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Current and Past Prairie Reconstruction Approaches

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Prairie Reconstruction History

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Introduction

The center of North America was once dominated by the prairie ecosystem, a grassland that has since been nearly entirely destroyed. After briefly considering the causes and consequences of the downfall of the prairie biome, the history of prairie reconstruction – the replanting and managing of prairie species in areas of their former range where they have been extirpated – is investigated.

The prairie biome of North America stretches across the center of the United States and Canada from the Rockies to the forests of Indiana and Wisconsin (Smith et al., 2010). Diverse grasses and forbs, herbaceous broad-leafed plants, underpin the ecosystem, which supports amphibians, birds, large herbivores like bison and elk, and predators like coyotes. This ecosystem evolved following the last glaciation approximately 8,300 years ago (Pannell, 2021). The prairie has been home to and managed by indigenous people like the Sioux and Ojibwa for generations ("Dakota and Lakhota," 2017; "Ojibwa," 2017; Pannell, 2021).

The history of prairie reconstruction begins with the destruction of the prairie. European colonization of the prairie region began earnestly in the 1830s, following the Black Hawk War in 1832. The short but bloody war between settlers and a group of Native Americans led by the Sauk warrior Black Hawk killed between 450 and 600 indigenous people and 70 European settlers. The war began in part from an 1804 treaty that ceded land in what is now Illinois to the US government, which the Sauk and Fox tribes did not view as legitimate. It ended with the massacre at Bad Axe of Black Hawk's band and yet another treaty, which ceded much of what is now Iowa to the US federal government (Lewis, 2014). In fact, "Billington suggested that the ruthless actions of the US forces in the final battle of the Black Hawk War may have discouraged resistance and expedited treaty making for the removal" of Native Americans living in the prairie region (Billington, 1960 as cited in Smith, 1990).

The area west of the Mississippi river was rapidly colonized by Europeans, fueled by additional treaties with Native Americans between 1832 and 1851 and the Homestead Act of 1860. Additional waves of settlement occurred after the Civil War in 1865 and during the Oklahoma Land Rush in 1888. While settlers originally preferred forests, thinking the lack of trees signaled a lack of soil fertility, the opposite was true and by 1840 cultivation of the prairie was well underway. A series of advancements in plow and thresher manufacturing occurred between 1837 and the 1870s; by 1900 most tallgrass prairie had been plowed (Smith, 1990, 1998).

European inhabitants believed the prairie had to be 'tamed', which reflected the Lockean idea that nature is waste and improved by labor. Many also believed the prairie to be endless: a grassland too large to be impacted by people. In a single generation, however, the frontier had closed, and the vast majority of rich prairie soil was cultivating settler monocultures of corn, oats, and wheat (Prince, 1997; Smith, 2001). What prairie remained was often too wet, sandy, or rocky to plow (Smith, 2014a).

Even the wet remnants were not protected for long; draining wet soils through the installation of drainage tile began in a large scale in 1888, and the pace of drainage increased after 1900. World War I increased demand for agricultural products, which prompted additional drainage; most wet prairies were drained by the 1920s. The depression of the 1930s slowed installation of tile, but drainage increased in pace again from 1945 to 1975 with the recovery of the economy (Smith, 1990, 1998; Prince, 1997).

Prairies survived in small remnant patches only in railroad and road rights-of-way, early cemeteries, hay fields, and pastures (Smith, 1998). Those patches of native prairie used for grazing rather than row cropping were often overgrazed or planted with invasive species (Jordan, 2010). The ecological integrity of these patches continues to be threatened by herbicide use, invasive species and woody plant encroachment, changes to hydrology, fire suppression, and the worldviews and policies that support and maintain monoculture farming (Faber et al., 2012; Smith, 2014a).

Through plowing, draining, and other European settler activities, prairies quickly became one of the most endangered ecosystems in the world, with less than 3 percent of the eastern tallgrass prairie remaining (Smith et al., 2010; McColpin et al., 2019). With such small populations of species, prairies are no longer ecologically functional. Reconstruction improves the viability of the prairie ecosystem through connecting and enlarging remnant prairies, which allows gene flow between populations (Helzer et al., 2010; Helzer, 2016; Niemuth et al., 2021).

Prairie reconstruction is the planting and managing of prairie species in areas where former prairie habitat has been completely removed, like former crop fields (Larson et al., 2017; McColpin et al., 2019). Though sometimes used interchangeably with reconstruction, prairie restoration is managing degraded, but remnant prairie (Larson et al., 2017). Reconstruction is the focus of this paper. It should be noted here that prairies are complex ecosystems that are impossible to fully reconstruct; some services and integrity are irretrievably lost when native prairies are lost (Larson et al., 2017).

Land Management Germinates

Initial interest in planting native species emerged from the field of land and range management at the end of the nineteenth century. Especially in the drier, western short grass prairies, erosion of soil and adequate forage crops for cattle were the most important considerations. One management option was the re-seeding of areas, which was done with both native and introduced forage species. Research and funding increased in response to the drought conditions of the 1930s. In 1933 the Soil Erosion Service and Civilian Conservation Corps were formed; both responded to the dust bowl conditions in the Midwest through range, soil, and water conservation activities. Again, a prominent treatment was range reseeding for cover, with no implicit focus on native species (Wasser, 1977). In fact, two prominent range researchers, Arthur Sampson and Lincoln Ellison, worked with native species in the early twentieth century before shifting their focus to introduced species (Jordan and Lubick, 2011).

Two other developments are notable from the 1930s. First, the century is considered the beginning of prairie reconstruction with the acquisition at the University of Wisconsin Madison Arboretum of what would become Curtis Prairie, though we will consider that reconstruction later. Secondly, the United States Fish and Wildlife Service (USFWS) initially acquired land in the prairie region in the 1930s. These National Wildlife Refuges were managed for waterfowl habitat by creating dense nesting cover and damming waterways to create standing bodies of water. Former croplands were often seeded with non-native species to create tall, dense nesting habitat and idled to protect nesting birds. These habitats were not sustainable, as for example, plots had to be re-seeded (Dixon et al., 2019).

In 1958 the Duck Stamp Act was amended, which allowed for the purchase of Waterfowl Production Areas (WPA). These lands, managed by the USFWS through Wetland Management Districts (WMD), were also designed to create waterfowl habitat. By the 1950s, as mentioned, drainage had become widespread, and WPAs were intended to save wetlands from drainage to protect waterfowl breeding. Ironically, the federal government was also subsidizing drainage at the same time through the US Department of Agriculture (USDA) (Dixon et al., 2019). Revegetating land and protecting wetlands points to an understanding that land had to be managed to prevent erosion or provide habitat; i.e. that the rampant use of as much land as possible for row cropping was unsustainable. However, these projects were concerned with specific range and game species, and not with reconstructing an entire ecosystem. In fact, non-game species only gained explicit protection in 1980 through the Fish and Wildlife Conservation Act ("History of the U.S. Fish and Wildlife Service," n.d.). To understand how the USFWS and others eventually undertook prairie reconstruction, we turn now to the earliest reconstruction projects.

Prairie Reconstruction Germinates

Jordan and Lubick (2011) identify at least six reconstruction projects that began between 1906 and 1934. However, only one project continued after the 1940s and only one project (though incidentally the same project) reconstructed a prairie ecosystem. That project, at the University of Wisconsin Madison Arboretum, was under the direction of Aldo Leopold, Norman Fassett, and Theodore Sperry. The University of Wisconsin Madison acquired in 1933 what would be named Curtis Prairie. In 1935, "Camp Madison" became home to over a hundred workers through the Civilian Conservation Corp program. The crew planted the prairie with seed and mature plants from relict prairies. In the 1940s fire was introduced to the area to control invasive species, in an attempt to make the area more ecologically functional (Jordan, 2010; Jordan and Lubick, 2011).

These earliest reconstruction projects were primarily private projects or tied, like Curtis Prairie, to academic institutions. This both allowed and restricted reconstruction work. On one hand, these projects allowed investigation of species considered uneconomic, unimportant, or nuisances. The projects could pursue knowledge for its own sake, without having to defend their choices as practical or economical. Leopold undertook a smaller, similar project on his property in Sauk County, Wisconsin, which provides an even more sheltered example. Jordan and Lubick (2011) note, “Leopold and his family could work - or play - there without having to justify the effort as research or beautification, as practical or even sensible.” However, in some ways these small (25 hectare) early restorations were restricted to what could be seen as academic gardening (“USA: Wisconsin: Curtis Prairie Restoration,” n.d.). Work – transplanting, watering, weeding - was almost entirely done by hand, making reconstructing even small areas extremely labor intensive (Jordan and Lubick, 2011). These early prairies were also often planted with mature plants from remnants, which would not be a suitable, sustainable source for larger reconstructions.

Prairie work of the 1930s and 1940s not associated with academia was less common and focused more on preservation. Even prairie preservation was difficult at points, as prairies have historically suffered from a perception as wasted space (Helzer, 2021). Plus, as Lubick and Jordan note, it takes considerably more willpower and funding to reconstruct a prairie than it does to preserve a prairie (Jordan and Lubick, 2011). As early as 1925, Dr. Bohumil Shimek suggested restoring Iowa prairies with seed from prairie hay fields. Shimek’s 1934 plan for prairie preservation actually went beyond preservation or restoration to include reconstruction of areas between remnants, though the plan was never implemented (Roosa, 1976; Smith, 1998). Iowa first dedicated a prairie preserve in 1947 (Smith, 1998).

Evolution of Reconstruction

There was a considerable lull in prairie reconstruction between the 1940s and 1970s. The CCC camp at UW-Madison closed in 1941, and while work continued at Curtis Prairie, it did so at a slower rate with fewer workers. In 1945 Henry Greene began reconstructing a second prairie at the UW-Madison Arboretum, though it was one of few projects during the era (M. R. Anderson and Cottam, 1968). The others were a prairie begun at the University of Illinois by Victor Shelford in 1942 and the Green Oaks project at Knox College in Galesburg, Illinois begun by Paul Shepard in 1955 (Schramm, 1968a). Many of the other early projects disbanded at this time, often when their primary organizers died or retired (Jordan and Lubick, 2011).

Jordan and Lubick (2011) assert that the value of reconstruction was not realized until the 1970s, which may have caused the mid-century disinterest. Prairie reconstruction seems unpractical and expensive without a realization of its unique benefits. Another cause of the lull may be the increased focus on specialization after World War II, which possibly stunted the highly-interdisciplinary work of reconstruction. By the 1960s, environmental awareness and activism was growing, but much attention was given to preservation of ecosystems. This also stunted the development of reconstruction, as preservationists pointed out that fully reconstructing any ecosystem is impossible. Reconstruction could be used as an excuse for destroying instead of preserving ecosystems, with the thought that the destroyed ecosystems could be rebuilt elsewhere.

Prairie reconstruction began to emerge again in the 1960s, perhaps less publicly than other forms of environmental action. Ray Schulenberg began a prairie reconstruction at Morton Arboretum in Lisle, Illinois in 1963 (Schulenberg, 1968). In 1965 both Iowa State University and Boerner Botanical Gardens in Hales Corners, Wisconsin began prairie reconstructions (Landers et al., 1968; Ode, 1968). In 1967 the Michigan Botanical Gardens began planting their prairie reconstruction, and the Southern Illinois University outdoor laboratory was seeded in 1968 (Bland, 1968; R. C. Anderson, 1970).

