frontiers | Frontiers in Sustainable Food Systems

TYPE Original Research
PUBLISHED 23 May 2025
DOI 10.3389/fsufs.2025.1499410

Check for updates

OPEN ACCESS

EDITED BY
Luuk Fleskens,
Wageningen University and Research, Netherlands

REVIEWED BY
Adam Canning,
James Cook University, Australia
Buddhi Gyawali,
Kentucky State University, United States

*CORRESPONDENCE
Rebecca Fudge
rebecca.fudge@tnc.org

RECEIVED 20 September 2024
ACCEPTED 25 April 2025
PUBLISHED 23 May 2025

CITATION
Fudge R, Lovdal A, Zimmerman E, Kushner L and Grossman J (2025) Environmental outcomes of landscape-scale agricultural transitions in the Upper Midwestern U.S. *Front. Sustain. Food Syst.* 9:1499410.
doi: 10.3389/fsufs.2025.1499410

Environmental outcomes of landscape-scale agricultural transitions in the Upper Midwestern U.S.

Rebecca Fudge[1,2]*, Alison Lovdal[2], Emily Zimmerman[2], Lisa Kushner[2] and Julie Grossman[1]

[1]Department of Horticultural Science, University of Minnesota, Saint Paul, MN, United States, [2]The Nature Conservancy, Arlington, VA, United States

The United States (U.S.) Corn Belt leads North America in row crop production, yet this high productivity comes at an environmental cost in terms of nitrate loss, soil erosion, and greenhouse gas emissions. In this study, we focus on the Upper Mississippi River basin within the U.S. Corn Belt, which represents a landscape scale for agricultural transformation. We outline a methodology to assess a suite of environmental outcomes associated with the transition from summer annual maize/soybean systems to incorporation of continuous living cover systems. We use and expand publicly available tools alongside empirical data to assess nitrate loss, soil erosion, and greenhouse gas emissions for four potential agricultural transition scenarios in the region, on an annual basis compared to a business-as-usual maize/soybean rotation. We consider the following four scenarios: incorporating (1) winter annual cover crops or (2) winter annual oilseeds into 50% of maize and soybean hectares in the region, or converting 50% of marginally productive maize and soybean hectares to (3) agroforestry or (4) pastured livestock systems. Our results indicate that all four systems are likely to reduce topsoil loss when compared to maize and soybean systems, and that the more transformative systems—agroforestry and pastured livestock—have the greatest potential to reduce nitrate loss. Yet, our results suggest that among these transitions, there are tradeoffs in environmental outcomes. For example, pastured livestock and winter annual oilseeds could potentially increase greenhouse gas emissions relative to maize/soybean systems. Our results illustrate that continuous living cover could improve environmental outcomes in the Upper Midwest, but there is tremendous uncertainty and variability surrounding those outcomes.

KEYWORDS

nitrate loss, soil erosion, greenhouse gas emissions, cover crop, agroforestry, livestock, winter annual oilseed, foodscape

1 Introduction

In the last century, agriculture in the United States (U.S.) Corn Belt has simultaneously experienced intensive and extensive growth. This growth is exemplified both by gains in per-acre productivity (Mueller et al., 2019), and the expansive and relatively homogenous row-crop agricultural system covering over 60 million ha (Hunt et al., 2020; USDA NASS, 2022). The growth of this increasingly simplified agricultural system, dependent upon high amounts of external, energy-intensive inputs, has driven negative environmental outcomes, especially soil degradation, erosion, and nutrient (nitrogen and phosphorus) loss (Robertson and Saad, 2021; Thaler et al., 2021). Excess sediment and nutrients from this agricultural system have costly local and regional impacts on human and ecosystem health (Rabotyagov et al., 2014). For example, ∼18–36% of the synthetic nitrogen (N) fertilizer applied to *Zea mays* L. (maize) leaches into groundwater as nitrate (Randall and Iragavarapu, 1995; Sexton et al., 1996; Masarik et al., 2014). Locally, excess nitrate in ground and surface water has led to concerns about safe drinking water (Hamlin et al., 2022) because of the relationship between nitrate exposure and adverse human health outcomes, including birth defects (Brender et al., 2013), infant methemoglobinemia (Comly, 1945), and cancer (De Roos et al., 2003; Inoue-Choi et al., 2015; Jones et al., 2016; Ward et al., 2018). Regionally, nitrate loss from agriculture in the Upper Mississippi River Basin is a primary contributor to the annual development of the Gulf of Mexico hypoxic zone (David et al., 2010; Robertson and Saad, 2021), which compromises the ecological integrity of aquatic communities (Rabalais et al., 2002).

While multiple factors contribute to these local and regional environmental challenges, in the dominant maize and *Glycine max* L. (soybean) cropping system, there are no living crops grown on the landscape during the late winter and spring (January–June), when precipitation events cause significant sediment and nutrient losses (David et al., 2010). Consequently, the overlapping challenges of soil degradation, erosion, and nutrient loss could be addressed through the integration of continuous living cover on the landscape (i.e., the presence of living roots in soil throughout the year). A variety of continuous living cover practices have been proposed for the U.S. Corn Belt that could be incorporated into, or even replace, maize and soybean systems: winter annual cover crops (Singer et al., 2007), winter annual oilseed crops (Liu et al., 2019), agroforestry systems (Mori et al., 2017), and pastured livestock (Sulc and Tracy, 2007). In contrast to annual cropping systems in the U.S. Corn Belt, which typically have extended periods of bare soil, continuous living cover practices prevent nutrient loss through plant uptake and provide erosion protection via reduced impacts of rainfall and wind (Basche and DeLonge, 2017). For example, winter annual cover crops—namely *Secale cereale* L. (cereal rye)—have been shown to reduce nitrogen and phosphorus loss from maize and soybean rotations by 28% and 29%, respectively (Iowa Department of Agriculture and Land Stewardship, 2017). Despite the potential environmental benefits of continuous living cover, adoption has been limited the U.S. Corn Belt, especially among row-crop farmers; in the U.S. Midwest, cover crops were planted on just 7.2% of farmland as of 2021 (Zhou et al., 2022). Many barriers to adoption of conservation practices, including continuous living cover, are related to broader system factors (e.g., prevailing markets, established infrastructure, agricultural policy) that will require a holistic approach to overcome (Ranjan et al., 2019).

For farmers to adapt their farms to meet the food, feed, fiber, and fuel needs of a growing population—while simultaneously producing the nature-based benefits upon which humanity depends—a foodscape may serve as a useful conceptual approach (Jung et al., 2024). Foodscapes are geographic areas with shared characteristics of agricultural production along biophysical and socioeconomic gradients (Jung et al., 2024). These areas offer a landscape scale for agricultural transformation, where niche innovation can be expanded to a region that shares macroeconomic patterns, political environments, and climatic conditions (Jung et al., 2024). Transitions are long-term processes that often take decades to unfold because radical innovations take a long time to develop from their emergence in small application niches to diffusion across a landscape (Köhler et al., 2019). In this case, radical innovations include continuous living cover systems that would transform the current dominance of maize/soybean monoculture systems and provide the environmental benefits increasingly needed by local and regional communities.

The Upper Mississippi River (UMR) Basin provides a key opportunity to apply the foodscape concept to begin landscape-scale transformation by integrating continuous living cover. This area encompasses southeast Minnesota, northeast Iowa, southwest Wisconsin, and northwestern Illinois, colloquially known as the Driftless region (Figure 1). The region has common biophysical characteristics, including its geology and soils, climate, and historical vegetation (Martin, 1965; Prior, 1991). The dominance of maize and soybean production in this region is a reflection of shared biophysical and cultural characteristics of the area and has been influenced by similar political and market structures. Based on these characteristics, this region is ripe for examining the foodscape concept. Importantly, the geographic scale of the foodscape allows for landscape-level change, which can ideally serve as a model for transformation of other row-crop production systems across the larger U.S. Corn Belt. The UMR Foodscape can serve as a niche for innovation outside of the prevailing regime, where bottom-up momentum can meet landscape-level pressure (Geels and Schot, 2007; Conway, 2023).

Stakeholders in the food and agriculture sector are increasingly engaging with local landowners and farmers to enhance environmental outcomes across the U.S. Corn Belt in landscapes like the UMR (Bossio et al., 2021; World Wildlife Fund, 2023). Some have ambitious goals to implement conservation practices (e.g., conservation practices on 50% of row-crop acres) and nearly all have articulated goals around enhancing environmental benefits associated with climate change mitigation, water quality and quantity, and/or biodiversity production (Prokopy et al., 2020). Yet, limited work has been done so far to examine environmental and economic benefits at the landscape scale in regions defined by agricultural production similarities.

