

Understanding the variation in vegetation composition of prairie restorations within crop fields

by

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The student author, whose presentation of the scholarship herein was approved by the program of study committee, is solely responsible for the content of this thesis. The Graduate College will ensure this thesis is globally accessible and will not permit alterations after a degree is conferred.

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ABSTRACT

Restoring native habitat within agricultural fields offers a way to balance the provisioning of ecosystem services like soil retention with the conservation of native habitat in extremely simplified landscapes, while simultaneously promoting sustainable crop production. As a farming conservation practice, these in-field restorations should be able to reliably achieve target goals across a wide variety of landscapes and contexts. For the vegetation community these goals could include species diversity and consistent native cover, which may facilitate ecosystem service provisioning and invasion resistance in highly disturbed areas.

For this thesis I performed a comprehensive sampling of farms that have implemented in-field tallgrass prairie restorations (hereinafter “prairie strips”) to understand which factors contribute to variation in the restoration outcomes of diversity and target species cover. I focused my analysis on the seed mix, since it is the largest economic investment a landowner will make in this type of restoration, but controlled for other differences between sites such as age, planting size, and planting season. I found that the seed mix richness was a strong predictor of diversity and target species richness across sites. I also found that the season in which a site was planted (spring, summer, fall) affected the establishment of target species, especially forbs, but that this effect was only detected in a subset of farms that had utilized the same seed mix. I found few predictors were associated with the abundance of non-target, or weedy, species across sites, and instead found that site differences explained the majority of variation in these outcomes. Non-target weedy richness and prairie species cover were negatively associated with one another, further underscoring the importance of controlling weedy vegetation for successful prairie establishment.

I explored which species were consistently detected across sites in Appendix C. I found that nine species were reliably detected across sites and seeded at least five times. I only found that two species were never detected across sites but seeded a number of times ($n = 4$), indicating they may be unreliable for this type of restoration practice.

In Appendix D I examined the results of a before-after survey that I distributed to my cooperating landowners and farmers to assess their confidence and interest in identifying plants. I found that the majority of my cooperators were interested in learning to identify more species and would find it most effective for their learning if another “knowledgeable” person taught them. I also found that the majority of my cooperators perceived their botanical literacy to have increased during the two years that this work took place.

In sum, this work demonstrates that certain restoration outcomes can be predictably achieved across a variety of farms through management decisions like the seed mix and that some species can reliably establish across these in-field restorations. More work will be needed to address the factors that consistently suppress non-target vegetation, as it continues to be an unpredictable and problematic aspect of prairie restoration and management.

CHAPTER 1. GENERAL INTRODUCTION

Restoration ecology is an evolving and increasingly relevant field for scientists, policy makers, land managers, and society alike given the increasing degradation and conversion of natural systems (Suding 2011, Suding et al. 2015). Early efforts at restorations tended to be site-specific undertakings with the goal of establishing vegetation that mirrored natural communities (Howell and Jordan 1991, Hobbs and Norton 1996). Many still argue that utilizing reference sites as benchmarks of success is a hallmark of restoration ecology (Balaguer et al. 2014). However, there have been multiple calls for restoration ecology to advance beyond efforts that use historical systems as templates for community composition (Higgs et al. 2014). First, many argue restoration ecology must transition from an ad-hoc approach to a science that is more applicable and predictable across landscapes and varied contexts (Hobbs and Norton 1996, Brudvig 2017). Second, there has been an increasing recognition that climate change and species introductions are resulting in “hybrid” or “novel” ecosystems that might be resistant to efforts to restore historical communities (Suding et al. 2004, Hobbs et al. 2009). And lastly, as scientists have continued to raise alarms over the numerous negative environmental consequences that human-driven land use change is having on the planet (Foley et al. 2005), there has been a growing emphasis of the role that restoration could play in provisioning ecosystem services in addition to conserving biodiversity (Rey Benayas et al. 2009).

Land use change due to agriculture is the largest reason for native habitat decline across the globe (Foley et al. 2005). Traditionally, working agricultural lands have been viewed as separate from conservation efforts (Scherr and McNeely 2008). Agricultural landscapes are often not considered hospitable environments for native flora and fauna due to practices like tillage, fertilizer application, and pesticide use (McLaughlin and Mineau 1995, Mozumder and Berrens

2007). Yet restoring pieces of native habitat in highly modified landscapes, like productive agricultural regions, could help reverse some of the damage that those very same land conversions caused (Hobbs and Norton 1996). Implementing ecological restorations within crop-fields may be a crucial step for reversing the damages caused by land-use simplification, preserving biodiversity, and maintaining sustainable food production (Kremen and Merenlender 2018).

The state of Iowa has one of the most modified landscapes in the USA. Within a lifetime of European settlement in the mid-1800s, tallgrass prairie, the dominant native habitat, was nearly completely converted to crop- or pasture-land. It is estimated that 28 of Iowa's 36 million acres were tallgrass prairie prior to European settlement and by 1940 27 million of those acres were in cropland (Smith 1998, U.S. Bureau of the Census & Census of Agriculture 1964). Cropland acres have remained steadily high since that time (U.S. Department of Agriculture 2015). Calls to protect the tallgrass prairie began soon after its demise in the early 1900s, but gained serious traction starting in the 1970s (Smith 1998). Iowa now has an estimated 54,000 acres of restored prairie planted on roadsides and over 96,000 acres planted in conservation areas (Kaul and Wilsey 2019), but native habitat restorations are still lacking in the dominant agricultural landscape.

The consequences of this massive shift from diverse native habitat to productive, simplified cropping system are quite evident. Agriculture in Iowa has been linked to environmental issues such as increased stream nitrate and pesticide concentrations, as well as increased soil erosion (Schilling and Libra 2000, Vecchia et al. 2009, Heathcote et al. 2013). Furthermore, climate change projections predict more intense precipitation events in the

Midwestern Corn Belt, potentially exacerbating soil loss and water quality issues as well as increasing crop damage in the years to come (Rosenzweig et al. 2002, O’Neal et al. 2005).

The idea of integrating restoration efforts into agricultural landscapes is not new (Hobbs and Norton 1996), yet utilizing restorations to mitigate agriculturally-caused environmental damage while simultaneously providing conservation habitat is a novel approach (Schulte et al. 2017). In Iowa, experiments testing the effects of sowing tallgrass prairie restorations into agricultural fields began in the mid-2000s (Liebman et al. 2013). These infield prairie restorations (hereinafter “prairie strips”) are based on the idea that converting a small amount of land to native perennial vegetation can offer disproportionate environmental and sociological benefits to landowners (Asbjornsen et al. 2014). Furthermore, taking a small amount of land out of production may actually increase profitability if the targeted land is low yielding (Brandes et al. 2016).

There is already ample evidence that prairie strips can be an effective farming conservation practice in certain locations (Schulte et al. 2017). As this practice expands across agricultural operations and contexts, it is important to monitor these restorations to ensure they reliably achieve their stated goals and targets. For the vegetation community these targets may be diversity and consistent native cover, which should facilitate multiple ecosystem service provisioning and resist invasion by agricultural weeds. Understanding the factors which explain variation in these restoration targets will aid in the reliable establishment and success of this farming conservation practice going forward.

Thesis organization

This work aims to understand the patterns in vegetation communities of in-field prairie restorations on working farms of Iowa with the goal of predicting restoration outcomes. Chapter 2 details the methodology and results of that investigation. Appendix A enumerates all the

species found during that investigation and their classifications. Appendix B has supplementary figures and tables from Chapter 2. Appendix C serves as a further exploration into identifying which plant species establish the most consistently across sites, with the goal of helping design more effective seed mixes for in-field restorations. Appendix D has the results of a survey that was distributed to cooperating landowners and farmers before and after the research project. This survey was intended to gauge cooperators' confidence and interest in identifying prairie plants and to see what methods would be helpful in increasing botanical literacy in farmers and landowners. Appendix E has the IRB approval for this survey.

References

- Asbjornsen, H., Hernandez-Santana, V., Liebman, M., Bayala, J., Chen, J., Helmers, M., Ong, C.K., Schulte, L.A., 2014. Targeting perennial vegetation in agricultural landscapes for enhancing ecosystem services. *Renew. Agric. Food Syst.* 29, 101–125. <https://doi.org/10.1017/S1742170512000385>
- Balaguer, L., Escudero, A., Martín-Duque, J.F., Mola, I., Aronson, J., 2014. The historical reference in restoration ecology: Re-defining a cornerstone concept. *Biol. Conserv.* 176, 12–20. <https://doi.org/10.1016/j.biocon.2014.05.007>
- 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. *Environ. Res. Lett.* 11, 014009. <https://doi.org/10.1088/1748-9326/11/1/014009>
- Brudvig, L.A., 2017. Toward prediction in the restoration of biodiversity. *J. Appl. Ecol.* 54, 1013–1017. <https://doi.org/10.1111/1365-2664.12940>
- Foley, J.A., DeFries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin, F.S., Coe, M.T., Daily, G.C., Gibbs, H.K., Helkowski, J.H., Holloway, T., Howard, E.A., Kucharik, C.J., Monfreda, C., Patz, J.A., Prentice, I.C., Ramankutty, N., Snyder, P.K., 2005. Global consequences of land use. *Science.* 309, 570–574. <https://doi.org/10.1126/science.1111772>
- Heathcote, A.J., Filstrup, C.T., Downing, J.A., 2013. Watershed sediment losses to lakes accelerating despite agricultural soil conservation efforts. *PLoS One* 8, e53554. <https://doi.org/10.1371/journal.pone.0053554>

- Higgs, E., Falk, D.A., Guerrini, A., Hall, M., Harris, J., Hobbs, R.J., Jackson, S.T., Rhemtulla, J.M., Throop, W., 2014. The changing role of history in restoration ecology. *Front. Ecol. Environ.* 12, 499–506. <https://doi.org/10.1890/110267>
- Hobbs, R.J., Norton, D.A., 1996. Towards a conceptual framework for restoration ecology. *Restor. Ecol.* 4, 93–110. <https://doi.org/10.1111/j.1526-100X.1996.tb00112.x>
- Hobbs, R.J., Higgs, E., Harris, J.A., 2009. Novel ecosystems: implications for conservation and restoration. *Trends Ecol. Evol.* 24, 599–605. <https://doi.org/10.1016/j.tree.2009.05.012>
- Howell, E.A., Jordan, W.R., 1991. Tallgrass prairie restoration in the North American Midwest, in: *The scientific management of temperate communities for conservation*. pp. 395–414.
- Kaul, A.D., Wilsey, B.J., 2019. Monarch butterfly host plant (milkweed *Asclepias* spp.) abundance varies by habitat type across 98 prairies. *Restor. Ecol.* 27, 1274–1281. <https://doi.org/10.1111/rec.12993>
- Liebman, M., Helmers, M.J., Schulte, L.A., Chase, C.A., 2013. Using biodiversity to link agricultural productivity with environmental quality: Results from three field experiments in Iowa. *Renew. Agric. Food Syst.* 28, 115–128. <https://doi.org/10.1017/S1742170512000300>
- Kremen, C., Merenlender, A.M., 2018. Landscapes that work for biodiversity and people. *Science*. 362, eaau6020. <https://doi.org/10.1126/science.aau6020>
- McLaughlin, A., Mineau, P., 1995. The impact of agricultural practices on biodiversity. *Agric. Ecosyst. Environ.* 55, 201–212. [https://doi.org/10.1016/0167-8809\(95\)00609-V](https://doi.org/10.1016/0167-8809(95)00609-V)
- Mozumder, P., Berrens, R.P., 2007. Inorganic fertilizer use and biodiversity risk: An empirical investigation. *Ecol. Econ.* 62, 538–543. <https://doi.org/10.1016/j.ecolecon.2006.07.016>
- O’Neal, M.R., Nearing, M.A., Vining, R.C., Southworth, J., Pfeifer, R.A., 2005. Climate change impacts on soil erosion in Midwest United States with changes in crop management. *CATENA* 61, 165–184. <https://doi.org/10.1016/j.catena.2005.03.003>
- Rey Benayas, J.M., Newton, A.C., Diaz, A., Bullock, J.M., 2009. Enhancement of biodiversity and ecosystem services by ecological restoration: A Meta-Analysis. *Science*. 325, 1121–1124. <https://doi.org/10.1126/science.1172460>
- Rosenzweig, C., Tubiello, F.N., Goldberg, R., Mills, E., Bloomfield, J., 2002. Increased crop damage in the US from excess precipitation under climate change. *Glob. Environ. Chang.* 12, 197–202. [https://doi.org/10.1016/S0959-3780\(02\)00008-0](https://doi.org/10.1016/S0959-3780(02)00008-0)
- Scherr, S.J., McNeely, J.A., 2008. Biodiversity conservation and agricultural sustainability: towards a new paradigm of ‘ecoagriculture’ landscapes. *Philos. Trans. R. Soc. B Biol. Sci.* 363, 477–494. <https://doi.org/10.1098/rstb.2007.2165>

- Schilling, K.E., Libra, R.D., 2000. The relationship of Nitrate concentrations in streams to row crop land use in Iowa. *J. Environ. Qual.* 29, 1846–1851.
<https://doi.org/10.2134/jeq2000.00472425002900060016x>
- 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>
- Smith, D., 1998. Iowa prairie: Original extent and loss, preservation and recovery attempts. *J. Iowa Acad. Sci.* 105, 94–108.
- Suding, K.N., 2011. Toward an era of restoration in ecology: successes, failures, and opportunities ahead. *Annu. Rev. Ecol. Evol. Syst.* 42, 465–487.
<https://doi.org/10.1146/annurev-ecolsys-102710-145115>
- Suding, K.N., Gross, K.L., Houseman, G.R., 2004. Alternative states and positive feedbacks in restoration ecology. *Trends Ecol. Evol.* 19, 46–53.
<https://doi.org/10.1016/j.tree.2003.10.005>
- Suding, K., Higgs, E., Palmer, M., Callicott, J.B., Anderson, C.B., Baker, M., Gutrich, J.J., Hondula, K.L., LaFevor, M.C., Larson, B.M.H., Randall, A., Ruhl, J.B., Schwartz, K.Z.S., 2015. Committing to ecological restoration. *Science.* 348, 638–640.
<https://doi.org/10.1126/science.aaa4216>
- U.S. Department of Agriculture, 2015. 2015 National Resource Inventory.
https://www.nrcs.usda.gov/Internet/NRCS_RCA/reports/nri_ia.html (Accessed 6 May 2020).
- U.S. Bureau of the Census & Census of Agriculture, 1964. Statistics for the state and counties, Iowa. <http://usda.mannlib.cornell.edu/usda/AgCensusImages/1964/01/16/1964-01-16.pdf> (Accessed 7 May 2020).
- Vecchia, A. V., Gilliom, R.J., Sullivan, D.J., Lorenz, D.L., Martin, J.D., 2009. Trends in concentrations and use of agricultural herbicides for corn belt rivers, 1996–2006. *Environ. Sci. Technol.* 43, 9096–9102. <https://doi.org/10.1021/es902122j>

CHAPTER 2. UNDERSTANDING THE VARIATION IN VEGETATION COMPOSITION OF PRAIRIE RESTORATIONS WITHIN CROP FIELDS

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Abstract

Strategically sowing strips of native vegetation within highly productive crop fields is a novel farming practice that can improve soil and water conservation while increasing biodiversity in highly simplified landscapes. As in-field ecological restorations are further adopted across a wide variety of farming operations, it is important to understand the factors that consistently predict restoration goals and outcomes. We performed a comprehensive vegetation survey of farms across Iowa, USA that have implemented on-farm tallgrass prairie restorations (“prairie strips”) to understand what factors were driving variation in the diversity and native cover of these plantings. We found that species richness of the seed mix was positively associated with plant species diversity at multiple scales as well as the target species richness. When we examined a subset of sites that utilized the same seed mix, we found that planting season drove variation in the richness and cover of target species, with fall plantings having higher relative cover of forbs. Our work demonstrates that management decisions, especially the composition of the seed mix, can have a strong and predictable effects on restoration outcomes across a wide variety of farming operations with independent managers.

Introduction

Restoring native habitat within working agricultural lands is increasingly being considered a crucial component for balancing sustainable production, biodiversity conservation, and ecosystem service provisioning (Schulte et al. 2017, Kremen and Merenlender 2018). In-field ecological restorations are often based on the idea that a small amount of native perennial cover, when placed strategically, can offer disproportionate beneficial effects on ecosystem services, such as soil conservation and nutrient retention, without strongly compromising agricultural yield (Asbjornsen et al. 2014). Furthermore, areas of restored habitat within agricultural landscapes have been shown to be effective at conserving biodiversity in insects (Kremen and M'Gonigle 2015), plants (Hirsh et al. 2013), birds (Harvey et al. 2006), and other fauna (Medina et al. 2007).

While the idea of restoring native habitat within working agricultural lands holds much promise for biodiversity conservation and ecosystem service provisioning, a large hurdle remains in restoration ecology: increasing the predictability of outcomes (Suding 2011, Brudvig 2017). Much of the variation that exists among restorations remains unexplained (Grman et al. 2013), and similar restoration practices may result in widely different outcomes in terms of species diversity or establishment (Stuble et al. 2017, Groves and Brudvig 2019). While unpredictability may always be an inherent part of ecological restoration (Norland et al. 2018), there is a growing consensus that we may be able to generate more reliable outcomes across contexts by interpreting and understanding the variation among restoration efforts (Brudvig et al. 2017, Brudvig 2017, Zirbel et al. 2017).

Variation among restoration efforts may be a result of both the restoration context and the outcomes being measured (Brudvig et al. 2017). In the context of tallgrass prairie restorations,

management decisions like the composition of the seed mix have been shown to greatly contribute to restoration outcomes like diversity and species' abundances (Grman et al. 2013, Meissen et al. 2019). Increasing the diversity of the seed mix has been linked to an increase in the diversity of the vegetation community (Larson et al. 2011, Carter and Blair 2012). Furthermore, diverse seed mixes can result in communities that are less dominated by weedy or exotic species (Carter and Blair 2012, Nemeč et al. 2013) due to the presence of one or more highly competitive species and/or the more complete resource use of a diverse community (Fargione and Tilman 2005).

In addition to management decisions like seed mix composition, site characteristics and the timing of management activities can alter prairie restoration outcomes. The initial configuration of a planting, such as its size and shape, may affect outcomes like diversity and invasion resistance. Larger restorations are likely to harbor more species due to the species-area relationship (Connor and McCoy 1979, Rosenzweig 1995), and a higher proportion of edge habitat could lead to more weeds, especially in highly disturbed environments like agricultural fields where weed pressure is presumably high (Rowe et al. 2013). The time of year prairie is seeded can also impact the establishment of different functional groups, given that some native species, forbs especially, need a cold-wet stratification period for germination (Rowe 2010, Kurtz 2013). Lastly, age is an important factor to control for in tallgrass prairie restoration outcomes, as restoration outcomes can change with succession (Young et al. 2001). For example, diversity often decreases over time in prairie restorations, usually due to the increasing dominance of warm season grasses (Sluis 2002, Camill et al. 2004, Grman et al. 2013, Kordbacheh et al. 2019).

In the Midwestern USA, strategically planting tallgrass prairie into and around annual row-cropped fields (hereinafter “prairie strips”) has received increasing attention as a farming conservation practice. Agriculture has replaced nearly all the original tallgrass prairie habitat in the region (Smith 1998) and prairie strips are a means of increasing native habitat within this agriculturally-dominated landscape. Annual row-crop agriculture in Iowa has also been linked to water quality issues like nitrate leaching (Jones et al. 2018). Prairie strips can help to mitigate environmental problems like water pollution by increasing the ecosystem service provisioning of soil retention (Helmets et al. 2012) and water filtration (Zhou et al. 2014). As prairie strips become a more widespread farming conservation practice across the Midwestern landscape, it is important that we monitor these restorations to understand what factors are driving their diversity and vegetation cover, two metrics we believe are important for both the conservation value and ecosystem service provisioning of these restorations.

