Wilding

THE SCIENCE AND VISION BEHIND THE WILD GRID

Co-authored by Dan Kane PhD, Science Director, Mad Agriculture
and Omar de Kok-Mercado MS, Wilding Director, Mad Agriculture

MAD!

Executive Summary

The expansion of row crop agriculture across the United States over the past two centuries has removed hundreds of millions of acres of native habitat from the landscape. The ecological consequences are measurable and ongoing. North American bird populations have declined by 29% since 1970 (Rosenberg et al., 2019). A third of native bee species are at elevated extinction risk (Cornelisse et al., 2025). Agricultural nutrient loading has become a primary driver of stream degradation and coastal hypoxia (US EPA, 2017). And billions of tons of carbon dioxide have been lost from soils (Sanderman et al., 2017). What habitat remains is severely fragmented, making it less likely to be useful for maintaining or restoring biodiversity.

Field-scale evidence indicates that strategically sited native perennial plantings can mitigate the ecological impacts of agriculture and provide crucial habitat resources. In small headwater catchments in central Iowa, prairie plantings covering 10% of the contributing area, sited at footslope and contour positions, reduced annual nitrate-N loss by 67%, total phosphorus loss by 90%, and sediment loss by more than 95% (Helmers et al., 2012; Zhou et al., 2014). Plantings can more than double bird species richness in working fields and increase pollinator abundance several-fold (Schulte et al., 2017). These outcomes are achievable on ground that is already underperforming agronomically and the economic case for continued row cropping is weak.

Scaling these outcomes beyond isolated parcels requires more than the existing conservation toolkit provides. Programs like the United States Department of Agriculture (USDA) Conservation Reserve Program (CRP) and the Environmental Quality Incentives Program (EQIP) operate at the parcel level and on contract timescales that fall short of what landscape-level connectivity and durability require.

This paper lays out a framework called ***Wilding:*** the strategic integration of perennial cover into working agricultural landscapes to build functionally connected, multi-use habitat network. Mad Agriculture's long-horizon commitment is to work with farmers and a broad coalition of partners to establish 65 million acres of Wilding through U.S. agricultural landscapes over fifty years, a network we call the ***Wild Grid***. The figure reflects the 20% semi-natural habitat threshold the ecological literature identifies as necessary to maintain essential ecosystem services (Garibaldi et al., 2021; Mohamed et al., 2024). Of the 12,412 majority-cropland sub-watersheds in the U.S., 70% currently fall below that threshold, representing an estimated planting gap of 22.9 million new acres within the larger 65 million acre network (Kane, 2026).

The paper works through how the Wild Grid gets built across four nested scales—field, community, regional, and continental—and what ecological evidence, financial structures, and governance precedents support the approach at each. A pilot project in the Driftless Area of southwest Wisconsin is the first test. Wilding is both a plan and a hypothesis. Our intent is to test it through demonstration.

WHAT IS WILDING?

Wilding is an ecological framework built around the strategic integration of perennial cover—native prairies, silvopasture systems, perennial crops, and riparian buffers—into working agricultural landscapes to reconstruct habitat connectivity and build a functionally connected, multi-use habitat network capable of delivering ecosystem services, supporting species movement, and maintaining biodiversity across watershed and regional scales. Sustaining that network requires financial structures and commercial opportunities that make coordinated stewardship economically viable for the farmers and landowners doing the work.

Wilding is not rewilding. Rewilding aims to restore landscapes toward pre-human conditions—removing human management, reintroducing apex predators, letting succession proceed without intervention. That framing assumes prior ecological states are clearly definable and recoverable, and that human presence must be removed or diminished. Wilding instead starts from the premise that human presence in agricultural landscapes is unavoidable. Thus, recovering prior ecological states in agricultural landscapes is not fully possible and would limit the possibility of using perennial cover for agricultural production while also meeting ecological goals.

Full rewilding of agricultural landscapes would also be economically and logistically untenable at the scale required. The United States currently produces food, feed, and fiber for hundreds of millions of people on more than 300 million acres of cropland. Removing human management from any substantial portion of that land would require either dramatic intensification elsewhere, with its own ecological costs, or significant reductions in agricultural output. Wilding sidesteps that tradeoff entirely. By targeting marginal, sub-profitable acres rather than productive farmland, it asks farmers to convert ground that is already underperforming economically, not ground that is feeding anyone reliably.

Rewilding, as defined by Carver et al. (2021), aims to rebuild ecosystems toward a state that would have existed had human disturbance not occurred, reducing human control until the system becomes self-sustaining. This framing has genuine merit in landscapes where human presence has receded or where large-scale predator reintroduction is feasible. Wilding does not argue against it. Wilding is complementary: a framework suited to working agricultural lands where human presence is permanent and the economic and social structure of rural communities depends on continued land use. In those landscapes, ecological recovery is not achieved by removing human management but by reorienting it. Some of those landscapes were actively shaped by Indigenous land stewardship for millenia, a history of ecological practice and displacement this paper acknowledges but does not address in full. What Wilding is working to rebuild is not a system without us, but one in which human land management is oriented toward enhancing ecological function rather than diminishing it. Wilding also differs structurally from the existing conservation toolkit. Programs like the United States Department of Agriculture (USDA) Conservation Reserve Program (CRP), Environmental Quality Incentives Program (EQIP), and conservation easements are built to operate at the parcel level—enrolling individual farms, contracting with individual landowners, verifying individual compliance. CRP pays farmers to remove environmentally sensitive land from production and establish permanent vegetative cover. EQIP provides cost-share funding for conservation practices including filter strips, cover crops, and nutrient management. Conservation easements permanently restrict development on ecologically valuable parcels.

Each program is well designed to incentivize individual landowners to make conservation improvements on their own land, and they provide critical resources to farmers (explore this topic further in our Cultivating Conservation resource). But, connectivity is not a parcel-level outcome. A watershed can have substantial CRP enrollment and still present a fully fragmented landscape to species that require linked habitat to persist. These programs have no mechanism for optimizing the spatial arrangement of habitat across property boundaries, for coordinating plantings so they function as a system rather than a collection of individual parcels, or for sustaining that coordination across the timescales that ecological function requires. It requires more intentional design and coordination to create that connectivity and sustain it into the future, and the approach to design and coordination necessarily changes as the work scales.

Wilding is less a set of practices than a framework for building ecological infrastructure from the ground up, beginning at the parcel level and scaling toward the watershed and beyond.

WHAT IS THE WILD GRID?

Mad Agriculture's long-horizon commitment is to work with farmers and a broad coalition of stakeholders to establish 65 million acres of functionally connected, multi-use corridors through U.S.

agricultural landscapes over a fifty-year period. That figure represents roughly 20% of total U.S. cropland, the threshold that the current ecological literature suggests is necessary to maintain essential ecosystem services, including pollination, pest control, water filtration, and soil erosion control (Garibaldi et al., 2021; Mohamed et al., 2024). While that literature addresses semi-natural habitat cover broadly rather than connected corridors specifically, the spatial arrangement of that 20% matters as much as the total amount. A fragmented 20% delivers diminished ecological function relative to a connected one: isolated patches cannot support viable wildlife populations, cannot facilitate species movement across the landscape, and are more vulnerable to local extinction. The connectivity dimension of the Wild Grid is designed to ensure that the same 20% threshold delivers the ecosystem services, biodiversity outcomes, and landscape-scale resilience that the literature suggests it should. Critically, native habitat reconstruction requires active, ongoing management. Wilding is built on the premise that this management could create economic opportunity through grazing, perennial crops, silvopasture, and ecosystem service markets, which can make stewardship financially viable for the farmers and landowners doing the work. The result is a mosaic in which ecological function and agricultural enterprise are nested within the same landscape, not competing for it. The Wild Grid is our vision for a continent-spanning network of functionally connected, multi-use habitat operating as ecological infrastructure across working agricultural lands.

The 65 million acre figure describes the total extent of the network, not the volume of new planting required to build it. Existing semi-natural habitat, remnant grassland, riparian corridors,

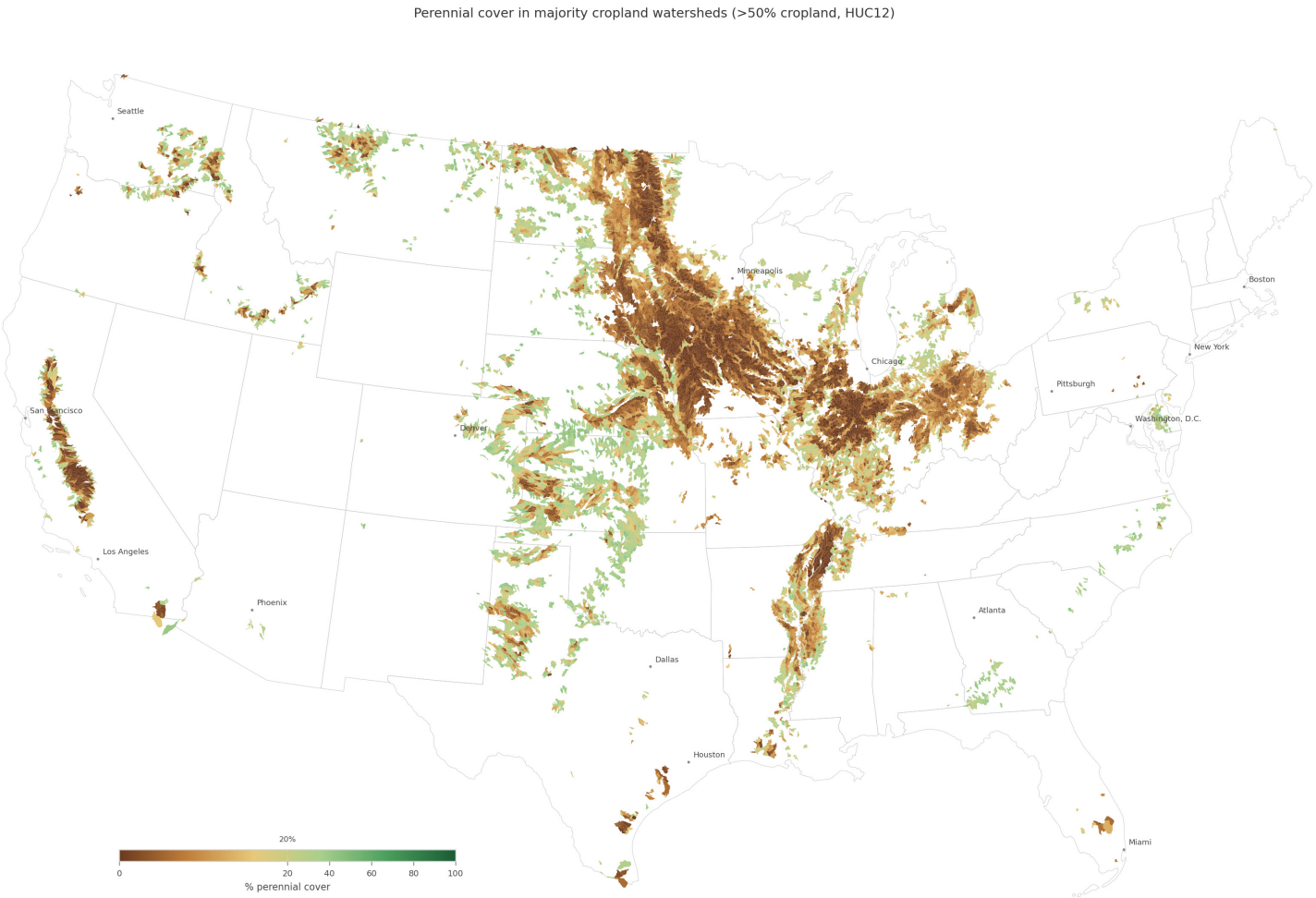

Figure 1. Perennial vegetation cover across majority cropland watersheds in the contiguous United States. Each polygon represents a HUC12 subwatershed with more than 50% of its area in cultivated crops. Color indicates the percentage of the watershed in perennial cover. Of the 12,412 majority-cropland watersheds, 8,729 (70%) have less than 20% perennial cover, encompassing approximately 151 million acres of cropland across roughly 195 million acres of watershed area. Land cover data from the USDA/USGS National Land Cover Database (NLCD 2024); watershed boundaries from the USGS National Watershed Boundary Dataset (WBD), HUC12 subwatershed level.

and conservation lands already contribute to the 20% threshold in some watersheds. To understand how much new ground would need to come into perennial cover, we analyzed the 12,412 sub-watersheds in the U.S. with majority cropland cover (Kane, 2026). Of those, 70.3%—representing 195 million acres—currently fall below the 20% perennial cover threshold. Bringing those sub-watersheds up to threshold would require establishing perennial cover on an estimated 22.9 million new acres. That is the planting gap Wilding is designed to close, situated within a larger network that incorporates and connects habitat that already exists.