Inspired by the growing interest, and recognizing the need for practitioners to communicate, Peter Schramm organized a conference on prairie reconstruction in 1968 (Jordan and Lubick, 2011; Smith, 2014a; Schramm, 2016). First called a Symposium on Prairie and Prairie Restoration, then the Midwest Prairie Conference, the ongoing biennial conference is now called the North American Prairie Conference (Schramm, 1968b; Zimmerman, 1970a; Hulbert, 1972; Wali, 1974; Glenn-Lewin and Landers, 1976; Stuckey and Reese, 1978; Kucera, 1980; Brewer, 1982).

The 1970s mark the true birth of prairie reconstruction, with more projects beginning than can be listed here. Reconstruction evolved in a number of ways during the 1970s, though of course not all projects fit these trends. First, reconstructions were planted increasingly with seed instead of seedlings or mature plants. For example, whereas SIU used prairie sod as a species source in 1965, and other 1960s projects used both transplants and seed, later projects used only seed (Landers et al., 1968; Schramm, 1968a; Schulenberg, 1968). The seed mixes were also becoming increasingly diverse; compare the 7 grass species planted in 1970 at the Allwine Prairie Preserve in Bennington, Nebraska to a prairie reconstruction planted in 2022 by The Nature Conservancy in Nebraska with 153 species (Bragg, 1976; Helzer, 2022).

This also highlights another point; early 1970s reconstructions tended to plant mainly grass species, in part because grass seed was most easily available (Smith, pers. comm.). Later reconstructions included more forbs with the understanding that it is difficult to add forbs to a site once grasses are established (Schramm, 2016). Included among the forbs of later reconstructions were species difficult to propagate. Early mixes often used primarily warm-season grasses, like big bluestem (*Andropogon gerardi*), the seed of which is fairly easy to harvest. Later diverse seed mixes included an increasing component of cool-season species like pasqueflower (*Pulsatilla patens*), which necessitated new methods to gather and propagate seed. Prairie phlox (*Phlox pilosa*), for example, flings ripe seeds away from the plant so the plants must be bagged before the seeds are ripe, flung, and lost (Zimmerman, 1970b). Diversity also extended beyond plants. Practitioners researched or added animals, fungi, and bacteria (Smith, 2014a). For example, Fermi National Accelerator Laboratory in Batavia, Illinois bought four bison in 1969 to graze their reconstruction (Jordan and Lubick, 2011). Finally, while mowing and burns were often used during the weedy first years of reconstructions (Schramm, 2016), disturbance has been recognized as continually necessary for the maintenance of prairie ecosystems (Jordan and Lubick, 2011).

Planting was also done increasingly mechanically. While transplanting had to be done by hand, and much seed was initially broadcast by hand, seed broadcasters and drills became increasingly popular ways to plant prairies. The Knox College Biological Field Station first used a Nisbet seed drill in 1970 (Schramm, 1970). Detailed procedures are now available to plant the many shapes and sizes of prairie seeds. Agronomic techniques are more practical and less labor-intensive, a key feature for prairie reconstructions to grow in size (Woehler and Martin, 1976; Smith et al., 2010; Schramm, 2016).

Prairie reconstructions did grow, by orders of magnitude. In 1974, Bob Betz began a 283 hectare prairie reconstruction within the accelerator ring at Fermi National Accelerator Laboratory, which has since grown to 486 hectares ("Tallgrass Prairie," n.d.; Jordan and Lubick, 2011; Smith, 2014a, 2014b). In 1991, the 3,480 hectare prairie at Neal Smith National Wildlife Refuge in Prairie City, Iowa was planted, and in 1996, reconstruction began on over 7,000 hectares at Midewin National Tallgrass Prairie in Wilmington, Illinois (Smith, 2014a).

These developments in prairie reconstruction were reflected in organizations beyond the North American Prairie Conference. In 1981 the journals *Restoration and Management Notes* (now *Ecological Restoration*) and *Natural Areas Journal* were first published. The first conference of the newly founded Society of Ecological Restoration met in 1989 (Jordan and Lubick, 2011). *Restoration Ecology* began publishing in 1993 ("Issue archive," n.d.).

Many of these early projects were based on trial and error and the hunches of practitioners, often because there was a feeling of urgency to respond to the massive loss of prairie (Smith, 2014a; Schramm, 2016). This approach has sometimes insinuated that prairie reconstruction is too variable to be replicable, though reconstruction has evolved to include more comparison of past reconstructions (Norland et al., 2015), plot-based comparative studies (Larson et al., 2017), and monitoring (McColpin et al., 2019) to guide future reconstructions.

Current Reconstruction

Reconstruction continues to evolve and improve, in part because of a number of collaborations designed to gather and share research about prairie reconstruction. The Grassland Restoration Network is a collaboration of prairie restoration practitioners organized in 2003 by The Nature Conservancy (Helzer et al., 2010). In 2012 the USFWS and others organized the Prairie Reconstruction Initiative (PRI) to research and improve the reconstruction process. The PRI has a standardized monitoring framework and database to allow research across all sites participating in the initiative (McColpin et al., 2019). There are also plans to coordinate prairie restoration on a scale beyond individual projects. For example, pollinators are heavily dependent on the size and connectivity of prairie patches, which has garnered research and conservation strategies concerned with prairie habitat beyond discrete reconstructions (Niemuth et al., 2021). Another example is the Minnesota Prairie Conservation Plan. The 25 year plan seeks to preserve and manage “functioning landscapes” across the entire prairie biome of Minnesota, with a specific focus on restoration and reconstruction, as the current protected prairies are too small to be ecologically functional (Chaplin et al., 2018).

“Growin’ Brome”

This evolution of techniques for prairie planting suggests that reconstruction has been fairly linear, germinating in the 1930s and maturing in the 1970s. That, however, is far from the case. One example of conflicting and evolving land management practices in the century after 1930 is the treatment and opinions on smooth brome. Smooth brome (*Bromus inermis*) is a central and northern Eurasian species, first introduced to the US in 1884, and now a difficult-to-control invader of prairie remnants and reconstructions (Wasser, 1977; Wasser and Dittberner, 1986; Dixon et al., 2019).

As early as 1906, the USDA recommended smooth brome be planted in rangelands in the western Dakotas, and 1941 research suggested that adding smooth brome to native ranges improved forage value (Wasser, 1977). A 1952 article titled “Growin’ Brome” in *Iowa Farm Science* illustrates the contemporary farmer’s opinions on smooth brome: "smooth bromegrass in recent years has become one of our most valuable grasses for hay, pasture and silage.” The plant was viewed as an important species to control erosion, though it was noted that smooth brome could become sod-bound. The grass was a valued pasture species when grown with alfalfa, another introduced species. In fact, the paper provides instructions on how to ‘renovate’ a pasture to smooth brome and alfalfa (Kalton et al., 1952), and the modern reader wonders what (perhaps native) pasture species were plowed under during the renovation.

A 1986 US Army Corps of Engineers Wildlife Resources Management Manual describes smooth brome as “useful for wildlife cover and soil conservation” as well as widely used for range reseeding. Much of the information mirrors that of 1952, though there were warnings that smooth brome grown alone outcompetes other vegetation and results in low diversity (Wasser and Dittberner, 1986).

Conclusion

The US Army Corps of Engineers manual is from the middle of the 1980s, which is arguably within the era of prairie reconstruction. However, society's opinions on and activities with prairies are not linear nor uniform. Often driven by high commodities prices, native prairie remnants continue to be plowed and degraded even as prairie preservation, restoration, and reconstruction are embraced and increasingly practiced by private landowners, conservation organizations, the USFWS, state departments of natural resources and departments of transportation (Faber et al., 2012; Wimberly et al., 2017).

Prairie reconstruction, first attempted in Wisconsin in the 1930s, developed into a widespread discipline in the decades from the 1970s onward. Improved techniques, monitoring, management, and research allows for the reconstruction of increasingly prairie-like grasslands. Reconstructions can never fully mimic what the prairie ecosystem once was, so remnant restoration and preservation continue to be important. However, given the fragmented and small size of remnants, the increase in permanent, diverse grassland cover that reconstruction offers will hopefully ensure the survival of prairie species into the future.

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Monitoring of Selected Morris Wetland Management District Prairie Reconstructions

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Introduction

The prairie biome once covered an area of over 150 million hectares, stretching across the United States and Canada from the Rockies to the forests of Indiana and Wisconsin (Smith et al., 2010; Smith, 2014a). European settlement between 1830 and 1900 transformed the area, as settlers plowed and drained the prairie for row cropping (Smith, 1990, 1998, 2014b; Prince, 1997). What prairie remnants remain are threatened by herbicide use, invasive species and woody plant encroachment, fire suppression, and monoculture farming (Faber et al., 2012; Smith, 2014b). As a result, prairie is one of the most endangered ecosystems; the most affected has been the eastern tallgrass prairie, of which less than 3 percent remains (Smith et al., 2010; McColpin et al., 2019).

As a result of this massive loss, the prairie ecosystem is no longer ecologically functional. Prairie remnants in settler cemeteries, railroad rights-of-way, and hay fields are often too small to support viable populations of prairie species, or risk negative effects like inbreeding. Prairie remnants are also extremely fragmented, which compounds the problem. Migration through the agricultural matrix and other forms of gene flow are unlikely, isolating what populations still exist (Helzer et al., 2010; Helzer, 2016; Niemuth et al., 2021).

One way to counteract this immense loss is prairie reconstruction, the planting and managing of prairie species in areas they have been fully extirpated, like former crop fields (Larson et al., 2017; McColpin et al., 2019). While prairie reconstruction can never fully recreate a native prairie, it provides habitat for prairie species and can connect prairie remnants to counter the effects of habitat loss and fragmentation (Helzer, 2016; Larson et al., 2017; Niemuth et al., 2021).

The United States Fish and Wildlife Service (USFWS) first acquired land in the prairie region in the 1930s. Lands were managed as National Wildlife Refuges to provide habitat for waterfowl, sometimes to the detriment of other species. For example, upland areas around wetlands were seeded with non-native plants like smooth brome to provide dense nesting habitat. In 1958 the Duck Stamp Act was amended, which allowed the purchase of Waterfowl Production Areas (WPA). These areas, overseen through the USFWS by Wetland Management Districts (WMD), were also managed as waterfowl habitat and were intended to counter the massive drainage of wetlands. Most WPAs were acquired in the 1960s and 1970s (Dixon et al., 2019), including the lands managed by the Morris WMD. The WMD opened in 1964 in Benson, before moving to Morris in the 1980s (Morris Wetland Management District, n.d., S. Vacek, pers. comm.).

While USFWS land management initially focused on only waterfowl breeding, management focuses shifted in the 1990s to broader ecological needs. Included in these changes was managing upland habitat for a variety of native prairie species. Native forbs and grasses were planted to attempt to reconstruct prairie habitat rather than planting native or non-native grasses solely for the benefit of waterfowl. The Morris WMD began prairie reconstruction projects on their WPAs in the late 1990s (S. Vacek, pers. comm.). In 2012 the USFWS and other collaborators launched the Prairie Reconstruction Initiative to research and improve the reconstruction process. The PRI has a standardized monitoring framework and database to allow meta-analysis across all sites participating in the initiative (Dixon et al., 2019; McColpin et al., 2019).

