WETLAND RESTORATION TECHNIQUES AND ASSOCIATED COSTS IN SOUTHEASTERN NORTH DAKOTA

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ABSTRACT

Degraded wetlands are a common occurrence throughout the Prairie Pothole Region of the United States. Many restoration attempts have been conducted to restore these unique ecosystems to their previous conditions. However, many restored wetlands fail to regain the appearance and functions of natural wetlands. Two studies were completed in southeastern North Dakota to determine if restoration of these areas is possible. Research objectives were to; (1) determine if one year of glyphosate application is enough to impact a soil seedbank of a previously cultivated wetland, and (2) estimate costs of three different vegetation restoration methods to better understand cost/benefit ratios of restoration methods. Seedbank analysis showed significant differences with one year of glyphosate application, and restoration costs were determined for the different techniques. These results will be utilized to help aid restoration efforts in the future to make them more time and cost effective.

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LITERATURE REVIEW

Wetlands and Losses

Wetlands are among the most important ecosystems on earth (Mitsch and Gosselink 2007). They provide numerous ecological functions to both humans and wildlife. These include shelter and habitat, food, protection from catastrophic flooding, irrigation, and carbon sequestration (Bobbink et al. 2006). Wetlands have been proven to aid in the filtration and purification of water as well as the recharge of underground aquifers (Mitsch and Gosselink 2007). Despite all the obvious values provided by the conservation of wetlands, it is difficult to get the general public to fully recognize their value without their benefits given a monetary value. This is an extremely difficult and highly subjective way to analyze wetlands because many of the services they provide are "invaluable," such as carbon sequestration and aesthetic values. The ecosystem services provided by wetlands around the globe have also been estimated and valued at $14,785 /hectare (ha) yr in 1994 (Costanza et al. 1997). At the present rate of inflation this value increases to $23,315.51 /ha yr (Bureau of Labor Statistics 2015).

As of 1991, there had been a net loss of 53% of the wetlands once found in the conterminous United States (Dahl 1990). Wetland losses from the 1950's to 1970's are estimated at 3.7 million ha, or approximately 185,000 ha per year during that period (Frayer et al. 1983). Most recent estimates (2004-2009) put wetland losses at a rate of 5,590 ha per year (Dahl 2011), which is a significant increase in loss compared to the previous estimates of a net gain of 12,900 ha per year in wetlands from 1998-2004 (Dahl 2006). However, Dahl (2006) commented in his report that despite this increases in acreage, the quality of those wetlands was not determined. The rate of wetland loss has decreased significantly in the past few decades, but overall wetland area is still being lost, as can be concluded from the most recent report on wetland status and trends (Dahl 2011). The greatest wetland losses are intertidal wetlands found along the coastal areas of Texas and Louisiana. A majority of these losses are not directly from human influence though, which shows an increase in protection from governmental and state agencies (Dahl 2000). The greatest impacts to these wetlands have been attributed to mostly oceanic influences, including land subsidence, coastal storms, and sea level rise. However, human related activities such as water, oil, and natural gas extraction have been shown to be a contributing factor of land subsidence (Dokka 2006). The largest contributors to freshwater wetland losses are from urban and rural development, and forested freshwater wetlands were impacted the most by silviculture operations (Dahl 2011).

The conversion of wetlands to agricultural land continues to be a major cause of wetland losses (Mitsch and Gosselink 2007). From the 1900's to the mid 1980's, wetland drainage in the United States (US) due to farms occurred at a rate of approximately 490,000 ha/yr (Office of Technology Assessment 1984). With the rising prices of grains in response to both the growing demand for food worldwide and the mounting need for biofuels, wetlands are still being converted to agricultural uses (Johnson 2013; Wright and Wimberly 2013). This problem is even more prevalent in the Prairie Pothole Region (PPR) of the North Central US and Canada. Seventy one percent of the wetlands found in the Canadian portion of the PPR are estimated to have been lost (National Wetlands Working Group 1988). According to Mitsch and Gosselink (2007), only 10% of the original wetlands once found in the region still exist since modern human settlement began, and more than half of these wetlands have been drained or altered primarily for agriculture. The PPR is known for its extremely fertile soils and substantial agricultural productivity. One contributing factor to that productivity is the abundance of water found in the numerous wetlands of the region (Winter 1989). However, in order for the land to be of utilized for agriculture, the water must be drained so crops can be planted. In some places, agricultural drainage networks are so vast and intense that they have potentially irreversibly altered the regional hydrology (Dahl 2014). Agriculture near wetlands can also have indirect impacts that aren't immediately noticeable or have immediate consequences. One such consequence is the application of pesticides onto crops and its potential impacts on invertebrates found in these wetlands, and its effects further up the food chain on waterfowl (Beyersbergen et al. 2004).