The 65 million acre target is not a goal Mad Agriculture will accomplish alone, nor is the pathway fully mapped. It is a design orientation large enough to matter ecologically and ambitious enough to require new institutions, new financial instruments, and sustained commitment across multiple generations of practitioners and landowners.

WHY DO WE NEED WILDING?

Millions of acres of habitat and perennial cover have been converted to croplands in the U.S. over the past two centuries. This conversion has resulted in measurable ecological losses, including species declines and diminished ecosystem services. It has also resulted in extensive habitat fragmentation, which can potentially amplify the effects of habitat loss on different species and reduce the effectiveness of isolated restoration efforts. Field-scale evidence indicates that strategically sited native perennial plantings can mitigate the ecological impacts of agriculture and provide crucial habitat resources. These outcomes are achievable on ground that is already underperforming agronomically and the economic case for continued row cropping is weak.

CROPLAND CONVERSION IMPACTS

The expansion of agriculture in the United States over the past four centuries since European colonization has been extensive. Historical reconstructions of land use in the United States indicate that the total area in cropland was negligible prior to colonization but rapidly expanded between 1850 and 1920, increasing by more than 270 million acres as settlers pushed west (Li et al., 2023; Meiyappan and Jain, 2012; Ramankutty and Foley, 1999). The primary sources of that new cropland were forest in the East and grassland in the Midwest and Plains.

According to USDA data, total cropland area peaked in 1949 at 478 million acres before contracting over the latter half of the 20th century to 338 million acres in 2017 (USDA NRCS, 2025). This decrease was largely attributable to the abandonment of cropland in coastal areas, urbanization, and the expansion of the USDA's Conservation Reserve Program (CRP). But starting around 2007, rising commodity prices and biofuel mandates spurred many farmers to put CRP acreage and other grasslands into cropland production in the Midwest and Northern Plains. Between 2008 and 2016 alone, nearly 5 million acres of grassland were converted to crop produc-

Figure 2. Male Bobolink (Dolichonyx oryzivorus), an obligate grassland nester whose North American population has declined by an estimated 65% since 1966. Conversion of grassland to row-crop agriculture is a primary driver. Restoring native perennial cover on subprofitable acres directly addresses the structural cause of the decline. Photo: Paul Danese, via Wikimedia Commons, CC BY-SA 4.0.

tion across twelve Midwestern states (Zhang et al., 2021). In the Northern Plains, roughly 770,000 acres of rangeland were converted to cultivated crops in just a decade (Claassen et al., 2011). That expansion occurred largely in grasslands or wetlands coming out of CRP contracts (Morefield et al., 2016) and on marginal agricultural acreage that provides only minimal gains in yield (Lark et al., 2020).

This massive and ongoing land use change has had enormous ecological impacts, including declines in animal populations, habitat fragmentation, degradation of water quality, and the emission of carbon dioxide from terrestrial carbon sinks such as soils. North American bird populations have declined by an estimated three billion individuals since 1970 with losses spanning biomes and extending to once-common species (Rosenberg et al., 2019). That figure is approximately 29% of 1970 total abundance, which is likely well below historical abundance figures giving shifting baselines and historical habitat loss. More than 22% of native pollinator species in North America face elevated extinction risk, including over a third of native bee species, and habitat loss due to agriculture is the primary driver (Cornelisse et al., 2025).

Habitat fragmentation shapes population persistence through mechanisms that habitat amount alone does not capture. In fragmented landscapes, the spatial arrangement of patches and the permeability of the matrix between them influence gene flow, dispersal behavior, and access to habitat resources (Collinge, 1996; Donald and Evans, 2006; Fahrig and Merriam, 1994). Local extinctions in small patches are common, and regional persistence depends on whether dispersers can move between them (Niebuhr et al., 2015). Habitat area and fragmentation in the surrounding landscape also shape population resilience to climatic extremes such as drought (Oliver et al., 2013). Fragmentation can disrupt the biological processes that agriculture depends on. Landscape structure can suppress natural predator populations more than the pest species they regulate, weakening the pest control services that intact, connected habitats provide (Wen et al., 2024). Conditions at patch edges, where habitat meets cropland or other disturbed land, can reduce plant reproductive output, with downstream consequences for the pollinators and seed dispersers that depend on those plants (Hulting et al., 2025).

Agriculture is the leading source of nutrient and sediment loading to U.S. surface waters. Conversion of perennial cover to row crops accelerates both, as it eliminates the infiltration, uptake, and physical filtration functions that perennial cover provides. These impacts are enhanced when cover is removed at field edges or slopes where flow accumulates. The five million acres of grassland converted across twelve Midwestern states between 2008 and 2016 increased annual soil erosion by nearly 8%, nitrogen loss by 3.7%, and soil organic carbon loss by 5.6% relative to existing cropland—disproportionate impacts from a 2.5% increase in total cropland area (Zhang et al., 2021). Downstream, the consequences are measurable: agricultural land use is a primary driver of degraded biological integrity in U.S. streams, and nutrient loading from row-cropped watersheds remains a principal cause of hypoxic zones in receiving waters, such as the Gulf of Mexico (Riseng et al., 2011; US EPA, 2017).

The conversion of perennial cover to annual row crops has also been responsible for substantial carbon dioxide emissions to the atmosphere through the decomposition of soil organic matter. After conversion, roots shift from deep and persistent to shallow and annual, reducing the belowground inputs that build stable organic matter over time, and tillage breaks open soil aggregates, exposing previously protected carbon to microbial decomposition (Crews and Rumsey, 2017). Estimates based on global soil inventories, digital soil mapping, and historical land-use reconstruction suggest that land use conversion to agriculture has been responsible for the loss of 133 Pg of carbon from soils globally, equivalent to 488 Gt of carbon dioxide emissions (Sanderman et al., 2017). That's a little more than the U.S.'s estimated historical emissions (435 Gt CO2) since 1751 (Ritchie and Roser, 2020).

PRAIRIE PLANTINGS: A CORNERSTONE PRACTICE

Native perennial plantings are among the more promising field-scale approaches to addressing the ecological impacts described above. While Wilding will grow to include a range of ecologically appropriate restoration and reconstruction strategies, prairie plantings are the cornerstone practice the program is built on for two reasons. First, they were historically the dominant vegetation type across the agricultural landscapes of the Midwest and Great Plains where Wilding is initially focused. Second, field evidence consistently identifies them as among the most cost-effective conservation practices available, delivering measurable improvements across water quality, biodiversity, and soil function (Schulte et al., 2017).

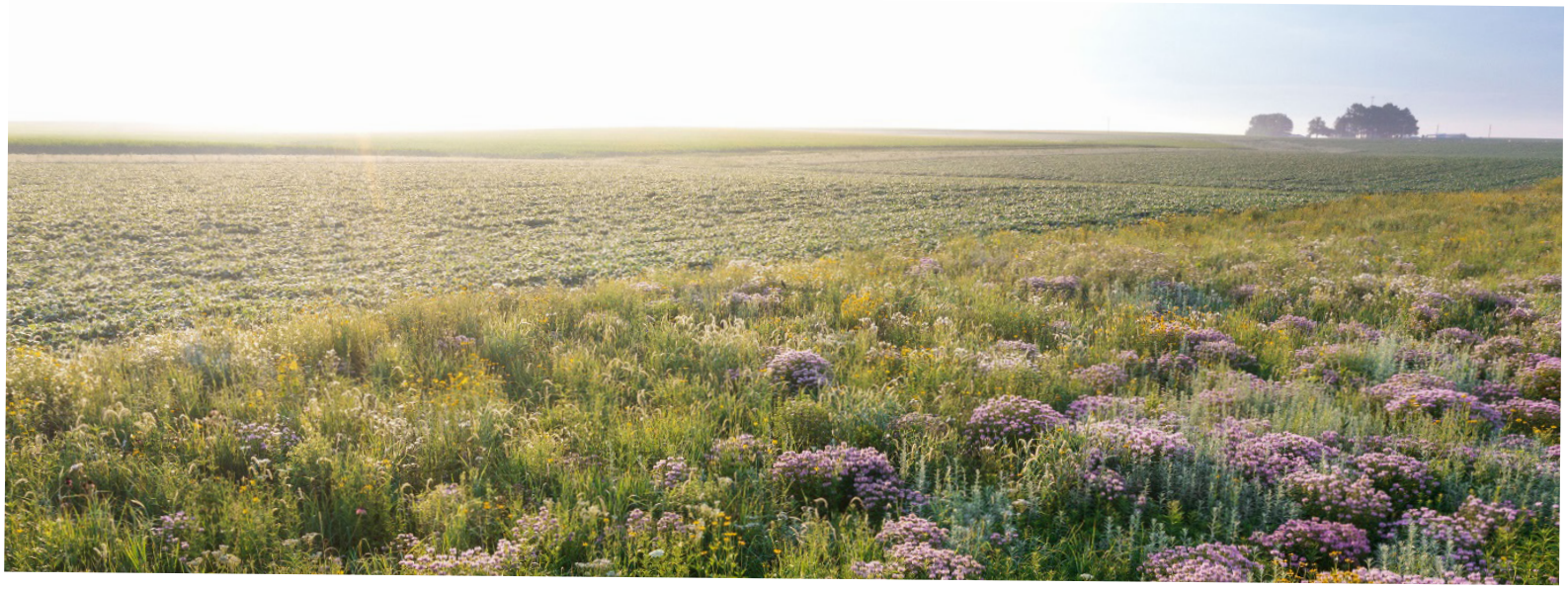

Figure 3. A prairie planting adjacent to a crop field in Grundy County, IA (photo credit: Omar de Kok-Mercado, Mad Agriculture, Courtesy of Iowa State University). Native perennial prairie plantings restore structural and floristic diversity to working landscapes, providing nesting and foraging habitat for grassland-dependent species, supporting pollinator communities, and intercepting sediment and nutrients before they reach surface waters.

Much of the field evidence on prairie plantings comes from the STRIPS program, a coalition led by Iowa State University of researchers from several universities, USDA scientists, conservation agencies, and participating farmers who have run trials across the Upper Midwest for several years.

In small experimental watersheds in central Iowa, prairie plantings covering 10% of the contributing area, sited at footslope and contour positions, reduced annual nitrate-N loss in surface runoff by 67%, total nitrogen by 84%, total phosphorus by 90%, and overland flow by 37% (Zhou et al., 2014). Sediment trapping efficiencies in the same trials reached 96.8% (Helmers et al., 2012). At plot and field scales, prairie strips have been shown to reduce nitrate leaching through the soil profile by 23% (Dutter et al., 2023, 2025; Nelson, 2022). Whether these field-scale outcomes aggregate proportionally as plantings extend across larger watersheds is an open empirical question, and one the Wild Grid is designed in part to answer.

In another trial, prairie plantings doubled bird species richness and increased pollinator abundance up to 3.5-fold compared to fields without plantings (Schulte et al., 2017). Butterfly diversity increased as well (Kemmerling et al., 2023), and field-level bird community composition measurably shifted toward grassland-dependent species (Giese et al., 2024).

