvopasture composition and design options in this region coupled with the time required

to study trees means that standard agricultural research and extension approaches are insufficient to support farmers practicing silvopasture. Rather, farmers, resource professionals, and researchers need to collaborate over the long term. This process of collaboration can begin using general principles derived from silvopasture, forestry, and grazing research and experience, but it must adaptively adjust those principles based both on formal measurements and on farmer observations. Because other continuous living cover systems also add temporal and species complexity, similar collaborative and adaptive approaches may be needed across the board to transform our agricultural monocultures to sustainable agroecosystems.

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

Ethics statement

The studies involving human participants were reviewed and approved by University of Wisconsin-Madison Institutional Review Board. Written informed consent for participation was not required for this study in accordance with the national legislation and the institutional requirements.

Author contributions

DM and KK conducted the data collection and analysis and drafted the article. All authors contributed

to editing, finalizing the article, conception, and design of the study.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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References

- Abbott, R. W. (1954). Woodland pasturing—are we going fast enough? *J. Forestry* 48, 431–433.
- Ahlgren, H. L., Wall, M. L., Muckenhirn, R. J., and Sund, J. M. (1946). Yields of forage from woodland pastures on sloping land in Southern Wisconsin. *J. Forestry* 44, 709–711.
- Arbuckle, J. G. (2009). “Cattle and trees don't mix!?: Competing agri-environmental paradigms and silvopasture agroforestry in the missouri ozarks,” in *Farming with Grass: Achieving Sustainable Mixed Agricultural Landscapes*, ed A. J. Franzluebbers (Ankeny, IA: Soil and Water Conservation Society), 116–133. Available online at: <https://www.swcs.org/resources/publications/farming-with-grass-online/>
- Ares, A., St Louis, D., and Brauer, D. (2003). Trends in tree growth and understory yield in silvopastoral practices with southern pines. *Agrofor. Syst.* 59, 27–33. doi: 10.1023/A:1026132918914
- Baah-Acheamfour, M., Carlyle, C. N., Bork, E. W., and Chang, S. X. (2014). Trees increase soil carbon and its stability in three agroforestry systems in Central Alberta, Canada. *For. Ecol. Manage.* 328, 131–139. doi: 10.1016/j.foreco.2014.05.031
- Baah-Acheamfour, M., Carlyle, C. N., Lim, S., Bork, E. W., and Chang, S. X. (2016). “Forest and grassland cover types reduce net greenhouse gas emissions from agricultural soils. *Sci. Total Environ.* 571, 1115–1127. doi: 10.1016/j.scitotenv.2016.07.106
- Blanco, J., Sourdril, A., Deconchat, M., Ladet, S., and Andrieu, E. (2019). Social drivers of rural forest dynamics: a multi-scale approach combining ethnography, geomatic and mental model analysis. *Landsc. Urban. Plan.* 188, 132–142. doi: 10.1016/j.landurbplan.2018.02.005
- Blesh, J., and Wolf, S. A. (2014). Transitions to agroecological farming systems in the mississippi river basin: toward an integrated socioecological analysis. *Agric. Human Values* 31, 621–635. doi: 10.1007/s10460-014-9517-3
- Brantly, S. (2014). *Forest Grazing, Silvopasture, and Turning Livestock into the Woods. Agroforestry Notes. National Agroforestry Center*. Available online at: <https://www.fs.usda.gov/nac/assets/documents/agroforestrynotes/an46si09.pdf> (accessed January 22, 2023).
- Braun, A. (2022). *Forestry Scenario Updates*. NRCS Wildlife/Forestry Committee Meeting (Madison, WI).
- Byrne, D., and Ragin, C. C. (2009). *Case-Based Methods*. Thousand Oaks, CA: SAGE Publications.
- Cavadini, J. (2022). *University of Wisconsin-Madison Extension Grazing Specialist, Personal Communication*.
- Cerf, M. (2011). Is participatory research a scientific practice? *J. Rural Stud.* 27, 414–418. doi: 10.1016/j.jrurstud.2011.10.004
- Clason, T. R. (1998). Silvopastoral practices sustain timber and forage production in commercial loblolly pine plantations of Northwest Louisiana, USA. *Agroforestry Syst.* 44, 293–303. doi: 10.1023/A:1006267114962
- Cubbage, F., Balmelli, G., Bussoni, A., Noellemeyer, E., Pachas, A. N., Fassola, H., et al. (2012). Comparing silvopastoral systems and prospects in eight regions of the world. *Agroforestry Syst.* 86, 303–314. doi: 10.1007/s10457-012-9482-z
- Dambach, C. A. (1944). A ten-year ecological study of adjoining grazed and ungrazed woodlands in Northeastern Ohio. *Ecol. Monogr.* 14, 255–270. doi: 10.2307/1948443
- Dana, S. T., and Fairfax, S. K. (1980). *Forest and Range Policy. Its Development in the United States*. 2nd ed. New York, NY: McGraw-Hill Book Company.
- DeDecker, J., Malone, T., Snapp, S., Thelen, M., Anderson, E., Tollini, C., et al. (2022). The relationship between farmer demographics, social identity and tillage behavior: evidence from Michigan soybean producers. *J. Rural Stud.* 89, 378–386. doi: 10.1016/j.jrurstud.2022.01.001

- Demchik, M., Thompson, D. M., and Schossow, R. (2005). "Forage Yield and Quality under Oak Crop Tree Management," in *Moving Agroforestry into the Mainstream: The 9th North American Agroforestry Conference Proceedings* (St Paul, MN: University of Minnesota).
- Doremus, H. (2011). Adaptive management as an information problem. *North Carol. Law Rev.* 89, 1455–1498. Available online at: https://heinonline-org.ezproxy.library.wisc.edu/HOL/Page?collection=journals&handle=hein.journals/nclr89&id=1466&men_tab=srchresults
- Elzen, B., Geels, F. W., Leeuwis, C., and Van Mierlo, B. (2011). Normative contestation in transitions in the making: animal welfare concerns and system innovation in pig husbandry. *Res. Policy* 40, 263–275. doi: 10.1016/j.respol.2010.09.018
- Ford, M. M., Zamora, D. S., Current, D., Magner, J., Wyatt, G., Walter, W. D., et al. (2019). Impact of managed woodland grazing on forage quantity, quality and livestock performance: the potential for silvopasture in Central Minnesota, USA. *Agroforestry Syst.* 3, 67–79. doi: 10.1007/s10457-017-0098-1
- Frey, G. E., Fassola, H. E., Nahuel Pachas, A., Colcombet, L., Lacorte, S. M., Pérez, O., et al. (2012). Perceptions of silvopasture systems among adopters in Northeast Argentina. *Agric. Syst.* 105, 21–32. doi: 10.1016/j.agry.2011.09.001
- Galleguillos, N., Keeley, K., and Ventura, S. (2018). Assessment of woodland grazing in Southwest Wisconsin. *Agric. Ecosyst. Environ.* 260, 1–10. doi: 10.1016/j.agee.2018.03.012
- Garrett, H. E., Kerley, K. P., Ladyman, K. P., Walter, W. D., Godsey, L. D., Van Sambeek, J. W., et al. (2004). *Hardwood Silvopasture Management in North America*. New Vistas in Agroforestry. Springer, 21–33.
- Geels, F. W. (2002). Technological transitions as evolutionary reconfiguration processes: a multi-level perspective and a case-study. *Res. Policy* 31, 1257–1274. doi: 10.1016/S0048-7333(02)00062-8
- Geels, F. W. (2011). The multi-level perspective on sustainability transitions: responses to seven criticisms. *Environ. Innov. Soc. Trans.* 1, 24–40. doi: 10.1016/j.eist.2011.02.002
- Grado, S. C., and Husak, A. L. (2004). Economic analyses of a sustainable agroforestry system in the Southeastern United States. *Valu. Agroforestry Syst.* 39–57. doi: 10.1007/1-4020-2413-4_3
- Green Lands Blue Waters (n.d.). *Continuous Living Cover*. Green Lands Blue Waters. Available online at: <https://greenlandsbluewater.org/continuous-living-cover/#why-clc> (accessed December 20, 2022).
- Gregory, R., Ohlson, D., and Arvai, J. (2006). Deconstructing adaptive management: criteria for applications to environmental management. *Ecol. Appl.* 16, 2411–2425. doi: 10.1890/1051-0761(2006)016[2411:DAMCFA]2.0.CO;2
- Guise, C. H. (1950). *The Management of Farm Woodlands*. 2nd ed. American Forestry. New York, NY: McGraw-Hill.
- Hart, A. (2019). Personal communication.
- Hawken, P. (2017). *Drawdown: The Most Comprehensive Plan Ever Proposed to Reverse Global Warming*. Penguin. Available online at: <https://www.drawdown.org/solutions-summary-by-rank> (accessed March 2022).
- Hoffmann, V., Probst, K., and Christinck, A. (2007). Farmers and researchers: how can collaborative advantages be created in participatory research and technology development? *Agric. Human Values* 24, 355–368. doi: 10.1007/s10460-007-9072-2
- Howlett, D. S., Moreno, G., Mosquera Losada, M. R., Nair, P. K. R., and Nair, V. D. (2011). Soil carbon storage as influenced by tree cover in the dehesa cork oak silvopasture of Central-Western Spain. *J. Environ. Monit.* 13, 1897. doi: 10.1039/c1em10059a
- Hsieh, H., and Shannon, S. E. (2005). Three approaches to qualitative content analysis. *Qual. Health Res.* 15, 1277–1288. doi: 10.1177/1049732305276687
- Ingram, J. (2015). Framing niche-regime linkage as adaptation: an analysis of learning and innovation networks for sustainable agriculture across Europe. *J. Rural Stud.* 40, 59–75. doi: 10.1016/j.jrurstud.2015.06.003
- Keeley, K. (2014). *Thoughts on the Back Forty: Diverse Perspectives on Farm Woods Drawn from in-Depth Interviews in the Driftless Area of Wisconsin*. Madison, WI: University of Wisconsin-Madison.
- Klerkx, L., Aarts, N., and Leeuwis, C. (2010). Adaptive management in agricultural innovation systems: the interactions between innovation networks and their environment. *Agric. Syst.* 103, 390–400. doi: 10.1016/j.agry.2010.03.012
- Loorbach, D., Frantzeskaki, N., and Avelino, F. (2017). Sustainability transitions research: transforming science and practice for societal change. *Annu. Rev. Environ. Resour.* 42, 599–626. doi: 10.1146/annurev-environ-102014-021340
- Lyon, A., Bell, M. M., Gratton, C., and Jackson, R. (2011). Farming without a recipe: wisconsin graziers and new directions for agricultural science. *J. Rural Stud.* 27, 384–393. doi: 10.1016/j.jrurstud.2011.04.002
- Mayerfeld, D., Rickenbach, M., and Rissman, A. (2016). Overcoming history: attitudes of resource professionals and farmers toward silvopasture in Southwest Wisconsin. *Agroforestry Syst.* 90, 723–736. doi: 10.1007/s10457-016-9954-7
- Montagnini, F., and Nair, P. K. R. (2004). Carbon sequestration: an underexploited environmental benefit of agroforestry systems. *Agroforestry Syst.* 61, 281–295. doi: 10.1007/978-94-017-2424-1_20
- Morgan, D., Fellows, C., and Guevara, H. (2008). "Emergent approaches to focus group research," in *Handbook of Emergent Methods*. New York, NY: The Guilford Press, 189–205.
- Mroczeni, M., Benrud, R., Marshall, L., Mapstone, P., and Rickard, B. (2009). Graze advisors: how do you manage heat stress? *Graze*. Available online at: <https://www.grazeonline.com>
- Nair, V. D., Haile, S. G., Michel, G., and Nair, P. K. R. (2007). Environmental quality improvement of agricultural lands through silvopasture in Southeastern United States. *Sci. Agric.* 64, 513–519. doi: 10.1590/S0103-90162007000500009
- Nelson, K. C., Brummel, R. F., Jordan, N., and Manson, S. (2014). Social networks in complex human and natural systems: the case of rotational grazing, weak ties, and eastern US dairy landscapes. *Agric. Human Values* 31, 245–259. doi: 10.1007/s10460-013-9462-6
- O'Neill, K., Steffen, A., Martin, M., Weaver, S., and Langmeier, J. (2013). Graze advisors: how do you deal with the heat? *Graze*. Available online at: <https://www.grazeonline.com>
- Orefice, J., Carroll, J., Conroy, D., and Ketner, L. (2017a). Silvopasture practices and perspectives in the Northeastern United States. *Agroforestry Syst.* 91, 149–160. doi: 10.1007/s10457-016-9916-0
- Orefice, J., Smith, R. G., Carroll, J., Asbjornsen, H., and Howard, T. (2019). Forage productivity and profitability in newly-established open pasture, silvopasture, and thinned forest production systems. *Agroforestry Syst.* 93, 51–65. doi: 10.1007/s10457-016-0052-7
- Orefice, J., Smith, R. G., Carroll, J., Asbjornsen, H., and Kelting, D. (2017b). Soil and understory plant dynamics during conversion of forest to silvopasture, open pasture, and woodland. *Agroforestry Syst.* 91, 729–739. doi: 10.1007/s10457-016-0040-y
- Paine, L. K., Klemme, R. M., Undersander, D. J., and Welsh, M. (2000). Wisconsin's grazing networks: history, structure, and function. *J. Nat. Res. Life Sci. Educ.* 29, 60–67. doi: 10.2134/jnlrse.2000.0060
- Patel-Weyand, T., Bentrup, G., and Schoeneberger, M. M. (2017). *Agroforestry: Enhancing Resiliency in U.S. Agricultural Landscapes Under Changing Conditions*. WO-GTR-96. Washington, DC: U.S. Department of Agriculture, Forest Service.
- Poncet, J., Kuper, M., and Chiche, J. (2010). Wandering off the paths of planned innovation: the role of formal and informal intermediaries in a large-scale irrigation scheme in Morocco. *Agric. Syst.* 103, 171–179. doi: 10.1016/j.agry.2009.12.004
- Porter, S., and Voskuil, A. (2022). *Double Trouble: Wisconsin's Land and Water Are Inundated With Pollution From Animal Manure and Excess Farm Fertilizer*. Environmental Working Group and Midwest Environmental Advocates. Available online at: <https://www.ewg.org/research/double-trouble-wisconsin-land-and-water-are-inundated-pollution-animal-manure-and-excess> (accessed April 2022).
- Provenza, F., Pringle, H., Revell, D., Bray, N., Hines, C., Teague, R., et al. (2013). Complex creative systems. *Rangelands* 35, 6–13. doi: 10.2111/RANGELANDS-D-13-00013.1
- Rissman, A. R., and Wardropper, C. B. (2021). Adapting conservation policy and administration to nonstationary conditions. *Soc. Nat. Resour.* 34, 524–537. doi: 10.1080/08941920.2020.1799127
- Röling, N. G. (2009). Pathways for impact: scientists' different perspectives on agricultural innovation. *Int. J. Agric. Sustain.* 7, 83–94. doi: 10.3763/ijas.2009.0043
- Röling, N. G., and Jiggins, J. (1998). "The Ecological Knowledge System," in *Facilitating Sustainable Agriculture: Participatory Learning and Adaptive Management in Times of Environmental Uncertainty*. Cambridge, UK: Cambridge University Press.
- Rubino, R. (1996). "Forest Grazing: Reflections on Its Evolution and the Future," in *Western European Silvopastoral Systems*, eds M. Étienne. Paris: INRA (Institut National de la Recherche Agronomique), 157–165.
- Sheffer, E., Erb, D., Witmer, P., Haugen, O., and Cooper, A. (2017). How do you deal with heat stress? *Graze*. 24. Available online at: <https://www.grazeonline.com>
- Shrestha, R. K., Alavalapati, J. R. R., and Kalmbacher, R. S. (2004). Exploring the potential for silvopasture adoption in South-Central Florida: an application of SWOT-AHP method. *Agric. Syst.* 81, 185–199. doi: 10.1016/j.agry.2003.09.004
- Snapp, S. S., DeDecker, J., and Davis, A. S. (2019). Farmer participatory research advances sustainable agriculture: lessons from Michigan and Malawi. *Agron. J.* 111, 2681–2691. doi: 10.2134/agronj2018.12.0769
- Stutzman, E., Barlow, R. J., Morse, W., Monks, D., and Teeter, L. (2019). Targeting educational needs based on natural resource professionals' familiarity, learning, and perceptions of silvopasture in the Southeastern U.S. *Agroforestry Syst.* 93, 345–353. doi: 10.1007/s10457-018-0260-4
- Teague, R., Provenza, F., Kreuter, U., Steffens, T., and Barnes, M. (2013). Multi-paddock grazing on rangelands: why the perceptual dichotomy between research results and rancher experience? *J. Environ. Manage.* 128, 699–717. doi: 10.1016/j.jenvman.2013.05.064
- USDA-NASS (2019a). *2017 Census of Agriculture, Table 8. Farms, Land in Farms, Value of Land and Buildings, and Land Use: 2017 and 2012*. US Department of Agriculture. Available online at: https://www.nass.usda.gov/Publications/AgCensus/2017/Full_Report/Volume_1,_Chapter_2_US_State_Level/ (accessed November 2022).

- USDA-NASS (2019b). *2017 Census of Agriculture, Table 43. Selected Practices: 2017 and 2012. US Department of Agriculture*. Available online at: https://www.nass.usda.gov/Publications/AgCensus/2017/Full_Report/Volume_1,_Chapter_2_US_State_Level/ (accessed November 2022).
- Whitt, C., and Wallander, S. (2022). *Rotational Grazing Adoption by Cow-Calf Operations. USDA ERS EIB 243*. Available online at: <https://www.ers.usda.gov/webdocs/publications/105077/eib-243.pdf?v=4030.5> (accessed December 2022).
- Wigboldus, S., Klerkx, L., Leeuwis, C., Schut, M., Muilerman, S., and Jochemsen, H. (2016). Systemic perspectives on scaling agricultural innovations. A review. *Agron. Sustain. Dev.* 36, 46. doi: 10.1007/s13593-016-0380-z
- Wilkens, P., Munsell, J. F., Fike, J. H., Pent, G. J., Frey, G. E., Addlestone, B. J., et al. (2022). Thinning forests or planting fields? Producer preferences for establishing silvopasture. *Agroforest. Syst.* 96, 553–564. doi: 10.1007/s10457-021-00665-z
- Wisconsin Department of Revenue, Division of Revenue (2022). *2022 Agricultural Assessment Guide for Wisconsin Property Owners*. Available online at: <https://www.revenue.wi.gov/DOR%20Publications/pb061.pdf> (accessed April 29, 2022).
- Yin, R. K. (2009). *Case Study Research Design and Methods*. Vol. 5. 4th ed. Applied Social Research Methods. Los Angeles: SAGE.



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EDITED BY

Christopher J. Whelan,
Moffitt Cancer Center, United States

REVIEWED BY

Lisa Schulte Moore,
Iowa State University, United States
Elizabeth M. Bach,
The Nature Conservancy, United States

*CORRESPONDENCE

Adena R. Rissman
✉ adena.rissman@wisc.edu

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Grassland and managed grazing policy review

Adena R. Rissman^{1*}, Ana Fochesatto², Erin B. Lowe¹, Yu Lu¹,
Regina M. Hirsch³ and Randall D. Jackson⁴

¹Department of Forest and Wildlife Ecology, University of Wisconsin-Madison, Madison, WI, United States, ²Nelson Institute for Environmental Studies, University of Wisconsin-Madison, Madison, WI, United States, ³Center for Integrated Agricultural Systems, University of Wisconsin-Madison, Madison, WI, United States, ⁴Department of Agronomy, University of Wisconsin-Madison, Madison, WI, United States

Perennial grasslands, including prairie and pasture, have declined with tremendous environmental and social costs. This decline reflects unequal policy support for grasslands and managed grazing compared to row crops. To create a resource for community partners and decision-makers, we reviewed and analyzed the policy tools and implementation capacity that supports and constrains grasslands and managed grazing in the U.S. Upper Midwest. Risk reduction subsidies for corn and soybeans far outpace the support for pasture. Some states lost their statewide grazing specialist when the federal Grazing Lands Conservation Initiative lapsed. The United States Department of Agriculture, Natural Resources Conservation Service support for lands with prescribed grazing practices declined after 2005 but remained relatively steady 2010–2020. These results reveal the policy disadvantage for grasslands and managed grazing in comparison with row crop agriculture for milk and meat production. Grassland and grazing policies have an important nexus with water quality, biodiversity, carbon and outdoor recreation policy. Socially just transitions to well-managed, grazed grasslands require equity-oriented interventions that support community needs. We synthesized recommendations for national and state policy that farmers and other grazing professionals assert would support perennial grasslands and grazing, including changes in insurance, conservation programs, supply chains, land access, and fair labor. These policies would provide critical support for grass-based agriculture and prairies that we hope will help build soil, retain nutrients, reduce flooding and enhance biodiversity while providing healthy food, jobs, and communities.

KEYWORDS

managed grazing, continuous living cover, perennial cover, policy and governance, systems change, grasslands, prairies, pasture

Introduction

Perennial grasslands have declined precipitously worldwide because they are planted to row-crops or converted to other land uses that degrade ecosystems and human cultural and economic relationships (Kwon et al., 2016; Lark et al., 2020; Winkler et al., 2021). Government, corporate, and non-governmental policies have contributed to grassland degradation, yet other policies aim to protect and restore grasslands. Policies are important aspects of grassland and agricultural governance because they provide incentives, regulations, market structures and standards, and assistance that shape farmer and land manager decisions about grasslands. Managed well, grasslands can enhance farmer profitability and quality of life, rural communities, food sovereignty, water quality and

flood reduction, wildlife, pollinator and plant habitat, and soil carbon (Rui et al., 2022; Sanford et al., 2022; Wepking et al., 2022). Focusing on the Upper Midwest of the United States, this policy review describes recent trends in policies, programs, and capacities that impact grasslands and provides recommendations for policy change to enhance grasslands and managed grazing. We include pasture, prairie, and savanna within the scope of this review.

Across North America, grasslands emerged as glaciers retreated (Strömberg, 2002). Indigenous communities actively managed grasslands with fire to increase food supply, manage grazing game (Fuhlendorf and Engle, 2004) and increase the visibility of enemies, promoting higher grassland productivity and more input of carbon and nutrients to soils (Frank and McNaughton, 1993). In the 1800s, the U.S. government's genocidal campaign against Indigenous communities included the destruction of bison (Hubbard, 2014), a keystone species for grassland ecosystems and Indigenous food systems and culture (Isenberg, 2000). Euro-American settlers replaced bison with cattle and row crops, parcelizing land into small and often insufficient homesteads. Overgrazing and plowing caused the degradation of grasslands (Holleman, 2017). Agricultural intensification during the Green Revolution drove more conversion from pastures to row crops. Meat and dairy markets have become highly consolidated through the increasing market share of international corporations which continues today (Lark et al., 2020), part of a major shift in global agricultural markets (Belk et al., 2014). These transitions track different ideas of production, reflecting different understandings of the value of intensive and extensive agriculture and the political economy of maximizing agricultural yields. Grassland succession into shrubs and forests along with urban and exurban housing developments have also reduced grassland area (Rajib et al., 2016).

In the Upper Midwest in particular, policies have caused grasslands to decline (Figure 1). Less than 1% of tallgrass prairie dominated by warm-season grasses remains (Samson and Knopf, 1994). While livestock were primarily raised on grass early in the 20th Century, policies in the latter half of the century incentivized farmers to transition the land to intensive production of corn and soybeans. The proliferation of subsidized corn and soybeans for animal feed in turn encouraged farmers to move cattle from pastures to confined barns and feedlots, accelerating the conversion of pasture to row crop agriculture (Gillon et al., 2016). Controls on crop supply were removed and farmers were encouraged to plant "fencerow-to-fencerow" and consolidate their operations. Corn and soybean subsidies and crop insurance expanded through U.S. Farm Bills (Imhoff and Badaracco, 2019), although subsidies were removed after international challenges through the World Trade Organization, crop insurance expanded (Schneppf, 2021). In an effort to improve domestic energy supply and provide governmental support for corn, a federal ethanol mandate required gasoline to include a percentage of renewable fuel including cellulosic ethanol from corn stover, incentivizing conversions of grassland to corn (Lark, 2020).

Rowcrops without livestock draw upon soil resources without making organic deposits sufficient to replenish reserves. However, overapplication of livestock nutrients from manure and urine

results in high soil nutrient levels and runoff that pollutes ground and surface waters. In contrast, well-managed grazed perennial grasslands can produce human food while making continuous but not excessive nutrient deposits into soil (Jackson, 2020). When well-managed, grazing has the capacity to regenerate soil organic matter, provide milk and meat, improve water quality, help stabilize climate, reduce flooding, and enhance biodiversity (Franzluubbers et al., 2012; Bengtsson et al., 2019). The grassland plants in these systems shunt much of the carbon they fix from the atmosphere into belowground tissues, creating a reserve of carbohydrates and nutrients that increases over-winter survival and regrowth after defoliation. Grassland roots and symbiotic fungi are continuously turning over and exuding carbon into the soil, which contributes to soil organic matter accumulation (Liang et al., 2016; Zhu et al., 2020), enhancing soil health. Carbon storage in grassland soils has the potential to contribute to climate mitigation, although the estimates from carbon accounting and life cycle analysis vary (Garnett et al., 2017; Mayerfeld, 2023).

Grassland loss has significantly degraded biodiversity and water quality. Grassland birds, pollinators, and monarch butterflies have declined dramatically with the loss of habitat and use of pesticides on row crops (Cox, 1991; Herkert et al., 1996; Ribic and Sample, 2001; Goulson et al., 2015; Boyle et al., 2019). Grazing and other grassland management approaches can help maintain grassland and savanna habitat, along with timber harvests, prescribed fire, mowing, and herbicide applications (Wisconsin Department of Natural Resources, 2016). Managed grazing can promote biodiversity and wildlife habitat, depending on the timing and intensity of grazing (Hardy et al., 2020). The Upper Midwest contributes significantly to the runoff of sediment containing nitrogen and phosphorus that expand the dead zone in the Gulf of Mexico (Rabalais et al., 2002). Climate change impacts include an increase in extreme storm events which have caused an increase in flooding (Bendorf et al., 2021), exacerbated by greater row crop production.

Grazing and grasslands can support farmer wellbeing, livelihoods, and vibrant rural communities with new and diverse farmers and grassland enterprises (Bardgett et al., 2021). Consolidation in agriculture has led many farmers and ranchers to lose their farms and increased rural depopulation. Grazing livestock on grassland offers a relatively profitable and low-cost opportunity for farmers whose access to high quality forage reduces their feed and manure management costs (Hanson et al., 1998; Soriano et al., 2001; Foltz and Lang, 2005). Demand for grass-fed products is increasing, creating new market opportunities. While beef and dairy receive most of the focus for managed grazing, smaller animals such as sheep, goats, and poultry, can offer an easier entry-point for new farmers because they require less up-front capital and infrastructure, reproduce more quickly, and are easier to manage. Additionally, these animals are culturally important for many immigrant communities and new farmers (on goats: Lu and Miller, 2019; on chickens: Haslett-Marroquin and Andreassen, 2017). Socially just transitions to well-managed grazed perennial grasslands require equity-oriented interventions that support the needs of all communities (Lowe and Fochesatto, 2023).

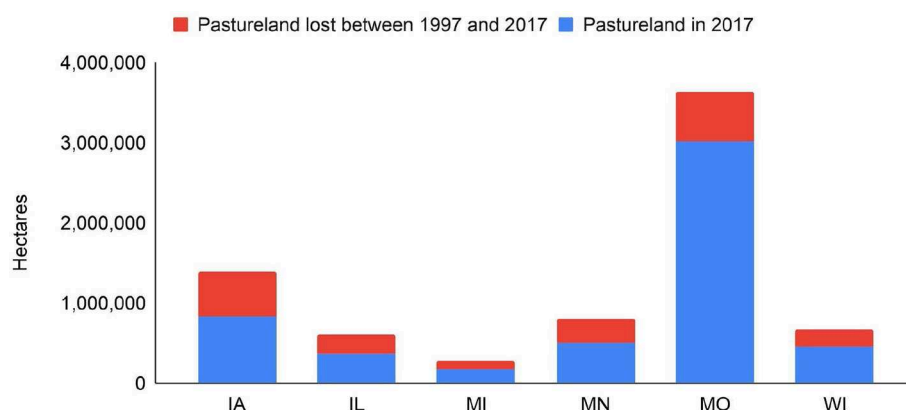


FIGURE 1

Non-woodland pasture declined across the Upper Midwest between 1997 and 2017. Data source: Agricultural Census 1997 and 2017 (USDA National Agricultural Statistics Service, 1997, 2017a).

The loss of grasslands has taken with it many cultural practices and social benefits that communities across the region are now working to recover. Tribal nations are actively re-establishing bison herds on the land and reconnecting tribal members with this ancestral practice and food source (Zontek, 2007). For example, the Intertribal Buffalo Council coordinates the transfer of surplus bison from national parks to tribal lands saying “to reestablish healthy buffalo populations on tribal lands is to reestablish hope for the Indian people” (InterTribal Buffalo Council, 2019). These initiatives contribute to seeing food as medicine, not just caloric content, through reaffirming ways of life and food sovereignty.

Land governance involves a multilayered system of policies and markets created and run by governments, private sector firms, and non-profit civil society organizations that influence the land management choices of individuals, families, and communities. Policies establish the rules of the game for agroecosystem management. Grassland policy is underdeveloped, especially outside of arid rangelands. Grassland and managed grazing are in need of a policy framework and policy advocacy coalition to increase grassland abundance and biodiversity and prevent further conversion to row crops and housing. As one indicator of this need, a Google Scholar search from 2022 reveals the number of records for “agricultural policy” (591,000) and “forest policy” (161,000) in comparison with grazing policy (1,800, with most focused on arid public land, not mesic private land), “grassland policy” (367), “pasture policy” (101), “prairie policy” (20), and “savanna policy” (3).

Given the need for greater attention to policies that support and constrain grassland and managed grazing, we synthesize programmatic information to review the policy landscape and draw on interview quotes for context. We then synthesize recommendations for policy change based on a literature review and extensive conversations with partners, interviewees, and workshop participants. The two objectives for this policy review are:

- 1) Review the policy tools and implementation capacity that supports and constrains grasslands, managed grazing, and prairies in the U.S. Upper Midwest.

- 2) Provide recommendations for enhancing policy support for grasslands and improved grassland governance.

Policy assessment

Policy review methods

We examined the grassland policy context in six Corn Belt and Great Lakes states of the tall grass prairie region: Illinois (IL), Iowa (IA), Michigan (MI), Minnesota (MN), Missouri (MO), and Wisconsin (WI), USA. Three of these states intersect with the United States Department of Agriculture (USDA) “Northern Crescent” region (MI, MN, WI) and four with the “Heartland” region (IA, IL, MN, MO). Iowa, Illinois, Missouri and Minnesota had substantial tallgrass prairie before European settlement (Transeau, 1935), while Wisconsin and Michigan had smaller patches of tallgrass prairie interspersed with oak savanna and hardwood forests (Cochrane and Iltis, 2000). Indigenous burning and grazing management likely expanded grassland area, reducing the size and density of forest cover (Changnon et al., 2003). This region’s land cover is dominated by agriculture, predominantly corn and soybean row crops. In 2022, corn covered 4.3 million hectares in IL, 5.1 in IA, 0.9 in MI, 3.4 in MN, 1.5 in MO, and 1.6 in WI (NASS, 2022). The central portions of MI, WI, and MN contain a grass-forest ecotone.

We identified policies relevant to grasslands and managed grazing and developed recommendations through a literature review and consultation with grazing farmers, advisors, and staff of civil society organizations, agricultural industry, and local, state, and federal government agencies as part of a larger project to promote grassland agriculture called Grassland 2.0. Policies were identified and discussed through multiple venues including Grassland 2.0 meta stakeholder meetings (regular meetings 2018–2023), Grassland 2.0 policy team (regular meetings 2019–2023), perennial policy leaders meeting (February 2021), three Just Transitions to Managed Grazing workshops (January, February, and March 2022, Lowe and Fochesatto, 2023), and a Farm Bill workshop (April 2022). We synthesized these conversations and prior literature to develop the policy categories in this manuscript.

The policies are not listed in order of priority; rather we focus first on the most common policy choices discussed by participants and end with the deeper structural drivers of land, capital, and labor. For programs expected to impact grasslands and managed grazing, we summarized publicly available data on trends in enrollment.

Programmatic information is supplemented with illustrative quotes about policies from a series of 130 semi-structured interviews (Lowe, 2022). Of the interviewees, 54% were from WI, 15% from IL, 14% from MN, 6% from MI, 5% from IA, and 2% from MO. An additional 4% of non-Midwesterners were interviewed to fill in specific gaps in expertise. All of these peoples' work intersected with agriculture in some capacity, and most worked specifically with animal agriculture. Interviews were conducted by Zoom or in-person, audio recorded and transcribed, with consent under IRB 2020-1687. Quotes from farmers and other professionals engaged in grasslands and managed grazing were selected to illustrate common perspectives on each policy tool. Job titles are accurate at the time of the interview.

We synthesized recommendation from these diverse sources. We also circulated a Wisconsin policy report and received feedback that we integrated into this manuscript's recommendations. Given the format of this policy piece, we present aggregated recommendations and not detailed coding of themes from interviews and workshops. Recommendations are not necessarily consensus perspectives, and ideas that faced the greatest criticism from participants are not included here. Drafts of this policy review and recommendations were circulated with community partners in advance of publication.

Federal subsidies, insurance, and renewable fuel standard

"Crop insurance...sucked the life out of grazing here in Illinois, because it puts a floor under what you're going to make or props prices up."—Cliff Schuette, Beef Grazier, IL

"More and more farmers are not being profitable in farming grains [but] whenever grain prices go up, we see land taken out of pasture [and] planted to corn...There really isn't...an economic motivation on transitioning away from corn and beans when we still have federal crop subsidies and crop insurance...There are no other government safety nets for grazing - nothing that compares to the subsidies given to grain farmers."—Meghan Filbert, Livestock Program Manager, Practical Farmers of Iowa & Diversified Grazier

Commodity subsidies and crop insurance

Commodity subsidies and crop insurance buffer price and yield losses for corn and soybeans, while support provided for pasture is scant. This incentivizes planting corn and soybeans despite market signals that might otherwise encourage farmers to grow different crops or pasture (Houser et al., 2020; Burchfield et al., 2022). Together, corn and soybeans have made up nearly half of this spending nationally (Schnepf, 2017). The amount of money

allocated to these programs amounts to 16% of Farm Bill spending, more than twice the amount (7%) allocated to all the other Farm Bill conservation programs discussed in this paper (USDA Economic Research Service, 2021). Because commodity subsidies and crop insurance reduce feed costs, they incentivize raising animals in confinement rather than on pasture or rangeland. Direct subsidies have been transitioned out (Figure 2). At the state level in 2016, the corn and soybean commodity and crop insurance subsidies were \$984M in IA, \$1,244M in IL, \$211M in MI, \$668M in MN, \$381M in MO, and \$327M in WI, compared with amount of the USDA Natural Resources Conservation Service's (NRCS) conservation program financial and technical assistance [including: Conservation Reserve Program (CRP), Conservation Stewardship Program (CSP), Wildlife Habitat Incentive Program (WHIP), Grassland Reserve Program (GRP), Regional Conservation Partnership Program (RCPP), Conservation Technical Assistance (CTA), and Environmental Quality Incentives Program (EQIP)] of \$0.1M in IA, \$0.08M in IL, \$0.04M in MI, \$0.14M in MN, \$0.09M in MO, and \$0.06M in WI (USDA Natural Resources Conservation Service, 2021; CSRA Science, 2022; USDA Risk Management Agency, 2022).

Pasture insurance could be provided through two programs, however adoption is very low and producers tend to find the programs unsupportive. The Pasture, Range, and Forage Program is designed to "cover replacement feed costs when a loss of forage for grazing or harvested for hay is experienced due to lack of precipitation" (United States Department of Agriculture, 2021). However, it insures only for loss of precipitation, not for heat or wind, all droughts, or other natural causes of livestock or feed loss, and it does not provide replacement costs for livestock lost. It also requires farmers to anticipate the months of likely loss of precipitation. Whole-Farm Revenue Protection (WFRP), a crop-neutral revenue insurance policy, was created in the 2014 Farm Bill and can support diversified farmers including graziers, but program rules, low payouts, farmer lack of familiarity, and paperwork requirements have hindered adoption. WFRP requires 5 years of farm tax records so can be limited for beginning farmers unless they took over an existing operation.

The lack of support for pasture relative to corn and soybeans makes it difficult for many farmers to justify growing anything other than row crops. Annual average insurance payments for corn and soybeans from 2005 to 2021 in the Midwest states were \$382M in IA, \$364M in IL, \$72M in MI, \$316M in MN, \$226M in MO, and \$122M in WI, compared with the amount for pasture of \$0.5M in IA, \$0.4M in IL, \$0.6M in MI, \$2M in MN, \$3M in MO, and, \$5M in WI (USDA Risk Management Agency, 2022). Furthermore, commodity subsidies were: \$580M in IA, \$484M in IL, \$83M in MI, \$204M in MN, \$111M in MO, and \$133M in WI compared with the amount for pasture of \$0 for all 6 states from 2005 to 2018 (Environmental Working Group, 2020a,b).

Federal spending on crop insurance and commodity programs is variable but increasing. Because they cover both price and yield loss, the cost of these programs increases as production increases and prices drop: between 1991 and 2017, taxpayer subsidies for crop insurance have increased from \$300 million to \$6.1 billion (National Sustainable Agriculture Coalition, 2017). Commodity subsidies and crop insurance are expected to increase

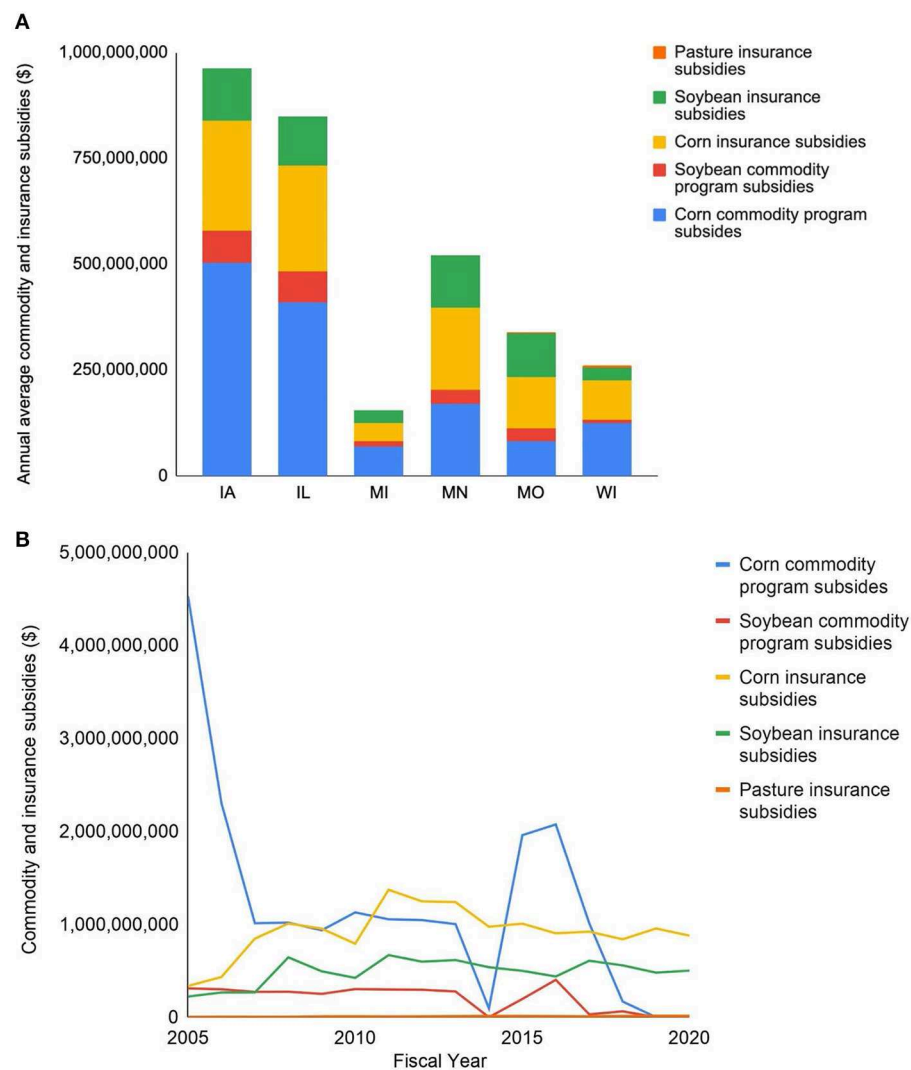


FIGURE 2

(A) Annual average commodity and insurance subsidies per year in Iowa, Illinois, Michigan, Minnesota, Missouri, and Wisconsin (\$). Annual average is between 2005 and 2021 for insurance (17 years) or 2005–2018 for subsidies (14 years). (B) Subsidies by USDA commodity and insurance programs in Iowa, Illinois, Michigan, Minnesota, Missouri, and Wisconsin (\$). Data source: Commodity subsidy data is from Environmental Working Group (Environmental Working Group, 2020a,b), including Direct Payments (DP) and Production Flexibility Contract (PFC) (1996–2013), Average Crop Revenue Election (ACRE) (2009–2013), Agricultural Risk Coverage (ARC) (authorized by 2014 Farm Bill, payments began in 2015), Price Loss Coverage (PLC) (authorized by 2014 Farm Bill, payments began in 2015), Price Support [Loan Deficiency Payments (LDP), Marketing Loan Gain (MLG), Commodity Certificates, and Counter-Cyclical Payments (CCPs)] (introduced in the 1996 Farm Bill). Insurance subsidy data is from the USDA Risk Management Agency (2022). Pasture insurance subsidies include forage production, forage seeding, and pasture, rangeland, forage.

greater than initially projected in coming years due to COVID-19, climate change impacts, crop price fluctuations, and trade wars (Taxpayers for Common Sense, 2022). A recent report estimates that eliminating crop insurance premium subsidies to farms with an adjusted gross income of >\$250,000 would save taxpayers \$20.2 billion over 10 years (National Sustainable Agriculture Coalition, 2022). Iowa farmers recognized as environmental leaders primarily supported incremental rather than transformative Farm Bill policy changes, though the majority supported conservation compliance on all lands receiving crop insurance, not just Highly Erodible Lands (Medina et al., 2020).

Renewable fuel standard

Federal mandates for ethanol have also contributed to grassland decline and row crop expansion (Wright et al., 2017). Ethanol is mandated in the Renewable Fuel Standard which originated with the Energy Policy Act of 2005 and was later extended under the Energy Independence and Security Act of 2007 (United States Department of Energy, 2022). Oil refiners and gasoline and diesel importers are required to sell specified volumes of renewable fuels enforced through significant fines. Renewable fuels include conventional, cellulosic, and advanced biofuel, and biomass-based diesel.

Financial and technical assistance

“We have way more applications than we have money for pasture land, whether it be the state or federal programs... I don’t know that that would ever completely go away, no matter how much money you threw at it.”—Selma Mascaro, State Grazing Specialist, NRCS Missouri

“If you’re part mechanical engineer and you can get through the rules and all of the tape, it’s great.”—Jen Falck, Wisconsin Partnership Program Coordinator, Oneida Nation

“...you could see the tremendous impact that having good grazing plans had on this establishment of successful grazing farms. [In] adjacent demographically similar counties [where] they didn’t...the difference was very stark...It’s really clear that what had made the difference really was GLCI (the Grazing Land Conservation Initiative).”—Margaret Krome, Policy Program Director, Michael Fields Agricultural Institute

The federal government provides financial and technical assistance for managed grazing and prairie restoration through conservation practices, activities, and enhancements under Farm Bill programs. The most notable programs are the Conservation Reserve Program (CRP), Environmental Quality Incentives Program (EQIP), Conservation Stewardship Program (CSP), Grasslands Reserve Program (GRP), and the Regional Conservation Partnership Program (RCPP). While these programs provide important support, they also create frustrations among farmers and their advisors due to long wait times to receive funding and a management plan, a high level of technical engineering for some practices, high up-front capital requirements, and higher support for cattle than other livestock (Reimer and Prokopy, 2014). An important advantage of the CSP program is that it allows for payments for farmers to maintain conservation practices they have already adopted, ensuring that early grazing adopters can still receive support.

Our analysis suggests declining or stable NRCS investments in financial assistance for grazing land conservation practices between 2005 and 2020, depending on the state (Figure 3). Missouri has the most non-woodland pasture of any state in our region, and also experienced the most dramatic decline of land area receiving NRCS funding for the specific conservation practice of prescribed grazing (Figure 4). While financial data was not available for all states, Wisconsin farmers received a total of \$24.3 million from the NRCS for pasture obligations from 2010 to 2019 through EQIP and CSP. This is a small fraction (6%) of total EQIP and CSP expenditures in Wisconsin. In FY20, NRCS applied conservation practices to 7,593 hectares of grazing land to improve the resource base in Wisconsin. Through EQIP, NRCS obligated \$968,461 for prescribed grazing across a count of 352 practices in FY20 such as fencing, water, and seeding (Legislative Fiscal Bureau, 2019).

Some local governments also provide grazing support. Districts or counties have the ability to cost-share managed grazing practices and provide technical assistance if it is identified as a local priority. Sometimes districts provide cost-share for livestock access lanes, stream crossings, watering facilities, and pasture establishment to promote rotational grazing.

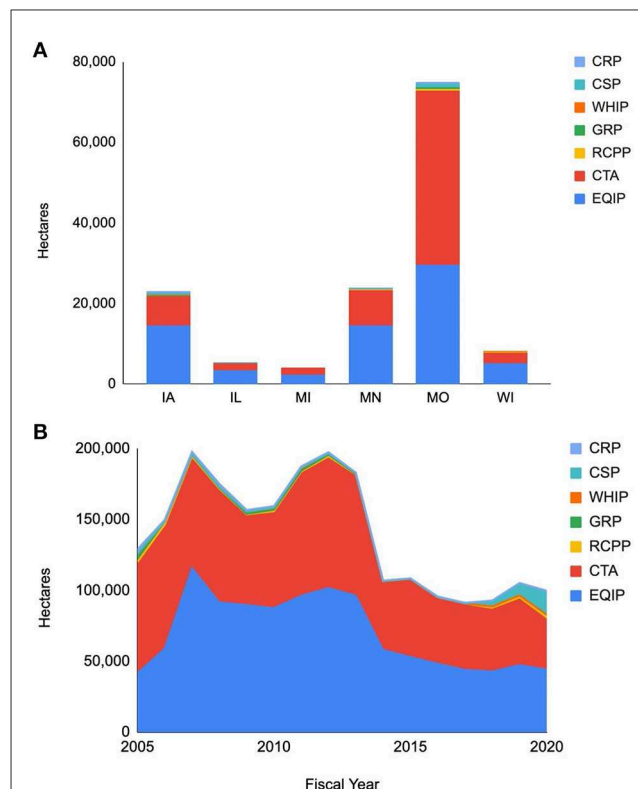


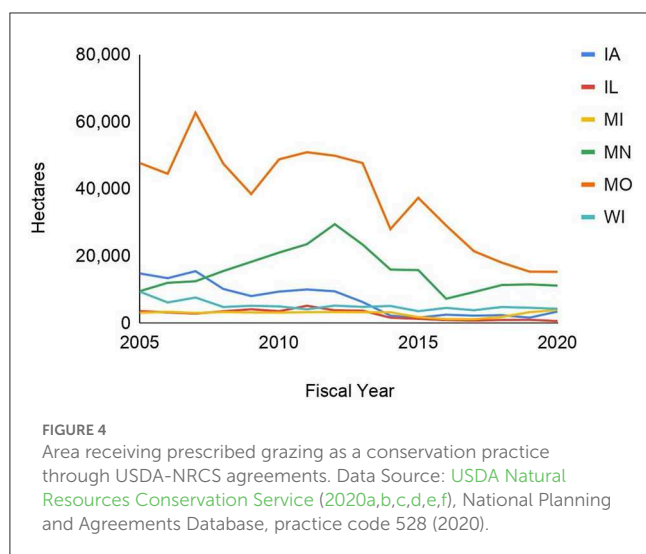
FIGURE 3

(A) Annual average land unit hectares per year receiving Grazing Land Conservation Practices in Iowa, Illinois, Michigan, Minnesota, Missouri, and Wisconsin by USDA-NRCS programs. Annual average is calculated between 2005 and 2020 for each program (16 years).

(B) Land unit hectares receiving Grazing Land Conservation Practices by USDA-NRCS programs in Iowa, Illinois, Michigan, Minnesota, Missouri, and Wisconsin. The programs are the Conservation Reserve Program (CRP), Conservation Stewardship Program (CSP), Wildlife Habitat Incentive Program (WHIP), Grassland Reserve Program (GRP), Regional Conservation Partnership Program (RCPP), Conservation Technical Assistance (CTA), and Environmental Quality Incentives Program (EQIP). Data source: USDA Natural Resources Conservation Service (2020a,b,c,d,e,f), National Planning and Agreements Database, October 2020. Grazing Land Conservation Practices. The 2014 Farm Bill was the first substantial reduction in conservation program funding since 1985.

Grazed cover crops can serve as a potential on-ramp for conventional farmers to start grazing or collaborate with graziers. Cover crops can be an important approach for increasing grass and other winter crop cover to reduce soil erosion, although they do not provide perennial grassland. Some farmers reported barriers with the EQIP program including long wait times to get a grazing management plan and receive EQIP funds, the need for up-front capital which can be prohibitive, lack of support before someone has livestock which makes it difficult to plan, challenges for row crop farmers to use cover crops as a stepping stone toward grazing, and lack of knowledge and support for livestock other than cattle.

Federal funds that support grazing networks and education have declined due to the end of funding for the Grazing Lands Conservation Initiative (GLCI) which was funded federally starting in 2004 and in some states extended until 2012. The GLCI supported state-based partnerships, network coordination, and education and technical assistance and education for graziers



and their service providers. \$14 million of the prior \$30 million appropriation for GLCI was restored through CTA in the FY22 Appropriations Package (CSRA Science, 2022).

Network of assistance organizations

Education, social norms, and farmer networks are important policy tools to help farmers make informed decisions with social support about how to transition and improve their managed grazing. Each state has a network of non-profit, university, and livestock association organizations that supports managed grazing, grasslands, and prairie. Some states have statewide member-based grazing organizations that provide leadership and education to farmers and consumers for the advancement of managed grazing including presentations, newsletters, field days, videos, an annual conference, and pasture walks (Grassworks, 2022; Minnesota Grazing Lands Conservation Association, 2022).

A number of organizations provide pasture walks, education, and information on grazing in their programming and publications including local Conservation Departments or Districts, NRCS, state natural resources and agricultural agencies, Grassworks, Marbleseed (formerly MOSES), Savannah Institute, University Extension, Resource Conservation & Development councils (RC&Ds), Michael Fields Agricultural Institute, and state Farmers Unions. The Wallace Center's Pasture Project has developed a pasture blueprint for Illinois and is expanding to other states. Green Lands Blue Waters is based in Minnesota and organizes information and hosts an annual meeting. Practical Farmers of Iowa is an important hub for conservation agriculture including grazing. The Missouri Center for Agroforestry is one of the world's leading centers on agroforestry, the integration of trees, crops, and livestock. The Dairy Grazing Apprenticeship program offers a recognized federal workforce development certification, which is based in Wisconsin and serves multiple states. GrassWorks in WI provides leadership and education to farmers and consumers for the advancement of managed grass-based agriculture. The Savanna Institute is researching and educating farmers about agroforestry. The UW-Madison Center for Integrated Agricultural Systems

(CIAS) has also been involved in agricultural education. University agricultural research stations house dairy heifers and beef herds that can be used for grazing research and to inform farmers. Universities and non-profits also develop decision support tools such as the Livestock Compass (Hendrickson and Munch, 2018) and Heifer Grazing Tool (Mulholland et al., 2022).

Grassland management and conservation are also supported by conservation and hunting organizations that provide information, prairie walks, and management training to landowners, such as Pheasants Forever, The Prairie Enthusiasts, state Departments of Natural Resources, US Fish and Wildlife Service's Partners for Fish and Wildlife, Aldo Leopold Foundation, state Prescribed Fire Councils, land trusts, grassland partnerships, and other bird and prairie conservation organizations.

Conservation Reserve Program

"Since I've been grazing for 20 years, I'm not eligible for CRP. Farmers that are thinking about transitioning - it would be beneficial for them."—Laura Paine, Grassland Farming and Outreach Lead, Grassland 2.0 & Beef Grazier, Paine Family Farm, WI

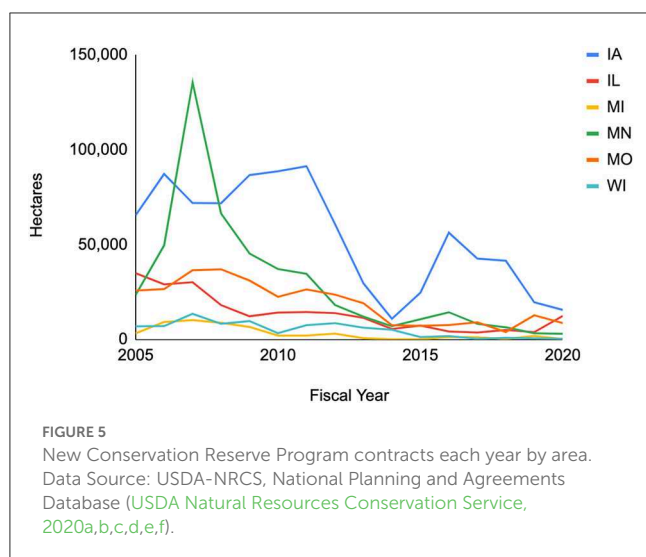
The CRP is the largest federal program managed by the Farm Services Agency. CRP provides an incentive to farmers to plant land into grassland cover and to take marginal lands out of production in order to protect water quality, provide flood control, and establish wildlife habitat. CRP operates through fixed term agreements, generally 10-years, that are connected to the deed so that they run with the land even if the owner changes. The program provides an annual payment to the landowner. CRP enrollments have not been resilient to increasing crop prices (Secchi and Babcock, 2007). The rising price of corn combined with price loss coverage in poor market years likely contributed to reduced enrollment in CRP in the upper Midwest between 2005 and 2020 (Figure 5). CRP promotes grassland conservation but only integrates moderately with grazing agriculture. It allows for emergency grazing during severe drought and non-emergency grazing every other year, limited to 50% stocking rate reduction during the bird breeding season (USDA Farm Service Agency, 2022).

Conservation easements and the Grassland Reserve Program

"Purchase of the development right is a great way for the landowner to have income and to be able to sell the land at a lower rate to a beginning farmer."—Kirsten Jurcek, Grazing Plan Writer & Beef Grazier, Brattset Family Farm, WI

"We're silent on who owns the land. \$0 provided for that...That's why we have the aggregation of land we have...Adding the ability for [ACEP-ALE] money to be used for acquisition of title to property...would go a long way."—Ian McSweeney, Director, Agrarian Trust

Conservation easements are perpetual or long term agreements that restrict development and can promote working land uses like



grazing and help farmers purchase agricultural land at a lower cost. Landowners typically receive a payment or tax reduction for the conservation easement. The 2002 Farm Bill introduced the Grassland Reserve Program (GRP), a voluntary easement program under which participants limit housing development and cropping to protect grasslands and their grazing and biodiversity benefits. For all six Midwest states the total number of GRP easements totaled 6,205 hectares for 114 contracts (37 in Missouri, 31 in Iowa, 22 in Wisconsin, 10 in Illinois, 10 in Michigan, and 4 in Minnesota) when the program was ended and brought under the Agricultural Conservation Easement Program (ACEP) (USDA Natural Resources Conservation Service, 2021). Nationally, the GRP supported prescribed grazing on 87% of its enrolled acreage while only 35% of ACEP Agricultural Land Easements received support for prescribed grazing.

Many states have programs to fund conservation easements and other types of long-term grassland reserves on private land. For example, Minnesota has a sales tax passed by state constitutional amendment, the Clean Water, Land and Legacy Amendment, that supports grassland conservation and other benefits. Missouri has the Parks, Soils, and Water Tax to support land, soil, and water conservation that can support grazing and grasslands.

Grass-fed and organic labels and certifications and supply chains

Labeling

Labels inform consumers about organic and grass-fed practices. Some labels are connected to formal governance systems through certification. For instance, milk and meat that are certified organic by the U.S. Department of Agriculture must have cows on pasture 120 days per year for 30% of their diet, specified in 2010 rulemaking. States vary in the number of organic farms (Table 1). Some programs require 100% grass-fed, such as Organic Plus Trust and American Grassfed Association (AGA, 2021). Midwest Organic Services Association (MOSA), based in Viroqua, Wisconsin, offers Grass-Fed Beef and Grass-Fed Dairy

certifications, which require at least 60% of each animal's feed to be from pasture. MOSA also offers Transitional Organic Verification cost-sharing for those who require support transitioning to an organic production system (MOSA, 2022).

Consumer demand for organic and grass-fed beef is rapidly increasing. The Nielsen Marketing Research firm found that sales of organic and non-organic grass-fed beef doubled each year between 2012 and 2016. In contrast, conventional beef sales increased by just 7% each year (Stone Barns Center for Food Agriculture, 2017). Despite the market potential for the grassfed industry, there is little governmental support for American producers (Stone Barns Center for Food Agriculture, 2017). While global consumer demand for organic milk is increasing, US dairies have been squeezed as costs increase more than prices with competition from New Zealand, Australia, and other countries (Askew, 2022).

Implementation of the Country of Origin Labeling (COOL) law for beef and pork is an important issue for many farmers raising animals. COOL previously required labeling of where meat was born, raised, and slaughtered. However, after a trade dispute under the World Trade Organization, USDA stopped enforcing country of origin labels for beef and pork in 2015. As a result, many companies are labeling meat raised abroad but repackaged at U.S. facilities as a U.S. product (United States Department of Agriculture, Agricultural Marketing Service, 2022). There is some dispute about the ramifications of reintroducing COOL for beef and whether it would lead to threats of sanctions from other countries. COOL does not apply to dairy products and while there have been some efforts to change it, the U.S. dairy industry has not been supporting the move as strongly as some cattlemen's associations (Myers, 2022; Progressive Farmer, 2022).

Supply chains

Consolidation is a major trend impacting dairy and meat production. The beef industry's processing is highly consolidating with four companies controlling the majority of the market, sparking antitrust challenges [In Re: Cattle and Beef Antitrust Litigation, case No. 0:22-md-03031-JRT-JFD (D. Minn)]. Four large meat-packing companies control over 80% of the market and have simultaneously been paying less to farmers while charging consumers more, leading to a Presidential Executive Order for a whole-of-government approach to increasing economic competition (The White House, 2021). Critics argue that lack of antitrust enforcement contributes to consolidation, as have agricultural education, research funding, and lending. Increasing access to regional meat processing is important for grass-based producers, which has been gaining policy attention.

Federal dairy programs have failed to address problems of oversupply. Without market signals that limit annual increases in milk production relative to demand, small and medium dairy farmers are being pushed out of the market. Milwaukee Journal Sentinel's journalist Rick Barrett documented the crisis in a Pulitzer Center series "Dairyland in Distress" (Barrett, 2019). The reports were sobering before the COVID-19 pandemic, and only worsened after. In 2018, Wisconsin led the nation in farm bankruptcies, and lost 700 dairy farmers—nearly two per day. In April 2019 he documented a loss of three per day. On average, milk costs

TABLE 1 Number of organic farms, sales, and land area by state in 2016 (USDA National Agricultural Statistics Service, 2017a,b,c,d,e,f).

	Iowa	Illinois	Michigan	Minnesota	Missouri	Wisconsin
Organic dairy farm	76	16	70	108	21	455
Organic beef farm	17	14	10	19	7	59
Organic dairy sales	\$15,549,114	\$298,665	Unknown	\$43,326,781	\$4,898,174	\$125,933,062
Organic beef sales	\$389,497	\$351,885	\$161,355	\$138,654	Unknown	\$700,896
Organic pasture or range (hectare)	5,484	1,502	3,856	7,553	4,056	20,991

\$17–22 per hundredweight (cwt, about 12 gallons) to produce, while the price farmers receive averages \$15.13. Economic research indicates that if a federal growth management policy was adopted, an average Wisconsin grazing dairy would realize a Net Farm Operating Income increase of up to 74%, and depending on the policy design, average annual milk prices would increase between \$0.73 and \$1.41/cwt for farms that stayed within production limits (Nicholson and Stephenson, 2021).

Grazing is a lower-input, lower-output form of agriculture than grain-fed livestock production. Grazing requires less machinery, fertilizer, and herbicide, although it does rely on fencing and sometimes some fertilization and seeding. Due to the lower inputs in grazing, it does not attract as much agribusiness interest, demonstrated by fewer industry sponsors at grazing conferences (Lu and Rissman, 2022).

Environmental policy interplay: Water, wildlife and plants, carbon

Water quality policy

Well-managed grasslands can reduce soil erosion and nutrient runoff, so perennial cover is a strategy for achieving goals of the federal Clean Water Act (CWA) and state water quality policy. However, grazing can also degrade water quality through overgrazing, compaction, erosion, and streambank destabilization. Under the CWA, state agencies develop watershed plans to achieve Total Maximum Daily Load (TMDL) of pollutants reaching impaired waterways, with approval from the Environmental Protection Agency. Smaller farms are primarily managed through voluntary, incentive-based water quality programs, while point sources such as Confined Animal Feeding Operations (CAFO), sewage treatment plants and cheese factories are mandated to meet permitted amounts of pollution. TMDLs model phosphorus and sediment loads from pasture/grassland and other land uses and point sources to establish a baseline and model the potential for water quality improvements. TMDLs rely on a variety of mechanisms for implementation, including state standards. For instance, Wisconsin's agricultural performance standards prevent unlimited livestock access to waters of the state in locations where high concentrations of animals prevent the maintenance of adequate or self-sustaining sod cover" (NR 151.08).

The EPA developed guidance in 2021 for states to use the Clean Water State Revolving Fund (CWSRF) for non-point source reduction (EPA, 2021). CWSRF received a major influx of funds under the Bipartisan Infrastructure Law. While non-point source pollution accounts for about 75% of water quality impairments in

the US, only 4% of the CWSRF has addressed non-point source pollution, an imbalance the EPA is seeking to remedy (EPA, 2021).

Water quality funding is more likely to subsidize manure storage, barnyards, and rooftops for confinement operations rather than incentivize transitions to lower-density managed grazing on perennial cover. States vary in their nutrient reduction strategies and state laws for phosphorus, nitrogen, and nutrient management planning. Wisconsin has a numeric phosphorus criteria and has developed a water quality trading program that allows point sources to fund conservation practices on agricultural land which can be cheaper than the marginal gains available in sewage treatment facilities and factories (Wu, 2021). Missouri and Iowa Nutrient Reduction Strategies include grazing and estimate its contributions to nutrient load reduction. States can develop standards for grazing management such as amount of residual dry matter. State erosion and phosphorus standards such as Wisconsin's NR151 also apply to grazed pastures, even though many grazing farms do not have a nutrient management plan. Examples of incentivizing grassland and pasture at the county scale include Dane County's Continuous Cover Program that provides cost share for establishment of both cool-season grass pastures and native prairie. Improving water quality is a primary goal of the program but also includes reducing soil erosion, sequestering carbon, and enhancing wildlife as outcomes.

Farmer-led watershed groups emerging in Iowa, Wisconsin and other states have stressed adoption of cover crops, no-till, prairie strips, and other practices compatible with corn and soybean plantings, while some are also educating members about pasturing livestock, such as farmer spotlights on grazing dairy heifers (WDATEP, 2022).

Wildlife, plant, and pollinator, and rare species conservation policy

Policies related to wildlife, plants and pollinators influence grasslands. Grasslands are critical for wildlife to sustain their populations and for human uses for hunting upland game such as pheasants and grouse, birdwatching, and hiking. Many grassland birds and plants have declined with the loss of grassland habitat. Wildlife is managed by states as a common resource not owned by individual landowners, with a system of hunting quotas and license fees. Federal Pittman-Robertson Act funds wildlife research and land stewardship, including for grassland-based wildlife, through an excise tax on hunting and fishing gear including firearms and ammunition.

The federal Endangered Species Act (ESA) and state-level endangered species laws aim to protect species at risk of extinction.

While ESA has been a powerful mechanism for preventing extinction on federal lands and due to federal actions such as dams, it has not been influential in preventing crop expansion into important habitats for species on the Threatened and Endangered Species list. For instance, the Poweshiek skipperling was once common, but the butterfly was listed as federally endangered in 2014. Its surviving populations have been extirpated from North Dakota, South Dakota, Minnesota, Iowa, and Illinois, and it remains in reduced numbers in Michigan, Manitoba and one area in Wisconsin. It continues to decline due to threats including loss of habitat, pesticides, climate change, invasive species, altered hydrology, and lack of disturbance, while recovery efforts are just beginning to understand the species' biology and recovery options (USFWS, 2019). The monarch butterfly depends on milkweed and its habitats have declined and been impacted by pesticides. In 2020 USFWS determined it is warranted for listing but that listing is precluded due to capacity constraints; it was listed in 2022 on the International Union for the Conservation of Nature Red List. States also maintain threatened and endangered species lists but rarely have regulatory authority over habitat loss due to agriculture. Migratory Bird Joint Ventures organize federal, state, and non-governmental partners to conserve bird habitat.

Animal and plant diversity can be enhanced or degraded due to grazing. Grazing can impair prairie plant diversity and grassland bird nest success through trampling and feeding if not well-managed for the site. While grazing is not appropriate for all prairies, it can be beneficial for some goals in the right contexts (MDNR, 2021). Pastures that include a broad seed mix including clover and other forbs can enhance diversity. Grazing is one tool for grassland management to prevent succession to woody species, along with prescribed fire, herbicide treatment. Grasslands and the species they support often require active stewardship that can be funded through a variety of policy mechanisms and supported by social networks of professionals and volunteers.

Carbon and other environmental markets

"If USDA gets involved, the carbon offset credits are going to come from just doing more of the same with some little tiny amendment... A CAFO is never going to be a carbon sink... it's just allowing polluters to keep polluting... The research just has not shown that these carbon market schemes actually reduce emissions. It's just a profit-making scheme that I think makes people feel better... we have farmers who are just like, 'I do not want to be in relationship with fossil fuel companies... that's not why I am doing soil health practices, I'm not doing it to bail them out'... I don't think that those farmers' voices are being heard."—Non-profit employee, Michigan, Interview #96

Markets that provide payments for ecosystem services including water quality and carbon storage and sequestration are increasingly piloted and discussed. The way environmental practices are accounted for in ecosystem models is pivotal to the payments farmers would receive. Based on the broad definition of carbon practices the USDA listed in a recent request for proposals for climate-smart commodities, many are concerned that carbon markets would incentivize conservation practices such as a cover crops that result in little soil carbon accumulation

over the long term (Jian et al., 2020; Blanco-Canqui, 2022), rather than incentivizing permanent conversion to perennial cover, such as through grasslands or well-managed pastures, that have the potential to provide long-term carbon storage (Rui et al., 2022; Sanford et al., 2022). The rise in private agri-environmental initiatives raises questions about how public programs can support and supplement them to ensure effective and equitable outcomes (Baylis et al., 2022).

Public and tribal lands

Public lands

Some local, state and federal lands allow conservation grazing in some wildlife management areas. For instance, the Minnesota Department of Agriculture provides opportunities for grazing and haying on certain public lands across the state through their Conservation Grazing Program (Minnesota Department of Agriculture, 2022). Missouri's Department of Conservation mentions the benefits of conservation grazing to manage natural grasslands and prairies (Missouri Department of Conservation, 2022). The management plans for public parks such as Ozark Highlands Southwest Prairie Area, Pawnee Prairie, Chapel View Prairie, and Robert E. Talbot Conservation Area all include prescribed grazing as a strategy to reach management goals. Wisconsin has a collaborative project with university extension and private graziers called Grazing Public Lands in Wisconsin (Grazing Public Lands in Wisconsin, 2018; Pasture Project, 2020). This project evaluates the opportunities and challenges of rotationally-grazed livestock for conservation on public grasslands. Illinois Department of Natural Resources (DNR) Wildlife Protection Program plans on using prescribed grazing to restore certain state-protected prairie lands, such as Prairie Ridge State Natural Area and Twelve-Mile Prairie (Illinois Department of Natural Resources, 2022). However, Iowa DNR only allows emergency haying and grazing on DNR managed land during times of disaster declared by the governor (Iowa DNR, 2013). The Michigan DNR has a Public Land Strategy that does not mention grazing (Michigan Department of Natural Resources, 2013).

Tribal lands

Tribal governments are important actors in developing policies and programs for grasslands and grazing agriculture. These programs are often structured to promote food sovereignty and support food banks, elders, and community. Self-governance is the core of sovereignty, and control of meaningful processes of food production is important for Native Nations. Efforts are underway to reform the Food Distribution Program on Indian Reservations (FDPIR) to promote food sovereignty. For instance, FDPIR 638 Self-Governance Demonstration Project has given certain Nations (including the Menominee and Oneida Nations in Wisconsin) control over what goes into food boxes, enabling them to provide their communities with culturally appropriate foods sourced from Native farmers (Indigenous Food and Agriculture Institute, 2022).

Several Native Nations pasture livestock to revitalize traditional foodways and provide healthy food and connections to land. For instance, in Iowa the Meskwaki Nation Natural Resources

Department manages a wildlife refuge that commonly has bison and is seeking to expand and create a new management plan for the herd (Meskwaki Department of Natural Resources, 2017). In Michigan, the Bay Mills Indian Community runs the Waishkey Bay Farm where they pasture poultry and raise grass-fed beef (Bay Mills Community College, 2022). In Minnesota, The Prairie Island Indian Community has 40 bison that roam on 55 hectares of tribal lands including pasture and prairieland. In Wisconsin, Oneida Nation educational farm Tsyunhehkwa has a herd of cattle (Tsyunhehkwa Agriculture, 2019). The Oneida Nation Farms and Agriculture Center raises steers, cow-calf pairs, and grass-fed bison (Oneida Nation, 2018). The Forest County Potawatomi own and operate a farm called Bodwéwadmí Ktëgan, where they raise pastured chickens, hogs, grass-fed cattle and bison. The Ho Chunk Nation used to have a bison herd at Badger Army Ammunition Plant, but this program ended due to financial challenges (Wisconsin Public Radio, 2010). The Menominee Nation has allocated land for farming operations, is actively developing a food production initiative including grazing, and building an agricultural degree program at the College of Menominee Nation. In both Illinois and Missouri there are no federally recognized indigenous nations. All indigenous nations that historically lived in Illinois and Missouri were violently forced from their lands and now reside in surrounding states (University of Missouri Libraries, 2022).

State and local plans and taxes

Plans

While all states have Wildlife Action Plans and Forest Action Plans, most states do not have Grassland Action Plans. USDA released the Northern Bobwhite, Grasslands and Savannas Framework for Conservation Action in 2022 to direct action toward priority counties in the central and eastern U.S. (USDA Natural Resources Conservation Service, 2022). The Minnesota Prairie Conservation Plan calls for protecting all native prairie from conversion, 40% grassland and 20% wetland in core and habitat complex areas, and 10% grassland in other areas of the state (Minnesota Prairie Plan Working Group, 2018). Some county or district land and water resource management plans have mentioned the benefits of grazing and grasslands and have set goals to promote grazing. State and county comprehensive plans designate land use areas but have limited regulatory authority. States have also developed pollinator plans that promote the conservation or reestablishment of native prairies and savannas (Locke et al., 2016; Minnesota Board of Water Soil Resources, 2019; Michigan Pollinator Protection Plan Steering Committee, 2022; Missourians for Monarchs Collaborative Steering Committee, 2022).

Property taxes

Agricultural land including grazing land has lower tax rates in our study states, however, prairie without grazing or haying is subject to higher taxes in some states. By Iowa law, the value of agricultural property taxes must be based on the current land

use rather than its highest and best use. Farmers can apply for the Agricultural Land Tax Credit that provides a tax credit in an amount determined by the county auditor to offset high farm taxes (Iowa Department of Revenue, 2022). There is also the Family Farm Credit that aims to provide \$10 million in property tax credits to landowners actively engaged in farming. Grazing land is taxed similarly to agricultural land. In Iowa, native prairie land and open prairie land are also eligible for tax credits or exemptions (Iowa Department of Revenue, 2022). In Illinois, the tax rate for cropland including rotational pasture is higher than the tax rate for permanent pasture. Illinois also has a conservation stewardship tax exemption for those with conservation plans approved by the Illinois Department of Natural Resources (2022). In Michigan, agricultural land has the potential to be exempt from certain local school operating taxes under the Qualified Agricultural Property Exemption program (Michigan Department of Natural Resources, 2013; State of Michigan, 2018). Land must be more than 50% in agricultural use to qualify and the definition of agricultural use includes grazing and pasture. Grasslands not under agricultural use do not qualify for this exemption. Minnesota has a few programs that allow agricultural land to be taxed at a lower rate. The Green Acres Program or Minnesota Agricultural Property Tax law states that farmers' properties should be valued using an agricultural lens rather than true market value which may be higher due to developmental pressures (Minnesota Department of Revenue, 2020). In conjunction with this program the Minnesota Department of Revenue (2018) also organizes the Rural Preserve Property Tax Program which provides the same relief for property taxes on rural land that is vacant, but still part of a farm. The Minnesota Department of Revenue (2022) also provides lower property taxes to special agricultural homesteads; land must be unoccupied and actively farmed to qualify for this program. In Missouri, property taxes are calculated as a percentage of the assessed market value of the land. Agricultural land including grazing land is taxed at 12% of market value of the property. Grain crops taxed as personal property are assessed at 0.05% of market value. Most property including grasslands are taxed at a rate of 32–33% of the assessed market value. Some agricultural producers can qualify for tax credits through the family farm breeding livestock program or the qualified beef tax credit program. Agricultural land including grazing land has lower tax rates in Wisconsin, but grassland without grazing or haying is subject to higher taxes. Farmers who graze woodlands are taxed at the agricultural rate and pay lower taxes than woodland owners without grazing in some states such as Wisconsin (Mayerfeld et al., 2016).

Access to land, capital, and fair labor

"There were both legal and illegal transfers that were enforced by State-sanctioned violence against Native people and through the forced labor of Africans that [have] never been atoned for...Land didn't just pop up and exist, and people were like, 'Oh, it's yours. It's free.' That's a story that we're told, but that's not the reality in most cases."—Neil Thapar, Co-Director, Minnow

"I was overwhelmed by the amount of farmers that [were interested in cooperative land ownership]...Many farmers, vegetable farmers, livestock farmers, crop farmers, said similar things."—Meghan Filbert, Livestock Program Manager, Practical Farmers of Iowa & Diversified Grazier

Land access

Land access is an important issue for bolstering grasslands and managed grazing as well as supporting the next generation of farmers and addressing financial and racial equity in landownership (Spratt et al., 2021). Land is increasingly out of reach, particularly for smaller farmers, because of decreasing farmland availability and skyrocketing costs. These trends are driven in part by consolidation in land ownership, urban development, and financial speculation in farmland. Subsidies and lending norms disproportionately increase the profits of large, commodity farms and CAFOs (Bekkerman et al., 2018; Azzam et al., 2021). This creates a positive feedback loop whereby as these farms gain land, they are able to leverage more capital, allowing them to acquire more land and driving smaller farms, such as many of those practicing managed grazing, out of business. Increased financialization of farmland has also led to a proliferation of landholding by companies, funds, and wealthy individuals, making it difficult for farmers, particularly smaller farmers, to compete (Ross, 2014; Fairbairn, 2020). Likewise, urban development can increase the cost of farmland especially near urban areas (Livanis et al., 2006). This is a particular issue for smaller, sustainable farmers who often rely on niche markets in cities and for many immigrant communities located in urban areas who are interested in farming. Also, if farmers are not profitable enough to create retirement accounts, that can increase the pressure to sell land for development or intensive agricultural use (Lowe, 2022).

Some land trusts, farm organizations, universities, and local, state, and Federal staff assist farmers in accessing land. This assistance includes technical support for succession planning and programs that help facilitate land transfers to beginning farmers such as FarmLink programs, state-level tax incentive programs, and the Conservation Reserve Program Transition Incentives Program (CRP-TIP). Some state or local-level programs around land access are funded through the Beginning Farmer and Rancher Development Program (BFRDP).

However, these types of support programs for land access are underdeveloped relative to other forms of technical and financial assistance for farmers (Lowe, 2022). This lack of support extends to the Farm Bill, which has no title or program focused on land access. What little funding is provided is scattered across programs like Agricultural Conservation Easement Program—Agricultural Land Easements (ACEP-ALE), CRP-TIP, and BFRDP without a coordinated approach. Moreover, very little effort has gone toward addressing the financialization of farmland and reducing consolidation in land ownership. A small number of land trusts and cooperative land stewardship programs seek to address these issues by purchasing land and enabling joint ownership by community members.