From 1997 to 2009, total wetland area in the PPR declined by 30,100 ha, or 2,510 ha/yr. Emergent wetlands and shrub wetlands in this area had the greatest declines (36,250 ha and 18,660 ha, respectively), but some of these losses were offset by an increase of 24,810 ha in forested wetlands (Dahl 2014).

While these recent statistics may provide evidence of a potentially bleak future, there is still hope for the remaining wetlands present around the country. Many different forms of legislation have been passed in the past half century to prevent the destruction of these systems, and also to provide the possibility of replacement for lost systems. Private organizations such as Ducks Unlimited and The Nature Conservancy, in addition to governmental agencies like the United States Fish and Wildlife Service (USFWS), have been purchasing high quality land to protect these wetland systems. Through the purchase of these high quality areas, the USFWS can designate these areas Waterfowl Production Areas (WPA), and prevent them from being drained or lost. Since its initiation, more than 274,000 hectares (677,000 acres) have been protected in nearly 7,000 WPA's across the country, with many of these falling in the PPR (USFWS 2007).

Wetland Restoration

The initiation of the "no net loss" federal policy by George H.W. Bush in 1989 was a major step towards the protection of our nation's wetlands. The goal of this policy was to achieve no overall loss of the nation's wetlands, or if the destruction of a wetland was unavoidable, the creation or restoration of a different wetland to mitigate the damage (National Wetlands Policy Forum 1988). While this policy allows for the mitigation of wetland alteration, these laws do not fully take into account the impacts that destruction of a wetland can have on an ecosystem. According to Mitsch and Gosselink (2007), "… the most common alterations to wetlands have been: (1) draining, dredging, and filing of wetlands; (2) modification of the hydrologic regime; (3) highway construction; (4) mining and mineral extraction; and (5) water pollution." This mitigation process allows the party destroying a wetland to replace or improve a wetland at another location. They have the option to create a new wetland where none existed, restore a lost wetland, increase the size of a functioning wetland, or enhance a poorly functioning wetland. Because wetlands are very unique habitats, they are very difficult to restore or replace once degraded or lost. It was once believed wetlands were easy to replace, but, even after 20 years post restoration, it has been shown that restored wetlands do not function to support the plant diversity of natural wetlands (Aronson and Galatowitsch 2008). Often wetland creation results in the production of a wetland that resembles the previous wetland, but lacks the functioning necessary to replace it. In order for a wetland mitigation to be deemed successful, the restored wetland must resemble and function like the original (Galatowistch and van der Valk 1994).

There are many barriers to overcome in order for a wetland restoration to be successful. Mitsch and Jørgensen (2004) outline seven key principles essential for success including designing the wetland system for function not form, and to conform the project to what the environment allows, among others. Once these principles have been addressed, the wetland creation or restoration has an improved chance of succeeding, although nothing in nature is guaranteed. Even the most carefully planned restorations have a chance of failure if any aspect is overlooked or unpredictable natural events, such as flooding occurs.

One of the greatest threats to a restoration is the chance of invasion by non-native species. Some non-native species are considered invasive species, and are described as species that quickly and efficiently take over an area following their introduction into a new location (Rejmanek and Richardson 1996). One reason restorations are so susceptible to invasive species is due to the large disruptions occurring to the plant communities at restoration sites. For example, the reflooding of a drained wetland results in a new environment being established, and thus a vegetation change from upland plants to more wetland species. This shift in the plant community generally results in upland species dying out and the presence of bare ground for recolonization by wetland species. Invasive species are generally fast colonizers and great competitors, often being well established in the new soil by the time native species begin to emerge and try to compete for resources. According to Galatowitsch et al. (1999), "Once established, invaders are difficult to remove, lowering the quality of existing wetlands, and reducing the effectiveness of restoration efforts." These invasive species generally create a monoculture and prevent desired native species from becoming established (Odum 1988).