Fostering biodiversity has been a longstanding target of restoration ecology, and the importance of biodiversity for ecosystem functioning has been increasingly demonstrated across systems (Rey Benayas et al. 2009, Isbell et al. 2017). In grassland systems, higher biodiversity has been linked to greater above- and belowground biomass (Mueller et al. 2013), soil C and N accumulation (Fornara and Tilman 2008), and resistance to invasion (Naeem et al. 2000). Furthermore, grassland systems that are more diverse can provide multiple ecosystem services simultaneously (Zavaleta et al. 2010). Given that prairie strips are intended to deliver multiple ecosystem services (Schulte et al. 2017), understanding the factors that affect the biodiversity of these plantings is important for their success.

Weeds and other invasive plants threaten prairie restorations across the Midwestern, USA (Grant et al. 2009), and weeds can cause large economic losses in annual cropping systems

(Oerke 2006). Prairie restorations undertaken within agricultural fields, which are often characterized by high nutrient levels and disturbance, are likely going to receive substantial competition from weeds (Baer et al. 2009). Furthermore, studies of community assembly and ‘priority effects’ have shown that if weeds establish before prairie species they can dominate and persist in restorations for years (Martin and Wilsey 2014, Wilsey et al. 2015). Therefore, understanding which factors help to promote diversity and cover of prairie species early, and conversely suppress weeds, in these plantings is essential for both the health of the restoration and the viability of the farming operation.

Here we report data from 25 sites across Iowa, USA that have integrated prairie strips into their agricultural operations. We undertook a comprehensive evaluation of the vegetation in these restorations to understand what factors contribute to variation in plant diversity and establishment of prairie species across a variety of locations. We focused our analysis on the seed mix, since it is the largest economic investment a landowner will make in this type of restoration (Phillips-Mao et al. 2015). However, we also evaluated some of the other factors that differed between sites such as age, size, shape, and planting season.

Methods

Site Information

We chose to do a comprehensive survey, visiting sites in Iowa that had planted prairie strips on their farming operation and were at least three years old. We visited 20 sites in 2018 and 25 sites in 2019. The sites were located within 24 counties in Iowa, USA (Figure 1) and were seeded between 2012 and 2016. We chose not to visit sites that were younger than three years old since it can take at least two years for native prairie plants to become established due to the growth of deep root systems (Schramm 1990) and, similar to other studies, we wanted to

compare plantings that were mature enough to observe vegetation dynamics and evaluate restoration success (Grman et al. 2013).

The majority of sites were in cultivation prior to seeding prairie strips. Although differences did occur across farms in terms of management (e.g. tillage regime, organic vs. conventional pest control and fertility regimes) we feel that our sites represent the variety of farming practices that occur on the Iowan landscape. Most sites were located in row-crop dominated landscapes. We did not take into account immediate features of the surrounding landscape, such as the proportion of land in forest, pasture, etc., as this hasn't been shown to affect prairie restorations in a similar context (Grman et al. 2013).

Field Methods

We surveyed the vegetation in each site once during the summer (mid-July through mid-August) of 2018 and 2019. Our later-season sampling time was intended to capture peak flowering of the prairie community but does mean that certain earlier flowering plants could already be senescing and reduced in biomass (Jarchow and Liebman 2012). The order in which sites were surveyed differed between years but was not entirely random due to coordination with landowners and farmers. At each site, 24 quadrats, or vegetation sampling points, were used. The quadrat measured 0.5 m by 1 m. Because sites differed in the number and shape of their prairie strips, sampling point locations were randomly chosen through the 'Create Random Points' function in ArcGis version 10.5.1. Random locations were at least 5 m from each other and 2 m from the prairie strip edge. If sites had more than one prairie strip, then the number of sampling points in a strip was proportional to that strip's area. Every strip had at least one sampling point. Sampling point locations were staked in the field using a Trimble GeoXT 2005 unit in 2018 and a Trimble Geo7x unit in 2019. The approximate location of sampling stayed constant between

years, but due to inaccuracies in GPS/poor satellite reception, quadrat locations may have varied by 1-2 m between years.

Within each quadrat we determined the identity and percent cover of each species. Percent cover estimates are based on a modified Daubenmire (1959) scale with 7 cover classes (Bailey and Poulton 1968). Because of multiple vegetation layers, total cover estimates for each sampling point often exceeded 100%.

Species were classified according to their life history attributes. We categorized species as native or introduced based on the Vascular Plants of Iowa database (Eilers and Roosa 1994). Species were categorized as “prairie” if they were listed in the University of Iowa Libraries’ “Iowa Prairie Plants” database (The University of Iowa Libraries 2020). For some native species that were not listed on this website but were included in seed mixes, those species were also classified as “prairie.” All other non-woody species were classified as “weedy” whether they were native or exotic. Our “prairie” versus “weedy” classification was due to a number of species that are native to Iowa but are considered noxious agricultural weeds (USDA and NRCS 2002), and the rise in recognition that endemic species can act like invasive species under anthropogenic changes (Simberloff et al. 2012). We believe our classification most accurately describes the target and the non-target vegetation in these agricultural plantings. A complete list of species and their classifications can be found in Appendix A and all survey data is publicly available (English et al. 2020).

In addition to the survey we performed via our randomized sampling points, we recorded all prairie species observed while walking between quadrats. While we did not exhaustively search the entirety of sites, recording additional species allowed us to more accurately estimate the proportion of the seed mix detected at each site.

Statistical Analysis

All analyses were performed in R version 3.6.2 (R Core Team 2019). Cover classes were converted to midpoints for the analysis. Diversity was calculated as Shannon Hill numbers ($e^{H'}$) (Jost 2006) using the {vegan} package in R (Oksanen et al. 2019). Diversity was calculated for each quadrat (alpha) and across quadrats (gamma), yielding 24 measures of alpha diversity and one measure of gamma diversity for each site in each sampling year. Beta diversity was calculated as gamma/average alpha diversity, using Shannon Hill numbers, which yields the effective number of distinct communities (Jost 2007). Our diversity calculations included all species, therefore we also calculated the species richness of both prairie and weedy species at the site (gamma species richness) and the quadrat (alpha species richness) level to parse out which species were driving patterns in diversity. We used the relative cover (cover/total cover) of different functional groups to compare species abundance across sites. We sub-classified prairie species by grasses (C₄ and C₃) and forbs (leguminous and non-leguminous). We sub-classified weedy species by their lifecycle (annual and perennial/biennial) due to the difficulty in managing perennial weeds in prairie restorations whether they are grasses (Salesman and Jessica 2011) or forbs (Larson et al. 2013). Biennial species were included with perennials. The proportion of species in the seed mix detected at each site (hereinafter the “detection rate”) was calculated as the number of seeded species found divided by the number of species seeded.

We ran mixed effect models using the {lmer} and {lmerTest} packages (Bates et al. 2015, Kuznetsova et al. 2017) for all our response variables. We included site as a random effect in all our models because we were more interested in the variation across our study locations than the mean at each location (Bolker et al. 2009). For quadrat level metrics (e.g., alpha diversity) we added quadrat nested within site as a random effect. We used up to 6 fixed effects

in our models. Sampling year was included as an independent variable in all models. Seed mix species richness was also kept in all models since it was our primary variable of interest. We were unable to procure information about the seed mix from two of our sites and they were largely excluded from the analysis (Figure 1). The other four fixed effects were covariates that we kept in models only if they significantly improved model fit (see explanation below): site age, the area planted to prairie strips (hereinafter site size), the average ratio of the perimeter to the area of the prairie strips, and the season in which a site was planted. Site age was calculated as the difference between planting date and 1/1/2020 and was converted to years in the R package `{lubridate}` (Grolemund and Wickham 2011). Site size and the average perimeter-to-area ratio were log transformed to account for large differences among sites. Fall and winter plantings were combined, yielding three levels of season planted (fall, spring, and summer). None of the independent variables had a variance inflation factor (VIF) over 3.

We started with full models and then eliminated non-significant covariates in a step-wise fashion, starting with the least significant covariate. Including non-significant covariates did not improve the model fit when compared in a likelihood ratio test. Random effects were also checked for significance using likelihood ratio tests. Marginal and conditional R^2 values, which are the variation explained by the fixed effects and the variation explained by the fixed and random effects, respectively, were calculated for every model using the `{performance}` package (Ludecke et al. 2020), which is based on the methods described by Nakagawa and Schielzeth (2013).

We checked assumptions of normality using the `{performance}` package (Ludecke et al. 2020) and transformed certain response variables to better fit with the assumptions of normality. Species richness at the site level was log-transformed (Ives 2015, Warton et al. 2016) and

proportions, e.g. relative cover of different functional groups and the detection rate, were logit transformed (Warton and Hui 2011).

Six of the sites surveyed in the summer of 2019 were planted with the same seed mix (Figure 1). Therefore, we ran a separate analysis on these sites only with our remaining covariates (age, size, perimeter-to-area ratio, and season planted). Given the small size of this sub-sample we ran separate simple linear regressions for each covariate.

Results

Diversity and richness

Gamma (site-level) diversity was positively associated with seed mix richness and site size, but not with our other covariates: site age, perimeter-to-area ratio, or season planted (Table 1). Increasing seed mix richness by one species increased gamma diversity on average by 0.37 (95% CI: 0.19-0.55, $p = <0.001$) after accounting for sampling year and site size. Doubling site size increased diversity by 1.79 (95% CI: 0.65-2.93, $p = 0.007$) after accounting for sampling year and seed mix richness.

Alpha diversity was negatively associated with site age and positively associated with the size of a site, but not associated with the number of species seeded or our other covariates (Table 1). For every year a site aged, alpha diversity declined on average by 0.63 (95% CI: 0.35-0.90, $p < 0.001$) after accounting for sampling year, seed mix richness and site size. Doubling site size increased alpha diversity by 0.49 (95% CI: 0.29, 0.69, $p < 0.001$) after accounting for sampling year, seed mix richness, and site age.

Beta diversity was only positively associated with the number of species seeded, and none of our other covariates (Table 1). Increasing the seed mix richness by one species increased

beta diversity, on average, by 0.05 (95% CI: 0.02-0.07, $p = 0.003$) after accounting for sampling year.

The gamma (site-level) richness of the prairie species was also significantly positively associated with the number of species seeded and the size of the site (Table 2). Increasing seed mix richness by one species increased median gamma prairie richness by 1.8% (95% CI: 0.95-2.58%, $p < 0.001$) after accounting for sampling year and site size. Doubling site size increased median gamma prairie richness by 5.6% (95% CI: 0.4-11%, $p = 0.05$) after accounting for sampling year and seed mix richness.

The alpha (quadrat-level) richness of prairie species was lower in summer plantings as compared to fall plantings (Table 2). Summer plantings had on average 2.3 (95% CI: 0.68 – 3.91 $p = 0.04$), less prairie species per quadrat as compared with fall plantings, after accounting for sampling year and seed mix richness.

Neither the gamma (site-level) nor alpha (quadrat-level) weedy richness were significantly associated with any of our predictor variables, besides the sampling year (Table 2).

Cover of Different Functional Groups

We found only a few of our fixed effects explained variation in the relative cover of different functional groups. Annual weeds and prairie legumes were both negatively associated with site age, after accounting for sampling year and seed mix richness (Table 3). Prairie C₃ grass relative cover was positively associated with site size and perimeter-to-area ratio, after accounting for sampling year and seed mix richness (Table 3). The seed mix richness did not significantly affect the relative cover of any grouping of prairie species or weedy species (Table 3, Table B1). Our random effect (site) explained much of the variation in most of our models of functional group cover (Table B1). Sampling year was statistically significant for the relative

cover of all prairie species, C₄ grasses, and perennial weeds, after accounting for seed mix richness (Table B1).

We examined the association between prairie and weedy species and found the abundance of prairie species and richness of weed species to be negatively associated (Figure 2A). This relationship held when we tested the association of the relative cover of prairie grasses and weedy richness (Figure 2B), but not the relative cover of prairie forbs and weedy richness (Figure 2C). Similarly, we found a negative association between the relative cover of weedy species and the richness of prairie species, although this trend was not as strong (Figure B1A, $p_{\text{cov}} = 0.02$, $R_m^2 = 0.16$). This relationship no longer held when we just examined relative cover of annual weeds (Figure B1B, $p_{\text{cov}} = 0.43$, $R_m^2 = 0.01$), however it was still statistically significant when examining perennials weeds only (Figure B1C, $p_{\text{cov}} = 0.02$, $R_m^2 = 0.15$).

Detection Rate

We found that none of our predictor variables were associated with variation in the detection rate of the seed mix. However, we did find that certain species reliably established across all the sites where they were seeded, while other species had lower detection rates across sites. Three grasses (*Andropogon gerardii*, *Elymus canadensis*, *Elymus virginicus*), five non-leguminous forbs (*Heliopsis helianthoides*, *Monarda fistulosa*, *Ratibida pinnata*, *Pycnanthemum virginianum*, *Silphium integrifolium*), and one legume (*Desmodium canadense*) were detected ubiquitously and seeded across at least five different sites. One non-leguminous forb (*Helenium autumnale*) and one legume (*Amorpha canescens*) were never detected and were seeded at four sites. Further information about the detection rate of specific species across sites can be found in Appendix C.

Seed mix subset

When examining the subset of sites where the same seed mix was sown ($n = 6$), we found that the season in which a site was planted explained variation in the richness and cover of prairie species, while age, size, and perimeter-to-area ratio were insignificant covariates.

Prairie species richness was significantly lower in summer planted sites compared with fall planted sites (Figure 3A), however there was no difference in the richness of weedy species between different planting seasons (Figure 3B). The relative cover of prairie species was significantly lower in summer planted sites compared with fall and spring planted sites (Figure 4A). There were no significant differences in the relative cover of grasses among planting seasons (Figure 4C), but the relative cover of forbs was highest in the fall plantings and lowest in the summer plantings, with spring plantings at an intermediate coverage (Figure 4B). Conversely, the relative cover of weedy species was significantly higher in the summer plantings as compared to the fall ($t_3 = 4.09$, $p = 0.05$) and spring ($t_3 = 5.2$, $p = 0.03$) plantings, however this trend was no longer significant when weeds were subdivided into annuals and perennials (Figure B2).

Discussion

Within crop-field restorations could serve as an essential component of strategies linking sustainable agriculture and biodiversity conservation, yet our ability to predict restoration outcomes is still lacking (Brudvig 2017). We compared the vegetation in 25 sites across Iowa, USA that have integrated tallgrass prairie restorations into their working agricultural fields to examine what factors contributed to variation in plant diversity and abundance among sites. Overall, we found that the diversity and richness of vegetation at our sites was a more predictable outcome than the abundance of different functional groups, given our model parameters.

Diversity Outcomes

The richness of the seed mix was important for some, but not all, of our diversity parameters. In accordance with our predictions, the seed mix richness was positively associated with the gamma- (site-level) diversity and richness of prairie species across sites (Table 1 and Table 2). In concert with the fact that gamma weedy richness was not associated with any of our predictor variables, this finding suggests that the relationships between gamma diversity and seed mix richness was driven by prairie species. This demonstrates once again that diversity begets diversity. Investing in a more speciose seed mix can result in a higher number of prairie species being found across sites (Grman et al. 2013). Site size was also positively associated with gamma diversity and prairie species richness (Table 1 and Table 2), which supports the species-area hypothesis, that the number of species found increases with increasing area (Rosenzweig 1995). This positive association wasn't due to a sampling effect since we had equal sampling effort across sites (Connor and McCoy 1979). Nonetheless, it is uncertain if the mechanism behind this trend is that our larger sites were more environmentally heterogenous than our smaller sites (Rosenzweig 1995) since we did not incorporate measurements of environmental heterogeneity into our analysis. However, it does suggest that if landowners can plant more area in prairie, they are likely to establish more target species successfully.

We also found support for the hypothesis that increasing the richness of the seed mix increases the beta diversity of sites (Table 1). This conforms with theory and experimental work suggesting that a larger species pool size will be associated with an increase in beta diversity (Chase 2003, Grman and Brudvig 2014). Recent work suggests that beta diversity is an important component of increasing ecosystem multifunctionality across systems (Mori et al. 2018) and specifically in prairie restorations (Grman et al. 2018). Interestingly, we did not find a

relationship between species pool size and local (alpha) diversity, as other work as demonstrated (Polley et al. 2005). Instead we found that age and alpha diversity were negatively associated. Previous work has demonstrated diversity declines with age, and this pattern could be due to our observation that annual species dropped out over time (see discussion below) (Sluis 2002, Middleton et al. 2010, Grman et al. 2013). Importantly, however, gamma and beta diversity were not negatively associated with age, indicating that different species are able to persist across these restorations through time.

Functional Group Cover Outcomes

Conversely, the seed mix richness was not a strong predictor of the abundance of different plant functional groups. Instead we found that age was negatively associated with the cover of annual weeds and prairie legumes (Table 2). The decline of annual weeds in prairie restorations with succession is a commonly observed pattern due to the maturation and competitive advantage of perennial prairie plants compared to early successional weeds (Camill et al. 2004, Blumenthal et al. 2005, Hirsh et al. 2013, Meissen et al. 2019). Our observation that legumes and age were negatively associated could be due to several factors. First, we found a significant negative association between the relative proportion of the seed mix devoted to legume species and age in a subset of our sites for which we had seeding abundance data available ($n = 15$, $R^2 = 0.56$, $p = 0.001$). This finding suggests that the negative cover-age relationship we observed in legumes could be an artifact of older sites merely having a lower amount of legume seed in their seed mix. Second, legume species have been shown to have lower photosynthetic rates under N fertilization compared to control treatments (Reich et al. 2003), and under repeated N fertilization they become less abundant over time (Kordbacheh et al. 2019). While prairie strips are not directly fertilized, the surrounding portions of agricultural

field continue to receive fertilization typical of the farming operation, and not all fertilizer stays where it is applied (Sebilo et al. 2013). Therefore, it is also possible that a decline in legume species over time is caused in part by inadvertent fertilizer leakage into the prairie community.

We found that the relative cover of all prairie species and C₄ grasses declined on average in the 2019 sampling year compared with the 2018 sampling year and that the relative cover of perennial weeds increased on average between our two sampling years (Table B1). None of the five most commonly found perennial weeds were the drivers of this pattern (Table B2), however when examining prairie C₄ grasses individually, the relative cover of both *Sorghastrum nutans* and *Bouteloua curtipendula* declined, on average, between sampling years (Table B3) which could help explain our observations of the decline of this functional group between sampling seasons. Although year-to-year variation in functional group cover occurs in prairie restorations, even when visiting the same sampling locations (Camill et al. 2004), continued monitoring to ensure that perennial weeds do not continue to increase at the expense of prairie grasses will be important for these in-field restorations.

Furthermore, we found that non-target species richness was negatively related to the relative abundance of target species (Figure 2A), indicating that prairie and weedy species can adversely affect the presence of one another. Grass cover appeared more strongly tied to a decline in weed diversity than forb cover (Figure 2B/2C), which indicates that grasses could be effective at lowering the diversity of weed species and/or filtering the weedy community for the most competitive species (Wilsey 2020). Other recent work has found that exotic species abundance is a strong negative predictor of prairie species diversity and establishment (Kaul and Wilsey 2020), further underscoring the importance of controlling non-target vegetation for target species success.

When we subset our data and examined only those sites that utilized the same seed mix we found that planting season affected the both richness and cover of prairie species, with forbs being especially responsive (Figure 3 and 4). Other work has found that dormant season seeding results in a higher cover of planted species whereas growing season seeding results in a higher cover of exotics (Larson et al. 2011, Martin and Wilsey 2012). While planting season did affect the richness of prairie species at the quadrat-level (Table 2), there was no pronounced effect of planting season on the relative cover of prairie species across all our sites (Figure B3). While our work demonstrates there are factors that can explain variation across a wide range of site conditions, it also indicates the importance of balancing observational studies like ours with controlled experiments to fully understand the relative impact that different factors have in predicting restoration outcomes.