Field borders that have never been converted to cropland consistently show higher soil organic carbon than adjacent crop fields (Chiartas et al., 2022; Drexler et al., 2024). A similar pattern has been documented in Conservation Reserve Program (CRP) lands. Early work by Gebhart et al. (1994) found that conversion from cropland to perennial grass cover under the CRP increased soil organic carbon across a range of Great Plains sites, and subsequent work in semi-arid CRP grasslands has confirmed measurable sequestration potential, though rates vary considerably by soil type and prior land use (Li et al., 2017). But the picture is more complicated in practice. Effects are not consistently detectable in the first several years of establishment, and surface carbon gains may be partially offset by losses deeper in the soil profile (Rutkoski and Evans, 2025; Yang et al., 2022). Nonetheless, perennial cover compares favorably to continued cropland use. In a long-term trial in WI, perennial grassland systems maintained soil carbon stocks over decades when annual row crop systems did not (Dietz et al., 2024). Analyses using biogeochemical process-based models suggest that perennial field borders are among the more promising long-term pathways to increasing soil carbon in cropland areas, with per-hectare removal rates that are relatively high even where applicable land area is constrained (Pett-Ridge et al., 2023).

THE LIMITS OF PARCEL-LEVEL CONSERVATION

The field-scale evidence reviewed above establishes that strategically sited native perennial plantings can substantially reduce sediment and nutrient loss, support biodiversity, and build soil carbon on the acres they occupy. What that evidence does not establish is whether those bene-

fits compound when plantings are coordinated across farms so as to reduce habitat fragmentation. As detailed above the spatial arrangement of habitat, not just the total amount, may be critical to determining its ecological function (Fahrig and Merriam, 1994; Niebuhr et al., 2015). A core hypothesis of Wilding is that those benefits do indeed compound and are worth pursuing, and that the degree of coordination required cannot be supported through existing conservation programs alone. Mad Agriculture will test this hypothesis through the work.

Programs like CRP and EQIP are built around the creation and execution of contracts at the parcel or farm level, enrolling individual farms, contracting with individual landowners, verifying individual compliance. These programs have no policy support or mandate to encourage the USDA, farmers, or technical service providers to implement them in a coordinated manner to reduce fragmentation. More recent initiatives, including USDA's Regional Conservation Partnership Program and Working Lands for Wildlife, have begun to address landscape-scale coordination, but participation remains voluntary and spatially uncoordinated at the scale required for functional corridor development. A watershed can have substantial CRP enrollment and still present a fully fragmented landscape to species that require linked habitat to persist. No existing program provides a mechanism for optimizing that arrangement across property boundaries at watershed scale or beyond.

Durability compounds the problem. CRP contracts run 10 to 15 years, and EQIP contracts typically run three to five years depending on practice. The timescales on which ecological benefits accrue are considerably longer. A long-term chronosequence at Cedar Creek found that even 91 years after agricultural abandonment, formerly ploughed grasslands had recovered only about three-quarters of the plant diversity and half the productivity of never-ploughed reference sites, and at observed rates of recovery, reaching 80% of reference diversity would take more than two centuries (Isbell et al., 2019). Belowground recovery of soil structure, microbial composition, and root biomass tracks similarly decadal chronosequences (Baer et al., 2015). Post-contract outcomes compound the problem. In the southern Great Plains, more than half of surveyed landowners with expired CRP contracts reported being unable to re-enroll when they tried, and cropland reversion became more likely as time since contract expiration increased (Barnes et al., 2020). The contract structure that makes enrollment tractable at the parcel level creates a durability problem at the landscape level.

The evidence base for native perennial plantings at the field scale is strong, but evidence for if and how those benefits compound when plantings are connected across farms and watersheds is weaker. Landscape-scale connectivity effects on biodiversity and ecosystem services are documented in modeling and observational studies, but controlled restoration experiments at watershed scale remain limited. Wilding is designed to operate in that gap—applying the field-scale evidence at larger scales while generating the data needed to evaluate whether the connectivity hypothesis holds in practice.

HOW DO WE BUILD THE WILD GRID?

In the previous sections we discussed what Wilding and the Wild Grid are, the evidence base for prairie plantings as a cornerstone practice of ecological reconstruction in agricultural landscapes, and how the existing conservation toolkit alone may not be well-suited to the goal of functionally connected multi-use corridors in agricultural landscapes. This section addresses the operational and finance questions that follow from that evidence base: how do individual plantings become a coordinated network, and what financial and governance structures are needed to create a network that's durable over time? The analogy to physical infrastructure is useful here. A road to nowhere has no network value; its value derives from what it connects to. The same logic can apply to habitat. Building toward connectivity requires a design approach that changes as the work scales.

At the field scale (tens to hundreds of acres), the challenge is identifying where to plant and making the financial case for Wilding. At the community scale (thousands to tens of thousands of acres), it is coordinating plantings across property boundaries and creating the shared financial structures that make that coordination durable. At the regional scale (hundreds of thousands to a few million acres) and continental scale (tens of millions of acres), it is aligning those community-level efforts with the larger landscape features—river systems, rights-of-way, public conservation lands—that can carry connectivity across jurisdictions.

FIELD SCALE

The Wild Grid does not begin with the most productive farmland. It begins on the acres that consistently underperform—the poorly drained corners that delay planting, the steep slopes

where erosion outpaces yield, the sandy knolls that exhaust early in a dry summer. These acres require the same inputs as the rest of the field but return less, often falling below breakeven in average or low-price years. Targeting them for conversion to perennial cover is both ecologically and economically sound. The goal is to find the places where the case for continued row cropping is already weak, and where native vegetation can take over without asking farmers to sacrifice reliable income and without meaningfully reducing the supply of agricultural products to the market.

IDENTIFYING MARGINAL ACRES

Research analyzing digital yield maps from combine harvester data across hundreds of Midwest fields has shown that within-field yield variability follows a consistent spatial pattern (Maestrini and Basso, 2018). Portions of fields with low topographic wetness index (i.e., exposed, drier positions) tend to produce yields that are consistently lower than the field average. Areas with very high wetness index (i.e., depressions and swales prone to waterlogging) tend to produce unstable yields, low in wet years when emergence is poor, higher in dry years when stored moisture matters. All other areas tend to be reliably productive zones.

When Basso et al. (2019) scaled this analysis to 8 million corn fields across 30 million hectares in ten Midwest states, those within-field yield variability patterns held. Roughly 26% of that total area was consistently low yielding, and another 28% were low-performing with inconsistent yields (i.e., sufficient yield in favorable conditions, low yield in poor conditions). Nitrogen use efficiency in stable low-yield areas averaged only 48 to 57%, compared to 76% in high-yield areas. That gap has a dollar value. The same study estimated that over-fertilizing chronically underperforming ground costs Midwest farmers roughly $485 million per year in unused inputs lost to the environment. The authors noted that these stable low-yield areas "may be better planted to conservation strips."

This research clearly indicates that factors such as soil type and topography are what determine which acres are marginal for crop production and that continuing to cultivate these areas represents a substantial financial liability on individual farmers and, in turn, the agricultural supply chain. Targeting such areas for perennial plantings can reduce this liability while also achieving other ecological outcomes, and a layered mapping workflow can help to identify these areas with readily available analytical tools and data. The foundation is multi-year yield mapping. Yield monitor data collected from GPS-equipped combine harvesters across three or more seasons can be classified into stable high-yield, stable low-yield, and unstable zones at 30-meter subfield resolution using satellite-derived vegetation indices (e.g., Normalized Differential Vegetation Index) as a proxy where combine data are unavailable

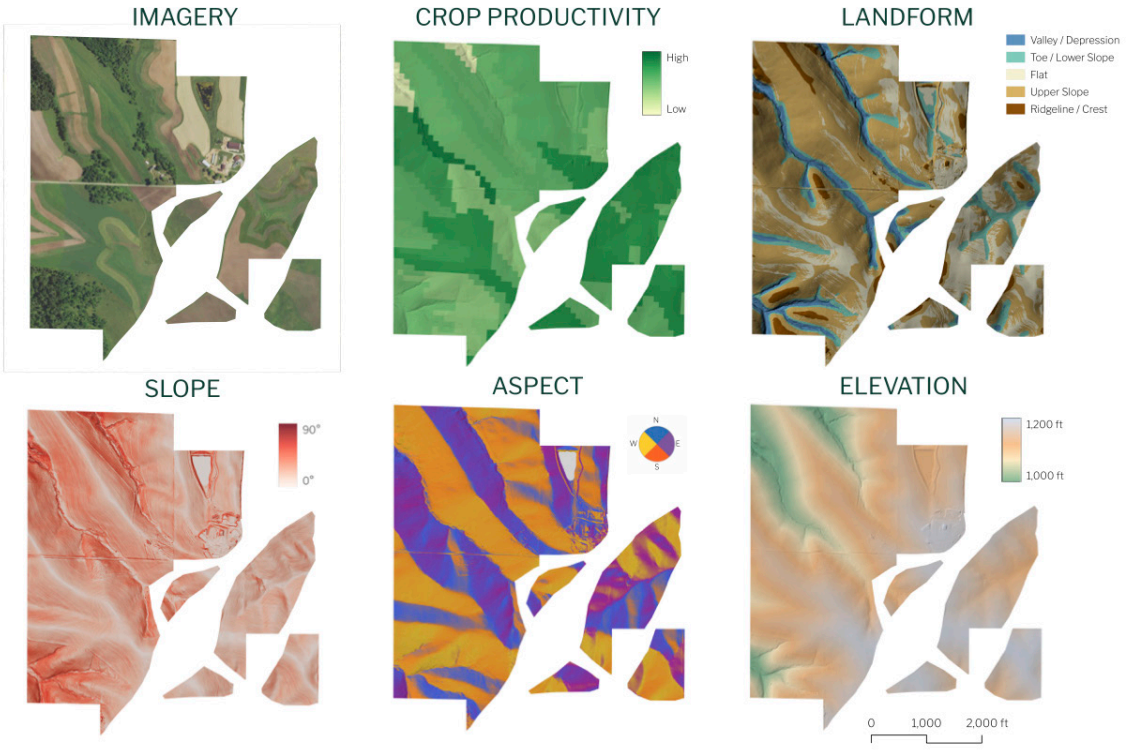

Figure 4. Geospatial layers used to identify sub-profitable acres on a parcel in the Driftless Region, Wisconsin. Clockwise from top left: aerial imagery (USDA National Agricultural Imagery Program), crop productivity index, landform classification (derived from Topographic Position Index, 50–55 m annulus), elevation, aspect, and slope (USGS 3DEP 1-meter DEM).

(Basso et al., 2019). Stable low-yield zones are structurally constrained—they occur in areas of shallow soils, compacted subsurface layers, and steep slopes. Unstable zones cluster in topographic depressions with high water flow accumulation, where performance swings with precipitation (Fowler et al., 2024; Maestrini and Basso, 2018).

These yield stability maps are then overlaid with publicly available spatial data to build a composite picture of marginality. Topographic wetness index and slope, derived from digital elevation models, identify the positions in the landscape where water accumulates or drains rapidly. USDA soil survey data add erodibility ratings, drainage class, and depth to restrictive layers. The USDA Cropland Data Layer time series flags fields with a history of frequent transitions between crop and non-crop cover—a land-use-change signature that independently identifies economically marginal land at national scale (Jiang et al., 2021; Khanna et al., 2021). Marginal acres are the areas where these layers converge. Those acres are the starting point for Wilding siting decisions.

THE FINANCIAL CASE

The financial logic for converting marginal acres is based on the fact that farmers spend money per acre but earn money per bushel. The average cost to produce an acre of corn in the US in 2025 is $897, including seed, fertilizer, fuel, equipment, and land opportunity cost (Swanson, 2025). That cost is incurred regardless of whether the acre yields 40 bushels or 200. At the three-year average price for conventional feed corn from 2020–2024 of $5.50 per bushel (USDA NASS, 2026), breaking even requires roughly 163 bushels per acre. In more local contexts, those numbers can quickly change. For example, in Wisconsin in 2025, conventional corn averaged $4.22 per bushel (USDA Risk Management Agency, 2026), raising the breakeven yield to approximately 213 bushels per acre, which is considerably higher than the state average of ~176 bushels per acre (USDA NASS, 2024). Stable low-yield subfield zones, by definition, fall well short of breakeven thresholds every year, while unstable subfield zones are only profitable in favorable years. In other words, some parts of a field consistently lose money, and other parts only pay off in good years.