This analysis seeks to summarize the condition of three reconstructions at the Morris WMD, which are monitored under the PRI framework. Ideally these reconstructions are both replicating as nearly as possible a native prairie ecosystem as well as providing habitat for waterfowl and other prairie wildlife. The PRI framework is relatively new, and so this analysis also seeks to assess the successes and limitations of the PRI monitoring protocol. These data provide an example of time commitment and information generated under the current monitoring protocol and may indicate adjustments to monitoring in the future.

Study Sites

The PRI monitoring framework has specific guidelines for how monitoring areas are named and delineated. The "overall site" is the preserve or office managing reconstructions. Here, the Morris WMD is the overall site. "Plantings units" are management units, and there are multiple planting units in the overall site. Monitoring information here is summarized for only three WPAs managed by the Morris WMD. These three WPAs, the Edwards WPA, the Pomme de Terre Lake WPA, and the Loen WPA, are the planting units. Finally, planting units are broken into "seed mix areas", those areas that received the same seeding and management treatment at the same time. There can be one or many seed mix areas in a planting unit. Each seed mix area is separately monitored (McColpin et al., 2019).

The Pomme de Terre Lake WPA is a 62 ha unit in Stevens County, MN (95.8776013 W, 45.6900108 N). The 12 ha planting unit consists of only one seed mix area, as the entire planting unit was drilled with the same seed mix in 2011. The Loen WPA is a 291 ha unit in Swift County, MN (95.4922614 W, 45.3788518 N). The planting unit consists of three seed mix areas, named for the year in which they were seeded: the 6 ha 2009 seed mix area was broadcast seeded, the 6 ha 2013 seed mix area was drilled, and the 6 ha 2017 seed mix area was broadcast seeded.

The Edwards WPA is a 221 ha unit in Stevens County, MN (95.8307588 W, 45.5653139 N). The planting unit consists of seven seed mix areas, six of which are summarized in this report. Multiple seeding mixes were designed for the unit to correspond to soil moisture classes, and problems during initial seeding resulted in some areas being reseeded. All areas with different seeding treatments are monitored as separate seed mix areas. The 1 ha Sculpted Seeding A seed mix area was drilled in 2015 with a mesic-wet seed mix, overseeded by hand broadcast in 2015 with a hand harvested seed mix, and drilled in 2016 with a mesic-wet seed mix. The 0.4 ha Sculpted Seeding B seed mix area was drilled in 2015 with a mesic seed mix and overseeded by hand broadcast in 2015 with a hand harvested seed mix. The 6 ha Sculpted Seeding C seed mix area was drilled in 2015 with a mesic seed mix. The 6 ha Sculpted Seeding D seed mix area was drilled in 2015 with a dry-mesic seed mix. The 0.4 ha Sculpted Seeding E seed mix area was drilled in 2015 with a dry-mesic seed mix and overseeded by hand broadcast in 2015 with a hand harvested seed mix. The 4 ha Shelterbelt A seed mix area was drilled in 2015 with a mesic seed mix (note that this seed mix area received a difference mesic seed mix than the Sculpted Seeding C seed mix area). The final seed mix area, the 0.4 ha Shelterbelt B area, was drilled and overseeded by hand broadcast in 2015 but is not summarized in this report (S. Vacek, pers. comm.).

Survey Methods

The PRI monitoring framework includes two forms of monitoring, a nested plot sampling method and a meandering walk sampling method. Here we compare only meandering walk data, collected by Sara Vacek, a wildlife biologist at the Morris WMD. During a meandering walk, the surveyor walks through the seed mix area and records all species present along with an estimate of species abundance and distribution. Species are recorded separately if there are multiple soil moisture categories within the seed mix area. The duration of the meandering walk and area searched are also recorded. This sampling method allows for more complete species lists than plot or transect methods (McColpin et al., 2019). While estimates of abundance and distribution are recorded during the meandering walk, they are not analyzed here because data collection was not consistent. When it was not possible to identify plants to species, the plant was listed by genus.

Analysis Methods

Seed mix areas represent unique sites, so all calculations are separated by seed mix area and monitoring year. The meandering walk data were first used to calculate total species richness, native species richness, and the percent richness of native species. Floristic Quality Assessment (FQA) metrics were then calculated. These include mean C (coefficient of conservatism), native mean C, FQI (Floristic Quality Index), and adjusted FQI. These values are all based on coefficients of conservatism, which are values given to species based on their tolerance of habitat degradation. They range from 0 to 10; non-native species are assigned a value of 0 and native species dependent on remnant habitats receive high C values. The C values used here were based on the Universal FQA 2017 update of the Dakotas, Iowa, and MN Wetland FQA lists.

The mean C value is the average coefficient of conservatism for all species in the seed mix area. The native mean C value is the average coefficient of conservatism for all native species in the seed mix area. This value excludes non-native species. High mean C values and native mean C values (closer to 10) reflect areas with more conservative species. An area with very low mean C values is likely highly disturbed or invaded by non-native species (Freyman et al., 2016). When plants were known only to genus, they were excluded from C value calculations.

FQI is $\bar{C}\sqrt{n}$. $\bar{C}$ is mean C, and *n* is total species richness. Adjusted FQI is $100\left(\frac{\bar{C}_n}{10}\right)\left(\frac{\sqrt{n_n}}{\sqrt{n_t}}\right)$. $\bar{C}_n$ is native mean C, n_n is native species richness, and n_t is total species richness. FQI is often used to assess the quality of natural – or in this case reconstructed – areas. Adjusted FQI is intended for areas with considerable human disturbance (Freyman et al., 2016). Higher FQI values reflect more conservative plant communities, and FQIs of less than 20 reflect degraded plant communities (Taft et al., n.d.).

FQA metrics allow comparisons in habitat quality between sites or over time. Different FQA metrics provide slightly different information, and thus calculating multiple FQA metrics provides more nuanced comparisons of plant community integrity. Two sites with the same FQI, for example, may have different mean C values. Mean C values are generally less dependent on area and are more suited to comparing sites of different sizes. Species richness alone is often highly dependent on area and may be especially misleading if the site includes many non-native species. FQI values are similarly dependent on area but are less impacted by species richness variance. Mean C values, richness values, and FQI values can also remain stable even as physiognomic classes change (Taft et al., n.d.), so grass:forb ratios are also calculated here.

Mean C, mean native C, FQI, and adjusted FQI were graphed by reconstruction in age order to monitor differences as reconstructions matured. The FQA metrics were also graphed by age of reconstruction on scatterplots to check for linear trends in the data; only mean C is shown here. The oldest reconstruction (Loen WPA 2009 seed mix area) was planted with a less diverse seed mix with a high grass:forb ratio, and so was excluded; the data were graphed in another scatterplot to check for linear trends in the remaining data.

The grass:forb ratio was calculated by species richness of each physiognomy, both for the seed mix and the meandering walk sampling. If a plant was known only to genus and the physiognomy was variable within the genus it was excluded from calculations. Sedges and rushes were included in the grass category, though other physiognomies were excluded from calculations. Early reconstructions often used primarily grass, because it was easily available, though a reconstruction should include a high forb component to reflect prairie diversity more accurately (Schramm, 2016; Smith, pers. comm.).

The species planted in each seed mix were compared with the species found during the meandering walk sampling to calculate the percent and number of seeded species that established. Seeding rate was not included in this analysis. There were no consistent abundance data for comparison and many species were seeded in trace amounts. Seeded species that were never encountered in the meandering walk surveys and a selection of seeded species that frequently failed to establish are listed in the appendix. Non-native species are also noted, especially those encountered most frequently in surveys.

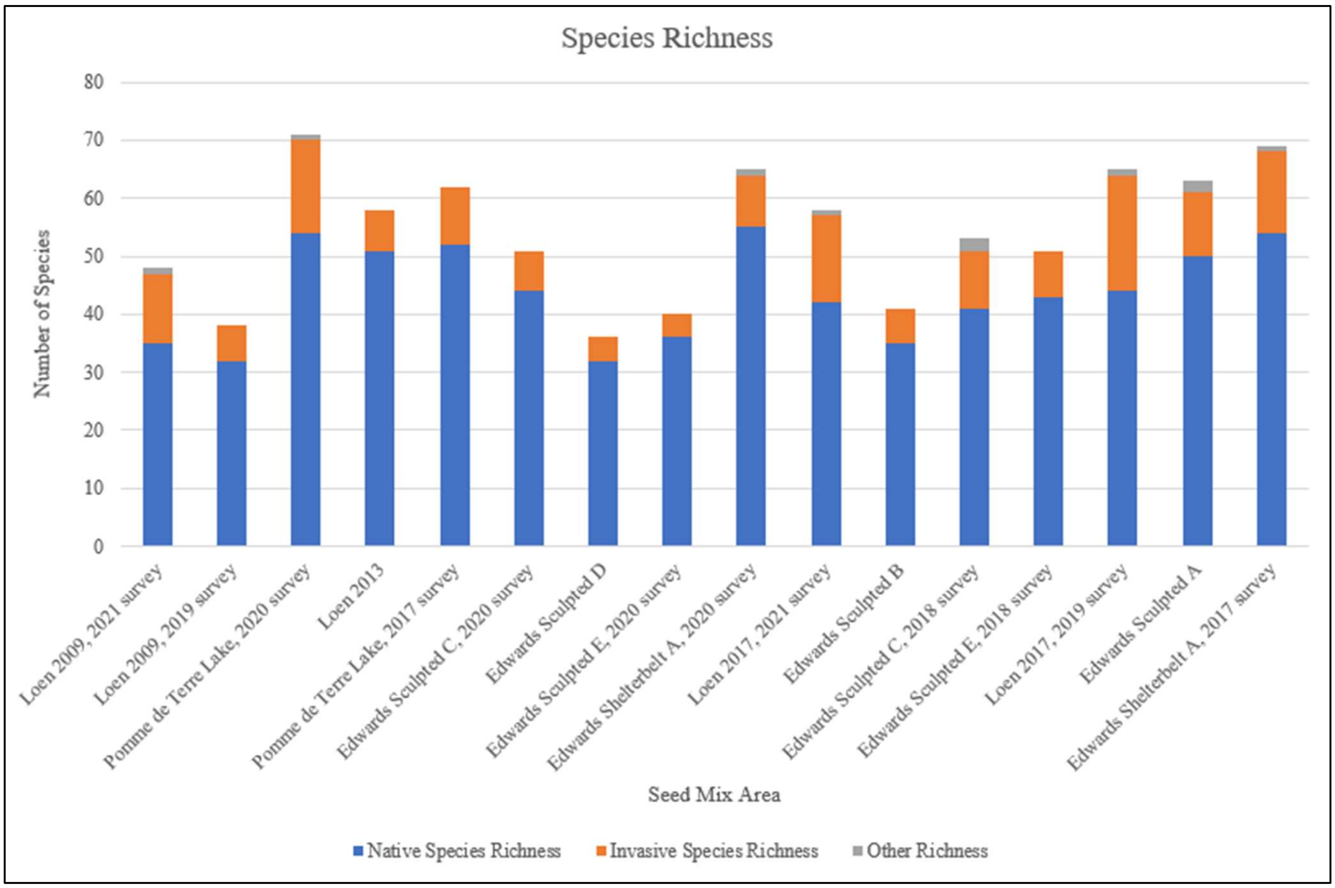
Results

Species Richness

Native species richness ranged between 32 species at the Loen WPA 2009 seed mix area during the 2019 survey and 55 species at the Edwards WPA Shelterbelt A seed mix area surveyed in 2020. Non-native species richness ranged between 4 and 20 species. Both the Edwards WPA Sculpted Seeding D seed mix area and the Edwards WPA Sculpted Seeding E seed mix area had four non-native species when surveyed in 2020. The Loen WPA 2017 seed mix area had 20 non-native species when surveyed in 2019. The 2020 survey of the Edwards WPA Sculpted Seeding D seed mix area had the lowest total species richness, of 36 species. The highest total species richness was found at the Pomme de Terre Lake WPA seed mix area when surveyed in 2020. That site had 71 total species (Appendix Table 1, Figure 1).