Other Factors Impacting Predictability

Factors outside the scope of this study could explain more of the variation that we observed, especially in weed abundance, which has long been identified as the one of the most unpredictable factors in a prairie restoration (Schramm 1990) and which we and others have shown is negatively associated with prairie species establishment and diversity (Figure 2, Figure B1, Kaul and Wilsey 2020). Recent work has outlined the importance of stochastic factors like first year temperature and precipitation in determining community assembly and the abundance of weedy species in particular (Groves et al. 2020). We also suspect that legacy effects, like the amount of weed seeds in the seedbank at the time of planting, are an integral part of predicting the abundance of non-target vegetation at a restoration site (Grman and Suding 2010). Ensuring a depleted weed seed-bank could be a good safeguard against early weed establishment and persistence (Zirbel et al. 2017). Pesticide drift or nutrient enrichment are other stochastic factors

specific to in-field restorations that could also affect the relative balance between target and non-target vegetation.

While we largely focused our analysis on covariates that are landowner-based decisions at the time of planting, there are other management decisions that could alter the trajectory of communities. The amount and timing of disturbance, like fire, are important components of maintaining prairie diversity and cover (Tix and Charvat 2005). In regard to weed management, however, fire on its own may not be an effective enough management tool to suppress the perennial herbaceous weedy species that can invade prairie strips (Ditomaso et al. 2006, Bahm et al. 2011). Therefore, future research should continue to explore the effects of stochastic factors as well as legacy effects and management practices like fire on suppressing perennial weed pressure in prairie restorations.

Conclusions

There are many factors that can contribute to prairie restoration outcomes, but here we have shown that across a realistic set of working Iowa farms, management decisions, like the seed mix richness and planting season, can have strong effects on important restoration goals like biodiversity. While higher richness seed mixes may be more expensive for landowners and farmers (Meissen et al. 2019), our results suggest that the added cost is worthwhile. Expanding the scope of ecological restoration to include in-field plantings will require continued monitoring and experimentation to further understand the drivers of variation in restoration outcomes in order to predict successful outcomes and to promote the goals of sustainable agriculture and biodiversity conservation.

References

- Asbjornsen, H., Hernandez-Santana, V., Liebman, M., Bayala, J., Chen, J., Helmers, M., Ong, C.K., Schulte, L.A., 2014. Targeting perennial vegetation in agricultural landscapes for enhancing ecosystem services. *Renew. Agric. Food Syst.* 29, 101–125. <https://doi.org/10.1017/S1742170512000385>
- Baer, S.G., Engle, D.M., Knops, J.M.H., Langeland, K.A., Maxwell, B.D., Menalled, F.D., Symstad, A.J., 2009. Vulnerability of rehabilitated agricultural production systems to invasion by nontarget plant species. *Environ. Manage.* 43, 189–196. <https://doi.org/10.1007/s00267-008-9167-6>
- Bahm, M.A., Barnes, T.G., Jensen, K.C., 2011. Herbicide and fire effects on Smooth Brome (*Bromus inermis*) and Kentucky Bluegrass (*Poa pratensis*) in invaded prairie remnants. *Invasive Plant Sci. Manag.* 4, 189–197. <https://doi.org/10.1614/IPSM-D-10-00046.1>
- Bailey, A.W., Poulton, C.E., 1968. Plant communities and environmental interrelationship in a portion of the Tillamook burn, Northwestern Oregon. *Ecology* 49, 1–13. <https://doi.org/10.2307/1933554>
- Bates, D., Mächler, M., Bolker, B., Walker, S., 2015. Fitting linear mixed-effects models using lme4. *J. Stat. Softw.* 67. <https://doi.org/10.18637/jss.v067.i01>
- Blumenthal, D.M., Jordan, N.R., Svenson, E.L., 2005. Effects of prairie restoration on weed invasions. *Agric. Ecosyst. Environ.* 107, 221–230. <https://doi.org/10.1016/j.agee.2004.11.008>
- Bolker, B.M., Brooks, M.E., Clark, C.J., Geange, S.W., Poulsen, J.R., Stevens, M.H.H., White, J.-S.S., 2009. Generalized linear mixed models: a practical guide for ecology and evolution. *Trends Ecol. Evol.* 24, 127–135. <https://doi.org/10.1016/j.tree.2008.10.008>
- Brudvig, L.A., 2017. Toward prediction in the restoration of biodiversity. *J. Appl. Ecol.* 54, 1013–1017. <https://doi.org/10.1111/1365-2664.12940>
- Brudvig, L.A., Barak, R.S., Bauer, J.T., Caughlin, T.T., Laughlin, D.C., Larios, L., Matthews, J.W., Stuble, K.L., Turley, N.E., Zirbel, C.R., 2017. Interpreting variation to advance predictive restoration science. *J. Appl. Ecol.* 54, 1018–1027. <https://doi.org/10.1111/1365-2664.12938>
- Camill, P., McKone, M.J., Sturges, S.T., Severud, W.J., Ellis, E., Limmer, J., Martin, C.B., Navratil, R.T., Purdie, A.J., Sandel, B.S., Talukder, S., Trout, A., 2004. Community- and ecosystem-level changes in a species-rich tallgrass prairie restoration. *Ecol. Appl.* 14, 1680–1694. <https://doi.org/10.1890/03-5273>

- Carter, D.L., Blair, J.M., 2012. High richness and dense seeding enhance grassland restoration establishment but have little effect on drought response. *Ecol. Appl.* 22, 1308–1319. <https://doi.org/10.1890/11-1970.1>
- Chase, J.M., 2003. Community assembly: when should history matter? *Oecologia* 136, 489–498. <https://doi.org/10.1007/s00442-003-1311-7>
- Connor, E.F., McCoy, E.D., 1979. The statistics and biology of the species-area relationship. *Am. Nat.* 113, 791–833. <https://doi.org/10.1086/283438>
- Daubenmire, R., 1959. A canopy-coverage method of vegetational analysis. *Northwest Sci.* 33, 43–64.
- Ditomaso, J.M., Brooks, M.L., Allen, E.B., Minnich, R., Rice, P.M., Kyser, G.B., 2006. Control of invasive weeds with prescribed burning. *Weed Technol.* 20, 535–548. <https://doi.org/10.1614/WT-05-086R1.1>
- Eilers, L.J., Roosa, D.M., 1994. *The vascular plants of Iowa*. University of Iowa Press.
- English, L., Liebman, M., Niemi, J., 2020. Improving the outcomes of prairie restorations within croplands database. <https://doi.org/10.25380/iastate.12287951.v1>
- Fargione, J.E., Tilman, D., 2005. Diversity decreases invasion via both sampling and complementarity effects. *Ecol. Lett.* 8, 604–611. <https://doi.org/10.1111/j.1461-0248.2005.00753.x>
- Fornara, D.A., Tilman, D., 2008. Plant functional composition influences rates of soil carbon and nitrogen accumulation. *J. Ecol.* 96, 314–322. <https://doi.org/10.1111/j.1365-2745.2007.01345.x>
- Grant, T.A., Flanders-Wanner, B., Shaffer, T.L., Murphy, R.K., Knutsen, G.A., 2009. An Emerging crisis across northern prairie refuges: prevalence of invasive plants and a plan for adaptive management. *Ecol. Restor.* 27, 58–65. <https://doi.org/10.3368/er.27.1.58>
- Grman, E., Suding, K.N., 2010. Within-year soil legacies contribute to strong priority effects of exotics on native California grassland communities. *Restor. Ecol.* 18, 664–670. <https://doi.org/10.1111/j.1526-100X.2008.00497.x>
- Grman, E., Bassett, T., Brudvig, L.A., 2013. Confronting contingency in restoration: management and site history determine outcomes of assembling prairies, but site characteristics and landscape context have little effect. *J. Appl. Ecol.* 50, 1234–1243. [https://doi.org/10.1111/1365-2664.12135@10.1111/\(ISSN\)1365-2664.ECOLOGICALRESTORATION](https://doi.org/10.1111/1365-2664.12135@10.1111/(ISSN)1365-2664.ECOLOGICALRESTORATION)

- Grman, E., Brudvig, L.A., 2014. Beta diversity among prairie restorations increases with species pool size, but not through enhanced species sorting. *J. Ecol.* 102, 1017–1024. <https://doi.org/10.1111/1365-2745.12267>
- Grman, E., Zirbel, C.R., Bassett, T., Brudvig, L.A., 2018. Ecosystem multifunctionality increases with beta diversity in restored prairies. *Oecologia* 188, 837–848. <https://doi.org/10.1007/s00442-018-4248-6>
- Grolemund, G., Wickham, H., 2011. Dates and times made easy with lubridate. *J. Stat. Softw.* <https://doi.org/10.18637/jss.v040.i03>
- Groves, A.M., Bauer, J.T., Brudvig, L.A., 2020. Lasting signature of planting year weather on restored grasslands. *Sci. Rep.* 10, 5953. <https://doi.org/10.1038/s41598-020-62123-7>
- Groves, A.M., Brudvig, L.A., 2019. Interannual variation in precipitation and other planting conditions impacts seedling establishment in sown plant communities. *Restor. Ecol.* 27, 128–137. <https://doi.org/10.1111/rec.12708>
- Harvey, C.A., Medina, A., Sánchez, D.M., Vílchez, S., Hernández, B., Saenz, J.C., Maes, J.M., Casanoves, F., Sinclair, F.L., 2006. Patterns of animal diversity in different forms of tree cover in agricultural landscapes. *Ecol. Appl.* 16, 1986–1999. [https://doi.org/10.1890/1051-0761\(2006\)016\[1986:POADID\]2.0.CO;2](https://doi.org/10.1890/1051-0761(2006)016[1986:POADID]2.0.CO;2)
- 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. <https://doi.org/10.2134/jeq2011.0473>
- Hirsh, S.M., Mabry, C.M., Schulte, L.A., Liebman, M., 2013. Diversifying agricultural catchments by incorporating tallgrass prairie buffer strips. *Ecol. Restor.* 31, 201–211. <https://doi.org/10.3368/er.31.2.201>
- Isbell, F., Gonzalez, A., Loreau, M., Cowles, J., Díaz, S., Hector, A., MacE, G.M., Wardle, D.A., O'Connor, M.I., Duffy, J.E., Turnbull, L.A., Thompson, P.L., Larigauderie, A., 2017. Linking the influence and dependence of people on biodiversity across scales. *Nature.* 546, 65–72. <https://doi.org/10.1038/nature22899>
- Ives, A.R., 2015. For testing the significance of regression coefficients, go ahead and log-transform count data. *Methods Ecol. Evol.* 6, 828–835. <https://doi.org/10.1111/2041-210X.12386>
- Jarchow, M.E., Liebman, M., 2013. Nitrogen fertilization increases diversity and productivity of prairie communities used for bioenergy. 5, 281–289. *GCB Bioenergy.* <https://doi.org/10.1111/j.1757-1707.2012.01186.x>

- Jones, C.S., Nielsen, J.K., Schilling, K.E., Weber, L.J., 2018. Iowa stream nitrate and the Gulf of Mexico. *PLoS One* 13, e0195930. <https://doi.org/10.1371/journal.pone.0195930>
- Jost, L., 2006. Entropy and diversity. *Oikos* 113, 363–375. <https://doi.org/10.1111/j.2006.0030-1299.14714.x>
- Jost, L., 2007. Partitioning diversity into independent alpha and beta components. *Ecology* 88, 2427–2439. <https://doi.org/10.1890/06-1736.1>
- Kaul, A.D., Wilsey, B.J. 2020. Exotic species drive patterns of plant species diversity in 93 restored tallgrass prairies. *Ecol. Appl.* in press.
- Kordbacheh, F., Jarchow, M., English, L., Liebman, M., 2019. Productivity and diversity of annually harvested reconstructed prairie communities. *J. Appl. Ecol.* 56, 330–342. <https://doi.org/10.1111/1365-2664.13267>
- Kremen, C., M’Gonigle, L.K., 2015. Small-scale restoration in intensive agricultural landscapes supports more specialized and less mobile pollinator species. *J. Appl. Ecol.* 52, 602–610. <https://doi.org/10.1111/1365-2664.12418>
- Kremen, C., Merenlender, A.M., 2018. Landscapes that work for biodiversity and people. *Science*. 362, eaau6020. <https://doi.org/10.1126/science.aau6020>
- Kurtz, C., 2013. *A practical guide to prairie reconstruction*, Second Edi. ed. University of Iowa Press, Iowa City.
- Kuznetsova, A., Brockhoff, P.B., Christensen, R.H.B., 2017. lmerTest package: tests in linear mixed effects models. *J. Stat. Softw.* 82. <https://doi.org/10.18637/jss.v082.i13>
- Larson, D.L., Bright, J.B., Drobney, P., Larson, J.L., Palaia, N., Rabie, P.A., Vacek, S., Wells, D., 2011. Effects of planting method and seed mix richness on the early stages of tallgrass prairie restoration. *Biol. Conserv.* 144, 3127–3139. <https://doi.org/10.1016/j.biocon.2011.10.018>
- Larson, D.L., Bright, J.B., Drobney, P., Larson, J.L., Palaia, N., Rabie, P.A., Vacek, S., Wells, D., 2013. Using prairie restoration to curtail invasion of Canada thistle: the importance of limiting similarity and seed mix richness. *Biol. Invasions* 15, 2049–2063. <https://doi.org/10.1007/s10530-013-0432-0>
- Ludecke, D., Makowski, D., Waggoner, P., 2020. performance: assessment of regression models performance. R package version 0.4.5. Cran R.
- Martin, L.M., Wilsey, B.J., 2012. Assembly history alters alpha and beta diversity, exotic-native proportions and functioning of restored prairie plant communities. *J. Appl. Ecol.* 49, 1436–1445. <https://doi.org/10.1111/j.1365-2664.2012.02202.x>

- Martin, L.M., Wilsey, B.J., 2014. Native-species seed additions do not shift restored prairie plant communities from exotic to native states. *Basic Appl. Ecol.* 15, 297–304. <https://doi.org/10.1016/j.baae.2014.05.007>
- Medina, A., Harvey, C.A., Merlo, D.S., Vílchez, S., Hernández, B., 2007. Bat diversity and movement in an agricultural landscape in Matiguás, Nicaragua. *Biotropica* 39, 120–128. <https://doi.org/10.1111/j.1744-7429.2006.00240.x>
- Meissen, J.C., Glidden, A.J., Sherrard, M.E., Elgersma, K.J., Jackson, L.L., 2019. Seed mix design and first year management influence multifunctionality and cost-effectiveness in prairie reconstruction. *Restor. Ecol.* rec.13013. <https://doi.org/10.1111/rec.13013>
- Middleton, E.L., Bever, J.D., Schultz, P.A., 2010. The effect of restoration methods on the quality of the restoration and resistance to invasion by exotics. *Restor. Ecol.* 18, 181–187. <https://doi.org/10.1111/j.1526-100X.2008.00501.x>
- Mori, A.S., Isbell, F., Seidl, R., 2018. β -Diversity, community assembly, and ecosystem functioning. *Trends Ecol. Evol.* 33, 549–564. <https://doi.org/10.1016/j.tree.2018.04.012>
- Mueller, K.E., Tilman, D., Fornara, D.A., Hobbie, S.E., 2013. Root depth distribution and the diversity-productivity relationship in a long-term grassland experiment. *Ecology*. <https://doi.org/10.1890/12-1399.1>
- Naeem, S., Knops, J.M.H., Tilman, D., Howe, K.M., Kennedy, T., Gale, S., 2000. Plant diversity increases resistance to invasion in the absence of covarying extrinsic factors. *Oikos* 91, 97–108. <https://doi.org/10.1034/j.1600-0706.2000.910108.x>
- Nakagawa, S., Schielzeth, H., 2013. A general and simple method for obtaining R^2 from generalized linear mixed-effects models. *Methods Ecol. Evol.* 4, 133–142. <https://doi.org/10.1111/j.2041-210x.2012.00261.x>
- Nemec, K.T., Allen, C.R., Helzer, C.J., Wedin, D.A., 2013. Influence of richness and seeding density on invasion resistance in experimental tallgrass prairie restorations. *Ecol. Restor.* 31, 168–185. <https://doi.org/10.3368/er.31.2.168>
- Norland, J.E., Dixon, C.S., Larson, D.L., Askerooth, K.L., Geaumont, B.A., 2018. Prairie reconstruction unpredictability and complexity: what is the rate of reconstruction failures? *Ecol. Restor.* 36, 263–266. <https://doi.org/10.3368/er.36.4.263>
- Oerke, E.C., 2006. Crop losses to pests. *J. Agric. Sci.* 144, 31–43. <https://doi.org/10.1017/S0021859605005708>
- Oksanen, J., Blanchet, F.G., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., Minchin, P.R., O'Hara, R.B., Simpson, G.L., Solymos, P., Stevens, M.H.H., Szoecs, E., Wagner, H., 2019. *vegan: Community ecology package*. R package version 2.5-2. Cran R.

- Phillips-Mao, L., Refsland, J.M., Galatowitsch, S.M., 2015. Cost-estimation for landscape-Scale restoration planning in the upper Midwest, U.S. *Ecol. Restor.* 33, 135–146. <https://doi.org/10.3368/er.33.2.135>
- Polley, H.W., Derner, J.D., Wilsey, B.J., 2005. Patterns of plant species diversity in remnant and restored tallgrass prairies. *Restor. Ecol.* 13, 480–487. <https://doi.org/10.1111/j.1526-100X.2005.00060.x>
- R Core Team, 2019. R: A language and environment for statistical computing. Vienna, Austria.
- Reich, P.B., Buschena, C., Tjoelker, M.G., Wrage, K., Knops, J., Tilman, D., Machado, J.L., 2003. Variation in growth rate and ecophysiology among 34 grassland and savanna species under contrasting N supply: a test of functional group differences. *New Phytol.* 157, 617–631. <https://doi.org/10.1046/j.1469-8137.2003.00703.x>
- Rey Benayas, J.M., Newton, A.C., Diaz, A., Bullock, J.M., 2009. Enhancement of biodiversity and ecosystem services by ecological restoration: A meta-analysis. *Science.* 325, 1121–1124. <https://doi.org/10.1126/science.1172460>
- Rosenzweig, M.L., 1995. Species diversity in space and time. Cambridge University Press.
- Rowe, H.I., 2010. Tricks of the trade: Techniques and opinions from 38 experts in tallgrass prairie restoration. *Restor. Ecol.* 18, 253–262. <https://doi.org/10.1111/j.1526-100X.2010.00663.x>
- Rowe, H.I., Fargione, J., Holland, J.D., 2013. Prairie restorations can protect remnant tallgrass prairie plant communities. *Am. Midl. Nat.* 170, 26–38. <https://doi.org/10.1674/0003-0031-170.1.26>
- Salesman, J.B., Thomsen, M., 2011. Smooth Brome (*Bromus inermis*) in tallgrass Prairies: A review of control methods and future research directions. *Ecol. Restor.* 29, 374–381. <https://doi.org/10.3368/er.29.4.374>
- Schramm, P., 1990. Prairie restoration: a twenty-five year perspective on establishment and management, in: Proceedings of the 12th North American Prairie Conference: Recapturing a Vanishing Heritage.
- Schulte, L., Niemi, J., Helmers, M., Liebman, M., Arbuckle, J., James, D., Kolka, R., O’Neal, M., Tomer, M., Tyndall, J., 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.* 201620229. <https://doi.org/10.1073/pnas.1620229114>
- Sebilo, M., Mayer, B., Nicolardot, B., Pinay, G., Mariotti, A., 2013. Long-term fate of nitrate fertilizer in agricultural soils. *Proc. Natl. Acad. Sci. U. S. A.* 110, 18185–18189. <https://doi.org/10.1073/pnas.1305372110>