These examples play out in the real world at the landscape as well. Brandes et al. (2016) analyzed subfield profitability across Iowa corn and soybean operations and found that as crop prices declined in 2015, up to 27% of Iowa acreage became outright subprofitable, losing money on a per-acre basis even before accounting for land costs. That number shifts with markets: in high-price years, more acres are carried along. But in average or low-price years, a meaningful share of every farm is running at a loss. Farmers may keep these acres in production anyway, as a hedge against the chance of a good year.

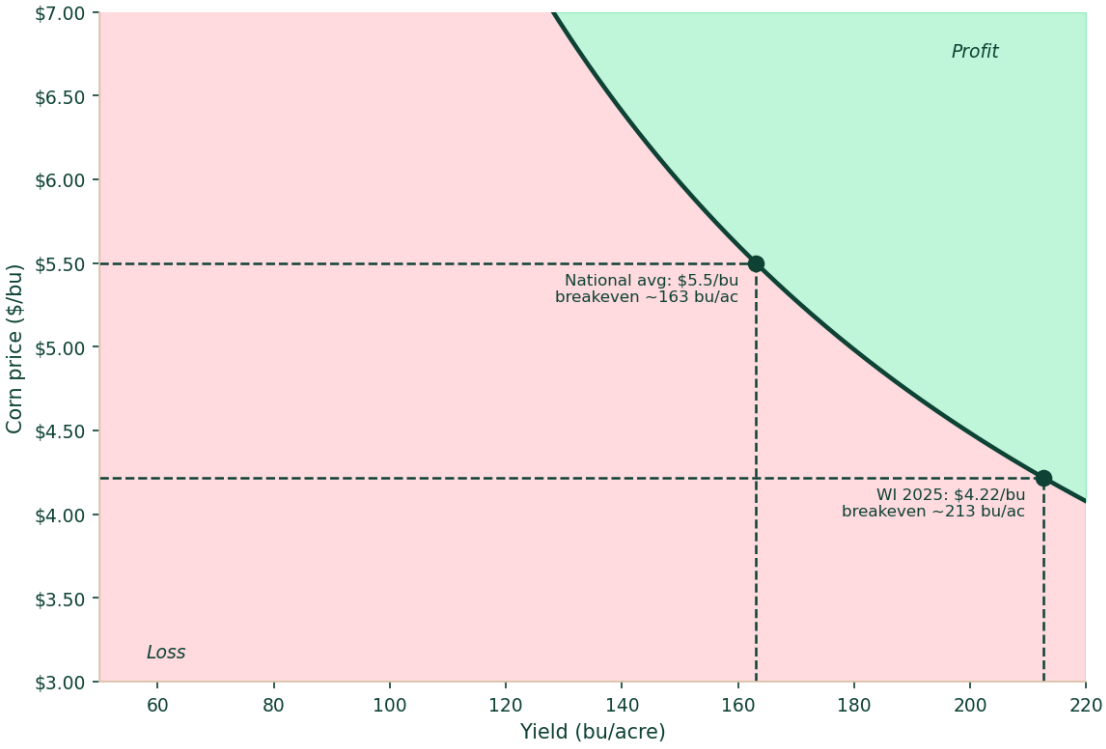

Figure 5. The curve marks the minimum corn price required to recover a production cost of $897/acre, the national average at a given yield. Any acre above and to the right of the curve turns a profit. At the national three-year average price ($5.50/bu), breaking even requires roughly 163 bushels per acre. At Wisconsin's average 2025 price ($4.22/bu), that threshold rises to ~213 bu/ac, a yield few marginal subfield zones reliably achieve.

The financial case for conversion to perennial cover depends on identifying alternative revenue streams from perennial plantings that can reliably outperform that hedge—not just in good years, but consistently enough to make it a rational choice. The most accessible entry point is existing USDA conservation programs such as EQIP, CSP, and CRP, which can cover installation and maintenance costs on a per-farm basis. But these programs have payment limits, are often oversubscribed, and may place limitations on agricultural land uses that reduce the opportunity to generate commercial revenue while maintaining perennial cover. Opportunity cost models that integrate yield stability data, marginality maps, local commodity prices, and land values can help identify the acres where the conversion argument is already strong and quantify how much additional revenue is needed to close the gap on the rest. Research confirms that when opportunity costs fall, farmer willingness rises: Luther et al. (2022) found that farmers are significantly more willing to convert lower-productivity areas to prairie plantings precisely because they are not being asked to sacrifice reliable income. The adoption barrier is not primarily attitudinal. It is economic, and addressing it requires making the per-acre financial case clearly and farm by farm.

Marginal acres are the right place to start, but they are likely not sufficient on their own to meet our full ambition for the Wild Grid. Jiang et al. (2021) estimated that approximately 25 million acres of cropland in the US could be confidently characterized as marginal. The 65 million acre Wild Grid target is well above that figure. Reaching it will require making inroads beyond the most obviously subprofitable ground while avoiding the displacement of food production into other areas, resulting in no net conservation gain. Marginal acres establish proof of concept and deliver the clearest early returns. The broader goal requires the financial and institutional logic to extend further into the working landscape.

COMMUNITY SCALE

A well-sited prairie planting on one farm intercepts runoff and supports local biodiversity. Those benefits are real but bounded by the parcel. Coordinating plantings across neighboring farms has the potential to produce outcomes that differ from single-parcel conservation in both magnitude and kind. Habitat patches that individually cannot support viable wildlife populations may do so when connected, and water quality interventions sited at critical points in a shared watershed can reduce nutrient loading at a scale no single farm can achieve alone. The community scale is where field-level ecological benefits begin to accumulate into landscape-level function.

That transition does not happen automatically. It requires two things that parcel-level conservation programs are not designed to provide: a shared spatial logic for where plantings should go to maximize their combined effect, and a financial structure that makes coordinated stewardship worth it for each participating landowner. The subsections below address each in turn.

IDENTIFYING PRIORITY AREAS AT THE COMMUNITY SCALE

Agricultural impact on water quality is a nonpoint source pollution problem. Pollutants enter waterways in small amounts from several areas across the landscape, making it very difficult to address the pollution at the source. Instead, the focus is on intercepting that pollution at downstream points where water and, thus, pollutants accumulate. Such areas are often termed critical source areas. They are the portions of a watershed where topography, soil type, and land use combine to generate disproportionate shares of runoff and nutrient export. They can be identified using mapping approaches that combine soil survey data, topographic wetness index, and flow accumulation modeling to map where surface runoff and shallow groundwater flows are most likely to reach streams (Qiu, 2009; Tomer et al., 2009). These maps allow practitioners to prioritize planting placement by both hydrological sensitivity and pollutant source intensity, and to compare placement scenarios for cost-effectiveness (Tomer et al., 2013). Spatially explicit water quality models can then simulate how converting those areas from cropland to perennial cover affects nutrient and sediment loads at the watershed outlet.

Research illustrates the potential of optimal placement at the field scale. In small experimental watersheds in central Iowa (0.5–3.2 ha), prairie plantings on 10% of the contributing area, sited at footslope and contour positions, reduced annual nitrate-N loss by 67% and total phosphorus by 90% (Helmers et al., 2012; Zhou et al., 2014). Whether comparable reductions are achievable when these placement principles are applied across larger watersheds, where subsurface flow, stream-channel processes, and concentrated flow paths come into play, is one of the open questions Wilding is designed to test.

Spatial analysis is also useful for targeting plantings to improve habitat connectivity for biodiversity outcomes. The underlying framework draws on metapopulation theory. Larger patches of

semi-natural habitat anchor local populations in the landscape, while the surrounding matrix determines how easily individuals can move between patches. That movement affects genetic exchange, recolonization after local extinction, and long-term population persistence (Fahrig and Merriam, 1994; Niebuhr et al., 2015). The planning task is to identify where existing habitat patches are, how well connected they are, and where new plantings would do the most to reduce isolation.

Identifying anchor habitats—the larger patches of semi-natural habitat that support source populations and anchor metapopulation dynamics in the landscape—is the starting point. Land use and land cover data, including the National Land Cover Database (USGS, 2020) and the Protected Areas Database (USGS, 2022), can be used to map existing semi-natural habitat, conservation easements, field edges, and remnant patches that currently support target species or guilds (i.e., a group of species that use the same resources or occupy similar ecological roles in a community . The specific habitat requirements of the target species or guild determine which land cover types qualify. Grassland obligates, woodland-dependent species, and generalist species all respond differently to the same landscape.

The surrounding land use and land cover matrix is then characterized using resistance surfaces. Resistance surfaces are maps that represent the relative cost of movement across different landscape elements for a given species, constructed by weighting land cover type, road density, waterways, and topography according to the movement ecology of the target species (Dutta et al., 2022; Wade et al., 2015). Matrix composition matters considerably. Even landscapes with adequate total habitat area can present effective isolation to species whose movement is constrained by the intervening land cover (Ricketts, 2001). A white-tailed deer moves readily across crop fields and roads. A grassland obligate like Henslow's Sparrow requires more continuously connected interior grassland to facilitate movement between patches.

Resistance surfaces can be combined with patch maps using network analysis tools such as Circuitscape (McRae et al., 2008) to identify existing movement corridors, pinpoint gaps where connectivity is lowest, and model where new plantings would most effectively reduce landscape resistance (Dutta et al., 2022). In fragmented grassland landscapes of the north-central U.S., network analysis using this approach identified a small number of keystone patches whose loss would have disproportionately large effects on overall connectivity, as well as stepping stone corridors linking major habitat concentrations (Wimberly et al., 2018).

Overlaying the water quality and connectivity analyses produces a composite siting framework. Acres that rank highly on both, falling in critical source areas and key connectivity gaps simultaneously, are the highest-priority candidates for Wilding and the Wild Grid, delivering water quality and biodiversity benefits from the same planting.

COORDINATION NEEDS FINANCIAL STRUCTURE

Research on conservation practice adoption in the United States has consistently found that economic factors and perceived risk, not attitudes or environmental awareness, are the primary determinants of farmer behavior (Prokopy et al., 2019; Ranjan et al., 2019). The decision to participate is not just about whether a farmer values wildlife or water quality, it is about whether joining a coordinated effort with neighbors is worth it financially and operationally. Siting plantings on marginal acres gets us partway there. Creating financial structures that sustain coordinated stewardship over time is the remaining challenge.
Habitat restoration generates benefits with real financial value for identifiable parties beyond the farmer. For example, municipalities drawing drinking water from affected watersheds may have lower water treatment costs, and food companies with supply chain exposure may have reduced climate-related or biodiversity-related risk exposure. Mechanisms exist to channel payments from these parties to the farmers and conservation practitioners who generate those benefits. Water quality trading programs allow point-source dischargers to purchase nutrient reduction credits from agricultural producers, and voluntary biodiversity and carbon markets create tradeable credits for verified ecological outcomes.

To be cost-effective, these mechanisms require a threshold of accumulated impact before they become financially viable. A single farm generating a modest water quality improvement is rarely enough to interest a municipal buyer or justify verification costs. But a connected corridor generating measurable, landscape-scale outcomes across a watershed is more likely to meet that threshold.

The structural barrier most of these mechanisms share is a timing mismatch. Restoration work requires upfront capital, while the ecological benefits that justify beneficiary payments take

Figure 6. The Forest Resilience Bond structure. Upfront private and philanthropic capital funds restoration work; downstream beneficiaries repay investors over time as ecological benefits accrue. The structure closes the timing gap between when restoration capital is required and when benefits can be verified and paid for. Adapted from Blue Forest Conservation.

years to materialize and even longer to verify. The Forest Resilience Bond (FRB), developed by Blue Forest Conservation, offers one proven approach to closing that gap. By combining concessional philanthropic capital with commercial investment and pre-committed beneficiary repayment agreements, the FRB provides practitioners with working capital upfront while repayment flows from downstream beneficiaries over time as the work is completed. The first deployment, which raised $4 million to treat 15,000 acres in California's North Yuba River Watershed, was completed in 2023 with full repayment to investors. A second raised $25 million in the same watershed, with follow-on instruments now being implemented across multiple western forests (Blue Forest Conservation, 2020).

Beyond ecosystem service markets, coordinated plantings in corridors present new commercial opportunities for farmers. Cooperative grazing is the most immediate economic entry point. When prairie strips and riparian corridors connect across neighboring farms, they accumulate enough continuous acreage to support a rotational grazing circuit—a small herd moving through connected ground from late spring through fall, generating lease revenue for each participating landowner while keeping vegetation in an actively managed state.