In our work, we evaluate multiple environmental benefits associated with the adoption of various types of continuous living cover across the UMR Foodscape. Specifically, we use publicly-available tools alongside empirical data to assess nitrate loss, soil erosion, greenhouse gas (GHG) emissions, and carbon sequestration for four potential continuous cover, agricultural transition scenarios in the UMR Foodscape. We focus on these environmental outcomes because of their centrality to agriculture's contributions to climate change and water quality issues, but methodologies to assess biodiversity outcomes at the landscape scale should also be developed. Two winter annual systems, winter annual cover crops and winter annual oilseeds, are analyzed under a widespread adoption scenario (50% of all

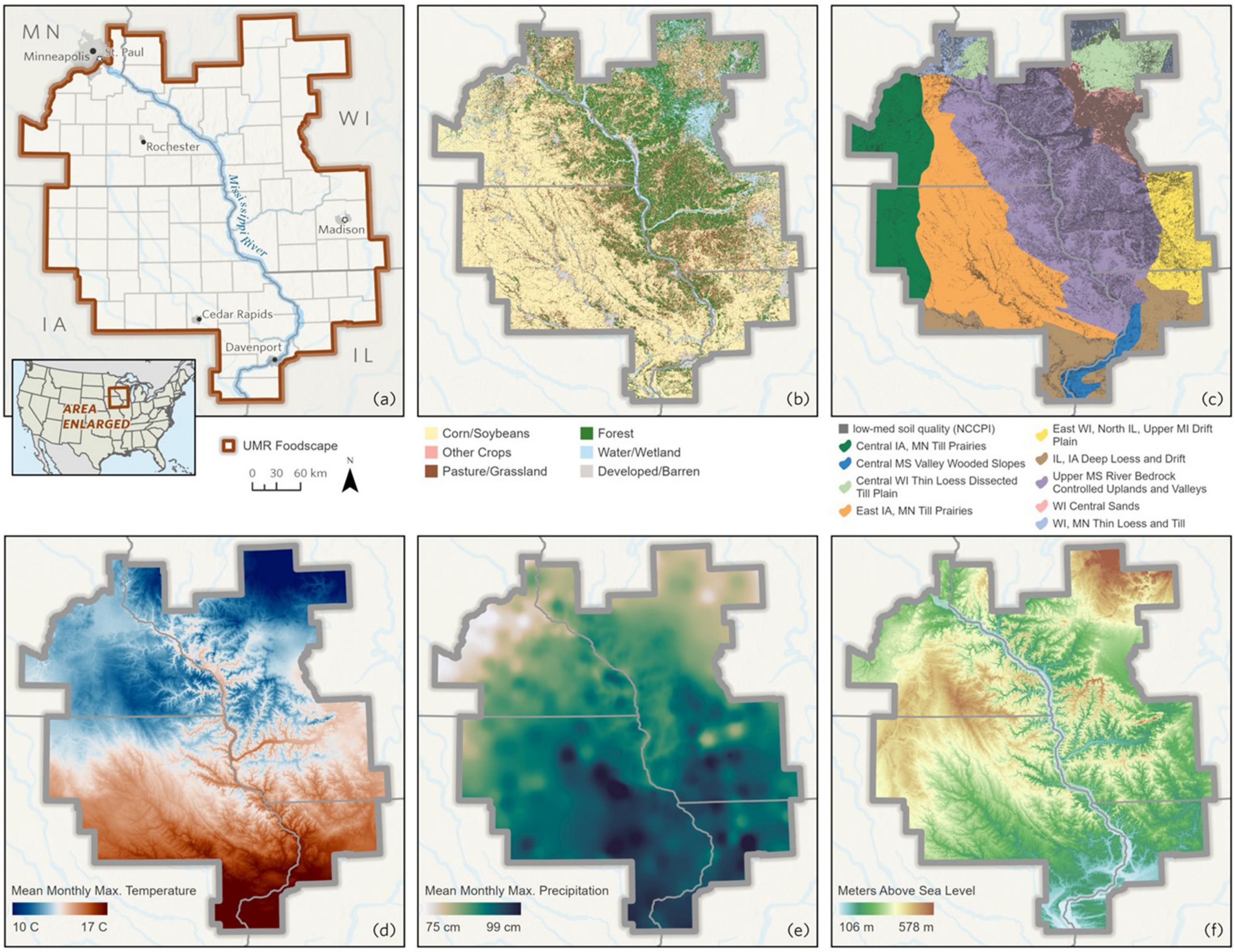

FIGURE 1
The geographic region and biophysical characteristics of the UMR Foodscape, including **(a)** counties included in the UMR Foodscape, **(b)** land use, **(c)** low-medium productivity soil and major land resource areas, **(d)** mean monthly maximum temperature, **(e)** mean monthly maximum precipitation, and **(f)** topography.

maize/soybean acreage in the UMR Foodscape). Two perennial systems, agroforestry and pastured livestock, are analyzed under a limited adoption scenario (50% of marginally productive maize/soybean area in the UMR Foodscape) because they are more transformative and would replace some maize/soybean (MS) production ha. We have two major goals with this work: (1) to provide a model for estimating environmental outcomes at the foodscape-scale in other global regions, and (2) to inform decision-makers and interested parties in the UMR Foodscape of the potential environmental outcomes associated with transition scenarios.

2 Methods

2.1 Study location

We focus on a region within the U.S. Corn Belt called the Upper Mississippi River (UMR) Foodscape. This area, situated in the UMR Basin, encompasses 83 counties across southeast Minnesota, northeast Iowa, southwest Wisconsin, and northwestern Illinois (Figure 1a; Supplementary Table 1). Of the 13.9 million ha in the UMR Foodscape, 9.1 million ha are cropland, comprising 65% of total land (Figure 1b). According to the 2022 USDA NASS Cropland Data Layer, cropland in the UMR foodscape is dominated by maize and soybeans, which occupy 6.6 million ha, or 72.5% of cropland. Hay and pasture take up another 1.88 million ha, or 20.7% of cropland. Other legumes and grain make up <1 million ha (6.5% cropland), and fruits and vegetables make up 0.3% of cropland. Tree crops comprise <1% of cropland.

We defined maize/soybean (**MS**) acreage in the UMR as hectares of maize and soybean occurring in the last 9 of 15 years, identified using the 2008–2022 USDA Cropland Data Layers (Figure 1a; USDA National Agricultural Statistical Service, 2008; USDA NASS, 2022). Marginally productive MS hectares were defined using the National Commodity Crop Productivity Index (NCCPI 3.0), which ranks land on a scale from 0 (low productivity) to 1 (high productivity; Albers et al., 2022). The average NCCPI score across the UMR Foodscape was 0.65 (Figure 1c).

2.2 Scope of transition scenarios

This study focuses on four continuous cover transition scenarios: winter annual cover crops, winter annual oilseeds, agroforestry, and pastured livestock (Table 1). We define winter annual cover crops as overwintering, non-harvested cover crops on MS hectares. This definition captures a variety of cover cropping practices and species, including cover crops grown as monocultures or in mixtures. We define a winter oilseed as an overwintering oilseed crop, either winter camelina (*Camelina sativa* L.) or pennycress (*Thlaspi arvense* L.), relay-cropped on MS hectares. These winter annual oilseeds have been proposed as new winter-hardy cash crops for the U.S. Midwest that can be grown in conjunction with a maize/soybean system (Forever Green Initiative, 2021).

For these two scenarios involving overwintering crops, we estimated environmental outcomes if these practices were adopted on 50% of MS hectares, in line with goals to implement regenerative practices on at least 50% of U.S. row crop hectares (Prokopy et al., 2020; Bossio et al., 2021; World Wildlife Fund, 2023). Because winter annual cover crops have already been adopted on some MS hectares, we estimated environmental outcomes for additional acreage that would equal 50% of MS hectares in total. We defined cover crop adoption on MS hectares as the 3-year average of cover crop use from 2017 to 2019, identified through data from Operational Tillage Information System (OpTIS) version 3.0. In the Foodscape, we estimate that cover crops are currently adopted on 2.1×10^5 ha, or 8% of MS hectares. Cover crops would need to be adopted on an additional 42% of acres to reach 50% of MS hectares in the Foodscape.