Figure 1: Species Richness

The number of native species, invasive species, and species known only to genus (listed as other richness). The figure is organized by the age of reconstruction, with the oldest sites on the left.

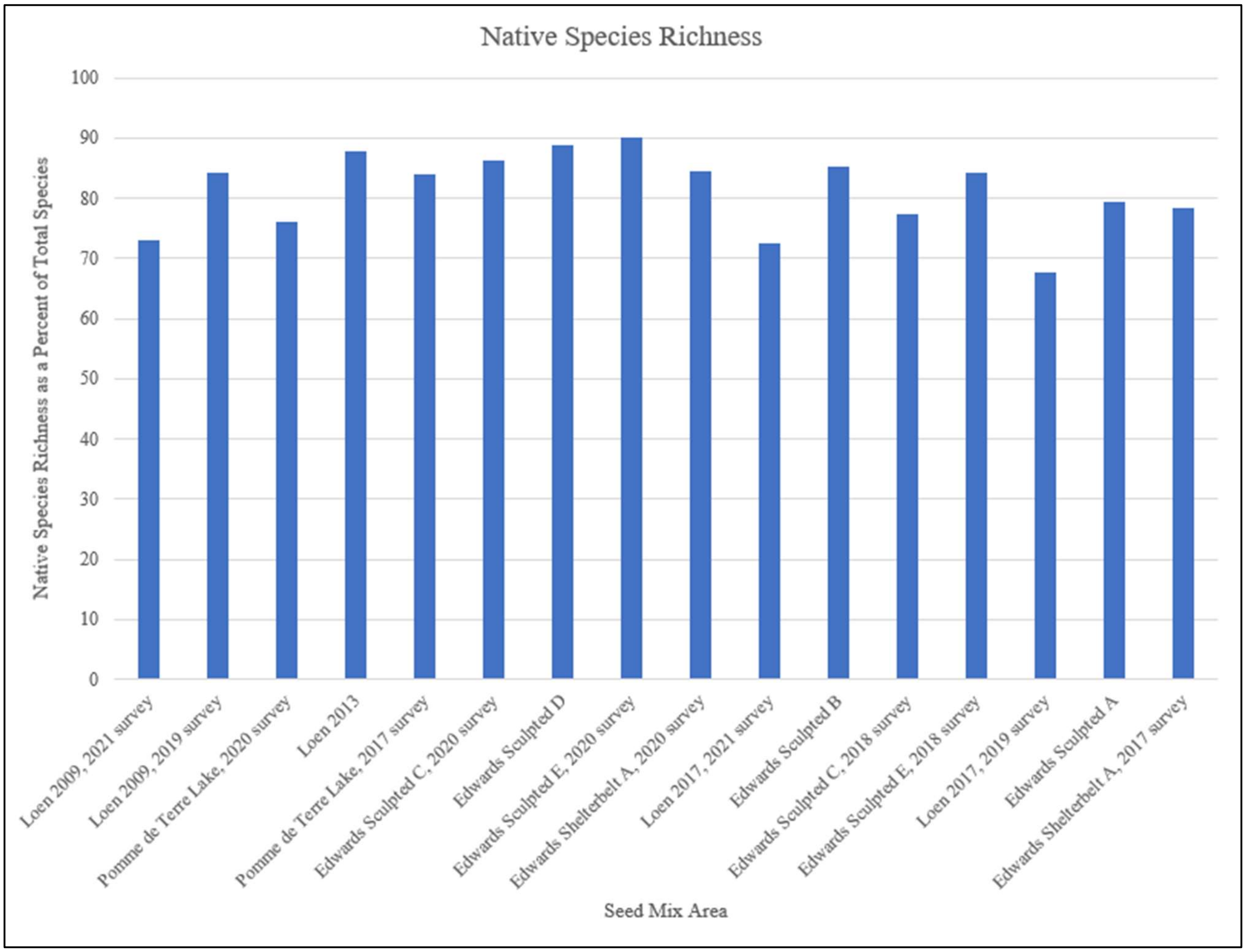


Percent Native Species

The Loen WPA 2017 seed mix area had the lowest percent native species when surveyed in 2019, at 68%. The 2020 survey of the Edwards WPA Sculpted Seeding E seed mix area had the highest percent of native species, at 90% (Appendix Table 2, Figure 2).

Figure 2: Native Species Richness as a Percent of Total Species

Native species richness expressed as a percent of total species richness. Plants identified only to genus were excluded from total species richness. The figure is organized by the age of reconstruction, with the oldest sites on the left.



<u>Mean C Values</u>

Mean C values ranged from 2.9 to 4.8, with the lowest value from the 2019 survey of the Loen WPA 2017 seed mix area and the highest value found both at the Edwards WPA Sculpted Seeding D seed mix area surveyed in 2020 and the Loen WPA 2013 seed mix area surveyed in 2021. Native mean C values ranged from 4.1 to 5.5. The 2021 survey of the Loen WPA 2017 seed mix area had the lowest value, and the 2021 survey of the Loen WPA 2013 seed mix area had the highest value. Seeded mean C values ranged from 5.6 in the Loen WPA 2009 seed mix to 6.3 in the Edwards WPA Sculpted Seeding E seed mix (Appendix Table 3, Figures 3, 4, 5).

Figure 3: Mean C Values

Mean C values from meandering walk surveys, native mean C values from meandering walk surveys, and mean C values of planted seed mix. Plants identified only to genus were excluded from averages. The figure is organized by the age of reconstruction, with the oldest sites on the left.

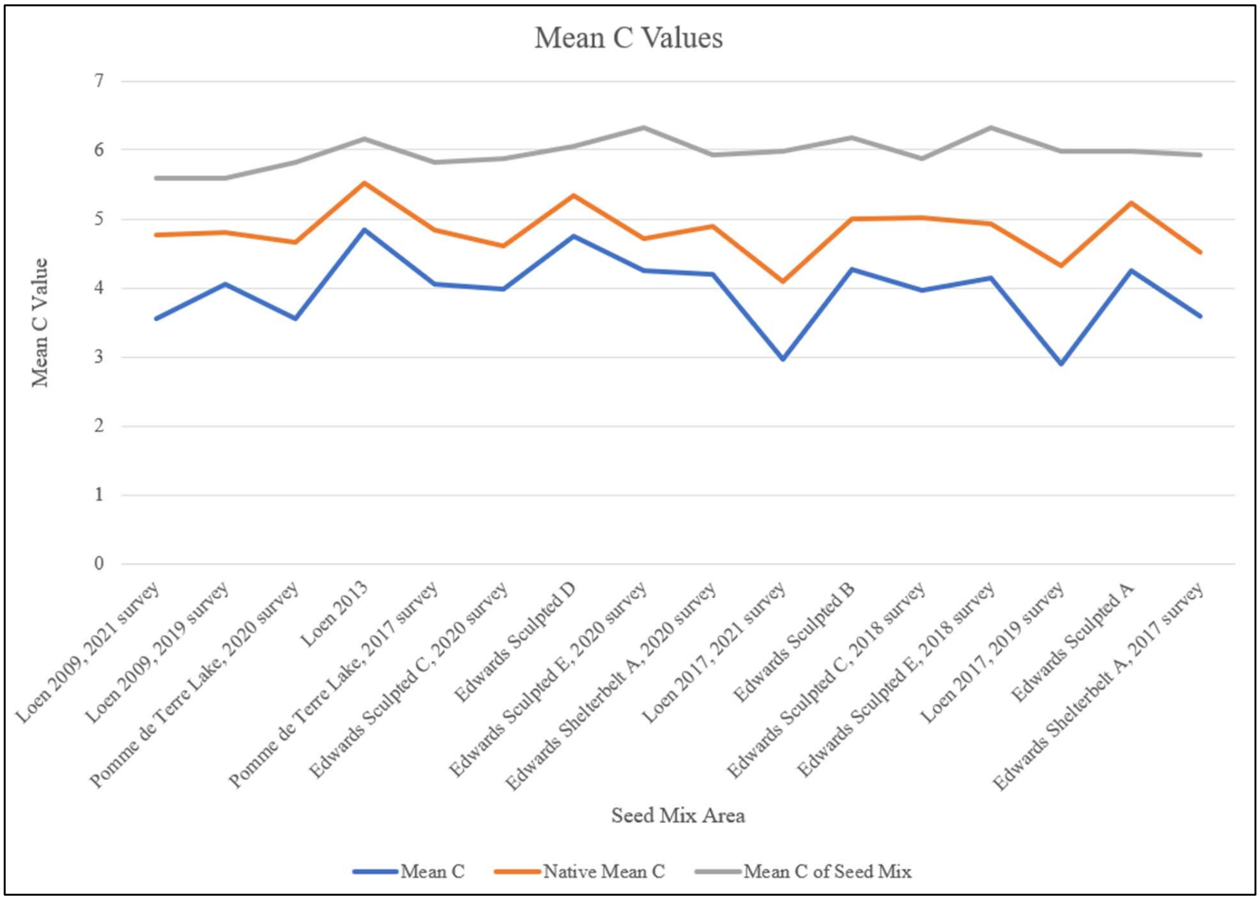
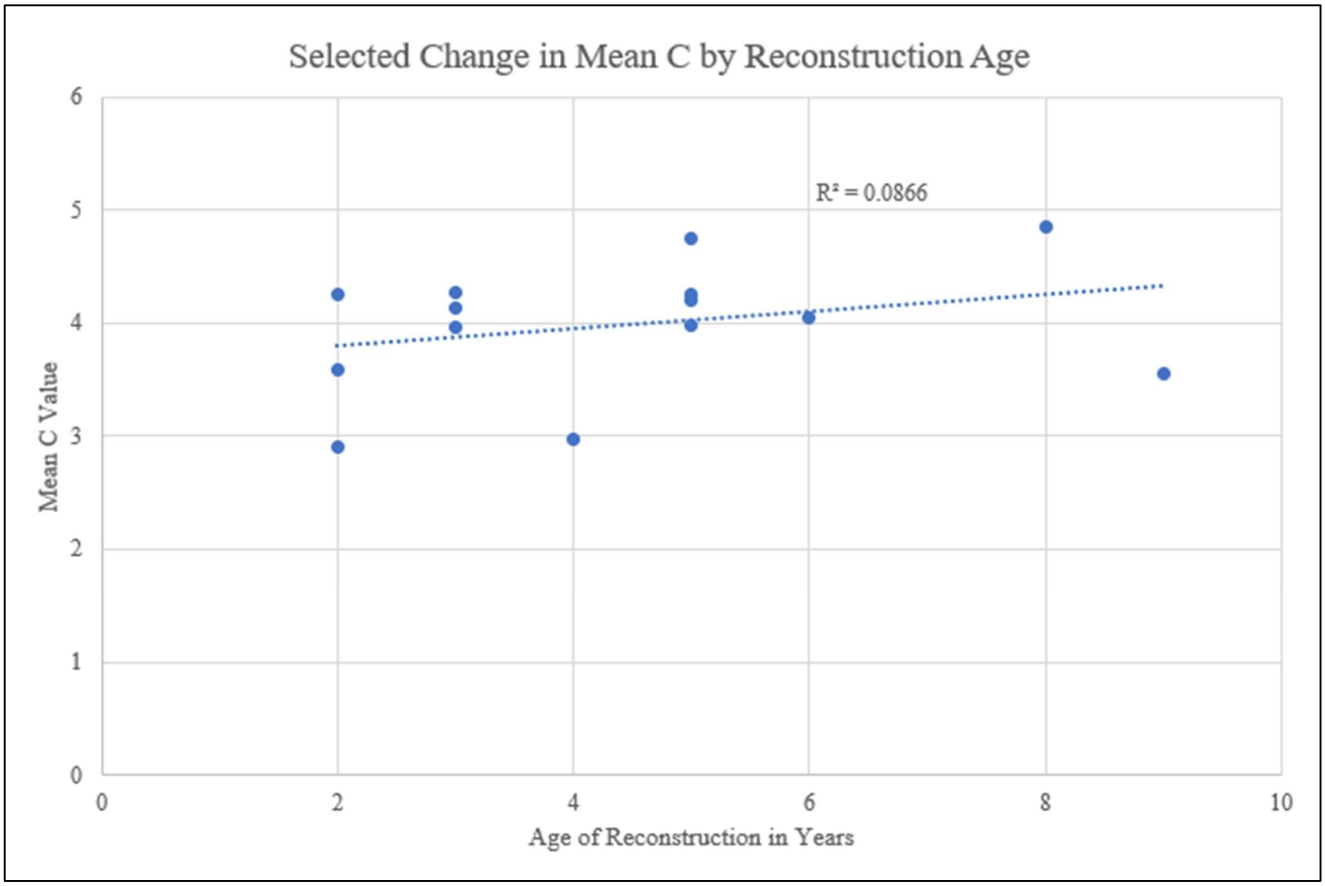