Grazing corridors may also function as a gateway to the adoption of additional beneficial practices in cropland acres. When perennial cover on marginal acres begins generating reliable lease revenue, it could change the economics for what occurs in the interior of the farm. Cover crops that might otherwise be planted solely for soil health gain additional value as supplemental forage for the grazing network. Synthetic input costs may decline as managed grazing closes nutrient loops through manure deposition. The corridor, in this model, is a potential reorganizing force for the production system as a whole, creating economic conditions under which in-field diversification becomes financially rational rather than purely aspirational.

Beyond individual revenue mechanisms, community-scale Wilding may require an institutional hub capable of aggregating and distributing value across participating farms. Regional grain processing infrastructure represents one candidate for that role. A mill that already sits at the center of a farmer network is positioned to extend that function to ecological value as well, bundling grazing leases, ecosystem service payments, and conservation program revenues across a network of participating landowners. We call this a millshed model, analogous to a watershed but organized around a processing hub rather than a drainage network. Wherever regional processing infrastructure exists within agricultural landscapes, it represents a potential organizational anchor for community-scale Wild Grid implementation.

SOCIAL BARRIERS TO COORDINATION

Financial barriers are the primary leverage point for farmer adoption, but not the only ones. Research on conservation practice adoption has documented a consistent set of non-economic barriers that operate alongside and independently of opportunity cost (Prokopy et al., 2019; Ranjan et al., 2019). Social norms around productive land use—the expectation that a well-managed field is planted fence row to fence row—create resistance to taking acres out of row crops re-

gardless of their economic performance. Many farmers identify themselves as "producers" first, and the adoption of conservation practices can conflict with that identity (Reimer et al., 2014). But peer influence can also cut both ways. In communities where early adopters are visible and trusted, adoption tends to accelerate. Where they are absent, the social proof needed to reduce perceived risk is missing.

Land tenure adds an additional structural constraint. A significant share of Midwest farmland is operated by tenants rather than owners. Tenants typically lack both the authority to make long-term conservation commitments and the financial incentive to invest in land they do not own. Non-operating landowners are often absentee and difficult to engage, and the power dynamics of the landlord-renter relationship complicate conservation decision-making in ways that neither party fully controls (Barnett et al., 2020). Addressing these barriers requires sustained investment in relationship-building, peer-to-peer learning, and landowner engagement. This is work that financial instruments can support but not substitute for.

Watershed-scale conservation initiatives offer clear examples of how to work through these barriers. Leach and Pelkey (2001), in a review of watershed partnerships across the US, identified trust, skilled facilitation, and sustained funding for coordination as the most consistent predictors of durable outcomes. Partnerships that relied on voluntary alignment alone rarely persisted beyond the tenure of their founding participants. Those that embedded facilitation capacity and dedicated funding into their structure held together across leadership transitions and shifting political conditions.

The Iowa Nutrient Reduction Strategy provides a recent and well-documented case. Upadhaya and Arbuckle (2021) surveyed 6,006 Iowa farmers and found that engagement in watershed management was predicted by access to public and private information sources, awareness of nutrient loss reduction strategies, farm contiguity to water bodies, and availability of cost-share and technical assistance. Reliance on press outlets primarily focused on agriculture as a primary information source, greater farmer age, and higher farm sales predicted lower engagement.

Community scale conservation succeeds when three elements are present: shared outcomes within a defined area, facilitation capacity that outlasts individual leaders, and accessible technical and financial support—including sustained revenue streams that outlast program contracts—that lowers the cost of participation over time. Without them, the non-economic barriers described above reassert themselves.

REGIONAL AND CONTINENTAL SCALE

At regional and continental scales, the challenges of building the Wild Grid resemble those at the community scale: coordinating dispersed efforts and sustaining them through shared financial structure and governance. What changes is the set of landscape features that come into view. River systems trace drainage networks across entire regions. Utility rights-of-way cross county and state lines. Public conservation lands anchor large blocks of habitat that community-scale projects can connect to but rarely encompass. The planning logic shifts from identifying where to create connections toward activating and enhancing connective features that already exist across the landscape.

At the regional scale, restoration and reconstruction efforts also begin to encompass a wider set of stakeholders with different positions and objectives in the agricultural landscape. The institutional challenge shifts accordingly from coordinating neighboring landowners around a shared watershed to aligning this wider set of stakeholders around a shared spatial framework that extends across jurisdictions and objectives. Likewise, the most instructive examples of successful efforts at the continental scale operate at regional and continental scales simultaneously, nesting regional efforts within a broader continental scale goal.

WORKING WITH WHAT EXISTS

River systems are among the most ecologically significant regional connectors. Riparian corridors possess unusually high species diversity relative to surrounding uplands, driven by variable flood regimes, geomorphic processes, and cross-ecosystem subsidies of water, nutrients, and organic material that support plant and animal communities (Naiman et al., 1993; Xiang et al., 2016). This ecological distinctiveness also makes them effective conduits for the movement of animals and dispersal of plants. In multi-species connectivity modeling, stream-associated riparian vegetation consistently produces pathways with higher ecological potential and greater feasibility for restoration than other landscape features (Liu et al., 2018). Major river corridors and their tributaries trace the drainage network of a region, providing a natural framework by which community-scale projects can be linked

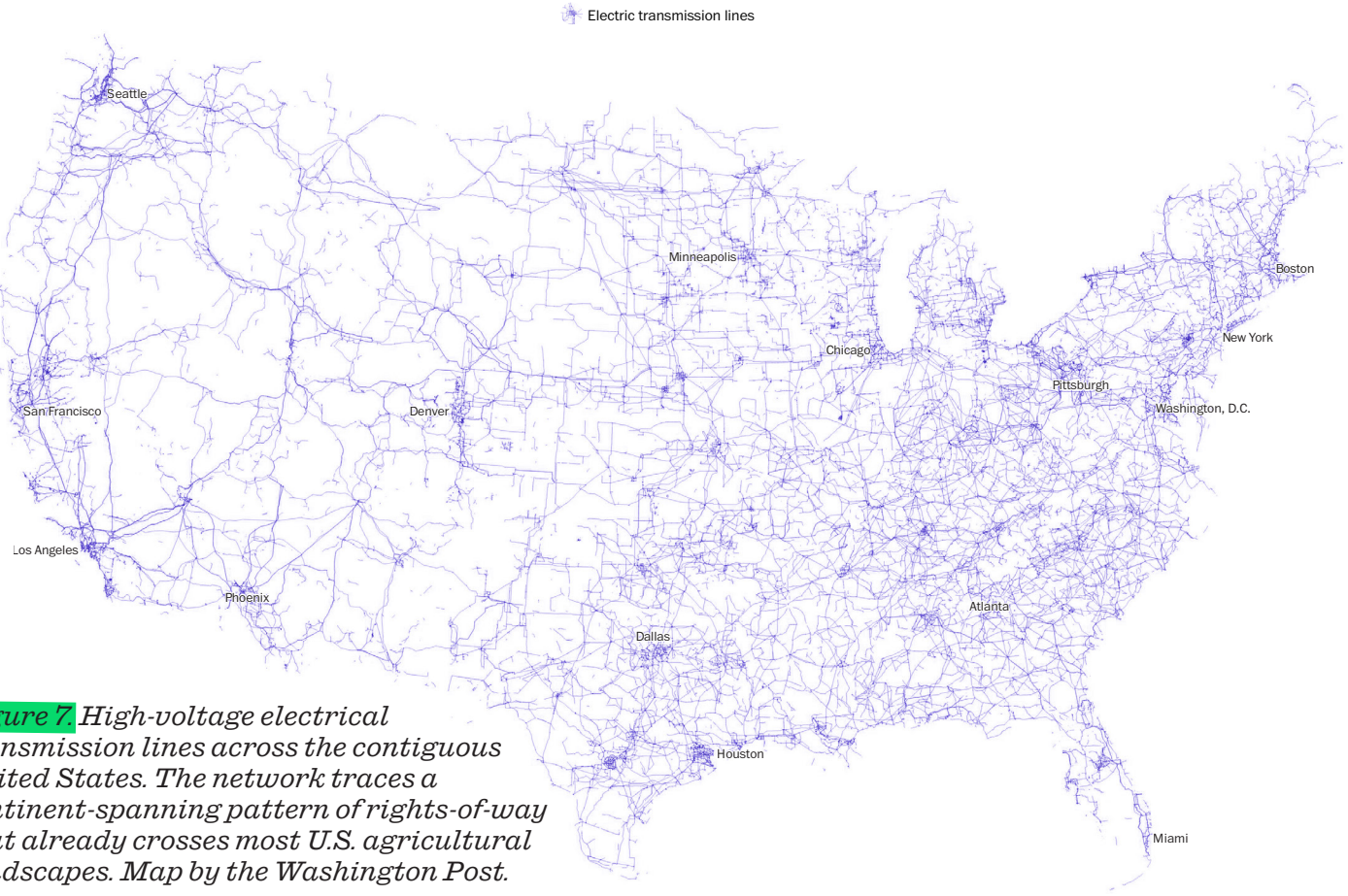

Figure 7. High-voltage electrical transmission lines across the contiguous United States. The network traces a continent-spanning pattern of rights-of-way that already crosses most U.S. agricultural landscapes. Map by the Washington Post.

into a continuous corridor.

Utility rights-of-way offer a complementary opportunity. Powerline easements in the United States cover a substantial area, and if managed for native low-growing vegetation rather than periodic mowing, they represent approximately five million acres of potential pollinator habitat (Russell et al., 2005). Where that management shift has occurred, the ecological response is measurable. A study along an 87-mile transmission corridor from Connecticut to New Hampshire found ten times the abundance and twice the species richness of wild bees in corridor plots compared to adjacent forest, with nearly half of all bee species known from New England present in the corridor (Wagner et al., 2019). Managed rights-of-way in New York, Massachusetts, and Maine had high bird species counts, particularly from guilds that utilize early successional habitat and are in decline elsewhere in the landscape (Confer and Pascoe, 2003). In fragmented landscapes, powerline clearings develop over time into novel open habitats with higher plant richness and greater representation of semi-natural grassland species than adjacent forests, with effects strongest at productive sites managed frequently (Eldegard et al., 2017).

Regional connectivity requires aligning community scale projects with features that already cross the landscape and enhancing their management to maximize plant diversity, and in turn, animal diversity. The physical pieces are present. What has been missing is the institutional architecture to align them around a shared goal.

REGIONAL GOVERNANCE

The governance challenge at regional scale is distinct from the community level in two important ways. First, the number and diversity of stakeholders expands dramatically. State agencies, utilities, conservation districts, tribes, municipalities, and food companies may all have a legitimate stake in regional corridor function and how it intersects with food systems, water management, and land use planning. Second, the time horizon lengthens. The conditions that make coordination possible at community scale—shared geography, interpersonal trust, a common problem—become harder to maintain as the network grows. At a regional scale, corridors must persist through land tenure change across dozens of properties, through shifts in state and federal policy, and through political administrations that may be indifferent or hostile to conservation priorities. Regional coordination requires more than

economic coherence. It requires institutions that can hold continuity through the inevitable turbulence that larger scales and longer time horizons introduce.

Precedents for this kind of governance exist, though none maps cleanly onto the Wild Grid's working-land context.

The North American Waterfowl Management Plan (NAWMP), signed in 1986 by the United States and Canada with Mexico joining in 1994, is the closest analog in ecological scope and governance structure (Anderson and Padding, 2015; Patterson, 1995). By 2024, more than 6,200 partners had contributed $3.75 billion in matching funds against $1.83 billion in federal NAWCA grants, conserving more than 30 million acres of wetlands and associated habitats across Canada, the U.S., and Mexico (NAWMP, 2024). What made the NAWMP effective was a quantified ecological target, a dedicated funding base built on NAWCA grants, federal Duck Stamp revenues, hunting license fees, and partner matching, and a governance structure that nested local implementation within regional joint ventures within a continental framework (Williams et al., 1999). Rather than centralizing management, NAWMP built a shared map, shared goals, and a mechanism that aligned federal appropriations with private and state funding to guide and enable these joint ventures. Critically for the Wilding context, NAWMP's reach into private working lands depended on programs explicitly designed around landowner needs and co-benefits that extended beyond wildlife habitat. Experience in NAWMP suggests that financial incentives are necessary but insufficient, and that programs generating returns for landowners beyond conservation payments tend to achieve greater durability (Brasher et al., 2019).