We define agroforestry for this study as alley cropping with a woody crop like hazelnut (*Corylus americana*) on MS hectares, converting at least 20% of MS hectares in a farm to an agroforestry system, based on the U.S. Department of Agriculture Natural Resource Conservation Service (USDA NRCS) agroforestry Conservation Practiced Standard (CPS) 311. We define pastured livestock systems as marginal MS hectares converted to a perennial grass/legume system rotationally grazed by cattle. In this system, we assume cattle are on pasture from calving to finishing at one farm, rather than finished on grain at a feed lot. For both of these scenarios, we estimate adoption on 50% of marginally productive MS hectares, defined as land with an NCCPI score < 0.55. Unlike winter annual crops, these scenarios represent a more significant transformation of MS land, and thus we estimate these practices would be adopted on land in the UMR Foodscape with the lowest opportunity cost.

2.3 Sources for estimating environmental outcomes

In this work we focused on three environmental outcomes: nitrate loss, soil erosion, and GHG emissions. When possible, we estimated environmental outcomes with publicly available tools to make the methodology accessible and useful for a variety of applications (Table 2). For the best estimates for the UMR Foodscape, we employed tools that were either developed specifically for the region or that could be spatially targeted to the region. We used the web platform AgEvidence (https://www.agevidence.org/) to estimate environmental outcomes related to cover crop adoption. AgEvidence is a web-based tool developed by The Nature Conservancy that uses meta-analyses and data visualizations to communicate the environmental and food production outcomes that result from implementing a variety of conservation agricultural practices, relative to a conventional row crop system (Atwood and Wood, 2020). The meta-analyses are based on peer-reviewed papers filtered by geography (the U.S. Corn Belt) and cropping system, published from 1980 to 2020 (Atwood et al., 2024).

We used the CarbOn Management and Emissions Tool (COMET)-Planner (http://comet-planner.com/) to estimate changes in GHG emissions due to transitioning MS hectares to winter annual cover crops, agroforestry, and pastured livestock systems. COMET-Planner is a web-based tool that provides estimates of GHG emissions of conservation practices across various agricultural landscapes (Swan et al., 2023). Created by Colorado State University in partnership with the USDA NRCS, the tool is widely used to estimate carbon benefits of conservation practices at large geographical scales (Swan et al., 2023). However, it is worth noting that the model relies on national and regional averages, rather than site-specific data on soils, climate, and management, so the model may over- or under-estimate carbon sequestration for specific parcels (Castle et al., 2025).

When publicly available tools were not available for a specific transition scenario/environmental outcome combination, we used published meta-analyses or empirical studies conducted in the UMR Foodscape region. For certain combinations of transition scenarios and environmental outcomes, we found no empirical studies specifically based in the UMR Foodscape. In those cases, we limited results to studies in states bordering the U.S. Corn Belt. The authors also looked at reputable gray literature (e.g., Ecotone Analytics et al., 2022) to identify other peer-reviewed literature relevant to our systems of interest.

2.4 Per hectare estimates of nitrate loss and soil erosion

To estimate baseline nitrate loss on MS hectares, we assumed that 18–36% of N applied as fertilizer is leached from the system, based on studies of MS systems in UMR Foodscape states (Randall and Iragavarapu, 1995; Sexton et al., 1996; Masarik et al., 2014). We estimated N fertilizer application rates using the most recent (2016) county fertilizer application data from the Nutrient Use Geographic Information System (NuGIS; Rund et al., 2010). To estimate annual top soil loss, we assumed an average annual top soil loss on MS hectares as 7.2 MT ha^{1} yr^{1} (Thaler et al., 2022).

We estimated per-hectare coefficients for changes in nitrate loss and soil erosion with AgEvidence for the winter annual cover crop scenario and with empirical studies for the remaining three scenarios; when applicable, citations are provided in Table 2 for empirical studies. For pastured livestock systems, scant literature exists to directly compare row-crop MS systems to pastured livestock because this is currently an uncommon transition in the

TABLE 1 Four transition scenarios outlined in this work.

Transition scenario	Detail	Scale of adoption
Winter annual cover crop	Overwintering, non-harvested cover crop on maize and soybean row crop acres	**Widespread**: 50% of maize/soybean acres in the UMR Foodscape
Winter annual oilseed	Winter oilseed cash cover crop (winter camelina or pennycress) on maize and soybean row crop acres	**Widespread**: 50% of maize/soybean acres in the UMR Foodscape
Agroforestry	Conversion from row crop to alley cropping (woody brush like hazelnut)	**Narrow**: 50% of low–medium productivity maize/soybean acres in the UMR Foodscape
Pastured livestock	Conversion from row crop to pasture-based cattle grazing	**Narrow**: 50% of low–medium productivity maize/soybean acres in the UMR Foodscape

Two scenarios involve winter annual crops (winter cover crops and winter annual oilseeds), which could be incorporated into a maize/soybean rotation with limited disruption. For those scenarios, we estimate environmental outcomes if they were adopted on 50% of corn/soy acres in the Foodscape, to match the most ambitious goals set by NGOs to adopt regenerative practices on 50% of acres. Two scenarios involve perennial systems (agroforestry and pastured livestock) which would replace maize/soybean acres. Because these systems are more disruptive, we estimate their adoption of 50% of maize/soy acres identified as low–medium productivity.

TABLE 2 Percent change in environmental outcomes relative to a maize/soybean system on a per-hectare basis.

Outcome	Winter cover crop		Winter oilseed		Agroforestry		Pastured livestock	
	Pct change	Model/ sources	Pct change	Sources	Pct change	Model/ sources	Pct change	Model/ sources
Nitrate loss	74% decrease—8% increase	Ag evidence	25% decrease—27% increase	Cecchin et al., 2021; Emmett et al., 2022	82–91% decrease	Wolz et al., 2018	68–86% decrease	Ecotone Analytics et al., 2022; Pilon et al., 2019
Soil erosion	28–92% decrease	Ag evidence	39–50% decrease	Cecchin et al., 2021	30–51% decrease	Ecotone Analytics et al., 2022	72–88% decrease	Ecotone Analytics et al., 2022; Pilon et al., 2017
Greenhouse gas emissions	Varies by county; see Section 3	COMET-planner	49% decrease—100% increase	Berti et al., 2017; Cecchin et al., 2021; Atwood and Wood, 2020	Varies by county; see Section 3	COMET-planner	Varies by county; see Section 3	COMET-planner; Stanley et al., 2018

region. As such, we first estimated the change in nitrate loss when MS hectares were converted to un-grazed, perennial systems. Then we estimated the change in nitrate loss in grazed systems vs. un-grazed systems to account for grazing-specific differences nitrate loss, such as biomass and manure accumulation. We combined these two estimates to calculate total change in nitrate loss and soil erosion.

2.5 Per hectare estimates of GHG emissions

To estimate baseline GHG emissions across MS hectares in the UMR Foodscape, we used the USDA U.S. Agriculture and Forestry Greenhouse Gas Inventory: 1990–2018 (U.S. Department of Agriculture, 2021). The GHG Inventory includes data on soil carbon stock changes, direct N_2O emissions, and indirect N_2O emissions by cropping system and state. We summed emissions across the UMR Foodscape counties and averaged total emissions by state across the years 2013–2015, the most recent data available. This approach allows us to estimate a percent change in emissions, representing a novel use of COMET-Planner. Typically, the tool is used to estimate a reduction in emissions, measured in MT CO_2e, but our calculation of baseline GHG emissions allows us to convert that figure to a percent reduction.

We used COMET-Planner to estimate changes in GHG emissions when converting MS hectares to continuous living cover (Swan et al., 2023). COMET-Planner aligns field-based studies to USDA NRCS CPS to estimate GHG effects of implementing these conservation practices on farms. For this analysis, we selected the CPS most closely aligned with each transition scenario. When possible, we calculated the range of GHG emissions changes by using COMET-Planner's estimates of maximum and minimum GHG reductions. In some cases, maximum and minimum values were not provided, so we used total GHG emission reductions. For the two CPSs associated with agroforestry, reduction estimates were not available for every county in the UMR Foodscape (values missing for 17 out of 83 counties). If COMET-Planner did not have data for a specific county, we used the average value of all UMR Foodscape counties within the same state. The estimation of GHG emissions through COMET-Planner incorporates multiple sources: soil carbon sequestration or soil organic carbon loss, woody biomass carbon sequestration, and CO_2/CO/N_2O/CH_4 emissions from biomass burning, liming, urea fertilization, and drained organic soils.