Figure 4: Change in Mean C by Reconstruction Age

Mean C values from meandering walk surveys, by the age of reconstruction. Plants identified only to genus were excluded from averages. Linear trendline shown has an R^2 value of 0.0079.

Figure 5: Selected Change in Mean C by Reconstruction Age

Mean C values from meandering walk surveys, by the age of reconstruction. Here the oldest reconstruction has been excluded as it was planted with a less diverse seed mix with a high grass:forb ratio. Plants identified only to genus were excluded from averages. Linear trendline shown has an R^2 value of 0.0866.

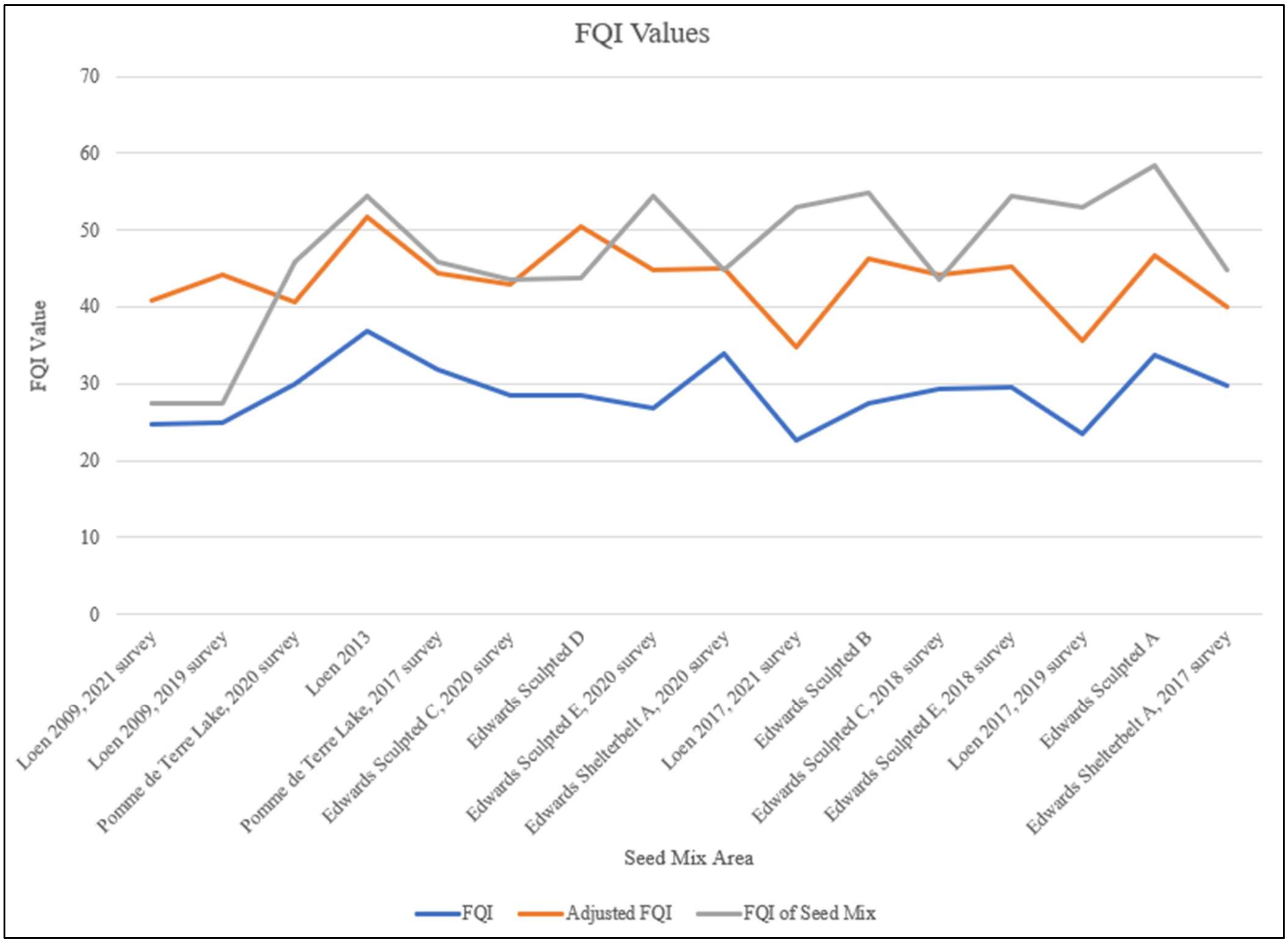


<u>FQI Values</u>

FQI values ranged from 22.58 to 36.88. The lowest value was found at the Loen WPA 2017 seed mix area surveyed in 2021. The highest value was found at the Loen WPA 2013 seed mix area surveyed in 2021. Adjusted FQI values ranged from 34.85 to 51.76. The minimum and maximum values correspond to the same sites as the un-adjusted FQI values. The lowest seeded FQI value was 27.43 in the Loen WPA 2009 seed mix. The highest value was 58.37 in the Edwards WPA Sculpted Seeding A seed mix (Appendix Table 4, Figure 6).

Figure 6: FQI Values

FQI values from meandering walk surveys, adjusted FQI values from meandering walk surveys, and FQI values of planted seed mix. Plants identified only to genus were excluded from C value calculations, though included in species richness. The figure is organized by the age of reconstruction, with the oldest sites on the left.

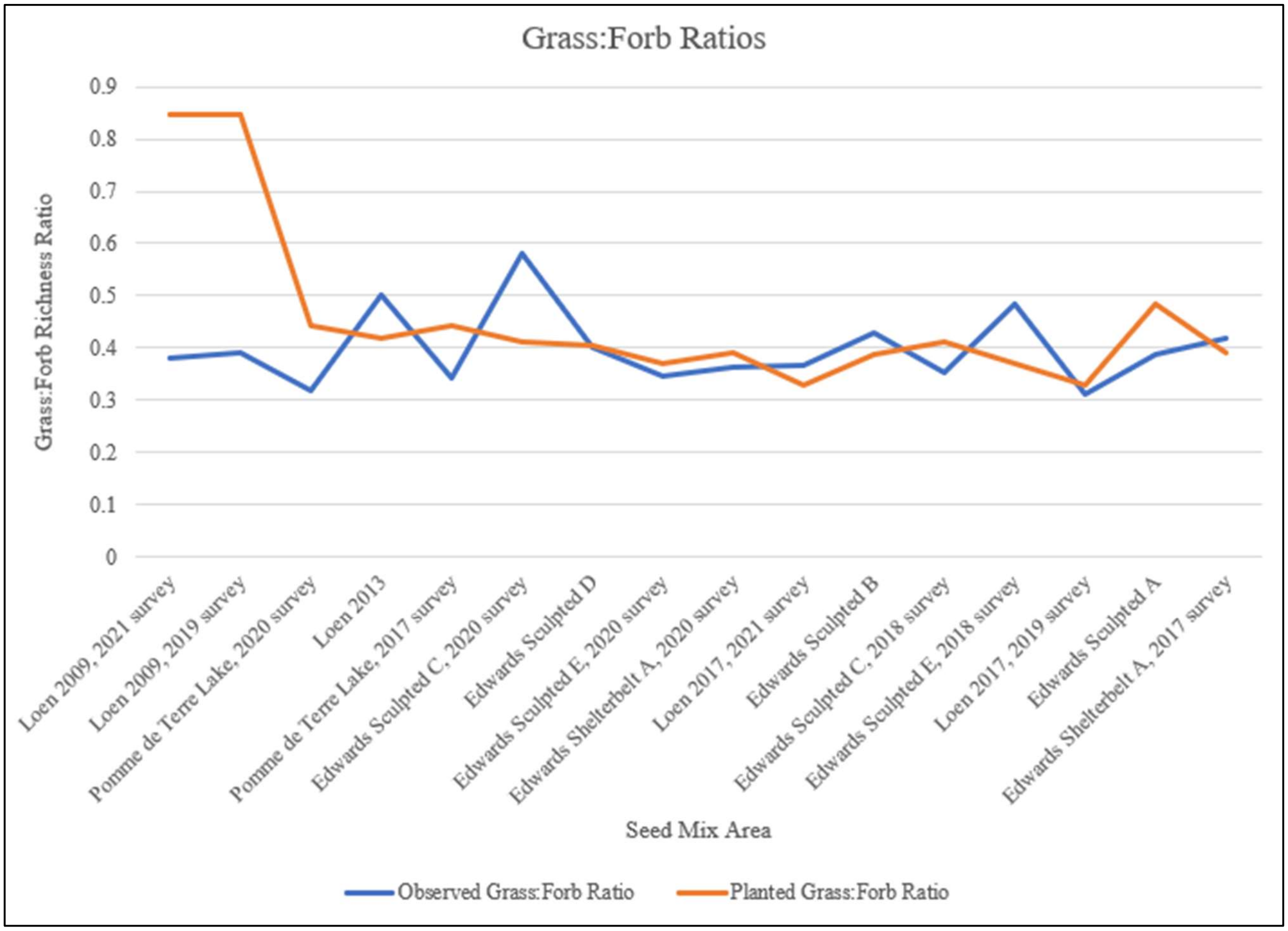


Grass:forb Ratio

The grass:forb ratio ranged from 0.31 at the 2019 survey of the Loen WPA 2017 seed mix area to 0.58 at the 2020 survey of the Edwards WPA Sculpted Seeding C seed mix area. The Loen WPA 2017 seed mix had the lowest seeded grass:forb ratio, of 0.33. The Loen 2009 WPA seed mix had the highest seeded grass:forb ratio, of 0.85 (Appendix Table 5, Figure 7).