The Yellowstone to Yukon Conservation Initiative (Y2Y) offers a second model. Spanning five U.S. states, two Canadian provinces, and more than 75 Indigenous territories, Y2Y has supported grizzly bear range expansion and population recovery (Hilty et al., 2024). A spatiotemporal counterfactual analysis found that the rate of protected area growth in the Y2Y region increased 90% following the initiative's founding in 1993, while growth in adjacent regions remained constant or declined, and by 2018, protected areas in the Y2Y region had grown by 26.5 million acres (Hebblewhite et al., 2022). At the field level, connectivity gains have been advanced through small, strategically placed private land acquisitions in key pinch points (Locke and Francis, 2012). Y2Y operates without regulatory authority or land ownership and is sustained through a network of more than 700 partner organizations held together by a small backbone organization that maintains a shared spatial framework and scientific foundation (Chester, 2015).

These examples demonstrate that coordination at regional to continental scale is achievable. Each also clarifies the distance between what has been demonstrated and what the Wild Grid would require. NAWMP operates primarily on wetlands with a well-defined hunting constituency that provides dedicated funding. Y2Y operates in a landscape anchored by large public land units with charismatic focal species that generate broad public support. The model for regional habitat corridors in agricultural lands does not yet fully exist. What these precedents suggest is what its components would need to be—sustained and diversified funding, skilled facilitation, a shared spatial framework, and a backbone organization capable of maintaining continuity, and financial incentives that generate returns for landowners beyond conservation payments alone.

Figure 8. A prairie planting adjacent to a crop field in Iowa (photo credit: Omar de Kok-Mercado, Mad Agriculture, Courtesy of Iowa State University).

BOX 1: WILDING PILOT PROJECT IN THE WISCONSIN DRIFTLESS

Mad Agriculture's first implementation of the Wilding framework is a three-year, 1,000-acre pilot in the Driftless Area of southwest Wisconsin, anchored in and around the Lowery Creek Watershed and the Military Ridge Prairie Heritage Area. Whole Foods Market and twenty additional corporate partners have committed $1.03 million to fund prairie planting across the pilot footprint.

The pilot is deliberately sited within one of the most ecologically significant grassland landscapes in the Midwest. The Military Ridge Prairie Heritage Area spans more than 95,000 acres across Dane and Iowa counties and has been identified by The Nature Conservancy and the Wisconsin DNR as the state's highest priority for landscape-scale grassland protection. The region supports 14 rare and declining grassland bird species and contains the highest concentration of native prairie remnants in the Midwest, providing an ecologically meaningful context for testing whether field-scale prairie plantings can begin to function as corridor infrastructure at community scale.

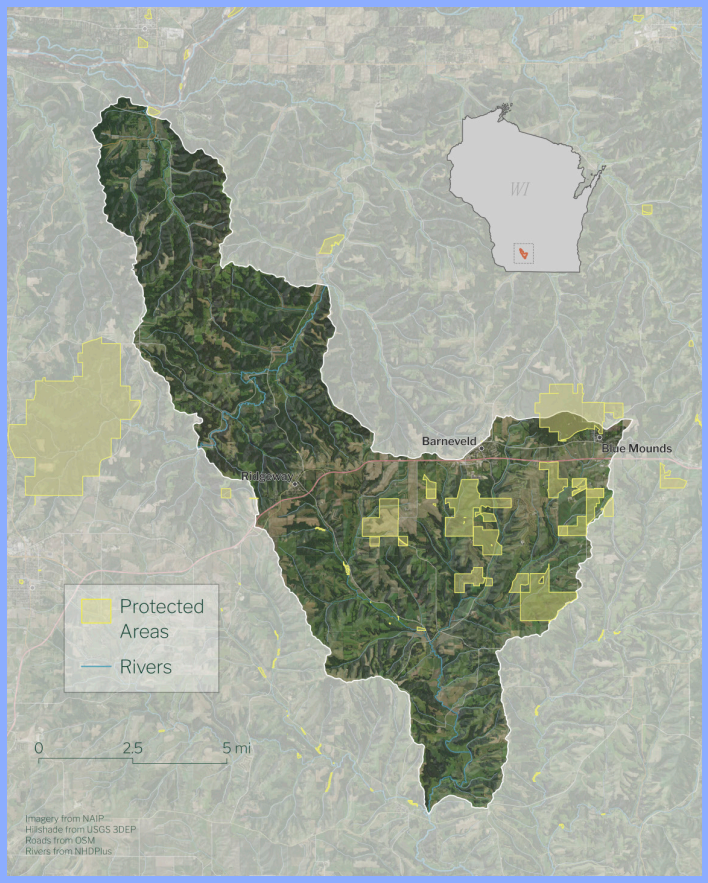

Figure 9. Wilding Pilot primary area of interest, southwestern Wisconsin: 58,553 acres across four contiguous HUC-12 sub-watersheds (Lowery Creek, Meudt Creek–Mill Creek, Ridgeway Branch, and Williams–Barneveld Creek–E. Branch Pecatonica River) extending from the Wisconsin River to Military Ridge. Yellow polygons indicate existing protected areas, including TNC's Barneveld Prairie and The Prairie Enthusiasts' Mounds View within the Military Ridge Prairie Heritage Area. Imagery from NAIP; hillshade from USGS 3DEP; rivers from NHDPlus.

The pilot operationalizes the framework laid out in this paper at the field and community scales, with an eye toward learning what regional-scale deployment will eventually require. At the field scale, Mad Agriculture and implementation partners identify marginal acres using yield stability and marginality mapping workflows, then co-design plantings with landowners. The region's steep slopes, thin ridgelines, and flood-prone lowlands produce persistent subfield yield variation that makes the financial case for conversion to perennial cover tractable on a meaningful share of the pilot footprint.

At the community scale, we will focus on clustering parcels in the Lowery Creek area and adjacent sub-watersheds. These sit in between the Wisconsin River to the north and two key prairie habitat preserves to the south and east—Barneveld Prairie and Mounds View grasslands, and represent key source water areas for downstream communities. This combination makes them an excellent potential testing ground for financial models similar to Blue Forest's Forest Resilience Bond that integrate ecosystem service payment schemes. Likewise, several efforts are alive in the region to stand up alternative commercial markets and production models for farmers, such as Grasslands 2.0 and Savanna Institute.

The regional scale is out of scope for a three-year, 1,000-acre pilot, but the work is deliberately situated to learn what regional deployment will require. The pilot sits between two of the continent's major river corridors—the Wisconsin River to the north and the Mississippi River to the west—positioning it within migratory flyways and hydrologic systems that extend well beyond the pilot footprint. It also sits within an active network of conservation and agricultural institutions. The Nature Conservancy has designated the Driftless a priority landscape for resilience and climate action. The Driftless Area Land Conservancy, The Prairie Enthusiasts, and the Southern Driftless Grasslands Partnership maintain active stewardship networks. Wisconsin's Water Quality Trading Clearinghouse provides a state-facilitated marketplace for testing nutrient credit generation from prairie plantings. Each represents a piece of the coordination, governance, and finance infrastructure that regional-scale Wilding will require. The pilot is designed to surface how these pieces fit together, which gaps remain, and what tools, partnerships, and institutional forms Mad Agriculture will need to facilitate building or convene to scale beyond the pilot footprint.

Conclusion

Decades of research document the extent to which the conversion of perennial cover to cropland has diminished ecological function, and demonstrate that strategically sited native perennial cover can produce substantial improvements in water quality, soil function, and biodiversity. Whether those benefits compound when plantings are coordinated into functionally connected networks across farms and watersheds is less well understood. Research also indicates that, to maintain the ecosystem services agriculture relies on and meet broader biodiversity goals, at least 20% of working landscapes should be maintained in semi-natural habitat (Garibaldi et al., 2021; Mohamed et al., 2024). Reaching that threshold in U.S. agricultural landscapes requires not only ecological design but the financial structures and commercial opportunities that make coordinated stewardship economically viable for the farmers and landowners doing the work and institutions that can define and maintain a shared purpose.

Wilding is a framework for reconstructing habitat connectivity in working agricultural landscapes through the strategic integration of perennial cover designed to build a functionally connected, multi-use habitat network capable of delivering ecosystem services, supporting species movement, and maintaining biodiversity across watershed and regional scales. The Wild Grid represents that vision at continental scale: 65 million acres of perennial habitat threaded through U.S. agricultural lands over fifty years. Reaching it depends on making that choice financially rational for farmers and stakeholders across the agricultural supply chain.

At each spatial scale at which the Wild Grid operates there are two challenges: determining how and where to reconstruct habitat and building the financial tools and institutions to ensure that habitat remains durable long enough for ecological function to compound. This paper has presented the evidence base, financial logic, and prior examples that inform Mad Agriculture's approach.

As much as Wilding is a framework, it is also a hypothesis, and one that requires testing at every scale. Mad Agriculture will test that hypothesis through the work itself. Ecological monitoring and modeling will track water quality, soil, and biodiversity outcome at field and watershed scales. Spatial analysis will refine how marginal acres and critical source areas are identified, and test those identifications against what farmers and the land itself reveal over time. Financial modeling will quantify the savings generated by removing subprofitable acres from production and the revenue required to sustain perennial cover as a durable land use. The aim is not to prove the framework is right. It is to learn, honestly and in public, where it holds and where it breaks, and to refine it across each cycle of work.

Scaling the Wild Grid to continental proportions will require tools, technologies, and institutional forms that do not yet exist. The interstate highway system required not just political will but engineering innovations and the growth of interstate supply chains and commerce that reinforced the need for it. Ecological infrastructure at continental scale will demand the same: new financial instruments, new governance models, and likely new physical technologies for aggregating and processing value from distributed working landscapes. The evidence and frameworks presented in this paper represent the current frontier of what is demonstrable. What lies beyond that frontier is not yet fully imaginable, and that is precisely the point. The Wild Grid is as much an invitation to invention as it is a plan.

Works Cited

Anderson, M., Padding, P., 2015. The North American approach to waterfowl management: synergy of hunting and habitat conservation. Int. J. Environ. Stud. 72, 1–20. https://doi.org/10.1080/00207233.2015.1019296

Baer, S.G., Bach, E.M., Meyer, C.K., Du Preez, C.C., Six, J., 2015. Belowground Ecosystem Recovery During Grassland Restoration: South African Highveld Compared to US Tallgrass Prairie. Ecosystems 18, 390–403. https://doi.org/10.1007/s10021-014-9833-x

Barnes, J.C., Sketch, M., Gramza, A.R., Sorice, M.G., Iovanna, R., Dayer, A.A., 2020. Land use decisions after the Conservation Reserve Program: Re- enrollment, reversion, and persistence in the south ern Great Plains. Conserv. Sci. Pract. 2, e254. https://doi.org/10.1111/csp2.254

Barnett, M.J., Spangler, K., Petrzelka, P., Filipiak, J., 2020. Power dynamics of the non-operating land owner-renter relationship and conservation decision-making in the midwestern United States. J. Rural Stud. 78, 107–114. https://doi.org/10.1016/j.jrurstud.2020.06.026

Basso, B., Shuai, G., Zhang, J., Robertson, G.P., 2019. Yield stability analysis reveals sources of large-scale nitrogen loss from the US Midwest. Sci. Rep. 9, 5774. https://doi.org/10.1038/s41598-019-42271-1

Blue Forest Conservation, 2020. Case Study: The Forest Resilience Bond. https://www.blueforest.org/wp-content/uploads/2024/02/The-Forest-Resilience-Bond-Case-Study.pdf

Brandes, E., McNunn, G.S., Schulte, L.A., Bonner, I.J., Muth, D.J., Babcock, B.A., Sharma, B., Heaton, E.A., 2016. Subfield profitability analysis reveals an economic case for cropland diversification. Envi ron. Res. Lett. 11, 014009. https://doi.org/10.1088/1748-9326/11/1/014009