Issues with land access and affordability disproportionately affect farmers of color who have been systematically deprived of land ownership through a variety of means including many U.S. government practices and policies. Centuries of governmental policy and practices have been used to systematically remove Native peoples from their homelands, redistribute that land to white farmers, and exclude other farmers of color from land ownership (Horst and Marion, 2019). These include treaties with Native Nations, the Indian Removal Act, the Homestead and Allotment Acts, slavery, immigration and labor policy, the Japanese Internment Act, heirs property laws, and USDA discrimination against farmers of color. Because of this, few people of color own farmland. Today, 97% of agricultural land is owned by white farmers although people of color make up the majority of the agricultural labor force (Horst and Marion, 2019). This dynamic makes it particularly difficult for farmers of color to build the wealth and access the capital necessary to purchase farmland.

Access to capital

Grazing operations require less capital than conventional livestock operations, but farmers still need capital for purchasing livestock and other equipment. Dairy farms require higher levels of capital for milking. Smaller ruminants such as poultry, sheep, and goats may have lower barriers to entry since smaller animals are less expensive and can cash flow faster.

Farmers can obtain loans from USDA Farm Service Agency, Farm Credit, and private banks for operations. Lenders are often familiar with conventional livestock operations' financial information but lack financial data on grazing operations, so it is still difficult for grazing farmers to get enough credit (Spratt et al., 2021). The FSA's Beginning Farmer and Rancher loan program offers financial assistance for beginning farmers with 3 years of farm management experience. However, farm labor is not counted as management experience, excluding many potential farmers who have extensive farming knowledge, including knowledge of animal agriculture.

Fair labor

The Fair Labor Standards Act (FLSA) and other labor laws often include exemptions that exclude agricultural workers from protections around minimum wage and overtime pay. Poor pay and workplace abuses are exacerbated by immigration laws that prevent workers from gaining citizenship, creating a situation in which workers are afraid to report abuses due to fear of deportation. As a result, 97% of profits made in agriculture are made by white farm owners, rather than being shared more equitably across the agricultural labor force (Horst and Marion, 2019).

State laws have expanded in some cases to increase overtime, minimum wage, and workers compensation for agricultural workers. Minnesota requires employers to pay overtime to many farmworkers unless they receive a salary or are not employees. However, the other states in the Upper Midwest region do not offer overtime pay to farmworkers. Overtime requirements for farmworkers have been expanding in states

such as New York, California, and Washington (Hoard's Dairyman, 2021; Farmworker Justice, 2022). Wisconsin includes agricultural workers in its minimum wage, while the other states in this region cover many but not all agricultural workers. Labor policy is particularly influential for grass-based dairy operations because of the extra labor involved in milking.

Fair labor standards are also being improved through private governance of supply chains, such as the Milk with Dignity campaign led by the Vermont-based organization, Migrant Justice. The Milk with Dignity campaign resulted in Ben and Jerry's signing onto fair labor standards with third party enforcement (Migrant Justice, 2022). Unlike other label-based fair trade standards, Milk with Dignity is farmworker centered, with a farmworker written code of conduct and a premium paid to farmers and their workers who join as members (Frye-Levine et al., 2019). We did not find evidence of a similar fair milk campaign in our study region.

Cooperative (co-op) ownership structures are important for enhancing farmer control and profit-sharing. Many agricultural co-ops play important roles in grass-based milk and meat. For instance, Organic Valley based in southwest Wisconsin is the nation's largest farmer-owned organic cooperative, including numerous small farms located in the upper Midwest and across the U.S. Minnesota-based Regenerative Agriculture Alliance is building a cooperative network of silvopasture chicken farms as well as processing facilities and marketing structures.

Actionable recommendations

Well-managed grasslands, savannas, and other forms of perennial agriculture are presently underutilized, yet have the ability to increase farmer profitability, grow strong, diverse rural communities, keep water clean and prevent flooding, build soil health and stabilize climate, revitalize wildlife and pollinator habitat and biodiversity, and produce high-quality milk and meat. If decision-makers want to support a transition to perennial grass-based agriculture, these recommendations from farmers and stakeholders in the grazing community suggest a variety of policy approaches. Further research is needed on these recommendations including quantitative modeling of their expected ecological and economic impacts and social science research on their perceived feasibility and legitimacy.

Federal subsidies, insurance, and renewable fuel standard

Reform crop insurance and subsidies

- Improve the financial safety net for grass-based agriculture including improved pasture and whole-farm crop insurance to increase farmer adoption.
- Reform crop insurance for corn and soybeans to reduce detrimental impacts on grasslands, including greater

flexibilities for base acres. Cap payment amounts and limit payments based on income.

Revise the ethanol mandate

- Revise the ethanol mandate to promote conservation agriculture and seek alternative domestic renewable energy sources.

Financial and technical assistance

Improve financial and technical assistance

- Expand the support for grassland and managed grazing in local, state and federal cost-share, grant, and loan programs to benefit grass-based livestock, clean water, flood mitigation, soil carbon, and habitat for wildlife and pollinators.
- Enhance Environmental Quality Incentives Program (EQIP), Conservation Stewardship Program (CSP), and other programs by reducing wait times and up-front capital requirements and lowering infrastructure standards for fencing.
- Establish a Perennial Crop Advisor Program within state and federal agencies to train crop advisors on how best to incorporate grasslands and other forms of perennial agriculture into existing cropping systems.
- Improve training about grass-based livestock systems for producers and public, private sector, and tribal advisors and conservationists, including silvopasture and livestock beyond cows.
- Enhance local technical assistance delivery through additional resources for soil and water conservation districts, university extension, and other local technical advisors.
- Enhance technical assistance for non-cow livestock such as sheep, pigs, and goats to better support beginning and socially disadvantaged farmers.
- Increase technical service support to socially disadvantaged farmers by focusing on building trust and hiring grazing experts from socially disadvantaged communities.
- Develop farmer to farmer training programs and networks for socially disadvantaged farmers.
- Prioritize perennial and grassland agriculture in cross-agency agricultural and conservation initiatives that support resilience to climate change.
- Develop and communicate quality standards for grass-based agriculture to achieve desirable environmental and social outcomes.

Enhance Conservation Reserve Program and conservation easements

- Promote Conservation Reserve Program (CRP) adoption to enhance environmental outcomes, with flexibility for working land uses when appropriate.
- Encourage conservation easements that secure grasslands while making managed grazing land more accessible and supporting appropriate public recreation opportunities.

Grass-fed and organic labels and certifications and supply chains

Enhance labels and certification and supply chains for grass-based farmers

- Further develop grass-based labels and certifications to enhance market share.
- Clarify labeling for consumers by enforcing the country of origin labeling.
- Address industry consolidation through antitrust legislation and updated legal frameworks.
- Develop grassland value-added supply-chains by supporting regional processors, aggregators, distributors, and marketers focused on grassland products and their stories.
- Establish and improve available financing and capital flows to assist small businesses engaged in establishing supply chains and markets for grasslands and other forms of perennial agriculture.
- Increase grants for start-up businesses that provide key supply chain infrastructure, such as processing, storage, and distribution.
- Enhance technical support and funding availability for business planning, lending, and marketing.
- Develop and increase support for cooperative farming and marketing structures.

Environmental policy interplay: Water, wildlife, plants, carbon

Prioritize perennial practices in water quality strategies

- Implement an all-of-government approach to prioritize perennial conservation practices in achieving water quality goals.
- Incorporate grazing and other perennial practices in state nutrient management strategies.
- Adopt pay for performance programs that reward farmers for sustainable management outcomes.

Enhance animal and plant diversity in grasslands

- Adopt pay for performance programs for plant and animal diversity on grazing and crop farms.
- Increase collaboration on threatened and endangered species recovery with agricultural agencies and managers.
- Increase investments in habitat stewardship to prevent extinction and future listings and keep common species common.

Ensure carbon and other environmental markets include perennial grasslands

- Ensure that carbon markets promote the long-term soil carbon benefits of perennial land cover and contribute to environmental co-benefits.

- Design carbon markets in ways that promote equity for smaller farm operations and inclusion of socially disadvantaged farmers.

Public and tribal lands

Consider well-managed grazing on publicly managed lands where appropriate

- Develop and test standards for environmentally sensitive grazing on a limited amount of public land that maintains wildlife and pollinator habitat.
- Expand grazing pilot programs by natural resource agencies as a conservation management strategy on publicly managed grasslands, where appropriate for achieving biodiversity, wildlife, and public recreation goals, with safeguards to ensure public benefits.

Support tribal grasslands and grazing

- Expand Native Nation land tenure and stewardship to restore prairie and grazing agriculture and improve food sovereignty.
- Create more positions for Tribal Liaisons (within NRCS) and invest in supporting organizations like the Wisconsin Tribal Conservation Advisory Council (which helps interface between Tribes and NRCS) for states across the upper Midwest.
- Increase coordination between the USDA and the Department of Interior Bureau of Indian Affairs to support grassland restoration and managed grazing on native lands.
- Expand Native Nation co-management of public grasslands to support food sovereignty.
- Increase Native Nation climate-smart perennial agriculture and forestry through institutional procurement and purchasing programs, such as expanding the FDPIR Self-Determination Demonstration Project.

State and local plans and taxes

Coordinate state-level planning, property taxes

- Develop state-level Grassland Action Plans to help guide agencies and partners in coordinating their efforts, modeled after the Forest Action Plans and Wildlife Action Plans that states must create to qualify for federal funds.
- Consider state property tax programs that ensure grazing is well-managed and provide property tax parity for well-managed woodlands, native prairies, and other grasslands.

Access to land, capital, and fair labor

Improve access to land, capital, and fair labor

- Increase availability and affordability of farmland by reducing farm consolidation, financial speculation, and urban sprawl.

- Improve infrastructure and programs to connect beginning and socially disadvantaged farmers to land that becomes available.
- Increase support for succession planning and decouple farmers' ability to retire from land sales.
- Increase incentives and culturally-responsive outreach strategies across land transfer programs including ACEP-ALE, CRP-TIP, and BFRDP.
- Provide beginning farmers with relief from student loan debt.
- Develop structures to help farmworkers build equity and modify programs like FSA's beginning farmer loan program to develop pathways to farm ownership.
- Encourage beginning and historically underserved farmers by providing stipends for mentor farmers, programs offering low-interest loans, debt relief, land access, assistance, and tax incentives, in order to ensure just transitions to perennial agriculture.
- Support cooperative and community-based models of land stewardship.

Conclusions

There is a critical need to revise agricultural policies if we are to restore grasslands and support managed grazing. Restoring and maintaining grasslands and grass-based agriculture is important for achieving water quality goals, protecting wildlife and pollinator habitat, stabilizing climate, providing flood resilience, enhancing rural communities, producing healthy food, and supporting viable farmer livelihoods. Current policies support row crops to the detriment of grasslands. Crop insurance and commodity subsidies, along with the federal mandate for ethanol, have injected billions of dollars into Upper Midwest agriculture to incentivize corn and soybean production. A number of conservation policies provide technical and financial assistance for grass-based agriculture and prairie restoration and further training and funding for grazing technical and financial assistance is needed; these changes offer high political feasibility with incremental rather than transformative impacts. Increasing regional meat processing capacity and clarity in grass-based labels would help support supply chains for grass-based milk and meat, which are also politically feasible options. At a deeper structural level, graziers would benefit from policies that address consolidation in the meat and dairy industries and increase access to land, capital, and fair labor to ensure they can steward land environmentally, provide fair wages and working conditions, and earn a profit. Taking these steps would help us transition toward agriculture that better supports farmers, eaters, ecosystems, and rural economies alike.

References

- AGA (2021). *Find a Certified AGA Producer - Wisconsin*. American Grassfed Association. Available online at: <https://www.americangrassfed.org/aga-membership/producer-members/> (accessed June 29, 2022).
- Askew, K. (2022). *US Organic Dairy Squeezed as Prices Fail to Cover Costs: 'There Is No Economic Reason for Dairies to Transition to Organic'*. Dairy Reporter. Available online at: [https://www.dairyreporter.com/Article/2022/06/23/US-organic-dairy-](https://www.dairyreporter.com/Article/2022/06/23/US-organic-dairy-squeezed-as-prices-fail-to-cover-costs-There-is-no-economic-reason-for-dairies-to-transition-to-organic)

Author contributions

AR, EL, and AF conceived the ideas and designed methodology. AF and EL conducted interviews. YL analyzed quantitative data and created figures. AR, EL, AF, RH, and YL researched policy programs. AR, EL, AF, RJ, RH, and YL wrote the manuscript. All authors contributed critically to the drafts and gave final approval for publication.

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Conflict of interest

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Azzam, A., Walters, C., and Kraus, T. (2021). Does subsidized crop insurance affect farm industry structure? Lessons from the U.S. *J. Policy Model.* 43, 1167–1180. doi: 10.1016/j.jpolmod.2021.06.003

- Bardgett, R. D., Bullock, J. M., Lavorel, S., Manning, P., Schaffner, U., Ostle, N., et al. (2021). Combating global grassland degradation. *Nat. Rev. Earth Environ.* 2, 720–735. doi: 10.1038/s43017-021-00207-2
- Barrett, R. (2019). *Dairy in Distress*. Milwaukee Journal Sentinel. Available online at: <https://projects.jsonline.com/topics/dairy-crisis/dairyland-in-distress.html> (accessed June 29, 2022).
- Bay Mills Community College (2022). *Waishkey Bay Farm*. Available online at: <https://bmcc.edu/about-bmcc/community-services/waishkey-bay-farm> (accessed February 1, 2023).
- Baylis, K., Coppess, J., Gramig, B. M., and Sachdeva, P. (2022). Agri-environmental programs in the United States and Canada. *Rev. Environ. Econ. Policy* 16, 83–104. doi: 10.1086/718052
- Bekkerman, A., Balasco, E. J., and Smith, V. H. (2018). Does farm size matter? Distribution of crop insurance subsidies and government program payments across U.S. farms. *Appl. Econ. Perspect. Policy* 41, 498–518. doi: 10.1093/aep/ppy024
- Belk, K. E., Woerner, D. R., Delmore, R. J., Tatum, J. D., Yang, H., and Sofos, J. N. (2014). The meat industry: do we think and behave globally or locally? *Meat Sci.* 98, 556–560. doi: 10.1016/j.meatsci.2014.05.023
- Bendorf, J., Hubbard, J. S., Kucharik, C. J., and VanLoocke, A. (2021). Rapid changes in agricultural land use and hydrology in the Driftless Region. *Agrosyst. Geosci. Environ.* 4:e20214. doi: 10.1002/agg2.20214
- Bengtsson, J., Bullock, J. M., Egoh, B., Everson, C., Everson, T., O'Connor, T., et al. (2019). Grasslands—more important for ecosystem services than you might think. *Ecosphere* 10, e02582. doi: 10.1002/ecs2.2582
- Blanco-Canqui, H. (2022). Cover crops and carbon sequestration: lessons from U.S. studies. *Soil Sci. Soc. Am. J.* 86, 501–519. doi: 10.1002/saj2.20378
- Boyle, J. H., Dalgleish, H. J., and Puzey, J. R. (2019). Monarch butterfly and milkweed declines substantially predate the use of genetically modified crops. *Proc. Natl. Acad. Sci. U. S. A.* 116, 3006–3011. doi: 10.1073/pnas.1811437116
- Burchfield, E. K., B. L., Schumacher, K., and Spangler, and, A., Rissing (2022). The state of US farm operator livelihoods. *Front. Sustain. Food Syst.* 5, 795901. doi: 10.3389/fsufs.2021.795901
- Changnon, S. A., Kunkel, K. E., and Winstanley, D. (2003). Quantification of climate conditions important to the tallgrass prairie. *Trans. Ill. State Acad. Sci.* 96, 41–54. Available online at: <https://ilacadofsci.com/wp-content/uploads/2013/08/096-04MS2216-print.pdf>
- Cochrane, T. S., and Iltis, H. H. (2000). Atlas of the Wisconsin Prairie and Savanna Flora. Technical Bulletin – Department of Natural Resources, Wisconsin, USA (No. 191). p. 226. Available online at: <https://www.cabdirect.org/cabdirect/abstract/20013031207>
- Cox, C. (1991). Pesticides and birds: from DDT to today's poisons. *J. Pestic. Reform* 11, 2–6.
- CSRA Science (2022). *Inside the FY22 Appropriations Package: Wins for Sustainable Agriculture*. Available online at: <https://csrascience.org/inside-the-fy22-appropriations-package-wins-for-sustainable-agriculture-national-sustainable-agriculture-coalition/> (accessed June 25, 2022).
- Environmental Working Group (2020a). *Farm Subsidy Information - Illinois*. Available online at: <https://farm.ewg.org/region.php?fips=17000> (accessed May 23, 2022).
- Environmental Working Group (2020b). *Farm Subsidy Information - Wisconsin*. Available online at: <https://farm.ewg.org/region.php?fips=55000&statename=Wisconsin> (accessed June 29, 2022).
- EPA (2021). *CWSRF Best Practices Guide for Financing Nonpoint Source Solutions*. EPA 841B21012. Environmental Protection Agency. Available online at: <https://www.epa.gov/system/files/documents/2021-12/cwsrf-nps-best-practices-guide.pdf> (accessed March 15, 2022).
- Fairbairn, M. (2020). *Fields of Gold, Financing the Global Land Rush*. Ithaca, NY: Cornell University Press, 1–232. Available online at: <https://ecommons.cornell.edu/bitstream/handle/1813/104007/9781501750106.pdf?sequence=1> (accessed June 29, 2022).
- Farmworker Justice (2022). *Overtime Map*. Available online at: <https://www.farmworkerjustice.org/overtime-map/> (accessed June 25, 2022).
- Foltz, J., and Lang, G. (2005). The adoption and impact of management intensive rotational grazing (MIRG) on Connecticut dairy farms. *Renew. Agric. Food Syst.* 20, 261–266. doi: 10.1079/RAF2005127
- Frank, D. A., and McNaughton, S. J. (1993). Evidence for the promotion of aboveground grassland production by native large herbivores in Yellowstone National Park. *Oecologia* 96, 157–161. doi: 10.1007/BF00317727
- Franzluubbers, A. J., Paine, L. K., Winsten, J. R., Krome, M., Sanderson, M. A., Ogles, K., et al. (2012). Well-managed grazing systems: a forgotten hero of conservation. *J. Soils Water Conserv.* 67, 100A–104A. doi: 10.2489/jswc.67.4.100A
- Frye-Levine, L., Ugoretz, S. J., and Miller, M. (2019). *Milk With Dignity: Worker-Centered Organizing for Social Responsibility*. Available online at: <https://cias.wisc.edu/wp-content/uploads/sites/194/2019/07/MWD.pdf> (accessed June 29, 2022).
- Fuhlendorf, S. D., and Engle, D. M. (2004). Application of the fire-grazing interaction to restore a shifting mosaic on tallgrass prairie. *J. Appl. Ecol.* 41, 604–614. doi: 10.1111/j.0021-8901.2004.00937.x
- Garnett, T., Godde, C., Muller, A., Röss, E., Smith, P., de Boer, I. J. M., et al. (2017). *Grazed and Confused? Ruminating on Cattle, Grazing Systems, Methane, Nitrous Oxide, the Soil Carbon Sequestration Question – and What It All Means for Greenhouse Gas Emissions*. Food Climate Research Network, Oxford, UK, University of Oxford. Available online at: https://www.oxfordmartin.ox.ac.uk/downloads/reports/fcrn_gnc_report.pdf
- Gillon, S., Booth, E. G., and Rissman, A. R. (2016). Shifting drivers and static baselines in environmental governance: challenges for improving and proving water quality outcomes. *Reg. Environ. Change* 16, 759–775. doi: 10.1007/s10113-015-0787-0
- Goulson, D., Nicholls, E., Botiasand, C., and Rotheray, E. L. (2015). Bee declines driven by combined stress from parasites, pesticides, and lack of flowers. *Science* 347, 1255957. doi: 10.1126/science.1255957
- Grassworks (2022). *Grassworks: Your Grazing Resource*. Available online at: <https://grassworks.org/> (accessed November 29, 2022).
- Grazing Public Lands in Wisconsin (2018). *Project Update*. Available online at: <https://grazingpubliclands.wisc.edu/?p=91> (accessed July 15, 2018).
- Hanson, G. D., Ford, S. A., Parsons, R. L., Cunningham, L. C., and Muller, L. D. (1998). Increasing intensity of pasture use with dairy cattle: an economic analysis. *J. Prod. Agric.* 11, 175–179. doi: 10.2134/jpa1998.0175
- Hardy, M. A., Broadway, M. S., Pollentier, C. D., Radeloff, V. C., Riddle, J. D., Hull, S. D., and Zuckerberg, B. (2020). Responses to land cover and grassland management vary across life-history stages for a grassland specialist. *Ecol. Evol.* 10, 12777–12791.
- Haslett-Marroquin, R., and Andreassen, P. (2017). *In the Shadow of Green Man: My Journey from Poverty and Hunger to Food Security and Hope*. Austin, TX: Acres USA Incorporated.
- Hendrickson, J., and Munch, R. (2018). *Livestock Compass*. Madison, WI: University of Wisconsin, Center for Integrated Agricultural Systems. Available online at: <http://www.veggiecompass.com/livestock-compass/> (accessed July 1, 2022).
- Herkert, J. R., Sample, D. W., and Warner, R. E. (1996). “Management of midwestern grassland landscapes for the conservation of migratory birds,” in *Management of Midwestern Landscapes for the Conservation of Neotropical Migratory Birds, General Technical Report NC-187*, ed F. R. Thompson (St. Paul, MN: North Central Forest Experiment Station USDA Forest Service), 89–116. Available online at: <https://www.srs.fs.usda.gov/pubs/10251> (accessed February 10, 2023).
- Hoard's Dairyman (2021). *DairyLivestream: What's the Future for Ag Labor?* Webinar Recording. Available online at: <https://www.youtube.com/watch?v=iee1PSRKiZc> (accessed May 15, 2022).
- Holleman, H. (2017). De-naturalizing ecological disaster: colonialism, racism and the global Dust Bowl of the 1930s. *J. Peas. Stud.* 44, 234–260. doi: 10.1080/03066150.2016.1195375
- Horst, M., and Marion, A. (2019). Racial, ethnic and gender inequities in farmland ownership and farming in the U.S. *Agric. Hum. Values* 36, 1–16. doi: 10.1007/s10460-018-9883-3
- Houser, M., Gunderson, R., Stuart, D., and Denny, R. C. H. (2020). How farmers “repair” the industrial agricultural system. *Agric. Hum. Values* 37, 983–997. doi: 10.1007/s10460-020-10030-y
- Hubbard, T. (2014). “Chapter 13. Buffalo genocide in nineteenth-century North America: ‘Kill, Skin, and Sell,’” in *Colonial Genocide in Indigenous North America*, eds A. Hinton, A. Woolford, and J. Benvenuto (New York, NY: Duke University Press), 292–305.
- Illinois Department of Natural Resources (2022). *Conservation Stewardship Program*. Available online at: <https://www2.illinois.gov/dnr/conservation/CSP/Pages/default.aspx> (accessed June 27, 2022).
- Imhoff, D., and Badaracco, C. (2019). *The Farm Bill - A Citizen's Guide*. Washington, DC: Island Press.
- Indigenous Food and Agriculture Institute (2022). *2018 Farm Bill Implementation Food Distribution Program on Indian Reservations: “638” Self-Governance Demonstration Project*. University of Arizona. Available online at: <https://indigenousfoodandag.com/wp-content/uploads/2020/07/IFAI-FDPIR-638-One-Pager.pdf> (accessed June 27, 2022).
- InterTribal Buffalo Council (2019). *Our History*. Available online at: <https://itbcbuffalonation.org/who-we-are/history/> (accessed June 29, 2022).
- Iowa Department of Revenue (2022). *Iowa Property Tax Overview*. Available online at: <https://tax.iowa.gov/iowa-property-tax-overview> (accessed June 27, 2022).
- Iowa DNR (2013). *Emergency Haying and Grazing on DNR-Managed Land. Conservation and Recreation Division*. Iowa Department of Natural Resources. Available online at: <https://drive.google.com/file/d/1A0-sYm0HA90H-B7prKCK6ingH8X5DDd/view?usp=sharing> (accessed June 29, 2022).
- Isenberg, A. (2000). *The Destruction of the Bison*. Cambridge: Cambridge University.

- Jackson, R. (2020). Soil nitrate leaching under grazed cool season grass pastures of the North Central US. *J. Sci. Food Agri.* 10, 5307–5312.
- Jian, J., Du, X., Reiter, M. S., and Stewart, R. D. (2020). A meta-analysis of global cropland soil carbon changes due to cover cropping. *Soil Biol. Biochem.* 143, 107735. doi: 10.1016/j.soilbio.2020.107735
- Kwon, H. Y., Nkonya, E., Johnson, T., Graw, V., Kato, E., and Kihui, E. (2016). “Global estimates of the impacts of grassland degradation on livestock productivity from 2001 to 2011,” in *Economics of Land Degradation and Improvement – A Global Assessment for Sustainable Development*, eds E. Nkonya, A. Mirzabaev, and J. von Braun (Cham: Springer), 197–214.
- Lark, T. J. (2020). Protecting our prairies: Research and policy actions for conserving America’s grasslands. *Land Use Policy* 97, 104727. doi: 10.1016/j.landusepol.2020.104727
- Lark, T. J., Spawn, S. A., Bougie, M., and Gibbs, H. K. (2020). Cropland expansion in the United States produces marginal yields at high costs to wildlife. *Nat. Commun.* 11, 1–11. doi: 10.1038/s41467-020-18045-z
- Legislative Fiscal Bureau (2019). *Dairy Assistance Programs (Agriculture, Trade, and Consumer Protection)*. Legislative Fiscal Bureau and Joint Committee on Finance. Available online at: https://docs.legis.wisconsin.gov/misc/lfb/jfcmotions/2019/2019_06_11/001_agriculture_trade_and_consumer_protection/002_paper_135_dairy_assistance_program (accessed December 10, 2022).
- Liang, C., Kao-Kniffin, J., Sanford, G. R., Wickings, K., Balser, T. C., and Jackson, R. D. (2016). Microorganisms and their residues under restored perennial grassland communities of varying diversity. *Soil Biol. Biochem.* 103, 192–200. doi: 10.1016/j.soilbio.2016.08.002
- Livanis, G., Moss, C. B., Breneman, V. E., and Nehring, R. F. (2006). Urban sprawl and farmland prices. *Amer. J. Agr. Economics* 88, 915–929. doi: 10.1111/j.1467-8276.2006.00906.x
- Locke, C., Meils, E., and Murray, M. (2016). *The Wisconsin Pollinator Protection Plan. Wisconsin Department of Agriculture, Trade and Consumer Protection*. Available online at: <https://datcp.wi.gov/Documents/PPPComplete.pdf> (accessed December 29, 2022).
- Lowe, E. (2022). *Agroecological Transformations: Pollinators, People, and Power. Publication no: 29213103* (doctoral dissertation). Madison, WI: University of Wisconsin-Madison. ProQuest Dissertations and Theses Global.
- Lowe, E., and Fochesatto, A. (2023). *Just Transitions to Managed Grazing: Needs and Opportunities for Change in the Midwestern United States*. Available online at: https://grasslandag.org/wp-content/uploads/sites/323/2023/01/Just-Transitions-Report_final.pdf (accessed January 1, 2023).
- Lu, C. D., and Miller, B. A. (2019). Current status, challenges and prospects for dairy goat production in the Americas. *Asian Aust. J. Anim. Sci.* 8, 1244–1255. doi: 10.5713/ajas.19.0256
- Lu, Y., and Rissman, A. (2022). “Who pays for the party? Conference sponsor networks in the Food-Energy-Water-Ecosystems Nexus”. In: *Presentation at the International Association for Society and Natural Resources Conference*.
- Mayerfeld, D. (2023). *Our Carbon Hoofprint: the Complex Relationship Between Meat and Climate*. Cham: Springer Nature. doi: 10.1007/978-3-031-09023-3
- Mayerfeld, D., Rickenbach, M., and Rissman, A. (2016). Overcoming history: attitudes of resource professionals and farmers toward silvopasture in southwest Wisconsin. *Agroforest. Syst.* 90, 723–736. doi: 10.1007/s10457-016-9954-7
- MDNR (2021). *Prairies of Minnesota Landowner Handbook*. Minnesota Department of Natural Resources. Available online at: <https://files.dnr.state.mn.us/assistance/backyard/prairierestoration/prairie-handbook.pdf> (accessed July 23, 2022).
- Medina, G., Isley, C., and Arbuckle, J. (2020). Iowa farm environmental leaders’ perspectives on the US farm bill conservation programs. *Front. Sustain. Food Syst.* 4, 497943. doi: 10.3389/fsufs.2020.497943
- Meskwaki Department of Natural Resources (2017). *2017–2027 Program Plan: Sac and Fox Tribe of the Mississippi in Iowa*, 1–139. Available online at: <https://drive.google.com/file/d/1UprFxZ51kTsvulbQ9-7Ewhxg6JIOBXzr/view> (accessed February 10, 2023).
- Michigan Department of Natural Resources (2013). *Managed Public Land Strategy*. Available online at: https://drive.google.com/file/d/1asN_9-0V2sQ-2eyt4u5y3WCruMQ6SaKu/view?usp=sharing (accessed October 11, 2020).
- Michigan Pollinator Protection Plan Steering Committee (2022). *Michigan Managed Pollinator Protection Plan*. Available online at: <https://pollinators.msu.edu/programs/protection-plan/> (accessed December 29, 2022).
- Migrant Justice (2022). *Milk with Dignity Campaign Video*. Available online at: <https://migrantjustice.net/milk-with-dignity-campaign> (accessed June 29, 2022).
- Minnesota Board of Water and Soil Resources (2019). *Pollinator Plan*. Available online at: <https://bwsr.state.mn.us/sites/default/files/2019-01/2019%20Revised%20Pollinator%20Plan%202012-26-18.pdf> (accessed December 29, 2022).
- Minnesota Department of Agriculture (2022). *Conservation Grazing Map*. Available online at: <https://www.mda.state.mn.us/conservation-grazing-map> (accessed June 29, 2022).
- Minnesota Department of Revenue (2018). *Rural Preserve Tax Program, Property Tax Fact Sheet 6*. Available online at: <https://www.co.houston.mn.us/?mdocs-file=2478> (accessed June 27, 2022).
- Minnesota Department of Revenue (2020). *Green Acres and Rural Preserve*. Available online at: <https://www.revenue.state.mn.us/green-acres-and-rural-preserve> (accessed April 29, 2020).
- Minnesota Department of Revenue (2022). *Special Agricultural Homestead*. Available online at: <https://www.revenue.state.mn.us/special-agricultural-homestead> (accessed June 27, 2022).
- Minnesota Grazing Lands Conservation Association (2022). *MN Grazing Lands Conservation Association*. Available online at: <https://www.mnglca.org/> (accessed December 29, 2022).
- Minnesota Prairie Plan Working Group (2018). *Minnesota Prairie Conservation Plan*. Available online at: https://files.dnr.state.mn.us/eco/mcbs/mn_prairie_conservation_plan.pdf (accessed June 27, 2021).
- Missouri Department of Conservation (2022). *Grassland Practices*. Available online at: <https://mdc.mo.gov/your-property/improve-your-property/habitat-management/grassland-management/grassland-practices> (June 27, 2022).
- Missourians for Monarchs Collaborative Steering Committee (2022). *Missouri Monarch and Pollinator Conservation Plan*. Available online at: <https://moformonarchs.org/wp-content/uploads/2022/03/M4M-Plan-0121-FINAL.pdf> (accessed December 29, 2022).
- MOSA (2022). *Grass-Fed Certification*. Midwest Organic Services Association. Available online at: <https://mosaorganic.org/images/documents/Grass-Fed-Certifications.pdf> (accessed January 15, 2022).
- Mulholland, C., Hendrickson, J., Munsch, J., and Barham, B. (2022). *Heifer Grazing Compass University of Wisconsin, Center for Integrated Agricultural Systems, Madison, Wisconsin*. Available online at: <https://cias.wisc.edu/our-work/farming-systems/farm-viability/heifer-grazing-compass/> (accessed July 1, 2022).
- Myers, V. G. (2022). *Beef Producers Gain Ground in Battle for Mandatory Country of Origin Labels*. Progressive Farmer. Available online at: <https://www.dtnpf.com/agriculture/web/ag/news/article/2022/05/01/label-wars> (accessed June 22, 2022).
- NASS (2022). *Acreage*. National Agricultural Statistics Service (NASS), Agricultural Statistics Board, United States Department of Agriculture. Available online at: https://www.nass.usda.gov/Publications/Todays_Reports/reports/acrg0622.pdf (accessed July 1, 2022).
- National Sustainable Agriculture Coalition (2017). *How Farm Subsidies Encourage the Big to Get Bigger*. Available online at: <https://sustainableagriculture.net/blog/farm-subsidies-encourage-big-get-bigger/> (accessed May 15, 2020).
- National Sustainable Agriculture Coalition (2022). *An Economic Analysis of Payment Caps on Crop Insurance Subsidies*. Available online at: <https://sustainableagriculture.net/wp-content/uploads/2022/07/Payment-Limit-Report-FINAL.pdf> (accessed July 30, 2022).
- Nicholson, C., and Stephenson, M. (2021). *Analyses of Proposed Alternative Growth Management Programs for the US Dairy Industry*. Program on Dairy Markets and Policy. Available online at: https://dairymarkets.org/GMP/GMP_Report.pdf (accessed June 30, 2022).
- Oneida Nation (2018). “Chapter 5: farmlands that provide,” in *The Live Sustain Grow Plan*, 78–99. Available online at: <https://oneida-nsn.gov/wp-content/uploads/2018/01/Chapter-5-Farmlands-that-Provide.pdf> (accessed June 30, 2022).
- Pasture Project (2020). *Grazing Public Lands in Wisconsin*. WI-DNR. Available online at: <https://pastureproject.org/wp-content/uploads/2020/07/Grazing-Public-Lands.pdf> (accessed June 30, 2022).
- Progressive Farmer (2022). *Beef Producers Gain Ground in Battle for Mandatory Country of Origin Labels*. Available online at: <https://www.dtnpf.com/agriculture/web/ag/news/article/2022/05/01/label-wars> (accessed June 29, 2022).
- Rabalais, N. N., Turner, R. E., and Wiseman, W. J. (2002). Gulf of Mexico Hypoxia, a.k.a. “The Dead Zone.” *Ann. Rev. Ecol. Syst.* 33, 235–263. doi: 10.1146/annurev.ecolsys.33.010802.150513
- Rajib, M. A., Abiablame, L., and Paul, M. (2016). Modeling the effects of future land use change on water quality under multiple scenarios: a case study of low-input agriculture with hay/pasture production. *Sustain. Water Qual. Ecol.* 8, 50–66. doi: 10.1016/j.swaqe.2016.09.001
- Reimer, A. P., and Prokopy, L. S. (2014). Farmer participation in US Farm Bill conservation programs. *Environ. Manage.* 53, 318–332. doi: 10.1007/s00267-013-0184-8
- Ribic, C. A., and Sample, D. W. (2001). Associations of grassland birds with landscape factors in southern Wisconsin. *Am. Midl. Nat.* 146, 105–121. doi: 10.1674/0003-0031(2001)146[0105:A0GBWL]2.0.CO;2
- Ross, L. (2014). Down on the farm wall street: America’s new farmer, eds A. Mittal, and M. Moore (Oakland, The Oakland Institute), 1–35. Available online at: https://www.oaklandinstitute.org/sites/oaklandinstitute.org/files/OI_Report_Down_on_the_Farm.pdf (accessed June 29, 2022).

- Rui, Y., Jackson, R. D., Cotrufo, M. F., Sanford, G. R., Spiesman, B. J., Deiss, L., et al. (2022). Persistent soil carbon enhanced in Mollisols by well-managed grasslands but not annual grain or dairy forage cropping systems. *Proc. Natl. Acad. Sci. U. S. A.* 119, 2118931119. doi: 10.1073/pnas.2118931119
- Samson, F., and Knopf, F. (1994). Prairie conservation in North America. *Bioscience* 44, 418–421. doi: 10.2307/1312365
- Sanford, G. R., Jackson, R. D., Rui, Y., and Kucharik, C. J. (2022). Land use-land cover gradient demonstrates the importance of perennial grasslands with intact soils for building soil carbon in the fertile Mollisols of the North Central US. *Geoderma* 418, 115854. doi: 10.1016/j.geoderma.2022.115854
- Schnepf, R. (2017). *Farm Safety-Net Payments Under the 2014 Farm Bill: Comparison by Program Crop*. Congressional Research Service Report No. 44914. Available online at: <https://sgp.fas.org/crs/misc/R44914.pdf> (accessed June 30, 2022).
- Schnepf, R. (2021). *Agriculture in the WTO: Rules and Limits on Domestic Support*. Congressional Research Service Report No. 45305. Available online at: <https://sgp.fas.org/crs/row/R45305.pdf> (accessed December 21, 2022).
- Secchi, S., and Babcock, B. A. (2007). Impact of high corn prices on conservation reserve program acreage. *Iowa Ag Rev.* 13, 4–5. Available online at: <https://dr.lib.iastate.edu/handle/20.500.12876/45182>
- Soriano, F. D., Polan, C. E., and Miller, C. N. (2001). Supplementing pasture to lactating Holsteins fed a total mixed ration diet. *J. Dairy Sci.* 84, 2460–2468. doi: 10.3168/jds.S0022-0302(01)74696-6
- Spratt, E., Jordan, J., Winsten, J., Huff, P., van Schaik, C., Jewett, J. G., et al. (2021). Accelerating regenerative grazing to tackle farm, environmental, and societal challenges in the upper Midwest. *J. Soil Water Conserv.* 76, 15A–23A. doi: 10.2489/jswc.2021.1209A
- State of Michigan (2018). *State Tax Commission Qualified Agricultural Property Exemption Guidelines*. Available online at: https://www.michigan.gov/-/media/Project/Websites/taxes/MISC/2005/2005_Qualified_Agricultural_Prop.pdf?rev=8329d8490fd04f81b95c48b5561c8388 (accessed June 30, 2022).
- Stone Barns Center for Food and Agriculture (2017). *Back to Grass: The Market Potential for U.S. Grass-Fed Beef*. Available online at: https://www.stonebarnscenter.org/wp-content/uploads/2017/10/Grassfed_Full_v2.pdf (accessed June 30, 2022).
- Strömberg, C. A. E. (2002). The origin and spread of grass-dominated ecosystems in the late Tertiary of North America: preliminary results concerning the evolution of hypsodonty. *Palaeogeogr. Palaeoclimatol. Palaeoecol.* 177, 59–75. doi: 10.1016/S0031-0182(01)00352-2
- Taxpayers for Common Sense (2022). *Cost of Farm Bills Continues to Skyrocket*. Available online at: <https://taxpayer.net/agriculture/cost-of-farm-bills-continues-to-skyrocket/> (accessed July 6, 2022).
- The White House (2021). *Executive Order on Promoting Competition in the American Economy*. Available online at: <https://www.whitehouse.gov/briefing-room/presidential-actions/2021/07/09/executive-order-on-promoting-competition-in-the-american-economy/> (accessed February 15, 2022).
- Transeau, E. N. (1935). The prairie peninsula. *Ecology* 16, 423–437. doi: 10.2307/1930078
- Tsyunhehkw Agriculture (2019). *Facebook*. Available online at: <https://www.facebook.com/Tsyunhehkw-Agriculture-299051071017373/> (accessed June 14, 2022).
- United States Department of Agriculture (2021). *Pasture, Rangeland, Forage*. United States Department of Agriculture. Available online at: <https://www.rma.usda.gov/en/News-Room/Frequently-Asked-Questions/Pasture-Rangeland-Forage> (accessed June 14, 2022).
- United States Department of Agriculture, Agricultural Marketing Service (2022). *Country of Origin Labeling (COOL)*. Available online at: <https://www.ams.usda.gov/rules-regulations/cool> (accessed June 29, 2022).
- United States Department of Energy (2022). *Renewable Fuel Standard*. United States Department of Energy and Alternative Fuels Data Center. Available online at: <https://afdc.energy.gov/laws/RFS> (accessed February 15, 2022).
- University of Missouri Libraries (2022). *Tribes of Missouri, Ioway Tribe*. Available online at: <https://libraryguides.missouri.edu/nativeamericanstudies/motribes> (accessed December 28, 2022).
- USDA Economic Research Service (2021). *Farm Bill Spending*. Available online at: <https://www.ers.usda.gov/topics/farm-economy/farm-commodity-policy/farm-bill-spending/> (accessed June 14, 2022).
- USDA Farm Service Agency (2022). *Non-emergency Haying and Grazing Conservation Reserve Program*. Available online at: https://www.fsa.usda.gov/Assets/USDA-FSA-Public/usdafiles/FactSheets/fsa_crp_haying_and_grazing_reference_resource_083021.pdf (accessed December 28, 2022).
- USDA National Agricultural Statistics Service (1997). *Census of Agriculture Historical Archive*. Available online at: https://agcensus.library.cornell.edu/census_year/1997-census/ (accessed May 23, 2022).
- USDA National Agricultural Statistics Service (2017a). *Certified Organic Survey – Illinois 2016*. Available online at: https://www.nass.usda.gov/Surveys/Guide_to_NASS_Surveys/Organic_Production/2016_State_Publications/IL.pdf (accessed June 29, 2022).
- USDA National Agricultural Statistics Service (2017b). *Certified Organic Survey – Iowa 2016*. Available online at: https://www.nass.usda.gov/Surveys/Guide_to_NASS_Surveys/Organic_Production/2016_State_Publications/IA.pdf (accessed June 29, 2022).
- USDA National Agricultural Statistics Service (2017c). *Certified Organic Survey – Michigan 2016*. Available online at: https://www.nass.usda.gov/Surveys/Guide_to_NASS_Surveys/Organic_Production/2016_State_Publications/MI.pdf (accessed June 29, 2022).
- USDA National Agricultural Statistics Service (2017d). *Certified Organic Survey – Minnesota 2016*. Available online at: https://www.nass.usda.gov/Surveys/Guide_to_NASS_Surveys/Organic_Production/2016_State_Publications/MN.pdf (accessed June 29, 2022).
- USDA National Agricultural Statistics Service (2017e). *Certified Organic Survey – Missouri 2016*. Available online at: https://www.nass.usda.gov/Surveys/Guide_to_NASS_Surveys/Organic_Production/2016_State_Publications/MO.pdf (accessed June 29, 2022).
- USDA National Agricultural Statistics Service (2017f). *2016 Certified Organic Survey – Wisconsin*. Available online at: https://www.nass.usda.gov/Surveys/Guide_to_NASS_Surveys/Organic_Production/2016_State_Publications/WI.pdf (accessed June 29, 2022).
- USDA Natural Resources Conservation Service (2020a). *Conservation Programs – Wisconsin*. National Planning and Agreements Database. Available online at: http://www.nrcs.usda.gov/Internet/NRCS_RCA/reports/cp_wi.html (accessed September 12, 2022).
- USDA Natural Resources Conservation Service (2020b). *Conservation Programs – Illinois*. National Planning and Agreements Database. Available online at: https://www.nrcs.usda.gov/Internet/NRCS_RCA/reports/cp_il.html (accessed June 12, 2022).
- USDA Natural Resources Conservation Service (2020c). *Conservation Programs – Iowa*. National Planning and Agreements Database. Available online at: https://www.nrcs.usda.gov/Internet/NRCS_RCA/reports/cp_ia.html (accessed June 12, 2022).
- USDA Natural Resources Conservation Service (2020d). *Conservation Programs – Michigan*. National Planning and Agreements Database. Available online at: https://www.nrcs.usda.gov/Internet/NRCS_RCA/reports/cp_mi.html (accessed June 12, 2022).
- USDA Natural Resources Conservation Service (2020e). *Conservation Programs – Minnesota*. National Planning and Agreements Database. Available online at: https://www.nrcs.usda.gov/Internet/NRCS_RCA/reports/cp_mn.html (accessed June 12, 2022).
- USDA Natural Resources Conservation Service (2020f). *Conservation Programs – Missouri*. National Planning and Agreements Database. Available online at: https://www.nrcs.usda.gov/Internet/NRCS_RCA/reports/cp_mo.html (accessed June 12, 2022).
- USDA Natural Resources Conservation Service (2021). *Grassland Reserve Program – Program Report*. RCA Data Viewer. Available online at: http://www.nrcs.usda.gov/Internet/NRCS_RCA/reports/fb08_cp_grp.html (accessed February 15, 2022).
- USDA Natural Resources Conservation Service (2022). *Northern Bobwhite, Grasslands, and Savannas. Working Lands for Wildlife*. Available online at: https://www.nrcs.usda.gov/wps/PA_NRCSConsumption/download/?cid=nrcsprd190246&ext=.pdf (accessed July 6, 2022).
- USDA Risk Management Agency (2022). *Summary of Business Reports*. United States Department of Agriculture, Risk Management Agency. Available online at: <https://www.rma.usda.gov/SummaryOfBusiness> (accessed February 15, 2022).
- USFWS (2019). *Poweshiek skipperling (Oarisma poweshiek) 5-Year Review: Summary and Evaluation*. Bloomington, MN: U.S. Fish and Wildlife Service (USFWS), Midwest Region. Available online at: https://ecos.fws.gov/docs/five_year_review/doc6278.pdf (accessed July 15, 2022).
- WDATCP (2022). *Business Case Study: Brey Cycle Farm, Raising Heifers With Management Intensive Rotational Grazing*. Wisconsin Department of Agriculture, Trade and Consumer Protection. Available online at: <https://uwdiscoveryfarms.org/wp-content/uploads/sites/1255/2022/06/BreyCycleFarmsCaseStudy.pdf> (accessed July 27, 2022).
- Wepking, C., Mackin, H. C., Raff, Z., Shrestha, D., Orfanou, A., Booth, E. G., et al. (2022). Perennial grassland agriculture restores critical ecosystem functions in the US Upper Midwest. *Front. Sustain. Food Syst.* 6, 1010280. doi: 10.3389/fsufs.2022.1010280
- Winkler, K., Fuchs, R., Rounsevell, M., and Herold, M. (2021). Global land use changes are four times greater than previously estimated. *Nat. Commun.* 12, 2501. doi: 10.1038/s41467-021-22702-2
- Wisconsin Department of Natural Resources (2016). *Wisconsin Wildlife Action Plan, 2016-2025*. Available online at: <https://p.widencdn.net/pd77jr/NH0938> (accessed April 25, 2022).

- Wisconsin Public Radio (2010). *Tribal Bison Herd Must Go, Say Ho Chunk*. Available online at: <https://www.wpr.org/listen/516171> (accessed June 29, 2022).
- Wright, C. K., Larson, B., Lark, T. J., and Gibbs, H. K. (2017). Recent grassland losses are concentrated around U.S. ethanol refineries. *Environ. Res. Lett.* 12, 044001. doi: 10.1088/1748-9326/aa6446
- Wu, Z. (2021). *Key Elements of Nutrient Credit Markets: An Empirical Investigation of Wisconsin's Market-Like Phosphorus Control Policy* (dissertation). The University of Wisconsin - Madison. Available online at: <https://www.proquest.com/docview/2532598068?pq-origsite=gscholarandfromopenview=true>
- Zhu, X., Jackson, R. D., DeLucia, E. H., Tiedje, J. M., and Liang, C. (2020). The soil microbial carbon pump: from conceptual insights to empirical assessments. *Glob. Change Biol.* 26, 6032–6039. doi: 10.1111/gcb.15319
- Zontek, K. (2007). *Buffalo Nation: American Indian Efforts to Restore the Bison*. Lincoln: University of Nebraska Press.



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EDITED BY

Jose G. Franco,
Agricultural Research Service (USDA),
United States

REVIEWED BY

Mrill Ingram,
Michael Fields Agricultural Institute,
United States
Mark Liebig,
Agricultural Research Service (USDA),
United States
Nicholas R. Jordan,
Independent Researcher, St. Paul,
MN, United States

*CORRESPONDENCE

Katja Koehler-Cole
✉ kkoehlercole2@unl.edu

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Comparing cover crop research in farmer-led and researcher-led experiments in the Western Corn Belt

Katja Koehler-Cole^{1*}, Andrea Basche², Laura Thompson³ and Jennifer Rees⁴

¹Nebraska Extension, University of Nebraska-Lincoln, Ithaca, NE, United States, ²Department of Agronomy & Horticulture, University of Nebraska-Lincoln, Lincoln, NE, United States, ³Nebraska Extension, University of Nebraska-Lincoln, Falls City, NE, United States, ⁴Nebraska Extension, University of Nebraska-Lincoln, York, NE, United States

Cover crops can mitigate soil degradation and nutrient loss and can be used to achieve continuous living cover in cropping systems, although their adoption in the Western Corn Belt of the United States remains low. It is increasingly recognized that cover crop integration into corn (*Zea mays* L.)-based crop rotations is complex, requiring site and operation specific management. In this review, we compared on-farm, farmer-led field scale trials to researcher-led trials carried out in small plots on University of Nebraska-Lincoln experiment stations. Although there is a range of cover crop research conducted in the state, there is no synthesis of the scope and key results of such efforts. Common cover crop challenges and goals in the state are similar to those reported nationwide; challenges include adequate planting timing, associated costs, and weather, while a top goal of cover crop use is to improve soil health. Farmer-led trials most frequently compared a cover crop to a no-cover crop control, likely reflecting a desire to test a basic design determining site-specific performance. Both researcher-led and farmer-led trials included designs testing cash crop planting timing, while some portion of farmer-led trials tested cover crop seeding rates, which are directly related to reported cover crop challenges. Farmer-led trials were carried out on a greater variety of soils, including sandy soils, whereas sandy soils were absent from researcher-led trials. More than half of farmer-led experiments were conducted on fields with slopes of 6–17% while most researcher-led experiments were conducted on fields with slopes of <1%. Mean cover crop biomass production was 600 kg/ha in farmer-led and 2,000 kg/ha in researcher-led trials. Crop yields were not significantly affected by cover crops in either farmer-led or researcher-led trials. Such comparisons demonstrate that in some instances, cover crop research is addressing challenges, and in some instances, it could be expanded. This synthesis expands our knowledge base in a way that can promote co-learning between different scales of experiments, and ultimately, reduce risks associated with cover crop management and further promote continuous living cover of agricultural landscapes.

KEYWORDS

field scale, experiment station, small plots, cover crops, cereal rye, mix cover crop

1. Introduction

Replacing fallow periods with a cover crop is one strategy toward continual living cover of the soil garnering significant recent attention, including investment from government and private-industry initiatives (Basche et al., 2020; Wallander et al., 2021) as well as expansive on-farm research initiatives (Bowman et al., 2022; Practical Farmers of Iowa, 2022). Cover crop research finds that replacing fallow periods improves a wide range of soil health and agronomic indicators, even after just a few years, including quantifiable increases to properties such as aggregation, infiltration, as well as reduced erosion, runoff, weed biomass, and enhanced nutrient cycling (Stewart et al., 2018; Nichols et al., 2020). However, cover crops are still only grown on approximately 3–4% of the cropland acres across leading commodity crop producing states such as Iowa, Illinois, and Nebraska (USDA-NASS, 2017). Researchers investigating the lack of adoption have focused on perceived biological, technical, or economic barriers to cover crops (Arbuckle and Roesch-McNally, 2015; Roesch-McNally et al., 2018). Successful adopters of cover crops often describe a more systems-based approach to soil health and crop management in general that accounts for other functions such as weed suppression, forage production and soil fertility (Church et al., 2020). However, success with cover crops also requires intentional shifts in multiple elements of cash crop management to optimize their benefits (Basche and Roesch-McNally, 2017). Overall, effective integration of cover crops is complex, requiring site and operation-specific adaptations.

The state of Nebraska, located in the Western Corn Belt and in the Northern Great Plains, is an especially useful region to understand cover crop use and adoption. The state contains climatic diversity from humid or semi-humid conditions in the southeast (approximately 850 mm annual rainfall) to semi-arid conditions in the west (approximately 400 mm annual rainfall), which is also represented in its commodities and cropping systems (Zomer et al., 2008; HPRCC, 2022). Nebraska is a top producing state for several major commodities in the United States including corn (*Zea mays* L.), soybean [*Glycine max* (L.) Merr.], cattle, and contains significant crop acreage for wheat (*Triticum aestivum* L.), alfalfa (*Medicago sativa* L.), and sorghum [*Sorghum bicolor* (L.) Moench] (USDA-NASS, 2021). The state also has more irrigated cropland acres than any other in the U.S. and irrigation is utilized on approximately one-third of harvested acreage (USDA-NASS, 2017). The propensity to livestock in Nebraska, the range of cropping systems and climatic regions, as well as its significant acreage utilizing irrigation suggests that many different agricultural regions of the U.S. might draw parallels from the cover crop research conducted in the state. Notably, a recent survey of producers, consultants, and agricultural researchers found that the three greatest challenges to cover crop adoption in the state of Nebraska are (1) the short window of time between cash crop harvest and cover crop planting; (2) input costs including the cost of cover crop seeding; and (3) weather issues (Das et al., 2022). Similar challenges have been reported by other Nebraska producers (Oliveira et al., 2019) and nationwide (Myers and Watts, 2015).

Decision-making processes in agriculture are not only based on biological and economic factors, but also social, cultural,

relational, and value-driven influences (Prokopy et al., 2008, 2019; Carlisle, 2016). The transfer of knowledge and innovative practices is enhanced in learning environments that provide in-group communication, community support and trusting relationships (Wick et al., 2019; Charatsari et al., 2020). A unique form of such learning environments are on-farm trials, where organizations with research capacity and expertise, including non-profit organizations or Land Grant Universities (i.e., extension educators or university researchers) collaborate with farmers to address specific research questions on the farmer's land. Recently, global networks of on-farm research practitioners have recognized the transformative value of this model of research and outreach to merge experiences, drive innovation, advance technology adoption, while improving profitability and environmental stewardship (Lacoste et al., 2022).

In Nebraska, an on-farm research program organized by the University of Nebraska Extension began in 1990 with a group of farmers in Eastern Nebraska and expanded in the early 2010s to include state-wide trials (Thompson et al., 2019). Trials are co-developed by farmers, University extension educators or researchers and sometimes other stakeholders such as Natural Resource Districts; and are motivated by a shared goal to address a specific research question. They are farmer-led in the sense that farmers manage the trials using their own equipment in large plots in their fields. In contrast, agronomic trials led by researchers at the University of Nebraska-Lincoln's agricultural experiment stations typically have small plots and do not involve producers, however they may also be informed by stakeholder involvement.

Producers often view small plot studies as less reliable than large scale or on-farm studies because they perceive small plot studies to be less representative on actual farm operations (Laurent et al., 2022). In contrast, interviews with participants in Nebraska's on-farm research program found that most trusted the results from their own studies and more than 50% of producers had made changes in their operation due to the study results (Thompson et al., 2019). However, most on-farm study findings from Nebraska have not been published, except at the local level. Making this information available to regional and national audiences could support knowledge sharing with the potential to increase adoption of cover crops. Including findings from on-farm or farmer-led studies in the scientific literature could also lead to a more comprehensive, nuanced view of cover crops than relying on researcher-led studies alone. For example, insight into which cover crop practices have been tested on farms could provide information for researchers to either further test promising practices or test alternatives. Additionally, evaluating the breadth of research in Nebraska, both farmer-led and researcher-led, can help determine how adequate ongoing research efforts are to address cover crop related challenges in the state across a range of climate conditions and cropping systems.

The objectives for our study were to compare farmer-led and researcher-led cover crop experiments from Nebraska, to identify similarities and differences in treatments evaluated, environments assessed as well as cover crop outcomes. We selected two outcomes, cover crop biomass and cash crop yield, as these are widely used indicators for agronomic performance and reported in most studies. This information can support addressing farmers' needs, informing objectives for future studies, and promoting

conservation practices that seek to increase continuous living cover in annual crop rotations. The unique, coordinated, and extensive database for on-farm research and reporting *via* the On-Farm Research Network lends itself well to a comparison with researcher-led trials. With this analysis we wanted to address the following research questions: (1) How do farmer-led and researcher-led cover crop experiments compare in terms of treatments evaluated and environments assessed? (2) How do farmer-led and researcher-led cover crop experiments compare in terms of management and outcomes such as cover crop biomass and yield impacts? In answering these questions, our work fills an important knowledge gap of strategically comparing researcher-led and farmer-led cover crop research to build a knowledge base that potentially reduces risks associated with cover crops and ultimately supports continuous living cover systems at a broader scale.

2. Methods and materials

2.1. Trial compilation

We built our database from two primary sources: The Nebraska On-Farm Research Network for farmer-led experiments and Web of Science for the researcher-led experiments in the state of Nebraska. The Nebraska On-Farm Research Network is the University of Nebraska Extension's on-farm research program (Nebraska On-Farm Research Network, 2022a,b). The program was initiated in 1990 with a group of farmers in Eastern Nebraska and has since expanded across the state. The on-farm trials are initiated either by farmers, researchers, and/or other stakeholders, or typically some combination thereof. Experiments are implemented on farmer's fields using their equipment and labor. University extension educators and researchers assist with trial design, data collection and data analysis (Thompson et al., 2019). Treatments in these trials reflect what farmers want to compare which does not always include a control or check plot, however, in some cases participating researchers may suggest or select treatments. The experimental design in these studies is randomized complete blocks with at least 3 replications or paired comparison designs with at least 5 replications. The plots are usually large, at least the width of the harvest equipment (often around 12 m) and are at least 100-m long to obtain an accurate estimate from the combine yield monitor. The large plot size sets them apart from the small plot studies found at experimental stations, which typically measure 6 × 10 m. Management information and experiment data are gathered from the farmers or university personnel collaborating with the farmers. Researchers or extension educators working with the On-Farm Research Network carry out the statistical analysis and write an annual report. The current On-Farm Research Network database (<https://on-farm-research.unl.edu/farm-research-results>) includes annual reports detailing experimental design, site and management information, measurements, statistical analysis, and results. Yield results are always included in on-farm reports, but often no other data are measured.

We carried out our search of the Nebraska On-Farm Research Network in March of 2022. To capture all types of cover crops, including green manures, we used the keywords of "cover crop", "green manure", and "catch crop". The latter two key words did

not return any entries. The key words "cover crop" resulted in 96 entries, each representing 1 year of a study at one site (field), with study years ranging from 2004 to 2020. From these 96 entries, we selected only studies that had a report and where the cover crop was grown in the same or the year before data was reported. We excluded 19 studies because they did not contain a cover crop and a no-cover crop control (check) treatment as an important goal of this work was to compare yield outcomes which could not be done for experiments without check treatments. Since many trials had more than two treatment comparisons (i.e. cover crop A vs. no cover crop; cover crop B vs. no-cover crop) a total of 89 site-year by treatment comparisons were included in the analysis.

We searched Web of Science for researcher-led, peer-reviewed publications, using the topic "cover crop*" and 1990–2020 (year published) and University of Nebraska Lincoln (affiliation). This returned 114 results, including studies that investigated green manures. To access publications by researchers affiliated with USDA-ARS, a second search with the topic "cover crop*" and 1990–2020 (year published) and United States Department of Agriculture (USDA) (affiliation) and Nebraska (all fields) was carried out, with 44 results, some of which were also returned in the first search. From these two searches, we selected publications reporting field trials in Nebraska (modeling studies or literature reviews were excluded), had replicated and randomized designs, compared the cover crop treatment(s) to a control (no cover crop) treatment, and reported cash crop grain yield and cover crop biomass data. Based on these selection criteria, nine studies were included in the analysis and can be found in Table 1. Although one of these experiments was conducted on a commercial farm, we considered these experiments to be primarily led by researchers given their inclusion in the peer-reviewed literature, although it is possible their designs were informed through partnership with farmers. The researcher-led studies included at least two sites and 2 years per site and often compared several cover crop treatments to a no-cover crop treatment. Thus, the researcher-led studies represent 290 individual site-year by treatment comparisons.

2.2. Database development

We categorized experiments based on their treatments (i.e., comparisons of cover crop species or termination methods) and management (i.e., crop rotations, cover crop species). We categorized crop rotations into the following groups: corn-soybean (where the cover crop is planted following a corn crop and the soybean is planted following the cover crop), continuous corn, small grains such as wheat or rye (*Secale cereale* L.) in rotation (uniquely counted even if rotation included corn or soybean), or other cash crops. We grouped cover crops by plant family including grasses, legumes, brassicas, or mixtures (any cover crop with more than one species present). We further extracted site-specific information on environments such as soils, field topography (slope), location and irrigation (yes/no). Locations were categorized according to the nine NOAA Climate Divisions within the state (NOAA, 2022).

To determine experimental outcomes, we extracted the cash crop yield and cover crop biomass data for each site-year. In

TABLE 1 List of researcher-led, peer-reviewed publications included in the database.

References	Crop rotation including cover crop species
Blanco-Canqui et al. (2017)	Continuous corn, cereal rye winter cover crop*
Kessavalou and Walters (1999)	Corn-soybean, Continuous corn, cereal rye winter cover crop
Koehler-Cole et al. (2017)	Soybean-winter wheat-corn, spring planted red and white clover cover crops
Koehler-Cole et al. (2020)	Corn-soybean, cereal rye winter cover crop and mixture cover crop of cereal rye, forage radish, hairy vetch, and winter pea
Nielsen et al. (2016)	Proso millet, spring cover crop of flax, oat, pea, rapeseed or mixture, winter wheat
Power et al. (1991)	Continuous corn, hairy vetch winter cover crop
Ruis et al. (2017)	Continuous corn, cereal rye winter cover crop
Williams et al. (2000)	Corn silage-soybean; barley, cereal rye, winter wheat, winter triticale, hairy vetch winter cover crops
Wortman et al. (2012)	Sunflower-soybean-corn; two-, four-, six-, eight-way mixture of spring planted cover crops including hairy vetch, buckwheat, mustards, field pea, radish, crimson clover, rape and chickling vetch

*Experiment conducted at on-farm location.

researcher-led trials, corn yield data was determined using plot combines that harvested the central two or three rows of each plot. In on-farm studies, plot yield is determined using a yield monitor on a full-size combine or a weigh wagon (Thompson, 2022, personal communication). Yields were adjusted to 15.5% moisture for corn, 13% moisture for soybean, and 13.5% moisture for wheat or rye. All researcher-led studies included cover crop biomass measurements compared to approximately half of the farmer-led experiments. Cover crop biomass in researcher-led trials was measured by cutting above-ground biomass in a known area, often a 0.3×1.5 m frame, drying the biomass in a forced air oven, and weighing the dried biomass. In farmer-led trials biomass was collected in a similar way, although it may have been air-dried instead of oven-dried. Biomass data was converted to kg/ha. We do not report other data collected from these experiments (such as soil health measurements) because such data were very limited, and the focus of our analysis was on comparing treatments, site-specific conditions, management, as well as yield and biomass outcomes between researcher-led and farmer-led trials.

Variables could have one or multiple observations, for example in farmer-led trials, the variable “location” had only one observation per study, but each researcher-led study could have two or more locations. We counted observations and presented them as percent totals for both farmer-led and researcher-led experiments. Where multiple observations or no information was included (such as two soil types at one location, or no soil type or slope given), the percent total represents the total number of sites or observations reporting information. Not all experiments could be categorized for all information due to incomplete data reporting.

2.3. Statistical analysis of cash crop impacts

To evaluate the effect of cover crops on cash crop yields in both types of experiments, we calculated response ratios for each site-experiment year that included yield information. The response ratio represents the natural log of the yield of the cash crop following a cover crop divided by the yield of the cash crop in the control treatment, a common metric utilized to compare results from different studies (Hedges et al., 1999). To calculate

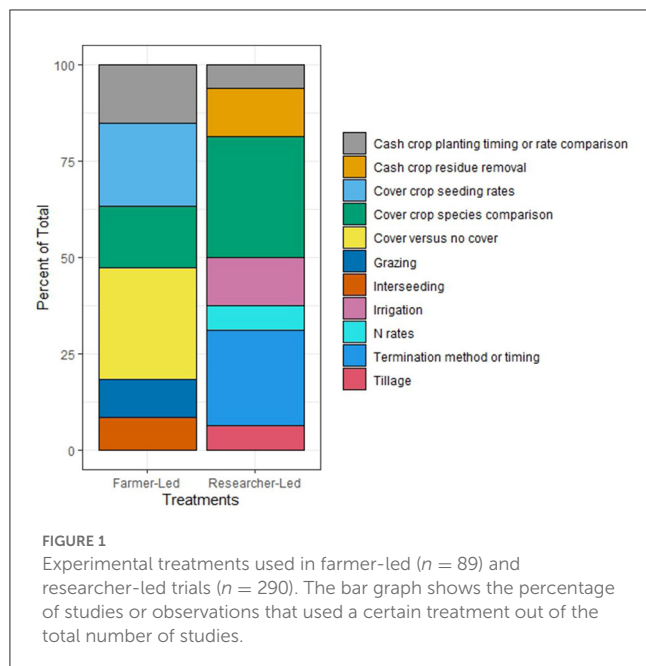
cover crop yield effects across experiments, we considered the effect of location as a random variable to account for similarities in each environment (St. Pierre, 2001). Studies were weighted by the number of experimental replications in the statistical model (Adams et al., 1997). Yield changes were back-transformed from the natural log and converted to percent changes to aid in interpretation of results.

3. Results and discussion

3.1. Treatments evaluated

A range of treatments were included in both the farmer-led and researcher-led experiments, from which we can infer goals of the trials (Figure 1). The most common treatment included in the researcher-led experiments was cover crop species comparisons (31%), while the most common treatment for the farmer-led experiments was cover crop compared to a no cover crop control (29%). Farmer-led experiments also compared cover crop species, but to a lesser extent (16%). Both types of experiments included trials evaluating cash crop planting timing and planting rates. Farmer-led experiments involved a diverse range of treatments, which included grazing (6%), cover crop seeding rates (12%), and interseeding cover crops (12%) (Figure 1). Researcher-led experiments included a divergent range of treatments evaluated, including tillage (6%), N rates (6%), irrigation (13%), and residue removal (13%), none of which were explicitly a part of any on-farm experiments in our database. Treatments evaluating irrigation, residue removal or tillage, for example, may not be as practical to conduct at the scale of a commercial farm as they might be on a smaller experiment scale.

The survey and interview work in the state provides insight into producer challenges and goals related to cover crops which can inform how well cover crop research is designed to address such goals and challenges. Important cover crop challenges reported in Nebraska were the short window of time for cover crop growth, cover crop input costs, and weather issues (Oliveira et al., 2019; Das et al., 2022). Both researcher-led and farmer-led trials included designs testing cash crop planting timing, while some farmer-led



trials tested cover crop seeding rates, which are directly related to these challenges.

Treatments focused on comparing cover crops vs. controls, as well as seeding rates, were included at higher percentages in on-farm experiments and could reflect the commonly cited goal of increasing efficiency and reducing costs for farmers participating in trials (Thompson et al., 2019). Managing input costs may have been the justification for seeding rate studies, while weather issues, in particular cold winters, are likely the rationale for interseeding, cover crop species comparisons as well as cash crop planting timing experiments. In general, however, we might assume that the large portion of farmer-led trials testing cover crop vs. no cover crop comparison are aimed at a central goal of determining cover crop performance on their specific farms.

3.2. Crop rotations including cover crops

The predominant cropping system for the farmer-led experiments was corn-soybean (77%), while researcher-led experiments were balanced between continuous corn and corn-soybean cropping systems (33% each) (Figure 2). Small grain crops such as wheat were included in both types of experiments (17% of researcher-led and 14% of farmer-led experiments), as were other cash crops including alfalfa, sunflower (*Helianthus annuus* L.), and proso millet (*Panicum miliaceum*) (17% of researcher-led and 4% of farmer-led). The most utilized cover crops in both types of experiments were grasses, representing 43% of researcher experiments and 65% of on-farm experiments. On-farm experiments were more likely to include mixtures (33%) compared to researcher experiments (21%). None of the on-farm experiments reported individually evaluating brassicas and only a limited few worked with monoculture legumes, while monoculture legumes and brassicas were included in researcher-led experiments

(Figure 2). Such cropping system patterns broadly align with the major field crops grown in the state, including corn (44% of harvested cropland), soybean (24% of harvested cropland), and wheat and alfalfa (4% each of harvested cropland) (USDA-NASS, 2022).

A survey of Nebraska producers conducted in 2014 found that the most frequently selected objectives of cover crop use were related to soil health—specifically, soil organic matter, soil erosion, and soil water holding capacity—while forage production was the fourth most common objective (Drewnoski et al., 2015). We might assume that including cover crops mixtures in farmer-led experiments are intended to meet soil health goals, few of these report data beyond cover crop biomass and yield. Although reporting on other outcomes (i.e., soil properties measured) was outside the scope of our study, there are initiatives within the state assessing and finding soil health improvements at cover crop on-farm experiments (Krupek et al., 2022a,b).

3.3. Experiment environments

The predominant region for both types of experiments was East Central Nebraska, representing 67% of researcher experiments and 75% of on-farm experiments. Remaining experiments were evenly distributed across the Southeast, Northeast, Central, South Central and Panhandle regions of the state (Figure 2). This is partly due to the distribution of farms across the state, with more, but smaller farms in the Eastern regions; and a greater proportion of pastureland in the western regions (USDA-NASS, 2021). In the more arid western regions of the state, perceived or reported negative cash crop effects of cover crops due to their water use (Nielsen et al., 2016) could limit research efforts.

Silty clay loam and silty loam are common soils in Eastern Nebraska and were represented in both kinds of trials. In contrast, sandy soils were not found in any researcher-led trials but comprised about 27% of the soils in farmer-led trials. Farmer-led experiments were more likely to be conducted on fields with greater slopes; approximately 54% of soil types reported in the on-farm experiments had 6 to 17% slopes. Most researcher-led experiment fields were relatively limited in topography, with 60% having <1% slopes and all with a maximum slope of 6% (Figure 2).

Research stations in Nebraska are mostly located on sites with fine-textured soils and little to no slope which has implications for soil health and plant productivity. More representative results on cover crop growth and effects on soil health and crop yields are obtained by including farmer-led trials in statistical analysis and subsequent management recommendations. In this context, farmer-led trials complement those led by researchers in painting a more realistic picture of opportunities and challenges associated with cover crops in this region, particularly for those grown under less optimal conditions (Laurent et al., 2022).

Irrigation was present on about half of the fields for the farmer-led and approximately 29% of the fields for the researcher-led experiments, closer to the state average of approximately 35% of farms with irrigation (USDA-NASS, 2017). Further, two of the nine peer-reviewed studies included experiments on both irrigated and non-irrigated sites (Figure 2).

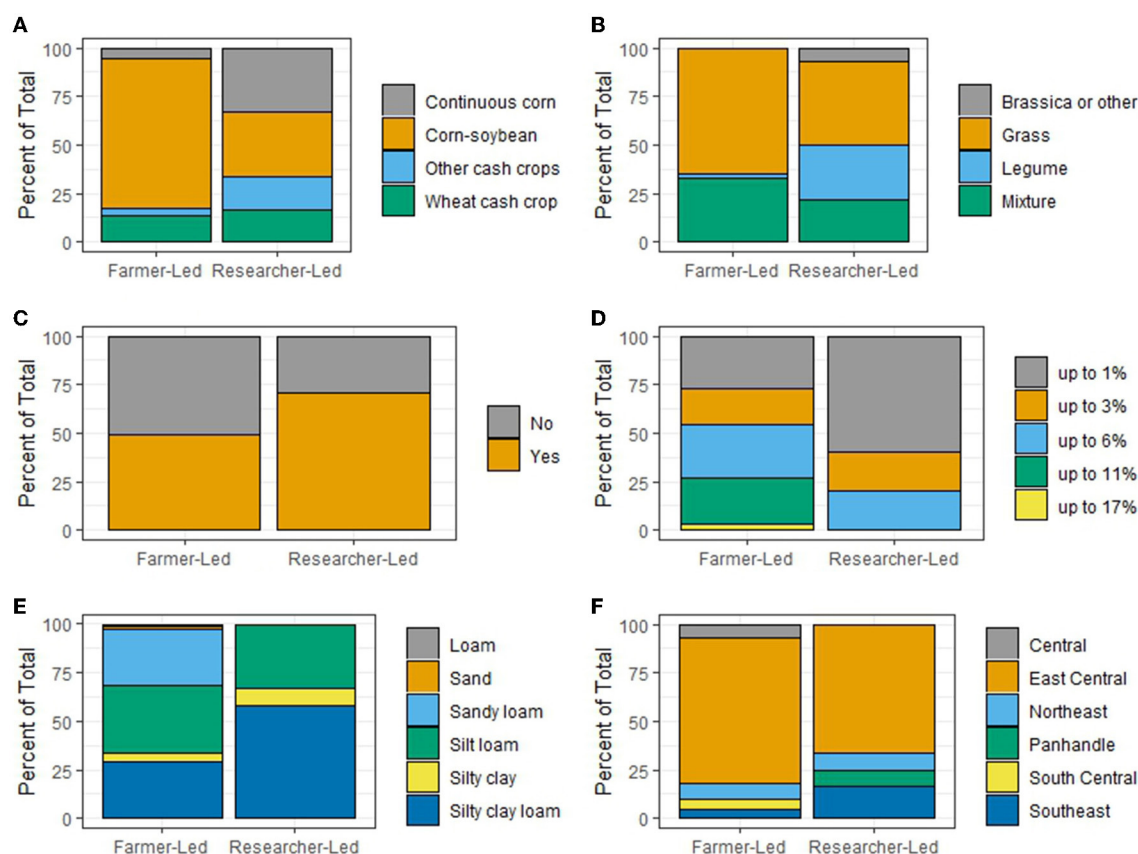


FIGURE 2

Management and environment information [(A) crop rotation, (B) cover crop species, (C) irrigation, (D) field slope, (E) soil type, and (F) Nebraska region] in farmer-led ($n = 89$) and researcher-led ($n = 290$) trials. The bar graph shows the percentage of studies or observations out of the total number of studies reporting information for each variable.

In general, we found a greater variety of environments (soils, climate regions) represented in the farmer-led compared to researcher-led experiments. This further emphasizes the value of on-farm experimentation in a state with a diverse environment such as Nebraska to test and validate management systems, and to demonstrate efficacy of practices under more variable (i.e., greater slopes, lesser soil quality) and potentially more challenging growing conditions. Although we might expect to find that researcher-led trials are more frequently conducted on homogeneous fields, such experiments can allow for studying management or collecting detailed data that would be difficult to do at a larger scale. Additionally, comparing the types of experiments and goals at these different scales can allow for reciprocal exchange of information—testing what has proven effective at a smaller scale on a larger scale, and vice versa, informing smaller scale research based on farmer interest.