Figure 7: Grass:Forb Ratios

The ratio of species richness of grasses, including sedges and rushes, to the species richness of forbs. Other physiognomies were excluded. The figure is organized by the age of reconstruction, with the oldest sites on the left.



Establishment of Seeded Species

The percent of the seed mix that established ranged between 38% and 70%. 38% of seeded species had established at the Edwards WPA Sculpted Seeding B seed mix area when surveyed in 2018. 70% of seeded species had established at the Edwards WPA Shelterbelt A seed mix area when surveyed in 2020. Because the seed mixes had different richness levels, the number of seeded species established is also included. The fewest seeded species established at the Loen WPA 2009 seed mix area. In 2019 15 seeded species had established; 16 seeded species were found in the 2021 survey. The most seeded species – 42 – were found in the 2021 survey of the Loen WPA 2013 seed mix area (Appendix Table 6, Figures 8, 9).

Figure 8: Percent Established

Seeded species that established as a percent of total species seeded. Total seeded species richness includes plants known only to genus. The figure is organized by the age of reconstruction, with the oldest sites on the left.

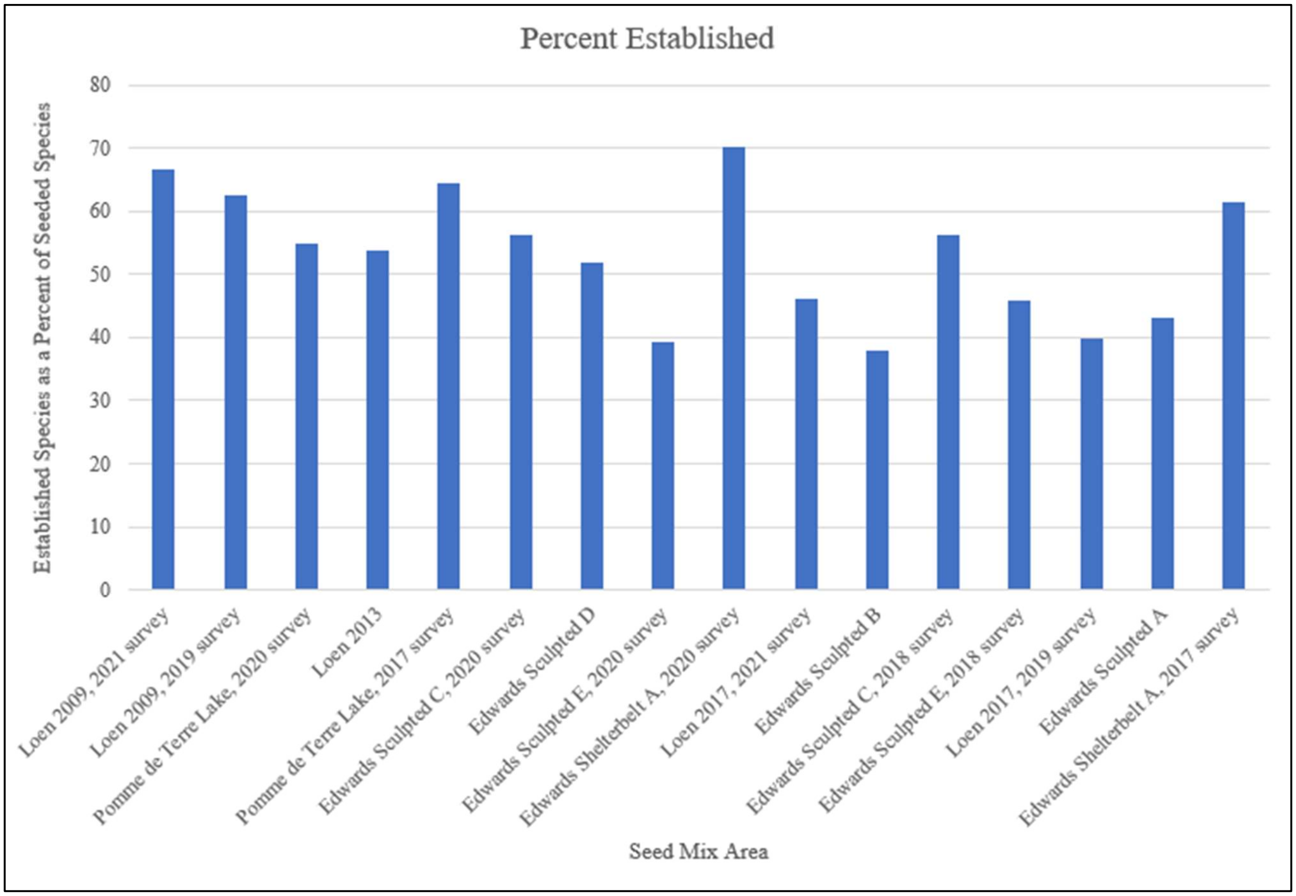
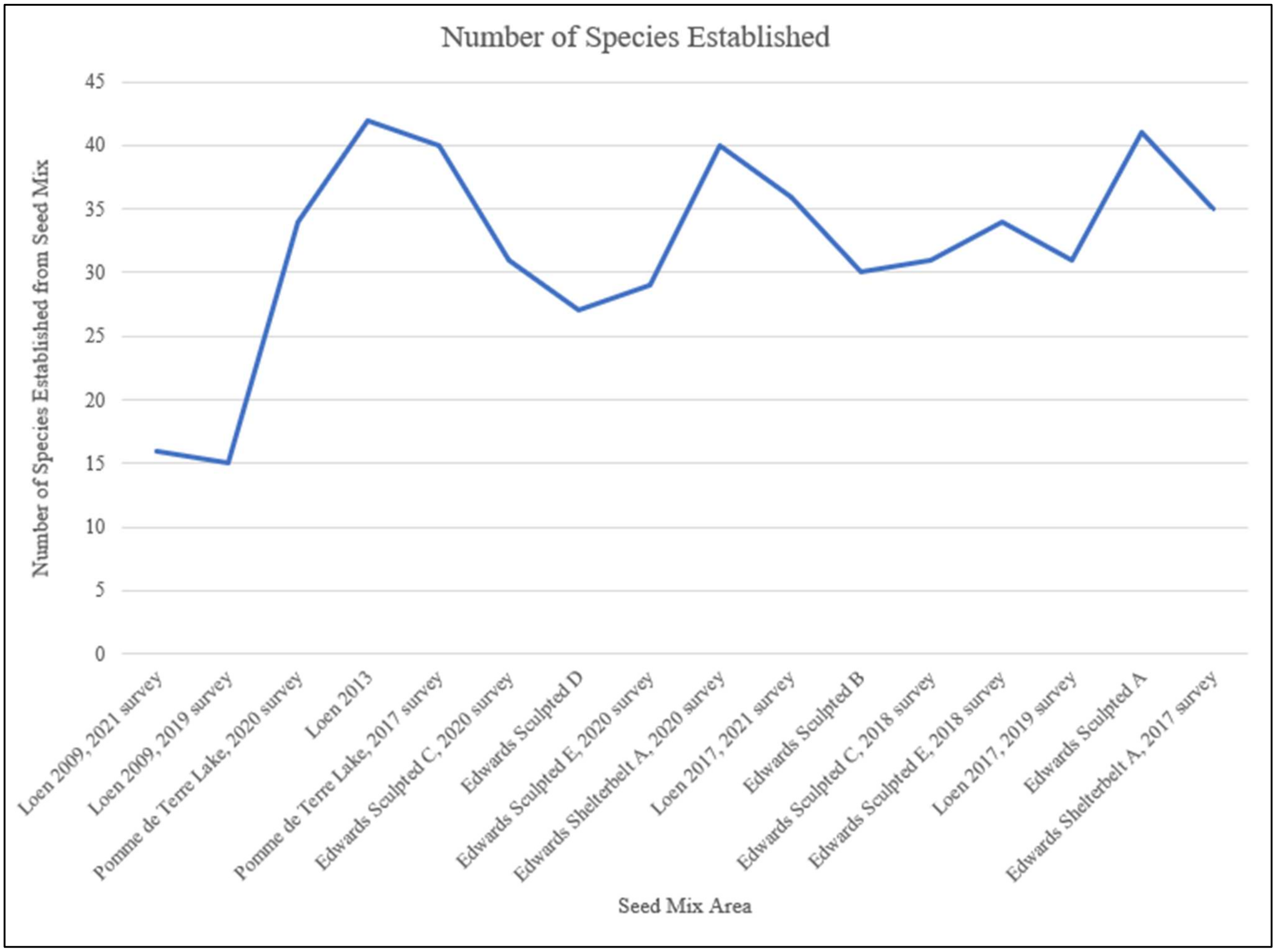


Figure 9: Number of Species Established

The number of seeded species that established. The figure is organized by the age of reconstruction, with the oldest sites on the left.



33 seeded species never established, including wild white indigo (*Baptisia alba*), prairie spiderwort (*Tradescantia occidentalis*), and culvers root (*Veronicastrum virginicum*) (Appendix Table 7). Other species, while found at some seed mix areas, frequently failed to establish when planted. These include lead plant (*Amorpha canescens*), prairie dropseed (*Sporobolus heterolepis*), and purple meadow rue (*Thalictrum dasycarpum*) (Appendix Table 8).

<u>Non-native Species</u>

All sites have non-native species, and there were at least 39 non-native species between the sites (plants known only to genus are excluded). Kentucky Bluegrass (*Poa pratensis*) was the most common non-native species, found in 15 of the 16 surveys. Canada thistle (*Cirsium arvense*) was found in 13 surveys, field sow thistle (*Sonchus arvensis*) in 12 surveys, and both sweet clover (*Melilotus officinalis*) and yellow foxtail (*Setaria pumila ssp. pumila*) in 10 surveys. Smooth brome (*Bromus inermis*) was found in 8 of the 16 surveys (Appendix Table 9). Note that these data refer to species presence or absence from a survey and not abundance within any given seed mix area.

Discussion

<u>FQA metrics</u>

Species richness shows no clear trend by age of reconstruction. At least 30 native species were found in every survey, and most surveys found over 40 native species. Importantly, non-native species never have a higher richness than native species. The percent richness of native species corroborates this finding, in that natives make up at least 68% of every survey. Most surveys had 70% to 80% natives, though there was no clear trend based on the age of reconstruction.

Mean C values were clustered around 4, with a difference of less than two between the highest and lowest values. As assumed, excluding non-native species (assigned C values of 0) raised the value of the mean native C. These values also showed little difference between sites and were clustered around 5. Excluding non-native species raised native mean C by less than two. This suggests the plant communities are missing highly conservative species, instead of a scenario where a highly conservative plant community was being invaded by non-native species. This is corroborated by the mean C of the seed mix, which represents the ideal mean C value should all planted species establish and no non-native species invade. The only way that a site could rise above the seeded mean C or FQI value is if native species introduced themselves, especially native species with high C values. This is unlikely, as species assigned high C values are restricted to remnant habitats. Ruderal species, that quickly introduce themselves, are assigned lower C values because of their capacity to spread to disturbed habitats. The seed mix mean C values were clustered around 6. Plants identified only to genus in both meandering walk and seed mix data were excluded from mean C value calculations, which decreases the accuracy of these comparisons. The scatterplots of mean C by reconstruction age show no clear trends as denoted by the high variance in the data and the very small R^2 values, even when the Loen WPA 2009 seed mix area was excluded.