Seedbanks

The seedbank of any area is one of the key components when determining the vegetation. A soil seedbank is described as a collection of all viable seeds in the soil that have not yet germinated (Roberts 1981). Seeds found in the soil are usually products of the vegetation found in the area, but may not necessarily reflect the present aboveground vegetation (Thompson and Grime 1979; Cardina and Sparrow 1996). It may also contain seeds from species no longer present in the above ground cover. The seeds of different plant species are highly variable in the amount of time they can remain viable in the seedbank, as well as the amount of time it takes them to germinate and begin to grow. Because of this, wetlands that appear similar may have drastically different plant communities. Also different types of wetlands (permanent vs. temporary) have different seedbanks due to different environmental conditions.

The presence or absence of surface water in a wetland is the determining factor of the type of vegetation found in that area (Weiher and Keddy 1998). Water is the key to the establishment of plants from a seedbank. Seedbanks may consist of persistent seedbanks with long-lived seeds and/or transient seedbanks with short-lived seeds (Thompson et al. 1997). Persistent seedbanks generally consist of seeds that can survive multiple growing seasons or wetting and drying events and remain viable to germinate when the conditions are right, generally when there is minimal competition. Transient seedbanks contain seeds which are viable only one growing season and germinate at their first opportunity. Temporary wetlands which dry rapidly and wet again following each successive rainfall favor persistent seedbanks while semi-permanent and permanent wetlands tend to favor transient seedbanks, due to predictable water regimes (Brock 2011). Seeds which do not germinate at their first opportunity continue to remain viable in the soil, and contribute to the residual seedbank of an area. This residual seedbank will continue to fluctuate as time goes on and as the reproductive success of present plants and germination of seeds vary (Bonis et al. 1995; Brock 1998; Leck and Brock 2000).

Planning a restoration based on recruitment from a present seedbank (i.e. natural revegetation) is often very difficult. Since many restorations are currently occurring on lands that were previously used for agriculture, seedbanks of these areas are often greatly reduced from years of herbicide, drainage, tillage, and grazing when compared to natural systems (Wienhold and van der Valk 1989; Kline 1997; Lunt 2003). These areas often have few native species left to help propagate restoration efforts. The effects of these missing native species from the ecosystem can be seen as a positive feedback loop between native species present and their ability to disperse. The fewer present, the less dispersal ability they have. Tilman (1997) found native species' abundance and richness were limited by recruitment and local biotic interactions. Recruitment limitation has been shown to have significant impacts on community composition in numerous studies (Menge and Sutherland 1987; Rejmánek 1989; Robinson et al. 1995). In addition to this lack of dispersal and recruitment, it has been shown that invasive species produce greater leaf area with reduced costs (Baruch and Goldstein 1999), but there is not significant evidence to conclude that overall native species grow slower (Daehler 2003). Because of this dispersal limitation and often limited reproductive success, seed stocking is helpful to increasing plant composition and abundance, and eventually, the seedbank when recruitment limitation is overcome (Tilman 1997).

Preparing a Restoration for Success

Many different methods have been utilized to prepare an area for restoration. In an analysis of restoration techniques from 38 restoration managers, Rowe (2010) found the most common practice was to plant the site into corn or soybeans for 2-3 years using conventional techniques. These lands were leased to local farmers to plant, and it helped considerably with alleviating costs, which can often be a driving factor in restoration. Many of these projects were fallow, previously farmed croplands. Wilson and Partel (2003) found that these fields are often dominated by introduced grasses, and these grasses can be difficult to control. The cropping process reduces perennial and annual weed abundance, as well as their seedbanks through disking (Farkas 2002), tilling (Gendron and Wilson 2007), and herbicide application (Schreiber 1992).

Fire has also been utilized as a technique to prepare for restoration. In areas where tilling or cropping are not applicable, fire can be utilized. Rowe (2010) found fire to be nearly universally applied in all restoration studies she looked at. C_4 plants have been shown to increase growth with spring burning (Robocker and Miller 1955), and the repetition of these burns on an annual or biennial basis can lead to a trend in the dominance of a native, warm season plant community (DiTomaso et al. 2006). However, only utilizing burning requires native species to still be present in the community. If there are shrubs or woody species present, frequent fire (every 1-2 years) is an effective tool to reduce their presence and prevent their future encroachment (Hartnett and Fay 1998; Peterson and Reich 2008).

Reestablishing the hydrologic function of a wetland is also necessary for its restoration. Much of the area that was previously tallgrass prairie has undergone extensive hydrological alteration due to agriculture (Urban 2005). The causes of these alterations can be twofold. The primary cause of these altered regimes is due to filling and draining of these sites to make the land accessible for agriculture (Mitsch and Gosselink 2007). Other contributing factors to this alteration can be lowered water tables or altered stream flows due to agricultural use (Rowe 2010). Undoing these alterations can be harder than initially thought. Removing drainage tiles from a field is a relatively simple process, but legislation can impede this process. Rowe (2010) found through communication with restoration managers that local regulations can often prevent the alteration or removal of these drainage tiles or ditches. With no way to remove these tiles, it can be impossible to restore hydrology to an area, even if the landowner wishes it.