- Simberloff, D., Souza, L., Nuñez, M.A., Barrios-Garcia, M.N., Bunn, W., 2012. The natives are restless, but not often and mostly when disturbed. *Ecology* 93, 598–607. <https://doi.org/10.1890/11-1232.1>
- Sluis, W.J., 2002. Patterns of species richness and composition in re-created grassland. *Restor. Ecol.* 10, 677–684. <https://doi.org/10.1046/j.1526-100X.2002.01048.x>
- Smith, D., 1998. Iowa prairie: original extent and loss, preservation and recovery Attempts. *J. Iowa Acad. Sci.* 105, 94–108.
- Stuble, K.L., Fick, S.E., Young, T.P., 2017. Every restoration is unique: testing year effects and site effects as drivers of initial restoration trajectories. *J. Appl. Ecol.* 54, 1051–1057. <https://doi.org/10.1111/1365-2664.12861>
- Suding, K.N., 2011. Toward an era of restoration in ecology: successes, failures, and opportunities ahead. *Annu. Rev. Ecol. Evol. Syst.* 42, 465–487. <https://doi.org/10.1146/annurev-ecolsys-102710-145115>
- The University of Iowa Libraries, 2020. Iowa prairie plants. Digit. Press Ed. <http://uipress.lib.uiowa.edu/ppi/index.php> (Accessed 1 March 2020)
- Tix, D., Charvat, I., 2005. Aboveground biomass removal by burning and raking increases diversity in a reconstructed prairie. *Restor. Ecol.* 13, 20–28. <https://doi.org/10.1111/j.1526-100X.2005.00004.x>
- USDA, NRCS, 2002. Iowa state-listed noxious weeds. <https://plants.usda.gov/java/noxious?rptType=State&statefips=19> (accessed 11 March 2020).
- Warton, D.I., Hui, F.K.C., 2011. The arcsine is asinine: the analysis of proportions in ecology. *Ecology* 92, 3–10. <https://doi.org/10.1890/10-0340.1>
- Warton, D.I., Lyons, M., Stoklosa, J., Ives, A.R., 2016. Three points to consider when choosing a LM or GLM test for count data. *Methods Ecol. Evol.* 7, 882–890. <https://doi.org/10.1111/2041-210X.12552>
- Wilsey, B.J., 2020. Restoration in the face of changing climate: importance of persistence, priority effects, and species diversity. *Restor. Ecol.* rec.13132. <https://doi.org/10.1111/rec.13132>
- Wilsey, B.J., Barber, K., Martin, L.M., 2015. Exotic grassland species have stronger priority effects than natives regardless of whether they are cultivated or wild genotypes. *New Phytol.* 205, 928–937. <https://doi.org/10.1111/nph.13028>

- Young, T.P., Chase, J.M., Huddleston, R.T., 2001. Community succession and assembly: comparing, contrasting and combining paradigms in the context of ecological restoration. *Ecol. Restor.* <https://doi.org/10.3368/er.19.1.5>
- Zavaleta, E.S., Pasari, J.R., Hulvey, K.B., Tilman, G.D., 2010. Sustaining multiple ecosystem functions in grassland communities requires higher biodiversity. *Proc. Natl. Acad. Sci. U. S. A.* 107, 1443–1446. <https://doi.org/10.1073/pnas.0906829107>
- 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>
- Zirbel, C.R., Bassett, T., Grman, E., Brudvig, L.A., 2017. Plant functional traits and environmental conditions shape community assembly and ecosystem functioning during restoration. *J. Appl. Ecol.* 54, 1070–1079. <https://doi.org/10.1111/1365-2664.12885>

Figures and Tables

ID	Sampling year	Seed mix richness	Age (years)	Size (ha)	Perimeter: Area ratio	Season planted
1	2018 & 2019	28	4.2	10.11	0.07	Fall
2	2018 & 2019	31	4.7	1.62	0.19	Spring
3	2018 & 2019	36	5.6	2.65	0.24	Spring
4	2018 & 2019	NA	5.1	0.76	0.27	Fall
5	2018 & 2019	25	4.8	0.90	0.23	Spring
6	2018 & 2019	42	3.1	3.92	0.12	Fall
7	2018 & 2019	42	3.9	2.40	0.23	Fall
8	2018 & 2019	29	7.6	1.69	0.18	Summer
9	2018 & 2019	17	7.8	1.73	0.23	Spring
10	2018 & 2019	NA	5.6	0.37	0.14	Summer
11	2018 & 2019	38	4.8	0.81	0.26	Spring
12	2019	54	4	5.36	0.17	Fall
13	2019	40	4.8	0.82	0.33	Spring
14	2018 & 2019	41	5.1	1.40	0.20	Fall
15	2019	40	4.6	0.76	0.34	Summer
16	2018 & 2019	40	5.1	6.50	0.09	Fall
17	2018 & 2019	34	4.6	0.57	0.09	Summer
18	2018 & 2019	40	3.7	0.68	0.12	Spring
19	2018 & 2019	27	5.7	0.51	0.34	Spring
20	2018 & 2019	40	3.5	1.24	0.17	Summer
21	2019	40	5.1	0.83	0.19	Fall
22	2018 & 2019	32	5.6	0.17	1.57	Summer
23	2018 & 2019	15	5.5	0.32	0.09	Summer
24	2018 & 2019	26	3.6	4.23	0.23	Spring
25	2019	45	5.5	0.83	0.34	Summer

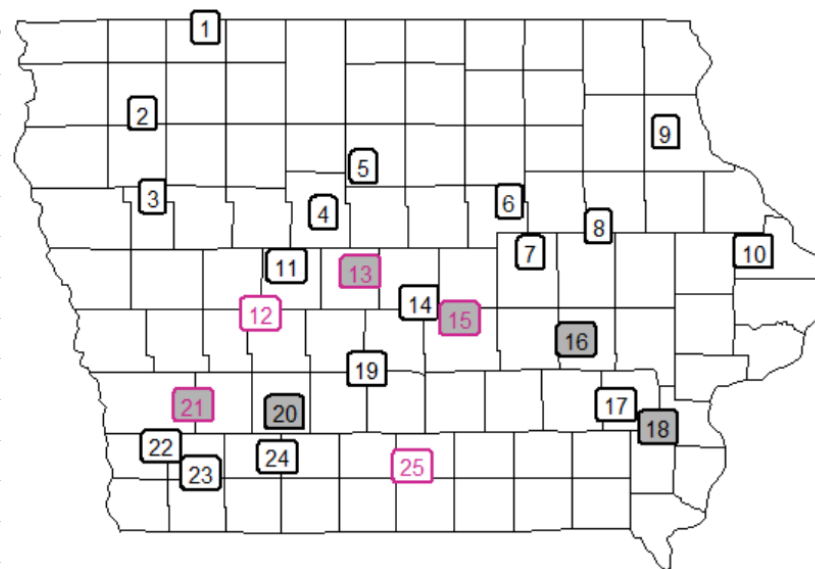


Figure 1. Map of site locations and corresponding site information. Pink labels on the map indicate sites that were only sampled in the summer of 2019, while gray-filled labels indicate sites that were seeded with the same seed mix. Seed mix richness indicates the number of species seeded at a site. Age was calculated as the difference between planting date and 01/01/2020. If a site had multiple strips the perimeter: area ratio was averaged across all strips. Fall planting includes winter seeding done through the end of January.

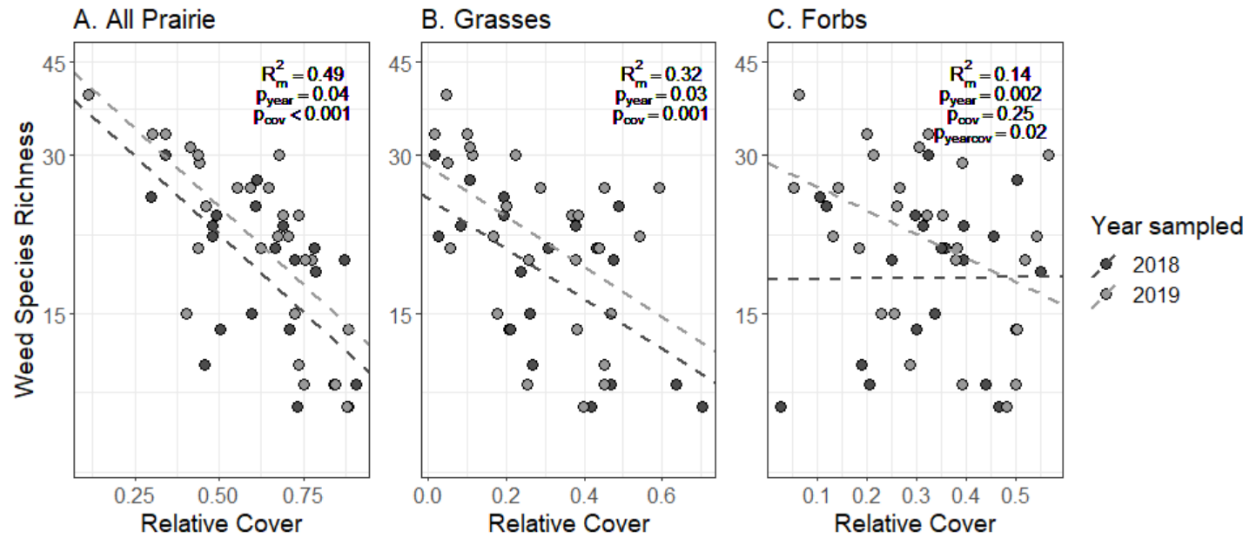


Figure 2. Relationship between the relative cover of all prairie species (A), prairie grasses (B), and prairie forbs (C) and the richness of the weedy community. Data are shown on a log-axis. Points are colored by sampling year. Site was included as a random effect to account for variation due to repeated sampling. R_m^2 indicates the variation explain by the fixed effects (sampling year and relative cover). P-values for year and relative cover (cov) were significant for all prairie species and grasses. The interaction between year and relative cover was significant for prairie forbs only.

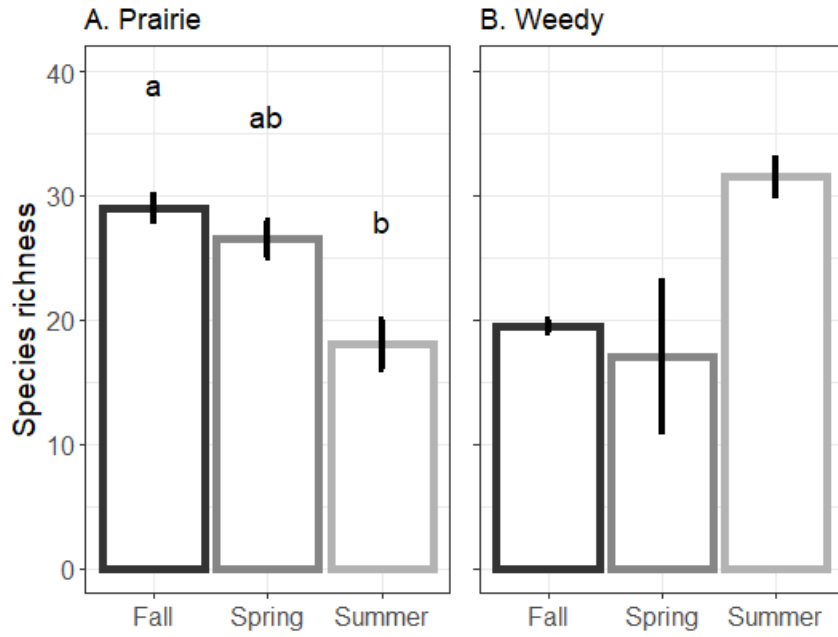


Figure 3. Prairie (A) and weedy (B) species richness as a function of the season in which a site was seeded. Data are a subset ($n = 6$) of surveyed sites that used the same seed mix and were all sampled in the summer of 2019. Standard errors are shown for each mean value and letters indicate statistically significant differences after Tukey adjustment.

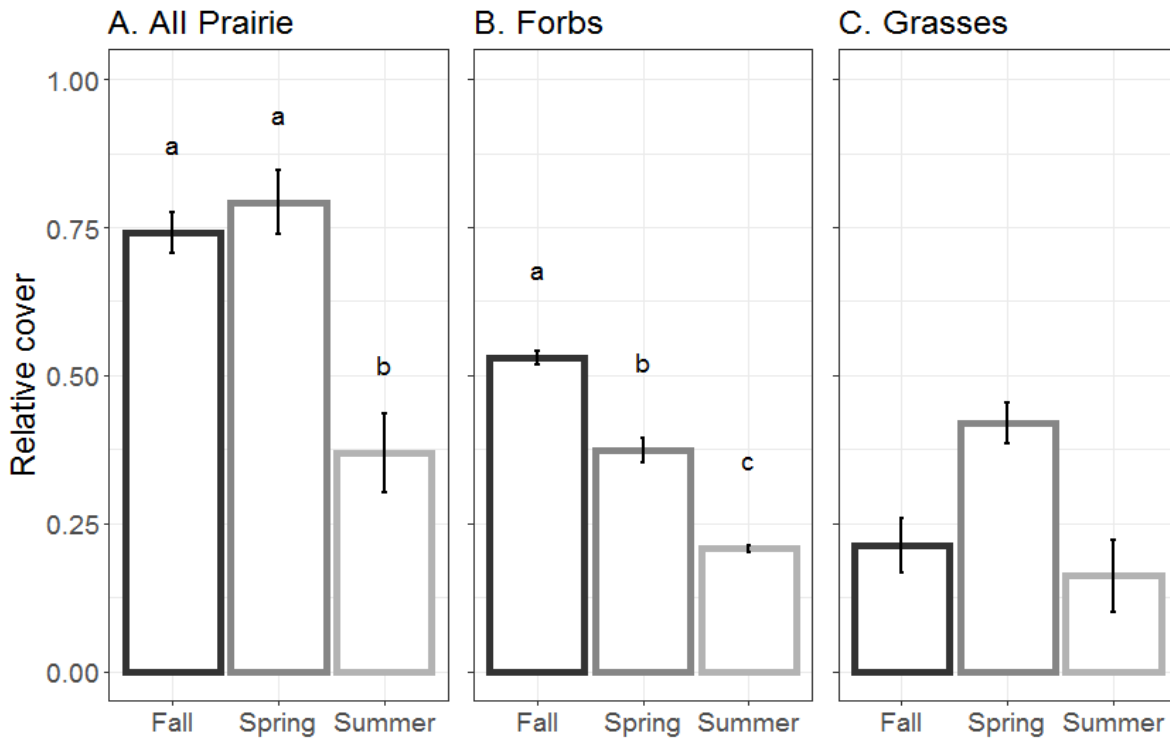


Figure 4. The relative cover of all prairie species (A), forbs (B), and grasses (C) as a function of the season in which a site was seeded. Data are a subset ($n = 6$) of surveyed sites that used the same seed mix and were all sampled in the summer of 2019. Standard errors are shown for each mean value and letters indicate statistically significant differences after Tukey adjustment. Data were logit transformed for statistical testing.

Table 1. Models of gamma- (γ site-level), beta- (β), and alpha (α local)-diversity. Fixed effects model estimates, 95% Confidence Intervals (CI) and p-values are reported. Non-significant covariates that were dropped from the final models are indicated by (—). P-values for random effects are generated from likelihood ratio tests. Marginal R^2 indicates the variation explained by the fixed effects listed and conditional R^2 indicates the variation explained by the fixed and random effects.

	γ			β			α		
	<i>Est.</i>	<i>95% CI</i>	<i>P</i>	<i>Est.</i>	<i>95% CI</i>	<i>P</i>	<i>Est.</i>	<i>95% CI</i>	<i>P</i>
FIXED EFFECTS									
Sampling year¹	1.38	-0.03, 2.85	0.07	0.14	-0.05, 0.33	0.16	0.12	-0.13, 0.37	0.35
Seed mix richness	0.37	0.19, 0.55	<0.001	0.05	0.02, 0.07	0.003	-0.01	-0.04, 0.03	0.64
Site age	—	—	—	—	—	—	-0.63	-0.90, -0.35	<0.001
Site size (ha)²	2.59	0.94, 4.23	0.007	—	—	—	0.71	0.42, 0.99	<0.001
Avg perimeter: area ratio²	—	—	—	—	—	—	—	—	—
Season planted	—	—	—	—	—	—	—	—	—
RANDOM EFFECTS									
Site			<0.001			<0.001			<0.001
Quadrat: Site	—	—	—	—	—	—			<0.001
R^2									
Marginal	0.56			0.31			0.2		
Conditional	0.88			0.85			0.45		

¹ Sampling year is a categorical variable with two levels (2018 & 2019). Therefore, estimates reported are the effect of 2019 relative to 2018.

² Size and perimeter: area ratio were log-transformed in the model. Estimates are the log transformed. In text estimates are calculated by taking $\beta_1 * \log(2)$.

Table 2. Models of prairie and weedy species richness at the site (γ) and quadrat (α) level. Fixed effects model estimates, 95% confidence intervals (CI) and p-values are reported. Non-significant covariates that were dropped from the final models are indicated by (—). P-values for random effects are generated from likelihood ratio tests. Marginal R^2 indicates the variation explained by the fixed effects listed and conditional R^2 indicates the variation explained by the fixed and random effects.

	PRAIRIE SPECIES						WEEDY SPECIES					
	γ^1			α			γ^1			α		
	<i>Est.</i>	<i>95% CI</i>	<i>P</i>	<i>Est.</i>	<i>95% CI</i>	<i>P</i>	<i>Est.</i>	<i>95% CI</i>	<i>P</i>	<i>Est.</i>	<i>95% CI</i>	<i>P</i>
FIXED EFFECTS												
Sampling year²	-0.05	-0.12, 0.03	0.22	-0.38	-0.62, -0.15	0.002	0.15	0.01, 0.29	0.04	0.10	-0.12, 0.31	0.39
Seed mix richness	0.02	0.01, 0.02	<0.001	0.002	-0.07, 0.08	0.96	0.01	-0.002, 0.03	0.10	0.05	-0.02, 0.12	0.16
Site age	—	—	—	—	—	—	—	—	—	—	—	—
Site size (ha)³	0.08	0.01, 0.15	0.05	—	—	—	—	—	—	—	—	—
Avg perimeter: area ratio³	—	—	—	—	—	—	—	—	—	—	—	—
Season planted (summer)⁴	—	—	—	-2.29	-3.91, -0.68	0.04	—	—	—	—	—	—
RANDOM EFFECTS												
Site			0.01			<0.001			0.01			<0.001
Quadrat: Site	—	—	—			<0.001	—	—	—			<0.001
R ²												
Marginal	0.50			0.09			0.16			0.04		
Conditional	0.82			0.66			0.65			0.57		

¹ Richness was log transformed. Shown are estimates from the log-transformed data.

² Sampling year is a categorical variable with two levels (2018 & 2019). Therefore, estimates reported are the effect of 2019 relative to 2018.

³ Size and perimeter: area ratio were log-transformed in the model. Estimates are the log transformed. In text estimates are calculated by taking $\beta_1 \cdot \log(2)$.

⁴ Season planted in a categorical variable with three levels (fall, spring, summer). Shown are the estimates of the summer treatment compared with the fall. P-value is Tukey adjusted.

Table 3. Models of the relative cover of leguminous forbs, C₃ grasses, and annual weeds. Fixed effect slope estimates (Est.), 95% confidence intervals (CI), and p-values (P) are reported. Non-significant covariates that were dropped from the final models are indicated by (—). P-values for random effects are generated from likelihood ratio tests. Marginal R² indicates the variation explained by the fixed effects listed and conditional R² indicates the variation explained by the fixed and random effects. Non-significant functional group information can be found in Table B1.