3.4. Cash crop yields after cover crops

Yield differences due to cover crops appeared smaller in farmer-led than in researcher-led trials. In farmer-led experiments, we calculated an average yield decline of 3.4% occurred across all cash crops (standard error 11%) while in researcher-led trials,

we calculated an average yield decline of 7.0% (standard error 5.6%) (Figure 3). However, neither of these differences were statistically different from zero. Laurent et al. (2022) similarly found few differences in crop yields when comparing small-plot trials to on-farm fungicide trials. In general, this trend of cash crop yield variability mirrors other studies which have found that grass cover crops can slightly decrease corn yields while legumes and mixes lead to neutral to positive impacts in corn (Miguez and Bollero, 2005; Marcillo and Miguez, 2017). Interestingly, Marcillo and Miguez (2017) found that yield declines in peer-reviewed experiments with corn following cover crops decreased in time, representing the learning curve expressed by farmers (Roesch-McNally et al., 2018). However, farmer self-reported data notes that cover crops consistently lead to cash crop yield improvements (CTIC, 2017), which aligns with the lower yield variability on-farm experiments compared to researcher-led experiments. This could also be a result of the fact that experiment stations often design trials in a factorial manner, vs. more of a “systems approach”, where farmers alter several aspects of management concurrently (Basche and Roesch-McNally, 2017; Church et al., 2020). Our analysis is unique in its inclusion both of on-farm (farmer-led) and experiment-station (researcher-led) studies; publication bias is often a concern in meta-analyses and systematic reviews (Philibert et al., 2012). While our analysis is

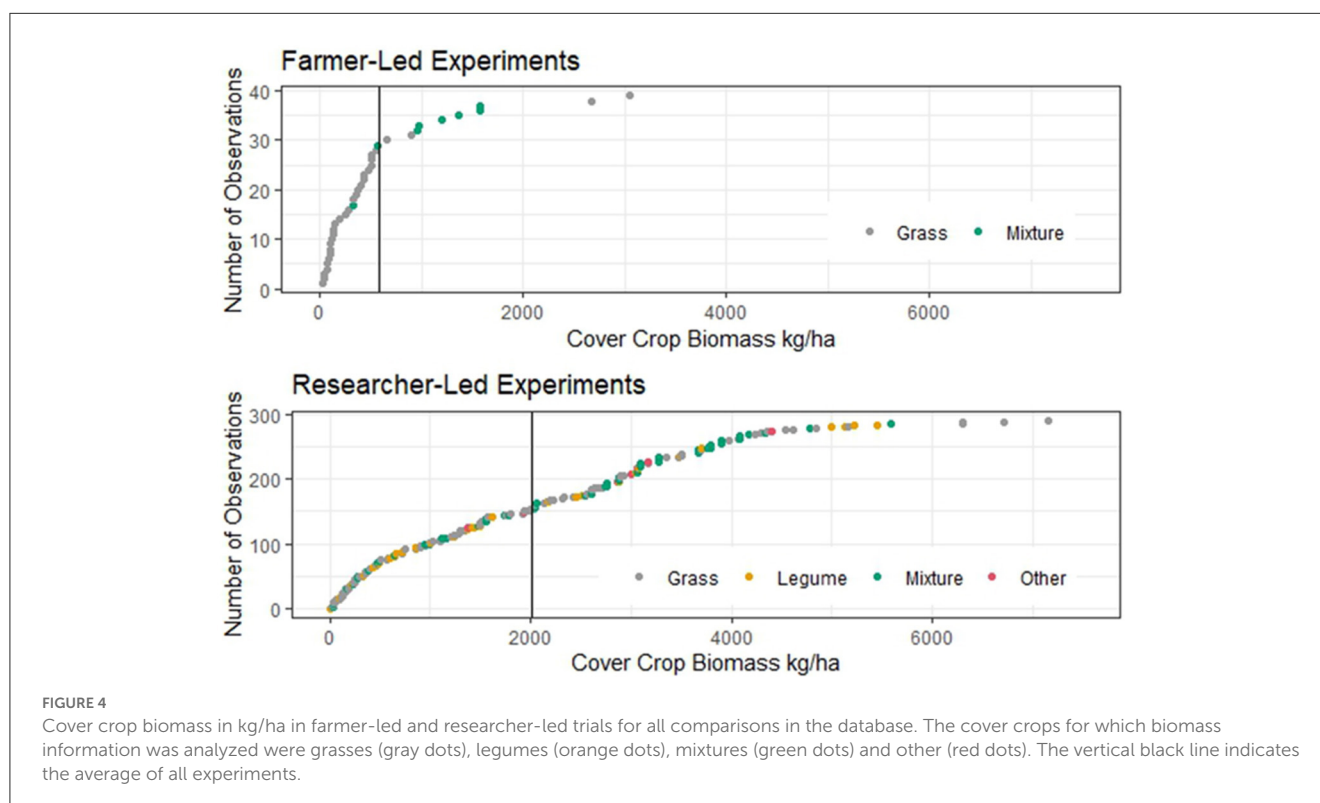
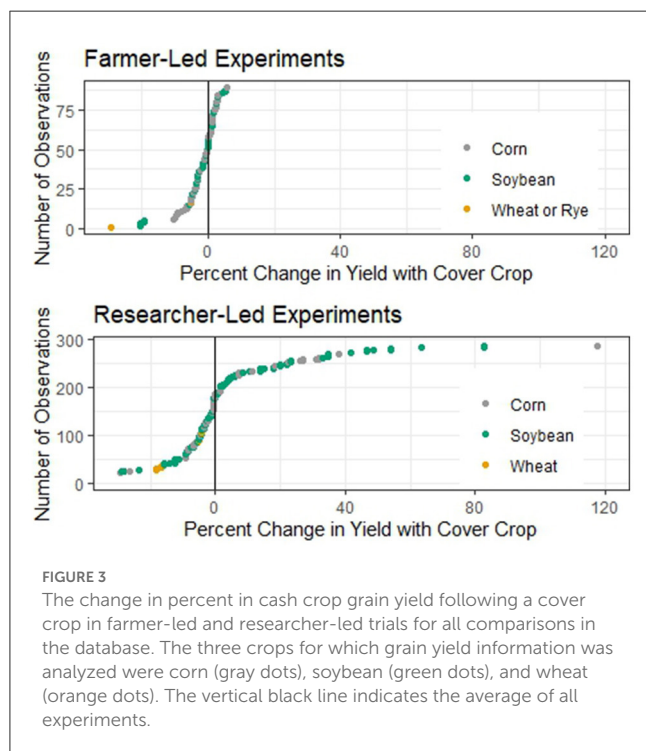
not purely reflective of either methodology, comparing farmer-led experiments published in reports vs. researcher-led experiments from peer-reviewed literature, provides insight into differences of scale, goals, management, and resulting outcomes. This can contribute to improving farmer trust and confidence in alternative

management that is tailored more specifically to their operations, and importantly, that such management can be profitable (Nilsen, 2010; SARE, 2017; Kyveryga, 2019).

3.5. Cover crop biomass

Cover crop biomass ranged from 31 to 3,054 kg/ha in farmer-led experiments with an average of 582 kg/ha (from 39 site-experiment years reporting cover crop biomass data) and from 9 kg/ha to 7,160 kg/ha at the researcher-led experiments with an average of 2,009 kg/ha (from all 290 site-experiment years included in the database) (Figure 4). These values are within the lower end of the range reported in a global assessment of cover crop biomass for semi-arid and cold climates most reflective of the state of Nebraska (annual precipitation <750 mm, USDA Plant Hardiness Zone <5), where mean cover crop biomass was estimated at $2,610 \pm 2,420$ Mg (Ruis et al., 2019). For the experiments in our database, there were several that found cover crop mixtures to have greater biomass than some of the grass only species experiments. Researcher-led experiments were more likely to include and report biomass for legume or other species (brassica, linaceae), which followed a similar distribution of biomass values compared to grass species (Figure 4).

Farmers manage their cover crops as part of profit-oriented system whereas researchers manage their cover crop to test a hypothesis. Farmers may terminate cover crops early to maximize the cash crop growing season, use cash crops with a long maturity group, and/or plant cover crops only after all cash crops on their operation are harvested. These practices shorten the available time



for cover crop growth and could explain the lower productivity in on-farm trials. Previous researcher-led studies in the Western Corn Belt emphasized the need to establish cover crops earlier to increase productivity for example by interseeding the cover crop into cash crop stands (Peterson et al., 2019; Ruis et al., 2019). Interestingly, while several on-farm trials have tested interseeding (Figure 1), at this time there is a lack of researcher-led, peer-reviewed Nebraska studies on this topic. Additionally, it could be that the less optimal environments found in on-farm experiments account for some of the lower cover crop biomass performance. This illustrates the complementary role farmer-led studies have in testing and refining innovative or emerging technologies. When producers and researchers collaborate, results from farmer-led studies can be peer-reviewed and published, extending the findings to a larger audience, and creating the opportunity for wider trust and acceptance of results.

3.6. Study limitations

Our database was limited by the desire to comprehensively assess cover crop outcomes in different scales of experiments for one important and diverse U.S. state. Our database was also limited by differences in reporting across farmer-led and researcher-led trials. We selected experiments that measured and reported cash crop yields and/or cover crop biomass and compared it to a no-cover crop control treatment. Our inclusion criteria (namely that a no-cover crop control and cash crop yields were requirements) resulted in the exclusion of several researcher-led and farmer-led studies that are not counted in terms of their experimental designs and environments. We realize that the exclusion of cover crop studies investigating research questions that do not necessitate a no-cover control may not have captured the breadth of cover crop studies conducted in Nebraska. We also recognize that all studies cannot measure or focus on each potential crop, soil, or other impact of cover crops. However, because our goal was to concurrently compare treatments, environments, and outcomes of cover crop experiments at two different scales, not all potential experiments fulfilled our database criteria. Regardless, this analysis includes 89 site-year by treatment farmer-led and 290 site-year by treatment researcher-led comparisons, representing a robust database that captures trends from across the state of Nebraska.

A related limitation is that objectives were rarely stated in on-farm studies, so we do not know what specific purpose cover crops were to fulfill, beyond our classification of treatments included. Farmers may target a specific area of their field for cover crops, for example to prevent erosion on a slope. This may have influenced how they managed their cover crops, impacting biomass and crop yields. In addition, data collection also differed between the two sets of studies, especially for yields. The considerably larger plot size of farmer-led trials may have reduced overall yield variability, suggesting that treatment differences may be easier to detect in on-farm studies (Laurent et al., 2022).

Despite the diversity of cropping systems and climates in Nebraska, the majority of both types of experiments were concentrated in the relatively wetter East Central region of the state. Cover crop research in drier environments can further inform management to reduce water-related risks often reported by

producers. For example, a more recent study that was not included in our review, suggested that non-winter hardy small grains may be a more productive, yet less water-intensive cover crop for Central and Western Nebraska than cereal rye (Rosa et al., 2021).

4. Conclusion

Although farmer-led and researcher-led cover crop trials differed with respect to treatments and management, we found many similarities between the two types of experiments. Cover crops did not significantly increase or decrease cash crop yields. We found that yield variability was lower at farmer-led compared to researcher-led experiments. Researcher-led experiments on average produced more cover crop biomass and included more brassica and legume cover crops, whereas farmer-led experiments included more mixtures. Farmer-led experiments were more likely to occur in a range of environmental conditions, across more variable landscapes and in some instances on soils of inherently lower productivity. Farmers may have multiple goals for cover crops, including forage for livestock, that may be more complex to conduct on a smaller scale. Conversely, researcher-led experiments assessed treatments such as irrigation, tillage and residue removal that are more complex or not possible to conduct at a larger scale. Identifying crop rotations, cover crop species and cultivars adapted to local soils and climates will be important to achieve continuous living cover in Nebraska's diverse cropping systems. Farmer-led trials due to their greater diversity in local soils and climates can play an important role in this endeavor. Future research should ensure greater representation of environmental conditions and agronomic systems, for example by including more farmer-led trials in research publications and greater collaboration between farmers and researchers.

Author contributions

KK-C and AB: conceptualization, research design, analysis, and writing manuscript. LT and JR: data collection and editing manuscript. All authors contributed to the article and approved the submitted version.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

The Supplementary Material for this article can be found online at: <https://www.frontiersin.org/articles/10.3389/fsufs.2023.1064251/full#supplementary-material>

References

- Adams, D. C., Gurevitch, J., and Rosenberg, M. S. (1997). Resampling tests for meta-analysis of ecological data. *Ecology* 78, 1277–1283. doi: 10.1890/0012-9658(1997)078[1277:RTFMAO]2.0.CO;2
- Arbuckle, J. G., and Roesch-McNally, G. (2015). Cover crop adoption in Iowa: The role of perceived practice characteristics. *J. Soil Water Conserv.* 70, 418–429. doi: 10.2489/jswc.70.6.418
- Basche, A., Tully, K., Álvarez-Berrios, N. L., Reyes, J., Lengnick, L., Brown, T., et al. (2020). Evaluating the Untapped Potential of U.S. Conservation Investments to Improve Soil and Environmental Health. *Front. Sustain. Food Syst.* 4, 547876. doi: 10.3389/fsufs.2020.547876
- Basche, A. D., and Roesch-McNally, G. E. (2017). Research topics to scale up cover crop use: Reflections from innovative Iowa farmers. *J. Soil Water Conserv.* 72, 59A–63A. doi: 10.2489/jswc.72.3.59A
- Blanco-Canqui, H., Sindelar, M., Wortmann, C. S., and Kreikemeier, G. (2017). Aerial interseeded cover crop and corn residue harvest: Soil and crop impacts. *Agron. J.* 109, 1344–1351. doi: 10.2134/agronj2017.02.0098
- Bowman, M., Poley, K., and McFarland, E. (2022). Farmers employ diverse cover crop management strategies to meet soil health goals. *Agric. Environ. Lett.* 7, e20070. doi: 10.1002/ael2.20070
- Carlisle, L. (2016). Factors influencing farmer adoption of soil health practices in the United States: A narrative review. *Agroecol. Sustain. Food Syst.* 40, 583–613. doi: 10.1080/21683565.2016.1156596
- Charatsari, C., Lioutas, E. D., and Koutsouris, A. (2020). Farmer field schools and the co-creation of knowledge and innovation: The mediating role of social capital. *Agric. Hum. Values* 37, 1139–1154. doi: 10.1007/s10460-020-10115-8
- Church, S. P., Lu, J., Ranjan, P., Reimer, A. P., and Prokopy, L. S. (2020). The role of systems thinking in cover crop adoption: Implications for conservation communication. *Land Use Policy* 94, 104508. doi: 10.1016/j.landusepol.2020.104508
- CTIC (2017). *Report of the 2016-17 National Cover Crop Survey. Joint publication of the Conservation Technology Information Center, the North Central Region Sustainable Agriculture Research and Education Program, and the American Seed Trade Association. West Lafayette, IN.* Available online at: https://www.ctic.org/files/2017CTIC_CoverCropReport-FINAL.pdf (accessed December 17, 2022).
- Das, S., Berns, K., McDonald, M., Ghimire, D., and Maharjan, B. (2022). Soil health, cover crop, and fertility management: Nebraska producers' perspectives on challenges and adoption. *J. Soil Water Conserv.* 77, 126–134. doi: 10.2489/jswc.77.2.00058
- Drewnoski, M., Muller, N., Saner, R., Jasa, P., Zoubek, G., Rees, J., et al. (2015). *Cover crop survey of Nebraska farmers.* Available online at: https://cropwatch.unl.edu/image/CW_News/2015/UNL-2015-Cover-Crop-Survey-Summary-2.pdf (accessed October 7, 2022)
- Hedges, L. V., Gurevitch, J., and Curtis, P. S. (1999). The meta-analysis of response ratios in experimental ecology. *Ecology* 80, 1150–1156. doi: 10.1890/0012-9658(1999)080[1150:TMAORR]2.0.CO;2
- HPRCC (2022). *ACIS Climate Maps.* Available online at: <https://hprcc.unl.edu/maps.php?maps=ACISClimateMaps> (accessed October 1, 2022).
- Kessavalou, A., and Walters, D. T. (1999). Winter rye cover crop following soybean under conservation tillage: Residual soil nitrate. *Agron. J.* 91, 643–649. doi: 10.2134/agronj1999.914643x
- Koehler-Cole, K., Brandle, J. R., Francis, C. A., Shapiro, C. A., Blankenship, E. E., and Baenziger, P. S. (2017). Clover green manure productivity and weed suppression in an organic grain rotation. *Renew. Agri. Food Syst.* 32, 474–483. doi: 10.1017/S1742170516000430
- Koehler-Cole, K., Elmore, R. W., Blanco-Canqui, H., Francis, C. A., Shapiro, C. A., Proctor, C. A., et al. (2020). Cover crop productivity and subsequent soybean yield in the western Corn Belt. *Agron. J.* 112, 2649–2663. doi: 10.1002/agi2.20232
- Krupek, F. S., Mizero, S. M., Redfearn, D., and Basche, A. (2022a). Assessing how cover crops close the soil health gap in on-farm experiments. *Agric. Environ. Lett.* 7, e20088. doi: 10.1002/ael2.20088
- Krupek, F. S., Redfearn, D., Eskridge, K. M., and Basche, A. (2022b). Ecological intensification with soil health practices demonstrates positive impacts on multiple soil properties: A large-scale farmer-led experiment. *Geoderma* 409, 115594. doi: 10.1016/j.geoderma.2021.115594
- Kyveryga, P. M. (2019). On-farm research: experimental approaches, analytical frameworks, case studies, and impact. *Agron. J.* 111, 2633–2635. doi: 10.2134/agronj2019.11.0001
- Lacoste, M., Cook, S., McNee, M., Gale, D., Ingram, J., Bellon-Maurel, V., et al. (2022). On-Farm Experimentation to transform global agriculture. *Nat. Food* 3, 11–18. doi: 10.1038/s43016-021-00424-4
- Laurent, A., Heaton, E., Kyveryga, P., Makowski, D., Puntel, L. A., Robertson, A. E., et al. (2022). A yield comparison between small-plot and on-farm foliar fungicide trials in soybean and maize. *Agron. Sustain. Dev.* 42, 86. doi: 10.1007/s13593-022-00822-3
- Marcillo, G. S., and Miguez, F. E. (2017). Corn yield response to winter cover crops: An updated meta-analysis. *J. Soil Water Conserv.* 72, 226–239. doi: 10.2489/jswc.72.3.226
- Miguez, F. E., and Bollero, G. A. (2005). Review of corn yield response under winter cover cropping systems using meta-analytic methods. *Crop Sci.* 45, 2318–2329. doi: 10.2135/cropsci2005.0014
- Myers, R., and Watts, C. (2015). Progress and perspectives with cover crops: Interpreting three years of farmer surveys on cover crops. *J. Soil Water Conserv.* 70, 125A–129A. doi: 10.2489/jswc.70.6.125A
- Nebraska On-Farm Research Network (2022a). Available online at: <https://on-farm-research.unl.edu> (accessed October 6, 2022)
- Nebraska On-Farm Research Network (2022b). On-farm research network database. Available online at: <https://resultsfinder.unl.edu> (accessed October 6, 2022)
- Nichols, V., Martinez-Feria, R., Weisberger, D., Carlson, S., Basso, B., and Basche, A. (2020). Cover crops and weed suppression in the U.S. Midwest: A meta-analysis and modeling study. *Agric. Environ. Lett.* 5, e20022. doi: 10.1002/ael2.20022
- Nielsen, D. C., Lyon, D. J., Higgins, R. K., Hergert, G. W., Holman, J. D., and Vigil, M. F. (2016). Cover crop effect on subsequent wheat yield in the central great plains. *Agron. J.* 108, 243–256. doi: 10.2134/agronj2015.0372
- Nielsen, R. L. (2010). *A Practical Guide to On-Farm Research.* West Lafayette, IN: Purdue University Department of Agronomy.
- NOAA (2022). *Physical Sciences Laboratory, Location of U.S. Climate Divisions.* Available online at: <https://psl.noaa.gov/data/usclimdivs/data/map.html> (accessed October 6, 2022).
- Oliveira, M. C., Butts, L., and Werle, R. (2019). Assessment of cover crop management strategies in Nebraska, US. *Agriculture* 9, 124. doi: 10.3390/agriculture9060124
- Peterson, A. T., Berti, M. T., and Samarappuli, D. (2019). Intersowing cover crops into standing soybean in the us upper midwest. *Agronomy* 9, 5. doi: 10.3390/agronomy9050264
- Philibert, A., Loyce, C., and Makowski, D. (2012). Assessment of the quality of meta-analysis in agronomy. *Agrosyst. Geosci. Environ.* 148, 72–82. doi: 10.1016/j.agee.2011.12.003

- Power, J. F., Doran, J. W., and Koerner, P. T. (1991). Hairy vetch as a winter cover crop for dryland corn production. *J. Prod. Ag.* 4, 62–67. doi: 10.2134/jpa1991.0062
- Practical Farmers of Iowa (2022). 2021 Cooperators reports. Available online at: <https://practicalfarmers.org/2022/08/2021-cooperators-program-report/> (accessed October 7, 2022).
- Prokopy, L. S., Floress, K., Arbuckle, J. G., Church, S. P., Eanes, F. R., Gao, Y., et al. (2019). Adoption of agricultural conservation practices in the United States: Evidence from 35 years of quantitative literature. *J. Soil Water Cons.* 74, 520–534. doi: 10.2489/jswc.74.5.520
- Prokopy, L. S., Floress, K., Klotthor-Weinkauff, D., and Baumgart-Getz, A. (2008). Determinants of agricultural best management practice adoption: Evidence from the literature. *J. Soil Water Cons.* 63, 300–311. doi: 10.2489/jswc.63.5.300
- Roesch-McNally, G. E., Basche, A. D., Arbuckle, J. G., Tyndall, J. C., Miguez, F. E., Bowman, T., et al. (2018). The trouble with cover crops: Farmers' experiences with overcoming barriers to adoption. *Renew. Agr. Food Syst.* 33, 322–333. doi: 10.1017/S1742170517000096
- Rosa, A. T., Creech, C. F., Elmore, R. W., Rudnick, D. R., Lindquist, J. L., Butts, L., et al. (2021). Contributions of individual cover crop species to rainfed maize production in semi-arid cropping systems. *Field Crops Res.* 271, 108245. doi: 10.1016/j.fcr.2021.108245
- Ruis, S. J., Blanco-Canqui, H., Creech, C. F., Koehler-Cole, K., Elmore, R. W., and Francis, C. A. (2019). Cover crop biomass production in temperate agroecozones. *Agron. J.* 111, 1535–1551. doi: 10.2134/agronj2018.08.0535
- Ruis, S. J., Blanco-Canqui, H., Jasa, P. J., Ferguson, R. B., and Slater, G. (2017). Can cover crop use allow increased levels of corn residue removal for biofuel in irrigated and rainfed systems? *BioEnergy Res.* 10, 992–1004. doi: 10.1007/s12155-017-9858-z
- SARE (2017). *How to Conduct Research on Your Farm or Ranch*. Available online at: <https://www.sare.org/wp-content/uploads/how-to-conduct-research-on-your-farm-or-ranch.pdf> (accessed October 5, 2022).
- Stewart, R. D., Jian, J., Gyawali, A. J., Thomason, W. E., Badgley, B. D., Reiter, M. S., et al. (2018). What we talk about when we talk about soil health. *Agric. Environ. Lett.* 3, 180033. doi: 10.2134/ael2018.06.0033
- St. Pierre, N. R. (2001). Invited review: integrating quantitative findings from multiple studies using mixed model methodology1. *J. Dairy Sci.* 84, 741–755. doi: 10.3168/jds.S0022-0302(01)74530-4
- Thompson, L. (2022). *E-mail Message December 15*.
- Thompson, L. J., Glewen, K. L., Elmore, R. W., Rees, J., Pokal, S., and Hitt, B. D. (2019). Farmers as researchers: in-depth interviews to discern participant motivation and impact. *Agron. J.* 111, 2670–2680. doi: 10.2134/agronj2018.09.0626
- USDA-NASS (2017). *Census of Agriculture*. Available online at: <https://www.nass.usda.gov/AgCensus/> (accessed October 5, 2022).
- USDA-NASS (2021). *State Agricultural Overview*. Available online at: https://www.nass.usda.gov/Quick_Stats/Ag_Overview/stateOverview.php?state=NEBRASKA (accessed October 5, 2022).
- USDA-NASS (2022). *Cropscape – USDA NASS Cropland Data*. Available online at: <https://nassgeodata.gmu.edu/CropScape/> (accessed October 7, 2022).
- Wallander, S., Smith, D., Bowman, M., and Claassen, R. (2021). *Cover crop trends, programs, practices in the United States. EIB 222, U.S. Department of Agriculture, Economic Research Service*. Available online at: <https://www.ers.usda.gov/webdocs/publications/100551/eib-222.pdf> (accessed October 7, 2022).
- Wick, A. F., Haley, J., Gasch, C., Wehlander, T., Briese, L., and Samson-Liebig, S. (2019). Network-based approaches for soil health research and extension programming in North Dakota, USA. *Soil Use Manage.* 35, 177–184. doi: 10.1111/sum.12444
- Williams, M. M., Mortensen, D. A., and Doran, J. W. (2000). No-tillage soybean performance in cover crops for weed management in the western Corn Belt. *J. Soil Water Conserv.* 55, 79–84.
- Wortman, S. E., Francis, C. A., Bernards, M. L., Drijber, R. A., and Lindquist, J. L. (2012). Optimizing cover crop benefits with diverse mixtures and an alternative termination method. *Agron. J.* 104, 1425–1435. doi: 10.2134/agronj2012.0185
- Zomer, R. J., Trabucco, A., Bossio, D. A., and Verchot, L. V. (2008). Climate change mitigation: A spatial analysis of global land suitability for clean development mechanism afforestation and reforestation. *Agric. Ecosyst. Environ.* 126, 67–80. doi: 10.1016/j.agee.2008.01.014



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EDITED BY

Jacob Jungers,
University of Minnesota Twin Cities,
United States

REVIEWED BY

Antonio DiTommaso,
Cornell University, United States
Atique ur Rehman,
Bahauddin Zakariya University, Pakistan

*CORRESPONDENCE

Cameron A. MacKenzie
✉ camacken@iastate.edu

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Identifying research priorities through decision analysis: A case study for cover crops

Gina A. Nichols^{1,2} and Cameron A. MacKenzie^{3*}

¹Department of Plant Sciences, University of California, Davis, Davis, CA, United States, ²Department of Agronomy, Iowa State University, Ames, IA, United States, ³Department of Industrial and Manufacturing Systems Engineering, Iowa State University, Ames, IA, United States

Introduction: In Midwestern maize (*Zea-mays* L.)-based systems, planting an over-wintering cover crop such as rye (*Secale cereale* L.) following fall harvests of summer crops maintains continuous soil cover, offering numerous environmental advantages. However, while adoption of cover crops has increased over the past decade, on a landscape-scale it remains low. Identifying where agronomic research could be most impactful in increasing adoption is therefore a useful exercise. Decision analysis (DA) is a tool for clarifying decision trade-offs, quantifying risk, and identifying optimal decisions. Several fields regularly utilize DA frameworks including the military, industrial engineering, business strategy, and economics, but it is not yet widely applied in agriculture.

Methods: Here we apply DA to a maize-soybean [*Glycine max* (L.) Merr.] rotation using publicly available weather, management, and economic data from central Iowa.

Results: In this region, planting a cover crop following maize (preceding soybean) poses less risk to the producer compared to planting following soybean, meaning it may be a more palatable entry point for producers. Furthermore, the risk of reduced maize yields when planting less than 14 days following rye termination substantially contributes to the overall risk cover crops pose to producers, but also has significant potential to be addressed through agronomic research.

Discussion: In addition to identifying research priorities, DA provided clarity to a complex problem, was performed using publicly available data, and by incorporating risk it better estimated true costs to the producer compared to using input costs alone. We believe DA is a valuable and underutilized tool in agronomy and could aid in increasing adoption of cover crops in the Midwest.

KEYWORDS

cover crop, soybean, risk, decision analysis (DA), Iowa (USA), maize (*Zea mays* L.)

1. Introduction

Many cropping systems in the United States (US) have undergone simplifications, now being composed of only a few, often annual, crops (Aguilar et al., 2015; Hijmans et al., 2016; Crossley et al., 2021). These systems frequently leave the soil fallow for some period of time, presenting notable environmental challenges including but not limited to increased risk of soil erosion and an increased potential for nutrient loss (Mitsch et al., 2001; Hatfield et al., 2009; Syswerda et al., 2012). The notion of “continuous living cover” has been used to encourage creative solutions to these issues by focusing cropping system re-design on eliminating these environmentally-challenging fallow periods. Planting cover crops to reduce fallow periods is one such tactic that could at least partially address many of the environmental problems presented by annual cropping systems.

The US produces approximately one-third of the world's maize (*Zea mays* L.) and soybean [*Glycine max* (L.) Merr.] [Food and Agriculture Organization (FAO), 2020], with five states in the Midwestern region contributing over half of that production [Feyereisen et al., 2006]. It follows that large amounts of agricultural land in the Midwestern US are dedicated to cropping systems that grow only maize and soybean [Boryan et al., 2011; USDA National Agricultural Statistics Service Cropland Data Layer (USDA NASS CDL), 2021]. Utilizing over-wintering cover crops in these systems has been shown to reduce soil erosion and nitrate leaching (Kaspar et al., 2007, 2012; Chen et al., 2022), is associated with a reduction in crop insurance losses due to drought, excess heat, and excess moisture (Aglasan and Rejesus, 2021), and possibly offer numerous other context-specific benefits such as increased soil infiltration rates, higher soil water-holding capacity, or increased soil organic matter content (Moore et al., 2014; Basche and DeLonge, 2017; Krupek et al., 2022; Nichols et al., 2022). However, the Midwestern maize-soybean systems present challenges to cover crop adoption. In some regions of the US, cover crop adoption on annual cropland is above 25% and growing (Hamilton et al., 2017). Meanwhile, states comprising the Midwestern US exhibit some of the lowest adoption rates, with most states well below 10% adoption (Hamilton et al., 2017; Rundquist and Carlson, 2017; Seifert et al., 2018).

Low adoption rates within the Midwest have been the subject of numerous studies, and it is clearly a complex issue involving economics, climate constraints, field operations, management, equipment, culture, and technical knowledge (Lee et al., 2018; Church et al., 2020; Nichols et al., 2020a; Thompson et al., 2021; Yoder et al., 2021; Park et al., 2022). One barrier we believe merits more attention is that of risk. Risk incorporates two components, uncertainty and negative consequences, and is frequently measured with probabilities describing the potential severity of consequences (Kaplan and Garrick, 1981; Bedford and Cooke, 2001; Hubbard, 2020). Cover crops present both direct, and indirect risks. Managerially, maize and soybean are both are planted in the late spring (April, May) and harvested in the fall (September, October, November). Producers typically fit over-wintering cover crops into these systems by planting a cover crop in the fall after the cash crop harvest, and terminating the cover crop in the spring before the next cash crop is planted [Sustainable Agriculture Research and Education (SARE), 2020]. Therefore, both the planting and termination of an over-wintering cover crop such as rye (*Secale cereale* L.) can conflict with cash crop management. As such, using a cover crop requires complex decision-making that balances risk and rewards in uncertain conditions. While perceived risks associated with cover cropping are often cited as barriers to adoption (Arbuckle and Roesch-McNally, 2015), quantifying those risks in economic terms is challenging (e.g., Bergtold et al., 2019; Plastina et al., 2020). Furthermore, while lists of cover crop research priorities have been proposed (e.g., Carlson and Stockwell, 2013; Basche and Roesch-McNally, 2017), a tool for ranking priorities would be useful. By quantifying the risk associated with each decision point for producers, research priorities can be set to address points posing the highest risk. The use of risk as a ranking tool would also help researchers and funding organizations assess how resources can be used most impactfully.

Furthermore, understanding how uncertainties around weather conditions elevate risks of profit loss is important for understanding both the mechanisms for delivering incentives, and the amount producers may require for meaningful participation.

Decision analysis is an interdisciplinary tool that can be applied to analyze decision-making under uncertain conditions (Howard, 1988; Clemen and Reilly, 2013; Howard and Abbas, 2015). It can leverage both quantitative information and expert knowledge, incorporate different degrees of risk aversion, and through sensitivity analyses can allow exploration of the decision space (Cegan et al., 2017; Shackelford et al., 2019). It is a recognized tool for coping with risk in agriculture (Hardaker et al., 2015) and has been applied to a range of agronomic-related topics including agroforestry adoption risks, nitrate pollution loading, cover crop species selection, optimal cropping system choices, and promoting sustainable agricultural practices (Almasri and Kaluarachchi, 2005; Gandorfer et al., 2011; Ramírez-García et al., 2015; Talukder et al., 2017; Do et al., 2020). However, to our knowledge decision analysis frameworks have had limited application regarding management decisions related to cover crops in the maize/soybean systems of the Midwestern US. Therefore, the objectives of this study were two-fold:

- 1) Provide a case study using publicly available data to demonstrate the process and utility of applying decision analysis to cover crop systems.
- 2) Use a basic analysis to suggest research priorities for cover crops in Central Iowa.

We chose to use Central Iowa as a case study because it has large areas in maize/soybean systems that are broadly representative of the US Midwest [USDA National Agricultural Statistics Service Cropland Data Layer (USDA NASS CDL), 2021], and currently demonstrates a moderate amount of cover crop adoption (Rundquist and Carlson, 2017). Furthermore, Iowa's land grant institution, Iowa State University, as well as the United States Department of Agriculture (USDA) National Laboratory for Agriculture and the Environment (NLAE) are located in Central Iowa and support a strong infrastructure for publicly funded agronomic research trials in this region that provide rich sources of public data.

2. Methods and materials

2.1. Decision set

We used cereal rye (*Secale cereale* L.) as our "model" over-wintering cover crop because it is the most used cover crop in Iowa and is one of the most widely used cover crop species in the Midwest (Singer, 2008). Assuming a producer has both the maize and soybean phase of a maize-soybean rotation growing at a given time, there are two scenarios for cover crop integration, each including three decision alternatives with unique benefits and challenges (Table 1). Concomitant benefits and challenges of each decision

TABLE 1 Two scenarios each including three decision alternatives related to cover cropping in a maize/soybean rotation with various benefits and challenges associated with each alternative.

Decision alternative	Description	Benefits	Challenges
In fields with a soybean crop			
1	Do not plant a cover crop following soybean harvest	No added costs or risks due to cover crop	Low residue from soybean crop leaves soil vulnerable to erosion (Dickey et al., 1985) Soil nitrogen is likely to be lost from the field in the spring to leaching (Qi et al., 2008) Low residue contributes minimally to non-chemical weed control
2	Plant a cover crop, plan to terminate early April	Soybeans are harvested earlier in the fall compared to maize [USDA National Agricultural Statistics Service (USDA NASS), 2022], allowing for earlier cover crop planting which increases likelihood of successful establishment and more cover crop growth (Chatterjee et al., 2020; Nichols et al., 2020b) Cover crop residue reduces soil erosion following soybeans (Kaspar et al., 2001) Cover crop residue may provide weed control following soybeans (Nelson and Bennett, 2018) Cover crop growth can uptake soil nitrate thus mitigating nutrient pollution (Qi et al., 2008; Kaspar et al., 2012; Martinez-Feria et al., 2019)	Cover crop may indirectly reduce subsequent maize yields by competing for workable field days and delaying maize planting, which often results in lower maize yields (Baum et al., 2019) Planting maize less than two weeks following cover crop termination may result in reduced yields, but the effect is unpredictable (Johnson et al., 1998; Acharya et al., 2017, 2020)
3	Plant a cover crop, plan to terminate late April	Enhances cover crop benefits due to more cover crop growth and biomass	Increases chances of delayed maize planting, and thus reduced maize yields
In fields with a maize crop			
4	Do not plant a cover crop following maize harvest	No added costs or risks due to cover crop	Soil nitrogen is likely to be lost from the field in the spring to leaching (Qi et al., 2008)
5	Plant a cover crop, plan to terminate early April	Maize can leave large nitrate reserves in the soil at harvest, and cover crop growth can uptake the nitrate thus mitigating nutrient pollution (Qi et al., 2008; Kaspar et al., 2012; Martinez-Feria et al., 2019) Soybean planting dates are less sensitive to planting dates compared to maize (Kessler et al., 2020), and rye does not increase risk of root disease in subsequent soybean crop (Araldi-Da-Silva et al., 2022)	Timely fall cover crop planting can be difficult following maize harvest Maize is harvested in late fall, and late-planted cover crops can result in low spring cover crop biomass (Chatterjee et al., 2020; Nichols et al., 2020b), and therefore minimal benefits, if terminated in early April
6	Plant a cover crop, plan to terminate late April	Enhances cover crop benefits due to more cover crop growth and biomass	Larger amounts of cover crop biomass may be more difficult to terminate uniformly

alternative highlights the need to use a quantitative approach to decision optimization, which can be achieved using decision analysis frameworks.

2.2. Decision structure

The decision set was translated into decision models with known states, uncertainties, and values, each described below.

2.2.1. Fall weather uncertainties

Cover crops are most often planted following cash crop harvests [Sustainable Agriculture Research and Education (SARE), 2020].

Soybean crops in Central Iowa are harvested in September or October, and maize in October or November [USDA National Agricultural Statistics Service (USDA NASS), 2022]. Planting cover crops into standing crops before harvest can increase the probability of establishment (Wilson et al., 2014) but requires specialized equipment that is not yet widely available. We therefore assume cover crop planting occurs after cash crop harvest.

Seeds require precipitation to germinate, and heat units to establish such that the plants emerge and survive the winter. Failure of a cover crop to germinate or establish in the fall results in wasted seed, wasted fuel, and possible weed problems the following spring. While the amount of precipitation needed for rye to germinate depends on soil moisture conditions at planting, crop advisors and producers often assume 1.27 cm (0.5 inches) is needed

(Sarrantonio, 1994), which is consistent with field studies (Fisher et al., 2011; Wilson et al., 2013) and simulation model assumptions (Feyereisen et al., 2006; Marcillo et al., 2019). While we assumed 1.27 cm was needed for our baseline analysis, this assumption was tested through a sensitivity analysis (see Section 2.3.2).

Growing degree days (GDDs) represent an estimation of the number of heat units accumulated above a threshold temperature specific to a crop. For rye the threshold is 0 or 1°C (Feyereisen et al., 2006). We acknowledge the number of GDDs required for rye to successfully over-winter will depend on several additional factors including soil texture and snow cover. A study in Minnesota suggested rye required at least 100 GDDs in the fall to produce biomass in the spring (Kantar and Porter, 2014). We therefore estimated rye requires 100 GDDs to successfully establish before winter, but tested the sensitivity of this assumption (see Section 2.3.2).

To estimate the probability of successful rye establishment, we used 30 years of historical weather data (1988–2019) collected at the AMES-8-WSW station from the Iowa Environmental Mesonet (IEM) (2022). We chose this dataset because it had previously undergone an extensive quality check (Archontoulis et al., 2020). Using 30 years of weather data, we calculated (i) the probability the site received 1.27 cm of rainfall during an allotted timeframe, and (ii) the probability of achieving 100 GDDs in the allotted timeframe. The timeframes differed by decision alternative to account for the generally earlier harvest dates for soybean compared to maize [USDA National Agricultural Statistics Service (USDA NASS), 2022]. The precipitation timeframes were 15-Oct through 30-Nov and 1-Nov through 30-Nov for rye following soybean and rye following maize, respectively. The GDD accumulation timeframes were 15-Oct through 1-Dec and 1-Nov through 1-Dec for rye following soybean and rye following maize, respectively. We chose to calculate the precipitation and GDD probabilities separately rather than as a joint probability to aid in assessing how breeding efforts could increase changes of establishment. We recognize this model for establishment is a simplification of the complex interactions between weather, soil, and management considerations. While more sophisticated modeling approaches have been utilized for predicting cover crop establishment (Baker and Griffis, 2009; Marcillo et al., 2019; Nichols et al., 2020b), they require specialized skillsets and a significant time commitment. Our goal in this exercise is to demonstrate how insights can be obtained using publicly available data and approachable methodologies.

2.2.2. Spring weather uncertainties

Iowa has a humid continental climate wherein a significant amount of precipitation occurs during the spring months. In addition to the direct constraints on management that precipitation exerts, performing field operations in wet soils can result in undesirable outcomes including long-term soil compaction and equipment malfunctions. The USDA National Agricultural Statistics Service (NASS) surveys producers to determine the number of days suitable for fieldwork (workable-field day; WFD) for each week throughout the year [USDA National Agricultural Statistics Service (USDA NASS), 2018]. A “suitable” day is defined

as one in which weather and field conditions allow producers to work in fields the majority of a given day. Determining whether a day is a “suitable” is subjective, but provides valuable information about the progress and constraints of agricultural production on a landscape level.

Historical data shows that in Iowa, the number of WFDs during the spring can severely restrict field activities (Urban et al., 2015; Edwards, 2020). To comply with governmental crop insurance cost-share policies, cover crops must be terminated before the cash crop is planted [Bergtold et al., 2019; USDA Risk Management Agency (USDA RMA), 2019]. Therefore, the presence of a living cover crop that must be terminated before the cash crop can be planted can potentially add to the spring workload for a producer. While this depends on whether producers typically have a pre-plant or pre-emergent herbicide pass, the operation is much less crucial when the goal is simply to eliminate weeds around cash crop planting compared to killing a live cover crop to comply with federal crop insurance requirements. To account for the increased importance of timely cover crop termination, in this exercise we assumed cover crop termination requires an additional set of field working-days compared to systems without a cover crop. However, because many producers do a pre-plant or pre-emergent herbicide pass in systems without cover crops, we did not assume extra herbicide or fuel costs associated with terminating the cover crop. In short, we assumed producers who plant a rye cover crop require two more spring WFDs than those who do not. This fact introduces an important component of risk that is often not accounted for explicitly in economic analyses.

The decision of cover crop termination timing will also affect WFDs, and therefore may indirectly affect cash crop yields. If a producer has WFDs in early April, the producer must choose whether to utilize them to terminate the cover crop, or wait in order to accrue more benefits from prolonged cover crop growth (Table 1). Societal-level benefits such as reduced nitrate leaching, as well as farm-level benefits such as the potential to off-set weed control costs, increase as spring cover crop termination dates are delayed and cover crop biomass increases (Finney et al., 2016; Thapa et al., 2018; Nichols et al., 2020b). However, by choosing not to utilize early April WFDs, the producer risks not having sufficient WFDs in late April to terminate the cover crop or plant the cash crop, resulting in delayed cash crop planting and a possible concomitant reduction in yields. Therefore, understanding the uncertainty around WFDs in the spring is an important component in assessing optimal decision alternatives.

In this analysis we only include the uncertainties associated with WFDs. In years with very low spring precipitation, delaying cover crop termination can also result in decreased cash crop yields due to the cover crop's use of stored soil water needed for cash crop production. While this risk is possible, due to climatic patterns it is not common in Central Iowa (Daigh et al., 2014; Martinez-Feria et al., 2016). Therefore, the risk of cover crops inducing drought-related yield reductions in the following cash crop is not considered in this exercise.

Workable field days are estimated by surveying farmers about how many days in the previous week were field-workable. The

data is therefore reported as a number of days within a 7-day calendar period, with this period being inconsistent between years. For the purposes of this exercise, we chose to take the total WFDs over the 7-day reporting period and divide the total by seven to assign a number of WFDs to each calendar day the reporting week included. We then created five spring categories (early April, late April, early May, and late May, June). Workable field day values were then summed within these spring calendar categories. More details, including R code, concerning this procedure can be found in [Supplementary material](#). We assumed cover crop termination would require two WFDs within a spring category, and cash crop planting would likewise require two WFDs. Therefore, cover crop termination and cash crop planting within a given window would require four WFDs. The probability of two and four WFDs being reported in a given spring category was calculated using 30 years of historical data (1988–2019).

2.2.3. Subsequent maize yield uncertainties

On average, winter cover crops such as rye have been shown to have a neutral effect on subsequent maize and soybean yields ([Marcillo and Miguez, 2017](#)). However, numerous studies have shown that under certain conditions, planting maize <10–14 days following cover crop termination can result in lower maize yields ([Johnson et al., 1998](#); [Pantoja et al., 2015](#); [Acharya et al., 2017](#); [Hirsh et al., 2021](#); [Quinn et al., 2021](#)). We assumed a producer would plant their maize crop as early as possible, regardless of the penalty that would be incurred due to the <14 day window. We made this assumption because conversations with producers confirmed that while they were aware there may be a yield penalty from a small termination-planting window, it was inconsistent and may not occur at all, and they were therefore more concerned with timely maize planting. We therefore assumed if there were four WFDs in a given spring category, the producer would plant maize but there would be a 50% chance of a 10% decrease in maize yield. We acknowledge that in our scenarios, the 10% yield penalty from the small termination-planting window is larger than the penalty incurred for delaying planting until late May, but we believe our decision structure captures the uncertainty currently associated with whether that yield penalty will be incurred. Soybeans are not impacted by the time between rye termination and soybean planting ([Acharya et al., 2020](#)), so no yield penalty was assigned in those circumstances.

2.2.4. Value

The main contributors to decision value were estimated using partial budgets and included the costs from planting a cover crop, the savings from planting a cover crop, and the income from the subsequent cash crop. Extension publications, farming group publications, and peer-reviewed literature were used to guide each estimation. Sensitivity analyses were performed on assumed values (Section 2.3.2), and instances where conclusions were overly sensitive to assumptions were noted.

To estimate the direct costs associated with planting and terminating a cover crop we used Iowa State University's "Economics of Cover Crops" decision tool ([Iowa State University Extension, 2018](#)). While these prices will fluctuate depending on the

price of fuel and labor, we feel they are sufficiently representative for this exercise ([Table 2](#)).

In order to account for the effect of cover cropping on income from crop yields, we needed to estimate the net revenue a producer expects per unit crop yield. The net revenue from a crop will depend on producer costs of production as well as market prices, both of which vary significantly across years. To overcome this variability, we looked at production costs ([Iowa State University Extension, 2022](#)) and market prices [[USDA National Agricultural Statistics Service \(USDA NASS\), 2022](#)] from 2013 to 2021, calculated the net revenue per unit crop yield for each year, then took the year with the maximum net revenue for each crop. By calculating the net revenue in this manner, when a rye cover crop negatively impacted cash crop yields our analyses represented the highest potential costs of those effects. All prices and calculations are available in [Supplementary material](#).

Maize was assumed to have a maximum yield of 10.7 dry Mg ha⁻¹ (200 bu ac⁻¹) and soybean a yield of 1.4 dry Mg ha⁻¹ (60 bu ac⁻¹), which are representative of the state average yields in Iowa [[USDA National Agricultural Statistics Service \(USDA NASS\), 2022](#)]. Maize yield is sensitive to planting date, with later planting dates being associated with lower yields ([Kucharik, 2008](#); [Baum et al., 2019](#)). We therefore assume a graduated yield penalty increasing 5–20% as maize planting occurs past April ([Supplementary Table S2](#)). In summary, the decision of whether to terminate the cover crop early or late impacts the available WFDs ([Table 3](#)), which impact whether the producer incurs a termination-planting penalty or a late-planting penalty, both of which impact the value of the decision.

Soybean yields are less sensitive to planting dates compared to maize ([Kessler et al., 2020](#)) and therefore was assumed to have a less severe graduated penalty as planting was delayed (5–10%; [Table 2](#)).

When the cover crop was followed by a maize crop (decision alternatives 1–3), we assumed herbicide costs were equal in the cover crop and no-cover alternatives (\$205 ha⁻¹). When the cover crop was followed by a soybean crop (decision alternatives 4–6), we utilized information from on-farm experiments showing producers reduced herbicide costs due to the mulch provided by a late-terminated cover crop. Therefore, in the decision alternative where the cover crop was terminated in late April or later followed by soybean planting (decision alternative 6), a \$37 ha⁻¹ savings in herbicides was applied ([Nelson and Bennett, 2018](#)).

There are currently no payments available to farmers in Iowa for the societal benefits reaped from delaying cover crop termination. However, other areas in the US have implemented payment structures that reward late termination due to the societal benefits gained from late termination ([Maryland Department of Agriculture, 2022](#)), so the potential for this payment in decision alternatives 3 and 6 was included in sensitivity analyses.

2.3. Decision analysis

2.3.1. Building decision trees

Decisions can be visualized and modeled using decision tree notation ([Clemen and Reilly, 2013](#); [Howard and Abbas, 2015](#)).

TABLE 2 Summary of economic assumptions for each scenario with relative cash crop yield assumptions provided in parentheses.

	No cover crop system	Cover crop system	
		14+ day gap	<14 day gap ^{a,b}
Cover crop			
Cover crop seed	–	\$20 ha ^{−1}	\$20 ha ^{−1}
Cover crop planting	–	\$32 ha ^{−1}	\$32 ha ^{−1}
Cost-shares/insurance discounts with cover crop planting	–	\$12–74 ha ^{−1}	\$12–74 ha ^{−1}
Cover crop preceding maize			
Herbicide costs	\$205 ha ^{−1}	\$205 ha ^{−1}	\$205 ha ^{−1}
Maize income (assumed \$2.14 net income per bushel)			
Planted early April	\$1057 ha ^{−1}	–	\$1057 ha ^{−1} /\$951 ha ^{−1} (90%)
Planted late April	\$1057 ha ^{−1}	\$1057 ha ^{−1}	\$1057 ha ^{−1} /\$951 ha ^{−1} (90%)
Planted early May ^c	\$1004 ha ^{−1} (95%)	\$1004 ha ^{−1} (95%)	\$1004 ha ^{−1} (95%)/\$889 ha ^{−1} (85%)
Planted late May	\$951 ha ^{−1} (90%)	\$951 ha ^{−1} (90%)	\$951 ha ^{−1} (90%)/\$846 ha ^{−1} (80%)
Planted June	\$846 ha ^{−1} (80%)	\$846 ha ^{−1} (80%)	\$846 ha ^{−1} (80%)/\$740 ha ^{−1} (70%)
Cover crop preceding soybean			
Herbicide costs ^d	\$205 ha ^{−1}	\$168 ha ^{−1}	\$168 ha ^{−1}
Soybean income (assumed \$4.06 net income per bushel)			
Planted early April	–	–	–
Planted late April	\$601 ha ^{−1}	\$601 ha ^{−1}	\$601 ha ^{−1}
Planted early May	\$601 ha ^{−1}	\$601 ha ^{−1}	\$601 ha ^{−1}
Planted late May ^e	\$571 ha ^{−1} (95%)	\$571 ha ^{−1} (95%)	\$571 ha ^{−1} (95%)
Planted June	\$541 ha ^{−1} (90%)	\$541 ha ^{−1} (90%)	\$541 ha ^{−1} (90%)

^aThe decision model for rye following soybean includes a 50% chance a <14 day maize yield reduction will not occur (first values listed), and 50% chance the <14 day maize yield reduction will occur (second values listed).

^bEstimated maize yield reduction due to termination-planting gap are based on Johnson et al. (1998), Hirsh et al. (2021), and Quinn et al. (2021).

^cEstimated maize yield reduction due to delayed maize planting are based on Kucharik (2008) and Baum et al. (2019).

^dEstimated reduction in herbicide costs based on Nelson and Bennett (2018).

^eEstimated soybean yield reduction based on Kessler et al. (2020).

The full decision model is available in [Supplementary material](#) and consists of building out a branch for each unique decision node and uncertainty outcome with probabilities, then assigning a value to each branch. We assume a risk-neutral decision maker which means that the decision maker should choose the alternative that maximizes his or her expected value. A square in the decision tree represents a choice between two or more alternatives, and a circle represents an uncertainty where each branch stemming from the uncertainty is assigned a probability. The first decision for the producer is whether or not to plant a cover crop in the fall ([Figure 1](#)). If the producers choose to plant a cover crop, there is an uncertainty about whether or not sufficient precipitation occurs followed by a second uncertainty about whether or not sufficient GDDs are accumulated. If sufficient precipitation and sufficient GDDs occur, the producer makes a second decision about whether or not to terminate in early April ([Supplementary Figure S1](#)). This decision is followed by uncertainties in the number of WFDs available in a given time frame, and whether there is a penalty when maize is planted in the same spring category as cover crop termination. The decision tree is solved using a “rollback” procedure starting from the right-hand side of the tree. If a decision

(square) node is encountered, the alternative with the largest expected monetary value is selected. If an uncertainty (circle) node is encountered, the expected monetary value is calculated using the probabilities on the branches as weights. This procedure results in identifying the alternative for a given decision (e.g., whether or not to plant a cover crop in the fall) that maximizes the producer's expected monetary value.

2.3.2. Sensitivity analyses

Sensitivity analysis on the uncertainty and parameter assumptions can provide insight into the criticality and importance of an assumption or variable to the decision. The sensitivity of outcomes was assessed for the precipitation required for rye germination (ranging from 0 to 3.5 cm in 1 mm increments), the number of GDDs needed for rye to over-winter (ranging from 0 to 300 in 5 GDD increments), the potential relative reduction in maize yields when maize was planted <14 days following rye termination (ranging from 0 to 20% in 5% increments), the incentive payments offered to plant rye (ranging from \$0 to 200

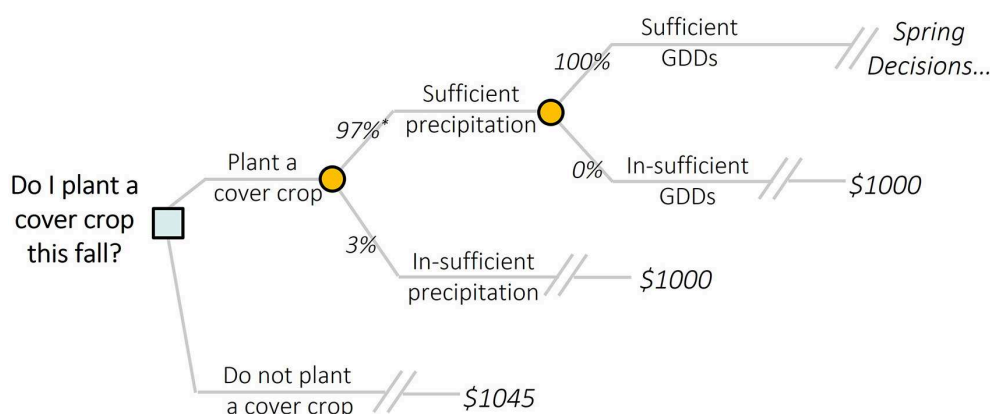


FIGURE 1

Decision tree visualization for planting a cover crop following a soybean crop. The first decision (light blue square) is whether or not to plant a cover crop. If the producer chooses to plant a cover crop, there is uncertainty about precipitation and growing degree days (GDDs); if the cover crop is successfully established the producer will have to decide whether to terminate the cover crop in early April, or to wait until late-April. Each decision branch has a monetary value. *See section 2.2 for calculation of these probabilities.

TABLE 3 Summary of probabilities of workable field days (WFDs) in a given timeframe based on 30 years of NASS survey data [USDA National Agricultural Statistics Service (USDA NASS), 2022].

Management window	Probability of two or more workable field days (WFDs)	Probability of four or more WFDs
1-Apr through 15-Apr (early April)	69%	48%
16-Apr through 30-Apr (late April)	71%	37%
1-May through 15-May (early May)	89%	45%
16-May through 31-May (late May)	87%	55%

ha⁻¹ in \$1 increments), and the incentive payments offered to delay termination of rye (ranging from \$0 to 200 ha⁻¹ in \$1 increments). Additionally, sensitivity analyses were performed on the assumed revenues and costs associated with each scenario to ensure conclusions were not overly sensitive to these assumptions (see [Supplementary material](#); Gupta, 2022 for details).

2.3.3. Value of information

In our decision model, if a producer has two WFDs within 14 days following cover crop termination, they have a 50% of incurring a 10% maize yield reduction if they choose to plant. This uncertainty is due to research gaps—we do not yet have sufficient information to provide a producer to help them determine whether this reduction will occur. By estimating the value of the decision if the producer knows whether the yield reduction will occur, one can estimate the “value of perfect information” (Repo, 1989). This provides an estimate of what that information would be worth to producers, thus allowing researchers to assess how impactful such research would be. We therefore estimated the value of knowing

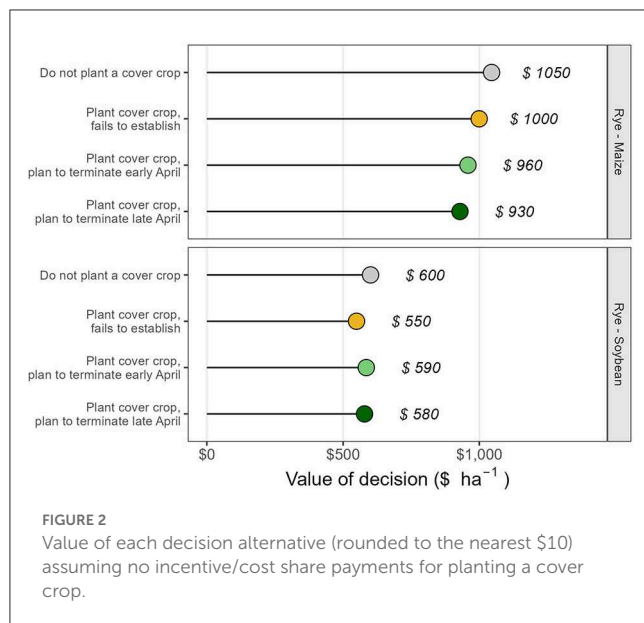
when there would not be a reduction in maize yields when planting <14 days after cover crop termination.

3. Results and discussion

3.1. Optimal decisions

Assuming there is no cost-share available for planting a cover crop and long-term or societal economic benefits are not accounted for, the overall expected monetary value of not planting a cover crop is greater than the expected monetary value of planting a cover crop, regardless of the sequencing scenario (Figure 2). This analysis shows that in addition to the cost of seed and fuel to plant the cover crop (\$52 ha⁻¹), when rye precedes maize there is an additional \$40–70 ha⁻¹ cost associated with the risk that the spring management of the cover crop will result in reduced maize yields (either through delayed maize planting due to insufficient WFDs or <14 day gap penalties). When rye precedes soybeans, the costs of planting the cover crop and risks of reduced yields due to delayed planting are partially compensated by through reduced herbicide costs. Within the decision sets that include the alternative of planting a cover crop, the value of the decision is always maximized if the cover crop is terminated in early vs. late April.

Many of the benefits reaped from planting cover crops (e.g., reduced soil erosion, reduced nitrate leaching, non-chemical weed control) are directly related to the amount of biomass the cover crop produces (Finney et al., 2016; Thapa et al., 2018; Nichols et al., 2020b). However, in areas that lack incentives for delaying cover crop termination to allow the cover crop to grow, our analyses show the optimal decision is to terminate the cover crop as soon as possible, even when there might be cost savings from reduced herbicide use (Rye-Soybean scenario in Figure 2). Notably, the termination decision differential is highest when the cover crop precedes maize, meaning the sequencing where society may benefit the most (higher mitigation of erosion and nitrate leaching, Table 1) would also require the highest incentives to



render late April termination the optimal decision. The US state of Maryland has created a tiered incentive system wherein producers are compensated more for early planting and late termination of cover crops (Maryland Department of Agriculture, 2022). Our analysis indicates having compensation rates differ by cropping sequence may also be an approach worth considering.

Our analyses also expose a potential moral hazard. If a producer chooses to plant a cover crop preceding a maize crop and receives a cost-share or incentive for doing so, failed cover crop establishment will lead to a better financial result than successful establishment (Rye-Maize scenario in Figure 2). It is important to provide support for producers as they learn to manage cover crops, and often cover crop establishment is out of a producer's control, but our analyses demonstrate the complexity in determining the best payment structures, and the need to include the risks the cover crop may pose to the subsequent crop yields.

3.2. Sensitivity to cost-share/incentives

If there are no cost-shares or incentive programs, the overall expected monetary value of not planting any cover crop is greater than the expected value of planting a cover crop, regardless of the sequencing scenario (in the top panel of Figure 3, this is seen from the “do not plant rye” alternative having a greater value when the cost share or incentive is \$0 on the horizontal axis). However, current incentive programs may be enough to make planting a cover crop preceding a soybean cash crop (“Rye-Soybean”) the optimal decision. If the incentive is greater than \$30 ha⁻¹, the expected monetary value of planting rye prior to soybeans is greater than not planting rye.

When a cover crop precedes a maize crop (“Rye-Maize” in top panel of Figure 3), within the current range of incentives the optimal decision is to not plant a cover crop. However, this recommendation is sensitive to the reduction in maize yield due to planting <14 days following cover crop termination (bottom panel

of Figure 3). If the potential reduction in yield were eliminated, the difference between the value of not planting a cover crop and planting a cover crop could be reduced from \$85 ha⁻¹ to \$60 ha⁻¹, bringing the difference into the range of current incentive programs in this area (\$12–74 ha⁻¹).

The exact causes of the reduced yield in maize are not yet clear and it is currently not possible to predict when they will manifest (e.g., Patel et al., 2019; Quinn et al., 2021). The value of perfect information is worth \$20–25 depending on the planned cover crop timing, which is roughly equal to the increased value from eliminating the yield penalty. This indicates that research that allows producers to accurately predict when the yield penalty will occur is equally as valuable as eliminating the yield penalty. Potential mechanisms include altered nutrient dynamics, disease pressure, allelopathy, rye stands that are not fully terminated, changes in soil temperature and/or moisture in a rye cover crop system. A meta-analysis of studies may aid in identifying factors that drive the variation in the effect. Our analyses demonstrate that this phenomenon poses a significant risk to producers, and a better understanding of the drivers and identification of ways to predict when yield declines are likely would greatly reduce the financial risk associated with planting a rye cover crop in these systems.

3.3. Sensitivity to weather

On average, Central Iowa received 7.4 and 4.2 cm of rain from 15-Oct and 1-Nov through 30-Nov, respectively. This equated to a high probability (>80%) of the rye cover crop receiving sufficient precipitation for germination (>1.27 cm) in both sequences (Figure 4, Supplementary Table S3). This result was robust against uncertainty in our assumptions; even if rye required almost double the assumed precipitation, the probability of receiving that amount of rainfall did not drop below 80% for either planting scenario (Figure 4). While the probability of accumulating sufficient GDDs (100) was 100% when the rye was planted following soybeans (15-Oct planting date), it dropped to 71% chance of success when planted following maize (1-Nov planting date; Supplementary Table S3). The probability of establishment was very sensitive to the sequencing (rye following soybeans or rye following maize). For the 1-Nov planting date, the results are very sensitive to the assumed GDDs required for establishment.

These results can be used to guide research efforts. Our analysis demonstrates that in most cases, precipitation is not the limiting factor for cover crop establishment in Central Iowa. Breeding varieties that require less precipitation to germinate would likely involve breeding for smaller seeds, which carries inherent tradeoffs (e.g., Carleton and Cooper, 1972; Mohler et al., 2009). A study done in Minnesota showed precipitation accounts for the highest amount of variation in rye establishment, followed by temperature (Wilson et al., 2013), demonstrating the value of evaluating weather-related risks locally. While our results do not account for how the precipitation is distributed across time and how that may impact germination, our results suggest this area of Iowa can support larger precipitation requirements for cover crops without

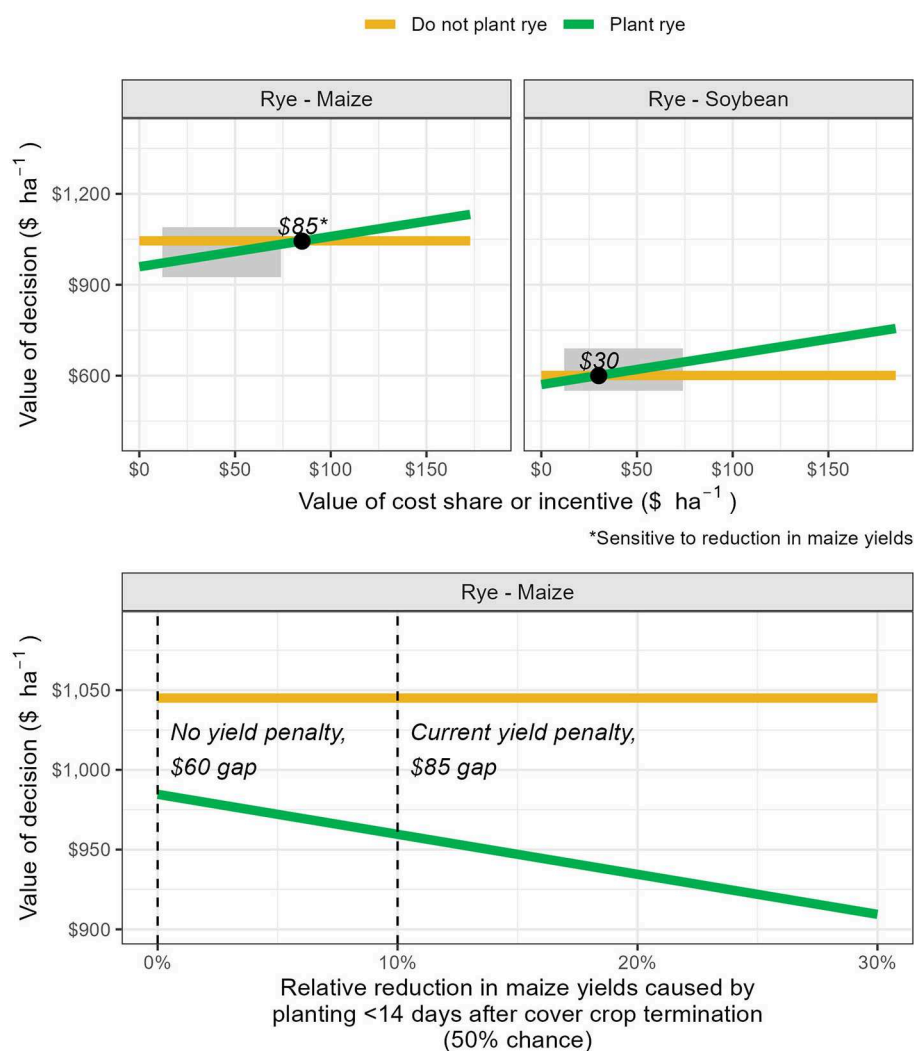


FIGURE 3

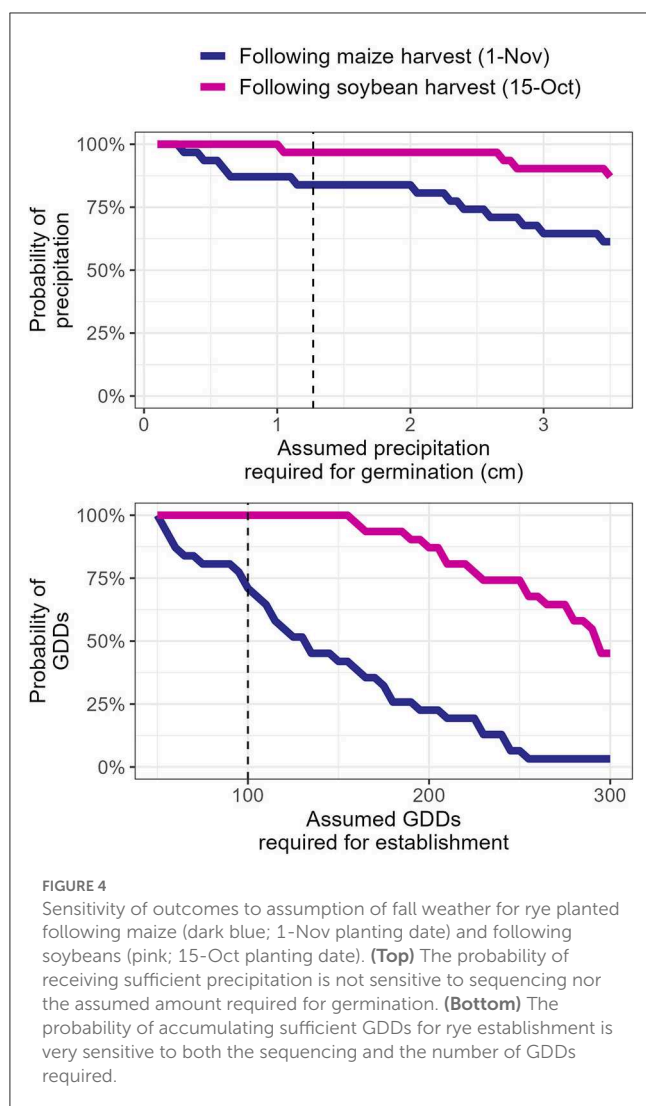
(Top) Planting a rye cover crop (green line) required \$85 ha⁻¹ and \$30 cost shares/incentives, respectively for a soybean-rye-maize and maize-rye-soybean scenario to make decision values equal to not planting a cover crop (gold line); gray box represents range of current incentive values. (Bottom) Current estimates show maize yields can be reduced by approximately 10% when maize is planted <14 days after terminating a cover crop; if agronomic research efforts were able to eliminate this yield reduction the difference in decision values would be within the range of current incentive programs.

experiencing a significant reduction in the probability of cover crop germination.

Our results also show when planting after soybean harvest, the cover crop is almost guaranteed to gain 100 GDDs in the fall (Figure 4). Conversely, after maize harvest the probability is very sensitive to how many GDDs are assumed to be needed. Our analyses highlight the need to better understand conditions that lead to successful establishment, particularly in the later months of the year. Additionally, research focused on identifying management tactics that allow for earlier cover crop planting may be most effective in increasing the probability of successful cover crop establishment in Central Iowa. For example, some producers report switching to earlier maturing soybean and maize varieties when adopting cover crops in order to plant the cover crop earlier (Plastina et al., 2020). Some areas have organized blocks of producers who share in aerial seeding costs, and custom seeding equipment/services that allows for

seeding into a standing crop are becoming more common. Our analyses indicate these types of activities are well-suited to reducing the risk associated with planting a cover crop in Central Iowa.

In the spring, the number of WFDs presented a great deal of uncertainty (Table 3). Averaged over the entire spring period (1-Apr through 31-May), there was a 79% probability of two or more WFDs in a given 2-week period, and only a 46% probability of four or more WFDs. We assumed two or more WFDs were needed to successfully complete a cover crop termination activity, and two additional WFDs were needed to complete cash crop planting activities. Therefore, producers wishing to terminate and plant within a 2-week period may not have sufficient WFDs to do so. The probability of two or more WFDs was higher in May compared to April, indicating paying producers to delay cover crop termination may also increase the chances the producer can terminate in their planned timeframe.



Our analyses indicate in Central Iowa, there is generally a high probability the fall conditions will foster cover crop establishment, and that the majority of risk occurs due to the potential for the additional management required in the spring to delay cash crop planting. A Midwestern focus group found some producers had been switching to winterkill cover crop varieties because of the difficulties associated with killing the cover crop and planting a cash crop in a timely manner in the spring (Plastina et al., 2020). For this analysis we assumed the rye cover crop could be terminated at any point, but the stage of rye growth will affect how easy it is to terminate, particularly when using mechanical termination (Creamer and Dabney, 2002; Mirsky et al., 2009). Decision support tools that help producers decide if early termination is the best choice could be beneficial in helping producers manage this risk.

4. Conclusions

Using publicly available data and reasonable assumptions, we were able to gain significant insight into localized priorities for

cover crop research. Using historical weather data, NASS surveys on WFDs, extension publications, and a partial budget for cover crop economics we were able to build a single-attribute decision model, and model decision values assuming a risk-neutral producer. Our analysis does not include possible long-term impacts such as the maintenance of productivity, long-term impacts on weeds or insects, or changes in yield stability over time, which could be incorporated in future applications of this framework. We found including only the costs of seed and fuel in cover crop economics underestimates the additional financial risk producers assume due to the extra spring work cover crops might entail in areas with limited numbers of WFDs during that time. We found there is minimal information on the number of GDDs required for a rye cover crop to successfully overwinter, and that this may have a large impact on risks associated with planting cover crops in Central Iowa. In Central Iowa, identifying ways to ensure early cover crop planting and managements that render maize yields less sensitive to rye cover crop termination timing, or that allow that reduction to be more predictable, could significantly help reduce the financial risk of planting cover crops. Furthermore, flat payments for planting cover crops may result in a moral hazard, wherein the decision value for planting a cover crop preceding a maize crop is maximized when the cover crop fails to establish in the fall. Policies that promote tiered payment structures could rectify this while still providing support for producers as they learn to manage cover crops.

Data availability statement

Publicly available datasets were analyzed in this study. The data and R code for this study are available in a public github repository: https://github.com/vanichols/Nichols_Frontiers_CoverCropRisk.

Author contributions

GN conceived of and designed the analyses, collected the data, performed the analyses, and wrote the first draft of the manuscript. CM designed the analyses and edited the manuscript. All authors contributed to the article and approved the submitted version.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

The Supplementary Material for this article can be found online at: <https://www.frontiersin.org/articles/10.3389/fsufs.2023.1040927/full#supplementary-material>

References

- Acharya, J., Bakker, M. G., Moorman, T. B., Kaspar, T. C., Lenssen, A. W., and Robertson, A. E. (2017). Time interval between cover crop termination and planting influences corn seedling disease, plant growth, and yield. *Plant Dis.* 101, 591–600. doi: 10.1094/PDIS-07-16-0975-RE
- Acharya, J., Moorman, T. B., Kaspar, T. C., Lenssen, A. W., and Robertson, A. E. (2020). Cover crop rotation effects on growth and development, seedling disease, and yield of corn and soybean. *Plant Dis.* 104, 677–687. doi: 10.1094/PDIS-09-19-1904-RE
- Aglasan, S., and Rejesus, R. M. (2021). "An analysis of crop insurance losses, cover crops, and weather in US Crop Production," in *2021 Agricultural and Applied Economics Association Annual Meeting* (Austin, TX).
- Aguilar, J., Gramig, G. G., Hendrickson, J. R., Archer, D. W., Forcella, F., et al. (2015). Crop species diversity changes in the United States: 1978–2012. *PLoS ONE* 10, e0136580. doi: 10.1371/JOURNAL.PONE.0136580
- Almasri, M. N., and Kaluarachchi, J. J. (2005). Multi-criteria decision analysis for the optimal management of nitrate contamination of aquifers. *J. Environ. Manage.* 74, 365–381. doi: 10.1016/j.jenvman.2004.10.006
- Araldi-Da-Silva, G., Kandel, Y. R., Han, G., Mueller, D. S., Helmers, M. J., Kaspar, T. C., et al. (2022). Field studies on the effect of rye cover crop on soybean root disease and productivity. *PhytoFrontiers*, 2, 192–201. Available online at: <https://apsjournals.apsnet.org/doi/epdf/10.1094/PHYTOFR-05-21-0038-R>
- Arbuckle, J. G., and Roesch-McNally, G. (2015). Cover crop adoption in Iowa: The role of perceived practice characteristics. *J. Soil Water Conserv.* 70, 418–429. doi: 10.2489/jswc.70.6.418
- Archontoulis, S. V., Castellano, M. J., Licht, M. A., Nichols, V., Baum, M., Huber, L., et al. (2020). Predicting crop yields and soil-plant nitrogen dynamics in the US Corn Belt. *Crop Sci.* 60, 721–738. doi: 10.1002/csc2.20039
- Baker, J. M., and Griffiths, T. J. (2009). Evaluating the potential use of winter cover crops in corn-soybean systems for sustainable co-production of food and fuel. *Agric. For. Meteorol.* 149, 2120–2132. doi: 10.1016/j.agrformet.2009.05.017
- Basche, A., and DeLonge, M. (2017). The impact of continuous living cover on soil hydrologic properties: a meta-analysis. *Soil Sci. Soc. Am. J.* 81, 1179–1190. doi: 10.2136/sssaj2017.03.0077
- Basche, A. D., and Roesch-McNally, G. E. (2017). Research topics to scale up cover crop use: Reflections from innovative Iowa farmers. *J. Soil Water Conserv.* 72, 59A–63A. doi: 10.2489/jswc.72.3.59A
- Baum, M. E., Archontoulis, S. V., and Licht, M. A. (2019). Planting date, hybrid maturity, and weather effects on maize yield and crop stage. *Agron. J.* 111, 303–313. doi: 10.2134/agronj2018.04.0297
- Bedford, T., and Cooke, R. (2001). *Probabilistic Risk Analysis: Foundations and Methods*. Cambridge: Cambridge University Press. doi: 10.1017/CBO9780511813597
- Bergtold, J. S., Ramsey, S., Maddy, L., and Williams, J. R. (2019). A review of economic considerations for cover crops as a conservation practice. *Renew. Agric. Food Syst.* 34, 62–76. doi: 10.1017/S1742170517000278
- Boryan, C., Yang, Z., Mueller, R., and Craig, M. (2011). Monitoring US agriculture: the US Department of Agriculture, National Agricultural Statistics Service, Cropland Data Layer Program. *Geocarto Int.* 26, 341–358. doi: 10.1080/10106049.2011.562309
- Carleton, A. E., and Cooper, C. S. (1972). Seed size effects upon seedling vigor of three forage legumes I. *Crop Sci.* 12, 183–186. doi: 10.2135/CROPSCI1972.0011183X001200020008X
- Carlson, S., and Stockwell, R. (2013). Research priorities for advancing adoption of cover crops in agriculture-intensive regions. *J. Agric. Food Syst. Commun. Develop.* 3, 125–129. doi: 10.5304/jafscd.2013.034.017
- Cegan, J. C., Filion, A. M., Keisler, J. M., and Linkov, I. (2017). Trends and applications of multi-criteria decision analysis in environmental sciences: literature review. *Environ. Syst. Decis.* 37, 123–133. doi: 10.1007/s10669-017-9642-9
- Chatterjee, N., Archontoulis, S. V., Bastidas, A., Proctor, C. A., Elmore, R. W., and Basche, A. D. (2020). Simulating winter rye cover crop production under alternative management in a corn-soybean rotation. *Agron. J.* 112, 4648–4665. doi: 10.1002/agj2.20377
- Chen, L., Rejesus, R. M., Aglasan, S., Hagen, S. C., and Salas, W. (2022). The of cover on soil erosion in the US Midwest. *J. Environ. Manage.* 324. doi: 10.1016/j.jenvman.2022.116168
- Church, S. P., Lu, J., Ranjan, P., Reimer, A. P., and Prokopy, L. S. (2020). The role of systems thinking in cover crop adoption: implications for conservation communication. *Land Use Policy* 94, 104508. doi: 10.1016/j.landusepol.2020.104508
- Clemen, R. T., and Reilly, T. (2013). *Making hard decisions with DecisionTools*. Beijing: Cengage Learning.
- Creamer, N. G., and Dabney, S. M. (2002). Killing cover crops mechanically: review of recent literature and assessment of new research results. *Am. J. Altern. Agric.* 17, 32–40. doi: 10.1079/AJAA200204
- Crossley, M. S., Burke, K. D., Schoville, S. D., and Radeloff, V. C. (2021). Recent collapse of crop belts and declining diversity of US agriculture since 1840. *Glob. Chang. Biol.* 27, 151–164. doi: 10.1111/gcb.15396
- Daigh, A. L., Helmers, M. J., Kladvivko, E., Zhou, X., Goeken, R., Cavdini, J., et al. (2014). Soil water during the drought of 2012 as affected by rye cover crops in fields in Iowa and Indiana. *J. Soil Water Conserv.* 69, 564–573. doi: 10.2489/jswc.69.6.564
- Dickey, E. C., Shelton, D. P., Jasa, P. J., and Peterson, T. R. (1985). Soil erosion from tillage systems used in soybean and corn residues. *Trans. ASAE* 28, 1124–1130. doi: 10.13031/2013.32399
- Do, H., Luedeling, E., and Whitney, C. (2020). Decision analysis of agroforestry options reveals adoption risks for resource-poor farmers. *Agron. Sustain. Dev.* 40, 1–12. doi: 10.1007/s13593-020-00624-5
- Edwards, W. (2020). *The Number of Days Suitable for Fieldwork in Iowa is Shrinking*. Iowa State University Extension and Outreach. Available online at: <https://www.extension.iastate.edu/agdm/articles/edwards/EdwMar20.html> (accessed September 2, 2022).
- Feyerisen, G. W., Wilson, B. N., Sands, G. R., Strock, J. S., and Porter, P. M. (2006). Potential for a rye cover crop to reduce nitrate loss in southwestern Minnesota. *Agron. J.* 98, 1416–1426. doi: 10.2134/agronj2005.0134
- Finney, D. M., White, C. M., and Kaye, J. P. (2016). Biomass production and carbon/nitrogen ratio influence ecosystem services from cover crop mixtures. *Agron. J.* 108, 39–52. doi: 10.2134/agronj15.0182
- Fisher, K. A., Momen, B., and Kratochvil, R. J. (2011). Is broadcasting seed an effective winter cover crop planting method? *Agron. J.* 103, 472–478. doi: 10.2134/agronj2010.0318
- Food and Agriculture Organization (FAO) (2020). *FAOSTAT Crops and Livestock Products*. Available online at: <https://www.fao.org/faostat/en/#data/QCL> (accessed September 1, 2022).
- Gandorfer, M., Pannell, D., and Meyer-Aurich, A. (2011). Analyzing the effects of risk and uncertainty on optimal tillage and nitrogen fertilizer intensity for field crops in Germany. *Agric. Syst.* 104, 615–622. doi: 10.1016/j.agry.2011.06.004
- Gupta, S. (2022). *Sensitivity Analysis in Excel. One and Two Variable Data Table*. Available online at: <https://www.wallstreetmojo.com/sensitivity-analysis-in-excel/> (accessed September 1, 2022).