Brasher, M.G., Giocomo, J.J., Azure, D.A., Bartuszevige, A.M., Flaspohler, M.E., Harrigal, D.E., Olson, B.W., Pitre, J.M., Renner, R.W., Stephens, S.E., Vest, J.L., 2019. The history and importance of private lands for North American waterfowl conservation. Wildl. Soc. Bull. 43, 338–354. https://doi.org/10.1002/wsb.1002

Carver, S., Convery, I., Hawkins, S., Beyers, R., Eagle, A., Kun, Z., Van Maanen, E., Cao, Y., Fisher, M., Ed wards, S.R., Nelson, C., Gann, G.D., Shurter, S., Aguilar, K., Andrade, A., Ripple, W.J., Davis, J., Sin clair, A., Bekoff, M., Noss, R., Foreman, D., Pettersson, H., Root-Bernstein, M., Svenning, J.-C., Taylor, P., Wynne-Jones, S., Featherstone, A.W., Fløjgaard, C., Stanley-Price, M., Navarro, L.M., Aykroyd, T., Parfitt, A., Soulé, M., 2021. Guiding principles for rewilding. Conserv. Biol. 35, 1882–1893. https://doi.org/10.1111/cobi.13730

Chester, C.C., 2015. Yellowstone to Yukon: Transborder conservation across a vast international land scape. Environ. Sci. Policy. https://doi.org/10.1016/j.envsci.2014.08.009

Chiartas, J., Jackson, L., Long, R., Margenot, A., O'Geen, A., 2022. Hedgerows on Crop Field Edges In crease Soil Carbon to a Depth of 1 meter. Sustainability 14, 12901. https://doi.org/10.3390/su141912901

Claassen, R., Carriazo, F., Cooper, J.C., Hellerstein, D., Ueda, K., 2011. Grassland to Cropland Conver sion in the Northern Plains: The Role of Crop Insurance, Commodity, and Disaster Programs (No. 120). USDA ERS.

Collinge, S.K., 1996. Ecological consequences of habitat fragmentation: implications for landscape architecture and planning. Landsc. Urban Plan. 36, 59–77. https://doi.org/10.1016/S0169-2046(96)00341-6

Confer, J.L., Pascoe, S.M., 2003. Avian communities on utility rights-of-ways and other managed shrub

lands in the northeastern United States. For. Ecol. Manag., Early-Successional Forests and Shru bland Habitats in the North Eastern United States :Critical Habitats dependent on Disturbance 185, 193–205. https://doi.org/10.1016/S0378-1127(03)00255-X

Cornelisse, T., Inouye, D.W., Irwin, R.E., Jepsen, S., Mawdsley, J.R., Ormes, M., Daniels, J., Debinski, D.M., Griswold, T., Klymko, J., Orr, M.C., Richardson, L., Sears, N., Schweitzer, D., Young, B.E., 2025. El evated extinction risk in over one-fifth of native North American pollinators. Proc. Natl. Acad. Sci. 122, e2418742122. https://doi.org/10.1073/pnas.2418742122

Crews, T.E., Rumsey, B.E., 2017. What Agriculture Can Learn from Native Ecosystems in Building Soil Organic Matter: A Review. Sustainability 9, 578. https://doi.org/10.3390/su9040578

Dietz, C.L., Jackson, R.D., Ruark, M.D., Sanford, G.R., 2024. Soil carbon maintained by perennial grass lands over 30 years but lost in field crop systems in a temperate Mollisol. Commun. Earth Envi ron. 5, 1–8. https://doi.org/10.1038/s43247-024-01500-w

Donald, P.F., Evans, A.D., 2006. Habitat connectivity and matrix restoration: the wider implications of agri-environment schemes. J. Appl. Ecol. 43, 209–218. https://doi.org/10.1111/j.1365-2664.2006.01146.x

Drexler, S., Thiessen, E., Don, A., 2024. Carbon storage in old hedgerows: The importance of be low-ground biomass. GCB Bioenergy 16, e13112. https://doi.org/10.1111/gcbb.13112

Dutta, T., Sharma, S., Meyer, N.F.V., Larroque, J., Balkenhol, N., 2022. An overview of computational tools for preparing, constructing and using resistance surfaces in connectivity research. Landsc. Ecol. 37, 2195–2224. https://doi.org/10.1007/s10980-022-01469-x

Dutter, C., Damiano, L.A., Niemi, J., Miller, B.A., Schulte, L.A., Liebman, M., Helmers, M.J., Cruse, R.M., McDaniel, M.D., 2023. Contour prairie strips affect adjacent soil but have only slight effects on crops. Field Crops Res. 296, 108905. https://doi.org/10.1016/j.fcr.2023.108905

Dutter, C.R., Rieke, E.L., Pottebaum, S., McDaniel, M.D., 2025. Prairie strips improve many measures of soil health in nearly a decade. J. Soil Water Conserv. 80, 388–404. https://doi.org/10.1080/002245 61.2024.2435683

Eldegard, K., Eyitayo, D.L., Lie, M.H., Moe, S.R., 2017. Can powerline clearings be managed to promote insect-pollinated plants and species associated with semi-natural grasslands? Landsc. Urban Plan. 167, 419–428. https://doi.org/10.1016/j.landurbplan.2017.07.017

Fahrig, L., Merriam, G., 1994. Conservation of Fragmented Populations. Conserv. Biol. 8, 50–59. https:// doi.org/10.1046/j.1523-1739.1994.08010050.x

Fowler, A., Basso, B., Maureira, F., Millar, N., Ulbrich, R., Brinton, W.F., 2024. Spatial patterns of his torical crop yields reveal soil health attributes in US Midwest fields. Sci. Rep. 14, 465. https://doi. org/10.1038/s41598-024-51155-y

Garibaldi, L.A., Oddi, F.J., Miguez, F.E., Bartomeus, I., Orr, M.C., Jobbágy, E.G., Kremen, C., Schulte, L.A., Hughes, A.C., Bagnato, C., Abramson, G., Bridgewater, P., Carella, D.G., Díaz, S., Dicks, L.V., Ellis, E.C., Goldenberg, M., Huaylla, C.A., Kuperman, M., Locke, H., Mehrabi, Z., Santibañez, F., Zhu, C.-D., 2021. Working landscapes need at least 20% native habitat. Conserv. Lett. 14, e12773. https:// doi.org/10.1111/conl.12773

Gebhart, D.L., Johnson, H.B., Mayeux, H.S., Polley, H.W., 1994. The CRP increases soil organic carbon. J. Soil Water Conserv. 49, 488–492. https://doi.org/10.1080/00224561.1994.12456893

Giese, J.C., Schulte, L.A., Klaver, R.W., 2024. Bird community response to field-level integration of prairie strips. Agric. Ecosyst. Environ. 374, 109075. https://doi.org/10.1016/j.agee.2024.109075

Hebblewhite, M., Hilty, J.A., Williams, S., Locke, H., Chester, C., Johns, D., Kehm, G., Francis, W.L., 2022. Can a large-landscape conservation vision contribute to achieving biodiversity targets? Conserv.

Sci. Pract. 4, e588. https://doi.org/10.1111/csp2.588

Helmers, M.J., Zhou, X., Asbjornsen, H., Kolka, R., Tomer, M.D., Cruse, R.M., 2012. Sediment Removal by Prairie Filter Strips in Row- Cropped Ephemeral Watersheds. J. Environ. Qual. 41, 1531–1539. https://doi.org/10.2134/jeq2011.0473

Hilty, J.A., Chester, C.C., Wright, P.A., Zenkewich, K., 2024. Uniting hearts and lands: advancing con servation and restoration across the Yellowstone to Yukon region. Front. Conserv. Sci. 4. https:// doi.org/10.3389/fcosc.2023.1264460

Hulting, K.A., Brudvig, L.A., Damschen, E.I., Levey, D.J., Resasco, J., Tewksbury, J.J., Haddad, N.M., 2025. Habitat edges decrease plant reproductive output in fragmented landscapes. J. Ecol. 113, 531–541. https://doi.org/10.1111/1365-2745.14452

Isbell, F., Tilman, D., Reich, P.B., Clark, A.T., 2019. Deficits of biodiversity and productivity linger a cen tury after agricultural abandonment. Nat. Ecol. Evol. 3, 1533–1538. https://doi.org/10.1038/ s41559-019-1012-1

Jiang, C., Guan, K., Khanna, M., Chen, L., Peng, J., 2021. Assessing Marginal Land Availability Based on Land Use Change Information in the Contiguous United States. Environ. Sci. Technol. 55, 10794–10804. https://doi.org/10.1021/acs.est.1c02236

Kane, D., 2026. kanedan29/wild-grid-analysis. https://github.com/kanedan29/wild-grid-analysis

Kemmerling, L.R., McCarthy, A.C., Brown, C.S., Haddad, N.M., 2023. Butterfly biodiversity increases with prairie strips and conservation management in row crop agriculture. Insect Conserv. Divers. 16, 828–837. https://doi.org/10.1111/icad.12675

Khanna, M., Chen, L., Basso, B., Cai, X., Field, J.L., Guan, K., Jiang, C., Lark, T.J., Richard, T.L., Spawn-Lee, S.A., Yang, P., Zipp, K.Y., 2021. Redefining marginal land for bioenergy crop production. GCB Bioenergy 13, 1590–1609. https://doi.org/10.1111/gcbb.12877

Lark, T.J., Spawn, S.A., Bougie, M., Gibbs, H.K., 2020. Cropland expansion in the United States produces marginal yields at high costs to wildlife. Nat. Commun. 11, 4295. https://doi.org/10.1038/s41467-020-18045-z

Leach, W.D., Pelkey, N.W., 2001. Making Watershed Partnerships Work: A Review of the Empiri cal Literature. J. Water Resour. Plan. Manag. 127, 378–385. https://doi.org/10.1061/(ASCE)0733-9496(2001)127:6(378)

Li, C., Fultz, L.M., Moore-Kucera, J., Acosta-Martínez, V., Horita, J., Strauss, R., Zak, J., Calderón, F., Wein dorf, D., 2017. Soil carbon sequestration potential in semi-arid grasslands in the Conserva tion Reserve Program. Geoderma 294, 80–90. https://doi.org/10.1016/j.geoderma.2017.01.032

Li, X., Tian, H., Lu, C., Pan, S., 2023. Four-century history of land transformation by humans in the Unit ed States (1630–2020): annual and 1 km grid data for the HIStory of LAND changes (HIS LAND-US). Earth Syst. Sci. Data 15, 1005–1035. https://doi.org/10.5194/essd-15-1005-2023

Liu, C., Newell, G., White, M., Bennett, A.F., 2018. Identifying wildlife corridors for the restoration of regional habitat connectivity: A multispecies approach and comparison of resistance surfaces. PLOS ONE 13, e0206071. https://doi.org/10.1371/journal.pone.0206071

Locke, H., Francis, W.L., 2012. Strategic Acquisition and Management of Small Parcels of Private Lands in Key Areas to Address Habitat Fragmentation at the Scale of the Yellowstone to Yukon Region. Ecol. Restor. https://doi.org/10.3368/er.30.4.293

Luther, Z.R., Swinton, S.M., Deynze, B.V., 2022. Potential Supply of Midwest Cropland for Conversion to In-Field Prairie Strips. Land Econ. 98, 274–291. https://doi.org/10.3368/le.98.2.082020-0129R1

Maestrini, B., Basso, B., 2018. Drivers of within-field spatial and temporal variability of crop yield across

the US Midwest. Sci. Rep. 8, 14833. https://doi.org/10.1038/s41598-018-32779-3

McRae, B.H., Dickson, B.G., Keitt, T.H., Shah, V.B., 2008. USING CIRCUIT THEORY TO MODEL CON NECTIVITY IN ECOLOGY, EVOLUTION, AND CONSERVATION. Ecology 89, 2712–2724. https://doi.org/10.1890/07-1861.1

Meiyappan, P., Jain, A.K., 2012. Three distinct global estimates of historical land-cover change and land-use conversions for over 200 years. Front. Earth Sci. 6, 122–139. https://doi.org/10.1007/s11707-012-0314-2