	Legumes ¹			C ₃ Grasses ⁷			Annual Weeds ⁷		
	<i>Est.</i>	<i>95% CI</i>	<i>P</i>	<i>Est.</i>	<i>95% CI</i>	<i>P</i>	<i>Est.</i>	<i>95% CI</i>	<i>P</i>
FIXED EFFECTS									
Sampling year²	-0.71	-1.61, 0.135	0.11	0.33	-0.49, 1.12	0.41	0.09	-0.40, 0.62	0.72
Seed mix richness	0.01	-0.07, 0.09	0.73	0.02	-0.04, 0.08	0.52	0.04	-0.01, 0.08	0.13
Site age	-1.05	-1.65, -0.44	0.006	—	—	—	-0.36	-0.67, -0.05	0.04
Site size (ha)³	0.65	0.02, 1.28	0.07	0.88	0.24, 1.52	0.02	—	—	—
Avg perimeter: area ratio³	—	—	—	1.49	0.53, 2.46	0.01	—	—	—
Season planted	—	—	—	—	—	—	—	—	—
RANDOM EFFECTS									
Site			0.09			0.14			0.24
R ²									
Marginal	0.46			0.30			0.32		
Conditional	0.71			0.56			0.53		

¹ Relative cover values were logit transformed for analysis.

² Sampling year is a categorical variable with two levels (2018 & 2019). Therefore, estimates reported are the effect of 2019 relative to 2018.

³ Size and perimeter: area ratio were log-transformed in the model. Estimates are the log transformed. In text estimates are calculated by taking $\beta_1 * \log(2)$.

CHAPTER 3. GENERAL CONCLUSION

Restoring native habitat within agricultural fields is a promising pathway to simultaneously address environmental issues like water quality while supporting biodiversity. In the Midwestern, USA “prairie strips” is a farming conservation practice that integrates native tallgrass prairie into working crop fields. Previous work has demonstrated that this practice can increase the provisioning of multiple ecosystem services such as soil retention and maintenance of pollinator habitat in agricultural landscapes (Schulte et al. 2017). Prairie strips recently became a codified practice under the Conservation Reserve Program (CRP) (USDA Farm Service Agency 2019), and therefore are likely to be more widely implemented across the Midwestern United States. To deliver multiple ecosystem services and to resist invasion pressure by non-target weeds these plantings should be diverse and consistently covered by native plants. Therefore, the success of this conservation practice hinges on our ability to reliably achieve these restoration outcomes across a wide variety of farming contexts.

There have been recent calls to transform restoration ecology from an ad-hoc science to a more predictive framework (Suding 2011, Brudvig 2017). Doing so requires that we improve our ability to explain variation amongst restoration sites (Brudvig 2017). Here we performed a comprehensive survey of farms that have integrated prairie strips into their farming operation to understand what factors were important determinants of two restoration outcomes: biodiversity and native cover. Given that restorations, and ecosystems in general, are inherently complex, there are a myriad of factors that could explain variation across sites. It was impossible for us to examine all the variables that differed among sites, so instead we chose to focus our analysis on the seed mix. The seed mix is estimated to be the largest economic investment a landowner makes in a prairie restoration (Phillips-Mao et al. 2015), and other work has demonstrated that

seed mix richness is positively associated with target species richness (Grman et al. 2013), beta-diversity (Grman and Brudvig 2014), and native cover (Carter and Blair 2012).

We found biodiversity and target species richness were, in general, more predictable outcomes than measures of native cover. The richness of the seed mix was positively associated with the gamma- and beta-diversity across our sites. Like other work, we found that diversity declined with site age (Sluis 2002, Middleton et al. 2010), but only at the local (alpha) level. Few of our explanatory variables explained the abundance of native species or different functional groups, and while we saw that higher native cover decreased weed species richness, like other work we did not see a direct link between seed mix richness and resistance to invasion (Larson et al. 2011).

We found that planting season affected the abundance of prairie species, especially forbs, but that signal was most prominent after we controlled for the effects of the seed mix by examining six sites that were seeded with the same mix. While our work demonstrated that certain management decisions can affect restoration outcomes across a wide variety of conditions, it also highlighted the importance of balancing observational studies with controlled experiments that can tease apart factors that may impact restoration outcomes but not produce a strong enough signal to overcome the variation across a large number of sites.

Our work focused primarily on how deterministic factors, specifically land manager decisions, were associated with restoration outcomes. Other recent work has highlighted the lasting effect that stochastic factors like first year precipitation have on the establishment of non-target species (Groves et al. 2020). While we believe that land manager decisions will be important determinants of prairie restoration outcomes, there will likely always be an inherent level of unpredictability due to stochastic factors like weather. Given this information, adoptees

of prairie strips should strive to utilize seed mixes that contain species that reliably establish across a wide variety of environmental conditions to ensure consistent target species cover even amidst varying weather patterns (APPENDIX C).

Future work should continue to parse out the importance of stochastic factors in determining community assembly of prairie strips, especially with regards to the establishment and persistence of non-target vegetation. Furthermore, understanding the nature of the weed propagule pressure at the time of planting may be essential to predicting the abundance and persistence of non-target vegetation. Efforts should continue to optimize seed mix design for these functional restorations, taking into account costs and species-specific establishment (APPENDIX C).

References

- Brudvig, L.A., 2017. Toward prediction in the restoration of biodiversity. *J. Appl. Ecol.* 54, 1013–1017. <https://doi.org/10.1111/1365-2664.12940>
- Carter, D.L., Blair, J.M., 2012. High richness and dense seeding enhance grassland restoration establishment but have little effect on drought response. *Ecol. Appl.* 22, 1308–1319. <https://doi.org/10.1890/11-1970.1>
- Grman, E., Bassett, T., Brudvig, L.A., 2013. Confronting contingency in restoration: management and site history determine outcomes of assembling prairies, but site characteristics and landscape context have little effect. *J. Appl. Ecol.* 50, 1234–1243. [https://doi.org/10.1111/1365-2664.12135@10.1111/\(ISSN\)1365-2664.ECOLOGICALRESTORATION](https://doi.org/10.1111/1365-2664.12135@10.1111/(ISSN)1365-2664.ECOLOGICALRESTORATION)
- Grman, E., Brudvig, L.A., 2014. Beta diversity among prairie restorations increases with species pool size, but not through enhanced species sorting. *J. Ecol.* 102, 1017–1024. <https://doi.org/10.1111/1365-2745.12267>
- Groves, A.M., Bauer, J.T., Brudvig, L.A., 2020. Lasting signature of planting year weather on restored grasslands. *Sci. Rep.* 10, 5953. <https://doi.org/10.1038/s41598-020-62123-7>
- Larson, D.L., Bright, J.B., Drobney, P., Larson, J.L., Palaia, N., Rabie, P.A., Vacek, S., Wells, D., 2011. Effects of planting method and seed mix richness on the early stages of tallgrass prairie restoration. *Biol. Conserv.* <https://doi.org/10.1016/j.biocon.2011.10.018>

- Middleton, E.L., Bever, J.D., Schultz, P.A., 2010. The effect of restoration methods on the quality of the restoration and resistance to invasion by exotics. *Restor. Ecol.* 18, 181–187. <https://doi.org/10.1111/j.1526-100X.2008.00501.x>
- Phillips-Mao, L., Refsland, J.M., Galatowitsch, S.M., 2015. Cost-estimation for landscape-scale restoration planning in the Upper Midwest, U.S. *Ecol. Restor.* 33, 135–146. <https://doi.org/10.3368/er.33.2.135>
- 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>
- Sluis, W.J., 2002. Patterns of species richness and composition in re-created grassland. *Restor. Ecol.* 10, 677–684. <https://doi.org/10.1046/j.1526-100X.2002.01048.x>
- Suding, K.N., 2011. Toward an era of restoration in ecology: successes, failures, and opportunities ahead. *Annu. Rev. Ecol. Evol. Syst.* 42, 465–487. <https://doi.org/10.1146/annurev-ecolsys-102710-145115>
- USDA Farm Service Agency, 2019. Fact sheet: Clean lakes, estuaries and rivers (CLEAR) initiative. Prairie Strip Practice (CP-43). https://www.fsa.usda.gov/Assets/USDA-FSA-Public/usdfiles/FactSheets/2019/crp_clear_initiative_prairie_strip_practice-fact_sheet.pdf (Accessed 6 May 2020).

APPENDIX A. SPECIES LIST

Table A1. Complete list of species found across all sites and all years. Functional group indicates prairie vs weedy classification.

Scientific Name	Common Name	Family	Native	Life Cycle	Functional Group	ID
<i>Abutilon theophrasti</i> Medik.	Velvetleaf	Malvaceae	Introduced	Annual	Weedy Forb	ABUTH
<i>Acalypha rhomboidei</i> Raf.	Rhomboid mercury	Euphorbiaceae	Native	Annual	Weedy Forb	ACARH
<i>Acer rubrum</i> L.	Red maple	Aceraceae	Native	Perennial	Woody	ACERU
<i>Achillea millefolium</i> L.	Western yarrow	Asteraceae	Native	Perennial	Prairie Forb	ACHMI
<i>Acroptilon repens</i> (L.) DC.	Russian knapweed	Asteraceae	Introduced	Perennial	Weedy Forb	ACRRE
<i>Agastache foeniculum</i> (Pursh) Kuntze	Anise hyssop	Lamiaceae	Native	Perennial	Prairie Forb	AGAFO
<i>Agrostis gigantea</i> Roth	Redtop	Poaceae	Introduced	Perennial	Weedy C3 Grass	AGRGI
<i>Alisma subcordatum</i> Raf.	American water plantain	Alismataceae	Native	Perennial	Wetland Forb	ALISU
<i>Alliaria petiolate</i> (Bieb.) Cavara & Grande	Garlic mustard	Brassicaceae	Introduced	Biennial	Weedy Forb	ALLPE
<i>Amaranthus retroflexus</i> L.	Redroot pigweed	Amaranthaceae	Native	Annual	Weedy Forb	AMARE
<i>Amaranthus tuberculatus</i> (Moq.) Sauer	Tall waterhemp	Amaranthaceae	Native	Annual	Weedy Forb	AMATU
<i>Ambrosia artemisiifolia</i> L.	Common ragweed	Asteraceae	Native	Annual	Weedy Forb	AMBAR
<i>Ambrosia psilostachya</i> DC.	Western ragweed	Asteraceae	Native	Perennial	Prairie Forb	AMBPS
<i>Ambrosia trifida</i> L.	Giant ragweed	Asteraceae	Native	Annual	Weedy Forb	AMBTR
<i>Amorpha canescens</i> Pursh	Lead plant	Fabaceae	Native	Perennial	Prairie Forb	AMOCA
<i>Andropogon gerardii</i> Vitman	Big bluestem	Poaceae	Native	Perennial	Prairie C4 Grass	ANDGE
<i>Anemone virginiana</i> L.	Tall thimbleweed	Ranunculaceae	Native	Perennial	Prairie Forb	ANEVI
<i>Apocynum cannabinum</i> L.	Common dogbane	Apocynaceae	Native	Perennial	Prairie Forb	APOCA
<i>Aquilegia canadensis</i> L.	Red columbine	Ranunculaceae	Native	Perennial	Prairie Forb	AQUCA
<i>Artemisia ludoviciana</i> Nutt.	White prairie sage	Asteraceae	Native	Perennial	Prairie Forb	ARTLU
<i>Asclepias incarnata</i> L.	Swamp milkweed	Asclepiadaceae	Native	Perennial	Prairie Forb	ASCIN
<i>Asclepias syriaca</i> L.	Common milkweed	Asclepiadaceae	Native	Perennial	Prairie Forb	ASCSY
<i>Asclepias tuberosa</i> L.	Butterfly milkweed	Asclepiadaceae	Native	Perennial	Prairie Forb	ASCTU

Table A1. (continued)

Scientific Name	Common Name	Family	Native	Life Cycle	Functional Group	ID
<i>Asclepias verticillate</i> L.	Whorled milkweed	Asclepiadaceae	Native	Perennial	Prairie Forb	ASCVE
<i>Astragalus canadensis</i> L.	Canadian milk vetch	Fabaceae	Native	Perennial	Prairie Forb	ASTCA
<i>Avena sativa</i> L.	Oat	Poaceae	Introduced	Annual	Weedy C3 Grass	AVESA
<i>Baptisia alba</i> (L.) Vent.	Wild white indigo	Fabaceae	Native	Perennial	Prairie Forb	BAPAL
<i>Baptisia bracteata</i> Muhl. ex Ell.	Cream false indigo	Fabaceae	Native	Perennial	Prairie Forb	BAPBR
<i>Barbarea vulgaris</i> Ait. f.	Wild rocket	Brassicaceae	Introduced	Biennial	Weedy Forb	BARVU
<i>Bidens frondosa</i> L.	Common beggarticks	Asteraceae	Native	Annual	Weedy Forb	BIDFR
<i>Bidens vulgata</i> Greene	Tall beggarticks	Asteraceae	Native	Annual	Weedy Forb	BIDVU
<i>Bouteloua curtipendula</i> (Michx.) Torrey	Side oats grama	Poaceae	Native	Perennial	Prairie C4 Grass	BOUCU
<i>Bromus inermis</i> Leyss.	Smooth brome	Poaceae	Introduced	Perennial	Weedy C3 Grass	BROIN
<i>Bromus japonicus</i> Thunb. ex Murr.	Field brome	Poaceae	Introduced	Perennial	Weedy C3 Grass	BROJA
<i>Bromus kalmii</i> Gray	Kalm's brome	Poaceae	Native	Perennial	Prairie C3 Grass	BROKA
<i>Bromus tectorum</i> L.	Downy brome	Poaceae	Introduced	Annual	Weedy C3 Grass	BROTE
<i>Calystegia sepium</i> (L.) R. Br.	Hedge bindweed	Convolvulaceae	Native	Perennial	Weedy Forb	CALSE
<i>Capsella bursa-pastoris</i> (L.) Medik.	Shepherd's-purse	Brassicaceae	Introduced	Annual	Weedy Forb	CAPBU
<i>Carduus nutans</i> L.	Musk thistle	Asteraceae	Native	Biennial	Weedy Forb	CARNU
<i>Carex annectens</i> (E.P. Bicknell) E.P. Bicknell	Yellow fox sedge	Cyperaceae	Native	Perennial	Prairie Sedge	CARAN
<i>Carex tribuloides</i> Wahlenb.	Blunt broom sedge	Cyperaceae	Native	Perennial	Prairie Sedge	CARTR
<i>Carex vulpinoidea</i> Michx.	Brown fox sedge	Cyperaceae	Native	Perennial	Prairie Sedge	CARVU
<i>Cerastium fontanum</i> ssp. <i>vulgare</i> (Hartm.) Greuter & Burdet	Mouse-ear chickweed	Caryophyllaceae	Introduced	Perennial	Weedy Forb	CERVU
<i>Chamaecrista fasciculata</i> (Michx.) Greene	Partridge pea	Fabaceae	Native	Annual	Prairie Forb	CHAFA
<i>Chenopodium album</i> L.	Lambsquarters	Chenopodiaceae	Introduced	Annual	Weedy Forb	CHEAL

Table A1. (continued)

Scientific Name	Common Name	Family	Native	Life Cycle	Functional Group	ID
<i>Cichorium intybus</i> L.	Chicory	Asteraceae	Introduced	Biennial	Weedy Forb	CICIN
<i>Cirsium altissimum</i> (L.) Spreng.	Tall thistle	Asteraceae	Native	Biennial	Prairie Forb	CIRAL
<i>Cirsium arvense</i> (L.) Scop.	Canada thistle	Asteraceae	Introduced	Perennial	Weedy Forb	CIRAR
<i>Cirsium discolor</i> (Muhl. Ex. Willd.) Spreng.	Pasture thistle	Asteraceae	Native	Biennial	Prairie Forb	CIRDI
<i>Cirsium vulgare</i> (Savi) Ten.	Bull thistle	Asteraceae	Introduced	Biennial	Weedy Forb	CIRVU
<i>Commelina communis</i> L.	Asiatic dayflower	Commelinaceae	Introduced	Annual	Weedy Forb	COMCO
<i>Convolvulus arvensis</i> L.	Field bindweed	Convolvulaceae	Introduced	Perennial	Weedy Forb	CONAR
<i>Conyza canadensis</i> (L.) Cronq.	Mare's tail	Asteraceae	Native	Annual	Weedy Forb	CONCA
<i>Coreopsis palmata</i> Nutt.	Prairie coreopsis	Asteraceae	Native	Perennial	Prairie Forb	CORPA
<i>Coreopsis tinctoria</i> Nutt.	Plains coreopsis	Asteraceae	Native	Perennial	Prairie Forb	CORTI
<i>Cyperus esculentus</i> L.	Yellow nutsedge	Cyperaceae	Native	Perennial	Weedy Sedge	CYPES
<i>Dactylis glomerata</i> L.	Orchard grass	Poaceae	Introduced	Perennial	Weedy C3 Grass	DACGL
<i>Dalea candida</i> Willd.	White prairie clover	Fabaceae	Native	Perennial	Prairie Forb	DALCA
<i>Dalea purpurea</i> Vent.	Purple prairie clover	Fabaceae	Native	Perennial	Prairie Forb	DALPU
<i>Daucus carota</i> L.	Queen Anne's lace	Apiaceae	Introduced	Biennial	Weedy Forb	DAUCA
<i>Desmanthus illinoensis</i> (Michx.) MacM.	Prairie mimosa	Fabaceae	Native	Perennial	Prairie Forb	DESIL
<i>Desmodium canadense</i> (L.) DC.	Showy tick trefoil	Fabaceae	Native	Perennial	Prairie Forb	DESCA
<i>Digitaria sanguinalis</i> (L.) Scop.	Large crabgrass	Poaceae	Introduced	Annual	Weedy C4 Grass	DIGSA
<i>Dracopis amplexicaulis</i> (Vahl) Cass.	Clasping coneflower	Asteraceae	Native	Annual	Prairie Forb	DRAAM
<i>Echinacea pallida</i> Nutt.	Pale purple coneflower	Asteraceae	Native	Perennial	Prairie Forb	ECHPA
<i>Echinacea purpurea</i> (L.) Moench.	Purple coneflower	Asteraceae	Native	Perennial	Prairie Forb	ECHPU
<i>Echinochloa crus-galli</i> (L.) Beauv.	Barnyardgrass	Poaceae	Introduced	Annual	Weedy C4 Grass	ECHCR

Table A1. (continued)

Scientific Name	Common Name	Family	Native	Life Cycle	Functional Group	ID
<i>Eleocharis palustris</i> (L.) Roem. & Schult.	Common spikerush	Cyperaceae	Native	Perennial	Rush	ELEPA
<i>Eleusine indica</i> (L.) Gaertn.	Goosegrass	Poaceae	Introduced	Annual	Weedy C4 Grass	ELEIN
<i>Elymus canadensis</i> L.	Canada wildrye	Poaceae	Native	Perennial	Prairie C3 Grass	ELYCA
<i>Elymus repens</i> (L.) Gould	Quackgrass	Poaceae	Introduced	Perennial	Weedy C3 Grass	ELYRE
<i>Elymus trachycaulus</i> (Link) Gould ex Shinnars	Slender wheatgrass	Poaceae	Native	Perennial	Prairie C3 Grass	ELYTR
<i>Elymus virginicus</i> L.	Virginia wildrye	Poaceae	Native	Perennial	Prairie C3 Grass	ELYVI
<i>Equistem arvense</i> L.	Common horsetail	Equisetaceae	Native	Perennial	Fern	EQUAR
<i>Erigeron annuus</i> (L.) Pers.	Annual fleabane	Asteraceae	Native	Annual	Weedy Forb	ERIAN
<i>Erigeron strigosus</i> Muhl. Ex Willd.	Daisy fleabane	Asteraceae	Native	Annual	Prairie Forb	ERIST
<i>Eriochloa villosa</i> (Thunb.) Kunth	Wooly cupgrass	Poaceae	Introduced	Annual	Weedy C3 Grass	ERIVI
<i>Eryngium yuccifolium</i> Michx.	Rattlesnake master	Apiaceae	Native	Perennial	Prairie Forb	ERYYU
<i>Eupatorium serotinum</i> Michx.	Late boneset	Asteraceae	Native	Perennial	Prairie Forb	EUPSE
<i>Euphorbia collorata</i> L.	Flowering spurge	Euphorbiaceae	Native	Perennial	Prairie Forb	EUPCO
<i>Euphorbia dentata</i> Michx.	Toothed spurge	Euphorbiaceae	Native	Annual	Prairie Forb	EUPDE
<i>Euphorbia maculate</i> L.	Spotted spurge	Euphorbiaceae	Native	Annual	Weedy Forb	EUPMA
<i>Euphorbia nutans</i> Lag.	Nodding spurge	Euphorbiaceae	Native	Annual	Weedy Forb	EUPNU
<i>Eutrochium purpureum</i> (L.) E.E. Lamont	Purple joe pye weed	Asteraceae	Native	Perennial	Prairie Forb	EUTPU
<i>Fraxinus americana</i> L.	White ash	Oleaceae	Native	Perennial	Woody	FRAAM
<i>Fraxinus pennsylvanica</i> Marsh.	Green ash	Oleaceae	Native	Perennial	Woody	FRAPE
<i>Gaillardia aristata</i> Pursh	Blanket flower	Asteraceae	Native	Perennial	Prairie Forb	GAIAR
<i>Galium aparine</i> L.	Catchweed bedstraw	Rubiaceae	Native	Annual	Weedy Forb	GALAP
<i>Galium boreale</i> L.	Northern bedstraw	Rubiaceae	Native	Perennial	Prairie Forb	GALBO
<i>Galium triflorum</i> Michx.	Sweet-scented bedstraw	Rubiaceae	Native	Perennial	Weedy Forb	GALTR