- Hamilton, A. V., Mortensen, D. A., and Allen, M. K. (2017). The state of the cover crop nation and how to set realistic future goals for the popular conservation practice. *J. Soil Water Conserv.* 72, 111A–115A. doi: 10.2489/jswc.72.5.111A
- Hardaker, J. B., Lien, G., Anderson, J. R., and Huirne, R. B. (2015). *Coping With Risk in Agriculture: Applied Decision Analysis*. Wallingford: CABI Publishing. doi: 10.1079/9781780645742.0000
- Hatfield, J. L., McMullen, L. D., and Jones, C. S. (2009). Nitrate-nitrogen patterns in the Raccoon River Basin related to agricultural practices. *J. Soil Water Conserv.* 64, 190–199. doi: 10.2489/jswc.64.3.190
- Hijmans, R. J., Choe, H., and Perlman, J. (2016). Spatiotemporal patterns of field crop diversity in the United States, 1870–2012. *Agric. Environ. Lett.* 1, 160022. doi: 10.2134/acl2016.05.0022
- Hirsh, S. M., Duiker, S. W., Graybill, J., Nichols, K., and Weil, R. R. (2021). Scavenging and recycling deep soil nitrogen using cover crops on mid-Atlantic, USA farms. *Agric. Ecosyst. Environ.* 309, 107274. doi: 10.1016/j.agee.2020.107274
- Howard, R., and Abbas, A. E. (2015). *Foundations of Decision Analysis*. Pearson. Available online at: <https://www.amazon.com/Foundations-Decision-Analysis-Ronald-Howard/dp/0132336243>
- Howard, R. A. (1988). Decision analysis: practice and promise. *Manage. Sci.* 34, 679–695. doi: 10.1287/mnsc.34.6.679
- Hubbard, D. W. (2020). *The Failure of Risk Management: Why It's Broken and How to Fix It*. New York: John Wiley and Sons. doi: 10.1002/9781119521914
- Iowa Environmental Mesonet (IEM) (2022). Available online at: <https://mesonet.agron.iastate.edu> (accessed September 1, 2022).
- Iowa State University Extension (2018). *Economics of Cover Crops Worksheets*. Available online at: <https://www.extension.iastate.edu/agdm/crops/html/a1-91.html> (accessed September 1, 2022).
- Iowa State University Extension (2022). *Estimated Costs of Crop Production*. Available online at: <https://www.extension.iastate.edu/agdm/crops/html/a1-20.html> (accessed September 1, 2022).
- Johnson, T. J., Kaspar, T. C., Kohler, K. A., Corak, S. J., and Logsdon, S. D. (1998). Oat and rye overseeded into soybean as fall cover crops in the upper Midwest. *J. Soil Water Conserv.* 53, 276–279.
- Kantar, M., and Porter, P. (2014). Relationship between planting date, growing degree days and the winter rye (*Secale cereale* L.) variety “Rymin” in Minnesota. *Crop Manage.* 13, 1–9. doi: 10.2134/CM-2013-0096-RS
- Kaplan, S., and Garrick, B. J. (1981). On the quantitative definition of risk. *Risk Anal.* 1, 11–27. doi: 10.1111/j.1539-6924.1981.tb01350.x
- Kaspar, T. C., Jaynes, D. B., Parkin, T. B., and Moorman, T. B. (2007). Rye cover crop and gamagrass strip effects on NO₃ concentration and load in tile drainage. *J. Environ. Qual.* 36, 1503–1511. doi: 10.2134/jeq2006.0468
- Kaspar, T. C., Jaynes, D. B., Parkin, T. B., Moorman, T. B., and Singer, J. W. (2012). Effectiveness of oat and rye cover crops in reducing nitrate losses in drainage water. *Agric. Water Manage.* 110, 25–33. doi: 10.1016/j.agwat.2012.03.010
- Kaspar, T. C., Radke, J. K., and Laflen, J. M. (2001). Small grain cover crops and wheel traffic effects on infiltration, runoff, and erosion. *J. Soil Water Conserv.* 56, 160–164. Available online at: <https://www.jswconline.org/content/56/2/160>
- Kessler, A., Archontoulis, S. V., and Licht, M. A. (2020). Soybean yield and crop stage response to planting date and cultivar maturity in Iowa, USA. *Agron. J.* 112, 382–394. doi: 10.1002/ajg2.20053
- Kruep, F. S., Mizero, S. M., Redfearn, D., and Basche, A. (2022). Assessing how cover crops close the soil health gap in on-farm experiments. *Agric. Environ. Lett.* 7, e20088. doi: 10.1002/acl2.20088
- Kucharik, C. J. (2008). Contribution of planting date trends to increased maize yields in the central United States. *Agron. J.* 100, 328–336. doi: 10.2134/agronj2007.0145
- Lee, D., Arbuckle, J. G., Zhu, Z., and Nowatzke, L. (2018). Conditional causal mediation analysis of factors associated with cover crop adoption in Iowa, USA. *Water Resour. Res.* 54, 9566–9584. doi: 10.1029/2017WR022385
- Marcillo, G. S., Carlson, S., Filbert, M., Kaspar, T., Plastina, A., and Miguez, F. E. (2019). Maize system impacts of cover crop management decisions: a simulation analysis of rye biomass response to planting populations in Iowa, USA. *Agric. Syst.* 176, 102651. doi: 10.1016/j.agry.2019.102651
- Marcillo, G. S., and Miguez, F. E. (2017). Corn yield response to winter cover crops: an updated meta-analysis. *J. Soil Water Conserv.* 72, 226–239. doi: 10.2489/jswc.72.3.226
- Martinez-Feria, R., Nichols, V., Basso, B., and Archontoulis, S. (2019). Can multi-strategy management stabilize nitrate leaching under increasing rainfall? *Environ. Res. Lett.* 14, 124079. doi: 10.1088/1748-9326/ab5ca8
- Martinez-Feria, R. A., Dietzel, R., Liebman, M., Helmers, M. J., and Archontoulis, S. V. (2016). Rye cover crop effects on maize: a system-level analysis. *Field Crops Res.* 196, 145–159. doi: 10.1016/j.fcr.2016.06.016
- Maryland Department of Agriculture (2022). *Maryland's 2022–2023 Cover Crop Program*. Available online at: https://mda.maryland.gov/resource_conservation/Pages/cover_crop.aspx (accessed September 1, 2022).
- Mirsky, S. B., Curran, W. S., Mortensen, D. A., Ryan, M. R., and Shumway, D. L. (2009). Control of cereal rye with a roller/crimper as influenced by cover crop phenology. *Agron. J.* 101, 1589–1596. doi: 10.2134/agronj2009.0130
- Mitsch, W. J., Day, J. W., Gilliam, J. W., Groffman, P. M., Hey, D. L., Randall, G. W., et al. (2001). Reducing nitrogen loading to the Gulf of Mexico from the Mississippi River Basin: strategies to counter a persistent ecological problem: ecotechnology—the use of natural ecosystems to solve environmental problems—should be a part of efforts to shrink the zone of hypoxia in the Gulf of Mexico. *Bioscience* 51, 373–388. doi: 10.1641/0006-3568(2001)051(0373:RNLTTG)2.0.CO;2
- Mohler, C. L., Liebman, M., and Staver, C. P. (2009). *Weed Life History: Identifying Vulnerabilities. Ecological Management of Agricultural Weeds*. Cambridge: Cambridge University Press, 40–98. doi: 10.1017/CBO9780511541810
- Moore, E. B., Wiedenhoef, M. H., Kaspar, T. C., and Cambardella, C. A. (2014). Rye cover crop effects on soil quality in no-till corn silage-soybean cropping systems. *Soil Sci. Soc. Am. J.* 78, 968–976. doi: 10.2136/sssaj2013.09.0401
- Nelson, H., and Bennett, S. (2018). *Cereal Rye Cover Crop for Reducing Herbicides in Soybeans. Practical Farmers of Iowa Cooperators' Program*. Available online at: <https://practicalfarmers.org/research/cereal-rye-cover-crop-for-reducing-herbicides-in-soybeans/> (accessed February 2022).
- Nichols, V., English, L., Carlson, S., Gailans, S., and Liebman, M. (2020a). Effects of long-term cover cropping on weed seedbanks. *Front. Agron.* 2, 591091. doi: 10.3389/fagro.2020.591091
- Nichols, V., Martinez-Feria, R., Weisberger, D., Carlson, S., Basso, B., and Basche, A. (2020b). Cover crops and weed suppression in the US Midwest: a meta-analysis and modeling study. *Agricultural and Environmental Letters* 5, e20022. doi: 10.1002/acl2.20022
- Nichols, V. A., Moore, E. B., Gailans, S., Kaspar, T. C., and Liebman, M. (2022). Site-specific effects of winter cover crops on soil water storage. *Agrosyst. Geosci. Environ.* 5, e20238. doi: 10.1002/agg2.20238
- Pantoja, J. L., Woli, K. P., Sawyer, J. E., and Barker, D. W. (2015). Corn nitrogen fertilization requirement and corn-soybean productivity with a rye cover crop. *Soil Sci. Soc. Am. J.* 79, 1482–1495. doi: 10.2136/sssaj2015.02.0084
- Park, B., Rejesus, R. M., Aglasan, S., Che, Y., Hagen, S. C., and Salas, W. (2022). Payments from agricultural conservation programs and cover crop adoption. *Appl. Econ. Perspect. Policy*. doi: 10.1002/aep.13248
- Patel, S., Sawyer, J. E., and Lundvall, J. P. (2019). Can management practices enhance corn productivity in a rye cover crop system? *Agron. J.* 111, 3161–3171. doi: 10.2134/agronj2019.03.0158
- Plastina, A., Liu, F., Miguez, F., and Carlson, S. (2020). Cover crops use in Midwestern US agriculture: perceived benefits and net returns. *Renew. Agric. Food Syst.* 35, 38–48. doi: 10.1017/S1742170518000194
- Qi, Z., Helmers, M. J., and Lawlor, P. A. (2008). “Effect of different land covers on nitrate-nitrogen leaching and nitrogen uptake in Iowa,” in *2008 Providence, Rhode Island, June 29-July 2, 2008* (p. 1). American Society of Agricultural and Biological Engineers.
- Quinn, D. J., Poffenberger, H. J., Leuthold, S. J., and Lee, C. D. (2021). Corn response to in-furrow fertilizer and fungicide across rye cover crop termination timings. *Agron. J.* 113, 3384–3398. doi: 10.1002/ajg2.20723
- Ramírez-García, J., Carrillo, J. M., Ruiz, M., Alonso-Ayuso, M., and Quemada, M. (2015). Multicriteria decision analysis applied to cover crop species and cultivars selection. *Field Crops Res.* 175, 106–115. doi: 10.1016/j.fcr.2015.02.008
- Repo, A. J. (1989). The value of information: approaches in economics, accounting, and management science. *J. Am. Soc. Inform. Sci.* 40, 68–85. doi: 10.1002/(SICI)1097-4571(198903)40:2<68::AID-ASIS2>3.0.CO;2-J
- Rundquist, S., and Carlson, S. (2017). *Mapping Cover Crops on Corn and Soybeans in Illinois, Indiana and Iowa, 2015–2016*. Environmental Working Group. Available online at: <https://www.ewg.org/research/mapping-cover-crops-corn-and-soybeans-illinois-indiana-and-iowa-2015-2016> (accessed September 1, 2022).
- Sarrantonio, M. (1994). *Northeast Cover Crop Handbook*. Kutztown, PA: Soil Health Series. Rodale Institute.
- Seifert, C. A., Azzari, G., and Lobell, D. B. (2018). Satellite detection of cover crops and their effects on crop yield in the Midwestern United States. *Environ. Res. Lett.* 13, 064033. doi: 10.1088/1748-9326/aac4c8
- Shackelford, G. E., Kelsey, R., Sutherland, W. J., Kennedy, C. M., Wood, S. A., Gennet, S., et al. (2019). Evidence synthesis as the basis for decision analysis: a method of selecting the best agricultural practices for multiple ecosystem services. *Front. Sustain. Food Systems* 3, 83. doi: 10.3389/fsufs.2019.00083
- Singer, J. W. (2008). Corn belt assessment of cover crop management and preferences. *Agron. J.* 100, 1670–1672. doi: 10.2134/agronj2008.0151

- Sustainable Agriculture Research and Education (SARE) (2020). *National Cover Crop Surveys*. Available online at: <https://www.sare.org/publications/cover-crops/national-cover-crop-surveys/> (accessed September 2, 2022).
- Syswerda, S. P., Basso, B., Hamilton, S. K., Tausig, J. B., and Robertson, G. P. (2012). Long-term nitrate loss along an agricultural intensity gradient in the Upper Midwest USA. *Agric. Ecosyst. Environ.* 149, 10–19. doi: 10.1016/j.agee.2011.12.007
- Talukder, B., Blay-Palmer, A., Hipel, K. W., and vanLoon, G. W. (2017). Elimination method of multi-criteria decision analysis (MCDA): a simple methodological approach for assessing agricultural sustainability. *Sustainability* 9, 287. doi: 10.3390/su9020287
- Thapa, R., Mirsky, S. B., and Tully, K. L. (2018). Cover crops reduce nitrate leaching in agroecosystems: A global meta-analysis. *J. Environ. Qual.* 47, 1400–1411. doi: 10.2134/jeq2018.03.0107
- Thompson, N. M., Reeling, C. J., Fleckenstein, M. R., Prokopy, L. S., and Armstrong, S. D. (2021). Examining intensity of conservation practice adoption: evidence from cover crop use on US Midwest farms. *Food Policy* 101, 102054. doi: 10.1016/j.foodpol.2021.102054
- Urban, D. W., Roberts, M. J., Schlenker, W., and Lobell, D. B. (2015). The effects of extremely wet planting conditions on maize and soybean yields. *Clim. Change* 130, 247–260. doi: 10.1007/s10584-015-1362-x
- USDA National Agricultural Statistics Service (USDA NASS) (2018). *National Crop Progress—Terms and Definitions*. Available online at: https://www.nass.usda.gov/Publications/National_Crop_Progress/Terms_and_Definitions/index.php (accessed September 1, 2022).
- USDA National Agricultural Statistics Service (USDA NASS) (2022). *Quick Stats*. Available online at: <https://quickstats.nass.usda.gov> (accessed September 1, 2022).
- USDA National Agricultural Statistics Service Cropland Data Layer (USDA NASS CDL) (2021). *Published Corn Data Layer*. Washington, DC: USDA-NASS. Available online at: <https://nassgeodata.gmu.edu/CropScape/> (accessed August 10, 2021).
- USDA Risk Management Agency (USDA RMA) (2019). *2020 Cover Crops Insurance and NRCS Cover Crop Termination Guidelines*. Available online at: <https://www.rma.usda.gov/en/News-Room/Frequently-Asked-Questions/2020-Cover-Crops-Insurance-and-NRCS-Cover-Crop-Termination-Guidelines> (accessed September 1, 2022).
- Wilson, M. L., Allan, D. L., and Baker, J. M. (2014). Aerially seeding cover crops in the northern US Corn Belt: limitations, future research needs, and alternative practices. *J. Soil Water Conserv.* 69, 67A–72A. doi: 10.2489/jswc.69.3.67A
- Wilson, M. L., Baker, J. M., and Allan, D. L. (2013). Factors affecting successful establishment of aerially seeded winter rye. *Agron. J.* 105, 1868–1877. doi: 10.2134/agronj2013.0133
- Yoder, L., Houser, M., Bruce, A., Sullivan, A., and Farmer, J. (2021). Are climate risks encouraging cover crop adoption among farmers in the southern Wabash River Basin? *Land Use Policy* 102, 105268. doi: 10.1016/j.landusepol.2020.105268



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EDITED BY

Carol Williams,
University of Wisconsin-Madison, United States

REVIEWED BY

Anne Elise Stratton,
University of Hohenheim, Germany
R. Eugene Turner,
Louisiana State University, United States

*CORRESPONDENCE

Tara Maireid Conway
✉ conwa304@umn.edu

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An agroecological turn in intermediating sustainability transitions with continuous living cover

Tara Maireid Conway*

Forever Green Initiative, Department of Agronomy and Plant Genetics, University of Minnesota, Saint
Paul, MN, United States

Continuous living cover's (CLC) perennial and winter annual crop varieties present a novel opportunity to increase the diversity and resiliency of agroecological systems in the Mid-Continent of North America. However, transforming the predominant agri-food regime remains a complex and daunting undertaking. In the face of such complexity, a recent body of literature highlights the particular importance of intermediaries in facilitating sustainability transition processes, which CLC agriculture's proponents can draw upon. Intermediaries can be defined as actors or organizations that positively influence sustainability transition processes by linking diverse entities, networks, institutions, activities and their related skills, knowledges, and resources. Simultaneously, agroecology, in the more political understanding of the term, can serve as an evaluative framework for agri-food transition processes to augment our understanding of intermediaries in sustainability transitions. This mini-review presents an overview of the emerging sustainability transition intermediary literature, an introduction to CLC agriculture's transition intermediaries, and the research gaps highlighted from an agroecological perspective. Integrating an agroecological lens attentive to the science, practice, and politics of intermediating agricultural transitions, this review proposes an adapted framework to understand and assess CLC agriculture's intermediaries. Thus, CLC agriculture presents a unique opportunity to iteratively draw upon and advance the sustainability transition intermediary literature.

KEYWORDS

sustainability transitions, intermediaries, agroecology, continuous living cover, food systems

1. Introduction

Continuous living cover (CLC) agriculture offers a compelling alternative agricultural paradigm amidst our food system's compounding ecological and socio-cultural crises. The dominant summer annual cropping systems of the Mid-Continent of North America are highly productive, yet leave soil exposed for the majority of the year, resulting in an array of environmental disservices (Crews et al., 2018). In response, the University of Minnesota's Forever Green Initiative, Green Lands Blue Waters (GLBW), and partners are working to develop a suite of perennial and winter annual crops to augment the prevailing summer annual system to enhance soil coverage and deliver additional income streams to farmers, providing sustainable water management and other critical socio-ecological benefits. This modified agronomic system is aptly referred to as continuous living cover, due to its premise of providing consistent plant cover to the soils of the Upper Midwest, described elsewhere as "plant cover on the soil and roots in the ground all year long" (Jewett and Schroeder, 2015).

The emergence of the term “Continuous living cover” is most readily tied to the formation of Green Lands Blue Waters, an organization with a singular focus on advancing CLC, in 2004 (greenlandsbluewater.org). However, continuous ground cover has long been practiced as an Indigenous agricultural technique (Citizen Potawatomi Nation, 2020), and thus the premise of continuously covering soil with plant matter extends far beyond GLBW’s inception. GLBW describes CLC as five strategies: agroforestry, perennial biomass, perennial forage, perennial grains, and cover crops/winter annuals, and emphasizes the on-farm integration and stacking of these strategies (Green Lands Blue Waters, 2021). CLC’s inclusion of a suite of strategies to achieve on-farm diversity, and particularly CLC’s incorporation of marketable winter annual crops such as winter camelina (*Camelina sativa*) and pennycress (*Thlaspi arvense*), differentiates the approach from a singular focus on perennialization or cover cropping. CLC has been invoked as an example of multifunctional agriculture, or the simultaneous production of both ecosystem services and agricultural commodities (Jordan and Warner, 2010), and as a pathway to landscape level change toward more resilient agricultural systems (Runck et al., 2013). However, novel crops and cropping systems alone do not change food systems nor do they guarantee a more just and equitable system (Streit Krug and Tesdell, 2020). As such, CLC, as a suite of crops and cropping systems, must be distinguished from the approaches taken to move CLC into the landscape and the resulting socio-ecological systems.

For example, the Forever Green Initiative, a primary driver of CLC crop domestication and improvement [in the U.S. Mid-Continent], understands change to be driven in part by market pull, or the profitability of CLC crops for farmers. In practice, this means devising entirely new supply chains for novel grain and oilseed crops such as Kernza™ (*Thinopyrum intermedium*) and winter camelina (Forever Green Initiative, 2020) alongside robust research and development that must span plant breeding, agronomy, food science, and more. This crop system scaling process has elsewhere been conceptualized as sustainable commercialization (Jordan et al., 2016) and new food crop domestication (Van Tassel et al., 2020), both of which call for an integrated attentiveness to a crop’s genetics, agronomics, and socio-cultural infrastructure, including markets, policies, educational practices, and values. Proponents of CLC agriculture must contend with the “wicked problem” (Rittel and Webber, 1973; Peterson, 2009) of the dominant agri-food regime, specifically the complex interdependencies, uncertainty, and contestation inherent to altering the prevailing system.

In the face of such complexity and uncertainty, there is a growing body of literature that highlights the particular importance of intermediary actors in facilitating many aspects of sustainability transition processes (Kivimaa et al., 2019a; Kanda et al., 2020; Sovacool et al., 2020). Sustainability transitions have been defined as radical shifts to new kinds of socio-technical systems away from unsustainable consumption and production patterns (Köhler et al., 2019). The concept of transition intermediaries has strong ties with Geels’s multi-level perspective (MLP), which understands sustainability transitions to arise through interactions between three analytical levels: niches, regimes, and socio-technical landscapes (Geels, 2019). Niches are understood to be spaces for

radical innovation that operate outside of the prevailing regime, which is the locus of stability for the dominant socio-technical system, made up of an established web of rules, beliefs, practices, and institutions. Meanwhile the landscape level represents the wider socio-technical context, such as macroeconomic patterns, political ideologies, and material realities like climate (Geels, 2011). The MLP theorizes that rare “windows of opportunity” for transformation of entrenched regimes (e.g., industrial agriculture) can arise when bottom-up momentum from the niche level is met with landscape level pressure. Intermediaries are considered to be significant actors in orchestrating this niche-regime-landscape alignment and thus may prove critical to the advancement of CLC agriculture. However, both the MLP and the associated field of transition intermediaries have been critiqued for their (1) assumption that green innovations are inherently positive, (2) lack of interrogation of the outcomes or consequences of a socio-technical shift toward “more sustainable” innovations, and (3) their disregard for distributional consequences (Avelino and Rotmans, 2009; Lawhon and Murphy, 2012; Geels, 2019; Magda et al., 2021). Therefore, the study of CLC transition intermediaries can benefit from an additional lens that is attentive to such shortcomings.

CLC agriculture’s proponents in Minnesota aspire toward, “healthy soils, clean water, and a more resilient and equitable agricultural economy” (Forever Green Partnership, 2022) and claim that “CLC, implemented equitably with people and communities at the center, can bring about both environmental and social changes sorely needed in agriculture” (Green Lands Blue Waters, 2021). CLC crops must scale both widely across the landscape and deeply into culture, values, and mindsets (Lam et al., 2020) in order to realize these aspirations. This gap between CLC as a suite of scientific enterprises and CLC as a driver of regional agricultural, environmental, and social transformation is perhaps best assessed through the lens of agroecology, which seeks systemic transformation to build just food system futures (Nicklay et al., 2023). Agroecology can be understood as the integration of sciences, practices, and politics (Wezel et al., 2009; Bell and Bellon, 2021) where things like plant breeding, relationship-building, and food justice activism can intermingle to seek transformation. Agroecology is a participatory, action-oriented, and transdisciplinary framework (Méndez et al., 2013) with a political orientation toward supporting transformations led through community self-organization and participatory governance processes (Anderson et al., 2019). Currently, CLC agriculture can be described as agroecological only in the narrowest understanding of agroecology as a scientific approach of applying ecological principles to agriculture (Wezel et al., 2009). It remains undetermined as to whether CLC agriculture can be described as agroecological in the more political understanding of the term as a transformative process that centers power, governance, and democracy (Anderson et al., 2019). Thus, agroecology presents itself as an evaluative framework to assess the process of transformation to CLC agriculture, where intermediaries ostensibly function as potent agents of transformation in regional agri-food systems. This mini-review presents an overview of the emerging sustainability transition intermediary literature, an introduction to CLC agriculture’s transition intermediaries, and the prospective advancements highlighted from an agroecological perspective.

2. Sustainability transition intermediaries

Intermediary is a general term that refers to any individual, organization, or thing that serves as a link between multiple entities. The term has been employed in diverse fields from finance (Boyd and Prescott, 1986) to social networking applications (Sylvain, 2018), and in the case of this mini-review, sustainability transitions. Sustainability transition intermediaries are more specifically defined as, “actors and platforms that positively influence sustainability transition processes by linking actors and activities, and their related skills and resources, or by connecting transition visions and demands of networks of actors with existing regimes in order to create momentum for socio-technical system change, to create new collaborations within and across niche technologies, ideas and markets, and to disrupt dominant unsustainable socio-technical configurations” (Kivimaa et al., 2019a). Noteworthy in the definition is the intrinsically positive understanding of transition intermediaries’ role in facilitating change, which warrants skepticism given that intermediaries are understood to have detrimental impacts in other fields ranging from agri-food supply chains (Huria and Pathania, 2018) to cultural taste-making (Edwards, 2012). This critique will be elaborated upon later in the mini-review, following an overview of the current literature on transition intermediaries.

Sustainability transition intermediaries are currently understood to advance transitions through bridging and brokering knowledge (Goodrich et al., 2020), transferring technology (Howells, 2006), enabling learning processes (Klerkx and Leeuwis, 2009), facilitating dialogue and social interaction among diverse stakeholders (Steyaert et al., 2016), creating new markets (Kivimaa et al., 2020a) mobilizing resources (Polzin et al., 2016), and political maneuvering (Kivimaa et al., 2020b). Their capacity to balance objectivity and subjectivity through clarifying and coordinating, while also eliciting diverse perspectives to inform their evolving understanding of complex situations is thought to be particularly important in the tackling of sustainability’s “wicked problems” (Steyaert et al., 2016). It remains contested whether transition intermediaries should strive for neutrality (Pielke, 2007; Klerkx and Leeuwis, 2009; Parag and Janda, 2014; Kant and Kanda, 2019) or if remaining neutral is possible given their inherent orientation toward change (Moss, 2009; Kivimaa, 2014). Despite the varying roles ascribed to intermediaries across the literature, they are consistently defined by *what they do* (Bergek, 2020), which is acting in-between networks, actors, institutions, scales, and/or spatial extents.

Although there is recognition that intermediaries can be formal or informal (Kivimaa et al., 2019a; Kanda et al., 2020) and range from individual actors to organizations (Köhler et al., 2019), the transition intermediary literature is primarily based on analyses of formal intermediary organizations in Europe. Representative examples include the Berlin Center of Competence for Water, which funds and coordinates regional water research and technology development (Moss, 2009); Doarpswurk, a semi-governmental organization that supports Frisian villages in resilient transition processes (Warbroek et al., 2018), and Malmo Cleantech

City, which supports the creation of jobs and employment in the clean technology sector (Kanda et al., 2020).

Kivimaa et al.’s (2019a) seminal systematic review of the sustainability transition intermediary literature resulted in a distillation of five types of intermediaries: systemic, which operate on all levels of a system and promote a change agenda; regime-based, which are tied to the prevailing regime but with a mandate to promote a transition; niche/grassroots, which attempt to experiment and advance a particular niche outside the predominant regime; process, which help facilitate a transition process in its day-to-day machinations; and user, which connect niche technologies to users and help articulate future demand to the broader socio-technical system. They found that while systemic and niche intermediaries hold particular importance, a robust ecology of all intermediary types is needed to support the multifaceted and dynamic process of a sustainability transition. Other research has indicated that intermediaries can have diverse and conflicting agendas (Kanda et al., 2020; Vihemäki et al., 2020) and that intermediary ecologies shift over time (van Lente et al., 2011; Kivimaa et al., 2019b). As such, interaction and coordination amongst various intermediaries is deemed essential (Mignon and Kanda, 2018). Additionally, transition processes are thought to have distinct phases, as in Kivimaa et al.’s (2019b) predevelopment, acceleration, and stabilization. Accordingly, intermediaries have particular roles in these phases, from supporting experimentation and making space for niche technologies in predevelopment (Kivimaa et al., 2019b) to creating markets, managing conflicts, and increasing cohesion during acceleration (Kivimaa et al., 2020a). The same intermediary may not be able to fulfill all these functions, potentially enabling excessive redundancy and competition amongst intermediaries as a transition process evolves (Kanda et al., 2020; van Boxtael et al., 2020). These diverse findings from the transition intermediary literature can be both drawn upon and advanced through applications to CLC agriculture.

3. Advancing the transition intermediary literature through CLC

Establishment of CLC crops and systems onto the landscape is a current, ongoing effort and as such, research related to CLC’s intermediaries is only recently emerging. For example, Muckey (2019) analysis of the viability of continuous living cover crop Kernza in Southern Minnesota cited effective communication and supply chain linkages as significant barriers to commercialization, while Ray’s (2020) research into CLC crop winter camelina’s supply chain development indicated a need for coordinated systems for research dissemination, collaboration with policy-makers, and general personnel capacity for systemic coordination. These early results indicated synergies between intermediaries and CLC supply chain development, specifically citing the lack of personnel to carry out intermediary functions as a significant barrier. Additionally, research on CLC crop technical service providers, who provide intermediary functions, indicates the critical importance of empathy, rapport, emotional intelligence, and relatability (Peters et al., 2021); elements that are mostly overlooked in the current sustainability transition intermediary literature.

More recently, emerging literature highlights that actors involved in the commercialization, adoption, and scaling of CLC agriculture actively identify as intermediaries (Cureton et al., *in review*) and that an intentional polycentric governance network is being built to systemically advance CLC agriculture (Jordan et al., *in review*). These contributions to the transition intermediary literature are novel, due to their reflexivity from an intermediary perspective and U.S. agri-food context.

Regarding reflexivity, Cureton et al. actively engage with Kivimaa et al. (2019a) intermediary typology and situate themselves, as CLC crop Kernza™ commercialization staff, as simultaneously operating within the systemic, niche, user, and niche-regime categories (Cureton et al., *in review*). Their contribution to the literature is rich with examples of CLC intermediary functions, with some examples including: brokering technology and technical resources to growers, cleaners, dehullers, millers, brewers, bakers, and more; observing and articulating innovation rhythms and trajectories to stakeholders; navigating tweaks to dominant policy regimes; incorporating novel crop varieties into cultural institutions; and systemically aligning niche-regime-landscape interactions to promote systemic transformation. While the authors consider Kernza™ to still be in early phases of commercial development, it is noteworthy that this group of CLC commercialization staff claims to transcend the boundaries of formerly established intermediary typologies. Additionally, Cureton et al.'s collective reflexivity is a welcome contribution to the literature, as there have been calls to intentionally introduce collective intermediary activities in research (Vilas-Boas et al., 2022), given the alleged importance of intermediary coordination. Relatedly, Jordan et al. (*in review*) cite the transition intermediary literature in their description of a "Learning and Experimentation Network" composed of individuals from various institutions working in the commercialization and scaling of CLC crops. This network convenes to share their experiences and learnings from sustainable supply chain development to help inform collective scaling efforts. However, Jordan et al. share reflections on the slow and difficult process of establishing the group as self-governing and self-directed, indicating that CLC intermediaries might not find much value in the group. Reflections on the complex, uncertain, and difficult work of scaling CLC crops is a worthwhile addition to transition intermediary literature, given the field's current focus on longitudinal, retrospective analyses that might flatten the lived complexity of sustainability transition processes (Murto et al., 2020).

3.1. CLC intermediaries as political actors

In addition to the ecological benefits of continuous living cover, such as improved soil health, water quality, and pollinator habitat, Jordan et al. (*in review*) underscore the importance of equity and social sustainability in their vision of diverse, regional CLC systems. Similarly, Cureton et al. highlight the incorporation of social sustainability research as a core approach to legitimacy-building for the novel perennial grain crop Kernza™ (Cureton et al., *in review*). Attentiveness to multiple aspects of sustainability has been mostly lacking in the current intermediary literature,

highlighted in Sovacool et al.'s findings that European transitions toward renewable energy systems have furthered injustice and intensified pre-existing vulnerabilities (Sovacool et al., 2021).

It should be noted that intermediaries, in their focus on linking diverse entities, are imminently concerned with building relations, and not all relationships are positive. For instance, food justice scholars point to racial capitalism and settler colonialism as defining sets of agri-food relations (Slocum and Cadieux, 2015; Black, 2022), indicating that our current system is not merely defined by an absence of connectivity but rather an undesirable set of relationships. Another prescient example lies in agricultural supply chain intermediaries, who have been charged with inflating food prices (Huria and Pathania, 2018), accumulating power through market consolidation, and exploiting farmers and farmworkers (De Fazio, 2016; Lakhani et al., 2021). Although supply chain intermediaries are distinct from transition intermediaries, the potential for negative outcomes through more connectivity in these examples warrants deeper consideration.

Instead, sustainability transition intermediaries are near-universally spoken of as inherently good despite the literature's occasional acknowledgment of intermediaries' diverse, conflicting agendas, competition, and potential for excessive redundancy. The seminal papers in the young field assert that intermediaries positively affect transition processes (Kivimaa et al., 2019a), are paramount during all phases of the transition process (Kivimaa et al., 2019b), and have a catalyzing effect on the processes they engage with (Klerkx and Leeuwis, 2009; Kanda et al., 2020). While there has been recognition that intermediaries can theoretically enable and disable transitions in equal measure (Janda and Parag, 2013; Kivimaa et al., 2020b) and calls to consider the negative impacts of intermediaries (Moss, 2009; Mignon, 2017), these suggestions remain mostly hypothetical and lacking in empirical engagement. This overly simplistic description of intermediaries is at odds with the complex nature of sustainability transitions, which understands change-making to be inherently political, defined by disagreements regarding the direction and steering of transition processes (Köhler et al., 2019), and always resulting in both winners and losers (Wigboldus et al., 2016). Cureton et al. acknowledge this dimension, citing their agency in potentially shaping CLC innovation trajectories and role in intervening when others attempt to change innovation trajectories in ways that are perceived to be at odds with more broadly shared values (Cureton et al., *in review*). Thus, Jordan et al. and Cureton et al.'s attentiveness to multiple and potentially conflicting aspects of sustainability transitions appears to be an important dimension to integrate in the transition intermediary literature. CLC agriculture provides an opportunity to further this area of inquiry, taking seriously the political agency of transition intermediaries.

3.2. CLC intermediaries as practitioners

Current findings are not yet robust enough to propose an alternative typology for CLC's transition intermediaries, primarily because the main findings to-date (Cureton et al., *in review*) are based on one CLC perennial grain crop Kernza™, which might not be representative of the much broader suite of

CLC crops, cropping systems, and stacking of these strategies. However, although Cureton et al. (in review) both draw upon the sustainability transition intermediary literature and self-identify as intermediaries in their practical theory of CLC crop commercialization, they cite the lack of conceptual and practical intermediary guidance as the impetus for their contribution to the literature. This observation is perhaps an inadvertent criticism of the current scope of the sustainability transition intermediary literature, which prioritizes systemic, retrospective analyses at the expense of actor-level perspectives on a transition in the making (Murto et al., 2020).

Similarly, Zolfagharian et al. (2019) critique the transitions literature for its lack of paradigmatic and methodological diversity, while Steyaert et al. (2016) call for increased attention to the assumed relationship between knowledge and action in the study of intermediaries. Conclusions in the transition intermediary literature are often directed toward policy-makers and researchers (e.g., Mignon and Kanda, 2018; Kant and Kanda, 2019; Kivimaa et al., 2019a) suggesting a paradigm that assumes policy-makers and researchers both can and will design effective intermediary bodies and their broader ecologies. This approach to knowledge production is somewhat incompatible with research findings, which indicate that intermediaries often arise naturally in response to gaps in coordination and knowledge (Moss, 2009; Kivimaa et al., 2019a; Kanda et al., 2020). These critiques, paired with Cureton et al.'s assertion that the literature is lacking in practical intermediary guidance, suggest that the transition intermediary literature should give consideration to intermediaries as action-research practitioners. Such a paradigmatic reframe would call for more diverse research artifacts that are attuned to application in sustainability transition processes, rather than merely describing transitions *post-hoc*. A practitioner-researcher positioning might be unique to intermediaries in U.S. agri-food contexts, where there is a history of Cooperative Extension providing some intermediary functions as an integral part of the Land Grant University system (Peters, 2014). This remains merely speculative but worthy of future inquiry. Summarily, the nascent research in CLC transition intermediaries indicates the need to bring more concerted attention to the practice and politics of transition intermediaries in future research, which finds great familiarity with agroecology's framework of science, practice, and politics.

4. An agroecological framework for intermediaries in CLC transitions

An understanding of agroecology as a triad of sciences, practices, and politics that align to achieve pragmatic goals (Bell and Bellon, 2021) can help advance the transition intermediary research in an agri-food context. Of course, the boundaries between these three categories are not strict, as science also has political and practical elements, just as practitioners are informed by science and politics. However, agroecology's triad remains a useful heuristic and, in this framing, the study of transition intermediaries can be understood as one of the many sciences that supports agroecological transformation. The framing also highlights the lack of attention to the politics and pragmatic practice of agroecological transition intermediaries within the current literature. While there

is a robust field of research devoted to agroecological transitions that remains outside the scope of this mini-review (e.g., Duru et al., 2015; Montenegro de Wit and Iles, 2016; Ollivier et al., 2018; Anderson et al., 2019; El Bilali, 2020), there is relatively less research that concertedly investigates intermediaries in such transitions. The emerging research that does investigate both agroecological transitions and intermediaries (e.g., Contesse et al., 2021; Iyabano et al., 2021; Groot-Kormelinck et al., 2022; Vilas-Boas et al., 2022) brings novel perspectives to the importance of non-human agency and farmer organizations in intermediary transition processes. However, there still remains a lack of attention to intermediaries as practitioners with political agency that could actively integrate frameworks to help guide their work. Such frameworks could address calls for more process-oriented approaches to sustainability transitions that acknowledge limits to scientific knowledge in complex problem solving (Bulten et al., 2021).

In that vein, this mini-review proposes a framework based on an adaptation of Anderson et al.'s (2019) notion of six "domains of transformation" in agroecology to understand and evaluate the role of intermediaries in CLC transitions (Table 1). Anderson proposes six primary, overlapping interfaces between the predominant agri-food regime and agroecological niches: knowledge and culture; systems of exchange; networks; discourse; equity; and access to farmland, plant material, and natural resources. Four of these categories (systems of exchange, networks, knowledge and culture, and discourse) fall directly within the purview of CLC transition intermediaries, who are tasked with building sustainable supply chains, forming networks, translating diverse knowledges, and continually framing CLC through their interactions with various stakeholders, from growers to policy-makers. Issues related to equity and access to natural ecosystems have important intersections with their work but do not currently define CLC intermediaries' role. The proposed framework augments Anderson et al.'s understanding of enabling and disabling conditions for transformation to reflect the three core functions of intermediaries based on Kivimaa et al.'s definition (Kivimaa et al., 2019a): (1) linking actors, activities, skills, and resources; (2) connecting transition visions with existing regimes to create momentum for socio-technical system change; and (3) disrupting dominant socio-technical configurations. The framework appreciatively builds off of the growing understanding of intermediaries in sustainability transitions, while explicitly adding a political dimension based on an agroecological understanding of what constitutes transformative change.

The resulting framework can serve as a starting point for CLC intermediaries attempting to make sense of their work, as well as a reflective, evaluative tool for ongoing transition efforts. The six core domains, adapted from Anderson et al. (2019), include:

- Construction, production, sharing, and mobilization of CLC knowledge.
- Profitable, fulfilling, accessible, and fair supply chains for CLC producers.
- Multi-stakeholder CLC networks that enable inclusive development of knowledge, markets, and discourse.
- CLC discourse, or the way in which language is used to frame CLC debates, policy, and action.

TABLE 1 Domains of transformation and associated practices for CLC transition intermediaries.

Domain	Definition	Enabling transition		Disabling transitions
		Practices linking actors, activities, skills, and resources	Practices creating momentum for socio-technical system change	Dominant socio-technical configurations to disrupt
Knowledge and culture	Construction, production, sharing, and mobilization of CLC knowledge	<ul style="list-style-type: none"> Promote horizontal processes of CLC food producer learning Invite diverse participation in CLC research processes 	<ul style="list-style-type: none"> Respect and employ knowledge from diverse stakeholders Solicit needs and aspirations of local food producers to inform CLC transition 	Disrupt: <ul style="list-style-type: none"> Promotion of centralized, researcher-produced knowledge Prioritization of knowledge from profit-led research agendas
Systems of exchange	Profitable, fulfilling, accessible, and fair supply chains for CLC producers	<ul style="list-style-type: none"> Embed CLC markets in local territories that allow for self-determination 	<ul style="list-style-type: none"> Construct CLC markets that value the ecological, social, economic, cultural, and political outputs of CLC agriculture Base CLC markets in democratic social relations 	Disrupt: <ul style="list-style-type: none"> Concentration of agricultural input markets Singular focus on economies of scale
Networks	Multi-stakeholder CLC networks that enable inclusive development of knowledge, markets, and discourse	<ul style="list-style-type: none"> Weave together networks driven by civil society actors 	<ul style="list-style-type: none"> Develop high-functioning polycentric, decentralized, governance models Develop policies that reach across constituencies to address agriculture, health, environment, and rural livelihoods 	Disrupt: <ul style="list-style-type: none"> Dominant regime that undermines local organization Research networks disconnected from food producers
Discourse	The way in which language is used to frame CLC debates, policy, and action	<ul style="list-style-type: none"> Employ discourse that promotes participation of local communities in shaping transitions 	<ul style="list-style-type: none"> Frame CLC agriculture holistically to include environmental, economic, and social goods 	Disrupt: <ul style="list-style-type: none"> Agricultural discourse with a singular focus on productivity
Equity	Dismantling dynamics of marginalization and inequality in multiple CLC-related arenas	<ul style="list-style-type: none"> Promote BIPOC and diverse gender participation in CLC decision making Promote participation by those historically excluded from U.S. agriculture 	<ul style="list-style-type: none"> Emphasize people-centered development of CLC systems 	Disrupt: <ul style="list-style-type: none"> Crop/ cropping system development models blind to existing inequalities Persistent inequity
Access to farmland, plant material, and natural resources	Food producer's access to CLC plant material, the ways in which CLC enables farmers to steward land, and how CLC actors align themselves with other land access initiatives	<ul style="list-style-type: none"> Enable food producer access to CLC crops and plants 	<ul style="list-style-type: none"> Promote synergies between CLC crop production and ecosystem services 	Disrupt: <ul style="list-style-type: none"> Unequal land access Farm consolidation Excessive private control of seeds and other aspects of biodiversity

Adapted from Anderson et al. (2019), integrating Kivimaa et al.'s (2019a) understanding of key intermediary functions.

- Dismantling dynamics of marginalization and inequality in multiple CLC-related arenas.
- Food producer's access to CLC plant material, the ways in which CLC enables farmers to steward land, and how CLC actors align themselves with other land access initiatives.

Critical reflection amongst these six domains and, specifically, the degree to which these six domains can be integrated for a given CLC crop or cropping system transition process can inform an understanding of how transformative or reinforcing a given CLC crop or system is to the predominant agronomic regime. The six domains and their associated intermediary practices remain suggestive and far from exhaustive. This framework welcomes modifications, additions, and

future iterations informed by the ongoing practice of CLC transition intermediaries.

5. Conclusion

CLC agriculture presents an exciting opportunity to iteratively draw upon and advance the burgeoning transition intermediary literature in an agroecological context. Agroecology's triad of science, practice, and movement provides a useful heuristic to expand current research approaches in the transition intermediary literature, while its explicitly political orientation can provide a framework to assess agri-food systems undergoing concerted transition efforts. More generally, such a framework can inform

the study of transition intermediaries in other contexts, spurring increased attention to the politics of transition intermediaries.

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Conflict of interest

The author declares that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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References

- Anderson, C. R., Bruil, J., Chappell, M. J., Kiss, C., and Pimbert, M. P. (2019). From transition to domains of transformation: getting to sustainable and just food systems through agroecology. *Sustainability* 11, 5272. doi: 10.3390/su11195272
- Avelino, F., and Rotmans, J. (2009). Power in transition: an interdisciplinary framework to study power in relation to structural change. *Eur. J. Soc. Theory* 12, 543–569. doi: 10.1177/1368431009349830
- Bell, M., and Bellon, S. (2021). "The rhetorics of agroecology: positions, trajectories, strategies," in *Agroecological Transitions, Between Determinist and Open-Ended Visions*, eds C. Lamine, D. Magda, M. Rivera-Ferre and T. Marsden (Hague: Peter Lang AG), 289–309.
- Bergek, A. (2020). Diffusion intermediaries: a taxonomy based on renewable electricity technology in Sweden. *Environ. Innov. Soc. Transit.* 36, 378–392. doi: 10.1016/j.eist.2019.11.004
- Black, S. T. (2022). Abolitionist food justice: theories of change rooted in place-and life-making. *Food Foodways* 30, 123–141. doi: 10.1080/07409710.2022.2030942
- Boyd, J. H., and Prescott, E. C. (1986). Financial intermediary-coalitions. *J. Econ. Theory* 38, 211–232. doi: 10.1016/0022-0531(86)90115-8
- Bulten, E., Hessels, L. K., Hordijk, M., and Segrave, A. J. (2021). Conflicting roles of researchers in sustainability transitions: balancing action and reflection. *Sustain. Sci.* 16, 1269–1283. doi: 10.1007/s11625-021-00938-7
- Citizen Potawatomi Nation. (2020). *Regenerative Agricultural Techniques Improve Tribal Land for Generations to Come* (February 13, 2020). Available online at: <https://www.potawatomi.org/blog/2020/02/13/regenerative-agriculture-techniques-improve-tribal-land-for-generations-to-come/> (accessed December 14, 2022).
- Contesse, M., Duncan, J., Legun, K., and Klerkx, L. (2021). Unravelling non-human agency in sustainability transitions. *Technol. Forecast. Soc. Change* 166, 120634. doi: 10.1016/j.techfore.2021.120634
- Crews, T. E., Carton, W., and Olsson, L. (2018). Is the future of agriculture perennial? Imperatives and opportunities to reinvent agriculture by shifting from annual monocultures to perennial polycultures. *Glob. Sustain.* 1, e11. doi: 10.1017/sus.2018.11
- Cureton, C., Peters, T., Skelly, S., Carlson, C., Conway, T., Tautges, N., et al. (in review). *Toward a Practical Theory for Commercializing Novel Continuous Living Cover Crops: A Conceptual Review Through the Lens of Kernza Perennial Grain, 2019–2022*.
- De Fazio, M. (2016). Agriculture and sustainability of the welfare: the role of the short supply chain. *Agric. Agric. Sci. Proc.* 8, 461–466. doi: 10.1016/j.aaspro.2016.02.044
- Duru, M., Therond, O., and Fares, M. (2015). Designing agroecological transitions; a review. *Agron. Sustain. Dev.* 35, 1237–1257. doi: 10.1007/s13593-015-0318-x
- Edwards, L. (2012). Exploring the role of public relations as a cultural intermediary occupation. *Cult. Sociol.* 6, 438–454. doi: 10.1177/1749975512445428
- El Bilali, H. (2020). Transition heuristic frameworks in research on agro-food sustainability transitions. *Environ. Dev. Sustain.* 22, 1693–1728. doi: 10.1007/s10668-018-0290-0
- Forever Green Initiative. (2020). *Forever Green Introduction Packet*. University of Minnesota. Available online at: https://forevergreen-umn.info/266_FG_Introduction_Packet_sm_0621.pdf (accessed December 10 2022).
- Forever Green Partnership. (2022). *About the Forever Green Partnership*. University of Minnesota. Available online at: <https://forevergreenpartnership.umn.edu/home> (accessed June 27, 2022).
- Geels, F. W. (2011). The multi-level perspective on sustainability transitions: responses to seven criticisms. *Environ. Innov. Soc. Trans.* 1, 24–40. doi: 10.1016/j.eist.2011.02.002
- Geels, F. W. (2019). Socio-technical transitions to sustainability: a review of criticisms and elaborations of the Multi-Level Perspective. *Curr. Opin. Environ. Sustain.* 39, 187–201. doi: 10.1016/j.cosust.2019.06.009
- Goodrich, K. A., Sjoström, K. D., Vaughan, C., Nichols, L., Bednarek, A., and Lemos, M. C. (2020). Who are boundary spanners and how can we support them in making knowledge more actionable in sustainability fields? *Curr. Opin. Environ. Sustain.* 42, 45–51. doi: 10.1016/j.cosust.2020.01.001
- Green Lands Blue Waters. (2021). *Our Journey to a Transformed Agriculture through Continuous Living Cover*. GLBW. Available online at: <https://greenlandsbluwaters.org/wp-content/uploads/2021/08/OurJourneyToTransformedAgThruCLC-GLBW2021.pdf> (accessed June 25, 2022).
- Groot-Kormelinck, A., Bijman, J., Trienekens, J., and Klerkx, L. (2022). Producer organizations as transition intermediaries? Insights from organic and conventional vegetable systems in Uruguay. *Agric. Hum. Values* 39, 1277–1300. doi: 10.1007/s10460-022-10316-3
- Howells, J. (2006). Intermediation and the role of intermediaries in innovation. *Res. Policy* 35, 715–728. doi: 10.1016/j.respol.2006.03.005
- Huria, S., and Pathania, K. (2018). Dynamics of food inflation: assessing the role of intermediaries. *Glob. Bus. Rev.* 19, 1363–1378. doi: 10.1177/0972150918788763
- Iyabano, A., Klerkx, L., Faure, G., and Toillier, A. (2021). Farmers' Organizations as innovation intermediaries for agroecological innovations in Burkina Faso. *Int. J. Agric. Sustain.* 20, 857–873. doi: 10.1080/14735903.2021.2002089
- Janda, K. B., and Parag, Y. (2013). A middle-out approach for improving energy performance in buildings. *Build. Res. Inform.* 41, 39–50. doi: 10.1080/09613218.2013.743396
- Jewett, J. G., and Schroeder, S. (2015). *Continuous Living Cover Manual*. Green Lands Blue Waters. Available online at: https://greenlandsbluwaters.org/wp-content/uploads/2019/08/CLC_Manual_FULL-1.pdf (accessed November 14, 2022).

- Jordan, N., and Warner, K. D. (2010). Enhancing the multifunctionality of US agriculture. *Bioscience* 60, 60–66. doi: 10.1525/bio.2010.60.1.10
- Jordan, N. R., Dorn, K., Runkel, B., Ewing, P., Williams, A., Anderson, K. A., et al. (2016). Sustainable commercialization of new crops for the agricultural bioeconomy. Sustainable commercialization of new bioeconomy crops. *Elem. Sci. Anthropol.* 4, 000081. doi: 10.12952/journal.elementa.000081
- Jordan, N. R., Wilson, D. S., Noble, K., Miller, K., Conway, T., and Cureton, C. (in review). A Polycentric Network Strategy for Regional Agricultural Diversification of Agriculture: Theory and Implementation.
- Kanda, W., Kuisma, M., Kivimaa, P., and Hjelm, O. (2020). Conceptualising the systemic activities of intermediaries in sustainability transitions. *Environ. Innov. Soc. Transitions* 36, 449–465. doi: 10.1016/j.eist.2020.01.002
- Kant, M., and Kanda, W. (2019). Innovation intermediaries: what does it take to survive over time? *J. Clean. Prod.* 229, 911–930. doi: 10.1016/j.jclepro.2019.04.213
- Kivimaa, P. (2014). Government-affiliated intermediary organisations as actors in system-level transitions. *Res. Policy* 43, 1370–1380. doi: 10.1016/j.respol.2014.02.007
- Kivimaa, P., Bergek, A., Matschoss, K., and van Lente, H. (2020a). Intermediaries in accelerating transitions: introduction to the special issue. *Environ. Innov. Soc. Transitions* 36, 372–377. doi: 10.1016/j.eist.2020.03.004
- Kivimaa, P., Boon, W., Hyysalo, S., and Klerkx, L. (2019a). Towards a typology of intermediaries in sustainability transitions: a systematic review and a research agenda. *Res. Policy* 48, 1062–1075. doi: 10.1016/j.respol.2018.10.006
- Kivimaa, P., Hyysalo, S., Boon, W., Klerkx, L., Martiskainen, M., and Schot, J. (2019b). Passing the baton: how intermediaries advance sustainability transitions in different phases. *Environ. Innov. Soc. Transitions* 31, 110–125. doi: 10.1016/j.eist.2019.01.001
- Kivimaa, P., Primmer, E., and Lukkarinen, J. (2020b). Intermediating policy for transitions towards net-zero energy buildings. *Environ. Innov. Soc. Transitions* 36, 418–432. doi: 10.1016/j.eist.2020.01.007
- Klerkx, L., and Leeuwis, C. (2009). Establishment and embedding of innovation brokers at different innovation system levels: insights from the Dutch agricultural sector. *Technol. Forecast. Soc. Change* 76, 849–860. doi: 10.1016/j.techfore.2008.10.001
- Köhler, J., Geels, F. W., Kern, F., Markard, J., Onsongo, E., Wiecek, A., et al. (2019). An agenda for sustainability transitions research: state of the art and future directions. *Environ. Innov. Soc. Transitions* 31, 1–32. doi: 10.1016/j.eist.2019.01.004
- Lakhani, N., Uteuova, A., and Chang, A. (2021). Revealed: the true extent of America's food monopolies, and who pays the price. *The Guardian*. Available online at: <https://www.theguardian.com/environment/ng-interactive/2021/jul/14/food-monopoly-meals-profits-data-investigation> (accessed April 20, 2022).
- Lam, D. P. M., Martín-López, B., Wiek, A., Bennett, E. M., Frantzeskaki, N., Horcea-Milcu, A. I., et al. (2020). Scaling the impact of sustainability initiatives: a typology of amplification processes. *Urban Transform.* 2, 1–24. doi: 10.1186/s42854-020-00007-9
- Lawhon, M., and Murphy, J. T. (2012). Socio-technical regimes and sustainability transitions: Insights from political ecology. *Prog. Hum. Geogr.* 36, 354–378. doi: 10.1177/0309132511427960
- Magda, D., Lamine, C., Marsden, T., and Rivera-Ferre, M. (2021). “Taking into account the ontological relationship to change in agroecological transitions,” in *Agroecological Transitions between Determinist and Open-ended Perspectives*, eds C. Lamine, D. Magda, M. Rivera-Ferre and T. Marsden (Hague: Peter Lang AG), 33–57.
- Méndez, V. E., Bacon, C. M., and Cohen, R. (2013). Agroecology as a transdisciplinary, participatory, and action-oriented approach. *Agroecol. Sustain. Food Syst.* 37, 3–18. doi: 10.1080/10440046.2012.736926
- Mignon, I. (2017). Intermediary–user collaboration during the innovation implementation process. *Technol. Anal. Strat. Manage.* 29, 735–749. doi: 10.1080/09537325.2016.1231299
- Mignon, I., and Kanda, W. (2018). A typology of intermediary organizations and their impact on sustainability transition policies. *Environ. Innov. Soc. Transitions* 29, 100–113. doi: 10.1016/j.eist.2018.07.001
- Montenegro de Wit, M., and Iles, A. (2016). Toward thick legitimacy: Creating a web of legitimacy for agroecology. *Agroecological Legitimacy. Elem. Sci. Anthropol.* 4, 000115. doi: 10.12952/journal.elementa.000115
- Moss, T. (2009). Intermediaries and the governance of sociotechnical networks in transition. *Environ. Plan. A* 41, 1480–1495. doi: 10.1068/a4116
- Muckey, E. (2019). *Kernza® in Southern Minnesota: Assessing Local Viability of Intermediate Wheatgrass*. University of Minnesota. Available online at: <https://hdl.handle.net/11299/202253> (accessed June 5, 2022).
- Murto, P., Hyysalo, S., Juntunen, J. K., and Jalas, M. (2020). Capturing the micro-level of intermediation in transitions: comparing ethnographic and interview methods. *Environ. Innov. Soc. Transitions* 36, 406–417. doi: 10.1016/j.eist.2020.01.004
- Nicklay, J. A., Perrone, S. V., and Wauters, V. M. (2023). Becoming agroecologists: A pedagogical model to support graduate student learning and practice. *Front. Sustain. Food Syst.* 7, 16. doi: 10.3389/fsufs.2023.770862
- Ollivier, G., Magda, D., Mazé, A., Plumecocq, G., and Lamine, C. (2018). Agroecological transitions: what can sustainability transition frameworks teach us? An ontological and empirical analysis. *Ecol. Soc.* 23, 18. doi: 10.5751/ES-09952-230205
- Parag, Y., and Janda, K. B. (2014). More than filler: middle actors and socio-technical change in the energy system from the “middle-out.” *Energy Res. Soc. Sci.* 3, 102–112. doi: 10.1016/j.erss.2014.07.011
- Peters, A., Barrett, E., and Stinogel, J. (2021). *Technical Assistance for Continuous Living Cover Agricultural Practices [Capstone Project]*. Humphrey School of Public Affairs in partnership with Green Lands Blue Waters. University of Minnesota. Available online at: <https://hdl.handle.net/11299/225836> (accessed June 20, 2022).
- Peters, S. J. (2014). Extension reconsidered. *Choices* 29, 1–6.
- Peterson, H. (2009). Transformational supply chains and the ‘wicked problem’ of sustainability: aligning knowledge, innovation, entrepreneurship, and leadership. *J. Chain Netw. Sci.* 9, 71–82. doi: 10.3920/JCNS2009.x178
- Pielke, R. A. Jr. (2007). *The Honest Broker: Making Sense of Science in Policy and Politics*. Cambridge: Cambridge University Press. doi: 10.1017/CBO9780511818110
- Polzin, F., von Flotow, P., and Klerkx, L. (2016). Addressing barriers to eco-innovation: Exploring the finance mobilisation functions of institutional innovation intermediaries. *Technol. Forecast. Soc. Change* 103, 34–46. doi: 10.1016/j.techfore.2015.10.001
- Ray, S. S. (2020). *Winter Camelina Supply Chain Development and Regional Opportunities for Oilseed Processing*. SARE. Available online at: <https://projects.sare.org/wp-content/uploads/Camelina-Supply-Chain-Development-and-Opportunities-for-Processing.pdf> (accessed June 11, 2022).
- Rittel, H. W., and Webber, M. M. (1973). Dilemmas in a general theory of planning. *Policy Sci.* 4, 155–169. doi: 10.1007/BF01405730
- Runkel, B., Kantar, M., Eckberg, J., Barnes, R., Betts, K., Lehman, C., et al. (2013). “Development of continuous living cover breeding programmes,” in *FAO Expert Workshop on Perennial Crops for Food Security Rome* (Italy).
- Slocum, R., and Cadieux, K. V. (2015). Notes on the practice of food justice in the US: understanding and confronting trauma and inequity. *J. Polit. Ecol.* 22, 27. doi: 10.2458/v22i1.21077
- Sovacool, B. K., Turnheim, B., Hook, A., Brock, A., and Martiskainen, M. (2021). Dispossessed by decarbonisation: reducing vulnerability, injustice, and inequality in the lived experience of low-carbon pathways. *World Dev.* 137, 105116. doi: 10.1016/j.worlddev.2020.105116
- Sovacool, B. K., Turnheim, B., Martiskainen, M., Brown, D., and Kivimaa, P. (2020). Guides or gatekeepers? Incumbent-oriented transition intermediaries in a low-carbon era. *Energy Res. Soc. Sci.* 66, 101490. doi: 10.1016/j.erss.2020.101490
- Steyaert, P., Barbier, M., Cerf, M., Levain, A., and Loconto, A. M. (2016). *Role of Intermediation in the Management of Complex Sociotechnical Transitions*. Available online at: <https://hal.archives-ouvertes.fr/hal-01470892>
- Streit Krug, A., and Tesdell, O. I. (2020). A social perennial vision: transdisciplinary inquiry for the future of diverse, perennial grain agriculture. *Plants People Planet* 3, 355–362. doi: 10.1002/ppp3.10175
- Sylvain, O. (2018). Intermediary design duties. *Conn. L. Rev.* 50, 203.
- van Boxtael, A., Meijer, L. L. J., Huijben, J. C. C. M., and Romme, A. G. L. (2020). Intermediating the energy transition across spatial boundaries: cases of Sweden and Spain. *Environ. Innov. Soc. Transitions* 36, 466–484. doi: 10.1016/j.eist.2020.02.007
- van Lente, H., Hekkert, M., Smits, R., and van Waveren, B. (2011). “Systemic intermediaries and transition processes,” in *Shaping Urban Infrastructures: Intermediaries and the Governance of Socio-Technical Networks*, eds S. Guy, S. Marvin, W. Medd, and T. Moss (London: Earthscan), 36–52.
- Van Tassel, D. L., Tesdell, O., Schlaughtman, B., Rubin, M. J., DeHaan, L. R., Crews, T. E., et al. (2020). New food crop domestication in the age of gene editing: genetic, agronomic and cultural change remain co-evolutionarily entangled. *Front. Plant Sci.* 11:789. doi: 10.3389/fpls.2020.00789
- Vihemäki, H., Toppinen, A., and Toivonen, R. (2020). Intermediaries to accelerate the diffusion of wooden multi-storey construction in Finland. *Environ. Innov. Soc. Transitions* 36, 433–448. doi: 10.1016/j.eist.2020.04.002
- Vilas-Boas, J., Klerkx, L., and Lie, R. (2022). Facilitating international animal welfare standards implementation in national contexts: the role of intermediaries in Brazilian pig production. *J. Rural Stud.* 90, 53–64. doi: 10.1016/j.jrurstud.2022.01.012
- Warbroek, B., Hoppe, T., Coenen, F., and Bressers, H. (2018). The role of intermediaries in supporting local low-carbon energy initiatives. *Sustainability* 10, 1–28. doi: 10.3390/su10072450
- Wezel, A., Bellon, S., Doré, T., Francis, C., Vallod, D., and David, C. (2009). Agroecology as a science, a movement and a practice. *Sustain. Agric.* 2, 27–43. doi: 10.1007/978-94-007-0394-0_3
- Wigboldus, S., Klerkx, L., Leeuwis, C., Schut, M., Muilerman, S., and Jochemsen, H. (2016). Systemic perspectives on scaling agricultural innovations. A review. *Agron. Sustain. Dev.* 36, 1–20. doi: 10.1007/s13593-016-0380-z
- Zolfagharian, M., Walrave, B., Raven, R., and Romme, A. G. L. (2019). Studying transitions: past, present, and future. *Res. Policy* 48, 103788. doi: 10.1016/j.respol.2019.04.012



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EDITED BY

Carol Williams,
University of Wisconsin-Madison, United States

REVIEWED BY

Rishikesh Singh,
Panjab University, India
Anne Elise Stratton,
University of Hohenheim, Germany

*CORRESPONDENCE

Nicholas R. Jordan
✉ jorda020@umn.edu

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A polycentric network strategy for regional diversification of agriculture: theory and implementation

Nicholas R. Jordan¹, David Sloan Wilson², Kate Noble³,
Keith Miller³, Tara Maireid Conway¹ and Colin Cureton¹

¹Agronomy and Plant Genetics Department, University of Minnesota, St. Paul, MN, United States,

²Binghamton University of New York, Binghamton, NY, United States, ³Terraluna Collaborative,
Minneapolis, MN, United States

Widespread and substantial diversification of current agroecosystems appears fundamental to meeting many grand challenges in agriculture. Despite urgent calls for diversification on regional scales, particularly in regions dominated by industrialized, low-diversity agriculture, strategies for diversification on such scales are in early stages of development, conceptually and practically. We outline such a strategy, and its implementation by the Forever Green Partnership, a public-private-NGO coalition in support of agricultural diversification in the U.S. Midwest region. Our strategy supports introduction and scaling of multiple novel crops in a region, which requires development of many interdependent supporting elements, including supportive markets, infrastructure, policy, finance, and R&D. The core of our strategy is development of *sustainable supply chains* (SSCs) for a set of novel crops. We define SSCs as rudimentary systems of these supporting elements for novel crops, linking on-farm crop production to end-use markets while advancing economic, environmental, and social sustainability criteria that are demanded by stakeholders. SSCs provide a scaffold upon which fully-developed support systems for multiple novel crops can be constructed, thus driving regional diversification. SSCs cannot be “built in a day”; rather they must evolve as production of novel crops expands over time and space, and as new challenges and opportunities emerge. Therefore, regional diversification requires a system to sustain this evolutionary process across time and multiple novel crops. We posit that an effective system can be built from two crucial elements: a process of conscious and concerted cultural evolution, and a polycentric network that organizes and supports that process. We outline this system and its conceptual basis, and its implementation by the Forever Green Partnership, and associated challenges and accomplishments. Three years after its inception, the Partnership has attracted substantial resources, developed a polycentric network, and some elements of the cultural-evolution process are in place. However, node development is uneven across the network, hindering its operation. In addition to advancing strategies for regional-scale diversification, the Partnership is seeking to advance conceptual and practical understanding of sustainability transitions in agriculture, and to explore the potential value of conscious cultural evolution in such transitions.

KEYWORDS

agroecology, polycentric governance, social networks, cultural evolution, sustainability transition

Introduction

Major transitions are needed in agriculture to address its grand challenges, including climate change adaptation and mitigation, restoration of soil, water, and biodiversity, enhancement of health through diet, and achieving equity and justice in agriculture, food, and bioproduct systems (Willett et al., 2019; Klerkx and Begemann, 2020; Rockström et al., 2020; Steiner et al., 2020). Diversification of current farm production systems appears fundamental to meeting these goals. Through a wide range of mechanisms, diversification can enable climate-change adaptation and mitigation, support dietary shifts, and improve the condition of soil, water, and biodiversity resources (Lin, 2011; Kremen and Miles, 2012; Bowles et al., 2020; Tamburini et al., 2020). Diversification also creates opportunities to enhance equity and other social dimensions of sustainability, if specific efforts to address social sustainability challenges are encompassed in diversification initiatives.

Herein, we write to advance strategic frameworks for diversifying agriculture at regional scales. The authors are affiliates of the *Forever Green Partnership*, (2023), a coalition of environmental, agricultural, research, and private-sector organizations working to advance agricultural diversification in the U.S. Midwest region. To guide the work of the Partnership, we have synthesized a regional-diversification strategy from multiple sources, both conceptual and practical, and describe ongoing implementation and assessment of the strategy. Development of such frameworks appears to be in early days, despite growing awareness of the value of diversified regional food systems (Blay-Palmer et al., 2018; Clancy and Ruhf, 2018; Nicol, 2020), and calls for diversification on regional scales (Prokopy et al., 2020). Specifically, we draw on frameworks from the emerging fields of sustainability transitions (Geels, 2019; Schlaili and Urmetzer, 2019; Wyborn et al., 2019), systemic approaches to innovation (Hermans et al., 2019) and the “science of scaling” of agricultural innovations (Barrett et al., 2020; Schut et al., 2020; Wigboldus et al., 2020). We integrate these by applying the emerging theory of conscious and concerted cultural evolution (Cox and Schoon, 2019; Wilson, 2019).

We address diversification at a regional level via introduction and scaling of additional crops in a region; these may be entirely novel crops, or new to the region. There are many barriers to such diversification (Lockeretz, 1988; Meynard et al., 2017, 2018; Jouan et al., 2019; Stefani et al., 2020; Mortensen and Smith, 2020). The fundamental conundrum is that, absent markets, farmers will not grow such novel crops, while without supply from farmers, market demand is unlikely to develop. Beyond markets, novel crops also lack most other pillars of support needed by any established crop: technologies and ecosystems of production (comprising crops, land and soil, and associated biodiversity); post-production infrastructure, and end-use product production; human “capital,” including interest and know-how; social and institutional capital (e.g., advocacy groups for the crop); and financial, political, legal, regulatory, and cultural support (Lockeretz, 1988; Montenegro de Wit and Iles, 2016; Blesh et al., 2023). The absence of such supporting elements creates strong ‘lock-in’ path dependence that sustains established crops (Meynard et al., 2018; Mortensen and Smith, 2020). To introduce and support a novel crop in a region, it is necessary to organize a new *socio-ecological-technical* system for the crop, comprising the above supporting elements.

Socio-ecological-technical systems are integrated sets of biophysical, technical and social elements that function together to

meet a societal need (Duru et al., 2015; Markolf et al., 2018; Ahlborg et al., 2019). Construction and scaling of socio-ecological-technical systems for diversification crops is a dynamic, contingent, and inherently risky undertaking, as many different elements must develop and cohere, in an integrated process of innovation and scaling (Jordan et al., 2016; Meynard et al., 2017; Blesh et al., 2023). Importantly, development of certain “pillars” (e.g., novel land valuation and financing mechanisms, Johnson, 2020), will be relevant to multiple novel crops for a region, creating interdependencies in socio-ecological-technical systems development among multiple crops. Therefore, the process of regional diversification can be framed as *establishment of a mutually supportive set of socio-ecological-technical systems for a set of novel crops*.

Accordingly, our strategy for regional diversification centers on interdependent construction of such supportive systems for each of a set of crops. The core of the strategy is a process of conscious and intentional cultural evolution (Cox and Schoon, 2019; Wilson, 2019), undertaken by a collective of actors relevant to construction of these supportive systems. Recently, this evolutionary approach to cultural change has emerged as a novel approach to sustainability transitions (Brooks et al., 2018; Jones et al., 2020). We apply this evolutionary perspective by viewing socio-ecological-technical systems as evolvable units of human culture that integrate beliefs, values, norms, knowledge, technologies, behaviors, and institutions (Montenegro de Wit and Iles, 2016; Barrett et al., 2020). Specifically, our strategy is designed to drive rapid regional diversification by efficiently evolving *sustainable supply chains* (SSCs) for novel crops. As we define them, SSCs are rudimentary socio-ecological-technical systems that link on-farm crop production to end-use markets, while advancing economic, environmental, and social sustainability criteria that are demanded by stakeholders. We propose that SSCs provide a scaffold upon which fully-developed socio-ecological-technical systems can be constructed and scaled, thus driving regional diversification. Below, we present the conceptual basis for this strategy, and then provide case study of ongoing implementation of the strategy by the Forever Green Partnership. We note that while our strategy is applicable to diversification by introduction of novel crops of any sort, the implementation case focuses on a set of perennial and winter-annual crop species being developed by the Partnership (2023).

Sustainable supply chains for novel crops

We define SSCs for novel crops as on-farm crop production and flows of agricultural commodities and ecosystem services that result from these farm activities, and associated institutions and infrastructure. Together, these elements constitute a rudimentary socio-ecological-technical system, consisting of three coupled and interactive subsystems (Duru et al., 2015).

A crop production subsystem comprising farmers and farms producing novel crops

During initial stages of SSC development for emerging crops, this subsystem should consist of spatially-aggregated clusters of farms producing these crops, as clusters provide mutual support and other advantages of aggregation (Manson et al., 2016). Such clustered production can be advantageously situated within areas on the scale

of a small watershed, as modestly-sized agricultural watersheds (*ca.* 10,000 ha) appear advantageous for coordinated implementation of agricultural diversification and conservation measures (Jordan et al., 2018; Ranjan et al., 2019).

A post-production commodity subsystem comprising post-production commodity supply-chain actors and associated infrastructure

This subsystem is an inter-organizational system that efficiently and effectively manages flows of material, information, and capital associated with the production of products, to meet economic interests of participating organizations while advancing environmental and social sustainability (Morais and Silvestre, 2018; Westermann et al., 2018). It links farm commodity production to end-use markets, and includes physical infrastructure (e.g., processing or storage facilities), and organizations and institutions involved in supply-chain operation or governance.

A socio-ecological subsystem comprising natural-resource management actors and natural resources affected by the supply chain

This subsystem comprises clusters of farms producing novel crops that produce some environmental benefit (e.g., improved condition of soil, water, and biodiversity resources), and one or more “customer(s)” for these benefits, e.g., a city affected by attributes of water in a watershed. The customer(s) will interact with farms to compensate them for these benefits, e.g., by monetary subsidies for new crop production. This subsystem also includes any non-local customers for environmental benefits (e.g., for soil carbon storage) and organizations and institutions involved in governance of relevant natural resources and systems for compensation (e.g., payment-for-ecosystem-service programs).

Our diversification strategy aims to drive regional diversification by multi-sector collective action to develop SSCs that advance economic, environmental, and social sustainability criteria that are demanded by stakeholders. As is broadly recognized (Hermans et al., 2019; Barrett et al., 2020), collective action across public, private, and NGO/philanthropy sectors is critical to sustainability transitions, such as regional diversification.

Building Sscs for regional diversification of agriculture: a system for interdependent development and scaling

Development and scaling of SSCs is a complex challenge

We presume that to attract and inspire broad collective action to advance regional diversification, SSC establishment and operation must provide multiple economic, environmental, and social benefits (Peterson, 2009; Boström et al., 2015). SSCs that produce this full range of sustainability benefits cannot be “built in a day.” There are many unknowns about SSC design and operation (Boström et al., 2015; Wigboldus et al., 2016), and SSCs must evolve as production of novel crops expands over time and space, adapting to new geographies, and

to new challenges and opportunities that emerge as scaling proceeds (Schut et al., 2020). Moreover, building fully supportive socio-ecological-technical systems for novel crops—including knowledge, economic, political, legal, and cultural domains—construction of fully-supportive systems is likely to be a prolonged process requiring a multiple evolutionary steps (Cooley and Papoulidis, 2017; Geels, 2019; Wilson, 2019; Barrett et al., 2020), via an iterative, learning-intensive process of prototyping, evaluation, and improvement (Seyfang et al., 2014; Gurzawska, 2019; Wilson, 2019; Barrett et al., 2020).

A development and scaling system for SSCs

To advance regional diversification by development and scaling of SSCs for multiple novel crops, effort must be sustained across crops, scales of implementation, and time. Drawing on a range of current models for scaling (Gurzawska, 2019; Tomich et al., 2019; Wilson, 2019; Woltering et al., 2019; Schut et al., 2020), we posit that a development and scaling system for SSCs can be built from two crucial elements. These are 1) active support of a process of intentional and conscious cultural evolution (Cox and Schoon, 2019; Wilson, 2019); and 2), a polycentric network (Carlisle and Gruby, 2019) that supports that process.

Developing SSCs through intentional and concerted cultural evolution

Recently, intentional facilitation of cultural evolution has emerged as a strategy for meeting complex sustainability challenges (Brooks et al., 2018; Wilson, 2019). The idea is to support cultural evolution by a selective process that supports desirable and replicable cultural innovations that meet sustainability challenges. For regional diversification of agriculture via novel crops, the relevant cultural innovation is in the structure and functioning of SSCs. Desirable SSC variants more efficiently and effectively advance sustainability goals of stakeholders. Such cultural evolution can be facilitated by creating variation through organized innovation and experimentation, imposing selection by “rewarding what works” through differential provision of resources, financial or otherwise (Cooley and Papoulidis, 2017; Sengers et al., 2019; Wilson, 2019; Barrett et al., 2020), and by supporting replication of favorable variants. We propose that, if undertaken collectively and in concert, and facilitated for efficiency and rapidity, these intentional processes of variation, selection, and replication will accelerate SSC development.