Post-Restoration Management

In order for a restoration to remain successful past its initiation, some sort of management plan must be put in place to ensure continued progression towards the originally desired goals. It is important to identify the objective of a wetland restoration prior to starting the project so progress of the wetland towards its desired state can be tracked (Mitsch and Gosselink 2007). Most early wetland restorations were completed with no attempt at long term management because it was originally believed restoring hydrologic conditions to a wetland would bring native plants back. However, recent research has shown this idea to be incorrect. Mulhouse and Galatowitsch (2003) visited sites with no predetermined management plan ten years after restoration, and found these sites dominated mostly by invasive plant species. It was found in restorations in the PPR that a majority of the flora found in restored wetlands will colonize in the first 12 years following restoration (Aronson and Galatowitsch 2008). Because of this, long-term monitoring of restored sites is necessary to ensure the restored area continues to progress towards the desired goal. Current monitoring for restorations due to mitigation is normally limited to 3-5 years to ensure the original conditions are met through the restoration (Mitsch and Wilson 1996). This, however, is not enough time. According to Jørgensen (1994), the further away a system is at its beginning state from the desired condition, the longer it will take for that system to reach the desired state. This means the amount of time necessary to monitor a restoration is highly variable, and depends entirely on the condition of the area prior to any restoration action. Severely degraded wetlands may take several decades to return to a desirable condition, while a minimally disturbed site may only take a few years. With no absolute quantity on the amount of time required to ensure the success of a restoration, each site will have to be evaluated individually and evaluators will have to design a management plan for each wetland independently.

Native Vegetation Seeding

Many early wetland restoration attempts in the PPR were initiated under the assumption that native vegetation would return if the water regime was restored, however, many studies have been conducted that show this often is not the case (Galatowitsch and van der Valk 1996a; 1996b; Aronson and Galatowitsch 2008). Decades after hydrologic function has been restored, many native species are still absent from the landscape. Studies of the seedbanks in these restored wetlands show that many of the native species found in natural undisturbed wetlands are absent from restored sites (van der Valk 2013). These missing species from the seedbank require supplementation in order to reestablish in restoration attempts.

Seeding of native species has been shown to greatly increase native diversity in restoration attempts (Kiehl et al. 2006). This seed addition allows native species to establish quickly, although often management is also needed to prevent interference from invasive species. For example, *Phalaris arundinacea* (reed canarygrass) is a major threat to many restoration attempts in the PPR (Adams and Galatowitsch 2006). By seeding these restored areas, native perennial vegetation can become established, and therefore limit the area available for reed canarygrass to infiltrate (Lindig-Cisneros and Zedler 2002). The quality of the seed to be used is also important to take into consideration. High diversity seed mixtures have been shown to be more effective at producing diverse communities than low diversity mixes (Leps et al. 2007).

Hay Transfer

The use of hay transfer to aid in restoration seeding is a relatively new practice that is being implemented more often. However, in many of the trials in which it was utilized, it aided the establishment of planted seed. According to a study done by Török et al. (2012), the use of hay transfer, in addition to seed planting, worked well for the suppression of weed cover, and also led to a decrease in weed species richness and overall biomass. This suppression of early weeds in a project is a typical goal of a restoration manager. The reasons a hay layer is potentially beneficial by suppressing weeds include the following: first, it acts to protect the soil surface from desiccation allowing native seeds (usually longer germination period than weeds) greater chance to germinate (Fowler 1988); second, it buffers the soil from fluctuations in temperature which is a germination signal for several weed species (Foster and Gross 1998); third, it may exhibit some allelopathic effect (Ruprecht et al. 2010); fourth, it can act as a physical barrier for wind dispersed weeds (Wedin and Tilman 1993); and last, it decreases the amount of light reaching the soil surface and available to weeds (Foster and Gross 1998).