Table A1. (continued)

Scientific Name	Common Name	Family	Native	Life Cycle	Functional Group	ID
<i>Gentiana alba</i>	White gentian	Gentianaceae	Native	Perennial	Prairie Forb	GENAL
<i>Geum canadense</i>	White avens	Rosaceae	Native	Perennial	Weedy Forb	GEUCA
<i>Gleditsia triacanthos</i> L.	Honey locust	Fabaceae	Native	Perennial	Woody	GLETR
<i>Glycine max</i>	Soybean	Fabaceae	Introduced	Annual	Weedy Forb	GLYMA
<i>Hackelia virginiana</i>	Stickseed	Boraginaceae	Native	Biennial	Weedy Forb	HACVI
<i>Helenium autumnale</i> L.	Sneezeweed	Asteraceae	Native	Perennial	Prairie Forb	HELAU
<i>Helianthus annuus</i> L.	Common sunflower	Asteraceae	Native	Annual	Weedy Forb	HELAN
<i>Helianthus giganteus</i>	Tall sunflower	Asteraceae	Native	Perennial	Prairie Forb	HELGI
<i>Helianthus grosseserratus</i> Martens	Saw-tooth sunflower	Asteraceae	Native	Perennial	Prairie Forb	HELGR
<i>Helianthus maximiliani</i> Schrader	Maximillain sunflower	Asteraceae	Native	Perennial	Prairie Forb	HELMA
<i>Helianthus pauciflorus</i>	Prairie sunflower	Asteraceae	Native	Perennial	Prairie Forb	HELPA
<i>Heliopsis helianthoides</i> (L.) Sweet	Oxeye sunflower	Asteraceae	Native	Perennial	Prairie Forb	HELHE
<i>Heuchera richardsonii</i>	Alumroot	Saxifragaceae	Native	Perennial	Prairie Forb	HEURI
<i>Hibiscus trionum</i> L.	Venice mallow	Malvaceae	Introduced	Annual	Weedy Forb	HIBTR
<i>Hordeum jubatum</i> L.	Squirreltail barley	Poaceae	Native	Perennial	Weedy C3 Grass	HORJU
<i>Hypericum perforatum</i> L.	Common st. john's wort	Hypericaceae	Introduced	Perennial	Prairie Forb	HYPPE
<i>Ipomoea hederacea</i> (L.) Jacq.	Ivyleaf morningglory	Convolvulaceae	Introduced	Annual	Weedy Forb	IPOHE
<i>Ipomoea purpurea</i> (L.) Roth	Common morning glory	Convolvulaceae	Introduced	Annual	Weedy Forb	IPOPU
<i>Juncus tenuis</i> Willd.	Slender rush	Juncaceae	Native	Perennial	Rush	JUNTE
<i>Juncus torreyi</i>	Torrey's rush	Juncaceae	Native	Perennial	Rush	JUNTO
<i>Juniperus virginiana</i>	Eastern red cedar	Cupressaceae	Native	Perennial	Woody	JUNVI
<i>Koeleria macrantha</i> (Ledeb.) Schultes	June grass	Poaceae	Native	Perennial	Prairie C3 Grass	KOEMA
<i>Lactuca canadensis</i> L.	Wild lettuce	Asteraceae	Native	Biennial	Prairie Forb	LACCA
<i>Lactuca serriola</i> L.	Prickly lettuce	Asteraceae	Introduced	Biennial	Weedy Forb	LACSE

Table A1. (continued)

Scientific Name	Common Name	Family	Native	Life Cycle	Functional Group	ID
<i>Leonurus cardiaca</i>	Motherwort	Lamiaceae	Introduced	Perennial	Weedy Forb	LEOCA
<i>Lespedeza capitata</i>	Roundheaded bush clover	Fabaceae	Native	Perennial	Prairie Forb	LESCA
<i>Liatris aspera</i> Michx.	Rough blazing star	Asteraceae	Native	Perennial	Prairie Forb	LIAAS
<i>Liatris pycnostachya</i> Michx.	Prairie blazing star	Asteraceae	Native	Perennial	Prairie Forb	LIAPY
<i>Linum lewisii</i>	Blue flax	Linaceae	Native	Perennial	Prairie Forb	LINLE
<i>Lobelia inflata</i>	Indian tobacco	Campanulaceae	Native	Annual	Weedy Forb	LOBIN
<i>Lonicera maackii</i> (Rupr.) Herder	Amur honeysuckle	Caprifoliaceae	Introduced	Perennial	Woody	LONMA
<i>Lotus corniculatus</i> L.	Bird's foot trefoil	Fabaceae	Introduced	Perennial	Weedy Forb	LOTCO
<i>Lythrum salicaria</i> L.	Purple loosestrife	Lythraceae	Introduced	Perennial	Weedy Forb	LYTSA
<i>Maianthemum stellatum</i>	Starry flase soloman's seal	Ruscaceae	Native	Perennial	Weedy Forb	MAIST
<i>Medicago lupulina</i> L.	Black medic	Fabaceae	Introduced	Perennial	Weedy Forb	MEDLU
<i>Medicago sativa</i>	Alfalfa	Fabaceae	Introduced	Perennial	Weedy Forb	MEDSA
<i>Melilotus officinalis</i> (L.) Pall.	Yellow sweetclover	Fabaceae	Introduced	Perennial	Weedy Forb	MELOF
<i>Melilotus alba</i>	White sweetclover	Fabaceae	Introduced	Perennial	Weedy Forb	MELAL
<i>Monarda fistulosa</i>	Wild bergamot	Lamiaceae	Native	Perennial	Prairie Forb	MONFI
<i>Morus alba</i> L.	White mulberry	Moraceae	Introduced	Perennial	Woody	MORAL
<i>Morus rubra</i>	Red mulberry	Moraceae	Native	Perennial	Woody	MORRU
<i>Muhlenbergia frondosa</i> (Poir.) Fern.	Wirestem muhly	Poaceae	Native	Perennial	Weedy C4 Grass	MUHFR
<i>Myosotis verna</i>	Spring scorpion grass	Boraginaceae	Native	Annual	Weedy Forb	MYOVE
<i>Myosoton aquaticum</i>	Giant chickweed	Caryophyllaceae	Introduced	Perennial	Weedy Forb	MYOQA
<i>Nepeta cataria</i> L.	Catnip	Lamiaceae	Introduced	Perennial	Weedy Forb	NEPCA
<i>Oenothera biennis</i> L.	Evening primrose	Onagraceae	Native	Biennial	Prairie Forb	OENBI
<i>Oenothera gaura</i> W.L Wagner & Hoch	Biennial beeblossum	Onagraceae	Native	Biennial	Prairie Forb	OENGA
<i>Oxalis stricta</i> L.	Yellow wood sorrel	Oxalidaceae	Native	Perennial	Weedy Forb	OXAST
<i>Panicum capillare</i> L.	Witchgrass	Poaceae	Native	Annual	Weedy C4 Grass	PANCA

Table A1. (continued)

Scientific Name	Common Name	Family	Native	Life Cycle	Functional Group	ID
<i>Panicum dichotomiflorum</i> Michx.	Fall panicum	Poaceae	Native	Annual	Weedy C4 Grass	PANDI
<i>Panicum virgatum</i> L.	Switch grass	Poaceae	Native	Perennial	Prairie C4 Grass	PANVI
<i>Parthenium integrifolium</i> L.	Wild quinine	Asteraceae	Native	Perennial	Prairie Forb	PARIN
<i>Pastinaca sativa</i> L.	Wild parsnip	Apiaceae	Introduced	Biennial	Weedy Forb	PASSA
<i>Penstemon digitalis</i> Nutt. ex Sims	Foxglove beardtongue	Scrophulariaceae	Native	Perennial	Prairie Forb	PENDI
<i>Phalaris arundinacea</i> L.	Reed canary grass	Poaceae	Introduced	Perennial	Weedy C3 Grass	PHAAR
<i>Phleum pratense</i> L.	Timothy grass	Poaceae	Introduced	Perennial	Weedy C3 Grass	PHLPR
<i>Physalis heterophylla</i> Nees	Clammy ground cherry	Solanaceae	Native	Perennial	Weedy Forb	PHYHE
<i>Physalis subglabrata</i> Mackenz. & Bush	Smooth ground cherry	Solanaceae	Native	Perennial	Weedy Forb	PHYSU
<i>Phyostegia virginiana</i> (L.) Benth.	False dragonhead	Lamiaceae	Native	Perennial	Prairie Forb	PHYVI
<i>Plantago lanceolata</i> L.	English plantain	Plantaginaceae	Introduced	Perennial	Weedy Forb	PLALA
<i>Plantago major</i> L.	Broadleaf plantain	Plantaginaceae	Introduced	Perennial	Weedy Forb	PLAMA
<i>Plantago rugelii</i> Dcne.	Black-seed plantain	Plantaginaceae	Native	Perennial	Weedy Forb	PLARU
<i>Poa pratensis</i> L.	Kentucky bluegrass	Poaceae	Introduced	Perennial	Weedy C3 Grass	POAPR
<i>Poa trivialis</i> L.	Meadow grass	Poaceae	Introduced	Perennial	Weedy C3 Grass	POATR
<i>Polygonum amphibium</i> L.	Water smartweed	Polygonaceae	Native	Perennial	Weedy Forb	POLAM
<i>Polygonum aviculare</i> L.	Prostrate knotweed	Polygonaceae	Introduced	Annual	Weedy Forb	POLAV
<i>Polygonum convolvulus</i> L.	Wild buckwheat	Polygonaceae	Introduced	Annual	Weedy Forb	POLCO
<i>Polygonum pensylvanicum</i> L.	Pennsylvania smartweed	Polygonaceae	Native	Annual	Weedy Forb	POLPE
<i>Populus deltoides</i> W. Bartram ex Marshall	Common cottonwood	Salicaceae	Native	Perennial	Woody	POPDE
<i>Portulaca oleracea</i> L.	Common purslane	Portulacaceae	Introduced	Annual	Weedy Forb	POROL
<i>Potentilla arguta</i> Pursh	Tall cinquefoil	Rosaceae	Native	Perennial	Prairie Forb	POTAR
<i>Potentilla norvegica</i> L.	Rough cinquefoil	Rosaceae	Native	Biennial	Weedy Forb	POTNO
<i>Potentilla recta</i> L.	Sulfur cinquefoil	Rosaceae	Introduced	Perennial	Weedy Forb	POTRE

Table A1. (continued)

Scientific Name	Common Name	Family	Native	Life Cycle	Functional Group	ID
<i>Potentilla simplex</i> Michx.	Old field cinquefoil	Rosaceae	Native	Perennial	Weedy Forb	POTSI
<i>Prunus serotina</i> Ehrh.	Wild black cherry	Rosaceae	Native	Perennial	Woody	PRUSE
<i>Pycnanthemum tenuifolium</i> Schrad.	Slender mountain mint	Lamiaceae	Native	Perennial	Prairie Forb	PYCTE
<i>Pycnanthemum virginianum</i> (L.) Rob. & Fernald	Common mountain mint	Lamiaceae	Native	Perennial	Prairie Forb	PYCVI
<i>Raphanus raphanistrum</i> L.	Wild radish	Brassicaceae	Introduced	Annual	Weedy Forb	RAPRA
<i>Ratibida columnifera</i> (Nutt.) Wootton & Standl.	Prairie coneflower	Asteraceae	Native	Perennial	Prairie Forb	RATCO
<i>Ratibida pinnata</i> (Vent.) Barnh.	Greyheaded coneflower	Asteraceae	Native	Perennial	Prairie Forb	RATPI
<i>Rhus glabra</i> L.	Smooth sumac	Anacardiaceae	Native	Perennial	Woody	RHUGL
<i>Rosa arkansana</i> Porter	Prairie rose	Rosaceae	Native	Perennial	Prairie Forb	ROSAR
<i>Rosa multiflora</i> Thunb.	Multiflora rose	Rosaceae	Introduced	Perennial	Weedy Forb	ROSMU
<i>Rubus occidentalis</i> L.	Black raspberry	Rosaceae	Native	Perennial	Woody	RUBOC
<i>Rudbeckia hirta</i> L.	Black eyed susan	Asteraceae	Native	Perennial	Prairie Forb	RUDHI
<i>Rudbeckia subtomentosa</i> Pursh	Sweet coneflower	Asteraceae	Native	Perennial	Prairie Forb	RUDSU
<i>Ruellia humilis</i> Nutt.	Wild petunia	Acanthaceae	Native	Perennial	Prairie Forb	RUEHU
<i>Rumex crispus</i> L.	Curly dock	Polygonaceae	Introduced	Perennial	Weedy Forb	RUMCR
<i>Sanicula canadensis</i> L.	Canadian black snakeroot	Apiaceae	Native	Biennial	Weedy Forb	SANCA
<i>Schedonorus arundinaceus</i> (Schreb.) Dumort.	Tall fescue	Poaceae	Introduced	Perennial	Weedy C3 Grass	SCHAR
<i>Schizachyrium scoparium</i> (Michx.) Nash	Little bluestem	Poaceae	Native	Perennial	Prairie C4 Grass	SCHSC
<i>Setaria faberi</i> Herrm.	Giant foxtail	Poaceae	Introduced	Annual	Weedy C4 Grass	SETFA
<i>Setaria pumila</i> (Poir) Roem. & Schult.	Yellow foxtail	Poaceae	Introduced	Annual	Weedy C4 Grass	SETPU
<i>Setaria viridis</i> (L.) Beauv.	Green foxtail	Poaceae	Introduced	Annual	Weedy C4 Grass	SETVI
<i>Sida spinosa</i> L.	Prickly side	Malvaceae	Introduced	Annual	Weedy Forb	SIDSP

Table A1. (continued)

Scientific Name	Common Name	Family	Native	Life Cycle	Functional Group	ID
<i>Silphium integrifolium</i> Michx.	Rosinweed	Asteraceae	Native	Perennial	Prairie Forb	SILIN
<i>Silphium laciniatum</i> L.	Compass plant	Asteraceae	Native	Perennial	Prairie Forb	SILLA
<i>Silphium perfoliatum</i> L.	Cup plant	Asteraceae	Native	Perennial	Prairie Forb	SILPE
<i>Solanum carolinense</i> L.	Horsenettle	Solanaceae	Native	Perennial	Weedy Forb	SOLCAR
<i>Solanum ptycanthum</i> Dun.	Eastern black nightshade	Solanaceae	Native	Annual	Weedy Forb	SOLPT
<i>Solidago canadensis</i> L.	Canada goldenrod	Asteraceae	Native	Perennial	Prairie Forb	SOLCA
<i>Solidago nemoralis</i> Ait.	Old-field goldenrod	Asteraceae	Native	Perennial	Prairie Forb	SOLNE
<i>Solidago rigida</i> L.	Stiff goldenrod	Asteraceae	Native	Perennial	Prairie Forb	SOLRI
<i>Sonchus arvensis</i> L.	Perennial sowthistle	Asteraceae	Introduced	Perennial	Weedy Forb	SONAR
<i>Sonchus asper</i> (L.) Hill	Spiny sowthistle	Asteraceae	Introduced	Annual	Weedy Forb	SONAS
<i>Sorghastrum nutans</i> (L.) Nash	Indian grass	Poaceae	Native	Perennial	Prairie C4 Grass	SORNU
<i>Sporobolus compositus</i> (Poir.) Merr.	Rough dropseed	Poaceae	Native	Perennial	Prairie C4 Grass	SPOCO
<i>Sporobolus cryptandrus</i> (Torr.) Gray	Sand dropseed	Poaceae	Native	Perennial	Prairie C4 Grass	SPOCR
<i>Sporobolus heterolepis</i> (Gray) Gray	Prairie dropseed	Poaceae	Native	Perennial	Prairie C4 Grass	SPOHE
<i>Symphyotrichum ericoides</i> (L.) Nesom	Heath aster	Asteraceae	Native	Perennial	Prairie Forb	SYMER
<i>Symphyotrichum laeve</i> (L.) Á. Löve & D. Löve	Smooth blue aster	Asteraceae	Native	Perennial	Prairie Forb	SYMLA
<i>Symphyotrichum lanceolatum</i> (Willd.) G.L. Nesom	Panicled aster	Asteraceae	Native	Perennial	Prairie Forb	SYMLAN
<i>Symphyotrichum novae-angeliae</i> (L.) G.L. Nesom	New england aster	Asteraceae	Native	Perennial	Prairie Forb	SYMNO
<i>Symphyotrichum oolentangiense</i> (Riddell) G.L. Nesom	Sky blue aster	Asteraceae	Native	Perennial	Prairie Forb	SYMNO

Table A1. (continued)

Scientific Name	Common Name	Family	Native	Life Cycle	Functional Group	ID
<i>Symphotrichum pilosum</i> (Willd.) Nesom	Frost aster	Asteraceae	Native	Perennial	Prairie Forb	SYMPI
<i>Taraxacum officinale</i> F. H. Wigg.	Common dandelion	Asteraceae	Introduced	Perennial	Weedy Forb	TAROF
<i>Teucrium canadense</i> L.	American germander	Lamiaceae	Native	Perennial	Prairie Forb	TEUCA
<i>Thlaspi arvense</i> L.	Field pennycress	Brassicaceae	Introduced	Annual	Weedy Forb	THLAR
<i>Toxicodendron radicans</i> (L.) Kuntze	Poison ivy	Anacardiaceae	Native	Perennial	Woody	TOXRA
<i>Tradescantia ohiensis</i> Raf.	Ohio spiderwort	Commelinaceae	Native	Perennial	Prairie Forb	TRAOH
<i>Trifolium campestre</i> Schreb.	Low hop clover	Fabaceae	Introduced	Annual	Weedy Forb	TRICA
<i>Trifolium hybridum</i> L.	Aslike clover	Fabaceae	Introduced	Perennial	Weedy Forb	TRIHY
<i>Trifolium pratense</i> L.	Red clover	Fabaceae	Introduced	Perennial	Weedy Forb	TRIPR
<i>Trifolium repens</i> L.	White clover	Fabaceae	Introduced	Perennial	Weedy Forb	TRIRE
<i>Ulmus americana</i> L.	American elm	Ulmaceae	Native	Perennial	Woody	ULMAM
<i>Urtica dioica</i> L.	Stinging nettle	Urticaceae	Native	Perennial	Weedy Forb	URTDI
<i>Verbena hastata</i> L.	Blue vervain	Verbenaceae	Native	Perennial	Prairie Forb	VERHA
<i>Verbena stricta</i> Vent.	Hoary vervain	Verbenaceae	Native	Perennial	Prairie Forb	VERST
<i>Verbena urticifolia</i> L.	White vervain	Verbenaceae	Native	Perennial	Prairie Forb	VERUR
<i>Veronia baldwinii</i> Torr.	Western ironweed	Asteraceae	Native	Perennial	Prairie Forb	VERBA
<i>Veronia fasciculata</i> Michx.	Ironweed	Asteraceae	Native	Perennial	Prairie Forb	VERFA
<i>Veronicastrum virginicum</i> (L.) Farw.	Culver's root	Scrophulariaceae	Native	Perennial	Prairie Forb	VERVI
<i>Vicia villosa</i> Roth	Hairy vetch	Fabaceae	Introduced	Biennial	Weedy Forb	VICVI
<i>Viola sororia</i> Willd.	Common blue violet	Violaceae	Native	Perennial	Weedy Forb	VIOSO
<i>Vitis riparia</i> Michx.	Riverbank grape	Viticeae	Native	Perennial	Woody	VITRI
<i>Xanthium strumarium</i> L.	Common cocklebur	Asteraceae	Introduced	Annual	Weedy Forb	XANST
<i>Zea mays</i> L.	Cultivated corn	Poaceae	Introduced	Annual	Weedy C4 Grass	ZEAMA
<i>Zizia aurea</i> (L.) W. Koch.	Golden alexander	Apiaceae	Native	Perennial	Prairie Forb	ZIZAU

APPENDIX B. SUPPLEMENTARY FIGURES AND TABLES

Table B1. Models of the relative cover of different functional groups. Slope estimates and p-values (in parentheses) included for each model. None of our other covariates (site age, size, perimeter: area ratio, or season seeded) significantly improved model fit and therefore are not included in the table. Relative cover values were all logit transformed prior to analysis and slope estimates remained transformed. Significant p-values for year sampled are asterisked. Marginal R² indicates the variation explained by the fixed effects listed and conditional R² indicates the variation explained by the fixed and random effects.