Facilitation of this evolutionary dynamic begins by supporting a cross-sector group in defining its goal: i.e., a paradigm of a fully-developed SSC for a novel crop, defined in terms of economic, environmental, and social aspects of sustainability. Once defined, prototypic supply chains can be evaluated against the goal, and supporting resources rewarded accordingly. As implemented in the Forever Green Partnership (described below), this group is a multi-sector collaborative, representing a range of societal sectors that have interests in diversification of a regional agriculture, and the ability to aggregate resources to support promising prototypic supply chains.

Variation is essential to evolution. Therefore, facilitation of SSC evolution should focus on generating variation relevant to the systemic SSC goal. This can be accomplished by organizing a system for creating and pilot-testing novel supply chains that address the systemic

SSC goal. Generally, such novel supply chains will integrate multiple innovations drawn from multiple domains, including the technical, social, organizational, and conceptual (Leeuwis and Aarts, 2011; Barrett et al., 2020). In the Forever Green Partnership, this integration is supported by an ongoing forum for persons professionally engaged in such integrative SSC innovation, as described below.

Finally, replication of selected variants is needed in any evolutionary process. Facilitation must ensure efficient replication of novel supply chains that advance toward the SSC goal. In practice, such replication can be accomplished by adding strong communication and “incubator” aspects to an integrative innovation forum, so that interested parties can develop new supply chain prototypes—e.g., for new crops or in new regions—built on successful novel SSCs.

If these elements of selection, variation, and replication can be established, closely coupled, and sustained over time, then an ongoing process of cultural evolution will drive SSCs toward the systemic goal. What is needed to establish and sustain these conditions, in practice? We propose that a polycentric network can well serve this purpose.

A polycentric network for efficient and forceful evolution of SSCs

Polycentric networks are emerging, in theory and practice, as a strategy for addressing complex sustainability challenges such as regional diversification of agriculture. The essential idea, quoting Ostrom (2010), is development of “*complex multi-level systems to cope with complex, multi-level problems*” (Ostrom, 2010; Dorsch and Flachsland, 2017; Carlisle and Gruby, 2019). Intentional concerted cultural evolution of SSCs is certainly such a problem, and therefore we posit that a multi-level polycentric network (Figure 1) can be designed to support the cultural evolution process outlined above. It is clear that cooperative cross-sector and cross-scale networks can advance innovation and sustainability transitions in agri-food systems (e.g., Blesh and Wolf, 2014; Bui et al., 2016; Home et al., 2017; Meynard et al., 2017). In particular, such networks can bring a range of complementary innovations together (e.g., novel diversified farming strategies and novel institutions) to advance agricultural socio-technical systems, typically at pilot scales, and to advance scaling of these systems (Bui et al., 2016; Home et al., 2017; Meynard et al., 2017). Most commonly, such networks have largely functioned as singular entities, focusing on development of place-based socio-ecological-technical systems (Melchior and Newig, 2021). In contrast, the polycentric network described below is conceptualized as a regional structure, engaging multiple networks operating at multiple scales, in order to support and systematize production, piloting, refinement, and possible scaling of multiple socio-ecological-technical systems in pursuit of agricultural diversification on regional scale. This project thus provides an additional case of deliberate experimentation with polycentric networks for sustainability transitions in agriculture (Marshall, 2009; Fasting et al., 2021; Heckelman et al., 2022). These reported cases, while different in many respects, aim to form systems of cooperation and mutual support among local-scaled sustainability networks and networks acting at broader scales. Therefore, polycentric networks can be seen as an effort to build on the successes of transition networks built around a single place-based project, by engaging multiple local-scaled networks in a polycentric “network of networks.” The goal is to provide particular benefits that emerge from effective polycentric structures, i.e., enhancing network-scale learning, innovation, and other collective

action, and supporting local self-determination in transition processes (Dorsch and Flachsland, 2017; Barrett et al., 2020). Similar work, if not explicitly framed as polycentric, is embodied in *La Via Campesina* (Rosset et al., 2019), and other extensive agroecology scaling networks (Mier y Terán Giménez Cacho et al., 2018).

For a multi-level polycentric network to support the cultural evolution process, it must provide a goal-setting and resource-provision group: i.e., a consortium of actors that can determine a shared goal for diversification of a region by novel crops, aggregate resources, and provide those resources to support emerging SSCs that best advance the goal. This consortium requires participation by actors that can command and aggregate resources, e.g., managers of corporations and firms, public institutions such as water infrastructure or economic development agencies, and NGOs, such as environmental NGOs. For example, private firms can actively cultivate markets for products of diversification crops that advance sustainability goals. Relevant resources include financial capital, and also include political capital, moral authority (“soft power”), and “integrative power” (ability to articulate compelling visions and bring actors together in collaborative efforts; Boström et al., 2015; Wigboldus et al., 2016; Geels, 2019). The principal incentive for participation is collective agency: the ability to achieve goals together by aggregating power across sectors, to better pursue their common interests in diversification.

At an intermediate level in the polycentric network, a system is needed that focuses on the variation and replication dimensions of managed cultural evolution. These functions can be provided by a consortium of actors—the integrative innovation forum described above—that can generate variation oriented to the SSC goal, assess performance of variant SSCs relative to the goal, promote replication of better-performing variants, and facilitate ongoing generation of new variation. This group should be drawn from actors that are actively involved in innovation, and in integration of innovations into novel co-innovation structures (Bui et al., 2016; Kivimaa et al., 2019), with an emphasis on enabling the “bundling” of complementary innovations in effective combinations (Barrett et al., 2020). Actors charged with innovation within dominant institutions in public, private, and NGO sectors are also key participants. We propose that such actors have collective ability to efficiently devise, test, and to provide nuanced evaluation of prototypic SSCs, as envisioned by Barrett et al. (2020). Moreover, by sharing their evaluations with the goal-setting and resource-provision group, they enable that group to carry out its key function of rewarding high-performing SSCs.

Finally, there is a third level in the polycentric network (Figure 1): innovation actors in a wide range of domains relevant to agricultural diversification. Emergence of key elements of SSCs frequently results from innovation at local scales that leverages creativity and local knowledge (e.g., building the base of supply chains by locally-tailored integration of novel crops into existing farming systems). These domains include development of new crops and new agricultural production systems, but also include supply-chain infrastructure, end-use innovation, and other economic, social, organizational, and policy innovation (Blesh and Wolf, 2014; Bui et al., 2016). In the context of agricultural diversification, such actors are increasingly organized in crop-specific networks that are focused on scaling of particular crops for diversification, via coupled and comprehensive innovation strategies (Meynard et al., 2017).

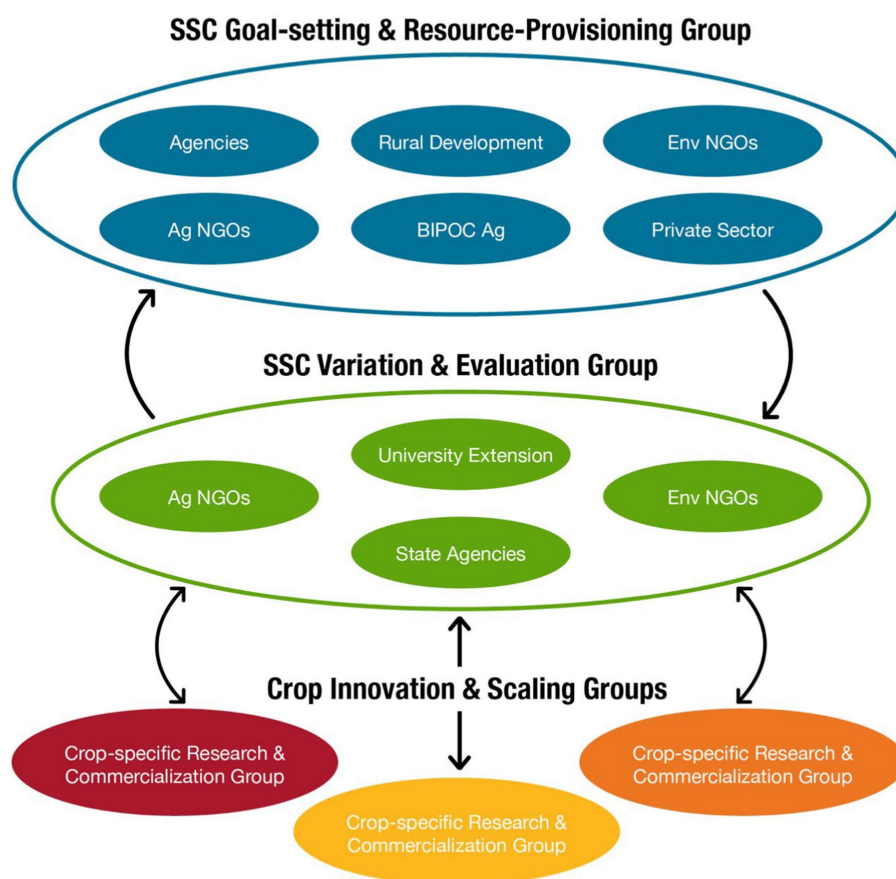


FIGURE 1

Conceptual model for polycentric network for evolution and scaling of SSCs. The network has three levels, as defined in text. Two levels are groups: a goal-setting and resource-provisioning group articulates goals for SSCs, and provides resources to support SSCs that advance its goals. This group is multi-sector, with relevant sectors indicated, as examples, including agriculturalists identifying with Black, Indigenous, and People of Color (BIPOC Ag), and environmental non-governmental organizations (Env NGOs), and agricultural non-governmental organizations (Ag NGOs). The SSC variation and evaluation group informs goal-setting/resource provisioning groups regarding “what works” in SSC development. This group unites a range of SSC innovators for exchange of SSC innovation and development approaches, and collective evaluation of these. This group is also multi-sector; again, relevant sectors are indicated as example. Crop-specific research & commercialization groups develop crops and implement SSCs for these crops. The SSC variation and evaluation group is informed by and provides feedback to crop-specific research & commercialization groups. Interactions occur between the goal-setting and resource-provisioning group and crop-specific research & commercialization groups but are less frequent and intense than the interactions described above.

An implementation case: the Forever Green Partnership

The Forever Green Partnership is an intentional experiment in applying the conceptual models outlined above in a project of regional diversification. This case study of the Partnership is intended to contribute to both the theory and practice of those engaged in the scaling of novel crops. Case studies allow researchers and practitioners to examine factors that influence a unit of analysis over time (Flyvbjerg, 2011). We use qualitative data (interviews and observations), aiming to support readers in forming naturalistic generalizations, i.e., transfers of knowledge that occur within the.

reader and their context (Stake and Trumbull, 1982). Such generalizations are based on context-specific settings and depend on the reader to apply the learnings, findings, and implications from the case study to their experiences (Stake, 2006). Specifically, we highlight the origins of the Partnership, its present structure

and functions, and comment on its progress to date. Our discussion of progress is informed by semi-structured interviews with members of the two major nodes of the Partnership network, which were conducted and analyzed by the Partnerships’ professional evaluators (co-authors Miller and Noble) during summer 2021 and summer 2022, *ca.* nine and 21 months, respectively after the key nodes of the Partnership had been established by an organizing group. Interviews explored understandings of the node that the interviewee was participating in, interactions with other nodes, and the nature and functioning of the Partnership as a whole. Interviews were conducted with 9 of 14 members (2021) and 9 of 18 members (2022) of the Strategic Steering Committee (see below), and 10 of 16 members of the Learning and Experimentation Network (see below). Interviews were recorded, transcribed, coded, and analyzed using qualitative methods. We have also drawn on observations of meetings of both nodes, which we attended as participant observers, recorded, and transcribed.

Context and diversification strategy

The Mid-Continent of North America is one of the most productive agricultural regions of the world, but cropping systems are dominated by short-lived summer annual crops. These systems leave soil exposed for much of the year, resulting in degradation of soil and biodiversity (Asbjornsen et al., 2014; Prokopy et al., 2020). These impacts threaten long-term food production in this global breadbasket, which may also be reduced by effects of climate change. Moreover, predominant cropping systems have major impacts on drinking water (Temkin et al., 2019), and diminish other ecosystem services related to water (Brauman, 2015), such as navigation and recreation. To protect the region from these mounting threats, and to sustain a significant element of the global food system, regional agricultural diversification is essential (Prokopy et al., 2020). The Forever Green Partnership has formed to pursue a particular diversification pathway: making farmland “forever green” with a set of crops that advance continuous living cover (CLC) agriculture in this region. CLC agriculture denotes agricultural systems in which there are living plants and roots in the ground throughout the entire year. Crops that advance CLC in this region include winter-hardy cover crops, which are generally defined as annual crops grown to enhance soil, water, and biodiversity without harvest of any agricultural commodity (e.g., seeds or biomass), other winter-hardy crops that produce such commodities, and perennial crops. Specifically, the Partnership is supporting development and commercialization of a portfolio of such crops for this cool-temperate region of North America, aiming to enhance a wide range of environmental and economic benefits to the region (Asbjornsen et al., 2014; Schulte et al., 2017). A leading developer of these crops is Forever Green Initiative (2023), a consortium of crop developers that is central to the Partnership. The Initiative is carrying out collaborative crop R&D efforts that span genomics, plant breeding, agronomy and agroecology, post-harvest handling and value-added processing.

Formation of the Forever Green Partnership

The Partnership was formally launched in 2018 by the co-directors of the Forever Green Initiative and several conservation groups (Friends of the Mississippi River and Minnesota Environmental Partnership) with financial support from the Minnesota Clean Water Council (a multi-sector governing body charged with distribution of public monies dedicated by statute to improving water resources in Minnesota). These conservation groups had grown increasingly interested in market-based diversification of agriculture as a pathway to meeting their water conservation goals. To pursue this vision, they proposed a coalition of environment, agriculture, research and business organizations in support of agricultural diversification via CLC agriculture. This coalition was also of interest to the Forever Green Initiative, as a complement to its crop R&D. After deliberation, a working charter for the Forever Green Partnership was established by late 2019. The charter established a “Strategic Steering Council” and “Learning and Experimentation Network” as two novel core elements of the Partnership, complementing the R&D capacities of the Forever Green Initiative. These two groups were organized in 2020, and began meeting monthly in the second half of that year. In 2019, the Forever Green Initiative received grant funding that supported

commercialization of the most advanced crops via development of markets, and supply chains to serve those markets. The current structure and activities of the Partnership (Figure 2) are described below, followed by a reflective account of the Partnership’s progress to date.

Strategic steering council

The Council is intended to function as the goal-setting and resource-provision group of the polycentric network described above—i.e., a consortium of actors that can set a goal for CLC agriculture, aggregate resources, and promote SSCs that best advance the goal, by differential allocation of these resources. At present, the group includes 17 active members (Table 1), drawn from state government, non-profit advocacy groups representing a range of interests including conservation, regional mainstream agriculture, rural community development, historically marginalized groups, the private sector, and the research and commercialization work of the Forever Green Initiative. The group aims to broaden the base of support for CLC agriculture across a wide range of societal sectors represented in the Council, so that these sectors can provide political, financial, and other forms of concrete support for advancing such agriculture. This support is intended to be provided selectively, providing support to SSCs and other CLC scaling efforts that accord with the Council’s shared vision for CLC agriculture. In interviews, members described themselves as wanting to be of use, experienced in thought and action leadership, and willing to offer their reputation, knowledge, capabilities, connections, and other resources to advancing CLC agriculture. Specifically, activities included discussion of goals and values (including social visions) for CLC agriculture, in pursuit of a shared vision for CLC agriculture in the region. The group has also held many learning sessions with innovators in relevant sectors (e.g., in rural development, and in new strategies for financing CLC agriculture) to develop shared understanding of these innovations and potential for engaging associated sectors in efforts to advance CLC agriculture. After these formative activities during the first year of operation, the Council turned its hand to definition and implementation of an agenda of “ambassadorship and advocacy” by which the multi-sector base of support for CLC agriculture could be broadened and deepened.

In interviews after the first year of operation, some Council members expressed appreciation for the Council as a forum for robust intersectoral exchange and cooperation around common interests in CLC agriculture. Illustrating this, one council member shared, “the original concept was that we would, through this interdisciplinary, iterative sort of workshopping model, we would bring all that expertise and come up with more of a synthetic pathway.” Another underscored the benefits of the diverse group, stating “there aren’t that many organizations that have that kind of potential reach across so many sectors. Summing up the unique potential of this group, one member shared, “[my] personal excitement is that I cannot find another group like this...that is building something and not just researching.”

While members see potential in the Steering Council, they also expressed frustration about barriers to working jointly, i.e., as a council, to scale CLC agriculture. Perceived barriers included lack of clarity about the role and autonomy of the Council within the Partnership, uncertainty about the ability of Council members to influence the strategic actions of the Partnership, and insufficient understanding of needs of researchers and commercialization staff. As

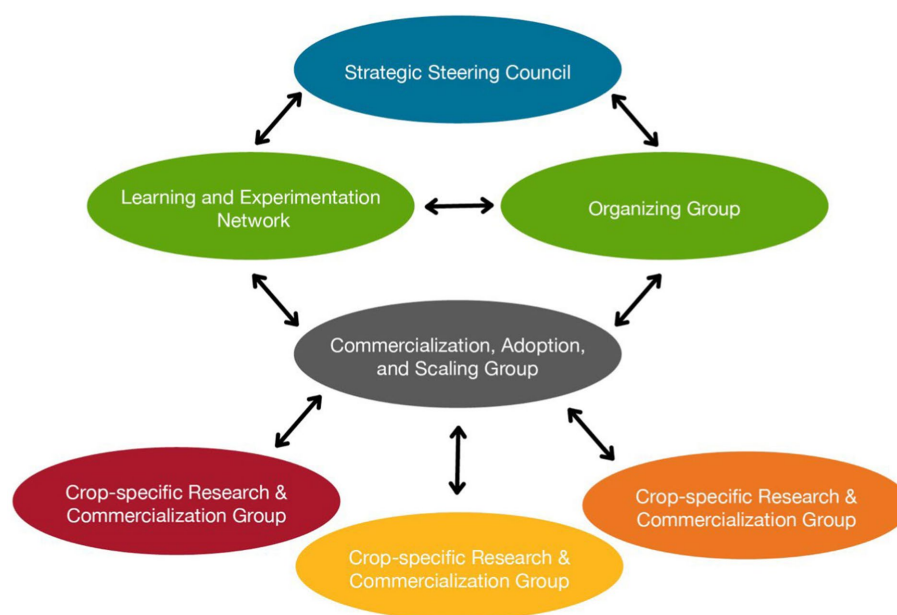


FIGURE 2

Current State of the Partnership. As presently implemented, the Partnership's polycentric network includes a Steering Council engaged in advocacy and ambassadorship in support of CLC agriculture, a Learning and Experimentation Network engaged in co-learning and action research on scaling CLC agriculture, a Commercialization, Adoption, and Scaling group building pilot supply chain projects, a set of crop-specific R&D teams advancing particular CLC crops, an organizing group, and a number of working groups addressing policy development and politics, equity and inclusion, and strategic communication. Major interactions are illustrated with two-headed arrows; for simplicity, working groups are omitted. Other inter-group interactions occur, but are less frequent and intense than those illustrated in the diagram.

one member stated, "I think it's important that everybody be on the same page about what their role is, and what we are trying to accomplish with the [Steering Council]. And I do not think we quite have that, yet." Another member spoke to the need to better understand the evidence behind the social, environmental, and economic benefits.

In response to the role uncertainty, over the course of the second year, the conveners of the Steering Council guided conversations and offered content to help the group determine how to operationalize its role in advocacy. There was largely agreement that the Steering Council's role in advocacy should be focused on building collaborative relationships with particular sectors around mutual interests in CLC agriculture. Several work streams came into focus during year 2, and after some experience attempting to launch such Council initiatives, it became clear that additional staff capacity was needed. Interviews after the second year indicated some appreciation of learning about topics and perspectives that are "outside of the circle" in which most Steering Council members operate. However, barriers to effective action by the Council were still perceived, namely the continued desire to firm up the Steering Council's purpose and the need to operationalize the advisory role. One member illustrates this by stating "There should be a 2-way conduit...these members should be taking their knowledge of the fears, aspirations, biases of their sector and bringing that to the Partnership so that if we are going astray so that we would know that." Other members spoke of the need to put boundaries around the scope of the conversations and clarify priorities: "We are opening up these wicked problems --while I really like those, I am wondering if we can bring this back to....how do we promote winter barley. Sometimes I think my mind sits in the area of 'the next steps of barley is this'....how do we move from niche to

bigger." In December, 2022, the Steering Council clarified that its purpose is to "advance Continuous Living Cover agriculture by contributing constructively to the development and sustainable commercialization of new cropping systems by: (1) Providing feedback to the Forever Green Initiative and the Partnership about strategic decisions, emerging issues and opportunities, and timely questions; (2) Providing resources to the Forever Green Initiative (relationships, financial, and other resources); (3) Acting as an ambassador for the Forever Green Initiative. To support this clarified role, in the coming year, leadership of the crop research and commercialization teams will identify emergent topics that would benefit from strategic input from the Steering Council. Through these developmental efforts, council members have advanced the Steering Council toward its intended goal-setting and resource-provision roles in the cultural evolution system outlined above—in particular, members have agreed on a goal for CLC agriculture—but their resource-provision roles has not yet been substantially implemented.

Learning and experimentation network

This group is intended to serve as the intermediate level of the polycentric network, focusing on the variation and replication dimensions of intentional cultural evolution. The Learning and Experimentation Network (referred to henceforth as the Network) is a group of persons professionally engaged in commercialization and scaling of CLC crops via market and supply-chain development. Members (16 as of this writing) are affiliated with five different organizations or advocacy groups (Table 1), and work together to share information and experience from their work to scale these crops. In parallel to the Steering Council, the Network began its work by sharing notions and visions about the nature of CLC agriculture, and then

TABLE 1 Participants in Forever Green Partnership's Strategic Steering Council and Learning and Experimentation Network, during 2020–2022.

Sector and Location	Organization	Participation
Agribusiness, Minnesota	Agribusiness (retired former executive)	Steering Council
Agribusiness, Minnesota	The Plant Pathways Company	Steering Council
Agribusiness, Minnesota	Worthwhile Ventures, Inc.	Steering Council
Agriculture NGO, Minnesota	Agricultural Resources Utilization Institute	Learning and Experimentation Network
Agriculture NGO, Minnesota	Intertribal Agriculture Council	Steering Council
Agriculture NGO, Minnesota	Kilimo Minnesota	Steering Council
Agriculture NGO, Minnesota	Minnesota Corn Growers Association	Steering Council
Agriculture NGO, Minnesota	Minnesota Farmers Union	Steering Council
Agriculture NGO, Minnesota	Naima's Farm	Steering Council
Climate NGO, Illinois	Solving for Pattern	Steering Council
Climate NGO, Minnesota	MN350	Steering Council
Environment NGO, Minnesota	Friends of the Mississippi River	Steering Council
Environment NGO, Minnesota	Minnesota Environmental Partnership	Steering Council
Environment, University Extension, Minnesota	Green Lands Blue Waters, University of Minnesota	Steering Council
Government, Minnesota	Minnesota Department of Agriculture	Steering Council
Research and commercialization, Wisconsin	Michael Fields Agricultural Institute	Learning and Experimentation Network
Research and commercialization, Wisconsin	Savannah Institute	Learning and Experimentation Network
Research and commercialization, Kansas	The Land Institute	Learning and Experimentation Network
Research and commercialization, Minnesota	University of Minnesota	Steering Council, Learning and Experimentation Network
Research and commercialization, Wisconsin	University of Wisconsin	Learning and Experimentation Network
Rural community development NGO, Minnesota	West Central Initiative	Steering Council

proceeded to a series of sessions focused on dialog on key aspects of day-to-day work. Topics have ranged widely, including framing and narrative for CLC agriculture, innovations in “green” finance, and developments in agricultural cooperatives. In interviews after the first year of operation, Network members voiced much appreciation for the learning and support that the group provided. They also expressed much uncertainty about the purpose and function of the Network, its role in the Partnership, and its autonomy. One Network member expressed this in saying, “I do not think there’s broad understanding in the Network of what the Network is supposed to be for or do. And so, that’s where I see the disjointed...confusion.” Several members spoke to the potential they saw in the Network; for example, one member stated, “How could that team spend 2h every other week to really inform one another what we are doing, solicit input on key decisions that I think they would have a good perspective on, get access to resources and relationships that we would not otherwise have, and start leveraging that.”

In the second year of its operation, the Network began a series of discussions focused on evolving challenges in commercialization and scaling, with each meeting featuring a central challenge narrated by a member. Recently, the Network has agreed to pursue an explicit program of action research (Touboulie and Walker, 2016) on particular challenges and opportunities in commercialization and scaling of CLC crops. In a group reflection conversation after the second year of operation, Network members articulated a clearer sense of the group’s purpose, value, and role in the Partnership. One member commented, “I truly see some really beautiful trust that has been built between this entire group. This is not an easy place to be vulnerable but we know that vulnerability

drives innovation and risk taking.” This statement is indicative of an apparently shared sentiment that trusting relationships have developed in the Network, and that this trust permits candid and vulnerable discussions of issues in scaling work. These trusting relationships were also seen as providing provided peer support that could be called on when needed. For instance, when one Network member was a panelist alongside another Network member, she felt she did a better job sharing her message because of the trust and collegiality she had built with this other person through the Network. Members also voiced a clearer sense of the Network’s identity and purpose: a forum and incubator for sharing experiences and insights in the work of scaling CLC agriculture, across a range of crops, ecosystems, and institutions.

Looking forward, the group was eager to share its emerging insights about its work, which they view as poorly understood by most other actors in the Partnership and agriculture generally. For example, the group hopes to influence policy development and other activities of the Partnership, such as strategic communications. These activities, if implemented, will help the group carry out its envisioned functions in the regional diversification strategy, namely to facilitate the variation and replication dimensions of intentional cultural evolution. To date, these activities are only partially implemented: the professional exchanges within the group are likely to be generating variation, as members transfer ideas for pilot-scale systems. For example, steward ownership (Sanders, 2022), an innovation in intellectual property ownership, originally applied to one crop, has recently been applied to another, as a result of communication among Network members. However, replication functions, and interactions

with the Steering Council leading to differential resource provision have not yet been robustly implemented.

Commercialization, adoption, and scaling group

As CLC crops developed by the R&D efforts of the Forever Green Initiative near commercial readiness, the Forever Green Commercialization, Adoption, and Scaling Group supports piloting, adoption, and scaling of these new crops and systems by growers, supply chain partners, end-users, and others. This group, now comprising five staff committing 100% effort, organizes and provides strategic technology transfer, risk-sharing, technical assistance, communication of technical properties, enterprise development, policy innovations, and extensive cross-sector partnership. At present, these activities focus primarily on the most commercialization-ready of Forever Green's portfolio of crops, including Kernza® perennial grain, the 'cash cover crops' pennycress, winter camelina and winter barley, perennial flax, elderberries, and hybrid hazelnuts. For these crops, pilot SSCs (Table 2) are being organized at a range of sites in the Upper Midwest region of the U.S. In each instance, this group convenes multiple supply-chain stakeholders—including end-use and intermediary firms, farmers, clients for environmental benefits produced by the crop, and other stakeholders—in collaborative efforts to develop a spatially-concentrated cluster of production of the focal crop, in a setting where there is active interest in the economic, environmental and social sustainability benefits that such a cluster could potentially provide. These clusters of production enable all parties to pilot and “debug” systems and innovations needed to create viable SSCs, e.g., post-production infrastructure or innovative public policy support for CLC agriculture. These activities are closely coordinated with the R&D teams for each of the above crops.

R&D teams

At present, *ca.* 75 scientists, primarily located at research universities in the Midwest region of the US, are developing 16 perennial and winter annual crops and associated cropping systems, and post-production handling and value-added processing systems, in affiliation with the Forever Green Initiative. Each effort is organized as a working team focused on a single crop or small group of crops, and includes geneticists, plant breeders, agronomists, environmental scientists, food scientists, and commercialization experts.

Ad-hoc working groups

The Partnership includes a number of working groups that have been developed since inception in 2019, all of which embody the cross-sector and cross-scale interactions integral to building and implementing a polycentric network for regional diversification. Working groups include an organizing group that provides overall coordination to the Partnership, and a newly-formed strategy group, with members drawn from most of the groups described above. The strategy group is charged with refining the strategy of the Partnership as a whole, and improving working relationships among the parts of the Partnership so as to enhance effective pursuit of its strategy. Another key group is striving to insure that commercialization and scaling of CLC agriculture proactively addresses justice, equity, and inclusion issues in agriculture. There are also standing groups for strategic communications, and a political working group that engages in policy advocacy and lobbying.

Progress of the Forever Green Partnership

The Partnership was implemented *de novo* in 2019. As noted, initial design, implementation, and operation of the Partnership were guided by the conceptual models outlined above. These models have been largely embraced, as working hypotheses, by the organizing group that provides overall coordination to the Partnership.

Challenges

Formation of a novel polycentric network is clearly an ambitious and inherently challenging project, and is expected to require some years of development before the network becomes effective in pursuit of its goals (Hileman and Bodin, 2019). At the time of writing, the Partnership is not yet fully functioning as a polycentric network for conscious cultural evolution, as envisioned in the regional diversification strategy outlined above. In essence, the Partnership has not yet developed certain “enabling conditions” that are important to effective polycentric networks (Carlisle and Gruby, 2019), such as agreed-upon rules of operation, cross-scale deliberation and learning, and mechanisms for accountability, all of which are important to facilitation of cultural evolution. These conditions appear essential to the processes of conscious cultural evolution (variation, selection, and replication of SSCs). These enabling conditions require agreements—and sustained collaborative activities—across nodes in the network, which highlights node development as a key milestone in the formation of effective polycentric networks. Ostrom's core design principles (Wilson et al., 2013) for effective cooperative groups offer a helpful touchstone for assessing development of effective network nodes. Principles most relevant to the initial development of individual network nodes include a shared understanding of a node's purpose and key activities, and processes for decision-making and distribution of costs and benefits of group participation. Achieving and implementing these shared understandings is likely to be complicated, particularly when a node represents a voluntary association in a “community” situation (Cabrera et al., 2018), as opposed to an organization whose leadership can mandate participation.

As may be expected from these considerations, development of the nodes of the Partnership has been complicated and slow. Interview data show that, for many participants in these nodes, shared understanding of each node's purposes and activities—and of interactions among nodes, and of the Partnership as a whole—has been slow to develop. Importantly, many participants express strong interest in taking action, and have been somewhat frustrated by deliberative activities, particularly in the Strategic Steering Council.

An important challenge is developing the nodes as semi-autonomous groups that are self-directed and self-governing, as opposed to being convened and directed by the project organizers, with relatively passive participants. In principle, this “semi-autonomous” attribute is critical to the ability of a node to function in a polycentric network on a sustained basis (Wilson et al., 2013). An important strategy for meeting this challenge has been to find ways for the node's activities to be valuable to participants even if the polycentric network is not yet functioning as a whole. Progress has been made in this respect for the Learning and Experimentation Network, whose members have actively embraced the opportunity to exchange experiences, information, and strategies regarding their work of developing new markets and supply chains for continuous-living-cover crops. This has been less successful for the Strategic

TABLE 2 Pilot supply-chain projects for various continuous-living-cover crops associated with the Forever Green Partnership in various US states (Illinois, IL; Iowa, IA, Kansas, KS; Minnesota, MN; Montana, MT; North Dakota, ND; South Dakota, SD; Wisconsin, WI).

CLC Crop	Location	Features	Number of farmers	Number of supply chain actors	Spatial extent
Kernza Perennial Grain	MN, KS, WI, MT	Technical and financial support for geographical clusters of piloting farms, farmer production cooperative, novel public (MN) seed capital fund for value-added enterprises CLC	82 approved growers in US; roughly 30 growers in MN	3 seed sources, 1 MN seed processor, 1 WI seed processor onboarding, 1 WI grain processor onboarding	~2,500 ha total licensed total, ~900 ha in MN (as of Oct 22)
Winter Camelina	MN, SD, ND, IA	Technical and financial support for geographical clusters of piloting farms	9 MN growers in 2021/22 pilot project;	2 seed sources, 2 seed processors, 4–6 major commercial actors conducting internal pilot production, 1 for-profit biotech business offering contracts	~40 ha in 2021/2022 pilot, 100 ha acres of industry pilots planted in 2022, multiple + 4,000 ha pilots planned for 2023
Hybrid Hazelnuts	WI, MN, IA	6 clusters of growers across Upper Midwest, pilot processing plant in Ashland, WI, network of leading ‘Go-First Farms’ in each cluster; piloting innovative germplasm ownership and land-access financing	50–75 growers across clusters, small number of growers and researchers (~10) account for roughly half of all production.	3–5 producer groups, 4 nurseries conducting propagation, one publicly-owned pilot processing line, 1 retail products brand, direct-to-consumer sales by growers and modest inclusion of Midwest-grown hazelnuts in limited-distribution food products	40–80 ha of hybrid research, early commercial, and hobbyist production
CLC Crop	Location	Features	Number of farmers	Number of supply chain actors	Spatial extent
Perennialized systems, including managed grazing (Grassland 2.0)	Primarily WI (some work in Driftless Region of IA, IL, and MN)	Partnering with farmer and citizen-led watershed groups to build shared ‘Story of Now’ and Vision for the future, and to identify and take action on pathways to the future.	Currently 10–15 farmers engaged in the grass-fed meat supply chain work in the Driftless started in 2022. Building out network in 2023. Custom dairy heifer grazing network in central/north-central WI ramping up. Currently 6 farmers, expanding in 2023.	Five local “learning hubs” built on watershed based groups. Two supply chain development pilot projects covering 3 of the 5 Learning Hubs. For the meat supply chain work in the Driftless, engagement with 3 processors in SW Wisconsin and 1 beef aggregation and sales cooperative that also has some processing.	~250 ha in Custom dairy heifer grazing network
Winter barley	MN	Early commercial scaling of new winter barley lines in partnership with regional seed companies and malting industry	10–20 at launch of first winter barley variety	Two seed company partners, early engagement with major (global) maltsters located in the region	Unsure

Steering Council, but there has been increasing energy around taking individual and collective action as advocates and ambassadors for continuous-living-cover agriculture and the Forever Green Partnership. The Commercialization, Adoption, and Scaling node has achieved self-direction and organization.

Crucially, we believe that node development has been limited by lack of resources for two key developmental activities. First, we have lacked capacity to engage with node participants in ongoing one-to-one discussions around their interests in node participation, questions, and concerns. These discussions appear important to stay in touch with participants as they engage in the slow, ambiguous, and

complicated work of node development. Second, there has also been a lack of resources for organizing and supporting cross-sector and cross-scale activities of the Partnership as a developing polycentric network. Such activities include information-sharing and other learning, carrying out initiatives that engage multiple nodes, and formation of shared understanding regarding collaboration between nodes in a polycentric network. Certainly, these activities and interactions are the lifeblood of effective polycentric networks (Carlisle and Gruby, 2019). In interviews, Steering Council members expressed that these activities were highly important to their ability to offer concrete support to scaling CLC agriculture, which is the core

purpose of the Council. These resource limitations may be particularly problematic in limiting “learn-by-doing” experiences for participants, as there is indication that participants in polycentric governance can increase the scope of their participation over time (Hileman and Bodin, 2019), after gaining experience. Recently, the Partnership has received new grant funding to support these cross-sector and cross-scale activities.

Accomplishments

Importantly, the main elements of the Partnership—as an implementation of the regional diversification strategy outlined above—have been formed, and certain key functional aspects of the network are coming into robust operation. First, new and highly-active elements of the Partnership have emerged, such as the Commercialization, Adoption, and Scaling group of the Forever Green Initiative, which was not part of the original design for the Partnership (Figure 1). That group and the crop-specific R&D teams have developed a set of pilot supply chains (Table 2), thus creating a set of variant SSCs, as is essential for the conscious cultural evolution process. For Kernza® perennial grain, these pilot supply chains have grown rapidly in the past several years, and now span thousands of acres, and many marketed products. Moreover, a parallel commercialization group for CLC crops has recently been initiated by the University of Wisconsin, demonstrating the replication that is key to cultural evolution. Second, the Partnership is achieving a growing reflexive capacity as a key tool for building an effective network, through the action-research methods that are being used by the Partnership’s evaluators, and by members of the Learning and Experimentation Network, and the recent formation of an evaluation group drawn from the network’s nodes, to assess functioning of the polycentric network as a whole. Finally, the Partnership has been successful in attracting and integrating resources, which is a fundamental purpose of polycentric networks (Carlisle and Gruby, 2019). These include ongoing operational support from the Clean Water Council of the State of Minnesota, and from philanthropic sources. A large research grant was obtained in 2021 for a participatory action-research (Touboulic and Walker, 2016) project focused on the Partnership, seeking to characterize and evaluate the Partnership through the eyes of participants. In the 2022 Minnesota Legislative session, new state financial support was given to the Partnership, because of broad political support for the Partnership and continuous-living cover agriculture. Very few other legislative proposals attracted such broad support, which spanned two political parties that share power in the Legislature. This success shows the resource-provision potential of the Steering Council, as members of the Council invested considerable political capital in organizing the necessary breadth of support.

Evaluation and reflexivity in the Forever Green Partnership

The Partnership seeks to build a collective critical awareness of its performance and to improve over time. These aspirations are implemented by ongoing, multi-faceted, collective evaluation of all levels of the polycentric system, and of its function as a whole, in terms of key functions, outputs, and outcomes. This evaluation is based on participatory action research (Touboulic and Walker, 2016),

implemented through developmental evaluation practices (Patton, 2010). These techniques serve to elucidate the experiences, perceptions, assumptions, and understandings of participants, and to create multiple deliberative settings for discussion of these, within nodes of the Partnership, and among nodes. Such wide-ranging and ongoing assessments are costly, requiring facilitation from skilled evaluators, and the investment of time, and cognitive and emotional engagement from all participants. In the face of the complexity of regional diversification, a particular focus of evaluation is supporting reflexivity, engaging participants in “questioning what we, and others, might be taking for granted—what is being said and not said—and examining the impact this has or might have.” (Cunliffe, 2016). Such reflexive work is widely seen as essential to addressing complex challenges (McLoughlin et al., 2020), such as development of “*complex multi-level systems to cope with a complex, multi-level problem*,” to quote Ostrom (2010) once again. In late 2022, the major nodes of the Partnership (Steering Council, Learning and Experimentation Network, Organizing Group, and Strategy Group each had gatherings for the purpose of reviewing Ostrom’s core design principles, with emphasis on articulation of each group’s purpose, autonomy of group, internal trust and equity, and give/get.

Discussion and conclusion

Fundamentally, our project is concerned with achieving a crucial, broadly-supported sustainability transition in agriculture: diversification at regional scales. Our effort to develop a regional-scale diversification strategy is part of a growing body of theory and practice addressing sustainability transitions in agriculture (El Bilali, 2020; Scoones et al., 2020). In this body of work, the multi-level perspective (Geels, 2019) is an overarching theoretical framework (El Bilali, 2019), underlying most current approaches. The multi-level perspective posits that sustainability transitions result from the joint operation of ‘top-down’ pressures for change in dominant systems (e.g., broad societal demand for climate mitigation and adaptation in agriculture), and the availability of scalable alternatives to dominant systems that meet such demand, typically resulting from ‘bottom-up’ innovation. In practice, however, most sustainability transition efforts in agriculture focus narrowly on particular scales or sectors, rather than attempting to coordinate activities across sectors and scales (El Bilali, 2020). Undoubtedly, this reflects the difficulty and cost of organizing the joint operation of effort broadly across sectors and scales (Schlaili and Urmetzer, 2019). By organizing a cross-scale and cross-sector project, we aim to advance understanding of sustainability transitions in agriculture.

We also aim to advance understanding of the value of conscious cultural evolution in sustainability transitions such as regional diversification, inspired by drawing on recent advances in understanding of conscious cultural evolution and its facilitation (Brooks et al., 2018; Atkins et al., 2019). Sustainability transitions frameworks often seek to support adaptation and evolution of fundamental societal systems. However, these frameworks have not explicitly united with the developing theory and practice of facilitated and intentional cultural evolution as a sustainability strategy (Schlaili and Urmetzer, 2019). This union offers much: if evolution and adaptation of cultural elements such as food systems is the goal, then attention to the fundamental drivers of cultural evolution and

adaptation is warranted. Specifically, we propose that intentional design for facilitated cultural evolution can markedly increase the likelihood of progress in the adaptation and evolution that is essential for transition in agriculture. Our project is thus part of a larger stream of work exploring conscious cultural evolution as a novel approach to sustainability transitions (Brooks et al., 2018; Jones et al., 2020). As Brooks et al. note, cultural evolution is a unifying framework, clarifying the logic and underlying dynamics of strategies such as adaptive management and innovation systems.

Finally, we seek to contribute to broader use of principles and practices of responsible innovation and scaling (Kuzma, 2019; Schut et al., 2020; Stilgoe et al., 2020) in addressing sustainability transitions such as regional diversification. Of course, innovation and scaling are of the essence in agricultural diversification, and calls for their “responsibility” acknowledge that all scaled innovations produce a mix of outcomes, some beneficial, others not (Herrero et al., 2020). The foundations of such responsibility are anticipation, reflexivity, inclusion, and responsiveness (Stilgoe et al., 2020). The use of polycentric governance and conscious cultural evolution provide many opportunities to implement these principles in practice. Via the internal deliberations of these networks, and ongoing feedback between the top-down and bottom-up scales in polycentric networks, there is much scope for anticipating consequences of particular diversification pathways via inclusive and participatory processes, and for collective reflexivity and responsiveness to perceived shortcomings of diversification strategies.

For example, a key value of the Partnership is to avoid diversification strategies that perpetuate current social injustices in agriculture. By implementing this value in goal-setting and resource-provisioning activities, and collaborating with farmers from historically-marginalized groups to develop diversification pathways (i.e., SSCs) that respect this value, the Partnership is striving to practice responsible innovation and scaling with respect to this goal. This requires engagement of multiple interested and affected parties in a holistic discussion of ends and means of innovation and scaling, participatory and inclusive anticipation of outcomes of alternative diversification pathways, and on-going mutual learning and reflection on the innovation and scaling process and its outcomes. These processes—albeit challenging, deliberative, and unpredictable—are all inherent in the Partnership’s polycentric and evolutionary approach. We argue that responsible innovation and scaling are essential to navigating sustainability transition projects in food and agriculture, and through

implementation of the Partnership’s strategy, we seek to build practical and conceptual approaches to taking such responsibility.

Data availability statement

The datasets presented in this article are not readily available because interview data were gathered under a confidentiality agreement. Summaries of interview data will be made available. Requests to access the datasets should be directed to jorda020@umn.edu.

Author contributions

NJ and DW developed the conceptual model. NJ, DW, KN, KM, TC, and CC contributed to the case study. KN, KM, and TC planned the evaluation research. KN and KM analyzed and interpreted interview data. All authors contributed to the article and approved the submitted version.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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References

- Ahlborg, H., Ruiz-Mercado, I., Molander, S., and Masera, O. (2019). Bringing technology into social-ecological systems research—motivations for a socio-technical-ecological systems approach. *Sustainability* 11:2009. doi: 10.3390/su11072009
- Asbjornsen, H., Hernandez-Santana, V., Liebman, M., Bayala, J., Chen, J., Helmers, M., et al. (2014). Targeting perennial vegetation in agricultural landscapes for enhancing ecosystem services. *Renewable Agric. Food Syst.* 29, 101–125. doi: 10.1017/S1742170512000385
- Atkins, P.W.D., Wilson, D.S., and Hayes, S.C. (2019). *Prosocial: Using Evolutionary Science to Build Productive, Equitable, and Collaborative Groups*. Oakland, CA: New Harbinger Publications.
- Barrett, C. B., Benton, T., Fanzo, J., Herrero, M., Nelson, R. J., Bageant, E., et al. (2020). “Socio-technical innovation bundles for Agri-food systems transformation,” in *Report of the international expert panel on innovations to build sustainable, equitable, inclusive food value chains* (Ithaca, NY, and London: Cornell Atkinson Center for Sustainability and Springer Nature).
- Blay-Palmer, A., Santini, G., Dubbeling, M., Renting, H., Taguchi, M., and Giordano, T. (2018). Validating the city region food system approach: enacting inclusive, transformational city region food systems. *Sustainability* 10:1680. doi: 10.3390/su10051680
- Blesh, J., Mehrabi, Z., Wittman, H., Kerr, R. B., James, D., Madsen, S., et al. (2023). Against the odds: network and institutional pathways enabling agricultural diversification. *One. Earth* 6. doi: 10.1016/j.oneear.2023.03.004
- Blesh, J., and Wolf, S. A. (2014). Transitions to agroecological farming systems in the Mississippi River basin: toward an integrated socioecological analysis. *Agric. Hum. Values* 31, 621–635. doi: 10.1007/s10460-014-9517-3
- Boström, M., Jönsson, A. M., Lockie, S., Mol, A. P., and Oosterveer, P. (2015). Sustainable and responsible supply chain governance: challenges and opportunities. *J. Clean. Prod.* 107, 1–7. doi: 10.1016/j.jclepro.2014.11.050
- Bowles, T. M., Mooshammer, M., Socolar, Y., Calderón, F., Cavigelli, M. A., Culman, S. W., et al. (2020). Long-term evidence shows that crop-rotation diversification

increases agricultural resilience to adverse growing conditions in North America. *One Earth* 2, 284–293. doi: 10.1016/j.oneear.2020.02.007

Brauman, K. A. (2015). Hydrologic ecosystem services: linking ecohydrologic processes to human well-being in water research and watershed management. *Wiley Interdiscip. Rev. Water* 2, 345–358. doi: 10.1002/wat2.1081

Brooks, J. S., Waring, T. M., Mulder, M. B., and Richerson, P. J. (2018). Applying cultural evolution to sustainability challenges: an introduction to the special issue. *Sustain. Sci.* 13, 1–8. doi: 10.1007/s11625-017-0516-3

Bui, S., Cardona, A., Lamine, C., and Cerf, M. (2016). Sustainability transitions: insights on processes of niche-regime interaction and regime reconfiguration in Agri-food systems. *J. Rural Studies* 48, 92–103. doi: 10.1016/j.jrurstud.2016.10.003

Cabrera, D., Cabrera, L., Powers, E., Solin, J., and Kushner, J. (2018). Applying systems thinking models of organizational design and change in community operational research. *Eur. J. Oper. Res.* 268, 932–945. doi: 10.1016/j.ejor.2017.11.006

Carlisle, K., and Gruby, R. L. (2019). Polycentric systems of governance: a theoretical model for the commons. *Policy Stud. J.* 47, 927–952. doi: 10.1111/psj.12212

Clancy, K., and Ruhf, K. Z. (2018). Digging deeper: new thinking on “regional”. *J. Agric. Food Sys. Community Dev.* 8, 1–5. doi: 10.5304/jafscd.2018.083.008

Cooley, L., and Papoulidis, J. (2017). Tipping the scales: shifting from projects to scalable solutions in fragile states. *Development* 60, 190–196. doi: 10.1057/s41301-018-0155-8

Cox, M., and Schoon, M. (2019). “Adaptive governance from an evolutionary perspective” in *Global challenges, Governance, and Complexity*. ed. V. Galaz (Northampton, MA: Edward Elgar Publishing Ltd), 78–93.

Cunliffe, A. L. (2016). “On becoming a critically reflexive practitioner” redux: what does it mean to be reflexive? *J. Manage. Educ.* 40, 740–746. doi: 10.1177/1052562916668919

Dorsch, M. J., and Flachsland, C. (2017). A polycentric approach to global climate governance. *Global Environ. Polit.* 17, 45–64. doi: 10.1162/GLEP_a_00400

Duru, M., Therond, O., and Fares, M. (2015). Designing agroecological transitions; a review. *Agron. Sustain. Dev.* 35, 1237–1257. doi: 10.1007/s13593-015-0318-x

El Bilali, H. (2019). The multi-level perspective in research on sustainability transitions in agriculture and food systems: a systematic review. *Agric* 9:74. doi: 10.3390/agriculture9040074

El Bilali, H. (2020). Transition heuristic frameworks in research on agro-food sustainability transitions. *Environ. Dev. Sustain.* 22, 1693–1728. doi: 10.1007/s10668-018-0290-0

Fasting, S., Bacudo, I., Damen, B., and Dinesh, D. (2021). Climate governance and agriculture in Southeast Asia: learning from a polycentric approach. *Front. Polit. Sci.* 3:698431. doi: 10.3389/fpos.2021.698431

Flyvbjerg, B. (2011). “Case study,” in *Encyclopedia of evaluation. The SAGE handbook of qualitative research*. eds. N. K. Denzin and Y. S. Lincoln (Thousand Oaks, CA: Sage), 301–306.

Forever Green Initiative. (2023). Available at: <https://forevergreen.umn.edu/>

Geels, F. W. (2019). Socio-technical transitions to sustainability: a review of criticisms and elaborations of the multi-level perspective. *Curr. Opin. Environ. Sustain.* 39, 187–201. doi: 10.1016/j.cosust.2019.06.009

Gurzawska, A. (2019). Towards responsible and sustainable supply chains—innovation, multi-stakeholder approach and governance. *Philos. of Manage.* 19, 267–295. doi: 10.1007/s40926-019-00114-z

Heckelman, A., Chappell, M. J., and Wittman, H. (2022). A polycentric food sovereignty-approach to climate resilience in the Philippines. *Elem Sci Anth.* 10:00033. doi: 10.1525/elementa.2020.00033

Hermans, F., Geerling-Eiff, F., Potters, J., and Klerkx, L. (2019). Public-private partnerships as systemic agricultural innovation policy instruments—assessing their contribution to innovation system function dynamics. *NJAS-Wageningen J. Life Sci.* 88, 76–95. doi: 10.1016/j.njas.2018.10.001

Herrero, M., Thornton, P. K., Mason-D'Croz, D., Palmer, J., Benton, T. G., Bodirsky, B. L., et al. (2020). Innovation can accelerate the transition towards a sustainable food system. *Nat. Food* 1, 266–272. doi: 10.1038/s43016-020-0074-1

Hileman, J., and Bodin, Ö. (2019). Balancing costs and benefits of collaboration in an ecology of games. *Policy Stud. J.* 47, 138–158. doi: 10.1111/psj.12292

Home, R., Bouagnimbeck, H., Ugas, R., Arbenz, M., and Stolze, M. (2017). Participatory guarantee systems: organic certification to empower farmers and strengthen communities. *Agroecol. Sustainable Food Syst.* 41, 526–545. doi: 10.1080/21683565.2017.1279702

Johnson, B. (2020). Global development and environment institute Tufts University. Land Value and Soil Quality: An Untapped Incentive Structure. Available at: <https://sites.tufts.edu/gdae/files/2020/06/Ben-Policy-Brief-13.pdf>

Jones, J. H., Ready, E., and Pisor, A. C. (2020). Want climate-change adaptation? Evolutionary theory can help. *Am J Hum. Biol.* 33:e23539. doi: 10.1002/ajhb.23539

Jordan, N. R., Dorn, K., Runck, B., Ewing, P., Williams, A., Anderson, K. A., et al. (2016). Sustainable commercialization of new crops for the agricultural bioeconomy. *Elem. Sci. Anth.* 4:000081. doi: 10.12952/journal.elementa.000081

Jordan, N. R., Mulla, D. J., Slotterback, C., Runck, B., and Hays, C. (2018). Multifunctional agricultural watersheds for climate adaptation in Midwest USA: commentary. *Renew. Agric Food Syst.* 33, 292–296. doi: 10.1017/S1742170517000655

Jouan, J., Ridier, A., and Carof, M. (2019). Economic drivers of legume production: approached via opportunity costs and transaction costs. *Sustainability* 11:705. doi: 10.3390/su11030705

Kivimaa, P., Hyysalo, S., Boon, W., Klerkx, L., Martiskainen, M., and Schot, J. (2019). Passing the baton: how intermediaries advance sustainability transitions in different phases. *Environ. Innov. Societal Trans.* 31, 110–125. doi: 10.1016/j.eist.2019.01.001

Klerkx, L., and Begemann, S. (2020). Supporting food systems transformation: the what, why, who, where and how of mission-oriented agricultural innovation systems. *Agric. Syst.* 184:102901. doi: 10.1016/j.agsy.2020.102901

Kremen, C., and Miles, A. (2012). Ecosystem services in biologically diversified versus conventional farming systems: benefits, externalities, and trade-offs. *Ecol. Soc.* 17:40. doi: 10.5751/ES-05035-170440

Kuzma, J. (2019). Procedurally robust risk assessment framework for novel genetically engineered organisms and gene drives. *Regul. Governance* 15, 1144–1165. doi: 10.1111/rego.12245

Leeuwis, C., and Aarts, N. (2011). Rethinking communication in innovation processes: creating space for change in complex systems. *J. Agric. Educ. Ext.* 17, 21–36. doi: 10.1080/1389224X.2011.536344

Lin, B. B. (2011). Resilience in agriculture through crop diversification: adaptive management for environmental change. *BioSci.* 61, 183–193. doi: 10.1525/bio.2011.61.3.4

Lockeretz, W. (1988). Agricultural diversification by crop introduction: the US experience with the soybean. *Food Policy* 13, 154–166. doi: 10.1016/0306-9192(88)90028-0

Manson, S. M., Jordan, N. R., Nelson, K. C., and Brummel, R. F. (2016). Modeling the effect of social networks on adoption of multifunctional agriculture. *Environ. Model. Softw.* 75, 388–401. doi: 10.1016/j.envsoft.2014.09.015

Markolf, S. A., Chester, M. V., Eisenberg, D. A., Iwaniec, D. M., Davidson, C. I., Zimmerman, R., et al. (2018). Interdependent infrastructure as linked social, ecological, and technological systems (SETs) to address lock-in and enhance resilience. *Earth's Future* 6, 1638–1659. doi: 10.1029/2018EF000926

Marshall, G. R. (2009). Polycentricity, reciprocity, and farmer adoption of conservation practices under community-based governance. *Ecol. Econ.* 68, 1507–1520. doi: 10.1016/j.ecolecon.2008.10.008

McLoughlin, C. A., Thoms, M. C., and Parsons, M. (2020). Reflexive learning in adaptive management: a case study of environmental water management in the Murray Darling basin. *Australia. River Res. Appl.* 36, 681–694. doi: 10.1002/rra.3607

Melchior, I. C., and Newig, J. (2021). Governing transitions towards sustainable agriculture—taking stock of an emerging field of research. *Sustainability* 13:528. doi: 10.3390/su13020528

Meynard, J. M., Charrier, F., Le Bail, M., Magrini, M. B., Charlier, A., and Messéan, A. (2018). Socio-technical lock-in hinders crop diversification in France. *Agron. Sustain. Dev.* 38:54. doi: 10.1007/s13593-018-0535-1

Meynard, J. M., Jeuffroy, M. H., Le Bail, M., Lefèvre, A., Magrini, M. B., and Michon, C. (2017). Designing coupled innovations for the sustainability transition of agrifood systems. *Agric. Syst.* 157, 330–339. doi: 10.1016/j.agsy.2016.08.002

Mier y Terán Giménez Cacho, M., Giraldo, O., Aldasoro, M., Morales, H., Ferguson, B., Rosset, P., et al. (2018). Bringing agroecology to scale: key drivers and emblematic cases. *Agroecol. Sustain. Food Syst.* 42, 637–665. doi: 10.1080/21683565.2018.1443313

Montenegro de Wit, M., and Iles, A. (2016). Toward thick legitimacy: creating a web of legitimacy for agroecology. *Elem. Sci. Anth.* 4. doi: 10.12952/journal.elementa.000115

Morais, D. O. C., and Silvestre, B. S. (2018). Advancing social sustainability in supply chain management lessons from multiple case studies in an emerging economy. *J. Clean. Prod.* 199, 222–235. doi: 10.1016/j.jclepro.2018.07.097

Mortensen, D. A., and Smith, R. G. (2020). Confronting barriers to cropping system diversification. *Front. Sustain. Food Syst.* 4:564197. doi: 10.3389/fsufs.2020.564197

Nicol, P. (2020). Pathways to scaling agroecology in the city region: scaling out, scaling up and scaling deep through community-led trade. *Sustainability* 12:7842. doi: 10.3390/su12197842

Ostrom, E. (2010). A multi-scale approach to coping with climate change and other collective action problems. *Solutions* 1, 27–36.

Partnership, Forever Green. (2023). Available at: <https://forevergreenpartnership.umn.edu>

Patton, M. Q. (2010). *Developmental Evaluation: Applying Complexity Concepts to Enhance Innovation and Use*. New York: The Guilford Press.

Peterson, H. (2009). Transformational supply chains and the 'wicked problem' of sustainability: aligning knowledge, innovation, entrepreneurship, and leadership. *J. Chain Net. Sci.* 9, 71–82. doi: 10.3920/JCNS2009.x178

Prokopy, L. S., Gramig, B. M., Bower, A., Church, S. P., Ellison, B., Gassman, P. W., et al. (2020). The urgency of transforming the Midwestern US landscape into more than corn and soybean. *Agric. Hum. Values* 37, 537–539. doi: 10.1007/s10460-020-10077-x

- Ranjan, P., Singh, A. S., Tomer, M. D., Lewandowski, A. M., and Prokopy, L. S. (2019). Lessons learned from using a decision-support tool for precision placement of conservation practices in six agricultural watersheds in the US Midwest. *J. Environ. Manag.* 239, 57–65. doi: 10.1016/j.jenvman.2019.03.031
- Rockström, J., Edenhofer, O., Gaertner, J., and DeClerck, F. (2020). Planet-proofing the global food system. *Nature Food* 1, 3–5. doi: 10.1038/s43016-019-0010-4
- Rosset, P., Val, V., Barbosa, L., and McCune, N. (2019). Agroecology and La via Campesina II. Peasant agroecology schools and the formation of a sociohistorical and political subject. *Agroecol. Sustain. Food Syst.* 43, 895–914. doi: 10.1080/21683565.2019.1617222
- Sanders, A., (2022). Binding capital to free purpose: Steward ownership in Germany (January 28, 2022). Available at SSRN: <https://ssrn.com/abstract=4144623>
- Schlaile, M. P., and Urmeter, S. (2019). “Transitions to sustainable development,” in *Decent Work and Economic Growth. Encyclopedia of the UN Sustainable Development Goals*. eds. W. Leal Filho, A. Azul, L. Brandli, P. Özuyar and T. Wall (Switzerland: Springer)
- Schulte, L. A., Niemi, J., Helmers, M. J., Liebman, M., Arbuckle, J. G., and James, D. E. (2017). Prairie strips improve biodiversity and the delivery of multiple ecosystem services from corn–soybean croplands. *PNAS* 114, 11247–11252. doi: 10.1073/pnas.1620229114
- Schut, M., Leeuwis, C., and Thiele, G. (2020). Science of scaling: understanding and guiding the scaling of innovation for societal outcomes. *Agric. Syst.* 184:102908. doi: 10.1016/j.agry.2020.102908
- Scoones, I., Stirling, A., Abrol, D., Atela, J., Charli-Joseph, L., and Eakin, H. (2020). Transformations to sustainability: combining structural, systemic and enabling approaches. *Opin. Environ. Sustain.* 42, 65–75. doi: 10.1016/j.cosust.2019.12.004
- Sengers, F., Wieczorek, A. J., and Raven, R. (2019). Experimenting for sustainability transitions: a systematic literature review. *Technol. Forecast. Soc. Chang.* 145, 153–164. doi: 10.1016/j.techfore.2016.08.031
- Seyfang, G., Hielscher, S., Hargreaves, T., Martiskainen, M., and Smith, A. (2014). A grassroots sustainable energy niche? Reflections on community energy in the UK. *Environ. Innov. Soc. Trans.* 13, 21–44. doi: 10.1016/j.eist.2014.04.004
- Stake, R. E. (2006). *Multiple case study analysis*. New York, NY: Guilford Press.
- Stake, R. E., and Trumbull, D. J. (1982). Naturalistic generalizations. *Rev. J. Philos. Soc. Sci.* 7, 1–12.
- Stefani, G., Nocella, G., and Sacchi, G. (2020). Piloting a Meta-database of Agroecological transitions: an example from sustainable cereal food systems. *Agriculture* 10:219. doi: 10.3390/agriculture10060219
- Steiner, A., Aguilar, G., Bomba, K., Bonilla, J.P., Campbell, A., Echeverria, R., et al. (2020). Actions to transform food systems under climate change. Wageningen, The Netherlands: CGIAR Research Program on Climate Change, Agriculture and Food Security (CCAFS).
- Stilgoe, J., Owen, R., and Macnaghten, P. (2020). “Developing a framework for responsible innovation,” in *The Ethics of Nanotechnology, Geoengineering and Clean Energy*. eds. A. Maynard and J. Stilgoe (Oxfordshire: Routledge), 347–359.
- Tamburini, G., Bommarco, R., Wanger, T. C., Kremen, C., van der Heijden, M. G., Liebman, M., et al. (2020). Agricultural diversification promotes multiple ecosystem services without compromising yield. *Sci. Adv.* 6:eaba1715. doi: 10.1126/sciadv.aba1715
- Temkin, A., Evans, S., Manidis, T., Campbell, C., and Naidenko, O. V. (2019). Exposure-based assessment and economic valuation of adverse birth outcomes and cancer risk due to nitrate in United States drinking water. *Environ. Res.* 176:108442. doi: 10.1016/j.envres.2019.04.009
- Tomich, T. P., Lidder, P., Dijkman, J., Coley, M., Webb, P., and Gill, M. (2019). Agri-food systems in international research for development: ten theses regarding impact pathways, partnerships, program design, and priority-setting for rural prosperity. *Agric. Syst.* 172, 101–109. doi: 10.1016/j.agry.2018.12.004
- Touboul, A., and Walker, H. (2016). A relational, transformative and engaged approach to sustainable supply chain management: the potential of action research. *Hum. Relat.* 69, 301–343. doi: 10.1177/0018726715583364
- Westermann, O., Förch, W., Thornton, P. K., Körner, J., Cramer, L., and Campbell, B. (2018). Scaling up agricultural interventions: case studies of climate-smart agriculture. *Agric. Syst.* 165, 283–293. doi: 10.1016/j.agry.2018.07.007
- Wigboldus, S., Klerkx, L., and Leeuwis, C. (2020). “Making scale work for sustainable development: a framework for responsible scaling of agricultural innovations in Adenle,” in *Science, technology, and innovation for sustainable development goals: Insights from agriculture, health, environment, and energy*. eds. A. A. Chertow, M. R. Moors and D. J. Pannell (New York: Oxford University Press), 518–544. doi: 10.1093/oso/9780190949501.003.0025
- Wigboldus, S., Klerkx, L., Leeuwis, C., Schut, M., Muilerman, S., and Jochemsen, H. (2016). Systemic perspectives on scaling agricultural innovations—a review. *Agron. Sustain. Dev.* 36:46. doi: 10.1007/s13593-016-0380-z
- Willett, W., Rockström, J., Loken, B., Springmann, M., Lang, T., Vermeulen, S., et al. (2019). Food in the Anthropocene: the EAT–lancet commission on healthy diets from sustainable food systems. *Lancet* 393, 447–492. doi: 10.1016/S0140-6736(18)31788-4
- Wilson, D.S. (2019). *This View of Life: Completing the Darwinian Revolution*. New York, Pantheon.
- Wilson, D. S., Ostrom, E., and Cox, M. E. (2013). Generalizing the core design principles for the efficacy of groups. *J. Econ. Behav. Organ.* 90, S21–S32. doi: 10.1016/j.jebo.2012.12.010
- Woltering, L., Fehlenberg, K., Gerard, B., Ubels, J., and Cooley, L. (2019). Scaling—from “reaching many” to sustainable systems change at scale: a critical shift in mindset. *Agric. Syst.* 176:102652. doi: 10.1016/j.agry.2019.102652
- Wyborn, C., Datta, A., Montana, J., Ryan, M., Leith, P., Chaffin, B., et al. (2019). Co-producing sustainability: reordering the governance of science, policy, and practice. *Annu. Rev. Environ. Resour.* 44, 319–346. doi: 10.1146/annurev-environ-101718-033103



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EDITED BY

Carol Williams,
University of Wisconsin – Madison,
United States

REVIEWED BY

Nicholas R. Jordan,
Independent Researcher,
St. Paul, MN, United States
Jennifer Blesh,
University of Michigan, United States

*CORRESPONDENCE

Mrill Ingram
✉ mingham@uwisc.edu

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Building cover crop expertise with citizen science in the upper Midwest: supporting farmer innovation in a time of change

Mrill Ingram^{1,2*}

¹Center for Integrated Agricultural Systems, University of Wisconsin-Madison, Wisconsin, United States,

²Michael Fields Agricultural Institute, East Troy, WI, United States

The use of cover cropping, as one element in a continuous living cover approach, has the potential to protect water quality and promote soil health, but overall U.S. acreage in cover crops as well as adoption rates remain low. Research on behavioral barriers to cover crop use indicates a lack of information about locally suitable practices and cover crop varieties, as well as the additional management complexity of cover cropping and a high degree of uncertainty in outcomes, especially in areas with shorter growing seasons. This paper describes the development of a citizen science project on cover cropping in Wisconsin designed to (i) generate more geographically distributed data on cover crop performance in the state; and (ii) build understanding of farmer decision-making around growing practices, barriers, and motivations for cover cropping. Citizen science, as it relies on physically distributed members of the public in data generation, is well established as an avenue for generating environmental data. We engage the approach as a tool for also researching influences on individual behavior and identifying potential leverage points for change, especially on-farm innovation and experimentation. I share project findings regarding cover cropping practices and biomass production, results on motivations and influences for cover cropping, as well as participatory approaches to share those results with farmers. This project also offers more general insights into how the citizen science model can be used to expand understanding of decision-making contexts, and to develop responsive outreach efforts that support participants in taking action.

KEYWORDS

cover crops, U.S. Midwest, agricultural transformation, participatory research, citizen science

1. Introduction – cover cropping and agricultural transformation

This paper shares the case study of an ongoing citizen science effort to improve understanding and use of cover cropping in Wisconsin. “CCROP,” or Cover Crops Research and Outreach Project, is a collaborative effort on cover crop research and outreach. The collaboration includes a citizen science element designed to generate more physically distributed data on the practices and results of cover cropping as well as a broader understanding of the context and processes of decision-making by farmers who cover crop, and how best to support on-farm innovation and engagement with environmentally sound practices. Objectives include linking information produced by farmers on their agronomic practices with researcher-produced data

from long-term agronomic studies on cover cropping in Wisconsin. Long-term agronomic studies include researcher-led work at the Michael Fields Agricultural Institute in Troy, Wisconsin, as well as the Wisconsin Integrated Cropping Systems Trials (WICST) at UW-Madison, a 24-hectare randomized and replicated experiment evaluating conventional, organic, grazing, and cover cropping systems, and one of the longest running cropping trials and associated databases about sustainable agriculture in the country.

Understanding barriers to conservation practices and how and why farmers overcome them is critical for transitioning to a more regenerative agriculture (Reimer et al., 2012; Blesh and Wolf, 2014; Roesch-McNally G. E. et al., 2018; Roesch-McNally G. et al., 2018). In this paper I describe how the CCROP project identified a lack of locally appropriate cover crop information and developed a citizen science effort to fill that gap. The project also explored the potential of the citizen science model in agriculture to expand understanding of how such data can be useful to farmers interested in innovative practices.

As the impacts of agriculture in both creating and potentially mitigating environmental harm are increasingly part of a broad conversation, so too is the role of farmers as critical agents in responding to that harm (Mottet et al., 2020; Petersen-Rockney et al., 2021). The U.S. food system produces large volumes of food and commodities at low per unit cost but accompanied by severe negative externalities including widespread fresh and marine water pollution, greenhouse gas production, and soil loss through erosion. This is especially notable in the highly specialized intensive corn and soybean landscapes of the Midwestern U.S. (Prokopy et al., 2020; Matson and VandenBrook, 2021). Individual on-farm decision making—about practices such as cover cropping, tilling the soil, and diversifying production—is being scrutinized in the context of a broader conversation on the role of agriculture in providing such public environmental goods as soil health, clean drinking water quality, and carbon capture (Vanni, 2014; Lamine and Dawson, 2018; Burchfield et al., 2022).

CCROP was initiated in 2017 to better understand the current use and conservation potential of cover cropping by farmers in Wisconsin, and to inform policymakers regarding the role of the state in supporting cover cropping as a practice beneficial to water quality and soil health. Advocates of cover cropping—planting a single or mix of plant species along with, or following, a cash crop—promote potential multiple benefits including building soil fertility, preventing soil compaction, erosion, and nutrient runoff from fields, boosting biodiversity by supporting pollinators and other wildlife, managing weeds and insect pests, as well as building ecological resilience in the context of climate extremes, including droughts and flooding. With 75% of the U.S. Midwest's agricultural land in corn and soybeans, cover cropping offers a tool in shifting current conventional agricultural practice toward a more holistic management approach that emphasizes continuous living cover. More consistent plant coverage on agricultural fields helps to store carbon, build soil health, and reduce erosion leading to water pollution, especially nutrient loading of waterways, which in turn threatens drinking water and the health of streams, rivers and ultimately, the ocean (Cates et al., 2018; Cates and Jackson, 2019). Cover cropping, as it requires more complex management approaches, can offer an “on ramp” to other site-sensitive production practices for farmers in intensive production systems (Roesch-McNally G. E. et al., 2018; Roesch-McNally G. et al., 2018; Thompson et al., 2021).

According to the 2017 USDA agricultural census, the percent of U.S. cropland planted with cover crops increased by 50% between 2012 and 2017, from just over 10 million acres to more than 15 million. But those acres still only account for about 4% of the nation's total cropland (Dunn et al., 2016). Nationally, cover crop adoption rates increased from 3.4% in 2012 to 5.1% in 2017 but vary a great deal across and within states as they are influenced by policy, environmental conditions, and other drivers. For example, Maryland, which has been heavily promoting and subsidizing cover crops for over a decade, especially within the Chesapeake Bay watershed, had an adoption rate of about 33% in 2017, while rates declined in other states. Cover crop adoption in Iowa is more common in the southeastern portion of the state where soils have lower organic matter and higher erodibility (Wallander et al., 2021). The diverse drivers of adoption suggest a dynamic and complex mix of benefits, costs, and policy influences on cover cropping decisions.

USDA estimates of cover crop use on cropped land in Wisconsin are 6% to 10% of cropped acres in most counties, with 10%–15%, and even over 15%, in a “hotspot” of central to western Wisconsin counties (Siefert, 2017; Wallander et al., 2021). Cover crop adoption rates in Wisconsin and Minnesota have not kept pace with Illinois, Indiana, and Ohio, and farmers in these more northern states are also more likely to stop using them (Seifert et al., 2018). Research on the biophysical impacts of cover cropping complicates easy conclusions about the benefits to soil health and water quality, especially across the wide range of conditions in which farmers grow (Myers et al., 2019; Vincent-Caboud et al., 2019; Sanford et al., 2022). Especially in the more northern areas of Wisconsin, a shorter growing season challenges farmers to plant a fall cover crop and for it to establish the biomass needed to prevent erosion and produce other cover crop benefits.

Thus, cover crop performance varies across different growing conditions, and on any particular farm a mix of variables impact “success” or “failure.” When it comes to cover crop management, one size does not fit all, or even the same person year to year. In a series of focus groups with corn belt farmers in Indiana, Iowa, and Illinois about barriers to conservation practices, Ranjan et al. (2020) reported that cover crops were not particularly popular, both because of the complex nature of the practice, and also due to dissatisfaction with the continuity of outreach and resources available to support sustained use of cover crops. In addition, decision-making path dependency and technological lock-in create barriers for farmers, especially those invested in intensive production systems (Gould et al., 2004; Roesch-McNally G. E. et al., 2018).

For some sectors like organic agriculture, including cover crops in rotations is a necessary form of soil fertility. But for many conventional farmers the uncertain outcomes and additional variables to manage, including the necessary investment of cost and time to experiment and fine tune them for each field, create multiple challenges in adding cover crops to a rotation.

Despite these challenges, however, benefits of cover cropping are well established (Myers, 2023), and the practice is increasingly encouraged and incentivized, promoted and funded via a variety of federal, state, and regional conservation programs (Siefert, 2018; Hellerstein et al., 2019; Wallander et al., 2021). In a 2021 address before Congress, President Joe Biden specifically mentioned “farmers planting cover crops” to capture carbon. A number of state government and private incentive programs support farmers in cover cropping; funding of federal and state conservation programs is highly correlated

to cover cropping rates (Ramirez et al., 2015; Zhou et al., 2022). Illinois, Indiana, Iowa, and Wisconsin in recent years have initiated programs to offer crop insurance rebates on fields with cover crops.

2. Identifying information gaps on cover cropping in Wisconsin

Interest in cover crops among Wisconsin farmers is strong despite aforementioned barriers. In 2020, the CCROP collaborative surveyed agricultural educators around Wisconsin to learn more about how educators viewed farmer interest, knowledge deficits, and perceived barriers to the use of cover crops in the state (Krome and Ingram, 2020). Of the 90 educators who responded, 40 were county conservation specialists, 8 were county ag extension agents, 7 were farmer-led watershed group collaborators with the others representing various university, agency, and crop consultant positions. We had roughly even representation in all quadrants of the state with 5 people reporting working statewide. Over 95% reported providing cover crop information to farmers in their area in the past year, with people receiving from under 5 to over 20 inquiries. In terms of preparedness, some 62% of 89 respondents indicated they did *not* have sufficient locally specific information on cover crops to answer farmers questions, but this differed according to location. Respondents working in the south and west of the state were more likely to respond saying they had the locally appropriate information to answer questions, while people in the north and east were more likely to report lacking appropriate information (86% and 65%, respectively; Figure 1). Respondents who reported covering the full state or multiple areas (17) were 82% more likely than those working in single

quadrants to indicate that they lacked sufficient locally appropriate information to answer farmers' cover crop questions.

A follow-up conversation about the survey results between members of the CCROP team and a subset of county conservationists offered specifics on what kinds of information farmers are lacking, as well as the complexity of decision making around cover crops. Participants noted that cover crop equipment setup is an area where they struggle to provide information. They identified producer-led groups as especially effective in providing equipment-related information, especially in explaining planter components, how to repurpose equipment, and working on a tight budget. One participant commented that information on cover crops targeted to farmers can be "fairly technical," presenting an additional barrier to more risk averse farmers. Farmer testimonials, including videos, may help make cover crops more accessible and provide a farmer-to-farmer perspective. Another participant noted that adopters of cover crops have encountered challenges which have left others hesitant to try cover crops—aerial seeding failures were specifically mentioned.

Participants in the conversation also shared that while costs associated with cover cropping is often raised by farmers and educators as an issue, they have effectively responded to such concerns by presenting cover crops as one element in a "systems approach" to overall farm sustainability. This observation echoes the qualitative results from farmer focus groups held by Roesch-McNally G. et al., (2018), who reported that for farmers who viewed challenges in implementing cover crops as creative management opportunities and took a trial-and-error approach, cover crops were just one piece in a larger dynamic "whole system."

With these Wisconsin survey results in mind, the CCROP collaborators launched a citizen science effort in 2020 to respond to the lack of locally appropriate information about cover crop

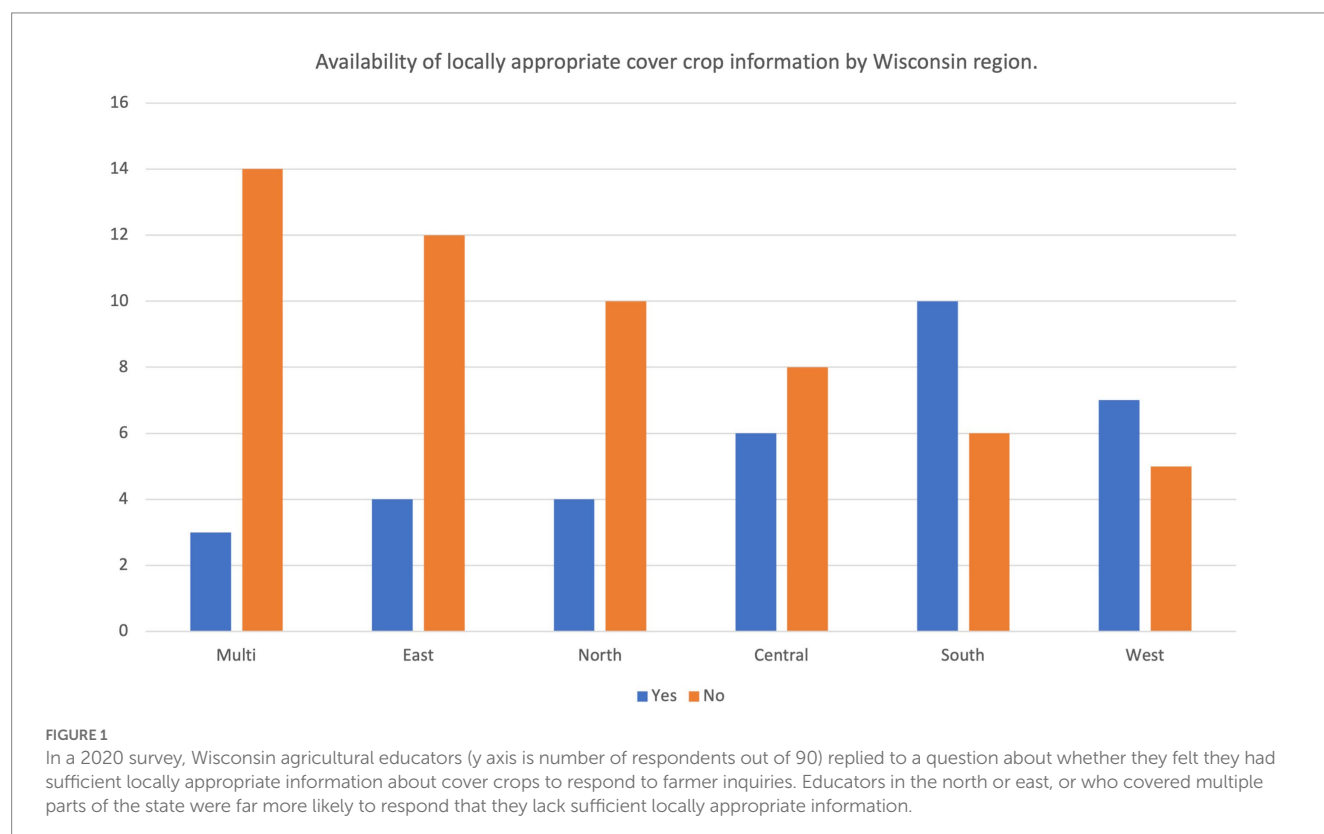




FIGURE 2

Participation and action continuum (adapted from Israel et al., 2005). The involvement of community members in projects at the left end of the spectrum can be limited to supplying data. In more community-driven and action-oriented projects, citizens help define issues and questions, and collect and analyze data in collaboration with researchers. The more dedicated the action focus, the more important participation becomes.

performance. We also sought to use the citizen science method to learn more about the context within which Wisconsin farmers were navigating the complexity of cover crop decisions.