To estimate GHG emissions reductions related to winter annual cover crops, we modeled outcomes with two CPS options: "Add Non-Legume Seasonal Cover Crop (with 25% Fertilizer N Reduction) to Non-Irrigated Cropland" (CPS 340) and "Add Legume Seasonal Cover Crop (with 50% Fertilizer N Reduction) to Non-Irrigated Cropland" (CPS 340). For the winter oilseed scenario, the COMET-Planner tool did not have a sufficiently applicable CSP due to the complicated climate change potential of these crops: because their intended use is as aviation fuel (Resurreccion et al., 2021), lifecycle assessments of these crops also include the potential reduction in emissions due to oilseed-derived jet fuel relative to petroleum-derived jet fuel (Berti et al., 2017; Cecchin et al., 2021; Ecotone Analytics et al., 2022). For only this crop, we included life cycle assessments from peer-reviewed studies to estimate GHG emissions.

For the agroforestry scenario, we selected two CPS options to represent a range of management choices: "Conversion of Annual Cropland to a Farm Woodlot" (CPS 612) and "Replace 20% of Annual Cropland with Woody Plants" (CPS 311). For the pastured livestock scenario, we used both COMET-Planner and empirical data. First, we calculated the change in GHG emissions due to conversion of annual row crops to a perennial system ("Conversion of Annual Cropland to Non-Irrigated Grass/Legume Forage/Biomass Crops"; CPS 512). Then, we used per-steer emissions estimates for a pasture-based system to estimate additional GHG emissions from steer, assuming 2.7 steers/ha (Stanley et al., 2018). This method does not consider emissions from manure, so it could represent an under-estimation of GHG emissions from the system.

3 Results

3.1 Spatial targeting for transition scenario adoption

In the UMR Foodscape, winter annual cover crops were used on 2.1×10^5 ha in 9 of the last 15 years, equal to 7% of total MS area (OpTIS v2023), aligning with previous research (Zhou et al., 2022). To reach 50% of MS hectares total, we calculated environmental outcomes for winter annual cover crop adoption on an additional 43% of MS acres (1.3×10^6 ha). For winter oilseeds, we calculated environmental outcomes given adoption on 50% of acres (1.5×10^6 ha). We estimate no current adoption of winter oilseeds in the UMR foodscape given lack of data, though some estimates suggest there are perhaps 2,000 ha in the region currently (Ecotone Analytics et al., 2022). For the agroforestry and pastured livestock scenarios, we estimated environmental outcomes based on adoption of these practices on 50% of marginally productive MS land in the UMR Foodscape. Counties varied in their quantity of marginal lands (Figure 1c), which could provide an opportunity for county-level targeting of agroforestry or pastured livestock practices. Most land in the UMR foodscape is highly productive; 50% of marginal land equals 9.6×10^4 ha or 3.1% of MS area.

3.2 Nitrate loss

Adoption of winter annual cover crops or winter oilseeds could represent a range of nitrate loss outcomes, but winter annual cover crops are generally expected to reduce nitrate loss to a greater extent than winter oilseeds (Table 3; Figure 2). The use of winter annual cover crops could lead to a range of nitrate loss outcomes on a per hectare basis: from a 74% decrease to an 8% increase relative to the current system (Table 2). If cover crops were adopted on 50% of MS acreage total in the UMR foodscape, that would translate to a 3.6×10^4 MT decrease in nitrate-N leaching to a 3.9×10^3 MT increase in leaching (Table 3; Figure 2). This wide range echoes findings of a global meta-analysis that found that many factors—including cover crop species, soil type, soil sand content, and tillage intensity—can affect the extent to which cover crops prevent nitrate loss, if at all (Nouri et al., 2022).

While winter annual cover crops are typically not fertilized, fertilizer application is recommended for winter annual oilseeds, which limits their utility in reducing nitrate loss. Fertilizer application for common winter annual oilseeds can range from 67 to 89 kg ha^{-1} (Ott et al., 2019; Gregg et al., 2024). However, winter annual oilseeds still provide cover and nutrient demand in the fallow season, which can reduce nutrient loss (Berti et al., 2017; Cecchin et al., 2021). As such, our estimates suggest a range of possible nitrate loss outcomes for winter oilseed systems, relative to MS systems: a 25% decrease to a 27% increase in leaching per hectare (Table 2). If winter oilseeds were adopted across 50% of MS acreage in the UMR foodscape, this could translate to a 1.4×10^4 MT-decrease to a 1.5×10^4 MT-increase in nitrate-N loss (Table 3; Figure 2).

Agroforestry and pastured livestock systems represent more transformative agricultural scenarios and have higher potential to reduce nitrate loss. Agroforestry systems could reduce nitrate loss by 82–91% per hectare (Table 2). If implemented across 50% of low-medium productive MS hectares in the UMR foodscape (3.1% of total MS area), nitrate loss could decrease by $1.3–2.9 \times 10^3$ MT (Table 3; Figure 2). On a per-hectare basis, pastured livestock could decrease nitrate loss by 68–86% relative to MS (Table 2). If implemented across the same hectares as the agroforestry scenario, pastured livestock could prevent $1.1–2.8 \times 10^3$ MT nitrate-N loss (Table 3; Figure 2).

3.3 Soil erosion

Winter annual cover crops and winter annual oilseed crops both offer benefits of preventing topsoil loss. In systems with winter annual cover crops, soil erosion could decrease by 28–92% per hectare relative to MS systems (Table 2). Adoption of winter annual cover crops on an additional 43% of MS hectare could prevent $2.6–8.5 \times 10^6$ MT of soil loss across the UMR foodscape (Table 3; Figure 2). Winter annual oilseeds could reduce soil erosion by 39–50% per hectare based on empirical evidence (Table 2). We estimate that adoption of winter oilseeds on 50% of MS hectares could represent $4.2–5.4 \times 10^6$ MT soil loss avoided (Table 3; Figure 2).

Agroforestry and pastured livestock systems also have the potential to reduce soil erosion. Available data suggest agroforestry systems could reduce topsoil loss by 30–51% per hectare relative to a MS system (Table 2). If agroforestry were adopted across 50% of low-medium productive MS acreage in the UMR foodscape, soil erosion could decrease by $2.1–3.5 \times 10^5$ MT (Table 3; Figure 2).

TABLE 3 Total estimated changes in environmental outcomes across the foodscape.

Outcome	Winter cover crop	Winter oilseed	Agroforestry	Pastured livestock
Change across foodscape				
Nitrate loss	3.6×10^4 MT decrease—3.9×10^3 MT increase	1.4×10^4 MT decrease—1.5×10^4 MT increase	$1.3–2.9 \times 10^3$ MT decrease	$1.1–2.8 \times 10^3$ MT decrease
Soil erosion	$2.6–8.5 \times 10^6$ MT decrease	$4.2–5.4 \times 10^6$ MT decrease	$2.1–3.5 \times 10^5$ MT decrease	$4.9–6.0 \times 10^5$ MT decrease
Greenhouse gas emissions	$4.9–7.7 \times 10^5$ MT CO_2e decrease	1.4×10^6 MT CO_2e decrease—2.8×10^6 MT CO_2e increase	3.1×10^5-1.1×10^6 MT CO_2e decrease	$2.5–5.6 \times 10^5$ MT CO_2e increase

For winter cover crops and winter oilseeds, these calculations were based on adoption estimates on 50% of maize/soybean acreage. For agroforestry and pastured livestock, these calculations were based on adoption estimates on 50% of marginal maize/soybean acreage (~3% of maize/soybean acreage).