	All prairie species	C₄ grasses	Non-leguminous forbs	All weedy species	Perennial weeds
Sampling year ¹	-0.20 (0.02*)	-0.30 (<0.001***)	0.02 (0.80)	0.14 (0.09)	0.26 (0.002**)
Seed mix richness	0.001 (0.95)	-0.03 (0.12)	0.03 (0.13)	-0.001 (0.98)	-0.02 (0.22)
R²					
Marginal	0.02	0.13	0.09	0.01	0.07
Conditional	0.92	0.95	0.91	0.92	0.93

¹ Sampling year is a categorical variable with two levels (2018 & 2019). Therefore, estimates reported are the effect of 2019 relative to 2018.

Table B2. Models of the relative cover of the top 5 most abundant perennial weeds as a function of sampling year and seed mix richness. Slope estimates and p-values (in parentheses) for each model included. Relative cover values for each species were logit transformed and estimates remain on the logit scale. Seed mix richness was included in each model since it was controlled for when modelling overall perennial weedy cover in Table B1.

	<i>Bromus inermis</i>	<i>Cirsium arvense</i>	<i>Poa pratensis</i>	<i>Taraxacum officinale</i>	<i>Phalaris arundinacea</i>
Sampling year ¹	0.33 (0.51)	0.74 (0.12)	0.33 (0.54)	-0.06 (0.75)	-0.36 (0.50)
Seed mix richness	-0.04 (0.54)	-0.02 (0.73)	-0.08 (0.21)	0.09 (0.08)	-0.002 (0.98)
R²					
Marginal	0.02	0.02	0.06	0.12	0.003
Conditional	0.71	0.77	0.68	0.93	0.75

¹ Sampling year is a categorical variable with two levels (2018 & 2019). Therefore, estimates reported are the effect of 2019 relative to 2018.

Table B3. Models of the relative cover of the top 5 most abundant prairie C₄ grasses as a function of sampling year and seed mix richness. Slope estimates and p-values (in parentheses) for each model included. Relative cover values for each species were logit transformed and estimates remain on the logit scale. Seed mix richness was included in each model since it was controlled for when modelling overall perennial weedy cover in Table B1.

	<i>Andropogon gerardii</i>	<i>Sorghastrum nutans</i>	<i>Panicum virgatum</i>	<i>Schizachyrium scoparium</i>	<i>Bouteloua curtipendula</i>
Sampling year ¹	0.08 (0.57)	-0.42 (0.004)	-0.13 (0.61)	-0.43 (0.19)	-0.93 (0.004)
Seed mix richness	-0.06 (0.11)	-0.02 (0.61)	-0.14 (0.02)	0.10 (0.05)	0.05 (0.40)
R²					
Marginal	0.10	0.03	0.21	0.14	0.05
Conditional	0.93	0.97	0.93	0.85	0.89

¹ Sampling year is a categorical variable with two levels (2018 & 2019). Therefore, estimates reported are the effect of 2019 relative to 2018.

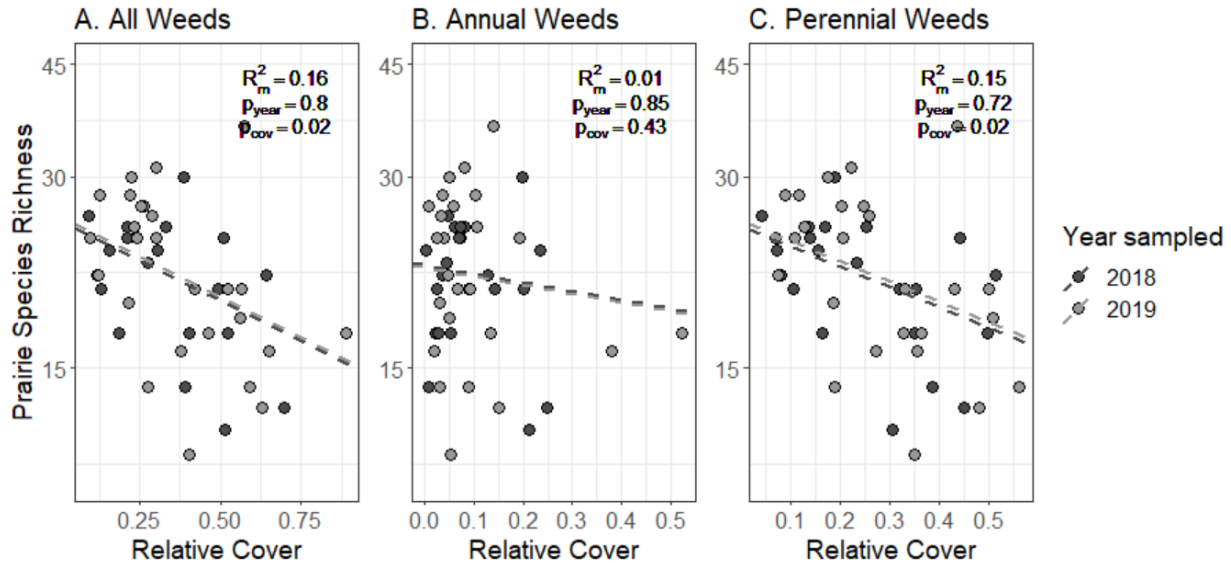


Figure B1. Relationship between the relative cover of all weedy species (A), annual weeds (B), and perennial weeds (C) and the species richness of the prairie community. Data shown on a log-scale. Points are colored by sampling year. Site was included as a random effect to account for variation due to repeated sampling. R_m^2 indicate the variation explain by the fixed effects (sampling year and relative cover). P-values for year and relative cover (cov) included.

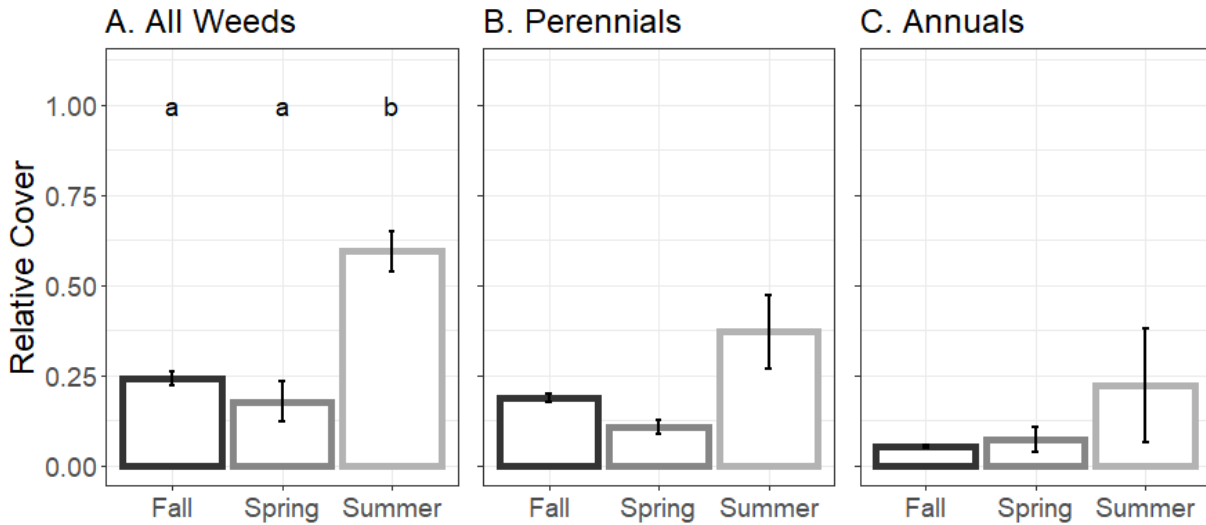


Figure B2. The relative cover of all weedy species (A), perennial weeds (B), and annual weeds (C) as a function of the season in which a site was seeded. Data are a subset ($n = 6$) of surveyed sites that used the same seed mix and were all sampled in the summer of 2019. Standard errors are shown for each mean value and letters indicated statistically significant differences after Tukey adjustment. Data were logit transformed for statistical testing.

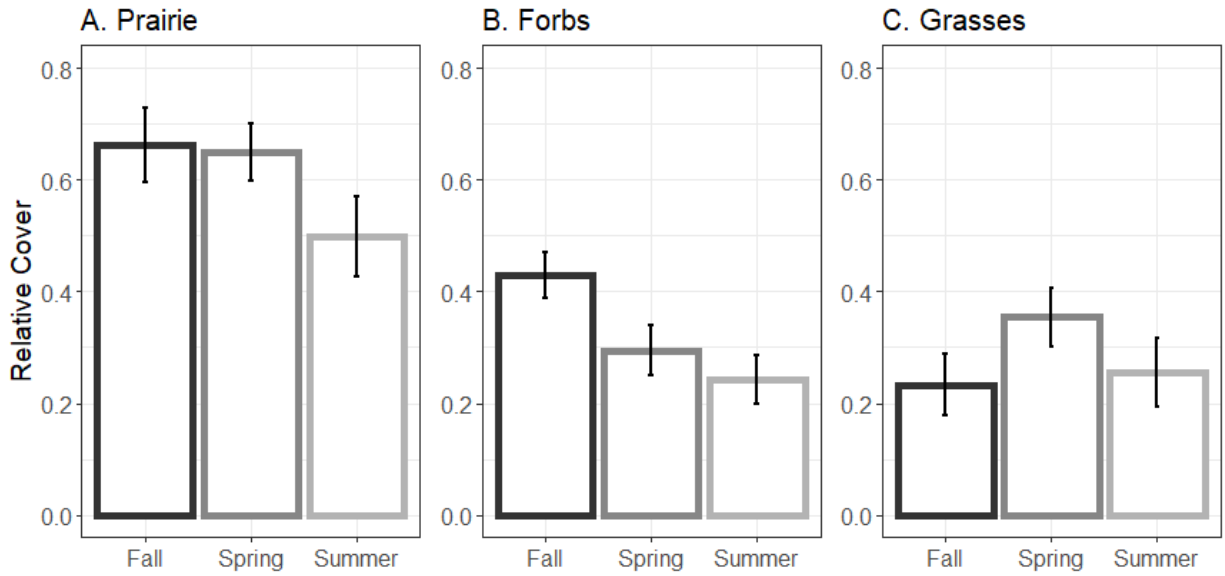


Figure B3. The relative cover of all prairie species (A), prairie forbs (B), and prairie grasses (C) as a function of the season in which a site was seeded. Data are from all sites sampled in the summer of 2019 ($n = 25$). Standard errors are shown for each mean value. Similar trends occur across all sites as seen in the subset of sites sown with the same seed mix in Figure 4, however here the pattern is not statistically significant due to the high variability in relative cover across sites.

APPENDIX C. SPECIES SPECIFIC DETECTION RATES

We used a survey of 22 sites that were sampled across Iowa in the summer of 2019 to assess the effectiveness of species establishment from seed mixes. In particular, we were interested in the detection rate of species across a variety of sites to deduce which species were consistent in their establishment. Given the high cost of a seed mix (Phillips-Mao et al. 2015), it's important that farmers and landowners are spending their money on seed that reliably establishes rather than seed that does not.

Species were considered “detected” at sites if they were present in the randomized vegetation survey, or if they were seen while walking through the site (see Methods in Chapter 2 for more details). The detection rate of each species was calculated as the number of sites where the species was detected divided by the number of sites where the species was seeded. We divided species into species that were detected at every site they were seeded, detected at some sites they were seeded, and detected at zero sites they were seeded.

Many of the species that were detected at every site they were seeded were not actually seeded many times (Figure 1). However, there were 9 species that were seeded at least 5 times and were detected everywhere, indicating these are reliable species for in-field restorations (Figure 1). These consistently establishing plants included three grasses (*Andropogon gerardii*, *Elymus canadensis*, *Elymus virginicus*), five non-leguminous forbs (*Heliopsis helianthoides*, *Monarda fistulosa*, *Ratibida pinnata*, *Pycnanthemum virginianum*, *Silphium integrifolium*), and one legume (*Desmodium canadense*).

Many species were detected at some, but not all sites where they were seeded. Most grasses were detected at least 50% of the time, the exception being *Koeleria macrantha* (Junegrass), which was detected less than <25% of the time (Figure 2B). *Koeleria macrantha* is

thought to be a shorter lived perennial (4-5 year lifespan) that prefers dry, well-drained soils, which may contribute to why it was not detected widely across sites (Houseal 2007). It may be that *K. macrantha* was seeded in areas that continually receive high moisture. Many forb species were detected at least 50% of the time. From those that were not, *Liatris aspera* (Rough blazing star) and *Veronicastrum virginicum* (Culver's root) were the only species that were widely seeded ($n = 10$ and $n = 16$, respectively), but not widely found ($n = 1$ and $n = 4$, respectively). It's hard to pinpoint the reasons for these trends, but other work has shown *L. aspera* to have low establishment across sites regardless of seeding density (Grman et al. 2015). Moreover, *V. virginicum* has very small seeds which require light for germination (Houseal 2007) so it may not readily establish if seeds are buried during planting.

Most of the species that were not detected anywhere were not seeded very many times (Figure 3), which is a reassuring finding. Only two species were seeded four times and not detected at any of those sites: *Helenium autumnale* (Sneezeweed) and *Amorpha canescens* (Leadplant). *Helenium autumnale* is small-seeded and requires light to germinate (Prairie Moon Nursery n.d.), therefore, like *V. virginicum*, it is possible seeds were buried or dried out, and did not readily establish across sites. *Amorpha canescens* has not been detected in other work examining establishment of prairie species in Conservation Reserve Program plantings (Hillhouse and Zedler 2011), and there is emerging evidence that it may be a species that relies on both rhizobia inoculum (Houseal 2007) and arbuscular mycorrhizal fungi (AMF) (Larimer et al. 2014) to establish and persist in prairie restorations, which could explain its absence in these in-field restorations.

Overall, these results demonstrate that many of the seeded prairie species establish well across in-field restorations. Only a couple species may not be worthwhile additions to seed mixes

due to their low establishment and detection. For species that do not reliably establish from seed in the field, the supplemental use of transplants (“plugs”) may be an effective way to increase diversity (Middleton et al. 2010).

References

- Grman, E., Bassett, T., Zirbel, C.R., Brudvig, L.A., 2015. Dispersal and establishment filters influence the assembly of restored prairie plant communities. *Restor. Ecol.* 23, 892–899. <https://doi.org/10.1111/rec.12271>
- Hillhouse, H.L., Zedler, P.H., 2011. Native species establishment in tallgrass prairie plantings. *Am. Midl. Nat.* 166, 292–308. <https://doi.org/10.1674/0003-0031-166.2.292>
- Houseal, G.A., 2007. Tallgrass prairie center’s native seed production manual. Tallgrass Prairie Center, Cedar Falls, IA.
- Larimer, A.L., Clay, K., Bever, J.D., 2014. Synergism and context dependency of interactions between arbuscular mycorrhizal fungi and rhizobia with a prairie legume. *Ecology* 95, 1045–1054. <https://doi.org/10.1890/13-0025.1>
- Middleton, E.L., Bever, J.D., Schultz, P.A., 2010. The effect of restoration methods on the quality of the restoration and resistance to invasion by exotics. *Restor. Ecol.* 18, 181–187. <https://doi.org/10.1111/j.1526-100X.2008.00501.x>
- Phillips-Mao, L., Refsland, J.M., Galatowitsch, S.M., 2015. Cost-estimation for landscape-scale restoration planning in the Upper Midwest, U.S. *Ecol. Restor.* 33, 135–146. <https://doi.org/10.3368/er.33.2.135>
- Prairie Moon Nursery, n.d. Prairie Moon Nursery Cultural Guide - All Species. <https://www.prairiemoon.com/blog/resources-and-information> (Accessed 27 March 2020).