Before turning to our methods, survey results, and discussion, I briefly review literature on the particular promise of citizen science in agricultural settings. This context is useful for understanding how the citizen science model can work not only as a data gathering tool but also as a method to support farmers in taking action, specifically to experiment with more complex environmentally responsive practices.

3. Agricultural citizen science

Thanks to the spread of communication networks and affordable connecting devices such as cell phones, the use of citizen science is expanding, especially in the environmental sciences (Strasser et al., 2018). Generally speaking, citizen science is the voluntary participation of members of the public in conducting scientific research. Although agriculture has seen relatively fewer such projects, citizen science engaging with farmers is on the rise, offering an opportunity to reflect on the unique potential of farmers as citizen scientists, as well as on the element of participation as it is realized in different projects (Ryan et al., 2018; Kimura and Kinchy, 2020; Mourad et al., 2020; van de Gevel et al., 2020; Ebitu et al., 2021).

Citizen science efforts fall on a spectrum of participation. Projects can range from a narrow engagement with participation that connects with citizens as primarily suppliers of data within a research framework defined by an external university-based researcher, to a much more collaborative and community-instigated research effort in which citizens define a research agenda and methods. Community-led movements have initiated important research—asking new questions and producing knowledge that challenges orthodox views (Gaventa, 2002; Ingram, 2007; Strasser et al., 2018).¹ The spectrum includes a

range of models for coproduced research design and knowledge creation (Harrison, 2011; Kasperowski et al., 2017; Ottinger, 2017).

A distinctive feature of the participatory nature of citizen science is the centering of new knowledge as it leads to action and application, and generally speaking the more participatory a project, the more action-oriented the results (Figure 2). Alan Irwin (2015) notably described citizen science as an avenue by which publicly funded research can be held accountable to the public good. Citizen science projects can affirm and build expertise outside of academia and be avenues via which academic expertise is made available to public concerns. Irwin also observed a potential connection between citizen science and collective action, observing how citizen science projects have the potential to help build alliances between groups and to “catch the attention of different parties and draw them in in a relatively sustained fashion,” (Irwin, 2015, p. 36). This promise, as it links knowledge generation to collective action, expands significantly on a more limited notion of citizen science as an individually oriented educational tool and avenue for building support for the scientific endeavor.

Farmers’ daily work making land management and agricultural production decisions can be understood as ongoing experimentation, generating “grounded expertise” (Bendfeldt et al., 2021). Strasser et al. (2018) have identified something similar in the “embodied” and “situated” knowledge resulting from personal experiences of community members involved in citizen science. Reviewing the literature on farmer adoption of conservation practices in the U.S., Thompson et al. (2021) note that most studies treat adoption as dichotomous—a farm has either adopted a practice or not. Citing Pannell et al. (2006), they argue for conceiving of farmer engagement with conservation practices as a “continuous learning process,” which includes an always ongoing series of activities gathering information, experimenting, and scaling up or dis-adopting.

This perception of farmers as continuously generating knowledge from ongoing experimentation, gathering data, and applying that information in trial-and-error suggests the appropriateness of an action-oriented citizen science effort in an agricultural context.

¹ Cooper et al. (2021) make a clarifying distinction between citizen science and “community science.” They write: “The term community science should be reserved for projects that focus on local priorities and local perspectives

and are able to maintain the locus of power in the community [such that] authority, power, and funding rests with communities.”

Knowledge “coproduction,” as it combines academic and nonacademic expertise in defining as well as solving problems, is increasingly central to sustainability research, and prioritizes action-oriented, context-based, and interactive knowledge generation (Norström et al., 2020), all of which can be featured in citizen science approaches.

Such action-oriented approaches to citizen science can also offer a correction to a “deficit” model in which farmers are viewed as passive, even reluctant targets of individually focused informational and behavioral change efforts (Schneider and Ingram, 1993). In a study of the use of climate forecasting tools, for example, Feldman and Ingram (2009) observed a lack of engagement by farmers even as they were facing new challenges related to drought and climate change. The authors suggested that farmers were not taking advantage of the tools at least in part due to a one-way delivery of the information—a “loading dock” model—and argued for the need for the sharing of new information and tools via “knowledge networks that are recursive, interactive, and end-to-end useful.” People operate within different “decision spaces” with both time and space dimensions, and delivery of information outside of such spaces do not do decision makers much good, they observed.

Understanding individual decision spaces and social networks is key to the “salience, reliability, and trust” of data (Cash and Buizer, 2005; Carolan, 2006; Silva and Tchamitchian, 2018; Jakku et al., 2019; Anderson et al., 2020; Rust et al., 2022). Farmers need information in a form and timeframe that fits their decision spaces as land managers, and when they are faced with risky decisions, hearing from trusted sources is important. Research on farmer attitudes about behavior change reveals the extent to which farmers themselves understand how their individual decisions are shaped by their social networks as well as cultural, policy, and economic contexts (Ranjan et al., 2020). Thus, our goals for the citizen science project included not only linking information from farmers to the state’s cover cropping databases but also learning about farmers’ decision-spaces—identifying key information networks and learning how to supply that data back to farmers in ways that support them in taking action, specifically supporting ongoing local innovation with cover crops.

4. Methods

With these goals in mind, we launched a hybrid citizen science project in 2020 to collect cover crop information supplied by farmers supplemented by project staff gathering biomass samples. We sought to learn about perceived barriers and how they were overcome, about trusted sources of information, and to identify potential avenues for supporting others interested in cover cropping. Our citizen science approach also involved a participatory element: gathering and using farmer feedback in survey design, supporting Extension staff in networking with farmers, as well as producing individualized reports, annual summaries, and opportunities for farmers to share results with other farmers. We developed a 35-question online survey via which farmer participants could share information. The survey questions were formatted to allow comparison to cover crop databases from Michael Fields Agricultural Institute and WICST. We included questions about timing, rotations, soil texture, cover crop species, manuring, and tillage. We also asked about seeding methods, rates, and costs as well as termination methods and timing. We included open-ended questions too, asking for example: “Please share any

other details regarding establishment, growth or management of cover crop species. Any interesting experiments, failures, equipment challenges?”

The survey collected background information including number of years’ experience with cover cropping, percent of farm in cover crops, and whether or not farmers were interested in expanding that amount. The survey included a number of qualitative questions aimed at building our understanding of the context of farmers’ decision making regarding cover crops. We relied on a Likert scale, asking farmers to select and rank as more and less important a list of sources of information on nutrient management and cover cropping, for example, on motivations for cover cropping, and potential positive influences. Potential influences we provided in our survey question included crop insurance breaks, additional information on equipment, cost reductions for the next cash crop (i.e., due to N credits or weed suppression; more time to experiment with cover crops; or support from additional county Extension personnel.) We also asked several open-ended questions; for example, if we had missed any significant motivations and their opinion of the survey itself as it attended to important considerations in cover cropping.

The comments sections generated rich data, which we supplemented with two extended interviews in 2022 with farmers we identified via farmer-led producer groups who were willing to share their cover crop experiences, ideas about how more farmers might begin using cover crops, and impressions of the survey. Our inductive content analysis for the qualitative data involved manually assigning labels, such as “cost,” or “grazing,” which we could quantify and out of which we identified key themes. Additional farmer interviews would be required to undertake narrative or discourse analysis but the two we pursued as well as a number of informal conversations were useful in iteratively verifying labels and identifying emergent themes. For example, from comments we were able to take the theme “cost” and identify subthemes related to time management, cost of seed, cost of equipment, and yield impact. We also cross-tabulated qualitative responses, for example, examining whether years of experience or location were correlated with more or less interest in expansion or need for information.

Participants also agreed to coordinate a November field visit with one of our collaborators from UW-Madison’s Nutrient and Pest Management program to collect a fall biomass sample from a chosen cover cropped field. We choose fall biomass to assess the cover crop growth of all cover crop species, including those that will not overwinter in Wisconsin, like oats, forage peas, and berseem clover. With limited time and monetary resources sampling in the spring is not yet an option. An in-person visit in the fall provided project staff an opportunity to visually assess the state of cover crops on different farms around the state, and in several cases to talk briefly with participants. We randomly sampled aboveground cover crop biomass from three 0.5-m² quadrats in each field. Within each quadrat, we used a gas-powered Stihl model 87 hedge trimmer to cut plants at the soil surface. Any weeds present were not separated from the samples. Samples were then dried at 49°C (120°F) for 2 weeks and weighed. We followed up with each farmer participant with their personal biomass estimate, along with a copy of an annual report sharing our general findings (Ingram et al., 2022).

Several strategies contributed a participatory element into our methods: (i) We asked respondents to identify relevant issues we missed and what new questions they might like to see, and then

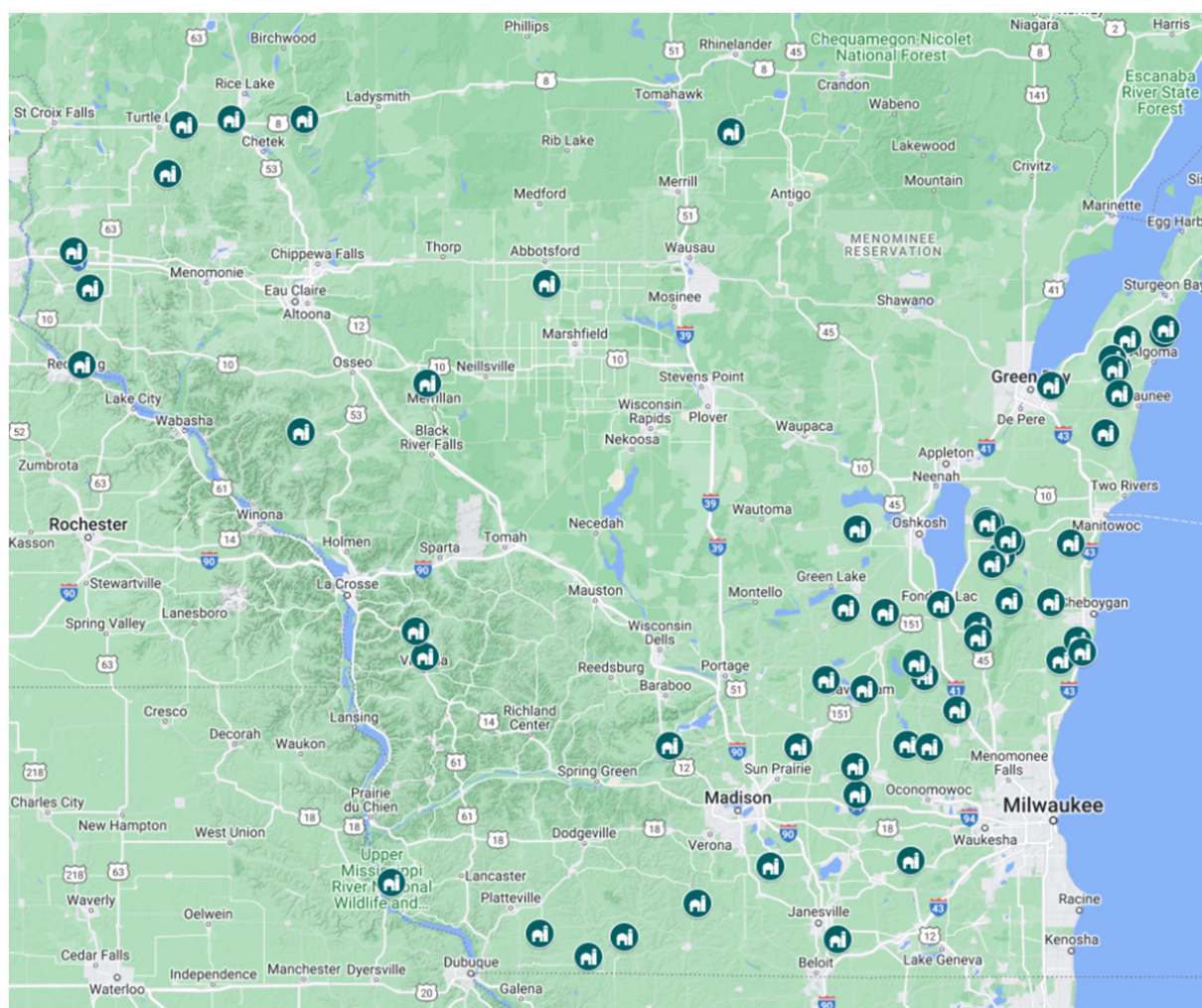


FIGURE 3

Locations of farmers participating in a cover crop citizen science effort in 2022. Farmer-produced data on cover crop practices are helping fill gaps in locally appropriate cover crop knowledge, especially needed in the eastern and northern parts of the state.

adjusting our survey accordingly for the following season. We also asked producers to review drafts of the survey. (ii) On the survey we asked farmers to identify their largest cover crop information gaps and concerns, which we then shared with Extension staff and other researchers via our annual report. We informally asked growers about the usefulness of the report. (iii) We supported communication about this project both by and between growers, aiming to build a self-awareness among participants of others engaged in a variety of cover cropping practices and experimentation. On the spectrum of community-based participation, this project falls between community “influencing” research design and “co-design” research questions and methods (Figure 2).

We also pursued participation via relationship-building with our citizen science participants, including project staff visits to farms and issuing individualized reports with participant’s biomass analysis results. An annual report written for farmers was shared widely via Extension networks. We also supported two participants in presenting about their cover crop experiences at a statewide cover crop conference and produced three webinars aimed at growers, agricultural educators, as well as other researchers.

Numbers of participants in the program are low compared to many conventional citizen science efforts, in part limited by project staff time to collect biomass samples but also by the complexity of our survey data and our goal of building the participatory element informed by that data. The number of participants has grown over time (more than doubling), and future goals for the project include augmenting with a spring biomass sample collected by participants themselves. In 2020, 15 farmers around the state participated in the survey, recruited via the state’s producer-led groups as well as extension and other agricultural educator networks. In 2021, 26 farmers located around Wisconsin joined, with 5 of them repeats from the previous year. The project launched a third season in 2022, with over 58 signups around the state, just under a quarter of them repeat participants (Figure 3).

5. Results

In response to farmer interest in a contextual presentation of biomass results, our annual reports include a table identifying county,

previous crop, cover crop species, planting method and resulting biomass (Table 1). We are currently working on producing an annual report from our 2022 data including generating an online map allowing farmers to see other participants in the same or nearby counties and providing information such as the crop previous to cover cropping in the field.

Farms surveyed in 2021 established cover crops following corn grain and corn silage, soybeans, and winter wheat. Of the 23 growers who reported species of cover crops used, 10 planted a cover crop mixture of 3 or more species. Mixtures tended to contain a grass, brassica, and legume with the most common species being crimson clover, red clover, oats, forage/field pea, and radish. Cereal rye, planted as a single species or with one other such as radish or oats, was planted as a cover crop on 8 of the 23 farms.

In terms of nutrient management and tillage, 16% (4) of responding farmers performed tillage and applied manure prior to establishing cover crops; 60% (15) of respondents used a drill to establish their cover crop, 4 farmers broadcast-seeded with no incorporation, 3 overseeded using aerial methods, and 1 used frost seeding (an option we added after receiving suggestions to do so the previous year). Manure was applied after cover crop planting on 32% (8) fields. Manure application rates ranged from 1.8 to 18 metric tons ha^{-1} of box manure (>20% DM) and 17,034 L ha^{-1} and 49,210 L ha^{-1} of liquid manure (4%–12% DM).

These data begin to provide needed information on local practices and experiments around cover cropping in more areas of Wisconsin. Farmer-provided data included information on cover crop species and contextualized with information about planting dates, nutrient management, and tillage, as well as challenges encountered.

Our 2021 survey respondents had a diverse range of years of experience with cover crops, ranging from 1–3 years to over 10 years. In 2021, cover crop acres planted by each farmer ranged from 10 acres to over 2,200 acres, representing from under 10% to 100% of all acres farmed. 80% of respondents said they'd like to expand the number of cover cropped fields, with 8 of 26 respondents already planting cover crops on at least 80% of all acres they farm. Three top incentives for cover cropping included reducing input costs for the next cash crop, for example, via nitrogen credits or weed suppression; cost sharing programs; and crop insurance breaks.

Most trusted sources of information for nutrient management were Agronomist or Certified Crop Advisor. For sources of knowledge about cover cropping, most respondents listed personal experience first, perhaps an indication of the demand to tailor cover cropping for any particular location, as well as the relative lack of locally sourced information and experience. Agronomists, UW Extension, and farmer-led networks were trusted sources of outside cover crop support (Figure 4). Interestingly, peers and other farmers were low on the list, another indication that experience with cover cropping remains low among many farmers, and that for many of our participants, farmer-to-farmer communication is happening via organized groups like the farmer-led networks. Most respondents selected or wrote in multiple sources.

Respondents selected from a list of “motivations” for cover cropping with most respondents selecting improving soil structure, organic matter, water quality, field trafficability, and weed suppression. If respondents said they were interested in expanding their cover-cropped acres, they were asked about “main barriers.” “Time” was listed by half of those growers as a main barrier, with several clarifying

that the season is too short following corn and soybeans, that it is “difficult to get covers in early enough,” and they have a “narrow planting window.” Other growers noted cost of seed as a barrier, as well as equipment challenges including irrigation to get covers established, too few planes available for aerial seeding, needing guidance technology (GPS) to plant corn into a green standing cover crop, and that a 15 foot no-till drill was too slow.

Our survey comment section, along with follow-up conversations with participating farmers illuminated how farmers were continuously experimenting with cover crops. For example, one survey respondent with 4–6 years of experience working with cover crops and interested in expanding his cover cropped acres, commented: “cold spring in 2022, rye took a very long time to begin growing. It wasn't until the first week of May that it even looked like any survived the winter. I let it grow an extra week while I planted other fields. The neighbor harvested the oats/rye forage in late fall. I plan on not doing that again.”

One of our interviewees, who has cover cropped for over a decade, described the challenges of his first attempt at cover cropping, “it was tillage radish, did it half-heartedly and nothing grew. I got back what I put in [with that experiment]. So next year I got out the seeder and was more successful.” Our second interviewed farmer's story offers another example of how cover cropping as an always ongoing learning process, as well as the importance of equipment: “First year I killed off all the wheat. So second year I had windrows of wheat super thick and nothing would grow. So then sprayed it, and we spread it or raked it up. We still are now testing a spreader on the back of our combine to do a better job. The rear of the combine is a big deal to get residue to spread evenly. If you have a thick mat behind it, affects the corn next year.”

Our results also provided information on social networks informing and shaping farmer's initial and ongoing decision-making about cover crops. For many, Wisconsin's state-supported producer-led watershed networks, which have tripled since 2016 from 14 to 43, were a valuable source of support (Figure 4), although one that does take time as our second interviewee emphasized: “For me it was the producer led network, absolutely. Taking the time to go to the meeting and talking with other farmers there. Especially after the event.”

He also described the importance of the supply chain in creating opportunities for farmers to learn from one another: “I learned from my seed dealer, but not directly. When Pioneer hosted a farmer thingy, one farmer at a breakout session there was spinning out rye on thousand acres. I thought if he could, I can.”

Our first interviewee, in describing the process by which he initially explored cover cropping, revealed the diversity of actors influencing his decisions: “I started taking control of my agronomic planning ... instead of hiring a consultant I taught myself on how to do it: I can read about it online, I can watch a video on YouTube, recordings of field days, and hear farmers speaking about what works and what does not. My county agronomist started sharing info on cover crops with me ... And my dad and my wife allow me to decide what to do. They've never said do not try something new.”

Along with a better understanding of the dynamic, ongoing nature of cover crop decision making we also gained more perspective on “time,” as a barrier. It can refer to the short growing season in the upper Midwest, or the constrained circumstances of a farmer in terms of taking risks to try new things. Our second interviewed farmer explained, “My son is too busy to go to those [producer-led] meetings. In my area I think of 4–5 young farmers all doing over

TABLE 1 Cover crop management and biomass production throughout Wisconsin during the 2021 growing season.

County	Previous crop	CC species	Planting		CC biomass			Precip (mm)	GDU ¹	CC termination
			Method	Date	Date	Metric tons DM/ac	Std err			
Grant	–	Annual ryegrass	Broadcast	18/9/2021	16/11/2021	1.2	0.0	142	898	Plant green
Green		Red clover	Frost seed	20/2/2021	16/11/2021	2.1	0.3	597	5,404	Plant green
Iowa		Multi-species mix	Drilled	24/8/2021	05/11/2021	2.7	0.2	208	1,501	Early, herbicide
Jefferson		–	Drilled	20/8/2021	03/11/2021	1.0	0.1	198	1,719	Plant green
Lafayette	Corn grain	Cereal rye, radish	Interseed	26/8/2021	16/11/2021	1.0	0.1	129	1,538	Early, crimp
Lafayette		Multi-species mix	Drill	1/8/2021	–	–	–	–	–	Plant green
Rock		Cereal rye	Interseed	13/9/2021	26/10/2021	0.6	0.0	155	963	Plant green
Trempealeau		Annual ryegrass	–	15/10/2021	01/12/2021	0.6	0.1	71	257	Plant green
Winnebago		Multi-species mix	Broadcast	17/9/2021	10/11/2021	1.3	0.4	46	922	Plant green
Jackson	Corn silage	Cereal rye	Drill	5/10/2021	09/11/2021	1.8	0.4	25	451	Graze
Manitowoc		Barley, winter wheat	Broadcast + Inc.	18/9/2021	10/11/2021	0.9	0.2	112	810	Plant green
Washington		Cereal rye, oats	Drill	19/8/2021	–	–	–	–	–	Winterkill
Winnebago		Cereal rye	Broadcast	10/9/2021	–	–	–	–	–	Plant green
Vernon	Forage sorghum	Multi-species mix	Interseed	10/9/2021	05/11/2021	0.8	0.1	66	939	Plant green
Green	Soybeans	Cereal rye	Drill	15/10/2021	–	–	–	–	–	Graze
Marathon		Multi-species mix	Broadcast	13/7/2021	09/11/2021	0.9	0.1	414	2,623	Plant green
Polk		Cereal rye	Drill	24/9/2021	9/11/2021	1.2	0.3	79	632	Plant green
Rock		Oats	Drill	30/9/2021	26/10/2021	0.7	0.1	145	531	Winterkill
St. Croix		Cereal rye	Drill	10/10/2021	10/11/2021	0.0	0.0	20.3	214	Plant green
St. Croix	Vegetables	Multi-species mix	drill	15/9/2021	09/11/2021	1.6	0.2	58	796	Early, crimp
Barron	Winter wheat	Multi-species mix	Drill	14/8/2021	09/11/2021	1.9	0.3	203	1,741	Plant green
Dodge		Multi-species mix	Drill	14/8/2021	09/11/2021	0.9	0.1	203	1,741	Plant green
Fond du Lac		Multi-species mix	Drill	17/8/2021	09/11/2021	0.6	0.0	224	1,590	Winterkill
Jefferson			Drill	26/7/2021	03/11/2021	2.5	0.1	307	2,513	Plant green
Pierce		Multi-species mix	Drill	17/8/2021	09/11/2021	1.4	0.1	224	1,590	Winterkill

¹Growing Degrees Units; base 4.4°C (40°F). Average biomass production was 1.36 metric tons (1.5 US tons) DM/ac.

1500 acres, so time for them to learn is valuable. I do not think an agronomist or a seed dealer or even the coop is going to persuade them cuz if it fails, they'll take the blame. So, they are very careful. I do not know about the incentive. But if you try and fail, it's hard."

In feedback on our reports and interviews, farmers emphasized the importance of narrative context, literally asking for "the story" accompanying the data. Given the range of variables in any cover crop approach, it is very difficult to compare data year to year. Farmers stated they need to understand biomass yield in the context of previous crop, for example, as well as tillage, fertility, and seeding method. As our first interviewed farmer explained: "For me the data means nothing without the story behind it. [In your reports] you are doing a pretty good job in terms of giving us county, precipitation, what crop preceded, what tillage, when it was planted, soil type, how did they feed it. Do not give me a bunch of numbers without the why behind them."

6. Discussion

Many of these results about the challenges of cover cropping will be familiar to both growers and researchers. This citizen science

approach is augmenting what is known with a more localized understanding of what cover crops are being experimented with, and building awareness among growers and others about ongoing practices and specific methods with which farmers are experimenting with cover crops. As described in the introduction, a number of incentive programs have championed cover crops to promote their adoption as a way to build soil and protect water quality. In addition, more Wisconsin farmers are aware of cover crops as a way to mitigate some of the challenges associated with climate change. The last two decades have been the warmest on record in Wisconsin, and the last decade has been the wettest (WICCI, 2021). Growers around the state are experiencing extreme rain events, groundwater flooding, declining snow cover, winter thaws, and more frequent extremely hot days and droughts. While we have not yet specifically surveyed growers about climate change as a motivation to use cover crops, we did observe that at cover crop conferences and in our survey comments section, farmers mentioned the benefits of cover crops to include earlier access to flooded fields in spring, for example, as well as protecting soil structure and preventing erosion and soil loss in heavy rains.

In response to farmer interest in contextualized knowledge, our annual report supplied back to participants included a table

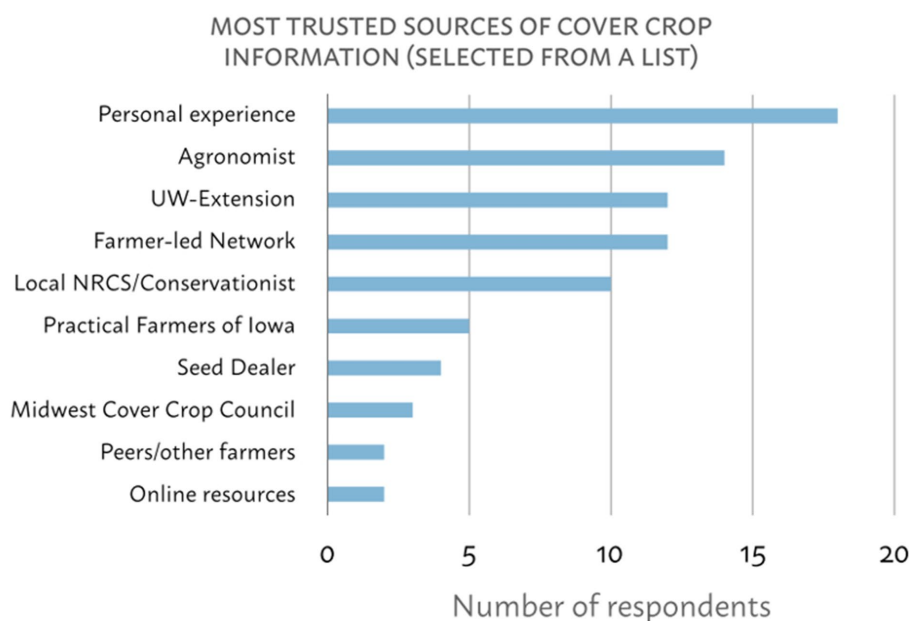


FIGURE 4

For trusted sources of information about cover cropping most 2021 respondents chose “personal experience,” with other sources including Agronomists, UW Extension, and farmer-led networks. Written in sources included books, OGRAIN, Michael Fields Agricultural Institute, Wisconsin Discovery Farms, and Iowa Learning Farms.

identifying county, previous crop, cover crop species, planting method and resulting biomass (Table 1). Future plans include developing an online interactive interface allowing participants to access map-based visualizations of the cover crop data they are helping generate. Data visualizations will be accompanied by videos and quotes from our farmer interviews as another contextual element. We plan on testing the interface with farmers, conservation agency staff, agronomists, and others to confirm the level of interest and to fine tune an accessible, effective sharing of information that participants will find useful and actionable within their decision spaces (Feldman and Ingram, 2009).

Other strategies in our participatory approach included using farmer feedback in improving our survey—adding questions related to termination, additional options such as frost seeding, and a question asking what it might take for growers to “stop using cover crops.” We also sought farmer input on technical challenges. We were able to provide resources for some of these challenges, and included links and a bibliography in our final reports. Other challenges require additional research, however, and we have shared these in presentations to research colleagues and amended the survey to inquire further; for example in more specific questions about equipment challenges. In response to interest from participants wanting to use cover crops as forage, we added a forage quality component to our most current sampling protocol.

Many of the comments from our participants were aimed toward other farmers. Seeing this as a network building opportunity, we compiled and shared comments in our annual reports and presentations so growers might see how their own interests and concerns were shared by others. As many have observed, farmers enjoy learning from others like them, and are more likely to trust the

information in contexts where they can observe how a farmer is putting new techniques into practice. We supported two of our participants in presenting their experiences at the Wisconsin Cover Crop Conference in 2022 and are gathering videos and additional narratives from participants.

Our results resonate with Thompson et al. (2021) and Pannell et al. (2006), who argue for conceiving of farmer engagement with conservation practices as a “continuous learning process.” The survey comments and interviews reveal how cover cropping involves an always ongoing set of activities: gathering information, experimenting, scaling up, or dis-adopting. Thus, while our citizen science project delivers annual “results” about cover cropping practice, equally important is the generation of awareness within the farmer research network of what kinds of experimentation and practices other farmers are engaged in, especially in similar locations and farm systems. These activities also point to the importance of continued policy, education, and networking efforts to provide a diversity of expertise and a continuity of support (Ranjan et al., 2020). Our outreach to researchers, educators, policy analysts, and sustainable farming advocates is motivated by our understanding of the need to build a diverse and reliable knowledge network supporting farmers in experimenting and engaging with cover cropping and other practices related to continuous living cover.

One clear limitation to generalizability of our findings is a self-selection bias towards farmers already interested in and using cover crops. Our results provide the most insight into the challenges of farmers who are already experimenting with cover crops in their crop rotations, or who are otherwise exploring options for more sustainable practices (although the shared challenges do suggest why some

farmers might stop using cover crops). Through our online survey and follow-up interviews and conversations, however, we did inquire about how farmers transitioned to new practices and what they saw as general barriers for other farmers.

7. Conclusion

Knowledge coproduction between researchers and farmers, as it generates needed agricultural information and supports on farm innovation is critical to supporting producers in developing more environmentally sustainable and resilient practices and surviving expanding uncertainties related to climate and markets. Bendfeldt and colleagues argue against an overemphasis on essentialist “best practices” and technocratic problem-solving in food systems research, stating, “The construction and expansion of farmer knowledge are not linear but rhizomatic and mycorrhizal in quality; therefore, scholar-practitioner responses to understanding and engaging with farmer knowledge systems should be amenable to a diversity of culturally dynamic systems of knowing that embody socio-eco relations and networks” (2021, p. 138).

We developed a hybrid citizen science project to respond to an information gap in locally suitable cover crop information, and also to learn more about farmer knowledge networks, and the specifics farmers’ decision spaces as they engage in cover cropping. Objectives included filling the knowledge gap with the participation of Wisconsin farmers, and then sharing that information in formats supportive of farmer action. Farmer-supplied data contributed to a more robust data set on cover cropping in Wisconsin, especially in the eastern and northern areas of the state. Farmers shared information on cover crop species selection, fertility methods, seeding methods, and tillage. Project staff visited farms to gather biomass samples in the fall. Qualitative questions in the citizen science survey sought information into challenges and perceived benefits of cover cropping in Wisconsin, as well as insight into how farmers might best consume new information on cover crops. We gathered specifics on the complexity of farmer decision making on cover cropping in the state and gained a better sense of ongoing experimentation and adjusting in response to weather and in the context of diverse growing systems. We built in participatory elements to our research effort including feedback on our own survey instrument and sharing data back to participants about their own results as well as the cover cropping practices around them. One goal is to create an awareness of an informal innovation network of Wisconsin farmers working with cover crops in diverse contexts. Results also emphasize the presence of a diversity of influential actors in the cover cropping decision space, including producer-led groups, seed and equipment dealers, as well as agricultural educators, advisors and family.

Agricultural citizen science has promise as a method for generating environmental information from dispersed sites and in an informational context that can support participants in taking action on that information. Specifically, this citizen science effort is providing much needed information about cover cropping as it is practiced in Wisconsin, along with information about how best to support ongoing farmer innovation in rapidly changing agricultural landscapes.

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

Ethics statement

The studies involving human participants were reviewed and approved by Institutional Review Board—University of Wisconsin–Madison. The patients/participants provided their written informed consent to participate in this study.

Author contributions

MI confirms being the sole author of this manuscript and has approved it for publication.

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Conflict of interest

The author declares that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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References

- Anderson, S., Colasanti, K., Didla, N., and Ogden, C. (2020). A call to build trust and center values in food systems work. Michigan State University Center for Regional Food Systems report. Available at: <http://foodsystems.msu.edu/resources/a-call-to-build-trust-and-center-values-in-foods-systems-work>
- Bendfeldt, E., McGonagle, M., and Niewolny, K. (2021). Rethinking farmer knowledge from soil to plate through narrative inquiry: an agroecological food systems perspective. *J. Agric. Food Syst. Community Dev.* 11, 137–151. doi: 10.5304/jafscd.2021.111.012
- Blesh, J., and Wolf, S. A. (2014). Transitions to agroecological farming systems in the Mississippi River basin: toward an integrated socioecological analysis. *Agric. Hum. Values* 31, 621–635. doi: 10.1007/s10460-014-9517-3
- Burchfield, E. K., Schumacher, B. L., Spangler, K., and Rissing, A. (2022). The state of US farm operator livelihoods. *Front. Sustain. Food Syst.* 5:795901. doi: 10.3389/fsufs.2021.795901
- Carolan, M. S. (2006). Social change and the adoption and adaptation of knowledge claims: whose truth do you trust in regard to sustainable agriculture? *Agric. Hum. Values* 23, 325–339. doi: 10.1007/s10460-006-9006-4
- Cash, D. W., and Buizer, J. (Eds.) (2005). *Knowledge-Action Systems for Seasonal to Interannual Climate Forecasting Summary of a Workshop*. National Academies Press, Washington, DC
- Cates, A. M., and Jackson, R. D. (2019). Cover crop effects on net ecosystem carbon balance in grain and silage maize. *Agron. J.* 111, 30–38. doi: 10.2134/agronj2018.01.0045
- Cates, A. M., Sanford, G. R., Good, L. W., and Jackson, R. D. (2018). What do we know about cover crop efficacy in the north Central United States? *J. Soil Water Conserv.* 73, 153A–157A. doi: 10.2489/jswc.73.6.153A
- Cooper, C. B., Hawin, C. L., Larson, L. R., Parrish, J. K., Bowser, G., Cavalier, D., et al. (2021). Inclusion in citizen science: the conundrum of rebranding. *Science* 372, 1386–1388. doi: 10.1126/science.abi6487
- Dunn, M., Ulrich-Schad, J. D., Prokopy, L. S., Myers, R. L., Watts, C. R., and Scanlon, K. (2016). Perceptions and use of cover crops among early adopters: findings from a national survey. *J. Soil Water Conserv.* 71, 29–40. doi: 10.2489/jswc.71.1.29
- Ebitu, L., Avery, H., Mourad, K. A., and Enyetu, J. (2021). Citizen science for sustainable agriculture – a systematic literature review. *Land Use Policy* 103:105326. doi: 10.1016/j.landusepol.2021.105326
- Feldman, D. L., and Ingram, H. M. (2009). Making science useful to decision makers: climate forecasts, water management, and knowledge networks. *Weather Clim. Soc.* 1, 9–21. doi: 10.1175/2009WCAS1007.1
- Gaventa, J. (2002). Exploring citizenship, participation and accountability. *IDS Bull.* 33, 1–14. doi: 10.1111/j.1759-5436.2002.tb00020.x
- Gould, K. A., Pellow, D. N., and Schnaiberg, A. (2004). Interrogating the treadmill of production: everything you wanted to know about the treadmill but were afraid to ask. *Organ. Environ.* 17, 296–316. doi: 10.1177/1086026604268747
- Harrison, J. H. (2011). Parsing ‘participation’ in action research: navigating the challenges of lay involvement in technically complex participatory science projects. *Soc. Nat. Resour.* 24, 702–716. doi: 10.1080/08941920903403115
- Hellerstein, D., Vilorio, D., and Ribaldo, M. (Eds.) (2019). *Agricultural Resources and Environmental Indicators, 2019. Economic Information Bulletin 288293*, United States Department of Agriculture. *Economic Research Service*. Available at: <https://ideas.repec.org/p/ags/uersib/288293.html>
- Ingram, M. (2007). Biology and beyond: the science of “Back to nature” farming in the United States. *Ann. Assoc. Am. Geogr.* 97, 298–312. doi: 10.1111/j.1467-8306.2007.00537.x
- Ingram, M., Smith, D., and Sanford, G. (2022). Building Knowledge about Wisconsin’s Cover Crops, a farmer citizen science research project. Report on the 2021 Season. UW-Madison, Center for Integrated Agricultural Systems. Available at: <https://cias.wisc.edu/wp-content/uploads/sites/194/2022/04/CCROP-Report-04.06.22-Final.pdf>
- Irwin, A. (2015). “Citizen science and scientific citizenship: same words, different meanings?” in *Science Communication Today – 2015*. eds. B. Schiele, J. L. Marec and P. Baranger (Nancy, France: Presses Universitaires de Nancy), 29–38. Available at: http://www.science-and-you.com/sites/science-and-you.com/files/users/documents/actes_sy_2015_complet.pdf#page=42
- Israel, B. A., Eng, E., Schulz, A. J., Parker, E. A., and Satcher, D. (Eds.) (2005). *Methods in Community-Based Participatory Research for Health*, 1st Jossey-Bass, San Francisco, CA.
- Jakku, E., Taylor, B., Fleming, A., Mason, C., Fielke, S., Sounness, C., et al. (2019). “If they don’t tell us what they do with it, why would we trust them?” Trust, transparency and benefit-sharing in smart farming. *NJAS – Wagen. J. Life Sci.* 90-91:100285. doi: 10.1016/j.njas.2018.11.002
- Kasperowski, D., Kullenberg, C., and Mäkitalo, Å. (2017). Embedding citizen science in research: forms of engagement, scientific output and values for science, policy and society. Preprint. *SocArXiv*, 27 February 2017. doi: 10.31235/osf.io/tfsgfh
- Kimura, A. H., and Kinchy, A. (2020). Citizen science in North American Agri-food systems: lessons learned. *Citiz. Sci. Theory Pract.* 5:4. doi: 10.5334/cstp.246
- Krome, M., and Ingram, M. (2020). CCROP Research Brief: “Identifying Needs of Wisconsin Cover Crop Information Providers.” Available at: https://www.covercropwi.org/_files/ugd/9ed610_97a9c83880664414ad00c941d7ccd3bc.pdf
- Lamine, C., and Dawson, J. (2018). The agroecology of food systems: reconnecting agriculture, food, and the environment. *Agroecol. Sustain. Food Syst.* 42, 629–636. doi: 10.1080/21683565.2018.1432517
- Matson, J., and VandenBrook, J. (2021). Toward a sustainable food system. *SSRN Electron. J.* doi: 10.2139/ssrn.3967155
- Mottet, A., Bicksler, A., Lucantoni, D., De Rosa, F., Scherf, B., Scopel, E., et al. (2020). Assessing transitions to sustainable agricultural and food systems: a tool for agroecology performance evaluation (TAPE). *Front. Sustain. Food Syst.* 4:579154. doi: 10.3389/fsufs.2020.579154
- Mourad, K. A., Hosseini, S. H., and Avery, H. (2020). The role of citizen science in sustainable agriculture. *Sustainability* 12:10375. doi: 10.3390/su122410375
- Myers, R. (2023). How conservation practices influence agricultural economic returns: implications for the farm finance community. AGree Research Paper, March. Available at: https://foodandagpolicy.org/wp-content/uploads/sites/17/2023/03/How_Conservation_Practices_Influence_Agricultural_Economic_Returns-1.pdf
- Myers, R., Weber, A., and Tellatin, S. (2019). Cover crop economics: opportunities to improve your bottom line in row crops (technical bulletin). Ag Innovations Series. SARE (Sustainable Agriculture Research and Education).
- Norström, A. V., Cvitanovic, C., Löf, M. F., West, S., Wyborn, C., Balvanera, P., et al. (2020). Principles for knowledge co-production in sustainability research. *Nat. Sustain.* 3, 182–190. doi: 10.1038/s41893-019-0448-2
- Ottinger, G. (2017). “Reconstructing or reproducing? Scientific authority and models of Change in two traditions of citizen science” in *The Routledge Handbook of the Political Economy of Science* eds. D. Tyfield, R. Lave, S. Randalls and C. Pauwles (Thorpe: Routledge). 351–364. doi: 10.4324/9781315685397
- Pannell, D. J., Marshall, G. R., Barr, N., Curtis, A., Vancley, F., and Wilkinson, R. (2006). Understanding and promoting adoption of conservation practices by rural landholders. *Aust. J. Exp. Agric.* 46:1407. doi: 10.1071/EA05037
- Petersen-Rockney, M., Baur, P., Guzman, A., Bender, S. F., Calo, A., Castillo, F., et al. (2021). Narrow and brittle or broad and nimble? Comparing adaptive capacity in simplifying and diversifying farming systems. *Front. Sustain. Food Syst.* 5:564900. doi: 10.3389/fsufs.2021.564900
- Prokopy, L. S., Gramig, B. M., Bower, A., Church, S. P., Ellison, B., Gassman, P. W., et al. (2020). The urgency of transforming the Midwestern U.S. landscape into more than corn and soybean. *Agric. Hum. Values* 37, 537–539. doi: 10.1007/s10460-020-10077-x
- Ramirez, M. J. G., Kling, C. L., and Arbuckle, J. G. (2015). Cost-share effectiveness in the adoption of cover crops in Iowa. Selected paper presented at the 2015 Agricultural and Applied Economics Association Annual Meeting, San Francisco, CA, 26–28 July. Available at: <https://ageconsearch.umn.edu/record/205876/files/GonzalezRamirezAAEA.Paper.pdf>
- Ranjan, P., Church, S. P., Arbuckle, J. G., Gramig, B. M., Reeling, C. J., and Prokopy, L. S. (2020). Conversations with non-choir farmers: implications for conservation adoption. Report for the Walton Family Foundation. Purdue University, West Lafayette. Available at: <https://core.ac.uk/download/pdf/343499013.pdf>
- Reimer, A. P., Weinkauff, K., and Prokopy, L. S. (2012). The influence of perceptions of practice characteristics: an examination of agricultural best management practice adoption in two Indiana watersheds. *J. Rural. Stud.* 28, 118–128. doi: 10.1016/j.jrurstud.2011.09.005
- Roesch-McNally, G. E., Arbuckle, J. G., and Tyndall, J. C. (2018). Barriers to implementing climate resilient agricultural strategies: the case of crop diversification in the U.S. Corn Belt. *Glob. Environ. Change* 48, 206–215. doi: 10.1016/j.gloenvcha.2017.12.002
- Roesch-McNally, G., Basche, A., Arbuckle, J., Tyndall, J., Miguez, F., Bowman, T., et al. (2018). The trouble with cover crops: Farmers’ experiences with overcoming barriers to adoption. *Renewable Agriculture and Food Systems*, 33, 322–333. doi: 10.1017/S1742170517000096
- Rust, N. A., Stankovics, P., Jarvis, R. M., Morris-Trainor, Z., de Vries, J. R., Ingram, J., et al. (2022). Have farmers had enough of experts? *Environ. Manag.* 69, 31–44. doi: 10.1007/s00267-021-01546-y
- Ryan, S. F., Adamson, N. L., Aktipis, A., Andersen, L. K., Austin, R., Barnes, L., et al. (2018). The role of citizen science in addressing grand challenges in food and agriculture research. *Proc. R. Soc. B Biol. Sci.* 285:20181977. doi: 10.1098/rspb.2018.1977
- Sanford, G. R., Jackson, R. D., Rui, Y., and Kucharik, C. J. (2022). Land use-land cover gradient demonstrates the importance of perennial grasslands with intact soils for building soil carbon in the fertile Mollisols of the north central US. *Geoderma* 418:115854. doi: 10.1016/j.geoderma.2022.115854
- Schneider, A., and Ingram, H. (1993). Social construction of target populations: implications for politics and policy. *Am. Polit. Sci. Rev.* 87, 334–347. doi: 10.2307/2939044
- Seifert, C. A., Azzari, G., and Lobell, D. B. (2018). Satellite detection of cover crops and their effects on crop yield in the Midwestern United States. *Environ. Res. Lett.* 13:064033. doi: 10.1088/1748-9326/aac4c8

- Silva, E. M., and Tchamitchian, M. (2018). Long-term systems experiments and long-term agricultural research sites: tools for overcoming the border problem in agroecological research and design. *Agroecol. Sustain. Food Syst.* 42, 620–628. doi: 10.1080/21683565.2018.1435434
- Strasser, B. J., Baudry, J., Mahr, D., Sanchez, G., and Tancoigne, E. (2018). “Citizen science”? Rethinking science and public participation. *Sci. Technol. Stud.* 32, 52–76. doi: 10.23987/sts.60425
- Thompson, N. M., Reeling, C. J., Fleckenstein, M. R., Prokopy, L. S., and Armstrong, S. D. (2021). Examining intensity of conservation practice adoption: evidence from cover crop use on U.S. Midwest farms. *Food Policy* 101:102054. doi: 10.1016/j.foodpol.2021.102054
- van de Gevel, J., van Etten, J., and Deterding, S. (2020). Citizen science breathes new life into participatory agricultural research. A review. *Agron. Sustain. Dev.* 40:35. doi: 10.1007/s13593-020-00636-1
- Vanni, F. (2014). *Agriculture and Public Goods*. Springer Netherlands, Dordrecht.
- Vincent-Caboud, L., Vereecke, L., Silva, E., and Peigné, J. (2019). Cover crop effectiveness varies in cover crop-based rotational tillage organic soybean systems depending on species and environment. *Agronomy* 9:319. doi: 10.3390/agronomy9060319
- Wallander, S., Smith, D., Bowman, M., and Claassen, R. (2021). Cover crop trends, programs, and practices in the United States, EIB 222, Washington, DC, Economic Information Bulletin (USDA, Economic Research Service, February 2021). Available at: <https://www.ers.usda.gov/webdocs/publications/100551/eib-222.pdf>.
- Wisconsin Initiative on Climate Change Impacts (WICCI) (2021). Wisconsin’s changing climate: impacts and solutions for a warmer climate. Assessment Report. UW-Madison’s Nelson Institute for Environmental Studies and the Wisconsin Department of Natural Resources. Available at: <https://wicci.wisc.edu/2021-assessment-report/full-report/>.
- Zhou, Q., Guan, K., Wang, S., Jiang, C., Huang, Y., Peng, B., et al. (2022). Recent rapid increase of cover crop adoption across the U.S. Midwest detected by fusing multi-source satellite data. *Geophys. Res. Lett.* 49:e2022GL100249. doi: 10.1029/2022GL100249



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EDITED BY

Jacob Jungers,
University of Minnesota Twin Cities,
United States

REVIEWED BY

Resham Thapa,
North Carolina State University, United States
Gabrielle Roesch-McNally,
American Farmland Trust, United States

*CORRESPONDENCE

Robert L. Myers
✉ myersrob@missouri.edu

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Farmer perspectives about cover crops by non-adopters

Robert L. Myers* and Kelly R. Wilson

Center for Regenerative Agriculture, University of Missouri, Columbia, MO, United States

The SARE/CTIC national farmer survey has assessed farmer experiences and perceptions about cover crops six times from early 2013 to early 2020. In most years, approximately 2,000 farmers responded to the survey questions, a majority of which were cover crop adopters, but a significant fraction (7% to 16%) were non-adopters: farmers not yet using cover crops. Survey reports previously focused on the experiences of cover crop adopters. In this paper, we synthesize responses of non-adopters to examine what factors influence or constrain adoption of cover crops. The non-adopters had similar farm sizes and land tenure situations compared to cover crop adopters, but were more likely to make use of conventional tillage and less likely to use continuous no-till compared to cover crop adopters. Non-adopters identified a number of concerns about cover crops, with the top concern being the time to plant and manage cover crops. Approximately 80% of non-adopters reported being open to considering cover crops. Factors cited to encourage non-adopters to adopt cover crops included incentive payments, tax breaks, crop insurance discounts, and soil carbon payments. Non-adopters wanted to gain a better understanding of how cover crops would benefit their particular farming operation and were interested to gain training through local cover crop workshops, local cover crop field demonstrations and one-on-one technical assistance. Non-adopters were particularly interested in how cover crops could boost soil organic matter and also wanted to know how cover crops could help with yields and reducing input costs.

KEYWORDS

cover crops, soil health, farmer adoption, conservation, tillage

1. Introduction

With heightened attention on climate change from both the public and private sectors, there is increased interest in understanding what strategies are effective to increase adoption levels of on-farm conservation practices. Conservation practices like cover crops are implemented by farmers to achieve a range of ecosystem services, helping to build soil health and mitigate the impacts of extreme weather events and shocks from climate change. There is a growing body of evidence of the on- and off-farm benefits of cover crops (Myers et al., 2019) and federal agencies, farm and conservation groups, in addition to major corporations are setting ambitious targets to increase adoption of cover crops on US farmland (Hamilton et al., 2017; Painter, 2020; Shroeder, 2021). However, adoption levels remain modest in many areas of the US and vary regionally (Wade et al., 2015; Hamilton et al., 2017).

There is a range of scholarship exploring what factors are associated with farmers' adoption of conservation strategies such as cover crops. Reported barriers to cover crop adoption include perceived lack of appropriate equipment/technology to manage cover crops, lack of perceived benefits, and time and labor constraints (Dunn et al., 2016; Roesch-McNally et al., 2018; Ranjan

et al., 2020; Thompson et al., 2021). Land tenure is often considered a key factor, with the assumption that farmers who rent land are less likely to adopt conservation practices than those who own the land they farm (Deaton et al., 2018). However, in their 2022 review of studies, Ranjan and colleagues find that quantitative studies on this topic are inconclusive and qualitative studies suggest a more complex picture (Ranjan et al., 2022). They find that renting land can be a barrier, but that other factors such as the stability of tenure, market dynamics, type of lease arrangements and timelines, producer relationships with landowners, and producer characteristics are also influential factors.

Besides land tenure, scholars report a complementary relationship between producers who use some type of conservation tillage and use of cover crops (Lee and McCann, 2019; Church et al., 2020; Thompson et al., 2021). In their review of conservation practices adoption studies, Propoky and colleagues find that factors most often positively associated with adoption are self-identifying as primarily motivated by land stewardship (or otherwise not primarily financially motivated), environmental attitudes, having a positive attitude toward the practice, having a propensity toward seeking and employing information, farming on vulnerable land, farm size, and higher levels of income and formal education (Prokopy et al., 2019). They also find that farmers who engage in marketing arrangement to maximize revenues or profits and those who expect that the practice will have a positive effect on yield are more likely to adopt conservation practices. Thomson and colleagues find a positive association with cover crop adoption and the perception/belief that cover crops reduce risk of nutrient loss to waterways (Thompson et al., 2021).

To better evaluate farmer experiences of cover crops and perceptions of farmers not yet using cover crops, a series of cover crop surveys were conducted by the Conservation Technology Information Center (CTIC) with financial support and input from the North Central Region Sustainable Agriculture Research and Education (SARE) program and the American Seed Trade Association (ASTA). Survey findings were reported each year through reports (CTIC and SARE, 2013, 2014, 2015, 2016, 2017, 2020). In this paper, we synthesize findings of these multi-year surveys to identify factors encouraging or constraining cover crop adoption.

2. Materials and methods

Surveys of farmers about cover crops were conducted in the U.S. in the winter or springs of the following years: 2013, 2014, 2015, 2016, 2017, and 2020. Since the surveys included questions about the previous crop year, the nomenclature for reporting on the surveys was reported as a two-year period, such as the 2019–2020 survey addressing crop results in 2019 but being done in the spring of 2020. With the exception of the first survey (2012–2013), all surveys were done through providing an online link to a Qualtrics or Survey Monkey survey through email distribution and farm media promotion. In most years, over 50,000 farmers were solicited for participation in the survey, with the largest number of farmers reached through distribution to farmer subscribers of Corn and Soybean Digest by Penton Media. Other sizable email lists of thousands of farmers and farm advisors were provided through CTIC and SARE. Press releases were used with the farm media to further promote the survey, and distribution also occurred through the

regional cover crop councils and other groups. Farmers who had filled out the survey in previous years were emailed the survey link and are believed to have constituted a majority of the farmers responding to the survey in subsequent years. The first year of the survey was done by a mixture of email surveys to a smaller list of farmers and handing out printed copies of the survey at five regional and national cover crop and no-till conferences where there were sizable numbers of farmers using cover crops.

It is important to note that the survey respondents do not represent a random sampling of the farming population or the cover crop using farmer population. Respondents self-selected whether to fill out the survey, but it is believed based on the respondent demographics that an effective cross-sample of farmers, particularly cover crop adopters, was obtained. In the first year of the survey, respondents were mostly from the Midwest. In subsequent years, there were respondents from across the lower 48 U.S. states, but the greatest numbers were from the Corn Belt states, reflecting the areas where the greatest number of farmers with cover crops are at (both Midwest and mid-Atlantic parts of the Corn Belt).

Survey questions were developed each year by a committee representing CTIC, SARE, ASTA staff and other experts on cover crops and survey methods. Each survey year, the survey questions and flow were reviewed for clarity. To maintain consistency, no major changes were made to individual questions, but we made minor changes where there was confusion over a question. As knowledge about cover crops and adoption increased over the years, questions were added to reflect the changing landscape and some were removed if they no longer seemed relevant. In years two to six of the survey, questions followed a branching tree pattern, such that non-adopters of cover crops answered only questions pertaining to the non-adopters, cover crop adopters with corn would answer questions about corn but not cotton, etc. Anyone who was not a farmer was thanked for opening the survey but instructed not to continue with the survey.

The number of farmer survey respondents varied by year, with 759 respondents for the 2012–2013 survey, approximately 2,000 respondents for each of the next four surveys (conducted in 2014–2017), and then 1,172 respondents for the 2019–2020 survey.¹ The smaller number that year was due to Penton Farm Media not participating in distribution of that survey due to discontinuation of the Corn and Soybean Digest publication. Not all farmers answered all questions, both based on the branching structure of the survey, and also their individual willingness to answer a particular question. Responses were generally not required to advance to the next question in the survey, but high percentages of survey respondents completed the questions relevant to their situation.

Questions specifically for non-adopters of cover crops dealt with the following topics: farm size, land tenure, tillage practices, concerns about cover crops, sources of information on cover crops, factors that might encourage adoption of cover crops, and other cover crop perceptions. Some questions were repeated for two or more years in the survey and others were only asked once.

Survey results were tabulated by CTIC staff and reports issued each year for a broad audience. Full survey reports are online at the

¹ For reference, the 2017 U.S. Census of Agriculture comprised 2,042,220 farms and reported a response rate of 71.8% (USDA, 2019).

SARE website at: <https://www.sare.org/publications/cover-crops/national-cover-crop-surveys/>.

3. Results

The percentage of survey respondents that were cover crop adopters vs. non-adopters varied each year, but generally about 7%–16% of the respondents were non-adopters (with a high of 272 non-adopters responding in 2014–2015). It is likely that the farmers not using cover crops who responded the survey were somewhat more interested in cover crops than the general farming population, but their responses still provide insights into factors keeping some farmers from adopting cover crops.

3.1. Farm size

In general, the distribution of farm sizes of farmers not using cover crops was very similar to farmers using cover crops and represented a reasonable cross section of crop farm sizes in the U.S. For example, in the 2019–2020 survey, of the farmers not using cover crops 13% farmed 2,000 or more acres, 16.9% farmed 1,000 to 1,999 acres, 20.8% farmed 500 to 999 acres, 14.3% farmed 180 to 499 acres, 6.5% farmed 50 to 179 acres, 10.4% farmed 10 to 49 acres, and 18.2% farmed 1 to 9 acres.

3.2. Land tenure

Land tenure was likewise fairly similar between cover crop adopters and non-adopters. Land tenure among non-adopters varied, but was relatively consistent in the 3 years it was surveyed (Figure 1). In the 2019–2020 survey, non-adopters of cover crops reported the following land tenure: 39% owned all the land they farmed, 10.4%

owned 76–99, 7.8% owned 51–75, 7.8% owned 26–50, 19.5% owned 1–25, and 15.6% owned none of the land they farmed. In relation to land tenure, in 2019–2020, non-adopters were asked about the following statement “It does not make sense for me to plant cover crops on ground I rent: 21% agreed or strongly agreed, but 28% disagreed or strongly disagreed with that statement and 51% were neutral.

3.3. Tillage

Tillage was a practice that differed between cover crop adopters and non-adopters, as shown in Figure 2. Cover crop adopters were more likely to make use of continuous no-till and non-adopters were more likely to be using conventional tillage.

3.4. Concerns about cover crops

One of the central questions about farmers not using cover crops is the concerns they have about cover crops, or what is holding them back from adoption (Figure 3). A question was asked about this topic in most of the surveys, and in every one, the top concern was the time and labor to plant and maintain the cover crop. In the 2019–2020 survey, 48% of the respondents listed that as their top concern. In most years, the second highest concern was usually related to cover crop economics, though the response choices varied some over the years on the economics question. Another common concern was using cover crops might increase production challenges and risks by adding weeds or making conditions harder to plant a cash crop in the spring. Farmers also felt unsure about their ability to effectively establish cover crops and to pick the right cover crop species.

While non-adopters reported a variety of concerns about cover crops, they also recognized potential benefits. In the 2014–2015 survey, non-adopters were asked what were the top three benefits they

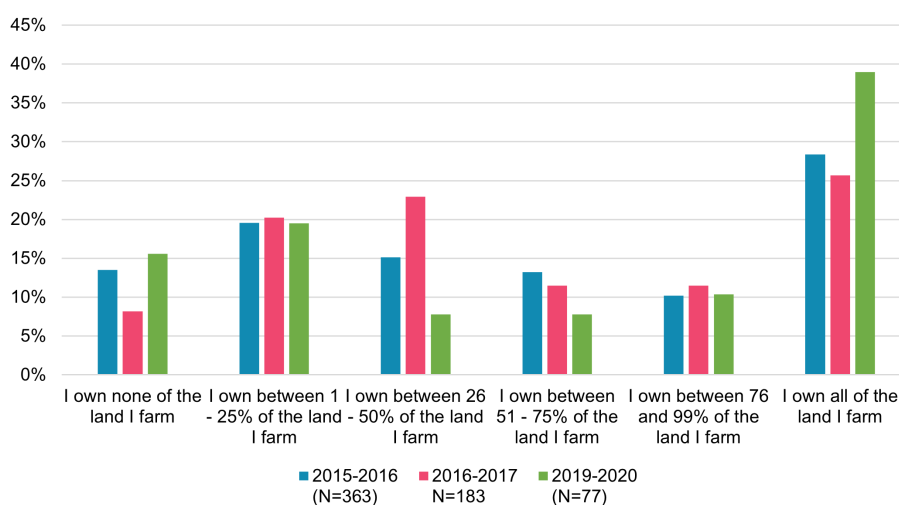


FIGURE 1

Land tenure of non-adopters in 2016, 2017, and 2020. According to the U.S. Census of Agriculture, approximately 34% of farms were rented or leased land in 2012 and approximately 32% were rented or leased land in 2017 in the U.S. (Vilsack and Clark, 2014; USDA, 2019).

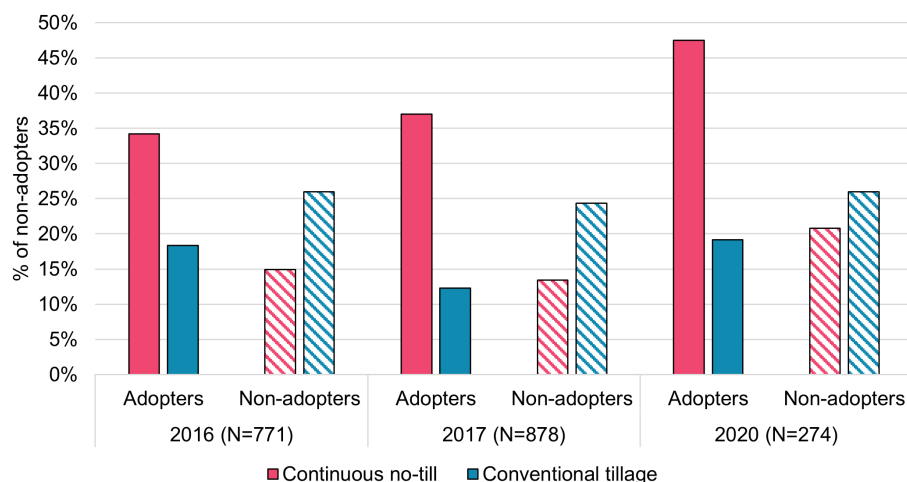


FIGURE 2

Tillage practices comparison between cover crop users and non-adopters in 2016, 2017, and 2020. According to the U.S. Census of Agriculture, 278,290 farms comprised cropland on which no-till practices were used compared to 405,692 farms that used intensive tillage in 2012. In 2017, 279,370 farms comprised cropland on which no-till practices were used compared to 264,893 farms that used intensive tillage.

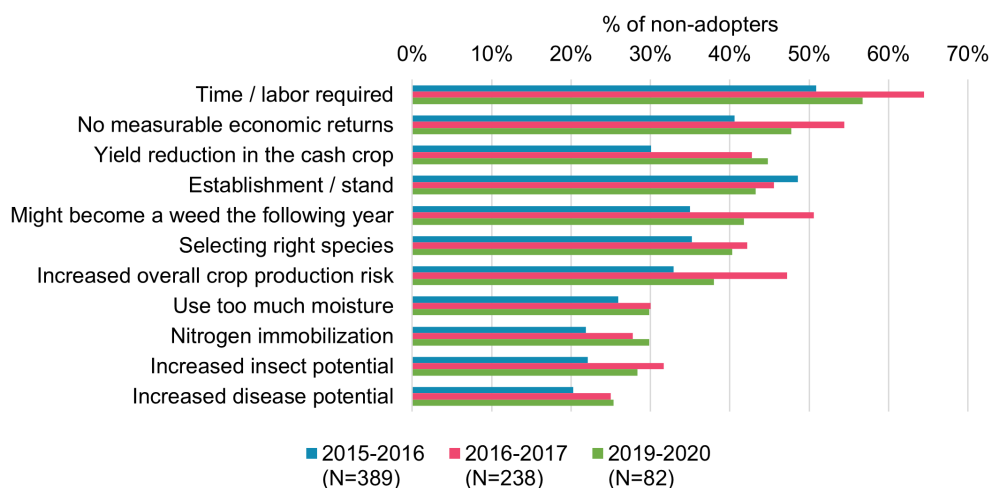


FIGURE 3

Non-adopters' reported "major concerns" about using cover crops. *N* represents total non-adopters responding to this question each year.

would look for from a cover crop. The top response was increased soil organic matter, named by 22%, followed by reduced soil erosion (18%), then a tie between increased yields in the following cover crops or reduced soil compaction, both at 11%. Smaller numbers were interested in cover crops for weed control (9%), nitrogen source (7%) and other factors.

3.5. Information sources

Participants were asked where they typically sought information about cover crops. Asked about information sources, in the 2019–2020 survey both cover crop adopters and non-adopters were asked to

check all that applied from among 12 categories of information sources. The highest number checked was "my own experience or trial and error" followed in decreasing importance by "other farmers," "ag media," "extension," "county natural resources conservation service," "SARE," "industry or retailer," "county farm service agency," and other options.

Related to information sources, non-adopters were asked about research priorities with cover crops in the 2014–2015 survey. The top response was "developing cover crops that fit my cash crop timing" followed closely by "developing cover crops that fit the climate in my area." Farmers were somewhat less interested in research on cover crops with improved ability to scavenge nitrogen, enhance cash crop disease resistance, or cover crops that fit common soil types.

3.6. Role of agriculture retailers

Farmers were also asked about the role of agriculture retailers (companies selling seed, chemicals, and/or fertilizers) in the 2013–2014 survey. Generally, responses to the ag retailer question were very similar between cover crop adopters and non-adopters on how ag retailers could be helpful with cover crop seed sales, cover crop planting, termination, and other services. The biggest contrast was whether ag retailers should encourage cover crop adoption: about 34% of cover crop adopters thought that should be a significant role for ag retailers, while only 23% of the non-adopters felt that ag retailers should be encouraging cover crop adoption. Cover crop adopters also felt more strongly that ag retailers should help with nutrient management plans to account for cover crops, while non-adopters thought cover crop termination advice and services were most important.

3.7. Motivations to use cover crops

Non-adopters were asked what would be “most helpful” to motivate them to use cover crops (Figure 4). The question was asked as a five-point Likert scale from not helpful to very helpful. In 2019–2020, the strongest positive response was to “cost-share or incentives to offset the cost of planting cover crops” with 54% responding that it would be moderately or very helpful. The next most favorable response was to “tax credits for planting cover crops,” with 70% ranking this approach as either very helpful or moderately helpful and 19% rating it as not helpful or somewhat helpful. Another type of financial incentive, payments for storing carbon, were also of interest with 63% ranking this approach as very helpful or moderately helpful to encourage them to use cover crops. The other type of financial inducement offered as a response was “discounted crop insurance

premiums,” with 61% saying it would be very helpful or moderately helpful to encourage adoption and 25% disagreeing.

Local demonstrations and advice were also viewed as beneficial to gaining encouragement to try cover crops; 65% said “that ‘local farm tours with cover crops so I can see how they work in my area’ would be very helpful or moderately helpful and 60% ranking ‘one-on-one technical assistance to select, plant or manage, cover crops’ as very helpful or moderately helpful.

Of least interest was having the ability to hire a local company or individual to do the cover crop seeding, with 34% interested in that option but 31% neutral and 35% not interested, despite concerns about the time it takes to plant and manage cover crops.

An earlier survey, in 2014–2015, found non-adopters responded that their willingness to use cover crops would be greatest if cover crops resulted in yield benefits for their primary cash crop. Tied for second were availability of cover crop incentive funds and availability of equipment for planting cover crops. They were least interested in having service providers or contractors to help plant cover crop seed, as was seen in 2019–2020.

3.8. Education opportunities

In 2013–2014, farmers were asked about the effectiveness of various educational opportunities (other years this question was not asked). Of the seven question response options given, non-adopters responded most favorably on two options, one being “local cover crop workshop where local experts and farmers who use cover crops present knowledge and share experiences” and the other being “trying things on my own and learning from successes and mistakes.” Both those responses were rated about 44% always effective and between 45% and 50% sometimes effective. Talking over the fence with a

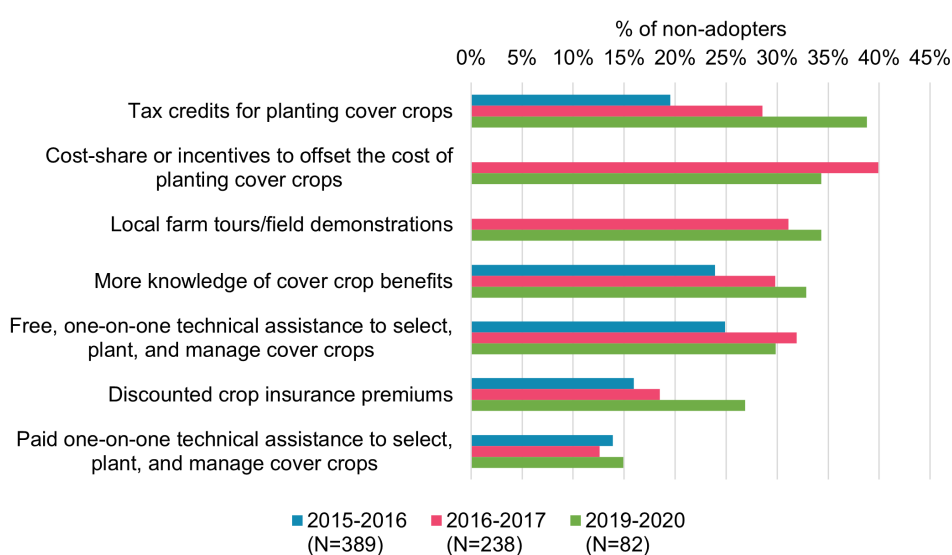


FIGURE 4

Non-adopters' ratings of what would be “most helpful” to motivate them to use cover crops. Non-adopters were asked to rank approaches that would best motivate them to use cover crops using a 5-point Likert scale that included (1) “not helpful,” (2) “somewhat helpful,” (3) “neither helpful nor detrimental,” (4) “moderately helpful,” and (5) “very helpful.” For this chart, we combined responses of “moderately helpful” and “very helpful” for each year. *N* represents total non-adopters responding to this question each year.

neighbor about his or her cover crops was viewed as much less effective.

3.9. Openness to using cover crops

In the two most recent surveys, non-adopters were asked if they had ever considered using cover crops on their farm. In 2016–2017, 82% said yes, in 2019–2020, 79% said yes. In 2016–2017, non-adopters were asked the degree to which they were interested in learning about how cover crops could benefit their farm: 38% strongly agreed, 36% agreed, 20% were neutral, and only 6% disagreed or strongly disagreed. As a follow-up question, they were asked: “If I better understood how cover crops would benefit my farm, I would be more likely to use them.” Sixty-nine percent agreed or strongly agreed with that statement, and only 7% disagreed or strongly disagreed. This was similar to a 2019–2020 finding, where 70% of non-adopters indicated they agreed or strongly agreed with the statement “I am interested in learning more about how cover crops can benefit my farm.”

In 2019–2020, non-adopters were also asked about their agreement on a series of other statements. In the strongest positive response, 75% of the non-adopters agreed or strongly agreed with the statement “If cover crops could help me reduce crop inputs (fertilizer, insecticide, herbicide, etc.) I would be more interested in using them on my farm.” Only 6% disagreed or strongly disagreed with that statement. Non-adopters had mixed opinions about the statement: “Concern about spread of herbicide-resistant weeds keeps me from using cover crops on my farm” with 32% agreeing or strongly agreeing, 40% neutral, and 28% disagreeing or strongly disagreeing with the statement.

4. Discussion

Synthesizing these SARE/CTIC multi-year farmer cover crop surveys offers important insight, particularly related to non-adopters perspectives. Perhaps most notable were the responses on land tenure, concerns about cover crops, and motivations for considering cover crops. While the non-adopters responding to the surveys were likely more interested in cover crops than non-adopter population as a whole, we found several common themes among our nonuser sample population.

Land tenure is frequently presented as a key factor impacting cover crop adoption. The assumption is that farmers are less likely to use these conservation practices on rented land, not knowing if they will retain access to any soil health benefits they might contribute to on a rented field. Along the same lines is the assumption that farmers who owned more of their land were more likely to use cover crops. However, these surveys showed there was little difference in land ownership percentage (or farm size) between cover crop adopters and non-adopters. When non-adopters were asked about their attitude on using cover crops for rented ground, as noted in results, only 21% agreed or strongly agreed that it “does not make sense for me to plant cover crops on ground I rent.” This result shows rental of land vs. ownership is likely not as big of a factor in cover crop use as some have assumed. As has been noted by other researchers, there is more complexity to the relationship between land tenure and adoption of conservation practices (Dunn et al., 2016; Deaton et al., 2018; Barnett et al., 2020; Ranjan et al., 2022). Considering that farmers increasingly

rent ground for a prolonged period of time, the notion that they do not care about stewarding their land is misconceived. The USDA Economic Research Service reported that “70 % of acres rented from operator landlords have been rented to the same tenant for over 3 years and 28% for over 10 years. Non-operator landlords tend to have even lengthier relationships with their tenants; 84 percent of acres have been rented to the same tenant for over 3 years and 41% for over 10 years” (Bigelow et al., 2016, p. 25).

Rather than rental arrangements, the number one concern that non-adopters have about cover crops is the time it takes to seed cover crops in the fall and to manage them. This finding was consistent in each of the four survey years, and was also noted by Lee and McCann (2019) in their research. This concern is perhaps understandable, as the fall time period when most cover crops are planted is one of the busiest on grain farms, with harvest operations often going up to the date of first frost or beyond, and many farmers wanting to do fall tillage and/or fertilizer applications after grain harvest. What is less apparent is why so many of the same non-adopters are reluctant to consider hiring someone to seed their cover crops, with low levels of interest in contracted cover crop seeding in the two survey years that option was asked about. It may be that many farmers do not want to feel dependent on someone else to do planting for them. However, to get past the hurdle of having more non-adopters resistant to cover crops, it will likely be necessary for more of them to start taking advantage of external cover crop seeding services, whether from aerial applications, fertilizer dealers, neighboring farmers or others who can do cover crop seeding.

Non-adopters consistently reported that cost-share or incentive payments for cover crop seeding would be the top positive inducement to start using cover crops. This aligns with the yearly expansion that federal and state agencies have made to their cover crop incentive programs in addition to the consistent demand for these funds, which usually outstrips supply. Further government investment in cover crop incentives will likely continue to help expand acreage of cover crops, based on survey responses.

Other financial inducements were also of interest, including tax credits, crop insurance discounts, and soil carbon payments. Tax credits have been discussed but not implemented in any large-scale fashion in the U.S. A pilot program on tax credits in one or more U.S. regions help to further assess the potential to drive adoption. Such tax credits could be property tax credits, which may motivate both owner-operators as well as non-operator landowners. Another approach could target income tax, either specifically for farm operators or split between operators and nonoperating landowners.

Cover crop discounts on crop insurance premiums were first offered in the state of Iowa in 2017 at \$5 per acre, and proved to be very popular, with demand for the program steadily growing (Iowa Department of Agriculture and Land Stewardship, 2021). Illinois is now also offering a similar crop insurance premium discount for planting cover crops. Nationally, USDA has provided a \$5 per acre benefit to farmers using cover crops who have crop insurance, but this incentive payment is distributed after cover crop use rather than before, as is the case in Iowa and Illinois. While the \$5 per acre payment is small compared to cover crop payments offered through the USDA Environmental Quality Incentive Program of \$40–50 per acre or more, it is a substantial percent discount on a crop insurance premium that might be in the \$15 per acre range.

Soil carbon payments are a more recent opportunity being offered to farmers, primarily through major food and agriculture companies

(Wongpiyabovorn et al., 2021; Oldfield et al., 2022). In the 2019–2020 survey, cover crop non-adopters indicated strong interest in soil carbon payments. However, challenge has been the widely divergent approaches taken among companies, leading to confusion among farmers about the options (Wongpiyabovorn et al., 2022). Some companies have also continued to modify their soil carbon payment programs, adding to further confusion. Going forward, improving clarity about these soil carbon payment options will likely lead to more use of them as an inducement for cover cropping by current non-adopters.

Providing learning opportunities and farmer-to-farmer networking through field days and workshops continues to be an important approach for non-adopters, based on their desire to learn from other farmers. Other studies have documented the value of these approaches, including a study on conservation field days and demonstrations in Indiana (Singh et al., 2018).

5. Conclusion

This survey analysis underscores that there is not one single approach that will dramatically increase cover crop adoption among current non-adopters. Some will be motivated by expanded incentive payments while others who are averse to government programs may prefer private sector payments. Further education and outreach efforts on cover crops will be important to help non-adopters better understand how cover crops can benefit their own personal situation. Continued use of local field days and workshops and direct engagement with producers on specific ways cover crops can work for them will be needed in combination with financial incentives to greatly expand the amount of cover crop acreage in the U.S.