Taft and colleagues (n.d.) note that reconstructions rarely have FQI values above 35, especially when they are relatively young. The FQI values here generally fell in Taft and colleagues' (n.d.) range between degraded ecosystems (20) and intensive reconstructions (35). Seed mix FQI values were usually high, which provides a basis for diverse reconstructions if conservative species establish and non-native encroachment can be mitigated. The Loen WPA 2009 seed mix area is unique in having seed mix and meandering walk FQI values within three digits of one another (seed mix FQI of 27.43, FQI of 24.98 in 2019, FQI of 24.62 in 2021). Though the mean C values do not set this site apart from others, the high FQI values compared to the seed mix FQI suggest that species richness is higher than expected at this site. This site was seeded with 24 native species but has 35 native species as of 2021. Unseeded native species include candle anemone (*Anemone cylindrica*), stiff sunflower (*Helianthus pauciflorus ssp. subrhomboideus*), and purple coneflower (*Echinacea angustifolia*), some of which were planted in the adjacent Loen 2013 seed mix area. This is an instance where calculating multiple FQA metrics instead of just mean C has provided a more complete understanding of the seed mix area.

The physiognomy of what was planted and what established was generally comparable. The only exception was the Loen WPA 2009 seed mix area, which was planted with a high richness of grasses.

These data do not show clear trends as reconstructions mature, though it should be noted that this monitoring was not designed to test hypothesizes about changes in reconstruction over time. This analysis had a very low sample size, especially of older reconstructions, and the oldest reconstruction was planted with a less diverse seed mix. Instead of analyzing the maturation of reconstructions, meandering walk data and FQA metrics are instead helpful to direct adaptive management actions to sites most in need, especially when scarce resources do not allow management of every site. For example, the high non-native species richness, low percent of natives, low mean C, and low FQI at the Loen 2017 seed mix area indicates that additional monitoring or management is needed at that site.

<u>Seed Mix Data</u>

Though between 38% and 70% of seeded species established, that range of values is misleading because of different seed mix richness levels and a number of sources of error. First, when plants were known to only genus in the seed mix or meandering walk data, they were assumed to correlate to species of the same genus in the other data set. This assumes that species established from planted seed and not from other sources, which may not be accurate. Secondly, the percent established was calculated based on seeded richness, which included plants known only to genus. Some seed mixes included, for example, *Liatris spp.* as well as seeds known to be specifically *Liatris punctata*. It is possible the unknown *Liatris sp.* was also *Liatris punctata*, which would change the species richness. The percent established reported here would then be lower than accurate. The percent established varied between surveys in the same seed mix area.

While this could suggest that species were continuing to establish or being lost between surveys, it may also represent a difference in survey effort, which was not considered in this analysis. Finally, FQA metrics are designed for species-level data; this monitoring protocol will produce the most conclusive information if plants in both seed mixes and meandering walks can be identified to species.

To avoid the error associated with percent established, the number of seeded species that established is also included. The fewest number of species established in the Loen WPA 2009 seed mix area, which also had the least diverse seed mix. The most seeded species established in the Loen WPA 2013 seed mix area, which actually had a lower percent established than the Loen WPA 2009 seed mix area due to the much higher seed mix richness in the Loen WPA 2013 seed mix. Seed mixes with higher richness are worthwhile for establishing greater species diversity, even if not all species establish. As many species as can be afforded or sourced should be planted to establish a diverse plant community. However, this analysis does not consider abundance. If obtaining a highly diverse seed mix comes at the cost of a low seeding rate, it is possible that some species will establish in low enough abundance that the species may become extirpated. An analysis of both seeding rates and current abundance is necessary to balance seed mix richness and seeding rate. In the future it may also be useful to analyze the number of native, unseeded species established, especially if all plants are identified to species. This may provide additional context to the percent and number of seeded species that establish, and the importance of seeding in matrixes that prevent the spread of native plants.

Seeded species that always or often failed to establish also present opportunities for future research. While it is beyond the scope of this analysis, analysis of abiotic or biotic factors in reconstruction sites may explain why some species never establish or may suggest conditions that promoted establishment of difficult species. Different seed mix areas were also planted with different seeding methods, which may have affected establishment.

Conclusions

While the non-native species list may provide a list of possible threats to reconstruction, this is likely the analysis that is most hindered by the lack of abundance and distribution data. All of the calculations here are based on richness and presence/absence data, which gives equal weight to all species. Some non-native species may be more of a problem than reflected in these data because of high abundance. Collecting abundance and distribution data consistently during meandering walks would help, though the best data would be collected by the nested plot survey method intended to be paired with meandering walks in the PRI framework.

Due to the extra time commitment required for nested plot surveys, the PRI framework recommends the nested plot surveys less frequently (every 3 to 5 years), and the meandering walk surveys annually (Taft et al., n.d.; McColpin et al., 2019). If possible, collecting meandering walk data annually in these Morris WMD reconstructions would provide more context for future analyses. More frequent data collection would allow statistical analyses of FQA metrics, decrease sampling error, and provide context on the influence of sampling effort.

The three Morris WMD prairie planting units analyzed here support dozens of native plant species, albeit not many highly conservative species. The success of species establishment suggests that future reconstructions be planted with as diverse a seed mix as possible. Of course, reconstructions must also be managed, for example to fight non-native encroachment. Monitoring is intended to identify such management needs. The PRI meandering walk monitoring protocol used here would be improved by more frequent monitoring, collection of abundance and distribution data, and consistent plant identification to species, but still results in

FQA metrics sensitive enough to identify sites in need of priority management, like the Loen 2017 seed mix area.

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Appendix

Table 1: Native and non-native species richness

Planting Unit	Seed Mix Area	Monitoring Year	Native Species Richness	Non-native Species Richness	Total Species Richness*
Loen WPA	2009	2019	32	6	38
Loen WPA	2009	2021	35	12	48
Loen WPA	2013	2021	51	7	58
Loen WPA	2017	2019	44	20	65
Loen WPA	2017	2021	42	15	58
Edwards WPA	Sculpted Seeding A	2018	50	11	63
Edwards WPA	Sculpted Seeding B	2018	35	6	41
Edwards WPA	Sculpted Seeding C	2018	41	10	53
Edwards WPA	Sculpted Seeding C	2020	44	7	51
Edwards WPA	Sculpted Seeding D	2020	32	4	36
Edwards WPA	Sculpted Seeding E	2018	43	8	51
Edwards WPA	Sculpted Seeding E	2020	36	4	40
Edwards WPA	Shelterbelt A	2017	54	14	69
Edwards WPA	Shelterbelt A	2020	55	9	65
Pomme de Terre Lake WPA		2017	52	10	62
Pomme de Terre Lake WPA		2020	54	16	71

*If a plant was known to only genus in which there are both native and non-native species, the plant was included in total richness, but not in native or non-native species richness.

Table 2: Percent native species richness

Planting Unit	Seed Mix Area	Monitoring Year	Percent Natives
Loen WPA	2009	2019	84
Loen WPA	2009	2021	73
Loen WPA	2013	2021	88
Loen WPA	2017	2019	68
Loen WPA	2017	2021	72
Edwards WPA	Sculpted Seeding A	2018	79
Edwards WPA	Sculpted Seeding B	2018	85
Edwards WPA	Sculpted Seeding C	2018	77
Edwards WPA	Sculpted Seeding C	2020	86
Edwards WPA	Sculpted Seeding D	2020	89
Edwards WPA	Sculpted Seeding E	2018	84
Edwards WPA	Sculpted Seeding E	2020	90
Edwards WPA	Shelterbelt A	2017	78
Edwards WPA	Shelterbelt A	2020	85
Pomme de Terre Lake WPA		2017	84
Pomme de Terre Lake WPA		2020	76

Table 3: Mean C values

Planting Unit	Seed Mix Area	Monitoring Year	Mean C	Native Mean C	Seeded Mean C
Loen WPA	2009	2019	4.1	4.8	5.6
Loen WPA	2009	2021	3.6	4.8	5.6
Loen WPA	2013	2021	4.8	5.5	6.2
Loen WPA	2017	2019	2.9	4.3	6.0
Loen WPA	2017	2021	3.0	4.1	6.0
Edwards WPA	Sculpted Seeding A	2018	4.3	5.2	6.0
Edwards WPA	Sculpted Seeding B	2018	4.3	5.0	6.2
Edwards WPA	Sculpted Seeding C	2018	4.0	5.0	5.9
Edwards WPA	Sculpted Seeding C	2020	4.0	4.6	5.9
Edwards WPA	Sculpted Seeding D	2020	4.8	5.3	6.1
Edwards WPA	Sculpted Seeding E	2018	4.1	4.9	6.3
Edwards WPA	Sculpted Seeding E	2020	4.3	4.7	6.3
Edwards WPA	Shelterbelt A	2017	3.6	4.5	5.9
Edwards WPA	Shelterbelt A	2020	4.2	4.9	5.9
Pomme de Terre Lake WPA		2017	4.0	4.8	5.8
Pomme de Terre Lake WPA		2020	3.6	4.7	5.8

Table 4: FQI values

Planting Unit	Seed Mix Area	Monitoring Year	FQI	Adjusted FQI	Seeded FQI
Loen WPA	2009	2019	24.98	44.16	27.43
Loen WPA	2009	2021	24.62	40.74	27.43
Loen WPA	2013	2021	36.88	51.76	54.40
Loen WPA	2017	2019	23.39	35.52	52.86
Loen WPA	2017	2021	22.58	34.85	52.86
Edwards WPA	Sculpted Seeding A	2018	33.77	46.59	58.37
Edwards WPA	Sculpted Seeding B	2018	27.33	46.20	54.96
Edwards WPA	Sculpted Seeding C	2018	29.41	44.19	43.55
Edwards WPA	Sculpted Seeding C	2020	28.43	42.85	43.55
Edwards WPA	Sculpted Seeding D	2020	28.50	50.38	43.68
Edwards WPA	Sculpted Seeding E	2018	29.57	45.26	54.35
Edwards WPA	Sculpted Seeding E	2020	26.88	44.80	54.35
Edwards WPA	Shelterbelt A	2017	29.81	39.97	44.77
Edwards WPA	Shelterbelt A	2020	33.89	44.99	44.77
Pomme de Terre Lake WPA		2017	31.88	44.35	45.79
Pomme de Terre Lake WPA		2020	29.99	40.59	45.79