Figures and Tables

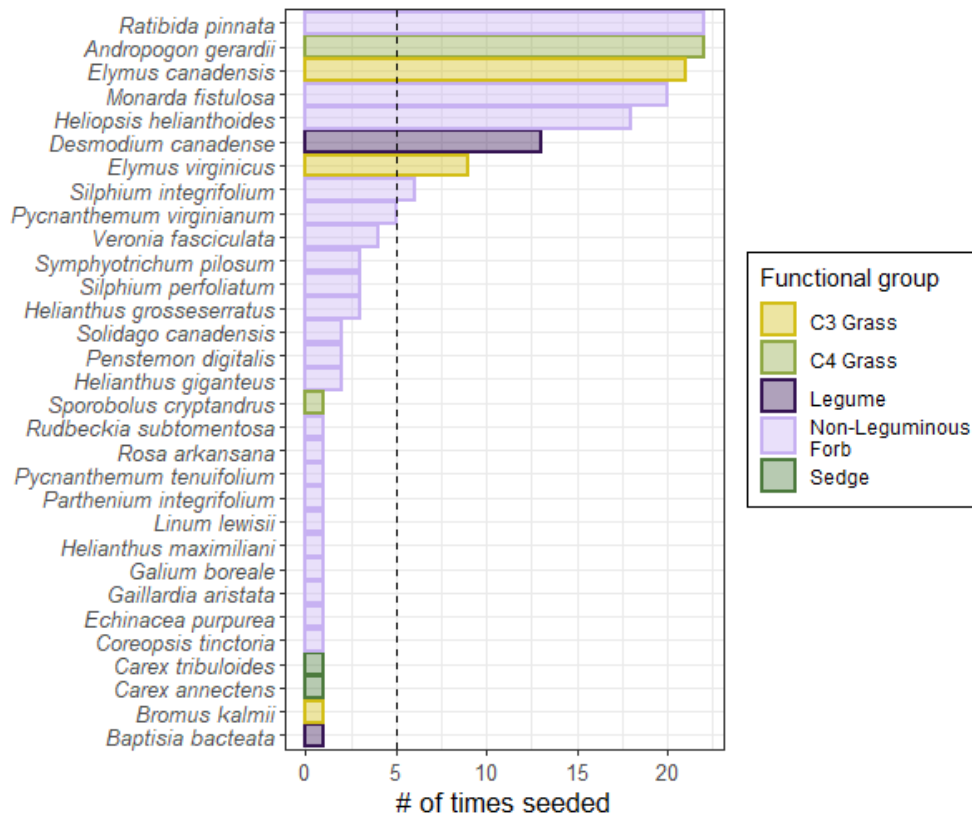
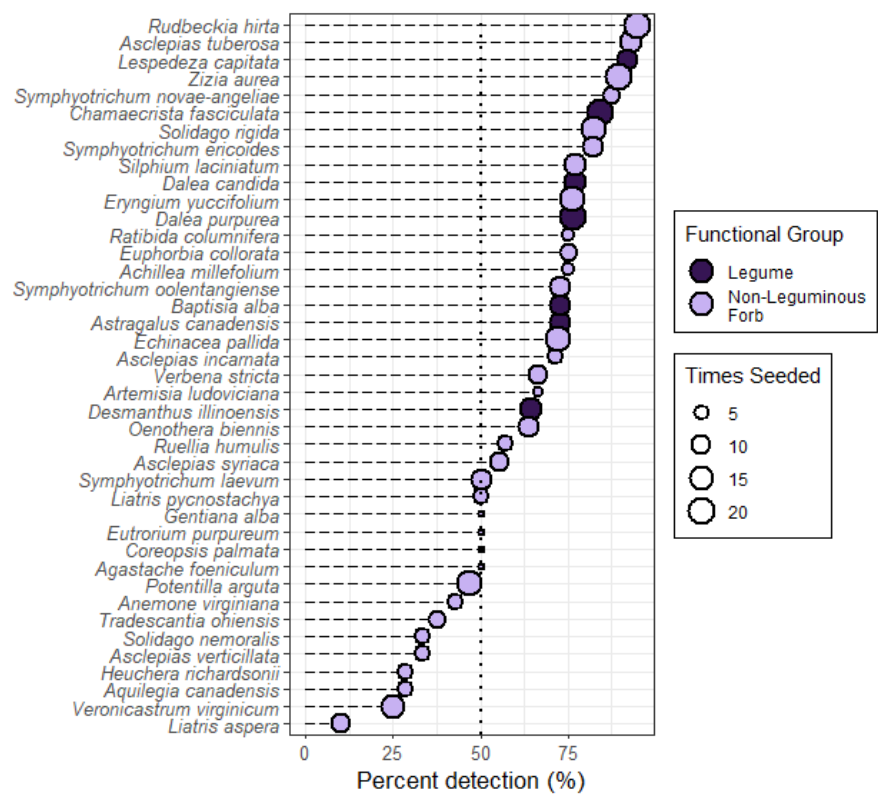


Figure C1. Seeded species that were always detected. Species are colored by their functional group. We drew a line at 5 to indicate species that were commonly seeded and always found, i.e. the most reliable of the seeded species.

A. Forbs



B. Grasses

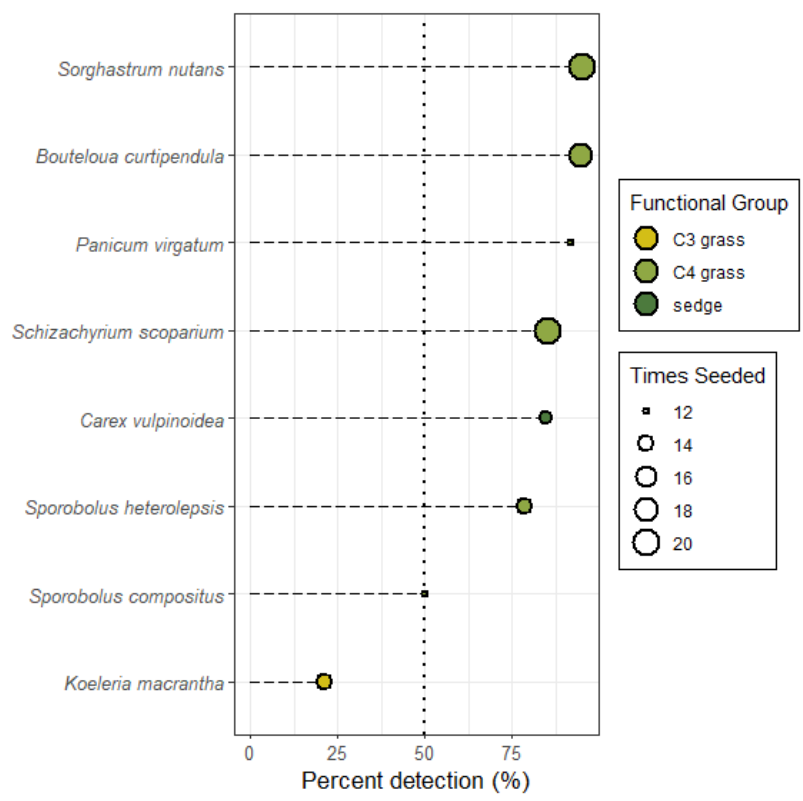


Figure C2. Seeded forbs (A) and grasses (B) that were detected in some of the sites they were seeded in. Species are colored by their functional group. The size of the circle indicates the number of times a plant was seeded. Dashed lines marked 50% detection across all sites

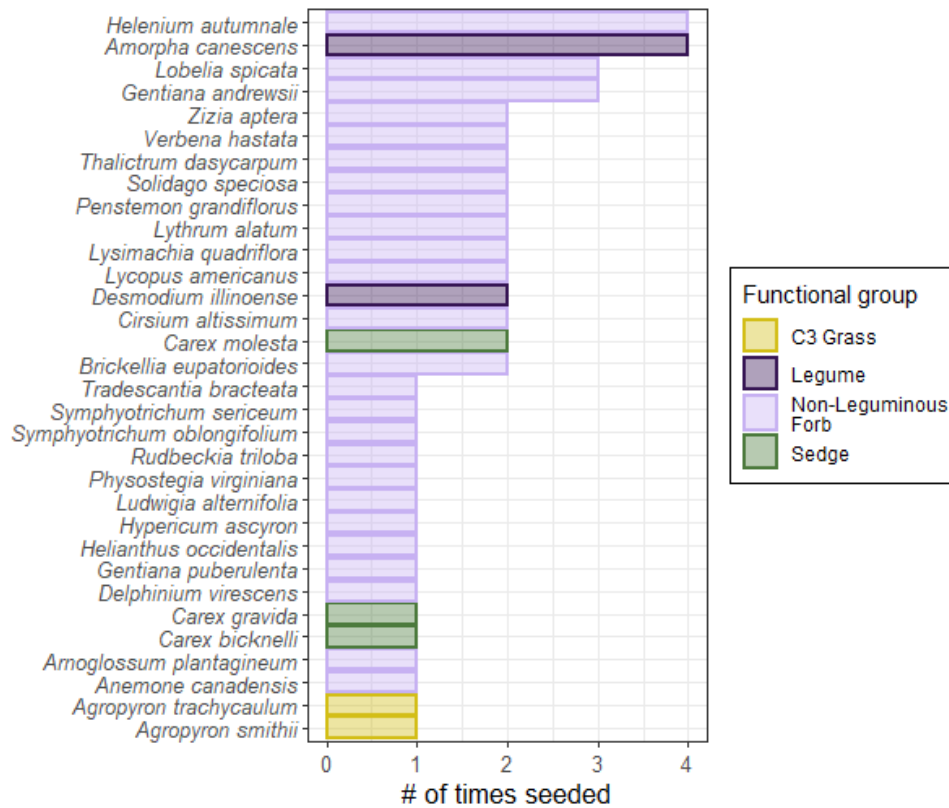


Figure C3. Seeded species that were never detected. Species are colored by their functional group. Note that the majority of these species were only seeded at 1 or 2 sites and were not commonly included in seed mixes. Two forbs, Sneezeweed (*Helenium autumnale*) and Leadplant (*Amorpha canescens*) are the only two species that were seeded a number of times ($n = 4$) and never detected.

APPENDIX D. COOPERATOR SURVEY RESULTS

We distributed a short survey during our initial summer of data collection (July – August 2018) and after our second season of data collection (December 2019). We sent our survey to all the cooperators whose sites we surveyed in 2018 ($n = 21$). The initial survey had 5 questions and was designed to gauge cooperators' confidence in identifying prairie plants, the methods they utilize to identify plants, and if they are interested in learning more about prairie plant identification. Our follow-up survey was identical to the initial survey, but also included a question that asked if cooperators identification skills had improved over the last two years (our study period). The IRB number for our survey is 18-284.

We had 17 responses to our initial survey (81%) and 16 responses (76%) to our final survey. Fourteen cooperators (67%) responded to both surveys.

Question 1 asked how many prairie grasses cooperators could identify with confidence. If respondents reported a range in values, we took the midpoint of the range. Responses ranged from 1 to 11 in the initial survey and 0 to 8 in the final survey. When examining the cooperators who responded to both surveys, the number of identifiable grasses increased for 6 people, decreased for 4 people, and stayed the same for 5 people (Figure 1A).

Question 2 asked how many prairie forbs cooperators could identify with confidence. Similar to question 1, we report the midpoint value if respondents wrote down a range. Responses ranged from 1 to 50 in the initial survey and 0 to 40 in the final study. The number of identifiable forbs increased for 9 people, decreased for 4 people, and stayed the same for 2 people (Figure 1B).

Question 3 asked cooperators to rank their interest, on a scale of 1 to 5, in learning to identify more species within their prairie strips, where 1 indicates completely uninterested and 5

indicates very interested. The majority of respondents chose rank 4 ($n = 6$) or 5 ($n = 7$) (Figure 2), meaning that most people were interested in learning more prairie plant identification. In the final survey, more respondents chose rank 5 ($n=10$), indicating they were very interested in learning more (Figure 2).

Question 4 asked cooperators to choose what kinds of strategies they employ to help identify unknown plants. The most common response in both surveys was to ask for someone else's help and the second most common response was to consult the internet or use a guidebook (Figure 3). Taking photos or samples of the plants were less commonly used strategies to help with plant identification.

Question 5 asked cooperators to list their favorite prairie plant. There was a wide diversity of responses (Table 1), indicating many people were already familiar with various prairie plants. Many respondents also indicated that it was hard to choose just one favorite (data not shown). Question 6 on the final survey asked cooperators to indicate if their knowledge of prairie plant identification increased over the study period. Answer choices included "increased", "stayed the same", and "decreased." The majority of respondents indicated that their knowledge increased over the study period. Three people indicated their knowledge stayed the same and no one responded that their knowledge decreased.

Overall, these survey responses indicate that many of early the adopters of prairie strips are interested in the vegetation within their plantings. Even though roughly half of respondents didn't indicate that the number of prairie grasses and forbs they could identify with confidence increased over the study period (Figure 1), the majority of respondents did indicate that they *perceived* their knowledge to have increased over the study period (Table 2). Furthermore, the majority of respondents are very interested in continuing to learn more plant identification

(Figure 2) and the most effective way to help them learn would be to connect them with another person who can help teach them (Figure 3).

Figures and Tables

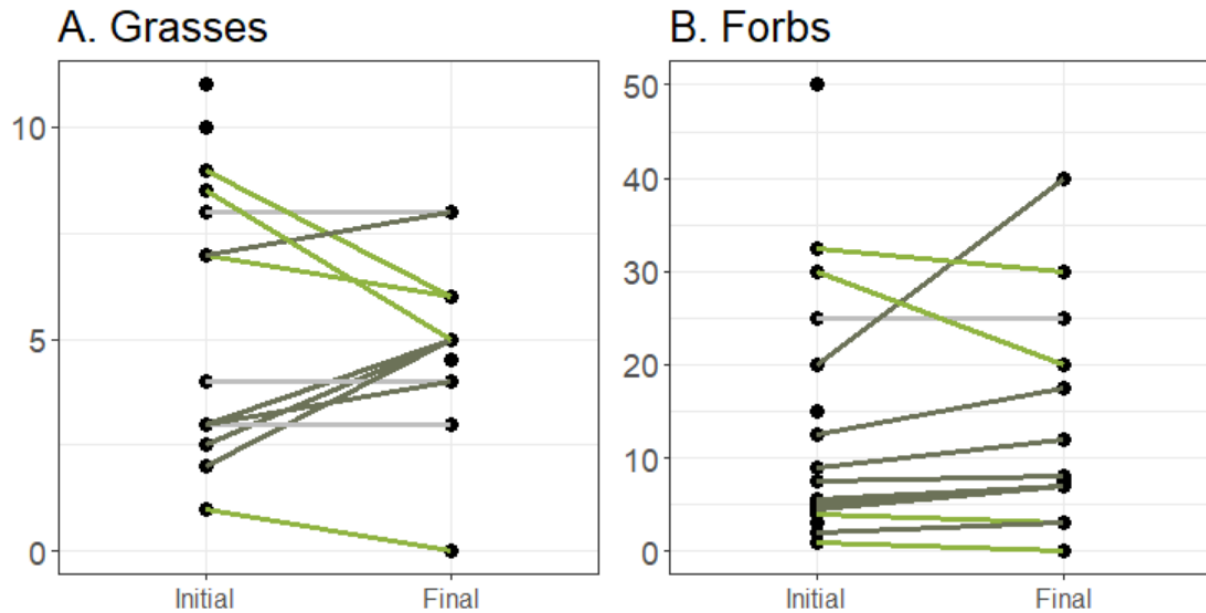


Figure D1. The number of prairie grass (A) and forbs (B) that cooperating landowners and farmers felt comfortable identifying with confidence. Responses are separated by initial and final surveys, which took place two years apart. Lines connect the same respondent and are colored by whether knowledge increased (dark green), decreased (light green), or stayed the same (grey) between survey periods.

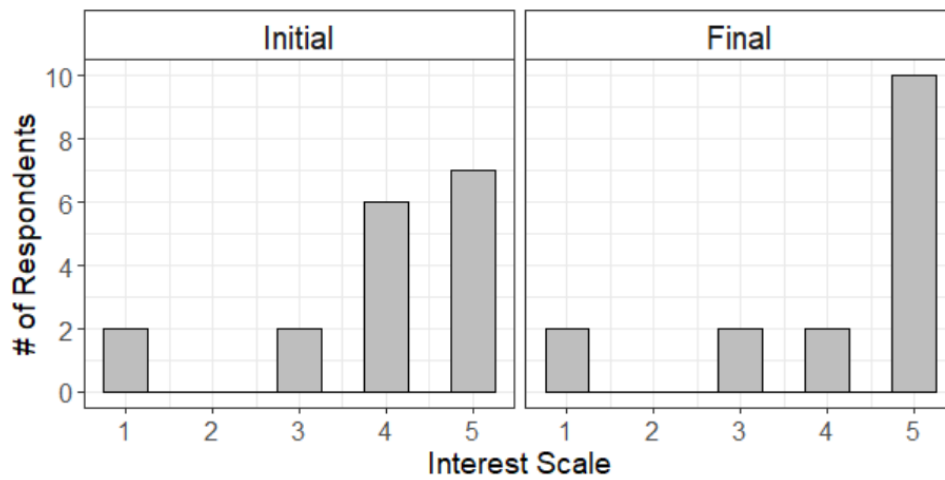


Figure D2. Histogram of responses ranking cooperators' interest in learning to identify more species in their prairie strips, where 1 = completely uninterested and 5 = very interested.

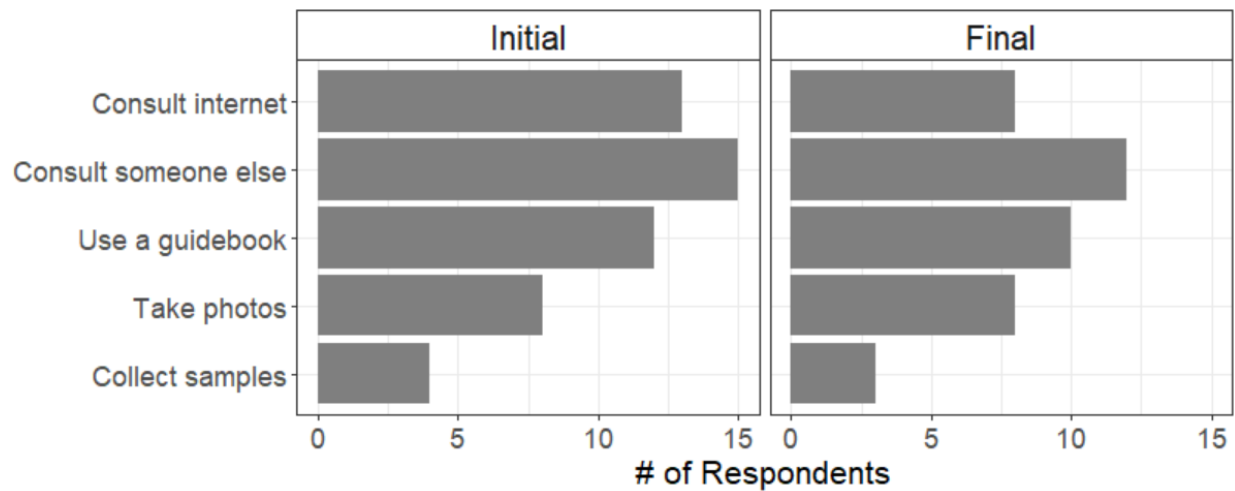


Figure D3. Histogram of responses when cooperators were asked to choose what strategies they use to identify unknown plants. Respondents could choose more than one response. Answers are separated by the initial and the final survey.

Table D1. List of “favorite prairie plants” as identified by the cooperating respondents.

Plant common name
Compass plant
Black-eyed Susan
Partridge pea
Prairie smoke
Wild bergamot
Butterfly milkweed
Purple prairie clover
Rattlesnake master
Ox-eye sunflower
Obedient plant
Little bluestem
Big bluestem
Canadian wild rye

Table D2. Responses to the question “Do you feel like your prairie plant identification skills have increased, decreased, or stayed the same over the past 2 years?” Respondents had to circle one of the three choices above.

Choices	Number of Responses
Decreased	0
Increased	13 ¹
Stayed the same	3

¹ Two respondents indicated that their identification only *slightly* increased

APPENDIX E. IRB APPROVAL FORM

IOWA STATE UNIVERSITY
OF SCIENCE AND TECHNOLOGY

Institutional Review Board
Office for Responsible Research
Vice President for Research
2420 Lincoln Way, Suite 202
Ames, Iowa 50014
515 294-4566

Date: 07/12/2018

To: Lydia English J. G Arbuckle

From: Office for Responsible Research

Title: Patterns of vegetation diversity in prairie strips across Iowa

IRB ID: 18-284

Submission Type: Initial Submission **Exemption Date:** 07/11/2018

The project referenced above has been declared exempt from the requirements of the human subject protections regulations as described in 45 CFR 46.101(b) because it meets the following federal requirements for exemption:

2: Research involving use of educational tests (cognitive, diagnostic, aptitude, achievement), survey procedures, interview procedures, or observations of public behavior, unless (i) Information obtained is recorded in such a manner that human subjects can be identified, and (ii) Any disclosure of the human subjects' responses outside the research could reasonably place the subject at risk of criminal or civil liability or be damaging to the subjects' financial standing, employability, or reputation.

The determination of exemption means that:

- You do not need to submit an application for annual continuing review.
- You must carry out the research as described in the IRB application. Review by IRB staff is required prior to implementing modifications that may change the exempt status of the research. In general, review is required for any *modifications to the research procedures* (e.g., method of data collection, nature or scope of information to be collected, changes in confidentiality measures, etc.), modifications that result in the *inclusion of participants from vulnerable populations*, and/or any *change that may increase the risk or discomfort to participants*. The purpose of review is to determine if the project still meets the federal criteria for exemption.

In addition, *changes to key personnel* must receive prior approval.

Detailed information about requirements for submission of modifications can be found on our [website](#). For modifications that require prior approval, an amendment to the most recent IRB application must be submitted in IRBManager. A determination of exemption or approval from the IRB must be granted before implementing the proposed changes.

Non-exempt research is subject to many regulatory requirements that must be addressed prior to implementation of the study. Conducting non-exempt research without IRB review and approval may constitute non-compliance with federal regulations and/or academic misconduct according to ISU policy.