The process of hay transfer can do more than just prevent the establishment of weedy species during a restoration. It can also be utilized as an additional or primary source of seed. Since many native and desirable plants have limited dispersal capabilities (Galatowitsch and van der Valk 1996a) the use of hay can help overcome this limitation by putting the seed in place. When using hay transfer as a form of seeding, the quality of the hay and timing of hay cutting are important. In order to maximize the potential success of the hay transfer, hay must be collected when the target species seeds are ripe and from an area where there is a high density of the target species (Kiehl et al. 2006; Rasran et al. 2006). Managers can manipulate sites using hay transfer to increase selected species richness by including them in the selected hay. Doing this inhibits the colonization of restored sites from other seed sources and relies almost entirely on contributions from the existing seedbank and the added hay for colony establishment (Klimkowska et al. 2010).

Soil Plug (Transplant) Restoration

The use of transplanted soil plugs as a restoration technique has been practiced for a considerable time, but published information utilizing these methods is minimal. There have been a few studies in both prairies (Christianson and Landers 1969; Clarke and Bragg 1994) and wetlands (Davis and Short 1997) that utilized soil transplanting as a method for restoration, but no simple, cheap, or consistent transplanting method has been discovered yet. The costs are often the primary limiting factor to the lack of soil transplant restorations. Despite the labor and cost intensive process required to transplant wetland vegetation, it is still highly encouraged in addition to naturally occurring vegetation and native inter-seeding (Williard et al. 1990). Various methods have been used to remove whole cores with vegetation and plants intact including plugs (Amon et al. 2006), tractor mounted tree spades (Fraser and Kindscher 2001), hand tools (Bragg 1988; Davis and Short 1997), and PVC pipe (Phillips 1990). Transplanted vegetation, in addition to seeding, has been shown to increase the native species richness, diversity, and quality of a restoration project when compared to areas that were seeded only (Middleton et al. 2010).

While the use of soil plugs for restoration has great potential, there are also drawbacks. Davis and Short (1997) mention the drastic impacts soil plug removal can have on the donor site. The removal of soil plugs leaves holes in the healthy donor sites, and thus can create areas for possible invasion or erosion if this impact is not addressed. One way to alleviate these effects can be the use of smaller plugs. By using many small plugs in an area there is less impact than removal of a large plug. By using small plugs, there is less continuous exposed area prone to invasion, and greater edge distance around the plug holes to allow surrounding native species to recolonize the removed area. Another way to minimize effects of transplants is to target vegetation removal from wetlands that will be removed due to mitigation projects to prevent a total loss of the wetland area (Fraser and Kindscher 2001).

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PAPER 1. CROPPING AS A TOOL FOR MANAGING SEEBANK COMPOSITION FOR WETLAND RESTORATION IN SOUTHEASTERN NORTH DAKOTA

Abstract

The process of restoring a degraded wetland to natural conditions is time consuming, tedious, and often an expensive process with variable results. Many restoration attempts involve restoration of the water regime and then allow the wetland to reestablish primarily through natural succession from its seedbank. However, after decades of disturbance the seedbanks of these wetlands can be severely depleted, often containing few, if any, native plants. This seedbank depletion is especially prevalent in wetlands that have been drained and/or utilized as hydric cropland. Prolonged cropping can severely impact a seedbank through commonly utilized agriculture practices such as herbicide treatment, fertilization, and tillage, and often leads to a transition to weedy, annual species and also provides an area for perennial invasive plant species to become established. Prior to initiation of a wetland restoration project it is important to determine if desirable species are still present in the seedbank. A seedbank study is one of the most effective methods to determine which species are present and viable in a seedbank. In this study we completed a seedbank study of a degraded wetland that was previously farmed and allowed to sit fallow for parts of the past 20 years, leading to a buildup of invasive and weedy species at the site. A seedbank study was completed with initial conditions, and once again a year later after the area was planted with Roundup Ready soybeans and treated with five applications of glyphosate. We look to address the impact five applications of glyphosate over one year can have on the seedbank of a degraded wetland, and if it is possible to restore these degraded wetlands to their natural form and function. Results of the seedbank study showed significant

differences ($P < 0.05$) between seedling counts, which suggests one year of herbicide application is enough time to significantly impact the seedbank of a degraded wetland.

Introduction

A seedbank is described as a collection of all viable seeds in the soil that have not yet germinated (Roberts 1981). The seedbank composition often provides information as to the ecological history of plant communities and is necessary to fully understand them (Hill and Stevens 1981; Thompson 1986; Milberg 1995). Frequently the seedbank is not a direct reflection of above ground vegetation (Thompson and Grime 1979; Cardina and Sparrow 1996). Many wetland species require precise cycles of wet and dry for their seeds to germinate, and thus may not be present in standing vegetation if conditions that year are not favorable. Hydrologic regimes in temporary or seasonal wetlands can fluctuate greatly in the Prairie Pothole Region (PPR), and this can have large impacts on seedbanks as well as emergent plants. Brock (2011) found that the composition of a seedbank can change profoundly the longer a wetland remains dry, or with an increase in the number successive germination events.