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

References

- Barnett, M. J., Spangler, K., Petzelka, P., and Filipiak, J. (2020). Power dynamics of the non-operating landowner-renter relationship and conservation decision-making in the midwestern United States. *J. Rural. Stud.* 78, 107–114. doi: 10.1016/j.jrurstud.2020.06.026
- Bigelow, D., Borchers, A., and Hubbs, T. (2016). U.S. farmland ownership, tenure, and transfer p. 53. U.S. Department of Agriculture, Economic Research Service. Available at: <https://www.ers.usda.gov/topics/farm-economy/land-use-land-value-tenure/farmland-ownership-and-tenure/>
- Church, S. P., Lu, J., Ranjan, P., Reimer, A. P., and Prokopy, L. S. (2020). The role of systems thinking in cover crop adoption: implications for conservation communication. *Land Use Policy* 94:104508. doi: 10.1016/j.landusepol.2020.104508
- CTIC and SARE. (2013). 2012–2013 cover crop survey. Conservation technology information center (CTIC), Sustainable Agriculture Research & Education (SARE). Available at: https://www.ctic.org/data/Cover_Crops_Research_and_Demonstration_Cover_Crop_Survey#:~:text=2019%2D2020%20Cover%20Crop%20Survey&text=Among%20farmers%20who%20planted%20green,and%202.625%20in%20spring%20wheat
- CTIC and SARE. (2014). 2013–2014 cover crop survey report. Conservation technology information center (CTIC), Sustainable Agriculture Research & Education (SARE). Available at: https://www.ctic.org/data/Cover_Crops_Research_and_Demonstration_Cover_Crop_Survey#:~:text=2019%2D2020%20Cover%20Crop%20Survey&text=Among%20farmers%20who%20planted%20green,and%202.625%20in%20spring%20wheat
- CTIC and SARE. (2015). 2014–2015 cover crop survey report. Conservation technology information center (CTIC), Sustainable Agriculture Research & Education (SARE). Available at: https://www.ctic.org/data/Cover_Crops_Research_and_Demonstration_Cover_Crop_Survey#:~:text=2019%2D2020%20Cover%20Crop%20Survey&text=Among%20farmers%20who%20planted%20green,and%202.625%20in%20spring%20wheat
- CTIC and SARE. (2016). 2015–2016 cover crop survey report. Conservation technology information center (CTIC), Sustainable Agriculture Research & Education (SARE). Available at: https://www.ctic.org/data/Cover_Crops_Research_and_Demonstration_Cover_Crop_Survey#:~:text=2019%2D2020%20Cover%20Crop%20Survey&text=Among%20farmers%20who%20planted%20green,and%202.625%20in%20spring%20wheat
- CTIC and SARE. (2017). 2016–2017 cover crop survey report. Conservation technology information center (CTIC), Sustainable Agriculture Research & Education (SARE). Available at: https://www.ctic.org/data/Cover_Crops_Research_and_Demonstration_Cover_Crop_Survey#:~:text=2019%2D2020%20Cover%20Crop%20Survey&text=Among%20farmers%20who%20planted%20green,and%202.625%20in%20spring%20wheat
- CTIC and SARE. (2020). 2019–2020 cover crop survey report. Conservation technology information center (CTIC), Sustainable Agriculture Research & Education (SARE). Available at: https://www.ctic.org/data/Cover_Crops_Research_and_Demonstration_Cover_Crop_Survey#:~:text=2019%2D2020%20Cover%20Crop%20Survey&text=Among%20farmers%20who%20planted%20green,and%202.625%20in%20spring%20wheat

Ethics statement

Ethical review and approval was not required for the study on human participants in accordance with the local legislation and institutional requirements. The patients/participants provided their written informed consent to participate in this study.

Author contributions

All authors listed have made a substantial, direct, and intellectual contribution to the work and approved it for publication.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Survey&text=Among%20farmers%20who%20planted%20green,and%202.625%20in%20spring%20wheat

Deaton, B. J., Lawley, C., and Nadella, K. (2018). Renters, landlords, and farmland stewardship. *Agric. Econ.* 49, 521–531.

Dunn, M., Ulrich-Schad, J. D., Prokopy, L. S., Myers, R. L., Watts, C. R., and Scanlon, K. (2016). Perceptions and use of cover crops among early adopters: Findings from a national survey. *J. Soil Water Conserv.* 71, 29–40.

Hamilton, A. V., Mortensen, D. A., and Allen, M. K. (2017). The state of the cover crop nation and how to set realistic future goals for the popular conservation practice. *J. Soil Water Conserv.* 72, 111A–115A. doi: 10.2489/jswc.72.5.111A

Iowa Department of Agriculture and Land Stewardship (2021). *Iowa Department of Agriculture and Land Stewardship's Cover Crop Insurance Discount Program Continues this Fall*. Available at: <https://iowaagriculture.gov/news/idals-offering-cover-crop-insurance-discount-fall-2021>

Lee, S., and McCann, L. (2019). Adoption of cover crops by US soybean producers. *J. Agric. Appl. Econ.* 51, 527–544. doi: 10.1017/aae.2019.20

Myers, R. L., Weber, A., and Tellatin, S. (2019). Cover Crop Economics; Opportunities to Improve Your Bottom Line in Row Crops. University of Missouri and North Central SARE, Columbia, Missouri: Sustainable Agriculture Research & Education (SARE). Available at: <https://www.sare.org/resources/cover-crop-economics/>

Oldfield, E. E., Eagle, A. J., Rubin, R. L., Rudek, J., Sanderman, J., and Gordon, D. R. (2022). Crediting agricultural soil carbon sequestration. *Science* 375, 1222–1225. doi: 10.1126/science.abl7991

Painter, K. L. (2020). Cargill joins regenerative agriculture movement, sets goal for 10 million acres. *StarTribune*. Available at: <https://www.startribune.com/cargill-joins-regenerative-agriculture-movement-sets-goal-for-10-million-acres/572432302/>

Prokopy, L. S., Floress, K., Arbuckle, J. G., Church, S. P., Eanes, F. R., Gao, Y., et al. (2019). Adoption of agricultural conservation practices in the United States: evidence from 35 years of quantitative literature. *J. Soil Water Conserv.* 74, 520–534. doi: 10.2489/jswc.74.5.520

Ranjan, P., Arbuckle, J. G., Church, S. P., Eanes, F. R., Floress, K., Gao, Y., et al. (2022). Understanding the relationship between land tenure and conservation behavior:

recommendations for social science research. *Land Use Policy* 120:106161. doi: 10.1016/j.landusepol.2022.106161

Ranjan, P., Church, S. P., Arbuckle, J. G., Gramig, B. M., Reeling, C. J., and Prokopy, L. S. (2020). Conversations with non-choir farmers: Implications for conservation adoption. Report for the Walton Family Foundation.

Roesch-McNally, G. E., Basche, A. D., Arbuckle, J. G., Tyndall, J. C., Miguez, F. E., Bowman, T., et al. (2018). The trouble with cover crops: farmers' experiences with overcoming barriers to adoption. *Renewable Agric Food Syst* 33, 322–333. doi: 10.1017/S1742170517000096

Shroeder, E. (2021). General Mills advances regenerative ag practices. *World-Grain. Com*. Available at: <https://www.world-grain.com/articles/15188-general-mills-advances-regenerative-ag-practices>

Singh, A., MacGowan, B., O'Donnell, M., Overstreet, B., Ulrich-Schad, J., Dunn, M., et al. (2018). The influence of demonstration sites and field days on adoption of conservation practices. *Journal of Soil and Water Conservation*, 73, 276–283.

Thompson, N. M., Reeling, C. J., Fleckenstein, M. R., Prokopy, L. S., and Armstrong, S. D. (2021). Examining intensity of conservation practice adoption: evidence from cover crop use on US Midwest farms. *Food Policy* 101:102054. doi: 10.1016/j.foodpol.2021.102054

USDA (2019). *2017 census of agriculture United States summary and state data Volume 1, Part 51*. Washington, DC: USDA, National Agricultural Statistics Service.

Vilsack, T., and Clark, C. (2014). *2012 census of agriculture*. National Agricultural Statistics Service, US Department of Agriculture.

Wade, T., Claassen, R., and Wallander, S. (2015). Conservation-Practice Adoption Rates Vary Widely by Crop and Region. EIB-147. Washington, DC: U.S. Department of Agriculture, Economic Research Service.

Wongpiyabovorn, O., Plastina, A., and Crespi, J. M. (2022). Challenges to voluntary ag carbon markets. *Appl. Econ. Perspect. Policy*

Wongpiyabovorn, O., Plastina, A., and Lence, S. H. (2021). *Futures Market for Ag Carbon Offsets under Mandatory and Voluntary Emission Targets*. Ames, Iowa: Center for Agricultural and Rural Development (CARD) Publications at Iowa State University. Available at: <https://ideas.repec.org/p/ias/cpaper/apr-fall-2021-4.html>



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EDITED BY

Siyabusa Mkuhlani,
International Institute of Tropical Agriculture
(IITA), Kenya

REVIEWED BY

Hannah Waterhouse,
University of California, Berkeley, United States
Carol Williams,
University of Wisconsin-Madison, United States

*CORRESPONDENCE

David J. Mulla
✉ mulla003@umn.edu

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Comparative simulation of crop productivity, soil moisture and nitrate-N leaching losses for intermediate wheatgrass and maize in Minnesota using the DSSAT model

David J. Mulla^{1*}, Muhammad Tahir¹ and Jacob M. Jungers²

¹Department of Soil, Water & Climate, University of Minnesota Twin Cities, St. Paul, MN, United States,

²Department of Agronomy & Plant Genetics, University of Minnesota Twin Cities, St. Paul, MN, United States

Perennial grain crops are a potential alternative source of staple foods and animal forage that can also provide additional environmental benefits over annual crops. Intermediate wheatgrass (IWG; *Thinopyrum intermedium*) is a new perennial dual-use crop for grain and forage, with growing interest among stakeholders as it produces grain in a more environmentally sound manner than current annual crops. DSSAT model simulations were performed for maize and a new DSSAT model for IWG based on data collected from field studies conducted during 2013–2015 at three different locations, i.e., Lamberton, Waseca and Crookston using low (zero), medium (60–80 kg ha⁻¹) and high fertilizer nitrogen (N) rates (120–160 kg ha⁻¹). The DSSAT CERES-Maize and CROPGRO-PFM models used as the basis for simulating IWG were calibrated at the high N rate to predict the yield/biomass, soil water balance, and soil nitrogen balance in maize and IWG, respectively, for the medium and low N rate treatments. Model predictions for maize yield and IWG biomass (0.89 >= Nash Sutcliffe Efficiency >= 0.58), soil profile moisture (0.81 >=NSE>=0.53) ranged from very good to satisfactory for maize and the high N rate in IWG, with nearly satisfactory accuracy for IWG under the medium and zero N rates. Simulation results indicate that low, medium and high N rates produced an average IWG biomass of 7.8, 9.7, and 10.5 t ha⁻¹, in addition to observed grain yield of 0.36, 0.49, and 0.45 t ha⁻¹, respectively. The corresponding N rates produced 5.9, 7.9, and 8.7 t ha⁻¹ maize yield. Soil profile moisture under IWG and maize averaged 0.25 and 0.29 m³m⁻³, respectively. Averaged over N rates and locations, IWG and maize had values for crop evapotranspiration (ET_c) of 592 vs. 517 mm; deep percolation of 100.8 vs. 154.5 mm; and nitrate-N leaching losses of 2.6 vs. 17.9 kg ha⁻¹, respectively. Results indicate that perennial IWG not only produced high biomass under rainfed conditions, but also reduced deep percolation by efficiently using soil profile moisture, leading to nitrate-N leaching losses six to seven times lower than for maize.

KEYWORDS

Kernza®, corn, modeling, evapotranspiration, deep percolation, nitrate leaching, Minnesota

Introduction

Modern societies now demand more from food systems—not only food, fuel, and fiber, but also a variety of ecosystem services. And although today's farming practices are producing unprecedented yields, they are also contributing to ecosystem problems such as physical and chemical soil degradation, greenhouse gas emissions, and water pollution. Nitrogen leaching from fertilized annual grain crops to groundwater is a health threat for citizens of rural communities relying on well water for drinking. The Midwestern United States is among the most highly productive and intensively farmed maize (corn) production areas in the world, with associated nitrate-N ($\text{NO}_3\text{-N}$) pollution of groundwater in Minnesota (MN). High nitrogen (N) fertilizer inputs on high organic matter soils in the rainy season result in high leaching and tile drainage losses of $\text{NO}_3\text{-N}$, leading to groundwater pollution, surface water eutrophication and algal blooms in downstream coastal areas (Kroening and Vaughan, 2019; Christopher et al., 2021). Nitrate concentrations in 27% of 728 river and stream sampling sites of Minnesota exceeded 10 ppm during the years 2000–2010 (Minnesota Pollution Control Agency, 2013). Similarly, 40% of groundwater wells in central Minnesota during recent years had $\text{NO}_3\text{-N}$ concentrations exceeding the permissible limit of 10 ppm. Moreover, groundwater contamination trends from the last decade have remained level (Kroening and Vaughan, 2019). A study conducted for a continuous corn rotation on sandy soil in Central Minnesota during 2011–2014 showed very high in-season $\text{NO}_3\text{-N}$ concentrations averaging 30.3 and 38.0 mg L^{-1} , and end-of-season leaching losses of 71.0 and 96 kg N ha^{-1} below the rooting zone, corresponding with urea N fertilizer applied at the economic optimum N rate (EONR) of 180 kg ha^{-1} or the agronomic optimum N rate (AONR) of 225 kg ha^{-1} , respectively (Struffert et al., 2016). In short, the practices accompanying annual row crop agriculture bring environmental problems, which argues for diversification of Midwestern agriculture to provide improved ecosystem services (Pennington et al., 2017; Prokopy et al., 2020).

There is a pressing need to reduce $\text{NO}_3\text{-N}$ losses through improved N management and alternative perennial cropping systems. Reducing N fertilization at EONR under continuous corn rotation in Midwestern row crop agriculture causes high $\text{NO}_3\text{-N}$ leaching losses, while further reducing N rates to bring sustainability within agroecosystems causes an economic loss in terms of reduced grain yields (Struffert et al., 2016). Replacing significant acreage of Midwestern row crops with perennial grain crops is an approach that has received far less attention than fertilizer management strategies for improving ecosystem services. IWG, a perennial grass trademarked by The Land Institute as Kernza®, has multiple environmental and economic benefits (Culman et al., 2013; Jungers et al., 2019). Kernza® not only produces animal forage in spring and fall, but also benefits farmers with high-value human-edible grain yield. Additional benefits of Kernza® include providing continuous living cover on the landscape and significantly higher below-ground biomass beyond those provided by annual cropping systems (Pinto et al., 2021). Perennial crops like Kernza® have an extended growing season, resulting in higher evapotranspiration, and lower runoff and deep percolation losses during late fall and early spring, compared to the fields that are fallow under annual crops. Extended growing time also increases the

assimilation of N during the time when it is susceptible to leaching (Huggins et al., 2001). Moreover, extensive rooting systems of Kernza® also result in lower $\text{NO}_3\text{-N}$ and water losses via deep percolation and surface runoff, and higher carbon sequestration, compared to the annual crops. Kernza® grown consecutively over three seasons needs only one pass of the tractor to initially plant seed, leading to reduced input expenses and reduced greenhouse gas emissions (Jungers et al., 2019; Lanker et al., 2020; Reilly et al., 2022).

IWG is a perennial cool-season forage grass and grain crop that is widely adapted throughout the USA and Canada (de Oliveira et al., 2020). Although IWG was initially implemented as a forage crop in the upper Midwest and northern Great Plains due to its winter hardiness and high forage quality, efforts were made to develop IWG into a grain crop by selecting for increased grain size and yield over the last decade (DeHaan et al., 2013). IWG can sustain high yields without replanting for numerous consecutive years, resulting in important climate mitigation benefits (de Oliveira et al., 2018). Reduced nitrate leaching compared with annual wheat (Culman et al., 2013) and maize (Jungers et al., 2019) has been observed with IWG due to its greater whole-crop N use efficiency (Sprunger et al., 2018) and water use efficiency (de Oliveira et al., 2020).

While efforts are being made to improve grain yield, seed size, threshability, shattering resistance, lodging resistance, and develop higher yielding cultivars for farmers (Zhang et al., 2017), much remains unknown about agronomic management, optimum N requirement, ET_c potential, and impacts of IWG on deep percolation and $\text{NO}_3\text{-N}$ leaching losses, compared to row crops. The short-term monitoring of yield and $\text{NO}_3\text{-N}$ leaching losses over small experimental areas under highly variable climatic conditions can produce incomplete assessments, while long-term monitoring needs laborious work to collect plant parameters and soil characteristics. Under these circumstances, combining experimental measurements with modeling is a highly useful approach for understanding the relationships among soil, plants, climate change, water quality and other components in agricultural systems, particularly for studying the effects of crop diversification over time (Prokopy et al., 2020). Crop models such as Decision Support System for Agrotechnology Transfer (DSSAT) are not only useful to understand management effects on overall production, but also environmental effects of perennial crops and their management options. These models work well when calibrated with site specific data (Zamora et al., 2009).

The DSSAT model has a modular structure consisting of cropping system, weather, soil, and crop management modules. DSSAT has been tested for various climatic conditions and crop varieties. The CERES-Maize and CROPGRO-PFM (perennial forage model) modules included in DSSAT version 4.7 have the ability to predict yield and biomass production of corn and perennial grasses, respectively. However, forage crop models are still under the development stage (Jones et al., 2003; Hoogenboom et al., 2019), and have not yet been applied for simulation of IWG.

The objectives of this study were to: (1) evaluate the accuracy of using the DSSAT-CERES-Maize and DSSAT-CROPGRO-PFM model to simulate yield and/or biomass, soil water, and soil N balance of IWG vs. corn at three locations in Minnesota; and (2) evaluate how N rates across a range of climatic conditions and soil types affect crop yield/biomass productivity, soil profile moisture, deep percolation and nitrate-N leaching losses for IWG vs. maize in Minnesota.

Materials and methods

Soil and weather data description

Experimental data for DSSAT model simulations were collected at three University of Minnesota Agricultural Experiment Station sites located in three different agroecological regions of Minnesota, United States. Soil series studied include a Knoke silty clay loam (Calciaquoll) at Lamberton (44.24, −95.30), a Webster clay loam (Haplaquoll) at Waseca (44.07, −93.53), and a Wheatville loam (Calciaquoll) at Crookston (47.81, −96.62). The Lamberton, Waseca and Crookston sites are situated in southwestern, southern, and northwestern Minnesota, in the Coteau/Drier Blue Earth Till, Rolling Moraine, and Northern Till/Inter-beach Sand Bar agroecoregions, respectively. The Crookston soil is lighter textured, drier and cooler than soil at the other two sites. Weather data including net radiation, maximum and minimum temperature, precipitation, relative humidity and wind speed were obtained from the Minnesota Agricultural Experiment Station weather stations located at each site. At the start of the experiment, soil data for modeling were obtained from soil samples collected at the start of experiment and the USDA soil survey geographic database (SSURGO). We relied on SSURGO for sand, silt, and clay proportion, soil organic carbon, water content at wilting point (drained lower limit), and field capacity (drained upper limit), soil saturated hydraulic conductivity, bulk density, soil pH, cation exchange capacity, water table depth, slope, and drainage conditions. Annual rainfall at Lamberton, Waseca and Crookston from 2013 to 2015 averaged 670 ± 68 mm, 976 ± 134 mm and 479 ± 43 mm, respectively. There was high variation in rainfall amount and distribution during different years (Figure 1). Surface soils ranged in textural class from silty clay loam at Lamberton to clay loam at Waseca, to loam at Crookston (Table 1). The Lamberton and Waseca soils were moderately to poorly drained (soil hydrological group of C/D), while Crookston soil was somewhat moderately drained (soil hydrological group of B/C). All fields were relatively flat (<2% slope) and naturally drained. The Lamberton and Waseca soils had higher soil organic matter, cation exchange capacity and soil water content at −33 and −1,500 kPa, with lower soil saturated hydraulic conductivity, compared to Crookston. The bulk density of soil decreased in the order of Crookston > Lamberton > Waseca.

Experimental management practices

The corn and IWG experiments were initiated in 2013 using three spring-applied fertilizer N rates, i.e., low (zero), medium (80 kg N ha^{-1}) and high (160 kg ha^{-1}). However, during the years 2014–2015, medium and high fertilizer N rates for IWG were reduced to 60 and 120 kg ha^{-1} , respectively. Corn received N fertilizer as pre-plant, while IWG was fertilized in late May. Both crops received N fertilizer urea as broadcast. Corn was planted in May each year at 76 cm row spacings with $86,500 \text{ seeds ha}^{-1}$. IWG (variety TLI-C2) was seeded in August 2011 at Lamberton and Waseca, and in May 2012 at Crookston, at 15-cm row spacing using $18 \text{ kg seeds ha}^{-1}$. Each crop was seeded in a plot size of 4.5 by 7.5 m. The preceding crop at all sites during 2012 was soybean, which received no N fertilizer. During 2012, IWG crop was mowed in the fall. IWG and corn were harvested in mid-August

and during last week of September, respectively. Crop yields were estimated annually for the years 2013–2015. The grain yield of IWG was determined by cutting the IWG in a 0.5 m^2 sample quadrat to a stubble height of 10 cm. Corn grain yield and biomass were obtained from a 3 m^2 area. Grain yield and biomass of corn and IWG were determined after drying at 60°C for 5 days. After yield samples were collected, all remaining biomass from all crop treatments was removed. The $\text{NO}_3\text{-N}$ concentration in the soil solution at 0.50 m depth was determined by collecting soil solution samples with a suction tube lysimeter with porous ceramic cups. We used measured data for soil moisture content and soil profile $\text{NO}_3\text{-N}$ as an initial input (Supplementary Table S1). Details of the IWG field experiments are given in Jungers et al. (2017, 2019), Tautges et al. (2018), and Frahm et al. (2018). Some agronomic parameters for modeling were obtained from IWG trials conducted at Winnipeg, Canada by the Univ. of Manitoba (Cattani and Asselin, 2018) and at Saint Paul, MN (Jungers et al., 2017, 2018). Moreover, we relied on detailed un-published experimental data with replications from the Lamberton, Waseca and Crookston sites for modeling (Supplementary Table S2).

DSSAT model description

The DSSAT model comprises crop simulation models that predict growth, development and yield for over 42 crops as well as soil water and N balances. DSSAT v4.7.5 has a modular structure consisting of cropping system, weather, soil, and crop management modules. The DSSAT modules CERES-Maize and CROPGRO-PFM (perennial forage model) were used to conduct simulations for maize and IWG, respectively. The perennial CROPGRO-Forage model developed by Rymph et al. (2004) was used in this study to simulate IWG, because it includes storage organs, for a better representation of carbon and N partitioning, and consequently patterns of re-growth. It also has proven ability to predict growth and tissue N composition of perennial grass in response to daily weather, N fertilization, harvest management, and allows winter dormancy or regrowth after 100% foliage harvest or freeze damage. The model code has been improved and used in development of model parameters to allow prediction of several tropical forage crops (Pequeno et al., 2014).

Model calibration procedure

Evaluation of leaching losses was carried out at each of the three sites receiving three N rates applied to corn and IWG during the spring. The DSSAT model was calibrated to assess crop yield/biomass and $\text{NO}_3\text{-N}$ leaching losses at a depth of 1.2 m. DSSAT was calibrated at each of the three sites for high ($120\text{--}160 \text{ kg ha}^{-1}$) N rates using 2013–2015 data to assess the yield/biomass, soil moisture content, deep percolation, and $\text{NO}_3\text{-N}$ leaching under maize and IWG crop. The model was subsequently validated for medium and low N rates applied at those three sites for the same years. DSSAT model calibration was based on: FAO-56 method for estimation of ETC; Ritchie 1-D tipping bucket method for infiltration; photosynthesis using leaf photosynthesis response curve; organic matter by Century model; hydrology by Ritchie water balance; and soil evaporation by Ritchie-Ceres method, using a modified soil profile. Due to the low slope, runoff potential of all three sites was selected as moderately low.

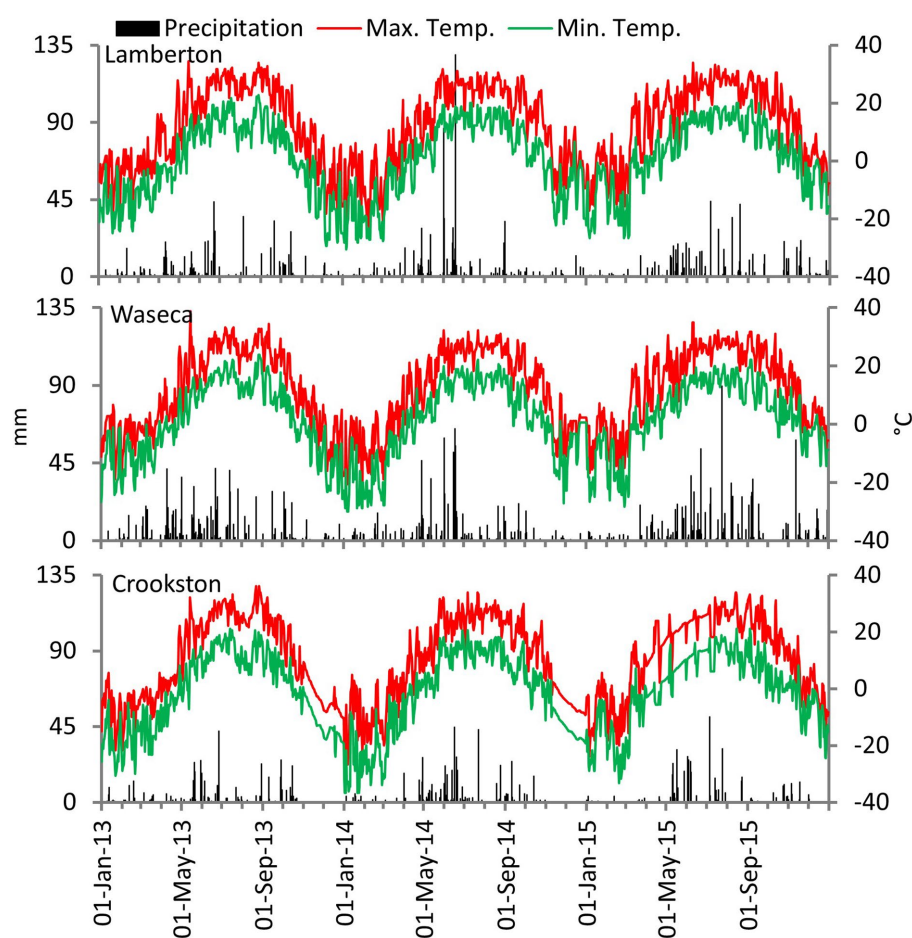


FIGURE 1

Daily precipitation at experimental sites located in Lamberton (top), Waseca (middle), and Crookston (lower), Minnesota from Jan. 2013 to Dec. 2015.

TABLE 1 Soil profile physico-chemical properties from three experimental field sites used as initial condition inputs for the DSSAT model calibration.

Location	Depth (m)	Soil type*	Sand	Clay	SOM	θ_{33kPa}	$\theta_{1500kPa}$	Ksat (mmhr ⁻¹)	B.D. (g cm ⁻³)	CEC (mEq/100 g)	H.G.
			%			v/v					
Lamberton	0.0–0.5	SiCL	10.0	32.0	3.8	0.33	0.19	3.5	1.30	25.9	C/D
	0.5–1.2	SiCL	20.0	30.0	1.8	0.30	0.18	3.5	1.36	22.8	C/D
Waseca	0.0–0.5	CL	25.0	30.0	4.5	0.34	0.20	14.5	1.25	24.0	C/D
	0.5–1.2	CL	26.5	31.0	1.7	0.31	0.19	14.5	1.35	20.0	C/D
Crookston	0.0–0.5	L	42.0	25.4	2.5	0.26	0.15	33.0	1.4	9.9	B/C
	0.5–1.2	CL	28.0	27.5	1.3	0.30	0.18	14.0	1.37	17.1	C

*USDA classification; SiCL, silty clay loam; CL, clay loam; L, loam; SOM, soil organic matter; Ksat, soil saturated hydraulic conductivity; B.D., bulk density; CEC, cation Exchange capacity; H.G. hydrological soil group.

The saturated upper limit (SUL) was considered equal to porosity, calculated from the soil bulk density. Drained upper limit (DUL) and lower limit (LL) were taken as SSURGO value of water content at field capacity, and wilting point, respectively (Table 1). DSSAT was calibrated using data from four replicates at each N rate. For corn simulation, genetic coefficients were calibrated to simulate the response of corn crop to weather and management conditions. The

observed experimental data were compared with the model simulation results. The CERES-maize cultivar calibration requires the estimation of six genetic coefficients, i.e., P_2 (delay in development with photoperiod above 12.5 h), P_5 (thermal time from silking to physiological maturity), PHINT (phyllochron interval), P_1 (thermal time from seedling emergence to the end of the juvenile growth period), G_2 (maximum possible number of kernels per plant) and G_3

(potential kernel growth rate). These coefficients were modified during calibration. First, the coefficients controlling phenology (P_1 , P_2 , P_5 , and PHINT) were modified to match anthesis and maturity dates, and leaf number. Later, the G_2 and G_3 parameters were adjusted to match the measured and modeled biomass and yield. At each site measured values of tillers m^{-2} (Supplementary Table S3) were used for site specific calibration.

To adapt CROPGRO-PFM for predicting the regrowth and yield of IWG, we followed an approach to develop the required data and cultivar traits based on: (i) genetic values and relationships reported in literature; and (ii) a comparison with observed experimental growth, biomass and NO_3 -N leaching losses data from IWG field experiments. Our starting point for IWG was the CROPGRO-PFM model adopted for *Brachiaria brizantha* (Marandu palisade grass) by Pequeno et al. (2014), included in DSSAT version 4.8 software (Hoogenboom et al., 2019). This module was applied to simulate IWG regrowth and soil water balance based on the experimental conditions for soil, weather, and crop management. As an initial input we considered optimized parameters of plant composition, phenology, and productivity from Pedreira et al. (2011) and Pequeno et al. (2014). Temperature, solar radiation, and photoperiod effects on vegetative partitioning, specific leaf area and photosynthesis were based on optimized parameters from Marandu palisade grass (Pedreira et al., 2014). Carbon and nitrogen mining parameters, and senescence parameters were also calibrated from Marandu palisade grass (Pedreira et al., 2011; Pequeno et al., 2014). Soil-water retention and hydraulic conductivity values obtained from SSURGO soil database (Table 1) were used in model calibration at each experimental location.

Model validation and performance assessment

Evaluation of model accuracy was performed using the Nash-Sutcliffe coefficient (NSE) as well as graphical comparison of measured and simulated outputs. The NSE equation (Nash and Sutcliffe, 1970) is:

$$NSE = 1 - \frac{\sum_{i=1}^n (O_i - S_i)^2}{\sum_{i=1}^n (O_i - O_m)^2}$$

where: S_i is simulated data, O_i is observed data, and O_m is mean of observed data. NSE values above 0.75 and 0.5 indicate very good and satisfactory model performance, respectively (Moriasi et al., 2007).

Results

Model calibration parameter analysis

Model calibration was limited to use of site specific soil data and optimization of key crop growth parameters. Soil texture, saturated upper limit (SUL), drained upper limit (DUL), lower limit (LL), soil organic matter, bulk density, and soil saturated hydraulic conductivity (K_{sat}) values at each site obtained from SSURGO soil databases for the 0–0.5 m and 0.5–1.2 m depths (Table 1) were important for accurate calibration of soil profile moisture and deep percolation losses. The

TABLE 2 Model performance (NSE values) for corn and IWG during calibration and validation.

Crop	Period	Yield/ biomass	Soil moisture	NO_3 -N leaching
Corn*	Calibration (N2)	0.86	0.81	0.76
	Validation (N1)	0.72	0.71	0.51
	(N0)	0.58	0.78	0.61
IWG	Calibration (N2)	0.89	0.71	0.63
	Validation (N1)	0.73	0.64	0.43
	(N0)	0.65	0.53	0.47

NSE values for corn were based on the grain yield, while above ground biomass was considered in case of IWG.

values of SUL were 0.51, 0.53, and 0.47 $m^3 m^{-3}$ for 0–0.5 m depth, and 0.49, 0.49, and 0.48 $m^3 m^{-3}$ for 0.5–1.0 m depth for Lamberton, Waseca and Crookston, respectively. The DUL values of the respective sites were 0.33, 0.34, and 0.26 $m^3 m^{-3}$ for 0–0.5 m depth, and 0.30, 0.31, and 0.30 $m^3 m^{-3}$ for 0.5–1.2 m depth. K_{sat} values at 0–0.5 and 0.5–1.2 m depths for Lamberton (3.5 $mm hr^{-1}$) were lower than at Waseca (14.5 $mm hr^{-1}$). Crookston had higher K_{sat} values at the 0–0.5 m depth (33.0 $mm hr^{-1}$), compared to the 0.5–1.2 m depth (14.0 $mm hr^{-1}$). The LL values for the 0–0.5 m and 0.5–1.2 m depths were 0.19 and 0.18 $m^3 m^{-3}$, 0.20 and 0.19 $m^3 m^{-3}$, and 0.15 and 0.18 $m^3 m^{-3}$ at Lamberton, Waseca, and Crookston, respectively. Corn yield was optimized using maize cultivar coefficient values of: $P1=220$; $P2=0.75$; $P5=850$; $G2=730$; $G3=9.6$; and PHINT = 36.0. The IWG optimized parameters included slight adjustment of DSSAT parameter values related to crop cultivar, eco file (Supplementary Table S4), and species file (Supplementary Tables S5–S8), related to plant composition, phenology, and productivity; temperature, solar radiation and photoperiod effects on vegetative partitioning, root growth, specific leaf area, and photosynthesis; carbon and nitrogen mining; and senescence parameter. Moreover, IWG crop simulation relied on crop management and measured parameters (Supplementary Tables S2, S3) and on initial estimates for soil moisture, soil NO_3 -N, soil slope, and water table depth.

Model performance assessment

DSSAT model accuracy in simulating grain yield of corn, and above ground biomass of IWG, soil profile moisture, deep percolation and NO_3 -N leaching losses across a range in fertilizer N rates and crop type at three Minnesota experimental sites is summarized in Table 2. The overall ability of model to estimate the corn yield and IWG biomass at harvest was very good for calibration (NSE, 0.86 and 0.89) and satisfactory (NSE, ≥ 0.58 and ≥ 0.65) for the validation treatments. NSE values indicate that DSSAT model performance during calibration and validation was good for estimating soil profile moisture under corn and IWG at all locations and N rates (NSE, >0.64), except for satisfactory performance (NSE = 0.53) for IWG receiving no N fertilizer. NO_3 -N leaching loss estimates by DSSAT had lower values of NSE, compared to other model output estimates. NO_3 -N leaching loss estimates by DSSAT for calibration under corn and IWG (NSE values ranging between 0.76 and 0.63) were very good and satisfactory, respectively. During validation, NSE values for NO_3 -N leaching losses

under corn (0.61–0.51) were satisfactory, while NSE values (0.47–0.43) for IWG showed close to acceptable accuracy at the medium and low N rates.

Grain yield and biomass

Simulated corn and IWG grain yield were affected by N rates and experimental site location (Table 3). Grain yield at the low N fertilizer rate averaged 5.25, 6.04, and 5.05 t ha⁻¹ for corn, and 0.37, 0.27, and 0.44 t ha⁻¹ for IWG at Lamberton, Waseca and Crookston, respectively. Relative to the low N fertilizer rate, corn yield at Lamberton, Waseca and Crookston increased by 31.8, 43.5, and 37.2% with medium N rates; while an increase of 60.4, 57.6, and 59.2% was observed with high N rates, respectively. Relative to low N fertilizer rates, IWG grain yield at Lamberton and Waseca increased by 40.5 and 55.6% in response to medium N fertilizer, while high rates caused a reduction in yield. However, IWG grain yield increased by 18.2 and 23.1% with the application of medium and high N fertilizer rates, respectively, compared to no fertilizer application. IWG biomass response to medium and high N rates was lower compared to corn stover biomass at all three Minnesota locations. At the low N rate, IWG produced 7.81, 7.11 and 8.51 t ha⁻¹ biomass at Lamberton, Waseca and Crookston, respectively. Applying medium N rates increased IWG biomass by 26.4, 30.2, and 17.7%, while a further increase of 6.1, 7.0, and 12.0% was observed at high N rates, compared to low N rates. Corn stover biomass with low fertilizer N rates averaged 5.71, 6.49, and 5.54 t ha⁻¹, at Lamberton, Waseca, and Crookston, respectively. Application of medium and high N rates increased the stover biomass at these sites by 36.1 and 48.5%, 36.4 and 49.0%, and 30.0 and 44.6%, respectively. The Waseca site had relatively higher corn yield and stover biomass across different N rates, compared to the Lamberton and Crookston sites, respectively. IWG grain yield and above ground biomass at Waseca, however, were lower across all N rates, compared to the Lamberton and Crookston sites, respectively.

Crop evapotranspiration

ET_c values were affected by the type of crop, N rates, as well as spatio-temporal variations in the soil and climatic conditions across experimental locations (Figure 2). Annual ET_c of the perennial IWG was greater on average than the annual ET_c of corn at all three sites. Averaged across all sites, three-year annual ET_c in Low N receiving

corn and IWG averaged 504.4 ± 59.3 mm and 573.2 ± 73.8 mm. The application of N fertilizer to corn and IWG increased ET_c by 6.6 and 3.9% at medium N rates, and by 8.7 and 5.6% at high N rates, respectively. Across all N rates, corn averaged 527.7 ± 25.5, 582.0 ± 24.3, and 442.4 ± 20.2 mm in annual ET_c at Lamberton, Waseca and Crookston, respectively. IWG at the respective sites averaged 15.8, 13.6 and 13.8% higher ET_c, compared to corn. Temporal variations in ET_c were also observed at all three sites. Averaged across N rates, ET_c under corn and IWG was as low as 432.2 and 481.9 mm at Crookston during the year 2013, and as high as 597.0 and 680.6 mm at Waseca during the year 2015, respectively. These two sites during the years 2013 and 2015 received 443 mm and 937 mm annual precipitation.

Deep percolation losses

Planting perennial instead of annual crops and application of optimum N fertilizer caused a marked reduction in deep percolation losses, though high temporal and spatial effects were observed under both cropping systems (Figure 3). Averaged across all N rates, corn at Lamberton, Waseca and Crookston had 168.6 ± 38.3 mm, 228.1 ± 42.4 mm, and 67.0 ± 18.2 mm of annual deep percolation at a 1.2 m depth, respectively. Regardless of N fertilizer rates, planting perennial IWG reduced deep percolation by 34.7% relative to corn, with a 33.9, 32.2, and 45.7% reduction observed at Lamberton, Waseca and Crookston, respectively. Averaged across all sites, deep percolation under high N rates for corn and IWG averaged 145.0 ± 74.4 mm and 95.3 ± 58.1 mm. Annual deep percolation under corn and IWG across years ranged from 121.3–206.6 mm and 68.1–146.5 mm at Lamberton; 196.0–280.1 mm and 133.2–194.0 mm at Waseca; and 55.5–89.2 mm and 29.8–44.7 mm at Crookston, respectively.

Nitrate-N leaching

Annual NO₃-N leaching losses were substantially reduced by planting perennial IWG, compared to the annual cropping system (Figure 4). In general, IWG resulted in 6–7 times lower annual NO₃-N leaching losses, compared to leaching losses observed under corn. Across all N rates during 3 years, corn averaged 17.9 kg ha⁻¹ in NO₃-N leaching losses against 2.6 kg ha⁻¹ under IWG. NO₃-N leaching losses were also reduced remarkably as N fertilizer rates decreased. Annual NO₃-N leaching losses in corn receiving low N averaged 11.3 ± 3.2, 14.3 ± 5.6, and 5.4 ± 2.2 kg ha⁻¹ at Lamberton, Waseca and Crookston,

TABLE 3 Grain and biomass of corn and IWG at different N rates applied at three experimental sites.

Crop	N level	Grain yield (t ha ⁻¹)			Biomass (t ha ⁻¹)		
		Lamberton	Waseca	Crookston	Lamberton	Waseca	Crookston
Corn	Low	5.25 ± 0.6 ^a	6.04 ± 0.6	5.00 ± 0.6	5.71 ± 0.5	6.49 ± 0.3	5.54 ± 0.5
	Medium	6.92 ± 0.4	8.67 ± 0.9	6.86 ± 0.5	7.77 ± 0.3	8.85 ± 0.8	7.20 ± 0.4
	High	8.42 ± 0.5	9.52 ± 1.7	7.96 ± 0.7	8.48 ± 0.3	9.67 ± 1.0	8.01 ± 0.5
IWG*	Low	0.37 ± 0.5	0.27 ± 0.4	0.44 ± 0.4	7.81 ± 1.2	7.11 ± 0.8	8.51 ± 1.1
	Medium	0.52 ± 0.5	0.42 ± 0.4	0.52 ± 0.4	9.87 ± 0.6	9.26 ± 1.1	10.02 ± 0.7
	High	0.38 ± 0.2	0.34 ± 0.3	0.64 ± 0.3	10.47 ± 0.7	9.91 ± 0.9	11.22 ± 0.5

^aMeans ± standard deviation; *Grain yield of IWG were measured values, all other data are simulated values.

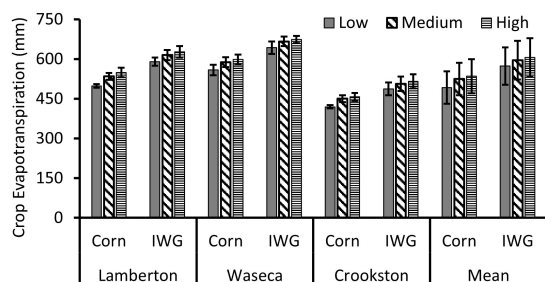


FIGURE 2
Annual predicted ET_c at three Minnesota experimental sites for low, medium and high N fertilizer application rates from Jan. 2013 to Dec. 2015.

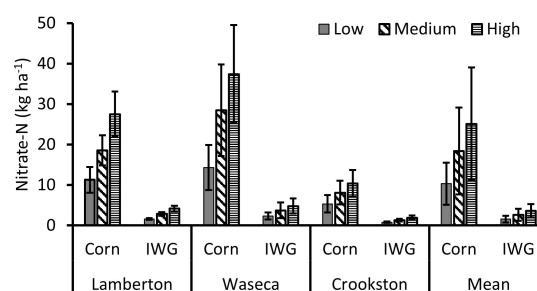


FIGURE 4
Annual predicted nitrate-N leaching at three Minnesota experimental sites receiving low, medium or high N fertilizer application rates from Jan. 2013 to Dec. 2015.

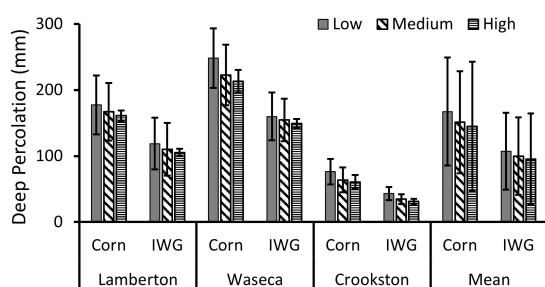


FIGURE 3
Annual predicted deep percolation at a depth of 1.2 m for three experimental sites receiving low, medium or high N fertilizer application rates in Minnesota from Jan. 2013 to Dec. 2015.

respectively, while nitrate-N leaching losses under IWG averaged only 1.6 ± 0.2 , 2.3 ± 0.9 , and $0.8 \pm 0.2 \text{ kg ha}^{-1}$, respectively. Averaged across all sites, application of N fertilizer at medium rates increased $\text{NO}_3\text{-N}$ leaching losses under corn and IWG relative to a control with no fertilizer by 64.3 and 80.6%, respectively, while increasing N fertilizer to high rates further increased the $\text{NO}_3\text{-N}$ leaching losses under respective crops by 48.3 and 46.2%. Climatic variability across years caused high temporal variations in nitrate-N leaching. $\text{NO}_3\text{-N}$ leaching losses during 2013, 2014, and 2015 averaged 13.1 ± 8.3 , 18.9 ± 9.3 , and $21.9 \pm 16.1 \text{ kg ha}^{-1}$ under corn, and 2.1 ± 1.1 , 2.6 ± 1.3 , and $3.1 \pm 2.1 \text{ kg ha}^{-1}$ under IWG, respectively. Under corn, the highest $\text{NO}_3\text{-N}$ leaching losses (52.2 kg ha^{-1}) were observed with high N rates at Waseca during 2015, while the lowest leaching losses (3.45 kg ha^{-1}) were observed at Crookston with low N rates during 2013. Nitrate-N leaching losses for IWG during the corresponding years at these sites had maximum and minimum values of 7.0 kg ha^{-1} and 0.63 kg ha^{-1} , respectively.

Soil profile moisture

Soil profile moisture was affected by cropping system, with high spatio-temporal variations observed for Lamberton, Waseca and Crookston sites (Figure 5). Field observations as well as DSSAT modeling indicate that IWG resulted in consistently lower soil moisture, compared to corn. During the months of April and early-May, and in October–November, the presence of perennial IWG

averaged substantially lower soil moisture, compared to soil in corn plots. DSSAT simulations of three-year (April–November) soil profile moisture at Lamberton, Waseca and Crookston under corn and IWG averaged 0.27 vs. $0.23 \text{ m}^3/\text{m}^3$, 0.34 vs. $0.29 \text{ m}^3/\text{m}^3$ and 0.26 vs. $0.22 \text{ m}^3/\text{m}^3$, respectively. Soil moisture was on average much higher at Waseca, compared to Lamberton and Crookston. Regardless of N fertilizer rates, soil profile moisture in corn plots during 2013, 2014, and 2015 averaged 0.26 , 0.27 , $0.29 \text{ m}^3/\text{m}^3$ at Lamberton; 0.33 , 0.32 , and $0.36 \text{ m}^3/\text{m}^3$ at Waseca; and 0.25 , 0.27 , and $0.27 \text{ m}^3/\text{m}^3$ at Crookston, respectively. IWG in 2013, 2014 and 2015 had an average soil profile moisture of 0.22 , 0.22 , and $0.24 \text{ m}^3/\text{m}^3$ at Lamberton; 0.29 , 0.26 , and $0.30 \text{ m}^3/\text{m}^3$ at Waseca; and 0.21 , 0.23 , and $0.22 \text{ m}^3/\text{m}^3$ at Crookston, respectively. Overall, there was a good correlation between the measured and simulated soil moisture contents, except for some cases where DSSAT over-predicted soil moisture under IWG.

Discussion

Parameter optimization and model performance

We calibrated the DSSAT model for estimation of grain yield and stover biomass of corn, and above-ground biomass of IWG, as well as for soil profile moisture before proceeding to assess deep percolation losses and then $\text{NO}_3\text{-N}$ leaching losses. The model was very good to satisfactory in estimating corn grain yield, corn and IWG biomass, and soil moisture under both crops. Generally good to satisfactory performance of the model was observed for estimating $\text{NO}_3\text{-N}$ leaching losses under corn, and IWG crops, respectively. Large variations were observed for soil solution $\text{NO}_3\text{-N}$ concentration measurements, compared to the variations in measured soil moisture. High variability in measured data contribute the lower NSE values associated with nitrate-N leaching in IWG receiving no N fertilizer (Jungers et al., 2019).

The DSSAT model has previously been used to simulate perennial grasses such as *Brachiaria brizantha* (Marandu palisade grass) by Pequeno et al. (2014), bermudagrass (*Cynodon* spp.) by Pequeno et al. (2017), and alfalfa (*Medicago sativa* L.) by Malik et al. (2018). However, this is first attempt to simulate IWG biomass using the DSSAT model. IWG is a relatively new crop with large genetic diversity, and much is still unknown about its adaptability to large variations of climatic and

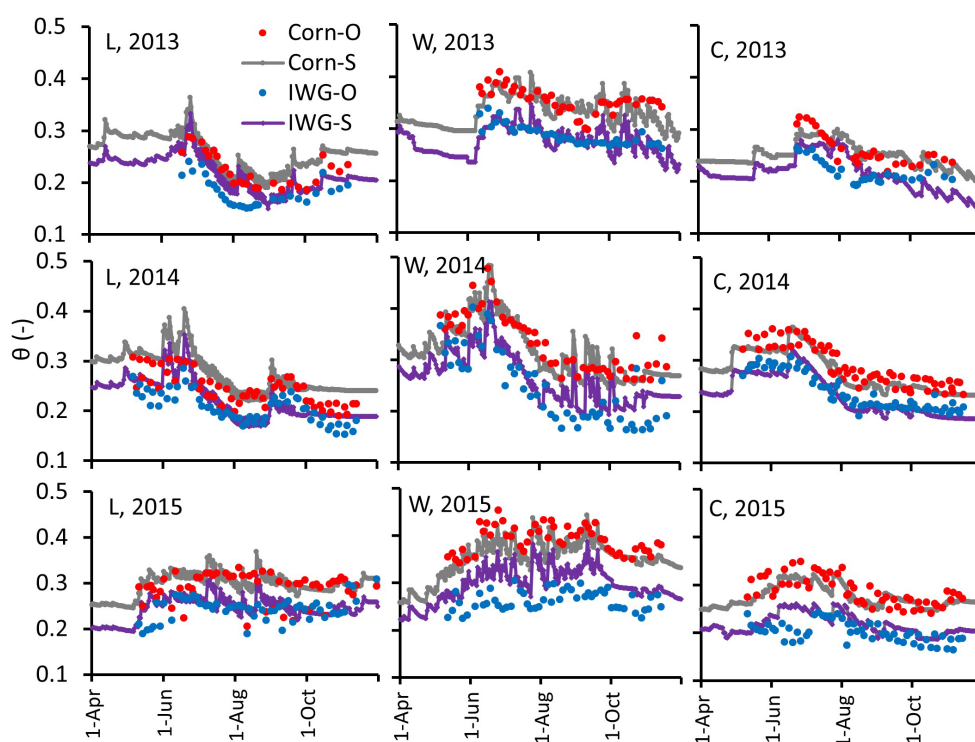


FIGURE 5

Field observed (O) and DSSAT estimated soil profile (1.2 m depth) moisture at Lamberton (L), Waseca (W), and Crookston (C) under corn and IWG from Apr. 1 to Nov. 30 of 2013 (top row), 2014 (middle row) and 2015 (bottom row).

soil conditions. This makes it challenging to calibrate the model for IWG yield as optimized IWG crop parameters are not available for a wide range in soil types and climatic conditions. More detailed studies of IWG are needed in future to improve model performance, particularly for grain yield and nitrate-N leaching. Nevertheless, model performance for IWG biomass was comparable to accuracy of DSSAT estimates for corn yield.

Effect of N fertilizer rates on crop yield and biomass

Increasing N fertilizer application from low to medium rates increased IWG grain yield and biomass by 40.5 and 26.4% at Lamberton, and 55.6 and 30.2% at Waseca, respectively. However, further increasing the N fertilizer to high rates at these sites caused a decline in grain yield and only caused a slight increase in IWG biomass. However, a low, but gradual increase of 18.2 and 23.1% in grain yield and 17.7 and 12.0% in biomass was observed at Crookston. IWG is vulnerable to substantial lodging at high N rates (Jungers et al., 2017). Additionally, higher precipitation at Lamberton and Waseca sites might have caused an increase in lodging, and decreases in grain yield, compared to Crookston. In contrast, corn responded strongly to increasing N fertilizer levels. At medium and high N rates, corn yield increased by 36.5 and 16.4%, and biomass by 36.9 and 12.0%, respectively. Previous findings by Jungers et al. (2017) showed that agronomically optimum N rates for maximizing IWG grain yield ranged from 61–96.4 kg N ha⁻¹. However, corn has much higher N requirements. Kaiser et al. (2016) issued N fertilizer guidelines for

rainfed continuous corn, indicating that the maximum return to N value (MRTN at 0.05 N price to crop value ratio) for Minnesota corn is 202 kg ha⁻¹, with an acceptable range of 179–224 kg ha⁻¹. Corn yield decreased across locations in the order Waseca > Lamberton > Crookston. This pattern followed the same trends across sites in soil organic matter (Table 1) and precipitation (Figure 1). The higher availability of N by mineralization from organic matter rich soil might have resulted in higher yield of corn. IWG yield and biomass, unlike corn yield and biomass, were not correlated to trends in organic matter or precipitation across sites. The Waseca site with highest organic matter did not show the maximum IWG biomass. The lower yield and biomass of IWG at Waseca was due to lower plant populations (Supplementary Table S3). IWG simulated yield and biomass were similar to values previously reported in Minnesota (Jungers et al., 2017).

Crop evapotranspiration

IWG showed higher annual DSSAT predicted ET_c than for corn at all locations (Figure 2). Annual ET_c for corn and IWG averaged 492.3 and 573.7 mm under low N fertilizer, and 535.2 and 605.8 mm under high N rates, respectively. Jungers et al. (2019) estimated no differences in seasonal ET_c under both crops with the Denitrification and Decomposition (DNDC) model, however, their data covered only the months between May–October. DSSAT model simulations in the present study, however, were based on annual ET_c estimates. Across all years and N rates, IWG averaged a 14.4% (74.5 mm) increase in annual ET_c, compared to corn. IWG is a C3 crop that

starts growth and transpiration in early spring, compared to C4 corn, which is planted in early May. Thus, higher ET_c by IWG during early spring should be considered while making comparisons between the annual and perennials. Application of N fertilizer at medium and high N rates increased crop evapotranspiration under both crops, compared to low N rates. Results indicate that water use efficiency increased with application of N both under corn and perennial IWG at all locations. Previous findings also indicate that water use efficiency of corn and perennial grasses including IWG increased with increasing fertilizer N rates (Ferchaud et al., 2015; Jungers et al., 2019). ET_c was highly variable across different locations, mainly because of precipitation. Waseca and Crookston sites showed highest and lowest ET_c under both crops, respectively. Moreover, annual average maximum and minimum temperatures at Crookston were 1.83°C and 2.26°C lower than at Lamberton, and 1.71°C and 3.17°C lower than at Waseca, respectively. Cooler weather at Crookston in northwestern Minnesota also contributed to lower ET_c than at southern (Waseca) or southwestern Minnesota (Lamberton) locations.

Deep percolation and nitrate-N leaching losses

Increases in biomass at higher N rates resulted in higher ET_c , and thus was associated with small reductions in deep percolation losses at all sites (Figure 3). However, this reduction in percolation at higher N rates did not cause reductions in NO_3 -N leaching losses (Figure 4). This suggests that the increase in NO_3 -N leaching losses with higher N fertilizer rates in maize and IWG were largely driven by the soil solution N concentrations, and that IWG has much lower NO_3 -N concentrations and NO_3 -N leaching losses compared to other crops under similar soil and climate conditions (Jungers et al., 2019). Moreover, most NO_3 -N leaching losses under IWG occurred before or after the active growing season (Supplementary Figures S1, S2). Though decreases in deep percolation for IWG at higher N rates may not be desirable in arid regions where groundwater recharge is important, it does result in more water being available for IWG uptake. Thus, higher production of IWG biomass was observed at Crookston under lower precipitation, compared to Waseca, which received the highest rainfall. Although reducing N fertilization to corn decreased NO_3 -N leaching, there was an economic loss in the form of reduced grain yields, which would not generally be acceptable for farmers. However, substituting corn with IWG produced both high biomass as well as reasonable grain production. NO_3 -N leaching under IWG was remarkably low even at higher N fertilizer rates. IWG showed 34.7% lower deep percolation compared to corn (154.5 mm vs. 100.8 mm), however, NO_3 -N leaching losses for IWG averaged across sites were 6–7 times lower than was observed for corn. This also implies that IWG may have much higher nitrogen use efficiency than corn under these soil and climatic conditions. Previously, higher N use efficiency in IWG has been observed compared to annual wheat and corn (Sprunger et al., 2018; Jungers et al., 2019), respectively. Increased N concentration in IWG aboveground biomass with 60–80 kg N ha⁻¹ fertilizer, compared to unfertilized IWG indicates that N fertilization in IWG increases N use efficiency (Tautges et al., 2018). Thus, reduced deep percolation and NO_3 -N leaching losses by IWG was caused by higher water and N use efficiency of IWG, compared to annual crops.

IWG may have higher water, and N use efficiency compared to corn due to a variety of reasons. Corn is a C4 crop and initiates growth and transpiration later in the spring compared to IWG, a C3 crop. Thus, corn could have lower water use efficiency and N fertilizer uptake in the spring, resulting in larger NO_3 -N leaching losses with percolating water. Compared to corn, perennial IWG has an extended growing season, which requires assimilation of soil available N during times when it is susceptible to leaching and annual crops are absent from the landscape. In particular, the absence of corn during the early spring, when precipitation is abundant, causes higher deep percolation and NO_3 -N leaching losses, compared to percolation and leaching losses when IWG is present. At Lamberton, Waseca and Crookston, 22.6, 23.8, and 26.7% of annual precipitation occurred during the months of April and May when an annual crop was either absent from the field or at an early growth stage, resulting in limited water and N uptake by corn. IWG is a deep-rooted cool-season grass with an extensive rooting system and has the capacity of capturing more water from deeper soil layers than annual crops. Ferchaud et al. (2015) indicated that higher rooting density and rooting depth are important factors, along with a prolonged period for increased water uptake of perennials, compared to annual crops. In Kansas, United States, de Oliveira et al. (2018) demonstrated the ability of IWG in maintaining a relatively high water-use efficiency throughout the whole growing season and having higher ET_c , compared to annual crops. Likewise, higher root biomass and distribution of that biomass to deeper depths by perennial grasses than annuals, can improve N utilization by perennials and limit NO_3 -N leaching losses (Jungers et al., 2019). Estimates using the DNDC model (Jungers et al., 2019) showed lower deep percolation and NO_3 -N leaching losses than estimates in the present study using DSSAT. However, they simulated only the growing season from May–October, while April received abundant precipitation at all sites.

As these sites were under rainfed conditions, deep percolation and NO_3 -N leaching losses in corn and IWG were correlated to precipitation amount and distribution. The reduction in precipitation from 814 mm to 479 mm from southern to northwestern Minnesota reduced deep percolation and NO_3 -N leaching by 3.2 and 3.9 times under corn, and by 3.1 and 2.7 times under IWG, respectively. Similarly, Lamberton results in southwestern Minnesota showed high NO_3 -N leaching losses. Results indicate that better N management is required for corn in southern and southwestern Minnesota under rainfed conditions, and that IWG would be a better option than N fertilizer management in corn for substantially reducing NO_3 -N leaching losses, in addition to the benefit of producing IWG grain yield and biomass for animal feed.

Soil profile moisture

Soil profile moisture (1.2 m depth) was higher under corn than IWG throughout the growing period, as indicated by model simulation as well as measured data (Figure 5). Perennial grasses have higher root biomass that extend to deeper soil layers than annual crops. Thus, water uptake capacity from the soil, especially in the deep layers is greater under perennials like IWG, compared to annual crops like maize (Ferchaud et al., 2015). Although Jungers et al. (2019) indicated that soil moisture in the upper 50 cm depth was statistically similar under IWG and corn with different N rates,

experimental soil moisture data for IWG showed consistently less soil moisture than for corn. Within the crops, high spatiotemporal variations were observed in soil moisture. The three Minnesota sites studied showed large differences in soil moisture. Regardless of temporal variations, a substantial decrease of 17.8 and 41.2% in water input was observed moving from southern Minnesota to southwestern and northwestern Minnesota, respectively. Likewise, moisture was generally highest in Waseca soil, followed by Lamberton, and lowest in the Crookston soil. Moreover, Waseca, Lamberton and Crookston sites with clay loam, silty clay loam and loamy surface soil texture had soil water retention that followed a similar trend. Thus, the soil profile moisture was affected by both soil water input from precipitation as well as soil water retention capacity.

Conclusion

An accurate DSSAT CERES-Maize (corn) and CROPGRO-PFM (IWG) model was optimized using field measured data under rainfed conditions at low, medium and high fertilizer N rates for the three sites in southwestern, southern and northwestern Minnesota. The calibrated model is able to accurately simulate corn grain and stover biomass, and IWG aboveground biomass, soil profile moisture, deep percolation and $\text{NO}_3\text{-N}$ leaching losses. Accuracy of model output decreased in the order grain yield/biomass > soil moisture > nitrate-N leaching. Model performance was better for calibration compared to validation, and of better accuracy for corn, compared to IWG at all three sites. Results indicated that corn and IWG grain yield averaged 5.4 and 0.36 t ha^{-1} , 7.5 and 0.49 t ha^{-1} , and 8.6 and 0.45 t ha^{-1} , at low, medium and high fertilizer N rates, respectively. The respective N rates also produced IWG biomass of 7.8, 9.7 and 10.5 t ha^{-1} . Averaged over N rates and sites, corn and IWG had 517 mm vs. 592 mm ET_c , 154.5 mm vs. 100.8 mm deep percolation, $0.29 \text{ m}_3 \text{ m}^{-3}$ vs. $0.25 \text{ m}_3 \text{ m}^{-3}$ soil profile moisture, and 17.9 kg ha^{-1} vs. 2.6 kg ha^{-1} $\text{NO}_3\text{-N}$ leaching losses, respectively. These results indicate that IWG has a potential for significantly reducing deep percolation and $\text{NO}_3\text{-N}$ leaching losses as a result of higher ET_c and N uptake efficiency, compared to corn. However, high spatio-temporal variations were observed. Moreover, results indicate that long-term assessment is required for addressing high temporal and spatial variations across sites, in order to facilitate use of the calibrated model at unstudied sites. Additional detailed studies are being carried out in Minnesota to provide more experimental data that can be used for further improvements in model simulations of IWG grain yield and nitrate-N leaching.

References

- Cattani, D. J., and Asselin, S. R. (2018). Has selection for grain yield altered intermediate wheatgrass? *Sustainability* 10:688. doi: 10.3390/su10030688
- Christopher, S. F., Tank, J. L., Mahl, U. H., Hanrahan, B. R., and Royer, T. V. (2021). Effect of winter cover crops on soil nutrients in two row-cropped watersheds in Indiana. *J. Environ. Qual.* 50, 667–679. doi: 10.1002/jeq2.20217
- Culman, S. W., Snapp, S. S., Ollenburger, M., Basso, B., and DeHaan, L. R. (2013). Soil and water quality rapidly responds to the perennial grain kernza wheatgrass. *Agron. J.* 105, 735–744. doi: 10.2134/agronj2012.0273
- de Oliveira, G., Brunsell, N. A., Crews, T. E., DeHaan, L. R., and Vico, G. (2020). Carbon and water relations in perennial Kernza (*Thinopyrum intermedium*): an overview. *Plant Sci.* 295:110279. doi: 10.1016/j.plantsci.2019.110279
- de Oliveira, G., Brunsell, N. A., Sutherlin, C. E., Crews, T. E., and DeHaan, L. R. (2018). Energy, water and carbon exchange over a perennial Kernza wheatgrass crop. *Agri. For. Meteorol.* 249, 120–137. doi: 10.1016/j.agrformet.2017.11.022
- DeHaan, L. R., Wang, S., Larson, S. R., Catton, D. J., Zhang, X., and Kantarski, T. (2013). "Current efforts to develop perennial wheat and domesticate *Thinopyrum intermedium* as a perennial grain" in *Perennial crops for food security*. eds. C. Batello, L. Wade, S. Cox, N. Pogna, A. Bozzini and J. Choptiany (Rome, Italy: Proc. the FAO Expert Wor), 390.
- Ferchaud, F., Vitte, G., Bornet, F., Strullu, L., and Mary, B. (2015). Soil water uptake and root distribution of different perennial and annual bioenergy crops. *Plant Soil* 388, 307–322. doi: 10.1007/s11104-014-2335-y

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

Author contributions

DM and MT contributed equally to the conceptualization of the article. JJ and MT led the data analysis. MT led the modeling. MT and DM led the writing and editing of the article. All authors contributed to the article and approved the submitted version.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

The reviewer CW is currently organizing a Research Topic with the author JJ.

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Supplementary material

The Supplementary material for this article can be found online at: <https://www.frontiersin.org/articles/10.3389/fsufs.2023.1010383/full#supplementary-material>

- Frahm, C. S., Tautges, N. E., Jungers, J. M., Ehlke, N. J., Wyse, D. L., and Sheaffer, C. C. (2018). Responses of intermediate wheatgrass to plant growth regulators and nitrogen fertilizer. *Agron. J.* 110, 1028–1035. doi: 10.2134/agronj2017.11.0635
- Hoogenboom, G., Porter, C. H., Shelia, V., Boote, K. J., Singh, U., White, J. W., et al. (2019). Decision support system for Agrotechnology transfer (DSSAT) version 4.7.5. DSSAT Foundation, Gainesville, FL. Available at: <https://DSSAT.net>.
- Huggins, D. R., Randall, G. W., and Russelle, M. P. (2001). Subsurface drain losses of water and nitrate following conversion of perennials to row crops. *Agron. J.* 93, 477–486. doi: 10.2134/agronj2001.933477x
- Jones, J. W., Hoogenboom, G., Porter, C. H., Boote, K. J., Batchelor, W. D., Hunt, L. A., et al. (2003). The DSSAT cropping system model. *Eur. J. Agron.* 18, 235–265. doi: 10.1016/S1161-0301(02)00107-7
- Jungers, J. M., DeHaan, L. R., Betts, K. J., Sheaffer, C. C., and Wyse, D. L. (2017). Intermediate wheatgrass grain and forage yield responses to nitrogen fertilization. *Agron. J.* 109, 462–472. doi: 10.2134/agronj2016.07.0438
- Jungers, J. M., DeHaan, L. H., Mulla, D. J., Sheaffer, C. C., and Wyse, D. L. (2019). Reduced nitrate leaching in a perennial grain crop compared to maize in the upper Midwest, USA. *Agric. Ecosyst. Env.* 272, 63–73. doi: 10.1016/j.agee.2018.11.007
- Jungers, J. M., Frahm, C. S., Tautges, N. E., Ehlke, N. J., Wells, M. S., Wyse, D. L., et al. (2018). Growth, development, and biomass partitioning of the perennial grain crop *Thinopyrum intermedium*. *Ann. Appl. Biol.* 172, 346–354. doi: 10.1111/aab.12425
- Kaiser, D. E., Fernandez, F., Lamb, J. A., Coulter, J. A., and Barber, B. (2016). Fertilizing corn in Minnesota. Univ. Minnesota Ext. AG-FO-3790-D (REVISED 2016) Available at: <https://extension.umn.edu/crop-specific-needs/fertilizing-corn-minnesota>
- Kroening, S., and Vaughan, S. (2019). The condition of Minnesota's groundwater quality, 2013–2017. Minnesota Pollution Control Agency (MPCA) Report. 82 Document number: wq-am1-10. Available at: <https://www.pca.state.mn.us/sites/default/files/wq-am1-10.pdf>.
- Lanker, M., Bell, M., and Picasso, V. D. (2020). Farmer perspectives and experiences introducing the novel perennial grain Kernza intermediate wheatgrass in the US Midwest. *Renew. Agri. Food Syst.* 35, 653–662. doi: 10.1017/S1742170519000310
- Malik, W., Boote, K. J., Hoogenboom, G., Caverio, J., and Dechmi, F. (2018). Adapting the CROPGRO model to simulate alfalfa growth and yield. *Agron. J.* 110, 1777–1790. doi: 10.2134/agronj2017.12.0680
- Minnesota Pollution Control Agency (2013). *Nitrogen in Minnesota surface waters: Conditions, trends, sources, and reductions*. <https://www.pca.state.mn.us/sites/default/files/wq-s6-26i.pdf>
- Moriasi, D. N., Arnold, J. G., Van Liew, M. W., Bingner, R. L., Harmel, R. D., and Veith, T. L. (2007). Model evaluation guidelines for systematic quantification of accuracy in watershed simulation. *Trans. ASABE* 50, 885–900. doi: 10.13031/2013.23153
- Nash, J. E., and Sutcliffe, J. V. (1970). River flow forecasting through conceptual models part I—A discussion of principles. *J. Hydrol.* 10, 282–290. doi: 10.1016/0022-1694(70)90255-6
- Pedreira, B. C., Pedreira, C. G. S., Boote, K. J., Lara, M. A. S., and Alderman, P. D. (2011). Adapting the CROPGRO perennial forage model to predict growth of *Brachiaria brizantha*. *Field Crops Res.* 120, 370–379. doi: 10.1016/j.fcr.2010.11.010
- Pennington, D. N., Dalzell, B., Mulla, D., Taff, S., Hawthorne, P., and Polasky, S. (2017). Cost-effective land use planning: optimizing land use and land management patterns to maximize social benefits. *Ecol. Econ.* 139, 75–90. doi: 10.1016/j.ecolecon.2017.04.024
- Pequeno, D. N. L., Pedreira, C. G. S., and Boote, K. J. (2014). Simulating forage production of Marandu palisade grass (*Brachiaria brizantha*) with the CROPGRO-perennial forage model. *Crop Pasture Sci.* 65, 1335–1348. doi: 10.1071/CP14058
- Pequeno, D. N. L., Pedreira, C. G. S., Boote, K. J., Alderman, P. D., and Faria, A. F. G. (2017). Species-genotypic parameters of the CROPGRO perennial forage model: implications for comparison of three tropical pasture grasses. *Grass Forage Sci.* 73, 440–455. doi: 10.1111/gfs.12329
- Pinto, P., De Haan, L., and Picasso, V. (2021). Post-harvest management practices impact on light penetration and Kernza intermediate wheatgrass yield components. *Agronomy* 11:442. doi: 10.3390/agronomy11030442
- Prokopy, L. S., Gramig, B. M., Bower, A., Church, S. P., Ellison, B., Gassman, P. W., et al. (2020). The urgency of transforming the Midwestern U.S. landscape into more than corn and soybean. *Agric. Hum. Values* 37, 537–539. doi: 10.1007/s10460-020-10077-x
- Reilly, E. C., Gutknecht, J. L., Tautges, N. E., Sheaffer, C. C., and Jungers, J. M. (2022). Nitrogen transfer and yield effects of legumes intercropped with the perennial grain crop intermediate wheatgrass. *Field Crops Res.* 286:108627. doi: 10.1016/j.fcr.2022.108627
- Rymph, S. J., Boote, K. J., Irmak, A., Mislavy, P., and Evers, G. W. (2004). Adapting the CROPGRO model to predict growth and composition of tropical grasses: developing physiological parameters. *Soil Crop Sci. Soc. Fla. Proc.* 63, 37–51.
- Sprunger, C. D., Culman, S. W., Robertson, G. P., and Snapp, S. S. (2018). How does nitrogen and perenniality influence belowground biomass and nitrogen use efficiency in small grain cereals? *Crop Sci.* 58, 2110–2120. doi: 10.2135/cropsci2018.02.0123
- Struffert, A. M., Rubin, J. C., Fernandez, F. G., and Lamb, J. A. (2016). Nitrogen management for corn and groundwater quality in upper Midwest irrigated sands. *J. Environ. Qual.* 45, 1557–1564. doi: 10.2134/jeq2016.03.0105
- Tautges, N. E., Jungers, J. M., DeHaan, L. R., Wyse, D. L., and Sheaffer, C. C. (2018). Maintaining grain yields of the perennial cereal intermediate wheatgrass in monoculture v. biculture with alfalfa in the upper Midwestern USA. *J. Agric. Sci.* 156, 758–773. doi: 10.1017/S0021859618000680
- Zamora, D. S., Jose, S., Jones, J. W., and Cropper, W. P. Jr. (2009). Modeling cotton production response to shading in a pecan alleycropping system using CROPGRO. *Agrofor. Syst.* 76, 423–435. doi: 10.1007/s10457-008-9166-x
- Zhang, X., Larson, S. R., Gao, L., Teh, S. L., DeHaan, L. R., Fraser, M., et al. (2017). Uncovering the genetic architecture of seed weight and size in intermediate wheatgrass through linkage and association mapping. *Plant Genome* 10, 1–15. doi: 10.3835/plantgenome2017.03.0022



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EDITED BY

Jose G. Franco,
Agricultural Research Service (USDA),
United States

REVIEWED BY

Kelly Mercier,
United States Department of Agriculture,
United States
Alan Rotz,
United States Department of Agriculture
(USDA), United States

*CORRESPONDENCE

Colin Cureton
✉ cure0012@umn.edu

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Towards a practical theory for commercializing novel continuous living cover crops: a conceptual review through the lens of Kernza perennial grain, 2019–2022

Colin Cureton^{1*}, Tessa E. Peters², Sophia Skelly²,
Constance Carlson³, Tara Conway¹, Nicole Tautges⁴,
Aaron Reser⁵ and Nicholas R. Jordan¹

¹Forever Green Initiative, Department of Agronomy and Plant Genetics, University of Minnesota, Saint Paul, MN, United States, ²The Land Institute, Salina, KS, United States, ³Regional Sustainable Development Partnerships, University of Minnesota, Saint Paul, MN, United States, ⁴Michael Fields Agricultural Institute, East Troy, WI, United States, ⁵Green Lands Blue Waters, University of Minnesota, Saint Paul, MN, United States

As agricultural scientists rapidly develop and deploy novel continuous living cover (CLC) crops and cropping systems such as perennial grains, a growing number of intermediaries are engaged in advancing the commercialization, adoption, and scaling of these novel CLC crops. However, these commercialization practitioners lack a conceptual and practical roadmap to help them achieve success. Through key concept review and practice narratives, this article presents the firsthand experience of primarily non-academic staff at several key public and nonprofit agricultural innovation platforms between 2019 and 2022 that have held core institutional responsibilities for facilitating the commercialization, adoption, and scaling of Kernza® perennial grain, North America's first commercially-viable perennial grain crop. Reviews of key concepts identified as relevant to the practice of commercializing novel continuous living cover crops are interwoven with practice narratives of the Kernza commercialization process through the lens of each concept, demonstrating the ways in which these concepts translate to specific activities, methods, and strategies, also noting remaining gaps, limitations, and areas for growth and learning. This narrative can move the growing community of CLC intermediaries and innovation brokers toward a 'practical theory' of CLC commercialization that lies at the intersection of technology transfer and adoption, innovation, and agri-food systems change processes. Such conceptual orientation and practical guidance stands to improve the efficacy of novel CLC crop commercialization intermediaries, accelerate wider efforts of agricultural innovation platforms to rapidly advance CLC agriculture, and provide fertile ground for further applied research.

KEYWORDS

continuous living cover, commercialization, stewardship, perennial grains, innovation, technology transfer, intermediaries, sustainability transitions

1. Introduction to commercialization of novel perennial and continuous living cover crops

Perennial and continuous living cover (CLC) crops and cropping systems provide year-round ground cover and long-lived roots, offering a host of water, soil health, biodiversity, pollinator, and climate benefits. At a landscape scale, CLC agriculture can better protect critical natural resources compared to agricultural systems based primarily on summer annual crops (Culman et al., 2013; Eberle et al., 2015; Basche and DeLonge, 2017; Jungers et al., 2019). CLC advocates' implicit theory of change is founded in the idea that new and improved CLC crops and cropping systems must be economically viable and significantly, if not primarily, market-driven. This indicates the need for agricultural production, supply chains, and markets for CLC crops along with significant and ongoing research and development to improve the crops and cropping systems. This process must navigate the notoriously high capital costs, high risk, path-dependency, and low-margin nature of agriculture and the food sector. Compounding issues arise when developing novel CLC crops such as perennial grains compared to, for example, alfalfa, grasslands, and existing winter-hardy crops since those crops do not require additional supply chain and infrastructure development. Social and philosophical dimensions of agriculture are also invoked, proposed, and negotiated as institutions developing novel CLC crops for economic, environmental, and social impact move their research and implementation forward in the world with thousands of actors with diverse interests and perspectives. Novel CLC crops also require incorporation in policy frameworks and in some cases more significant policy innovations. While the benefits of a CLC agricultural system would be tremendous, challenges abound for arriving at this CLC landscape.

Institutions developing CLC crops and systems therefore have their work cut out for them. Following several hard-fought decades of research and development on a novel CLC crop or cropping system, proponents are confronted with a series of systemic technical, economic, regulatory, and cultural barriers to deploying this new crop and its products in the marketplace. Research institutions understand that developing a new crop or cropping system requires dozens of scientists working in well-organized transdisciplinary teams. What's becoming increasingly clear is that it also requires well-supported teams to commercialize novel CLC crops and systems. This process includes crop development scientists as well as growers, engineers, chemists, food scientists, marketers, economists, start-ups, established firms, finance/investors, policymakers, and, the focus of this article, commercialization staff whose purpose it is to weave these actors together to support the adoption and scaling process for novel CLC crops and systems. Commercialization staffs' work stands to benefit from both guiding concepts and practices informed by peers engaged in this work.