Table 5: Grass:forb ratio

Planting Unit	Seed Mix Area	Monitoring Year	Meandering Walk Grass:forb Ratio	Seeded Grass:forb Ratio
Loen WPA	2009	2019	0.39	0.85
Loen WPA	2009	2021	0.38	0.85
Loen WPA	2013	2021	0.50	0.42
Loen WPA	2017	2019	0.31	0.33
Loen WPA	2017	2021	0.37	0.33
Edwards WPA	Sculpted Seeding A	2018	0.39	0.48
Edwards WPA	Sculpted Seeding B	2018	0.43	0.39
Edwards WPA	Sculpted Seeding C	2018	0.35	0.41
Edwards WPA	Sculpted Seeding C	2020	0.58	0.41
Edwards WPA	Sculpted Seeding D	2020	0.40	0.41
Edwards WPA	Sculpted Seeding E	2018	0.48	0.37
Edwards WPA	Sculpted Seeding E	2020	0.34	0.37
Edwards WPA	Shelterbelt A	2017	0.42	0.39
Edwards WPA	Shelterbelt A	2020	0.36	0.39
Pomme de Terre Lake WPA		2017	0.34	0.44
Pomme de Terre Lake WPA		2020	0.32	0.44

Table 6: Percent and number of seeded, established species

Planting Unit	Seed Mix Area	Monitoring Year	Percent Established	Number of Species Planted*	Number of Species Established
Loen WPA	2009	2019	63	24	15
Loen WPA	2009	2021	67	24	16
Loen WPA	2013	2021	54	78	42
Loen WPA	2017	2019	40	78	31
Loen WPA	2017	2021	46	78	36
Edwards WPA	Sculpted Seeding A	2018	43	95	41
Edwards WPA	Sculpted Seeding B	2018	38	79	30
Edwards WPA	Sculpted Seeding C	2018	56	55	31
Edwards WPA	Sculpted Seeding C	2020	56	55	31
Edwards WPA	Sculpted Seeding D	2020	52	52	27
Edwards WPA	Sculpted Seeding E	2018	46	74	34
Edwards WPA	Sculpted Seeding E	2020	39	74	29
Edwards WPA	Shelterbelt A	2017	61	57	35
Edwards WPA	Shelterbelt A	2020	70	57	40
Pomme de Terre Lake WPA		2017	65	62	40
Pomme de Terre Lake WPA		2020	55	62	34

*Includes plants known only to genus.

Table 7: Seeded species that failed to establish

Species Name	Common Name	C Value
Acorus americanus	sweet flag	0
Agastache scrophulariifolia	purple giant hyssop	9
Alisma subcordatum	common water plantain	2
Allium canadense var. canadense	wild onion	8
Allium cernuum	nodding onion	8
Amorpha nana	dwarf wild indigo	9
Astragalus crassicarpus var. crassicarpus	groundplum	7
Baptisia alba	wild white indigo	6
Baptisia bracteata	cream wild indigo	7
Beckmannia syzigachne	american sloughgrass	1
Calamovilfa longifolia	prairie sandreed	5
Eutrochium maculatum var. bruneri	spotted joe-pye weed	9
Gentiana puberulenta	downy gentian	10
Glyceria grandis	tall mannagrass	4
Glyceria striata	fowl mannagrass	6
Heuchera richardsonii	alumroot	8
Iris versicolor	blue flag iris	4
Koeleria pyramidata	junegrass	7
Leersia oryzoides	rice cutgrass	2
Linum spp.	flax species	5-7
Lygodesmia juncea	skeletonweed	2
Mimulus ringens	alleghany monkeyflower	6
Pedicularis canadensis	wood betony	10
Physalis spp.	groundcherry species	0-8
Rosa blanda	smooth wild rose	8
Scirpus cyperinus	woolgrass	10
Scolochloa festucacea	sprangletop	6
Scutellaria lateriflora	maddog skullcap	6
Sisyrinchium spp.		8-10
Sparganium eurycarpum	giant burreed	4
Sphenopholis obtusata	prairie wedgegrass	7
Tradescantia occidentalis	prairie spiderwort	5
Veronicastrum virginicum	culvers root	10

Table 8: Selected species that frequently failed to establish

Species Name	Common Name	C Value
Amorpha canescens	lead plant	9
Sporobolus heterolepis	prairie dropseed	10
Thalictrum dasycarpum	purple meadow rue	7
Sporobolus compositus	rough dropseed	4
Solidago speciosa	showy-wand goldenrod	10
Potentilla arguta	tall cinquefoil	8
Calamagrostis canadensis	bluejoint	5

Table 9: Non-native species*

Scientific Name	Common Name	Seed Mix Area
Agrostis stolonifera	redtop	Edwards Sculpted A Edwards Sculpted B Edwards Sculpted C, 2018 survey Edwards Sculpted C, 2020 survey Edwards Shelterbelt A, 2020 survey
Arctium minus	common burdock	Loen 2017, 2019 survey Loen 2017, 2021 survey
Artemisia absinthium	wormwood	Loen 2017, 2019 survey Loen 2017, 2021 survey Edwards Shelterbelt A, 2020 survey Pomme de Terre Lake, 2017 survey Pomme de Terre Lake, 2020 survey
Berteroa incana	hoary false alyssum	Pomme de Terre Lake, 2020 survey
Bromus inermis	smooth brome	Loen 2009, 2019 survey Loen 2009, 2021 survey Loen 2013 Loen 2017, 2021 survey Edwards Sculpted C, 2018 survey Edwards Shelterbelt A, 2017 survey Pomme de Terre Lake, 2017 survey Pomme de Terre Lake, 2020 survey
Bromus tectorum	downy brome	Loen 2017, 2019 survey
Cannabis sativa	hemp	Loen 2017, 2019 survey
Carduus acanthoides	plumeless thistle	Loen 2017, 2019 survey Loen 2017, 2021 survey Edwards Sculpted A Edwards Sculpted C, 2018 survey Edwards Sculpted C, 2020 survey
Carduus nutans	nodding thistle	Edwards Shelterbelt A, 2017 survey
Cirsium arvense	Canada thistle	Loen 2009, 2021 survey Loen 2013 Loen 2017, 2019 survey

		Loen 2017, 2021 survey Edwards Sculpted A Edwards Sculpted B Edwards Sculpted C, 2018 survey Edwards Sculpted C, 2020 survey Edwards Sculpted D Edwards Sculpted E, 2018 survey Edwards Sculpted E, 2020 survey Edwards Shelterbelt A, 2017 survey Edwards Shelterbelt A, 2020 survey
Cirsium vulgare	bull thistle	Loen 2017, 2019 survey Edwards Shelterbelt A, 2017 survey
Dianthus armeria	deptford pink	Pomme de Terre Lake, 2017 survey Pomme de Terre Lake, 2020 survey
Elymus repens	quackgrass	Loen 2009, 2021 survey Loen 2017, 2019 survey Edwards Sculpted B Edwards Sculpted E, 2018 survey Edwards Shelterbelt A, 2017 survey Edwards Shelterbelt A, 2020 survey Pomme de Terre Lake, 2020 survey
Euphorbia esula	leafy spurge	Loen 2017, 2019 survey Edwards Shelterbelt A, 2020 survey
Glechoma hederacea	ground ivy	Loen 2017, 2019 survey Loen 2017, 2021 survey
Lotus corniculatus	birds-foot trefoil	Pomme de Terre Lake, 2020 survey
Medicago lupulina	black medick	Loen 2013 Loen 2017, 2021 survey Edwards Sculpted A
Medicago sativa	alfalfa	Loen 2009, 2019 survey Loen 2009, 2021 survey Loen 2017, 2019 survey Loen 2017, 2021 survey
Melilotus officinalis	sweet clover	Loen 2009, 2021 survey Loen 2013 Loen 2017, 2021 survey Edwards Sculpted A Edwards Sculpted C, 2018 survey Edwards Sculpted E, 2018 survey Edwards Shelterbelt A, 2017 survey Edwards Shelterbelt A, 2020 survey Pomme de Terre Lake, 2017 survey Pomme de Terre Lake, 2020 survey
Morus alba	white mulberry	Loen 2009, 2019 survey
Nepeta cataria	catnip	Loen 2017, 2019 survey Loen 2017, 2021 survey

Phleum pratense	timothy	Edwards Sculpted C, 2020 survey Edwards Shelterbelt A, 2017 survey Pomme de Terre Lake, 2020 survey
Poa compressa	Canada bluegrass	Edwards Shelterbelt A, 2017 survey Pomme de Terre Lake, 2017 survey
Poa pratensis	Kentucky bluegrass	Loen 2009, 2019 survey Loen 2013 Loen 2017, 2019 survey Loen 2017, 2021 survey Edwards Sculpted A Edwards Sculpted B Edwards Sculpted C, 2018 survey Edwards Sculpted C, 2020 survey Edwards Sculpted D Edwards Sculpted E, 2018 survey Edwards Sculpted E, 2020 survey Edwards Shelterbelt A, 2017 survey Edwards Shelterbelt A, 2020 survey Pomme de Terre Lake, 2017 survey Pomme de Terre Lake, 2020 survey
Polygonum convolvulus	wild buckwheat	Pomme de Terre Lake, 2020 survey
Rhamnus cathartica	common buckthorn	Loen 2009, 2019 survey Loen 2009, 2021 survey Pomme de Terre Lake, 2020 survey
Rumex crispus	curly dock	Loen 2009, 2019 survey Loen 2017, 2019 survey
Securigera varia	crown vetch	Edwards Sculpted D Edwards Sculpted E, 2018 survey Edwards Sculpted E, 2020 survey Pomme de Terre Lake, 2020 survey
Setaria pumila ssp. pumila	yellow foxtail	Loen 2009, 2021 survey Loen 2013 Loen 2017, 2019 survey Loen 2017, 2021 survey Edwards Sculpted A Edwards Sculpted B Edwards Sculpted C, 2018 survey Edwards Sculpted E, 2018 survey Edwards Shelterbelt A, 2017 survey Pomme de Terre Lake, 2020 survey
Setaria viridis	green foxtail	Loen 2009, 2021 survey Loen 2013 Edwards Sculpted A Edwards Sculpted C, 2018 survey Edwards Sculpted E, 2018 survey
Silene latifolia	white campion	Pomme de Terre Lake, 2017 survey

		Pomme de Terre Lake, 2020 survey
Sonchus arvensis	field sow thistle	Loen 2009, 2021 survey Loen 2017, 2019 survey Loen 2017, 2021 survey Edwards Sculpted A Edwards Sculpted B Edwards Sculpted C, 2018 survey Edwards Sculpted C, 2020 survey Edwards Sculpted D Edwards Sculpted E, 2018 survey Edwards Sculpted E, 2020 survey Edwards Shelterbelt A, 2017 survey Edwards Shelterbelt A, 2020 survey
Taraxacum officinale	common dandelion	Loen 2017, 2019 survey Loen 2017, 2021 survey Edwards Sculpted A Edwards Sculpted C, 2018 survey Edwards Shelterbelt A, 2017 survey
Thlaspi arvense	field pennycress	Pomme de Terre Lake, 2017 survey
Tragopogon dubius	goats beard	Loen 2009, 2021 survey Edwards Sculpted A
Trifolium pratense	red clover	Loen 2017, 2019 survey Edwards Shelterbelt A, 2017 survey Edwards Shelterbelt A, 2020 survey
Trifolium repens	white clover	Loen 2017, 2019 survey Edwards Shelterbelt A, 2017 survey
Ulmus pumila	Siberian elm	Loen 2009, 2021 survey Edwards Sculpted C, 2020 survey Pomme de Terre Lake, 2017 survey Pomme de Terre Lake, 2020 survey
Verbascum thapsus	common mullein	Loen 2009, 2021 survey Loen 2017, 2021 survey Pomme de Terre Lake, 2017 survey Pomme de Terre Lake, 2020 survey

*This list excludes plants known only to genus in which there are both native and non-native species.