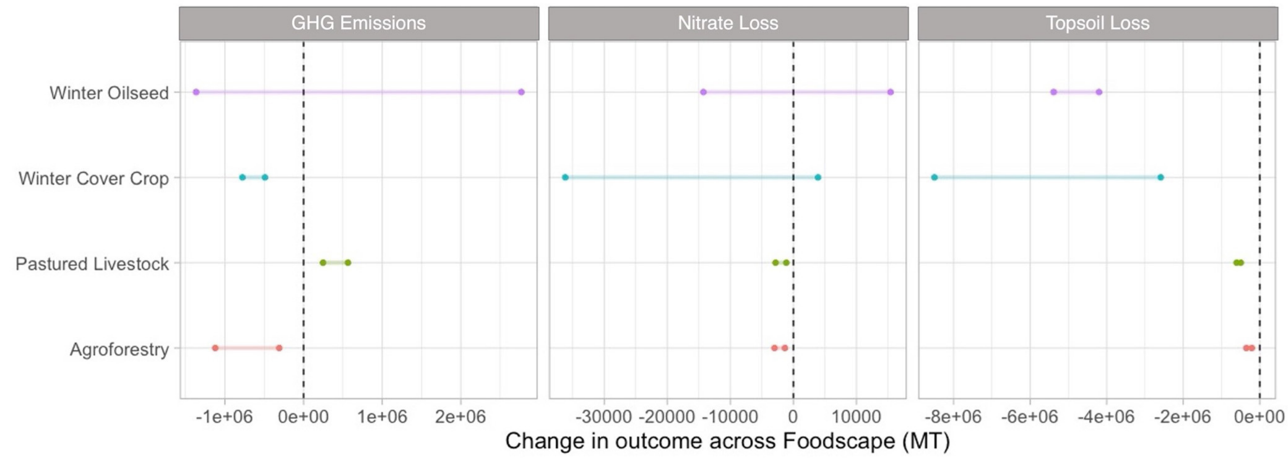

FIGURE 2
Estimated changes in GHG emissions, nitrate loss, and topsoil loss across the Foodscape area. Winter annual oilseeds and winter annual cover crop outcomes reflect adoption on 50% of maize/soybean acreage. Pastured livestock and agroforestry outcomes reflect adoption on 50% of low-medium productivity maize/soybean acreage, equal to 3.1% of maize/soybean acreage.

Pastured livestock could reduce soil erosion by 72–88.4% relative to MS systems, which would translate to $4.9–6.0 \times 10^5$ MT avoided soil loss if implemented across the same acreage as agroforestry in the UMR foodscape (Table 3; Figure 2).

3.4 Greenhouse gas emissions

Winter annual cover crops are likely to reduce GHG emissions, while winter annual oilseeds could lead to a range of emissions outcomes. If implemented on an additional 43% of hectares in the foodscape, winter annual cover crops could reduce GHG emissions by $4.9–7.7 \times 10^5$ MT CO_2e yr^{-1} (Table 3; Figure 2). For winter oilseeds implemented across 50% of MS hectares, GHG emissions could decrease by 1.4×10^6 MT CO_2e yr^{-1} or could increase by 2.8×10^6 MT CO_2e yr^{-1} (Table 3; Figure 2). This range represents uncertainty around these new crops, especially over their fertilization rates which will affect emissions, as well as their downstream use—and potential emissions savings—as replacements for petroleum-based aviation fuel.

Agroforestry systems represent a potentially large reduction in GHG emissions, even with limited adoption. This range was created using two different CPSs in COMET-Planner: CPS 612 (convert the whole property to woodlot) and CPS 311 (convert 20% of annual cropland to agroforestry). If agroforestry were adopted across 50% of low-medium productive MS acreage in the UMR foodscape, GHG emissions could potentially be reduced by 3.1×10^5-1.1×10^6 MT CO_2e per year (Table 3; Figure 2).

Livestock systems represent a potential increase in GHG emissions. Emissions from enteric fermentation largely offset expected reductions due to converting MS hectares to a perennial system. If implemented across the same area as the agroforestry scenario, pastured livestock systems could increase GHG emissions by $2.5–5.6 \times 10^5$ MT CO_2e per year (Table 3; Figure 2). We estimated the conversion of an annual row crop system to a perennial forage system or un-grazed pasture to reduce GHG emissions ($1.3–4.5 \times 10^5$ MT CO_2e yr^{-1} avoided), but grass-fed cattle are estimated to emit 2,700 kg CO_2e yr^{-1} per steer. At an assumed stocking rate of 2.7 steers/ha (Stanley et al., 2018), we estimate the emissions of livestock to outpace the carbon sequestration benefits.

4 Discussion

4.1 Utility of modeling approach

Our approach, using a combination of empirical studies and publicly available tools, represents an accessible and useful way to scale up spatially based models for landscape-level change at the scale of a foodscape. Our results provide three major outcomes with utility for stakeholders: (1) A broad overview of potential environmental outcomes for an agricultural transitional scenario focused on continuous cover, (2) a preliminary overview of areas with marginally productive maize and soybean production that could be targeted for more transformative transition scenarios, and (3) an example methodology for foodscape-scale projects in other regions to assess environmental outcomes and their potential implications. This work has the potential to set the stage for broader, systems-level interventions (e.g., development of farmer-to-farmer and advisor learning, creation of markets and infrastructure, policy creation and implementation, etc.) to facilitate changes in our agriculture and food systems.

We recognize numerous tools exist to estimate environmental outcomes across various spatial scales, from sub-field, field, small watershed, to large basin (e.g., Perez and Cole, 2020). Many of those tools presently lack the capacity to assess the continuous cover practices and their associated environmental outcomes assessed here, and/or require extensive technical skills and empirical data, often for validation and calibration, to run. The approach presented here overcomes those challenges to provide estimations that can guide future conservation planning efforts and public-private investment. The applications of this work are particularly timely, in order to address the challenge of ensuring that public funds are applied in the right practices and places to achieve desired environmental outcomes (e.g., Jones et al., 2018).

4.2 Transition scenario tradeoffs for winter annual cover and oilseed crops

Our approach to analyzing environmental outcomes allows high-level comparisons between transition scenarios, which are useful for conservation organizations considering where to invest their time and resources in a regenerative agriculture transition. Winter annual cover crops and winter oilseeds were compared directly because they are two conservation practices that fit into maize/soybean rotations in a manner that would minimally disrupt the system. The wide range of outcomes related to winter annual cover crops, relative to winter annual oilseeds, could represent the variety of winter cover crop species studied across multiple locations (Atwood and Wood, 2020). This study also does not take into account potential changes in N fertilizer demand for subsequent maize/soybean crops after cover crop use. Winter annual oilseeds, by comparison, have been studied much less extensively (Cecchin et al., 2021; Emmett et al., 2022). Further field studies on winter oilseeds would elucidate the true range of nitrate loss and soil erosion outcomes. So far, life cycle estimates for winter annual oilseed crops assume their use as jet fuel and account for lower emissions from oilseed-derived jet fuel relative to petroleum-derived jet fuel (Resurreccion et al., 2021). At the same time, winter oilseeds have the potential to increase GHG emissions relative to maize/soybean systems, especially due to their fertilizer use (Cecchin et al., 2021).

This analysis focused on environmental outcomes across transition scenarios but did not incorporate economic data. Winter oilseeds have the greatest economic potential of the four transition scenarios we investigated, given our current agricultural system and its associated policy, markets, and infrastructure. Soybean systems relay-cropped with winter camelina or pennycress oilseed crops have been found to generate an equivalent net income to monoculture soybean systems (Ott et al., 2019). Fertilizer application on oilseeds constitutes a major expense (48% of material costs; Ott et al., 2019), so reducing fertilization rate could make the system more profitable and reduce nitrate loss from the system. The system could prove profitable over the longer term given that: (1) winter cash crops only minorly disrupt summer cash crop yields and, thus, profits, (2) winter cash crops do not require capital expenditures on specialized equipment, and (3) these crops could be harvested to generate revenue greater than the additional costs to include them in a rotation. Given the lack of market for these oilseeds currently, future prices for these crops will affect the profitability of the system (Gesch et al., 2014). As of now, winter oilseeds are more successful when relay-cropped with soybean than maize because of better light penetration during establishment in soybean systems (Mohammed et al., 2020); future cultivars could be developed for successfully interseeding in maize systems.

Winter annual cover crops have both costs and benefits associated with them, though they lack the direct profits associated winter annual oilseeds. The largest direct costs of cover crops are associated with planting and management (Bergtold et al., 2019). High seed cost was identified as one of the highest barriers of adoption for farmers [Conservation Technology Information Center (CTIC) and Sustainable Agriculture Research and Education (SARE), 2013]. Cover crops can provide direct benefits, such as higher cash crop yields, though that is highly dependent on environmental factors (Bergtold et al., 2019). Cover crops can provide indirect benefits by adding or recovering nutrients to the soil to reduce N input need, though cost savings are dependent on fertilizer costs (Snapp et al., 2005). Cover crops can also provide indirect benefits through weed control and consequent herbicide savings. Cost-share programs are often cited as a method to incentivize cover crop adoption by reducing short-term risks (Plastina et al., 2020). Payments through the Environmental Quality Incentives Program (EQIP) were found to have a statistically significant, positive effect on cover crop adoption at the county-level in the U.S. Corn Belt, but payments through the Conservation Stewardship Program (CSP) reduced county-level cover crop adoption (Park et al., 2023; Surdoval et al., 2024). These opposing effects underscore that not all cost-share programs are created equal. One proposed reason for the success of EQIP is its narrow focus on new projects likely to be successful in encouraging new conservation practices (Park et al., 2023). However, cost-share programs should not just remove barriers to initial adoption but also help provide a path toward long-scale feasibility (Thompson et al., 2021).