Kernza® Perennial Grain is furthest along in navigating these commercialization challenges. Several institutions are collaboratively developing novel perennial grain and CLC crops and systems in the pursuit of a much wider sustainability transition in the agri-food system. Kernza is the trade name of grain, seed, and products derived from varieties of Intermediate Wheatgrass (IWG), *Thinopyrum intermedium*, improved for use as a food-grade grain. IWG is a Eurasian forage grass initially brought to the United States in the early 1900's. It has been under development as a commercially-viable

perennial grain crop for over 30 years by The Rodale Institute, The Land Institute (TLI), the University of Minnesota (UMN), and increasingly other institutions across the world. Since its inception, the Kernza trademark has been owned and managed by TLI and, since 2019, effort has been made to increase the involvement of other early-adopter institutions, growers, processors and end-users in exploring how to manage the trademark more collaboratively.

The relationship between TLI and the UMN is woven together by long-running personal, professional, and institutional relationships. UMN and several of its respective entities, such as the Forever Green Initiative (FGI) and Green Lands Blue Waters (GLBW), hold critical roles in developing CLC crops and systems, developing networks to advance CLC, and supporting the commercialization, adoption, and scaling of CLC agriculture. The education and professional development of key researchers in the Upper Midwest was strongly influenced or supported by TLI, and vice versa. What had been long-running informal or project-specific research collaborations were recently crystallized through the 2020 funding of a major five-year project, KernzaCAP, funded by the United States Department of Agriculture (USDA), National Institute For Agriculture, Sustainable Agricultural Systems Coordinated Agricultural Projects (CAP) program. KernzaCAP takes an integrative approach to further developing Kernza's germplasm, agronomy, food science, understanding of ecosystem services, education, extension, policy, and supply chains and economics. Separate philanthropic and public funding has provided a preceding and ongoing base of support for commercialization staff.

The experience of commercialization and stewardship staff during this critical phase of Kernza perennial grain's development can provide valuable insights for CLC crops and systems that are soon to follow. This paper provides an account of the experience of a self-organized team representing UMN, TLI, and the Michael Field Agricultural Institute (a Wisconsin-based nonprofit) that have led many core commercialization activities for Kernza since 2019 (Table 1). There is a growing recognition that new tools such as perennial grains and oilseeds, woody perennials, and winter annuals will be valuable for advancing the cross-societal commitments to soil health and regenerative agriculture (Crews et al., 2018). The ecosystem of actors advancing CLC continues to expand, and it is critical that these actors have a combination of theoretical framing and practical guidance provided by peer practitioners that includes clear methodologies and strategies that can be iterated and adapted across CLC crops. This article is intended to provide an orientation to the nature of CLC crop commercialization as well as practical guidance on strategies, approach, timelines, mindset, skill sets, and other aspects of CLC commercialization. Taken together, this review may move the CLC community toward what Berkman and Wilson (2021) describe as a 'practical theory' for novel CLC crop commercialization. Such practical theories reside between basic and applied theory, and suggest actionable steps toward solving a problem that currently exists in a particular context in the real world. Practical theory recognizes that theory and practice are not a dichotomy, but rather co-constitutive (Miller and King, 1998). The problem of effectively supporting the launch, adoption, and scaling of novel crops with unique agronomic, physical, environmental, etc. characteristics is one such problem around which actionable steps are needed, the practice of which can improve our understanding of future iterations of novel CLC crop commercialization.

TABLE 1 Commercialization milestones for Kernza® perennial grain.

Year	Event
2009	The Land Institute files for “Kernza” trademark (officially registered in 2011).
2010	Harvest of the first large-scale Kernza field (30 acres) in Kansas occurs, filling a semi-truck with grain, a key proof-of-concept moment.
2011	Food science research begins at the University of Minnesota, launching research in support of Kernza product innovation.
2013	Birchwood Cafe in Minneapolis, MN adds a savory Kernza waffle to the menu, the first Kernza product on a restaurant menu.
2014	The first grower contracts for commercial Kernza production (with grass seed growers in northern Minnesota) are established by Patagonia Provisions.
2016	An early version of a Kernza Grower Guide is made available to farmers and technical assistance providers.
2016	Long Root Ale from Patagonia Provisions becomes the first packaged Kernza product.
2016	General Mills begins experimenting with Kernza in an R&D facility for product development; leading to future launch of limited-run Cascadian Farm brand cereals.
2018	Sustain-A-Grain, a farmer-centered company in Kansas is founded and begins to sell Kernza seed.
2020	Direct-to-consumer Kernza flour and grain sales are made possible online via a new Kernza processing and food brand, Perennial Pantry.
2021	A group of Kernza growers founds The Perennial Promise Grower’s Cooperative.
2022	State support for Kernza supply chain partners becomes a new opportunity when the Minnesota Legislature approves a Continuous Living Cover Value Chain Development Fund.
2022	Stakeholder driven discussions begin to explore the formation of a Kernza Stewards Alliance (KSA).

Adapted from KernzaCAP (2023).

First, this paper provides a brief introduction to the early, pre-2019 commercialization of Kernza, which contextualizes the concerted cross-institutional support for commercialization that followed. The remainder of the paper is devoted to an overview of concepts that elucidate and inform the function of commercialization staff in this collaborative endeavor with accompanying narratives from 2019–2022 that bridge these theories into practice, contributing to a practical theory for novel CLC crop commercialization (Table 2). Early commercialization of Kernza perennial grain, pre-2019.

Following over 20 years of basic research and development on IWG to breed for its viability as a perennial grain crop, the food industry and wider society began to take note of the potential for perennial grains. Early commercialization activity was characterized by a small group of champions (farmers, food businesses and a wide array of other dedicated partners) working through the early hurdles together in committed yet challenging attempts to bring Kernza to market. These early champions demonstrated that growing and creating products with Kernza was possible. As early as 2008, a national, sustainability-minded food company

conducted recipe testing on Kernza tortillas, culminating in a pilot at one store location. Their engagement catalyzed commercial activity at TLI and beyond in the ensuing decade. In 2013, a Minneapolis-based cafe began featuring Kernza waffles on their menu. This provided proof of concept to Minnesota cross-sector stakeholders with budding interest in Kernza, opening the door to new consumer awareness and additional food businesses piloting Kernza, as well as catching the interest of policymakers and nonprofits.

Considering commercialization alongside basic research was not accidental. IWG germplasm development and associated research (e.g., agronomics, food science) at TLI and UMN in the 2010s was coupled with an ecosystem of Minnesota partners such as the Minnesota Institute for Sustainable Agriculture and GLBW to help facilitate early farmer and commercial piloting in Minnesota and the wider Midwest. For Kernza to emerge beyond the academic environment, such entities were needed to serve key logistical roles, including distribution of seed and grain to businesses and farmers and information dissemination, and the cautious but dogged cheerleading role for the potential of the crop. Early commercial experimentation with Kernza, like other novel crops, involved a tremendous amount of troubleshooting such that it was unlikely to be profitable for farmers or food businesses to trial the crop without support. Additional troubleshooting was required to process the grain, which includes cleaning, dehulling, testing for appropriate seed and grain quality, and in some cases milling, before an end user could consider working with the ingredient. The experience of these early actors foreshadowed the need for and functions of the dedicated Kernza commercialization staff that would follow in later years.

Early Kernza commercialization would not have happened without the boldness of a few key farmer and food business leaders willing to go the extra mile to trial a risky and experimental grain. The first grower contracts were established with Minnesota producers in 2014 and by 2015, several Minnesota businesses were piloting Kernza products (e.g., beer, noodles, crackers) and a local mill soon took on milling and distribution. In 2016, the first widely distributed Kernza product, a Kernza beer, hit regional West Coast markets, a major multinational company was testing Kernza as an ingredient at their research and development (R&D) facility, and the media were taking note. The ensuing excitement about Kernza resulted in an influx of interest from businesses and farmers alike in 2016 and 2017.

It soon became clear that the entities developing Kernza perennial grain needed support to facilitate commercialization activities. From 2016 to 2018, TLI contracted a small grain logistics company to increase Kernza acreage by working with existing growers and enlisting new ones. During this phase, growers faced hurdles related to early-stage germplasm, accessing seed, and a lack of sufficient agronomic knowledge and support. Plantings were geographically spread out and relationships with processing partners and buyers were nascent. In the absence of efficient systems to buy, clean, and market Kernza grain and provide farmer technical assistance, the logistics company also stepped into those roles which was a tall order.

The challenges encountered by this company and the wider Kernza community were multifaceted and capacity was limited, straining the existing goodwill of Kernza stakeholders. However, many early partners remained committed and the successes with Kernza during these years piqued the interest of additional restaurants, smaller companies, and major industry. Efforts to keep good communication flowing between stakeholders and to emphasize a

TABLE 2 Key concepts underpinning commercialization of new crops such as Kernza®.

Concept	Definition	Sub-concepts	Implementation/milestones
Technology Transfer	The sharing or introduction of a technology followed by the spread or expanded utilization of the new technology Molnar and Jolly (1988)	Intellectual property, variety releases, and licensing	<ul style="list-style-type: none"> • Release of first commercial Kernza variety, MN-Clearwater • Development of co-exclusive licenses to three regional seed companies
		Commercial trademark	<ul style="list-style-type: none"> • Development of Kernza trademark • Built transparent process of grower vetting and trademark licensing • Annual reevaluation of vetting and licensing priorities
		Physical transfer of Kernza seed and grain	<ul style="list-style-type: none"> • Began highly informally • Evolved into formalized multi-partner process reliant on request intakes and material transfer agreements • Early commercial sales conducted through unique cross-sector partnership with state crop improvement association and seed company • Increasingly, requests are fulfilled by market partners as business development opportunities rather than solely university and NGO partners as technology transfer • Commercial sales of seed and grain by private actors replace institutional tech transfer roles
		Education and programming	<ul style="list-style-type: none"> • Annual call series and development of Kernza informational resources to provide grower support • Development of communication network among early-adopter Kernza growers • Formation of state-supported Kernza technical assistance team • Development of technical resources to support technology adoption along the entire supply chain, spanning from dehulling to baking
		De-risking support	<ul style="list-style-type: none"> • Developing and deploying State support to provide producers with environmental benefit payments, risk management payments, seed and grain testing services, and agronomic support • Developing and piloting a value chain development fund to support post-farmgate entrepreneurs and businesses
Innovation	The commercial introduction of a new product Perez (2010) , as opposed to the invention produced by science and technology. Understood here as the ways in which Kernza and Kernza's associated knowledge find footing in the world in the form of viable products, businesses, and new value propositions, and how Kernza, in turn, informs institutional, public, firm, and consumer priorities, assumptions, and possibilities.	Innovation Systems	<ul style="list-style-type: none"> • Intentional cultivation of a regional system that encompasses many of the actors needed to construct innovative grain systems • System supported by consistent strategic communication and coordination • Innovation system enabled by strong social capital, civic engagement, and state investment in MN.
		Innovation Management	<ul style="list-style-type: none"> • Communicating and integrating learnings, needs, and challenges across R&D, supply chains, and other stakeholders. • Multi-stakeholder collaboration established a baseline understanding of harvest methods, mycotoxin levels, cleanout rates, and more. • Initiated collaborative project to assess evolving harvesting, seed cleaning, processing, milling, and sifting needs in response to improved Kernza germplasm.
Intermediaries	Actors and institutions that positively influence sustainability transition processes by linking entities and their related resources and skills, creating new collaborations across niche technologies like Kernza, linking technologies to markets, and generally creating momentum for system change	Innovation brokers	<ul style="list-style-type: none"> • An early reserve of Kernza from state-supported water trials was provided to Kernza entrepreneurs as 'start-up grain', which helped them launch a business that is on the forefront of Kernza innovation • Dozens of dialogs with prospective end-user firms pursuing innovations in product development and marketing, linking them to technical expertise and high-quality information.
		Systemic intermediary	<ul style="list-style-type: none"> • Navigating tweaks to policy regimes to better incorporate novel crops like Kernza

(Continued)

TABLE 2 (Continued)

Concept	Definition	Sub-concepts	Implementation/milestones
Legitimacy	The broad acceptance and wide adoption of Kernza Montenegro de Wit and Iles (2016) . Thick legitimacy requires the passing of credibility tests in multiple arenas, ranging from legal to scientific.	Scientific	<ul style="list-style-type: none"> • Creating Kernza meetings where practitioners and researchers can share findings and collaborate.
		Civic	<ul style="list-style-type: none"> • Developing legislated risk mitigation strategies (EECO, Conservation Stewardship Program Enhancements) • Leading development of Kernza Stewards Alliance
		Legal	<ul style="list-style-type: none"> • Implementing and managing trademarks • Seed contracting
		Social	<ul style="list-style-type: none"> • Instigating social sustainability research and sustainable supply chain evaluation
Multi-level perspective & sustainability transitions	Transitions to qualitatively different, more sustainable systems is immensely difficult and requires concerted alignment of niche and	Niche-regime interactions	<ul style="list-style-type: none"> • Development of Forever Green Partnership • Implementation of LEN • Incorporating Kernza into key cultural institutions, e.g., state and county fairs
		Landscape-regime interactions	<ul style="list-style-type: none"> • USDA Farm Service Agency certification of Kernza acres for conservation practices • Engaging with state agencies such as Soil and Water Conservation Districts
Scaling readiness	A framework for understanding, visualizing, and strategizing around the maturity of core innovations, and the many accompanying innovations needed for its success. Scaling readiness encompasses both evaluative measures that assess the readiness and use of an innovation and methodologies or that result in adoption, niche and regime change, and have implications for legitimacy.	Innovation packages	<ul style="list-style-type: none"> • Used as a framework to assess major weaknesses in overall early commercial Kernza ecosystem, like seed shortages, and develop rapid solutions

“join us on the journey” framing across early partners helped build tolerance for working through hurdles together and embracing a long-game, collaborative approach toward perennial agriculture.

Still, in our opinion there remained a general underestimation of the type of capacity and investment needed to commercialize a novel perennial grain. A cross-scale, cross-sector ecosystem of actors was needed to guide commercial activity in tandem with germplasm development, agronomic best practices, farmer support, processing R&D, product development, consumer awareness, and more. In response, key institutions developing Kernza implemented a strategy for the development of a multi-site commercialization team by late 2018, whose subsequent work is detailed across the practice narratives in this article. In turn, these narratives highlight the utility of certain theories that provide conceptual guidance to this commercialization work.

2. Conceptual review and practice narratives

Commercializing and stewarding novel CLC crops and systems such as Kernza is a fundamentally pragmatic endeavor and thus engages with knowledge and ideas to the extent that they can enable successful action ([Zolfagharian et al., 2019](#)). This paper reflects the pragmatic process, outlining the various theories that contextualize the commercialization and stewardship team's efforts to build Kernza's

commercial development. Methodologically, this is described as “following the problem” with whichever approaches work. It is the complex and untidy work of bridging theory into action. Concepts from the fields of technology transfer and adoption, innovation management and brokering, intermediaries, sustainability transitions, multi-level perspective, legitimacy, and scaling readiness are relevant to understanding the nature of commercializing novel CLC crops and cropping systems and designing practical approaches to advance this practice ([Table 1](#)). An overview of these concepts is interwoven with pertinent reflection on the practice of novel CLC crop commercialization and stewardship staff through a narrative case study of key activities on Kernza commercialization from 2019–2022. This interweaving illustrates how conceptual frames have proven relevant in practice, and notes areas in need of further conceptual development, in light of our practical experience. While hundreds of individuals and entities have contributed to Kernza's early commercial development, only authoring entities are named to respect the confidentiality and potentially varying perspectives of these many other stakeholders.

2.1. Technology transfer and adoption in CLC

Before new agricultural technologies can be scaled, they must be successfully adopted. Prior to being adopted, these technologies

must be transferred. Thus, any entity seeking to develop and scale a technology such as CLC crops must, at a minimum, have effective technology transfer and adoption strategies. Too often, the opportunity and promises of innovation and scale obfuscate the detailed, nuanced, specific work of effective technology transfer and adoption that necessarily precedes achieving larger impact.

Molnar and Jolly (1988) define technology transfer as the sharing or introduction of a technology followed by the spread or expanded utilization of the new technology, generally proceeding from the central points to the periphery. Technology transfer is explicitly a multi-level process of communication involving a variety of senders and receivers of ideas and materials. Moreover, community absorption of new technology involves significant selection or modification in the course of adaptation to local conditions and preferences.

Technology transfer in agriculture has been closely studied for nearly a hundred years. Comprehensive reviews of technology transfer between universities, industry, and society detail its many challenges and characteristics (Hoenen et al., 2018). The nature of agriculture presents numerous challenges to technology transfer, including protracted timelines, the need for regional adaptation of crop varieties and cropping systems, and complications due to weather, soil type, pests, equipment, management, and markets. Technology transfer is a nuanced, layered process that extends well beyond patenting and licensing. It is also influenced by grower attitudes and resources, industry and university fields, inventor motivations, firm characteristics and culture, the structure of cross-sector collaboration, and staffing (*Ibid*). Cramb (2000) notes, “successful adoption depends on more than careful planning in research and the use of appropriate methodologies in extension. It depends on the timely formation of coalitions of key actors whose interests converge sufficiently that they can focus their resources and efforts on achieving change in agricultural systems.” While this article focuses primarily on commercialization staff’s roles and activities, the importance of these key actor coalitions across growers, supply chain actors, and end-users cannot be overstated and deserve subsequent inquiry in their own right. Recent research emphasizes the role of agricultural scientists as well in the political work of constructive collective action to address grand challenges, such as those targeted by CLC crops and systems (Jordan et al., 2021).

Studies of technology transfer have dispelled simple unidirectional processes (Schmoch et al., 1997), transfer of new technologies free from the need for complementary innovations (Sartas et al., 2020), and highlighted that new technologies are bound up with social and institutional processes. The field of technology transfer and related critique led to subsequent conceptual development of the socio-technical system (Geels, 2004) and more recently scaling readiness (Sartas et al., 2020). The classical notion of technology transfer has been criticized as inadequate for understanding the sources and solutions to increasingly complex contemporary problems, giving way to understanding of agricultural innovation systems and “intermediaries” as innovation facilitators and brokers—concepts introduced in subsequent sections (Koutsouris, 2018). Despite these criticisms, the concept of technology transfer can be helpful for highlighting the specific activities of CLC commercialization staff at the point of technology ‘handoff,’ details which are at risk of being lost in more complex theoretical framing.

Barriers to adopting agricultural conservation practices, including living cover crops, are well documented (Roesch-McNally et al., 2018;

Prokopy et al., 2019). These findings and the associated strategies for overcoming the barriers, such as technical assistance (Peters et al., 2021), can reasonably be assumed to extend to other CLC crops, though more research is warranted. Practical barriers to the adoption of conservation practices in US agriculture include farmland lease terms and rental dynamics, partial information, cognitive and interpersonal factors, and financial concerns. These barriers vary by actor in the agricultural system, such as non-operating landowners versus operators (Ranjan et al., 2019) and relative to gender (Carter, 2019). Field tours, or field days, can be an effective strategy to support grower adoption, though the design of such projects and attendee characteristics are important factors in shaping new technology adoption (Forte-Gardner et al., 2004).

Since 2019, a substantial portion of Kernza commercialization staff’s activities have focused on detailed technology transfer and adoption strategies for Kernza perennial grain. To do this, they interface closely with researchers, growers, industry, university technology transfer office staff, agricultural utilization experts, community partners, and others. Between 2019 and 2022, the primary strategies to support technology transfer and adoption among Kernza commercialization staff included: (1) Intellectual property, variety releases, and licensing, (2) management of a commercial trademark, (3) transfer of Kernza seed and grain to support technology adoption, and (4) educational forums, programming, resources, and dialog, and (5) de-risking support for growers and supply chain actors. To date, recruitment of growers has not been the target of a technology transfer strategy because numerous growers are interested in Kernza and recruitment has not been a limiting factor.

Perhaps because of this, little research has focused on factors informing adoption of Kernza. Lanker et al. (2020) conducted 10 in-depth interviews with early Kernza growers in 2017, finding that all were interested in the economic and ecological benefits of Kernza and had a positive attitude toward experimentation and new practices. They also found that early adopters reduced risk and cost to their operation by utilizing marginal land and resources. Growers cited the need for information on production practices, forage value, weed management, as well as economic assessments and market information—foreshadowing the need for a robust commercialization team. Wayman et al. (2019) found that across the United States and France, potential Kernza growers’ interest was motivated by both farm profitability and soil health.

Cross sector coordination of intellectual property and licensing strategies was most evident leading up to and following UMN’s release of the first commercial Kernza variety, MN-Clearwater between 2019 and 2022. Prior to the 2019 release, newly hired commercialization staff organized disparate stakeholders to accelerate UMN toward a release. This entailed development of internal and external communications strategies targeting early-adopter growers and stakeholders, open conference calls between the crop R&D team, early adopter growers, and other stakeholders to develop relationships and build knowledge, and physical transfer of seed. Concurrent to the release, commercialization staff coordinated with growers and researchers on variety increase lots sown around Minnesota in conjunction with water quality trials. This included post-harvest management logistics, seed and grain testing protocol, and on-boarding a new Kernza seed cleaner. In 2020, due to the disruptions of the global pandemic UMN had still not licensed the variety to actors in the marketplace and so commercialization staff filled this

critical gap by collaborating with TLI to vet and approve growers, to execute MN-Clearwater seed sales directly to growers in partnership with the state crop improvement association, and to fulfill orders via the seed cleaner. Over 10,000 lbs. of seed were sold through this fragmented yet functional model, with roughly 1,000 acres planted in 2020, mostly in Minnesota. This was then the largest concentrated regional planting of Kernza, and roughly a five-fold increase of existing production in Minnesota.

In winter of 2020, commercialization staff developed UMN's strategy for time-delimited (four-year) co-exclusive licenses to three regional seed companies. A co-exclusive model offered growers options as well as the right balance between protection and competition for licensees. Regional seed companies were chosen because of their proximity to the Kernza research community in the Upper Midwest. Commercialization staff regularly work with licensed seed companies to promote Kernza to their customers and to troubleshoot seed supply regulations, lot certifications, and other issues. For example, in 2022 commercialization staff aggregated market information, identifying a likely seed shortage, and promoted strategies to mitigate this shortage.

The Kernza trademark, established in 2013 as a mechanism to protect the novel perennial grain in the marketplace from cooptation or dilution, has been another key tool for facilitating the technology transfer and adoption process. The Kernza trademark's benefits to include the ability to rigorously vet grower and industry partners, differentiate Kernza in the marketplace, build consumer awareness, ensure quality, regulate nefarious actors, gather market data, and build shared identity among private actors across the value chain. Downsides include additional paperwork, time and cost to manage and administer the trademark, additional nuance in achieving policy support, and Kernza stakeholders' perceptions and/or misunderstandings on the nature of trademarks. Since 2013, TLI has owned and managed the Kernza trademark, increasingly opening that process to key partners such as UMN in 2019. In 2019, the newly formed TLI-UMN commercialization team developed a transparent and consistent process of grower vetting and trademark licensing to better organize the technology adoption process, boost legitimacy, maximize chances of success in early production, and improve the resulting grain quality and overall integrity of Kernza in the marketplace. Stakeholders shared that prior to 2019 there was a real or perceived situation in which accessing Kernza seed was a murky, exclusive, or unclear process and that only the most well-connected, lucky, or persistent growers were able to access seed. Since 2019, this process has been implemented as consistently as possible to boost transparency, reduce favoritism, and pursue fairness in technology deployment while also stating institutional and organizational priorities of, for example, adoption in particular geographic regions. Trademark vetting criteria for growers are based on practical considerations of analogous experience, appropriate equipment, scale, support, and readiness. Annually, commercialization staff revisit grower vetting and licensing priorities, adapting as appropriate, and consistently communicate these priorities to the grower community. For example, in 2021 vetting guidelines were made significantly more flexible in an effort to widen accessibility and engagement with Kernza of producers with different priorities, scales, backgrounds, and experience. Since this grower application system was instituted, over 300 growers have applied for a Kernza trademark license. Notably, roughly 80% of growers applying have not been approved at first due

to lack of alignment, capacity, equipment, experience, location, or other factors. While somewhat restraining rapid scaling of production, this process has established standards, transparency, and consistency for adoption of Kernza perennial grain. Moving forward, TLI is considering unique forms of steward ownership to transfer the ownership and management of the Kernza trademark to licensed Kernza growers, handlers, distributors, and makers.

Commercialization staff's third main technology transfer strategy has been physical transfer of bin-run (hull-on) seed, de-hulled grain, flour, and other Kernza ingredients. This process began highly informally with university pick-ups and parking lot hand-offs, and has since grown into a multi-partner process involving request intakes, execution of material transfer agreements (MTAs), fulfillment of sample requests by university and nonprofit partners, and, increasingly, an uptake of this process by market partners as a means of customer relations and market development. Since 2019, thousands of pounds of sample Kernza perennial grain have been transferred for experimentation in cleaning, processing, milling, sifting, brewing, distilling, baking, feed trials, and other food and non-food product development activities. Such transfers help potential partners move forward with Kernza while physically stitching together sustainability transition relationships and processes across sectors. The physical transfer process requires consistent and clean communication, legal support for MTAs, small-scale food grade cleaning equipment and storage, packaging, and fulfillment. Receiving entities often require technical support from food scientists and other entrepreneurs, requiring a degree of cooperation across sectors and in some cases direct competitors. This experience suggests that a system for physical distribution of sample grains is neither quick, simple, cheap, nor easy, and is fundamentally collaborative.

The fourth main technology transfer strategy between 2019 and 2022 for Kernza perennial grain has been educational forums, programming, resources, and dialog. Tactics include: (1) formal and informal cross-sector partnerships with researchers, industry, growers, and entrepreneurs, (2) development and dissemination of technical information and support to growers, processors, and end-users, and (3) winter call series, summer field days, and increasingly visible public events. Foundational resources include a winter series of annual documents and associated call-series and/or in-person events. Annual documents lay out the state of Kernza, institutional priorities for the coming year, how to become a Kernza grower, and how to access seed, technical support available, and other resources. An accompanying annual call series, initially oriented toward growers, was started in the summer of 2019 as MN-Clearwater was poised for commercial release. These were structured as relatively open conference calls between UMN and TLI Kernza breeders, agronomists, environmental scientists, and early adopter growers. These conversations helped to build trust among early adopter growers and institutional actors.

In early 2020, UMN again hosted a series of conference calls with Minnesota growers who either had Kernza growing on their farms or were interested in growing Kernza. These calls continued informally between UMN and early-adopter growers, which created a runway for growers to move from curious participants to engaged leaders. The calls formed a foundation of communication among early adopter Kernza producers in the region, which growers subsequently took the lead on, not long after forming a producer-owned and led cooperative. Growers began taking on peer-to-peer technical assistance and new grower mentor roles. A state-supported Kernza technical assistance

team was formed soon after with coop leadership, community partners, a part-time specialist, and university and NGO researchers. Taken together, these practices provided foundational resources for growers to successfully plant, harvest and market Kernza, but also created a feedback loop between growers and researchers built on trust and clear communication. This vignette demonstrates how well-organized technology transfer strategies can foster successful technology adoption, subsequent grower-led diffusion, and commercial activity.

A Minnesota-based agricultural utilization entity has been a critical partner in the development of technical specifications and associated resources regarding processing, food science, forage and co-product uses, business development, and other applications. These efforts have been critical to effective technology transfer and adoption across the value-chain. Several phases of critical support from a State of Minnesota legislative commission helped weave together development of Kernza's agronomic management, water quality impacts, proof-of-concept pilot commercial production, and baseline agricultural utilization information. Technical resources developed through these projects are publicly-available and provide important, often more rare support for technology adoption among value chain actors seeking to clean, dehull, mill, malt, brew, bake, or otherwise utilize Kernza. Their role and impact suggests that technology transfer support must extend well beyond the farmgate in order for markets to develop, thereby supporting grower uptake.

Finally, the fifth main strategy for supporting technology adoption of Kernza perennial grain has been several channels of de-risking support. The first channel deploys support from State of Minnesota to early adopter Kernza growers to maximize the chances of commercial success and protect water quality in areas with vulnerable and/or impaired drinking water. These regions have been dubbed Economic and Environmental Clusters of Opportunity, or EECOs, with the goal of concentrating production in these areas to maximize environmental benefit, achieve economic efficiencies, and foster regional innovation and community leadership. This Forever Green EECO Implementation program provides both an environmental benefit payment and risk management payment in the event of losses taken on-farm or in the market, coupled with a diverse technical assistance team, seed and grain quality testing services, and targeted supplies and equipment. A forthcoming channel of support results from a 2021 policy initiative. This initiative is prototyping a new CLC value-chain development fund that supports entrepreneurs and businesses beyond the farmgate to adopt and/or scale-up their work with Kernza perennial grain and several other leading-edge CLC crops and systems. The initiative was funded by the Minnesota state government because of well-organized advocacy by a coalition of CLC-focused entrepreneurs and a separate coalition of environmental advocacy groups, supported by UMN and the Minnesota Department of Agriculture.

This overview of the commercialization team's technology transfer activities confirms the well-established definition of technology transfer as multi-level, multi-actor, and multi-directional. Technology transfer of a novel CLC crop is ongoing and occurs across the value chain. The technology transfer and adoption process requires trusting relationships, well-organized teams able to traverse a wide range of stakeholders, topics, and skills, technical knowledge or the ability to marshal it where needed, the design and execution of educational and outreach resources, and the development of strategic and transparent frameworks to guide important commercialization processes. These

frameworks include intellectual property and variety release strategies, sample grain provision to end-users, development and management of seed and grain quality testing systems, strategic deployment of grain reserves, management of trademark and identity preserved programs where applicable, ongoing cross-sector partnerships, and outreach and engagement strategies. Before anyone can reasonably think about scaling, the above must happen while navigating the many standard challenges of agriculture as well as the new hurdles of bringing a paradigm-shifting, regime-challenging novel CLC crop to market. While the initial technology transfer process may be considered over at some point, commercialization staff expect that these technology transfer strategies will roll directly into the longer-term processes of innovation management and sustainability transitions, discussed later in the paper.

2.2. The role of innovation in the transition to CLC: definition, trajectories, rhythms, management, and regional systems

Understanding the nature of *innovation* better equips commercialization staff to facilitate its acceleration and anticipate the likely impacts of deploying novel CLC crops and systems. Similar to technology transfer and adoption, innovation has been a major topic of inquiry in economics and business since the early 20th century. Joseph Schumpeter –well-known for his consideration of ‘creative destruction’– was concerned with the role of innovation as spurring major transitions in economic development and society at large. [Perez \(2010\)](#) summarizes, “Schumpeter strongly distinguished innovation, seen as the commercial introduction of a new product or a ‘new combination,’ from invention, which belongs to the realm of science and technology,” and further distinguishing that, “The meaningful space in which technical change in society needs to be studied, therefore, is that of innovation, at the convergence of technology, the economy, and the socio-institutional context.” Innovation can occur in business structure, products, processes, branding, and other dimensions. Common characteristics of innovation include being interactive, located, a learning and integrative process, often or largely non-technical, social, cultural, and based on creative destruction ([Romanowski, 2019](#)). This distinction between invention and innovation highlights the gulf between the scientific development of a perennial grass into a grain crop (invention) and all that is required in physical processing, product, business, and market development, marketing, and effectively positioning this package in the socio-institutional environment to meet evolving consumer, industry, and public priorities (innovation). This gray space between invention and innovation is where commercialization staff call home. Crucially, commercialization staff are primarily facilitators of others across the food and agricultural system catalyzing innovation around novel CLC crops. With any success, the transition to CLC will be characterized by innovation, entrepreneurship, and creativity among actors outside the research enterprise that will take novel CLC technologies on-farm, to the market, into policy arenas, and to-scale.

In the latter 20th and early 21st centuries, the field of neo-Schumpeterian economics generated rich insights into the nature of innovation processes. Its researchers identified the ways in which entrepreneurship and innovation create dynamic and uncertain environments in which, “the set of possibilities itself is subject to

unexpected change,” through which, “more complex modes of behavior which include ‘potential surprises’ become relevant.” (Hanusch and Pyka, 2007). The field readily recognizes that the most visible type of innovation, that of technological innovation and change, is intimately bound up with organizational, institutional, and social innovation. Innovations tend to, “not only modify the business space, but also the institutional context and even the culture in which they occur” (Perez, 2010). This is particularly relevant to the innovations in policy needed for CLC crops, which span notions of productive agriculture and agricultural conservation, and the cultural change needed for perennializing 10,000 years of heretofore annual row crop agriculture.

Neo-Schumpeterians assert that the process of introducing innovations is decidedly nonlinear, proposing logistic (S-curve) “innovation rhythms” in which initial slow innovation reflects interlinked actors’ iterative learning processes. The emergence of dominant designs lead to cascading changes and scaling, followed by a slow-down at an innovation’s maturation and saturation. Moreover, these innovation rhythms develop one or more “trajectories” in large possibility spaces in which uses, standards, relative costs, accompanying practices, and market acceptance are defined. These trajectories are defined as incremental innovations that build on original radical innovations. These concepts provide insight into the fundamental uncertainties and rhythms of dynamic change across modes of innovation that are likely to occur as a novel CLC crop with transformative potential makes headway. While much focus in the Kernza and CLC community focuses on the scientific development of new and improved crops and associated knowledge, those concerned with innovation might put their locus of study on the ways in which these inventions and new knowledge find footing in the world in the form of viable products, businesses, and new value propositions, and how that innovation in turn informs institutional, public, firm, and consumer priorities, assumptions, and possibilities.

Innovation management can influence firms’ and institutions’ competitiveness and success. Within the context of TLI and UMN, innovation management may be understood as the wide range of activities that occur at the intersection of portfolio-level and crop-specific interfacing between basic researchers, commercialization staff, and novel CLC crop early adopters and stakeholders. Seven common dimensions of innovation management include inputs management, knowledge management, innovation strategy, organizational culture and structure, portfolio management, project management, and commercialization (Adams et al., 2006). Innovation management models have been conceived as technology (push), market pull, coupling, integrated, networking, open innovation, and open innovator (Romanowski, 2019). Kline’s (1986) “chain-linked” integrated model reflects the iterative, dialectical, almost folding nature of innovation management in which CLC commercialization staff cross-walk the research and commercial environment with timely information, new knowledge, opportunities, and resources.

Innovation systems are, “interlinked sets of people, processes, assets, and social institutions that enable the introduction and scaling of new ideas, products, services, and solutions capable of facilitating impact” (Thiele et al., 2022). The notion of innovation systems coupled with regional sciences developed the concept of regional innovation systems (López-Rubio et al., 2020), which focus on the interdependencies between regionally co-located firms, human capital, context, institutions, networks, and other inter-relationships

and the potential positive externalities thereof. The regional innovation systems lens suggests that the entities developing novel CLC crops may benefit from strategically deploying such crops in concentrated geographic areas as a means of reducing transaction costs, finding efficiencies, and spurring innovation.

In practice, innovation and related concepts are ever-present in the process of commercializing novel CLC crops. The work is full of newness—new observations, problems, challenges, work-arounds, uses, products, value propositions, partnerships, policies, and cascading impacts. The enthusiasm and drive of entrepreneurs on-farm, in grain processing facilities, kitchens, breweries, bakeries, and food companies are critical forces needed to transform CLC inventions into CLC innovations. Shepherding a fundamentally innovative process, CLC commercialization staff are constantly instigating, fostering, communicating, and adapting to innovation in their work. A focus on innovation requires paying close attention to the details of nuanced processes such as grain harvest, post-harvest management, seed cleaning and processing, milling, malting, brewing, baking, and marketing. Experimentation, iteration, and sometimes accidents lead to valuable insights and innovations. Such processes often create closely-guarded innovations that provide an edge or differentiation to actors in the marketplace whereas others are shared widely, creating spillover effects that catalyze advancement of the wider enterprise. Rarely are commercialization staff the ones developing these innovations, but often they are the actors, communicating, and adapting CLC communities to the impacts of innovation.

The innovation process surrounding Kernza perennial grain reflects many of the principles of neo-Schumpeterian economics. For example, the actions of Kernza growers, entrepreneurs, and buyers regularly invoke dynamic and uncertain pathways for Kernza’s role in the market as well as the organizational, institutional, and social environments in which this novel grain and its formative value proposition is taking root. Debates during the price discovery phase have highlighted varying innovation trajectories for Kernza as, alternatively, a high-value non-commodity grain that substantially supports growers and rural communities in the transition to organic and regenerative organic agriculture; a widely-consumed and modestly more affordable climate-smart food used at higher inclusion rates in products; or a scalable market-driven tool for water quality protection. None of these three trajectories are mutually-exclusive, but all have implications for involved actors. Similarly, start-up businesses focused on Kernza are closely considering strategies for relationship development with customers, customer engagement in product design, grain-based product bundles, and innovations intended to circumvent relatively non-transparent, extractive grocery distribution supply chains. State and federal policymakers are recognizing a number of ways public programs and investments may require reform and innovation to account for perennial grains and other novel market-based CLC crops. At the cultural level, Kernza entrepreneurs, consumers, and champions are beginning to ask what a perennial grain economy and society might look like. Where such ideas lead no one quite yet knows. All such examples indicate shifting innovation trajectories, possibility sets, and technical innovations with the potential to domino into social and cultural innovations. These developments accord with contemporary understanding of innovation as a multi-faceted process in which technical, social, cultural and organizational innovations cohere in “new effective combinations”

that enable technical innovations to achieve scale and societal impact (Leeuwis and Aarts, 2011; Herrero et al., 2020).

CLC commercialization and stewardship staff's role in this process is to observe and articulate emerging innovation rhythms and trajectories, mirroring back to stakeholders enmeshed in the innovation process the larger arc, potential pathways, and their dynamic effects on the enterprise as a whole. Between 2019 and 2022 this has been accomplished for Kernza through detailed narrative documents provided by UMN FGI written for Kernza stakeholders at large, as well as integrated call series and webinars hosted by these staff that, through their design and execution tell the story of Kernza's emerging innovation trajectories in its stakeholders own voices. Commercialization staff observe if not anticipate inflection points in innovation rhythms, messaging to stakeholders and the public as appropriate. Bending the arc on these trajectories may be important for retention of critical partners, staving off consolidation of power and resources by single actors or supply chain segments, or otherwise maintaining the pursuit of public benefits of the novel CLC crop.

Kernza commercialization staff must construct practical innovation management strategies between crop R&D teams, commercial interests, and other stakeholders to communicate the latest learnings, needs, and challenges in multiple directions. Examples in Kernza from 2019–2022 include close collaboration with growers, processors, and researchers to document and communicate various harvest methods, mycotoxin levels, the impact of processing (dehulling) on mycotoxin levels, cleanout rates, and test weights. These early learnings were generated through informal collaboration over several years by early adopter growers, start-up partners, and researchers seeking to set a common baseline understanding of working with this new novel CLC crop. Winter call series and events convening growers, researchers, value chain actors, policymakers, and community partners are a key programmatic mechanism for innovation management. In 2021 and 2022, the grower-researcher call series, designed primarily for technology transfer and adoption with growers, expanded to a strategic integrated presentation on the status of Kernza to a wider range of Kernza stakeholders. What was previously self-directed by market actors through piecemeal informal collaboration is coalescing into systematized innovation management processes. For example, in the ensuing years, UMN Kernza breeders, food scientists, and commercialization staff have designed a project to streamline collaboration with Kernza growers, agricultural utilization partners, processors, and food companies to systematically assess needed alterations to harvesting, seed cleaning, processing, milling, and sifting in response to germplasm improvement. Similar to the way growers steward their fields and researchers steward their labs, these are examples of commercialization staff's role stewarding the innovation process. Finally, the literature on innovation management suggests that agricultural innovation platforms developing novel CLC crops have significant room to grow in articulating and implementing explicit innovation management strategies (Biggs et al., 2012).

The Upper Midwestern US, and specifically Minnesota, stands out to Kernza commercialization staff as an active regional innovation system for Kernza perennial grain. It is characterized by relatively tight geographic presence of many if not all of the types of actors necessitated to innovate in grain systems: breeders, agronomists, natural resource scientists, food scientists, growers, farmer groups and agricultural nonprofits, processors, millers, brewers, food companies of all scales, funders, investors, state support, engaged policymakers,

consumers, and communities. This regional innovation ecosystem is no accident, it has been intentionally cultivated over several decades by institutional actors at UMN including the Forever Green Initiative, Green Lands Blue Waters, the Regional Sustainable Development Partnerships, the Minnesota Institute for Sustainable Agriculture, not to mention the well-known presence of strong social capital and civic engagement in the state of Minnesota. While Minnesota may not have the greatest climatic comparative advantage for producing Kernza perennial grain, comparative success there warrants subsequent research on the degree to which, for example, strong bridging social capital may greatly accelerate CLC crop innovation. Cultivating and maintaining a regional innovation system around novel CLC crops requires constant support from commercialization staff. For Kernza between 2019 and 2022, this has included annual strategic communications to frame regional success, progress, bottlenecks, and priorities; regular communication and coordination to support key partnerships like an emerging cooperative; onboarding and incorporation of new entrepreneurial energy; and transparency, accountability, and self-awareness to steward the system as a whole rather than choosing favored actors.

The benefits of a regional innovation system are highlighted by the experience of Kernza adoption in Wisconsin, where local commercialization intermediaries identify the lack of such a system. Wisconsin neighbors Minnesota and because of similar latitude, topography, and soil history, agriculture is generally deemed quite similar between the two states. However, Kernza adoption and production has diverged considerably among Minnesota and Wisconsin. While Minnesota is currently home to nearly 1,300 acres of Kernza situated on over 40 farms, just three growers are actively growing Kernza in Wisconsin on a total of just over 200 acres. Despite Wisconsin having a perennial crops program, university researchers with a history of collaboration with TLI, experience in sustainability innovation in farm cooperatives and organic production, and a relatively high degree of farm diversity compared to other states, this has not yet been enough to spur significant Kernza adoption.

Wisconsin commercialization staff feel that Wisconsin has thus far lacked the institutional commitment, civic and public support, and investment to support a regional innovation system. In comparison, Minnesota's state and other investments stimulated a wide variety of projects and partnering organizations, including university-municipal partnerships to deploy Kernza for its environmental benefits, university-farmer collaborations, civic-sector support, and private business startups to support commercialization efforts. The comparable lack of funding in Wisconsin stymied Kernza's pre-adoption pipeline, as university researchers were restricted to relying on federal grant programs to support agronomic and extension work. Even fewer resources are available to support Kernza's integration into business supply chains. Wisconsin's smaller grower community has identified significant challenges accessing the seed supply and post-harvest processing options predominantly located in or adjacent to Minnesota. A takeaway lesson has been that investing in post-harvest processing, infrastructure, and grain handling recommendations, and building human capital and desire to refine grain post-harvest, is just as important as developing agronomic research and production guidelines, which is often where investment is directed early on with new crops. This results in imperfect options: risk holding grain until localized facilities emerge or erode profit by shipping longer distances. The relative lack of grower adoption has

created compounding challenges, such as insufficient grain volume to interest potential cleaners, processors, and end users to perform tests with Kernza, ultimately impacting Wisconsin Kernza sales. In the absence of state support, growers assume the entire risk out of a devotion to environmental protection and investment in improving the impact of their farming operations. The dedication of these individuals, like those in MN, cannot be overstated.

In sum, Wisconsin serves as an example of a region where enthusiasm from some researchers, growers, supply chain actors and advocates has lacked sufficient support to develop a regional innovation system, hindering the regional adoption and scaling of this novel CLC crop. By comparison, Minnesota demonstrates that well-supported, concentrated, dynamic innovation systems can accelerate commercial development, reduce transaction costs, and de-risk adoption, and that such activity provides positive externalities for wider actors.

2.3. Intermediaries and innovation brokers

CLC commercialization staff can be understood to operate as intermediaries in sustainable agriculture innovation systems. The concept of intermediaries arises from a growing body of literature that highlights the particular importance of intermediary actors in facilitating transitions to more sustainable systems (Moss, 2009; Steyaert et al., 2016; Mignon and Kanda, 2018; Kivimaa et al., 2020). Intermediaries are thought to positively influence sustainability initiatives by linking diverse entities and their related resources and skills, creating new collaborations across niche technologies like Kernza, linking technologies to markets, and generally creating momentum for system change (Kivimaa et al., 2019). Others have underscored the importance of intermediaries in brokering and transferring knowledge, aggregating lessons, and mobilizing resources (Klerkx and Leeuwis, 2009; Goodrich et al., 2020; Kanda et al., 2020). Despite the various roles ascribed to them across the literature, intermediaries are near-unanimously defined by their ability to span boundaries (Bergek, 2020), be it across actors, networks, institutions, spatial extents, or scales.

There is a general understanding that a full ecology of intermediaries, from those that operate on a systemic policy level to those who support particular niche technologies, is needed to support a transition process and that the network of intermediaries shifts over time. Given the emergent and uncertain change processes in scaling new technologies and systems, intermediaries can act in conflicting roles barring sufficient coordination (Kanda et al., 2020). The mounting body of evidence suggests that intermediaries and their coordination may play a critical role in a transition to CLC agriculture. Additionally, given transition intermediaries' normative orientation toward change, they can never be fully neutral actors (Moss, 2009; Steyaert et al., 2016), and as such, intermediaries must recognize their power to be both a guide and a gatekeeper to various entities (Sovacool et al., 2020).

More specifically, Kernza commercialization staff often operate as innovation brokers, a particular type of intermediary that, "from a relatively impartial third-party position, purposefully catalyze innovation through bringing together actors and facilitating their interaction" (Klerkx and Gildemacher, 2012). These actors institutionalize facilitation of innovation systems for system-level

impact, expanding the nature of extension activities from one-to-one to many-to-many. Common functions of these actors are to analyze context, articulate demand, compose networks, and facilitate interaction. A typology of innovation brokers spans innovation consultants working with individual or groups of agricultural producers and enterprises, peer network brokers, research and innovation councils, and several others. The potential impact of innovation brokers is significant, but their 'ghost in the machine' nature often leaves their role poorly understood by funders and innovation system stakeholders.

Brokering innovation is decidedly different from other key systemic intermediary activities of commercialization staff such as navigating important but largely technocratic tweaks to policy regimes to better incorporate novel CLC crops. It often involves brokering resources—physical, financial, relationships, information, or otherwise—to spur commercial activity and entrepreneurship, indicating that theories of intermediation have evolved from the seminal field of technology transfer. For example, an early reserve of grain from state-supported water quality trials was, not by accident, provided by Kernza innovation brokers as 'start-up grain' to a team of Kernza entrepreneurs, which helped them launch a business that continues to be on the forefront of Kernza innovation. This entailed discernment of potential system-level impact, and targeted brokering of physical assets. Dozens of such small and large examples exist in which commercialization staff benevolently broker interests, skills, expertise, information, resources, and access to novel crop technologies with disparate actors. A number of activities previously discussed in the technology transfer section and otherwise could, when taken together with considerations of innovation processes, be recast as workflows in the milieu of innovation brokering.

2.4. Legitimacy

For new crops and cropping systems, developing authoritative knowledge and building systems for its acceptance is an arduous process. Montenegro de Wit and Iles (2016) discuss how, even after decades of concerted and organized effort, the organic movement has attained only 'thin legitimacy' based primarily on market demand and policy intervention. Both of these are important, but for new crop adoption to happen on a temporal and spatial scale that can catalyze meaningful change, legitimacy must be expanded and built on additional, credible, and authoritative processes, dubbed 'thick legitimacy'. Some of these processes include drawing in consumers and companies, enacting government rules that recognize or support the transfer and adoption of new crops and cropping systems, increasing scientific interest and the number and types of research that are happening, and attracting farmers to the new system (*ibid*).

In practice, commercialization and stewardship staff can bend the arc of legitimacy, but rarely are its primary determinants. Legitimacy is achieved first and foremost through the viability of a technology to perform on farms and in products, and its ability to achieve given societal outcomes. The series of actors needed to adopt, prove, promote, plant, and purchase the novel CLC crop are integral to achieving legitimacy. While somewhat more removed from the direct enterprise, policymakers and agencies are essential actors in advancing or hindering novel CLC crop legitimacy. As discussed below with regard to the Multi-Level Perspective (MLP), achieving legitimacy is

contingent on the times as well as the technology. For example, Kernza is much better positioned to achieve legitimacy in the rise of the regenerative agriculture movement and age of climate instability than it would have been 50 years prior in a yield-centric paradigm.

Commercialization and stewardship staff working to build legitimacy for Kernza have acted to stabilize areas in which various actors have made inroads to authority. In the scientific arena, we have worked to create spaces for scientific sharing and collaboration, including organizing and hosting two international Kernza meetings in collaboration with other scientists from 2019–2022. These spaces build community, but also present criticisms, identify missing areas of research, and build strategies for addressing these. Additionally, they are a space to discuss civic engagement, policy and advocacy needs, and create plans to obtain funding.

From these Kernza discussions, it is clear that two critical areas of research are social sustainability and ecosystem service payments for farmers. The addition of social sustainability research is critical to advancing Kernza in the marketplace. One definition of social sustainability is when, “people are not subject to conditions that systematically undermine their capacity to meet their needs” (Missimer et al., 2017). New research on how and whether Kernza supply chain actors are engaging in socially sustainable practices is ongoing. The aforementioned Forever Green EECO Implementation Program is an example of legitimacy being built for both Kernza perennial grain and ecosystem service payments via state government entities endorsing and funding such programs. Additionally, Kernza has seen recent successes as perennial grains have been added to the NRCS’s Conservation Stewardship Program Enhancements for 2021 and the Farm Service Agency has begun to allow growers to certify IWG as a grain crop for data collection and whole farm insurance purposes, opening a window to further support by USDA programs and continue building legitimacy.

Kernza commercialization and stewardship staff have made inroads in developing civic legitimacy by leading the early development of a Kernza Stewards Alliance (KSA) composed of supply chain stakeholders, from growers to food product manufacturers, that is to some extent modeled on other commodity organizations designed to promote products. However, unlike other commodity organizations, the KSA is moving toward a steward-owned model that will create mechanisms to shift power out of the hands of institutions, such as TLI and UMN, and into the hands of supply chain stakeholders by transferring ownership and governance of the Kernza trademark to its licensees. UMN and TLI will maintain a voice through a perpetual purpose trust, a body committed to the long-term benefits of Kernza perennial grain. The engagement, involvement, and enthusiasm of these actors for this process and its goals is a demonstration of legitimacy. However, the lack of precedent for the establishment of this complex entity provides a clear need for additional legal clarity and legitimacy, indicating that various modes of legitimacy are intimately bound up with one another.

2.5. Multi-level perspective and sustainability transitions

Geels’s Multi-Level perspective is a critical framework used to understand socio-technical transitions and sustainability transformations and thus provides an important basis to understand

the commercialization, adoption, and scaling of Kernza. The MLP claims that there are three critical levels in a socio-technical transition effort: niches, regimes, and a landscape (Geels, 2002; Geels, 2019). The MLP posits that stable regimes like industrial agriculture are notoriously hard to disrupt, however niche innovations that operate outside of the dominant culture have the potential to destabilize regimes if sufficient bottom-up momentum is met with top-down pressure from the landscape level. Landscape level pressure can be endogenous (e.g., major policy changes) or exogenous (e.g., pandemic, climate change). Thus, the MLP suggests that Kernza, as a niche, will need concerted alignment with the regime and landscape to open a window of opportunity to effectively establish itself. Additionally, this theory offers a four-phase understanding of a transition, demarcated into: experimentation, stabilization, diffusion and disruption, and anchoring or institutionalization; which can orient actors in transition efforts that often span multiple decades (Geels, 2019). The MLP has proved formative to studies of sustainability transitions (Köhler et al., 2019) and although it has been applied to agri-food systems, it is based primarily in a socio-technical systems framework that may not best account for socio-ecological systems, here being the ecological realities inherent to agriculture, and the unique market and decision-making structures of agricultural systems (Duru et al., 2015; El Bilali, 2020). It may prove best to engage with the MLP critically in the hopes of augmenting the framework to better describe transitions in an agricultural context.

In practice, MLP highlights that the viability of novel CLC crops is contingent on landscape changes and regime acceptability in addition to technological readiness. CLC commercialization staff must therefore focus on the regime and landscape factors as well as shepherding the niche solution. More specifically, they must situate themselves as competent interpreters and instigators of niche-regime, regime-landscape, and niche-landscape interactions, a role described as systemic intermediary above (e.g., Kivimaa et al., 2019).

Engagement of incumbent regime actors and ideas is constant, detailed, and necessary. For example, commercialization staff often lead or support methodical work to make “lateral,” technocratic inroads into highly structured agency and policy mechanisms needed to either legitimize or bring online support for a novel CLC crop or system. Several examples include working closely with USDA Farm Service Agency to allow growers to certify their IWG acres for grain production, incorporating IWG into agricultural conservation practices and incentive programs. With the first official commercial variety release in 2019 and less than 5,000 acres in production, Kernza is in an early phase of developing these support mechanisms. Achieving these milestones also often requires development of associated tools such as enterprise budgets, economic models, harvest reports, and contributing to peer-reviewed literature. As often as commercialization staff may talk to prospective growers or end-users, they are equally as likely to engage Soil Water Conservation Districts and other state agency staff. They may also engage legislators, peer organizations and institutes, academics.

Finally, commercialization staff and many other Kernza stakeholders invest significant time incorporating Kernza into key cultural events and institutions such as state and county fairs, FarmFests, museums, arboretums, school programming, and more. Since the mid-2010’s, Kernza’s presence being served by a farm-to-table restaurant in partnership with a grower advocacy group was a key regime inroad to Minnesota’s culinary and public conversation. In

Minnesota, Kernza is also now growing at both a premier science museum, with accompanying exhibit on CLC agriculture in development, as well as a premier arboretum and onsite at several schools, associated with Future Farmers of America (FFA) programs. While adjacent or parallel to core activities of developing commercial production, supply chains, and markets for Kernza, these activities require substantial interfacing with commercialization staff and build further regime acceptance.

2.6. Scaling readiness

Given overwhelming potential activities and often limited instruction, it is helpful to couch tactical commercialization actions in broader, strategic context. A recently developed framework that supports this is scaling readiness. It encompasses both evaluative measures that assess the readiness and use of an innovation or innovation package (Sartas et al., 2020) and methodologies or processes that result in adoption, niche and regime change, and have implications for legitimacy both in terms of scaling out and scaling up (Wigboldus and Leeuwis, 2013). Alternatively put, scaling readiness provides innovation brokers with a framework for understanding, visualizing, and strategizing around the maturity of core innovations, and the many accompanying innovations needed for its success. In the context of CLC and Kernza, the framing of innovation packages used by Sartas et al. (2020) is particularly useful. In this frame, the scaling of an individual innovation (e.g., Kernza perennial grain) requires the scaling of related innovations such as new varieties, seed handling and distribution best practices, harvest methods, processing infrastructure and methods, on-farm storage solutions, markets, marketing strategies, business structures, and policy strategies. This framework of scaling readiness further validates and operationalizes the aforementioned and accompanying notions and necessarily interlinked technical, economic, social, cultural, policy, and institutional innovations (Leeuwis and Aarts, 2011; Meynard et al., 2017; Herrero et al., 2020).

The scaling readiness framework has helped Kernza commercialization staff navigate their complex work of tracking the development of innovation packages and directing resources to building out components lacking in readiness or use. Specifically, the framework was used to identify an innovation system components lacking in readiness, seed supply, which created a systemic bottlenecks. The identification of the seed supply bottleneck informed intensive efforts to alleviate Kernza seed shortages and improve quality in the seed supply, as well as expand markets for Kernza perennial grain. An excellent subsequent exercise would be to more explicitly map Kernza's innovation package by readiness and use utilizing the scaling readiness methodology (Sartas et al., 2020; Schut et al., 2022) in partnership with Kernza stakeholders to identify key bottlenecks worthy of attention, support, and investment.

In particular, challenging notions of context independent scaling is important in CLC agriculture where social complexity and technical complexity are both great. Novel CLC crops are not substituting one crop or variety for another or one management practice for another. Novel CLC crops and their stakeholders are creating new systems of growing, managing, processing, distributing, creating, and valuing food. This requires an innovation package that is adaptable based on physical production and processing technologies, product

development, policy support, consumer awareness and values, and institutional mechanisms.

3. Discussion

This investigation of relevant fields in the light of our practice narratives suggests that multiple related processes are inherent in CLC commercialization staff's work such as technology transfer, intermediating between niche technologies and regime actors, innovation management at the institutional portfolio level, innovation brokering of specific novel CLC technologies across innovation systems, and building legitimacy within slowly transitioning regimes. In practice, these processes may be occurring all at once (e.g., in the same room) via capacity-delimited programming. This suggests that the multiple functions of commercialization roles need to be integrated in practice with adaptive, multifunctional, nimble staffing.

In the course of this work, CLC commercialization staff often find themselves confronted with the need to make specific choices that may shape innovation trajectories in particular directions or, alternatively, find themselves tasked with intervening in attempted plays to change innovation trajectories that significantly diverge with institutional or otherwise broadly shared narratives and values. Acting as a steward of this process rather than a gatekeeper that hinders innovation and regime transformation is a delicate dance.

Notably, the various activities of novel CLC crop commercialization staff are often highly disparate, and in some cases greatly so. For example, physically transferring a novel perennial grain crop to spur entrepreneurship and innovation is distinct from navigating local, state, and federal institutional and policy environments to generate CLC portfolio-level support at the regime level. The CLC crop or cropping system may be a throughline but otherwise the actors, goals, strategies, and cultures of such processes can vary wildly. Among all concepts reviewed, the intense experience of CLC commercialization staff members is perhaps best characterized as being simultaneous innovation brokers of novel CLC crops to their stakeholders as well as transition intermediaries navigating and surmounting the mazes, riddles, and roadblocks of regimes.

Since these many activities closely relate to but function outside the process of developing new technologies (i.e., crop varieties) and knowledge (i.e., agronomic best management practices), the scale and nature of support needed to move novel CLC crops from invention to innovation to scale across landscapes and markets may not be immediately apparent. However, the practice narratives suggest the need for coordinated action to shape and advance the adoption and scaling of novel CLC crops. Critically, the design principles of such coordinated action need to be based on responsiveness, flexibility, adaptation, and dynamism.

These concepts and practice narratives support development of a practical theory for bridging new CLC technologies into the food and agricultural sector and society at large. Layered and nuanced intermediation theories can obscure the critical, practical need for a clear technology transfer strategy with defined actors and adequate support. Similarly, institutional actors, intermediaries, and policymakers may wax poetic about innovation without providing the practical financial and staffing support to meet the significant needs of entrepreneurs to wrap their arms and minds around novel CLC crops and develop go-to-market strategies. Overall, these fields of

literature suggest that a robust if decentralized architecture of intermediation must be constructed and sustained to facilitate technology transfer of novel CLC crops, the innovation likely induced thereof, CLC crop enterprise scaling, and niche-regime-landscape interaction and transformation. Such architecture boosts the chances that CLC inventions more rapidly translate to CLC adoption, innovation, scaling, and impact.

Several important caveats are in order. First, while attempting to crystallize early learnings, commercialization staff still consider Kernza to be in its early phase of commercial development and the best practices for facilitating novel CLC crop commercialization are emergent. Second, these staff readily acknowledge that the basic research and development work on CLC crops such as Kernza precedes and continues alongside their commercial development. Long-term, continued advancements in breeding, agronomic best management, clear understanding of environmental benefits, and robust food science are key factors in Kernza perennial grain's commercial viability, and this will likely hold for most other perennial grains and oilseeds. Without significant and sustained investment in developing and improving CLC crops and cropping systems, much of what's discussed in this article is moot. At the same time, development of a new crop without investment in supply chain and markets fails to deliver the novel CLC crop's intended impact. Also, the independent actions of private actors in the market often precede or supersede the actions and relevance of commercialization staff, and indeed are necessary for novel CLC crops to move forward. In a best case scenario, commercialization staff function as integral parts of the CLC crop development enterprise, serving as stewards of the innovation process that sit between these researchers and private actors to move novel CLC crops from the lab to the field and market.

4. Conclusion

Taken together, the concepts reviewed and practice narratives indicate that a 'practical theory' of novel CLC crop commercialization:

- Is technology (crop or system) specific, and spanning many dimensions thereof
- Is built on a robust research and development platform for said technology
- Requires collaboration across disciplines, sectors, and the entire value chain
- Is likely, at least at first, regionally situated in specific geographic, social, cultural, environmental, economic, and institutional conditions
- Must understand, account for, support, and navigate multifaceted innovation processes
- Requires effective innovation management strategies at the institutional or systemic level
- Is aided by the existence and further fomenting of regional innovation systems that offer appropriate degrees of protection, risk, and dynamic interplay between stakeholders
- Requires well-supported teams of innovation brokers and other intermediaries to interface with R&D efforts and myriad novel CLC crop stakeholders
- Must be attuned to incumbent landscape pressures, regime arrangements, sustainability transitions underway therein, and

the need to consistently build legitimacy for the novel CLC crop or system within actively changing regimes

- Demands systems-level understanding of core and accompanying innovations, with strategic focus on addressing the most underdeveloped elements of innovation packages

For all the described activities, nationwide Kernza commercialization staff has consisted of a small group that notably perform similar functions for institutional portfolios of 6–10 novel or improved CLC crops and systems in addition to Kernza, resulting in thin capacity for any one crop or function. Marshaling adequate resources in the form of multiple well-qualified people, associated facilities, and institutional support prior to or in the earliest days of novel CLC crop commercialization is crucial yet challenging. Yet again, the endemic 'chicken-egg' problem in new crop commercialization may strike in which such investments are too hard to justify, given the many and good alternative uses. The innovation broker role may often find formal or informal alignment with existing institutional roles, which may come with synergistic benefits as well as drawbacks. Our conceptual review and practice narratives highlight the abundant support needed and value of investing in innovation broker capacities.

Moving forward, this article suggests that novel CLC crop commercialization activities already do and will continue to present a wealth of case studies from which to refine a practical theory of novel CLC crop commercialization. Recognizing major differences across regional innovation systems and CLC crops in their portfolios, CLC commercialization staff have begun constructing developmental frameworks for innovation packages and commercialization model typologies. Further research is warranted to explicate these ideas. Similarly, several crucial concepts omitted from this initial paper include concepts for balancing economic, environmental, and social values in the commercialization process, such as sustainable commercialization, as well as governance of novel CLC cropping systems. This suggests that novel CLC crop commercialization is not only a robust field of practice but also fertile ground for critical applied research on practical methodologies and frameworks that can support a wide range of actors to drive a rapid transition toward continuous living cover agriculture at scale.

Author contributions

CCu conceived the project, conducted literature review, and was the primary manuscript writer and editor. SS contributed to the early commercialization narrative and figure creation. TP, NT, and CCa contributed to writing practitioner narratives, AR helped refine the scope, contributed to the early commercialization narrative, and edited. TC contributed to literature review and editing. NJ helped with theoretical framing and scope refinement. All authors contributed to the article and approved the submitted version.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

References

- Adams, R., Bessant, J., and Phelps, R. (2006). Innovation management measurement: a review. *Int. J. Manag. Rev.* 8, 21–47. doi: 10.1111/j.1468-2370.2006.00119.x
- Basche, A., and DeLonge, M. (2017). The impact of continuous living cover on soil hydrologic properties: a meta-analysis. *Soil Sci. Soc. Am. J.* 81, 1179–1190. doi: 10.2136/sssaj2017.03.0077
- Bergek, A. (2020). Diffusion intermediaries: a taxonomy based on renewable electricity technology in Sweden. *Environ. Innov. Soc. Trans.* 36, 378–392. doi: 10.1016/j.eist.2019.11.004
- Berkman, E. T., and Wilson, S. M. (2021). So useful as a good theory? The practicality crisis in (social) psychological theory. *Perspect. Psychol. Sci.* 16, 864–874. doi: 10.1177/1745691620969650
- Biggs, R., Schlüter, M., Biggs, D., Bohensky, E. L., BurnSilver, S., Cundill, G., et al. (2012). Toward principles for enhancing the resilience of ecosystem services. *Annu. Rev. Environ. Resour.* 37, 421–448. doi: 10.1146/annurev-environ-051211-123836
- Carter, A. (2019). “We Don’t equal even just one man”: gender and social control in conservation adoption. *Soc. Nat. Resour.* 32, 893–910. doi: 10.1080/08941920.2019.1584657
- Cramb, R. A. (2000). “Processes influencing the successful adoption of new technologies by smallholders (no. 433-2016-33502)” in *Working with farmers: The key to the adoption of forage technologies*. ed. B. Hacker, vol. 95 (Canberra, Proceedings: Australian Centre for International Agricultural Research (ACIAR)), 11–22. doi: 10.22004/ag.econ.135365
- Crews, T. E., Carton, W., and Olsson, L. (2018). Is the future of agriculture perennial? Imperatives and opportunities to reinvent agriculture by shifting from annual monocultures to perennial polycultures. *Global Sustainability* 1:11. doi: 10.1017/sus.2018.11
- Culman, S. W., Snapp, S. S., Ollenburger, M., Basso, B., and DeHaan, L. R. (2013). Soil and water quality rapidly responds to the perennial grain Kernza wheatgrass. *Agron. J.* 105, 735–744. doi: 10.2134/agronj2012.0273
- Duru, M., Therond, O., and Fares, M. H. (2015). Designing agroecological transitions: a review. *Agron. Sustain. Dev.* 35, 1237–1257. doi: 10.1007/s13593-015-0318-x
- Eberle, C. A., Thom, M. D., Nemecek, K. T., Forcella, F., Lundgren, J. G., and Gesch, R. W., .. & Eklund, J. J. (2015). Using pennycress, camelina, and canola cash cover crops to provision pollinators. *Ind. Crop. Prod.* 75, 20–25. doi: 10.1016/j.indcrop.2015.06.026
- El Bilali, H. (2020). Transition heuristic frameworks in research on agro-food sustainability transitions. *Environ. Dev. Sustain.* 22, 1693–1728. doi: 10.1007/s10668-018-0290-0
- Forte-Gardner, O., Young, F. L., Dillman, D. A., and Carroll, M. S. (2004). Increasing the effectiveness of technology transfer for conservation cropping systems through research and field design. *Renewable Agric. Food Sys.* 19, 199–209. doi: 10.1079/RAFS200485
- Geels, F. W. (2002). Technological transitions as evolutionary reconfiguration processes: a multi-level perspective and a case-study. *Res. Policy* 31, 1257–1274. doi: 10.1016/S0048-7333(02)00062-8
- Geels, F. W. (2004). From sectoral systems of innovation to socio-technical systems: insights about dynamics and change from sociology and institutional theory. *Res. Policy* 33, 897–920. doi: 10.1016/j.respol.2004.01.015
- Geels, F. W. (2019). Socio-technical transitions to sustainability: a review of criticisms and elaborations of the multi-level perspective. *Curr. Opin. Environ. Sustain.* 39, 187–201. doi: 10.1016/j.cosust.2019.06.009
- Goodrich, K. A., Sjöström, K. D., Vaughan, C., Nichols, L., Bednarek, A., and Lemos, M. C. (2020). Who are boundary spanners and how can we support them in making knowledge more actionable in sustainability fields? *Curr. Opin. Environ. Sustain.* 42, 45–51. doi: 10.1016/j.cosust.2020.01.001
- Hanusch, H., and Pyka, A. (2007). Principles of neo-Schumpeterian economics. *Camb. J. Econ.* 31, 275–289. doi: 10.1093/cje/bel018
- Herrero, M., Thornton, P. K., Mason-D'Croz, D., Palmer, J., Benton, T. G., Bodirsky, B. L., et al. (2020). Innovation can accelerate the transition towards a sustainable food system. *Nature Food* 1, 266–272. doi: 10.1038/s43016-020-0074-1
- Hoenen, S., Kolympiris, C., Wubben, E., and Omta, O. (2018). “Technology transfer in agriculture: the case of Wageningen University,” in *From agriscience to agribusiness*. Eds. N. Kalaitzandonakes, E. G. Carayannis, E. Grigoroudis and S. Rozakis (Cham: Springer), 257–276.
- Jordan, N., Gutknecht, J., Bybee-Finley, K. A., Hunter, M., Krupnik, T. J., Pittelkow, C. M., et al. (2021). To meet grand challenges, agricultural scientists must engage in the politics of constructive collective action. *Crop Sci.* 61, 24–31. doi: 10.1002/csc2.20318
- Jungers, J. M., DeHaan, L. H., Mulla, D. J., Sheaffer, C. C., and Wyse, D. L. (2019). Reduced nitrate leaching in a perennial grain crop compared to maize in the upper Midwest, USA. *Agric. Ecosyst. Environ.* 272, 63–73. doi: 10.1016/j.agee.2018.11.007
- Kanda, W., Kuusma, M., Kivimaa, P., and Hjelm, O. (2020). Conceptualising the systemic activities of intermediaries in sustainability transitions. *Environ. Innov. Soc. Trans.* 36, 449–465. doi: 10.1016/j.eist.2020.01.002
- KernzaCAP (2023). “Celebrating 40 years; the story of Kernza perennial grain in 40 milestones.” Available at: <https://kernza.org/celebrating-40-years-kernza-perennial-grain-in-40-milestones/>
- Kivimaa, P., Bergek, A., Matschoss, K., and van Lente, H. (2020). Intermediaries in accelerating transitions: introduction to the special issue. *Environ. Innov. Soc. Trans.* 36, 372–377. doi: 10.1016/j.eist.2020.03.004
- Kivimaa, P., Boon, W., Hyysalo, S., and Klerkx, L. (2019). Towards a typology of intermediaries in sustainability transitions: a systematic review and a research agenda. *Res. Policy* 48, 1062–1075. doi: 10.1016/j.respol.2018.10.006
- Klerkx, L., and Gildemacher, P. (2012). “The role of innovation brokers in the agricultural innovation system,” in *Improving agricultural knowledge and innovation systems: OECD conference proceedings*. A. Brizzi, W. Janssen, A. Watkins, M. Lantin and J. Wadsworth Eds. (Paris: OECD Publishing)
- Klerkx, L., and Leeuwis, C. (2009). Establishment and embedding of innovation brokers at different innovation system levels: insights from the Dutch agricultural sector. *Technol. Forecast. Soc. Chang.* 76, 849–860. doi: 10.1016/j.techfore.2008.10.001
- Kline, S. J. (1986). “An overview of innovation” in *The positive sum strategy: Harnessing technology for economic growth*. eds. R. Landau and N. Rosenberg, vol. 1 (Washington, DC: National Academy Press), 275–305.
- Köhler, J., Geels, F. W., Kern, F., Markard, J., Onsongo, E., Wiczorek, A., et al. (2019). An agenda for sustainability transitions research: state of the art and future directions. *Environ. Innov. Soc. Trans.* 31, 1–32. doi: 10.1016/j.eist.2019.01.004
- Koutsouris, A. (2018). “Role of extension in agricultural technology transfer: a critical review,” in *Innovation, technology, and knowledge management*. Eds. N. Kalaitzandonakes, E. G. Carayannis, E. Grigoroudis and S. Rozakis (From Agriscience to Agribusiness, Springer), 337–359.
- Lanker, M., Bell, M., and Picasso, V. (2020). Farmer perspectives and experiences introducing the novel perennial grain Kernza intermediate wheatgrass in the US Midwest. *Renewable Agric. Food Sys.* 35, 653–662. doi: 10.1017/S1742170519000310
- Leeuwis, C., and Aarts, N. (2011). Rethinking communication in innovation processes: creating space for change in complex systems. *J. agricul. educ. extension* 17, 21–36. doi: 10.1080/1389224X.2011.536344
- López-Rubio, P., Roig-Tierno, N., and Mas-Tur, A. (2020). Regional innovation system research trends: toward knowledge management and entrepreneurial ecosystems. *Int. J. Quality Innov.* 6, 1–16. doi: 10.1186/s40887-020-00038-x
- Meynard, J. M., Jeuffroy, M. H., Le Bail, M., Lefèvre, A., Magrini, M. B., and Michon, C. (2017). Designing coupled innovations for the sustainability transition of agrifood systems. *Agric. Syst.* 157, 330–339. doi: 10.1016/j.agry.2016.08.002
- Mignon, I., and Kanda, W. (2018). A typology of intermediary organizations and their impact on sustainability transition policies. *Environ. Innov. Soc. Trans.* 29, 100–113. doi: 10.1016/j.eist.2018.07.001
- Miller, H. T., and King, C. S. (1998). Practical theory. *Am. Rev. Public Adm.* 28, 43–60. doi: 10.1177/027507409802800103
- Missimer, M., Robèrt, K. H., and Broman, G. (2017). A strategic approach to social sustainability—part 1: exploring the social system. *J. Clean. Prod.* 140, 32–41. doi: 10.1016/j.jclepro.2016.03.170
- Molnar, J. J., and Jolly, C. M. (1988). Technology transfer: institutions, models, and impacts on agriculture and rural life in the developing world. *Agric. Hum. Values* 5, 16–23. doi: 10.1007/BF02217173

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- Montenegro De Wit, M., and Iles, A. (2016). Toward thick legitimacy: creating a web of legitimacy for agroecology. *Elementa* 4, 1–24. doi: 10.12952/journal.elementa.000115
- Moss, T. (2009). Intermediaries and the governance of sociotechnical networks in transition. *Environ Plan A* 41, 1480–1495. doi: 10.1068/a4116
- Perez, C. (2010). Technological revolutions and techno-economic paradigms. *Camb. J. Econ.* 34, 185–202. doi: 10.1093/cje/bep051
- Peters, A., Barrett, E., and Stinogel, J. (2021). Technical assistance for continuous living cover agricultural practices. Retrieved from the University of Minnesota Digital Conservancy, Available at: <https://hdl.handle.net/11299/225836>.
- Prokopy, L. S., Floress, K., Arbuckle, J. G., Church, S. P., Eanes, F. R., Gao, Y., et al. (2019). Adoption of agricultural conservation practices in the United States: evidence from 35 years of quantitative literature. *J. Soil Water Conserv.* 74, 520–534. doi: 10.2489/jswc.74.5.520
- Ranjan, P., Wardropper, C. B., Eanes, F. R., Reddy, S. M. W., Harden, S. C., Masuda, Y. J., et al. (2019). Understanding barriers and opportunities for adoption of conservation practices on rented farmland in the US. *Land Use Policy* 80, 214–223. doi: 10.1016/J.LANDUSEPOL.2018.09.039
- Roesch-McNally, G., Basche, A., Arbuckle, J., Tyndall, J., Miguez, F., Bowman, T., et al. (2018). The trouble with cover crops: farmers' experiences with overcoming barriers to adoption. *Renewable Agric. Food Sys.* 33, 322–333. doi: 10.1017/S1742170517000096
- Romanowski, R. (2019). The nature of innovation management. In *Managing Economic Innovations - Ideas and Institutions*. Ed. R. Romanowowski (Bogucki Wyd. Nauk, Poznań), 6–21.
- Sartas, M., Schut, M., Proietti, C., Thiele, G., and Leeuwis, C. (2020). Scaling readiness: science and practice of an approach to enhance impact of research for development. *Agric. Syst.* 183:102874. doi: 10.1016/j.agsy.2020.102874
- Schmoch, U., Reid, P. P., Encarnacao, J., and Abramson, H. N. (Eds.). (1997). *Technology transfer systems in the United States and Germany: Lessons and perspectives*. National Academies Press. NW Washington
- Schut, M., Leeuwis, C., Sartas, M., Taborda Andrade, L. A., van Etten, J., Muller, A., et al. (2022). “Scaling readiness: learnings from applying a novel approach to support scaling of food system innovations,” in *Root, tuber and Banana food system innovations: Value creation for inclusive outcomes*. Eds. G. Thiele, M. Friedmann, H. Campos, V. Polar and J. Bentley (Springer), 71–102.
- Sovacool, B. K., Turnheim, B., Martiskainen, M., Brown, D., and Kivimaa, P. (2020). Guides or gatekeepers? Incumbent-oriented transition intermediaries in a low-carbon era. *Energy Res. Soc. Sci.* 66:101490. doi: 10.1016/j.erss.2020.101490
- Steyaert, P., Barbier, M., Cerf, M., Levain, A., and Marie Loconto, A. (2016). Role of intermediation in the management of complex sociotechnical transitions. *AgroEcological Transitions*, Wageningen University Research. Available at: <https://hal.archives-ouvertes.fr/hal-01470892>.
- Thiele, G., Friedmann, M., Campos, H., Polar, V., and Bentley, J. W. (2022). *Root, tuber and Banana food system innovations: Value creation for inclusive outcomes* (p. 561). Springer Nature. New York City
- Wayman, S., Debray, V., Parry, S., David, C., and Ryan, M. R. (2019). Perspectives on perennial grain crop production among organic and conventional farmers in France and the United States. *Agriculture* 9:244. doi: 10.3390/agriculture9110244
- Wigboldus, S. A., and Leeuwis, C. (2013). *Towards responsible scaling up and out in agricultural development: An exploration of concepts and principles*. Wageningen (The Netherlands): Centre for Development Innovation.
- Zolfagharian, M., Walrave, B., Raven, R., and Romme, A. G. L. (2019). Studying transitions: past, present, and future. *Res. Policy* 48:103788. doi: 10.1016/j.respol.2019.04.012

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