Mohamed, A., DeClerck, F., Verburg, P.H., Obura, D., Abrams, J.F., Zafra-Calvo, N., Rocha, J., Estrada-Car mona, N., Fremier, A., Jones, S.K., Meier, I.C., Stewart-Koster, B., 2024. Securing Nature's Contri butions to People requires at least 20%–25% (semi-)natural habitat in human-modified landscapes. One Earth 7, 59–71. https://doi.org/10.1016/j.oneear.2023.12.008

Morefield, P.E., LeDuc, S.D., Clark, C.M., Iovanna, R., 2016. Grasslands, wetlands, and agriculture: the fate of land expiring from the Conservation Reserve Program in the Midwestern United States. Environ. Res. Lett. 11, 094005. https://doi.org/10.1088/1748-9326/11/9/094005

Naiman, R.J., Decamps, H., Pollock, M., 1993. The Role of Riparian Corridors in Maintaining Regional Biodiversity. Ecol. Appl. 3, 209–212. https://doi.org/10.2307/1941822

NAWMP, 2024. 2024 North American Waterfowl Management Plan Update: Expanding the Partner ship. https://nawmp.org/sites/default/files/2025-01/nawmp-update-2024-en-web.pdf

Nelson, J., 2022. Monitoring Soil Movement at the Field-Scale in Row Crop Systems with Prairie Strips. Iowa State University, Ames, IA. https://dr.lib.iastate.edu/entities/publication/1e3e416e-6424-49e7-be95-098926bd1994

Niebuhr, B.B.S., Wosniack, M.E., Santos, M.C., Raposo, E.P., Viswanathan, G.M., Da Luz, M.G.E., Pie, M.R., 2015. Survival in patchy landscapes: the interplay between dispersal, habitat loss and fragmenta tion. Sci. Rep. 5, 11898. https://doi.org/10.1038/srep11898

Oliver, T.H., Brereton, T., Roy, D.B., 2013. Population resilience to an extreme drought is influenced by habitat area and fragmentation in the local landscape. Ecography 36, 579–586. https://doi.org/10.1111/j.1600-0587.2012.07665.x

Patterson, J.H., 1995. The North American Waterfowl Management Plan and Wetlands for the Ameri cas programmes: a summary. Ibis 137, S215–S218. https://doi.org/10.1111/j.1474-919X.1995.tb08447.x

Pett-Ridge, J., Kuebbing, S., Mayer, A., Hovorka, S., Pilorgé, H., Baker, S., Pang, S., Scown, C., Mayfield, K., Wong, A., Aines, R., Ammar, H., Aui, A., Ashton, M., Basso, B., Bradford, M., Bump, A., Busch, I., Calzado, E., Chirigotis, J., Clauser, N., Crotty, S., Dahl, N., Dai, T., Ducey, M., Dumortier, J., Ellebracht, N., Egui, R., Fowler, A., Georgiou, K., Giannopoulos, D., Goldstein, H., Harris, T., Hayes, D., Hellwinckel, C., Ho, A., Hong, M., Hunter-Sellars, E., Kirkendall, W., Langholtz, M., Layer, M., Lee, I., Lewis, R., Li, W., Liu, W., Lozano, J., Lunstrum, A., McNeil, W., Nico, P., O'Rourke, A., Paus tian, K., Peridas, G., Pisciotta, M., Price, L., Psarras, P., Robertson, G., Sagues, W., Sanchez, D., Schmidt, B., Slessarev, E., Sokol, N., Stanley, A., Swan, A., Toureene, C., Wright, M., Yao, Y., Zhang, B., Zhang, Y., 2023. Roads to Removal: Options for Carbon Dioxide Removal in the United States (No. LLNL--TR-852901, 2301853, 1080440). https://doi.org/10.2172/2301853

Prokopy, L.S., Floress, K., Arbuckle, J.G., Church, S.P., Eanes, F.R., Gao, Y., Gramig, B.M., Ranjan, P., Singh, A.S., 2019. Adoption of agricultural conservation practices in the United States: Evidence from 35 years of quantitative literature. J. Soil Water Conserv. 74, 520–534. https://doi.org/10.2489/jswc.74.5.520

Qiu, Z., 2009. Assessing Critical Source Areas in Watersheds for Conservation Buffer Planning and Ri parian Restoration. Environ. Manage. 44, 968–980. https://doi.org/10.1007/s00267-009-9380-y

Ramankutty, N., Foley, J.A., 1999. Estimating historical changes in land cover:North American crop lands from 1850 to 1992. Glob. Ecol. Biogeogr. https://doi.org/10.1046/j.1365-2699.1999.00141.x

Ranjan, P., Church, S.P., Floress, K., 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. https://doi.org/10.1080/08941920.2019.1648710

Reimer, A., Thompson, A., Prokopy, L.S., Arbuckle, J.G., Genskow, K., Jackson-Smith, D., Lynne, G., McCann, L., Morton, L.W., Nowak, P., 2014. People, place, behavior, and context: A research agenda for expanding our understanding of what motivates farmers' conservation behaviors. J. Soil Water Conserv. 69, 57A-61A. https://doi.org/10.2489/jswc.69.2.57A

Ricketts, T.H., 2001. The matrix matters: effective isolation in fragmented landscapes. Am. Nat. 158, 87–99. https://doi.org/10.1086/320863

Riseng, C.M., Wiley, M.J., Black, R.W., Munn, M.D., 2011. Impacts of agricultural land use on biological integrity: a causal analysis. Ecol. Appl. 21, 3128–3146. https://doi.org/10.1890/11-0077.1

Ritchie, H., Roser, M., 2020. CO_2 emissions [WWW Document]. Our World Data. URL https://ourworldindata.org/co2-emissions (accessed 4.23.26).

Rosenberg, K.V., Dokter, A.M., Blancher, P.J., Sauer, J.R., Smith, A.C., Smith, P.A., Stanton, J.C., Panjabi, A., Helft, L., Parr, M., Marra, P.P., 2019. Decline of the North American avifauna. Science 366, 120–124. https://doi.org/10.1126/science.aaw1313

Russell, K.N., Ikerd, H., Droege, S., 2005. The potential conservation value of unmowed powerline strips for native bees. Biol. Conserv. 124, 133–148. https://doi.org/10.1016/j.biocon.2005.01.022

Rutkoski, C.E., Evans, S.E., 2025. Newly established prairie strips intermittently increase soil carbon beneath the strip, but not in surrounding cropland. Soil Sci. Soc. Am. J. 89, e70056. https://doi.org/10.1002/saj2.70056

Sanderman, J., Hengl, T., Fiske, G.J., 2017. Soil carbon debt of 12,000 years of human land use. Proc. Natl. Acad. Sci. 114, 9575–9580. https://doi.org/10.1073/pnas.1706103114

Schulte, L.A., Niemi, J., Helmers, M.J., Liebman, M., Arbuckle, J.G., James, D.E., Kolka, R.K., O'Neal, M.E., Tomer, M.D., Tyndall, J.C., Asbjornsen, H., Drobney, P., Neal, J., Van Ryswyk, G., Witte, C., 2017. Prairie strips improve biodiversity and the delivery of multiple ecosystem services from corn–soybean croplands. Proc. Natl. Acad. Sci. 114, 11247–11252. https://doi.org/10.1073/pnas.1620229114

Swanson, K., 2025. High Production Cost Series, Part 1 [WWW Document]. Natl. Corn Grow. Assoc. URL https://ncga.com/stay-informed/media/the-corn-economy/article/2025/08/high-production-cost-series-part-1 (accessed 4.27.26).

Tomer, M.D., Dosskey, M.G., Burkart, M.R., James, D.E., Helmers, M.J., Eisenhauer, D.E., 2009. Methods to prioritize placement of riparian buffers for improved water quality. Agrofor. Syst. 75, 17–25. https://doi.org/10.1007/s10457-008-9134-5

Tomer, M.D., Porter, S.A., James, D.E., Boomer, K.M.B., Kostel, J.A., McLellan, E., 2013. Combining precision conservation technologies into a flexible framework to facilitate agricultural watershed planning. J. Soil Water Conserv. 68, 113A-120A. https://doi.org/10.2489/jswc.68.5.113A

Upadhaya, S., Arbuckle, J.G., 2021. Understanding Factors Influencing Farmers' Engagement in Watershed Management Activities. Front. Sustain. Food Syst. 5. https://doi.org/10.3389/fsufs.2021.669571

US EPA, 2017. National Water Quality Inventory: Report to Congress (No. EPA 841-R-16-011). US EPA.

USDA NASS, 2026. USDA NASS QuickStats [WWW Document]. URL https://quickstats.nass.usda.gov/

(accessed 4.27.26).

USDA NASS, 2024. Wisconsin Ag News: 2023 Corn County Estimates. https://www.nass.usda.gov/Statistics_by_State/Wisconsin/Publications/County_Estimates/2024/WI-CtyEst-Corn-02-24.pdf

USDA NRCS, 2025. 2022 National Resources Inventory Summary Report. USDA NRCS. https://www.nrcs.usda.gov/sites/default/files/2026-02/2022%20NRI%20Summary%20Report.pdf

USDA Risk Management Agency, 2026. RMA Price Discovery Tool [WWW Document]. RMA Price Discov. Tool. URL https://public-rma.fpac.usda.gov/apps/PriceDiscovery/ (accessed 4.27.26).

USGS, 2022. Protected Areas Database U.S. [WWW Document]. Prot. Areas Database US - Overv. URL https://www.usgs.gov/programs/gap-analysis-project/science/pad-us-data-overview (accessed 4.23.26).

USGS, 2020. National Land Cover Database [WWW Document]. Natl. Land Cover Database. URL https://www.usgs.gov/centers/eros/science/national-land-cover-database (accessed 4.23.26).

Wade, A.A., McKelvey, K.S., Schwartz, M.K., 2015. Resistance-surface-based wildlife conservation connectivity modeling: Summary of efforts in the United States and guide for practitioners (No. RMRS-GTR-333). U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ft. Collins, CO. https://doi.org/10.2737/RMRS-GTR-333

Wagner, D.L., Metzler, K.J., Frye, H., 2019. Importance of transmission line corridors for conservation of native bees and other wildlife. Biol. Conserv. 235, 147–156. https://doi.org/10.1016/j.biocon.2019.03.042

Wen, Z., Fu, Y., Zheng, H., Ouyang, Z., 2024. Metapopulation networks unlock the effects of landscape fragmentation on agricultural pests and natural predators. Glob. Ecol. Conserv. 50, e02849. https://doi.org/10.1016/j.gecco.2024.e02849

Williams, B.K., Koneff, M.D., Smith, D.A., 1999. Evaluation of Waterfowl Conservation under the North American Waterfowl Management Plan. J. Wildl. Manag. 63, 417–440. https://doi.org/10.2307/3802628

Wimberly, M.C., Narem, D.M., Bauman, P.J., Carlson, B.T., Ahlering, M.A., 2018. Grassland connectivity in fragmented agricultural landscapes of the north-central United States. Biol. Conserv. 217, 121–130. https://doi.org/10.1016/j.biocon.2017.10.031

Xiang, H., Zhang, Y., Richardson, J., 2016. Importance of Riparian Zone: Effects of Resource Availability at Land-water Interface. Riparian Ecol. Conserv. 3, 1–17. https://doi.org/10.1515/remc-2016-0001

Yang, Y., Loecke, T., Knops, J.M.H., 2022. Surface soil organic carbon sequestration under post agricultural grasslands offset by net loss at depth. Biogeochemistry 159, 303–313. https://doi.org/10.1007/s10533-022-00929-5

Zhang, X., Lark, T.J., Clark, C.M., Yuan, Y., LeDuc, S.D., 2021. Grassland-to-cropland conversion increased soil, nutrient, and carbon losses in the US Midwest between 2008 and 2016. Environ. Res. Lett. 16, 054018. https://doi.org/10.1088/1748-9326/abecbe

Zhou, X., Helmers, M.J., Asbjornsen, H., Kolka, R., Tomer, M.D., Cruse, R.M., 2014. Nutrient removal by prairie filter strips in agricultural landscapes. J. Soil Water Conserv. 69, 54–64. https://doi.org/10.2489/jswc.69.1.54