4.3 Transition scenario tradeoffs for perennial cropping systems

Because agroforestry and pastured livestock systems represent a more significant transformation in the current MS system than either winter annual cropping system, we conservatively estimate their environmental impacts on 50% of marginally productive MS acreage. We estimate that pastured livestock systems and agroforestry systems would offer similar environmental benefits in terms of nitrate loss, but pastured livestock systems could have an order-of-magnitude greater reduction in topsoil loss. These results account for the presence of animals by considering grazed vs. un-grazed systems, but there is uncertainty around these outcomes because of the paucity of studies directly comparing maize/soybean systems to pastured livestock systems. More research is necessary to fully understand this relationship and its impacts on environmental outcomes.

While pastured livestock systems have tremendous potential for C sequestration—more than any other tested agricultural practice on the Mollisol soils of the region (Sanford, 2014)—they are also expected to increase GHG emissions relative to a MS system due to the emissions from steers. Some research indicates that emissions per steer can be offset by soil carbon sequestration (Stanley et al., 2018). In this analysis, we only account for C sequestration due to conversion of row crops to perennial forage; we do not estimate additional C sequestration due to the specific impacts of adding cattle to the landscape. Most grazing-mediated effects on soil C sequestration have been attributed to effects on the relative abundance of C3 vs. C4 grasses in grassland systems (Derner et al., 2006, 2019), which may not be relevant in managed forage system. However, livestock enteric fermentation plus CH_4 and N_2O from manure management account for about a third of agricultural GHG emissions in the U.S. (U.S. Department of Agriculture, 2021). The decrease in pasture-based livestock operations in the Midwest over the last 60 years demonstrates the economic challenges of this system, given current agricultural incentives (Sulc and Tracy, 2007), and the proposed increase in pastured livestock in the UMR Foodscape would outstrip current abattoir capacity. However, with cost-sharing programs, these systems can represent a cost-reduction in rearing livestock due to the reduction in feed costs compared to an operation which must purchase grain feed (Winsten, 2024). Several policy interventions have been proposed to lower the cost of production for small and midsize regenerative grazing operations, such as improving grass-fed labeling standards to better accommodate the needs of small-scale farmers and encouraging supply chain innovations like cost-share for new infrastructure or state-run meat processing (Spratt et al., 2021).

Our analysis suggests agroforestry systems have the greatest potential to reduce GHG emissions compared to MS systems on a per-hectare basis, due to the combined potential of trees to sequester C (Udawatta and Jose, 2011) and the reduction in fertilizer needs for the system (Kim and Isaac, 2022). Agroforestry also reduces nitrate and topsoil loss, suggesting the greatest environmental value of the four systems studied. However, there is currently limited economic viability for agroforestry systems in the region. The most likely agroforestry crop for the region is hybrid hazelnut (*C. americana* × *C. avellana*; Braun et al., 2019). Despite growing global demand for hazelnuts, most hazelnuts grown in the U.S. are cultivated in Oregon (Smith and Mehlenbacher, 2023), and the current yield of hybrid hazelnuts in the UMR foodscape region is too low to support commercial production (Fischbach et al., 2011). Efforts are underway to improve hybrid hazelnut germplasm (Braun et al., 2019) for cultivation in this region, with additional financial support from USDA Climate-Smart Commodity grants for agroforestry establishment and production. Hazelnut production requires large upfront expenditures, and trees take at least 4–5 years to bear fruit. Hazelnuts must be harvested and de-husked with specialized equipment that is not yet widespread in the region, though groups in the UMR Foodscape are working to develop a processing line (Braun and Jensen, 2015). The broad environmental benefits of agroforestry systems suggest that there would be regional benefits to implementing cost-share programs to assist producers with start-up costs alongside investments into hazelnut processing infrastructure.

4.4 Environmental impacts beyond our assessment

We estimated environmental outcomes with the best available sources, but limitations exist in data availability for this work. For one, the small number of empirical studies and large standard deviations in data underscore the importance of continuing to collect data on these novel agricultural systems, especially through field trials. Also, this work focused on agricultural transition scenarios in isolation, but producers could, and will likely need to if we are to meet ambitious environmental goals, incorporate several transition pathways at once, such as incorporating livestock into an agroforestry system. Evidence suggests that implementing multiple diversification strategies, rather than one strategy, improves cash crop yields, biodiversity, and ecosystem services (Estrada-Carmona et al., 2022; Jones et al., 2023; Rasmussen et al., 2024). Additional studies on the environmental outcomes of stacking practices, as well as effective long-term monitoring, are lacking and will be critical to transformation.

To scale up any of these transition scenarios, the region would need to undergo significant infrastructure changes, and many approaches would come with their own potential environmental externalities. For example, scaling up winter annual cover crop adoption to 50% of all land currently occupied by MS systems would require a proportional increase in seed production, possibly necessitating the conversion of cultivated lands from cash crops or natural systems to produce cover crop seeds (Runck et al., 2020). While maize seed production takes <0.5% of land devoted to the crop, even a high-seed-yielding cover crop, such as rye, would need an average of 12 times as much land (Runck et al., 2020). Scaling up winter annual oilseed adoption would also require a proportional increase in seed production, though there are currently fewer estimates of seed yield among these plants to predict how much land would be needed. This reality underscores the critical role that cover crop and oilseed breeding research can play in improving seed yield, though pennycress and winter camelina are both currently bred with seed yield as a consideration (Ott et al., 2019).

These realities highlight the importance of a systems-level approach to agricultural transformation. None of the proposed transition scenarios can be implemented without tremendous changes to infrastructure, markets, and agricultural policy, or without increasing farmers' participation in decision-making and adoption of practices. To carry out this transition in the UMR foodscape, an assessment of environmental outcomes is one early step in a multi-sector effort that must combine scientific research into improved cultivars, economic incentives to build out supply chains, and opportunities for farmers to test these strategies without risking their livelihoods. We have focused on reducing local impacts, including local production of greenhouse gases. But widespread adoption of these new systems would have global consequences, which are critical to explore in future work.

Data availability statement

Publicly available datasets were analyzed in this study. This data can be found here: COMET-Planner: https://pln-50-ui-231219-dot-comet-201514.appspot.com/download; AgEvidence: https://www.agevidence.org/#/us-corn-belt; OpTIS: https://www.ctic.org/optis.

Author contributions

RF: Conceptualization, Data curation, Formal analysis, Funding acquisition, Investigation, Methodology, Project administration, Resources, Software, Validation, Visualization, Writing – original draft, Writing – review & editing. AL: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Supervision, Validation, Visualization, Writing – review & editing. EZ: Formal analysis, Methodology, Visualization, Writing – review & editing. LK: Conceptualization, Resources, Supervision, Writing – review & editing. JG: Resources, Supervision, Writing – review & editing.

Funding

The author(s) declare that financial support was received for the research and/or publication of this article. Salary was provided through AFRI NIFA predoctoral fellowship (1030772). Publishing costs were supported through a grant from the Ballmer Family Foundation.

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 author(s) declared that they were an editorial board member of Frontiers, at the time of submission. This had no impact on the peer review process and the final decision.

Publisher's note

Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fsufs.2025.1499410/full#supplementary-material

References

Albers, M. A., Dobos, R. R., and Robotham, M. P. (2022). *User Guide for the National Commodity Crop Productivity Index (NCCPI), Version 3.0*. Washington, DC: United States Department of Agriculture, Natural Resources Conservation Service, Soil and Plant Science Division. Available online at: https://www.nrcs.usda.gov/sites/default/files/2023-01/NCCPI-User-Guide.pdf (accessed May 13, 2025).

Atwood, L., Gannett, M., and Wood, S. A. (2024). AgEvidence: a dataset to explore agro-ecological effects of conservation agriculture. *Sci. Data* 11:581. doi: 10.1038/s41597-024-03415-9

Atwood, L. W., and Wood, S. A. (2020). *AgEvidence US: Agro-Environmental Responses of Conservation Agricultural Practices Published from 1980 to 2020*. Arlington, VA: The Nature Conservancy.

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

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

Berti, M., Johnson, B., Ripplinger, D., Gesch, R., 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

Bossio, D., Obersteiner, M., Wironen, M., Jung, M., Wood, S., Folberth, C., et al. (2021). *Foodscapes: Toward Food System Transition*. The Nature Conservancy, International Institute for Applied Systems Analysis, and SYSTEMIQ.

Braun, L., and Jensen, J. (2015). *Growing Hybrid Hazelnuts*. Rural Advantage.

Braun, L. C., Demchik, M. C., Fischbach, J. A., Turnquist, K., and Kern, A. (2019). Yield, quality and genetic diversity of hybrid hazelnut selections in the Upper Midwest of the USA. *Agroforest Syst.* 93, 1081–1091. doi: 10.1007/s10457-018-0209-7

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

Castle, S. E., Miller, D. C., and Wardropper, C. B. (2025). Mapping the social-ecological suitability of agroforestry in the US Midwest. *Environ. Res. Lett.* 20:024041. doi: 10.1088/1748-9326/adab09

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 a maize-soybean cropping system. *Agric. Syst.* 189:103062. doi: 10.1016/j.agsy.2021.103062

Comly, H. H. (1945). Cyanosis in infants caused by nitrates in well water. *JAMA* 129:112. doi: 10.1001/jama.1945.02860360014004

Conservation Technology Information Center (CTIC) and Sustainable Agriculture Research and Education (SARE) (2013). *2012 - 2013 Cover Crop Survey*. Available online at: https://www.sare.org/wp-content/uploads/SARE-CTIC-CC-Survey-Report-V2.8.pdf (accessed May 13, 2025).

Conway, T. M. (2023). An agroecological turn in intermediating sustainability transitions with continuous living cover. *Front. Sustain. Food Syst.* 7:1009195. doi: 10.3389/fsufs.2023.1009195

David, M. B., Drinkwater, L. E., and McIsaac, G. F. (2010). Sources of nitrate yields in the Mississippi River Basin. *J. Env. Qual.* 39, 1657–1667. doi: 10.2134/jeq2010.0115

De Roos, A. J., Ward, M. H., Lynch, C. F., and Cantor, K. P. (2003). Nitrate in public water supplies and the risk of colon and rectum cancers. *Epidemiology* 14, 640–649. doi: 10.1097/01.ede.0000091605.01334.d3

Derner, J. D., Augustine, D. J., and Frank, D. A. (2019). Does grazing matter for soil organic carbon sequestration in the Western North American great plains? *Ecosystems* 22, 1088–1094. doi: 10.1007/s10021-018-0324-3

Derner, J. D., Boutton, T. W., and Briske, D. D. (2006). Grazing and ecosystem carbon storage in the North American great plains. *Plant Soil* 280, 77–90. doi: 10.1007/s11104-005-2554-3

Ecotone Analytics, Forever Green Partnership, and Friends of the Mississippi River (2022). *Putting Down Roots: Analyzing the Economic and Environmental Benefits of Continuous Living Cover for Minnesota's Farmers, Water and Climate*. Available online at: fmr.org/CLC-Report (accessed May 13, 2025).

Emmett, B. D., O'Brien, P. L., Malone, R. W., Rogovska, N., Kovar, J. L., Kohler, K., et al. (2022). Nitrate losses in subsurface drainage and nitrous oxide emissions from a winter camelina relay cropping system reveal challenges to sustainable intensification. *Agric. Ecosyst. Environ.* 339:108136. doi: 10.1016/j.agee.2022.108136

Estrada-Carmona, N., Sánchez, A. C., Remans, R., and Jones, S. K. (2022). Complex agricultural landscapes host more biodiversity than simple ones: a global meta-analysis. *Proc. Natl. Acad. Sci. U.S.A.* 119:e2203385119. doi: 10.1073/pnas.2203385119

Fischbach, J., Demchik, M., Braun, L., and Wyse, D. (2011). *Hazelnut Production Potential in the Upper Midwest: A Report on Hybrid Hazelnut Yields*. Ashland/Bayfield County: UW Extension.

Forever Green Initiative (2021). *Forever Green Introduction Packet*. St. Paul, MN: University of MInnesota. Available online at: https://forevergreen.umn.edu/crops (accessed July 25, 2024).

Geels, F. W., and Schot, J. (2007). Typology of sociotechnical transition pathways. *Res. Policy* 36, 399–417. doi: 10.1016/j.respol.2007.01.003

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

Gregg, S., Strock, J. S., Gesch, R. W., Coulter, J. A., and Garcia Y Garcia, A. (2024). Rate and time of nitrogen fertilizer application for winter camelina. *Agron. J.* 116, 1804–1816. doi: 10.1002/agj2.21610

Hamlin, Q. F., Martin, S. L., Kendall, A. D., and Hyndman, D. W. (2022). Examining relationships between groundwater nitrate concentrations in drinking water and landscape characteristics to understand health risks. *GeoHealth* 6:e2021GH000524. doi: 10.1029/2021GH000524

Hunt, E. D., Birge, H. E., Laingen, C., Licht, M. A., McMechan, J., Baule, W., et al. (2020). A perspective on changes across the U.S. Corn Belt. *Environ. Res. Lett.* 15:071001. doi: 10.1088/1748-9326/ab9333

Inoue-Choi, M., Jones, R. R., Anderson, K. E., Cantor, K. P., Cerhan, J. R., Krasner, S., et al. (2015). Nitrate and nitrite ingestion and risk of ovarian cancer among postmenopausal women in Iowa. *Int. J. Cancer* 137, 173–182. doi: 10.1002/ijc.29365

Iowa Department of Agriculture and Land Stewardship (2017). *Iowa Nutrient Reduction Strategy*.

Jones, C. S., Nielsen, J. K., Schilling, K. E., and Weber, L. J. (2018). Iowa stream nitrate and the Gulf of Mexico. *PLoS ONE* 13:e0195930. doi: 10.1371/journal.pone.0195930

Jones, R. R., Weyer, P. J., DellaValle, C. T., Inoue-Choi, M., Anderson, K. E., Cantor, K. P., et al. (2016). Nitrate from drinking water and diet and bladder cancer among postmenopausal women in Iowa. *Environ. Health Perspect.* 124, 1751–1758. doi: 10.1289/EHP191

Jones, S. K., Sánchez, A. C., Beillouin, D., Juventia, S. D., Mosnier, A., Remans, R., et al. (2023). Achieving win-win outcomes for biodiversity and yield through diversified farming. *Basic Appl. Ecol.* 67, 14–31. doi: 10.1016/j.baae.2022.12.005

Jung, M., Boucher, T. M., Wood, S. A., Folberth, C., Wironen, M., Thornton, P., et al. (2024). A global clustering of terrestrial food production systems. *PLoS ONE* 19:e0296846. doi: 10.1371/journal.pone.0296846

Kim, D.-G., and Isaac, M. E. (2022). Nitrogen dynamics in agroforestry systems. A review. *Agron. Sustain. Dev.* 42:60. doi: 10.1007/s13593-022-00791-7

Köhler, J., Geels, F. W., Kern, F., Markard, J., Onsongo, E., Wieczorek, 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

Liu, R., Wells, M. S., and Garcia Y Garcia, A. (2019). Cover crop potential of winter oilseed crops in the Northern U.S. Corn Belt. *Arch. Agron. Soil Sci.* 65, 1845–1859. doi: 10.1080/03650340.2019.1578960

Martin, L. (1965). *The Physical Geography of Wisconsin, 3rd Edn*. Madison, WI: University of Wisconsin Press.

Masarik, K. C., Norman, J. M., and Brye, K. R. (2014). Long-term drainage and nitrate leaching below well-drained continuous corn agroecosystems and a prairie. *JEP* 05, 240–254. doi: 10.4236/jep.2014.54028

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

Mori, G. O., Gold, M., and Jose, S. (2017). "Specialty crops in temperate agroforestry systems: sustainable management, marketing and promotion for the midwest region of the U.S.A.," in *Integrating Landscapes: Agroforestry for Biodiversity Conservation and Food Sovereignty*, ed. F. Montagnini (Cham: Springer International Publishing), 331–366. doi: 10.1007/978-3-319-69371-2_14

Mueller, S. M., Messina, C. D., and Vyn, T. J. (2019). Simultaneous gains in grain yield and nitrogen efficiency over 70 years of maize genetic improvement. *Sci. Rep.* 9:9095. doi: 10.1038/s41598-019-45485-5

Nouri, A., Lukas, S., Singh, S., Singh, S., and Machado, S. (2022). When do cover crops reduce nitrate leaching? A global meta-analysis. *Glob. Chang. Biol.* 28, 4736–4749. doi: 10.1111/gcb.16269

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

Park, B., Rejesus, R. M., Aglasan, S., Che, Y., Hagen, S. C., and Salas, W. (2023). Payments from agricultural conservation programs and cover crop adoption. *Appl. Eco Perspect. Pol.* 45, 984–1007. doi: 10.1002/aepp.13248

Perez, M., and Cole, E. (2020). *A Guide to Water Quality, Climate, Social, and Economic Outcomes Estimation Tools: Quantifying Outcomes to Accelerate Farm Conservation Practice Adoption*. Washington, DC: American Farmland Trust. Available online at: farmlandinfo.org/publications/guide-to-outcomes-estimation-tools (accessed July 23, 2024).

Pilon, C., Moore, P. A., Pote, D. H., Martin, J. W., Owens, P. R., Ashworth, A. J., et al. (2019). Grazing management and buffer strip impact on nitrogen runoff from pastures fertilized with poultry litter. *J. Env. Qual.* 48, 297–304. doi: 10.2134/jeq2018.04.0159

Pilon, C., Moore, P. A., Pote, D. H., Pennington, J. H., Martin, J. W., Brauer, D. K., et al. (2017). Long-term effects of grazing management and buffer strips on soil erosion from pastures. *J. Env. Qual.* 46, 364–372. doi: 10.2134/jeq2016.09.0378

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

Prior, J. (1991). *Landforms of Iowa*. University of Iowa Press.

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

Rabalais, N. N., Turner, R. E., and Wiseman, W. J. (2002). Gulf of Mexico Hypoxia, A.K.A. "the dead zone." *Annu. Rev. Ecol. Syst.* 33, 235–263. doi: 10.1146/annurev.ecolsys.33.010802.150513

Rabotyagov, S. S., Campbell, T. D., White, M., Arnold, J. G., Atwood, J., Norfleet, M. L., et al. (2014). Cost-effective targeting of conservation investments to reduce the northern Gulf of Mexico hypoxic zone. *Proc. Natl. Acad. Sci. U.S.A.* 111, 18530–18535. doi: 10.1073/pnas.1405837111

Randall, G. W., and Iragavarapu, T. K. (1995). Impact of long-term tillage systems for continuous corn on nitrate leaching to tile drainage. *J. Env. Qual.* 24, 360–366. doi: 10.2134/jeq1995.00472425002400020020x

Ranjan, P., Church, S. P., Floress, K., and Prokopy, L. S. (2019). Synthesizing conservation motivations and barriers: what have we learned from qualitative studies of farmers' behaviors in the United States? *Soc. Nat. Resour.* 32, 1171–1199. doi: 10.1080/08941920.2019.1648710

Rasmussen, L. V., Grass, I., Mehrabi, Z., Smith, O. M., Bezner-Kerr, R., Blesh, J., et al. (2024). Joint environmental and social benefits from diversified agriculture. *Science* 384, 87–93. doi: 10.1126/science.adj1914

Resurreccion, E. P., Roostaei, J., Martin, M. J., Maglinao, R. L., Zhang, Y., and Kumar, S. (2021). The case for camelina-derived aviation biofuel: sustainability underpinnings from a holistic assessment approach. *Ind. Crops Prod.* 170:113777. doi: 10.1016/j.indcrop.2021.113777

Robertson, D. M., and Saad, D. A. (2021). Nitrogen and phosphorus sources and delivery from the Mississippi/Atchafalaya River Basin: an update using 2012 SPARROW models. *J. Am. Water Resour. Assoc.* 57, 406–429. doi: 10.1111/1752-1688.12905

Runck, B. C., Khoury, C. K., Ewing, P. M., and Kantar, M. (2020). The hidden land use cost of upscaling cover crops. *Commun. Biol.* 3:300. doi: 10.1038/s42003-020-1022-1

Rund, Q. B., Williams, R., and Fixen, P. E. (2010). *A Preliminary Nutrient Use Geogrpahic Information System (NuGIS) for the U.S. International Plant Nutrition Institute*. Available online at: http://www.ipni.net/ipniweb/portal.nsf/0/5d3b7dfafc8c276885257743005aa07a/FILE/1203%20FIXEN%20GPSFC%20NuGIS%20FINAL.pdf (accessed June 5, 2024).

Sanford, G. R. (2014). "Perennial grasslands are essential for long term SOC storage in the Mollisols of the North Central USA," in Soil Carbon, eds. A. E. Hartemink and K. McSweeney (Cham: Springer International Publishing), 281–288. doi: 10.1007/978-3-319-04084-4_29

Sexton, B. T., Moncrief, J. F., Rosen, C. J., Gupta, S. C., and Cheng, H. H. (1996). Optimizing nitrogen and irrigation inputs for corn based on nitrate leaching and yield on a coarse-textured soil. *J. Env. Qual.* 25, 982–992. doi: 10.2134/jeq1996.00472425002500050008x

Singer, J., Nusser, S. M., and Alf, C. (2007). Are cover crops being used in the US corn belt? *J. Soil Water Conserv.* 62, 353–358. doi: 10.1080/00224561.2007.12435983

Smith, D. C., and Mehlenbacher, S. A. (2023). Hazelnut genetic improvement at Oregon State University, a summary of the breeding effort since 1969. *Acta Hortic.* 35–40. doi: 10.17660/ActaHortic.2023.1379.6

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

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

Stanley, P. L., Rowntree, J. E., Beede, D. K., DeLonge, M. S., and Hamm, M. W. (2018). Impacts of soil carbon sequestration on life cycle greenhouse gas emissions in Midwestern USA beef finishing systems. *Agric. Syst.* 162, 249–258. doi: 10.1016/j.agsy.2018.02.003

Sulc, R. M., and Tracy, B. F. (2007). Integrated crop–livestock systems in the U.S. Corn Belt. *Agronomy J.* 99, 335–345. doi: 10.2134/agronj2006.0086

Surdoval, A., Jain, M., Blair, E., Wang, H., and Blesh, J. (2024). Financial incentive programs and farm diversification with cover crops: assessing opportunities and challenges. *Environ. Res. Lett.* 19:044063. doi: 10.1088/1748-9326/ad35d8

Swan, A., Toureene, C., Easter, M., Chambers, A., Brown, K., Williams, S. A., et al. (2023). *COMET-Planner: Carbon and Greenhouse Gas Evaluation for NRCS Conservation Practice Planning*. Available online at: www.comet-planner.com (accessed May 13, 2025).

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 Future* 10:e2021EF002396. doi: 10.1029/2021EF002396

Thaler, E. A., Larsen, I. J., and Yu, Q. (2021). The extent of soil loss across the US Corn Belt. *Proc. Natl. Acad. Sci. U.S.A.* 118: e1922375118. doi: 10.1073/pnas.1922375118

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

U.S. Department of Agriculture (2021). *Data from: U.S. Agriculture and Forestry Greenhouse Gas Inventory: 1990–2018*.

Udawatta, R. P., and Jose, S. (2011). "Carbon sequestration potential of agroforestry practices in temperate North America," in *Carbon Sequestration Potential of Agroforestry Systems*, eds. B. M. Kumar and P. K. R. Nair (Dordrecht: Springer Netherlands), 17–42. doi: 10.1007/978-94-007-1630-8_2

USDA NASS (2022). *Census of Agriculture*.

USDA National Agricultural Statistical Service (2008). *Cropland Data Layer (CDL)*. Available online at: https://croplandcros.scinet.usda.gov/ (accessed May 13, 2025).

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. *IJERPH* 15:1557. doi: 10.3390/ijerph15071557

Winsten, J. R. (2024). Low-overhead dairy grazing: a specific solution to a vexing problem. *J. Soil Water Conserv.* 79, 27A−31A. doi: 10.2489/jswc.2024.0122A

Wolz, K. J., Branham, B. E., and DeLucia, E. H. (2018). Reduced nitrogen losses after conversion of row crop agriculture to alley cropping with mixed fruit and nut trees. *Agric. Ecosyst. Environ.* 258, 172–181. doi: 10.1016/j.agee.2018.02.024

World Wildlife Fund (2023). *The Role of Regenerative Agriculture to Drive Food Systems Transformation*. World Wildlife Fund. Available online at: https://www.worldwildlife.org/publications/wwf-perspectives-paper-the-role-of-regenerative-agriculture-to-drive-food-systems-transformation (accessed May 20, 2024).

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