Seedbanks also tend to vary greatly between types of wetlands. Temporary, short lived wetlands often have a much different seedbank composition than wetlands that hold water continuously. Short lived wetlands tend to favor a persistent, long lived seedbank that can survive long periods of dry before germination, while wetlands with nearly permanent inundation tend to favor transient seedbanks that often remain viable for one year or less (Thompson et al. 1997). Often areas with more regular hydrologic regimes will contain a mixture of plant species contributing to both the transient and persistent seedbank (Leck and Simpson 1987; 1995).

Analyzing the composition of a seedbank can also explain important ecological shifts that may be occurring in a community. By comparing a seedbank to the present above ground vegetation, a determination of floristic similarity can be determined. This can be used to then determine whether the seedbank is driving vegetation composition, or if the vegetation is influencing the seedbank (Leck and Simpson 1987; Henderson et al. 1988). This is useful to determine whether or not a trend, either good or bad, exists across a temporal scale. However, it is highly likely in wetlands containing both transient and persistent seedbanks, to fluctuate greatly from year to year depending upon hydrologic conditions, thus greatly influencing floristic similarity scores from year to year (Leck and Simpson 1995).

It has been suggested that wetland restoration is a process that can occur naturally by utilizing the seedbank of an area and secondary succession (van der Valk 1999). However, transient portions of a seedbank have been shown to remain viable for only short periods of time (one year or less), and fail to germinate if they remain in the seedbank for longer (Thompson et al. 1997). Because of this, the longer a wetland remains dry or in a degraded condition, the more likely the transient seedbank is to be depleted. If these conditions are allowed to persist for decades or longer, it is possible to eliminate portions of the persistent seedbank as well. Larkin et al. (2012) showed that *Typha* x *glauca* was able to prevent seedlings from surviving by creating a litter layer that changed physical and chemical properties of the soil and shaded out other species. By failing to create seed yearlyto contribute to the seedbank, eventually these persistent species can disappear from the seedbank.

Ability to disperse can also significantly influence on the species present in a seedbank. After years of drainage, many species that were present prior to draining are lost from the seed rain that aids in these species survival (Kettenring and Galatowitsch 2011). Wienhold and van der Valk (1989), and van der Valk et al. (2009) found that after completing a restoration, only a subset of the expected or desired wetland species would be present in the seedbank. Supplementation of a seedbank can occur in a multitude of natural ways including water movement (Huiskes et al. 1995; Middleton 1999; 2000), waterfowl (Mitsch and Wilson 1996; Figuerola et al. 2002; Muelller and van der Valk 2002), wind (Fenner 1985; Wienhold and van der Valk 1989; van der Valk et al. 2009), and other animals (DeVlaming and Proctor 1968; Crawley 1983). While dispersal of some wetland species happens easily, other species historically abundant in the PPR (*Carex* spp.) have been shown to not easily colonize these areas due to the loss of hydrologic connections across the landscape (Galatowitsch and van der Valk 1995; 1996a) and their reproduction primarily through clonal growth.

It is important to consider the present seedbank when planning a restoration, and design a plan to control it. Seedbanks of previously farmed hydric croplands are often dominated by introduced grasses that are difficult to control once they become established (Wilson and Partel 2003). By cropping these areas with traditional farming practices for 2-3 years prior to starting a restoration, it is possible to reduce their seedbanks and increase chances of restoration success (Rowe 2010). The use of disking (Farkas 2002), tilling (Gendron and Wilson 2007), and herbicide (Schreiber 1992) have been shown to reduce perennial and annual weed cover and their seedbanks in fallow fields. If the existing seedbank is not addressed prior to planting of native seed, already established introduced species may outcompete native seedlings and hinder their establishment.

In this study we look to determine which species are left in a soil seedbank of a degraded wetland that was utilized for decades primarily for agriculture in the past and left fallow to be taken over by invasive wetland species such as *Typha* spp. and *Phalaris arundinacea*. We also were looking to see the impact one year of Roundup Ready soybeans and five applications of glyphosate can have on the soil seedbank.

Methods

Study Site

This study was conducted on an 18.86 hectare area located on the Albert Ekre Grassland Preserve in Richland County, North Dakota; approximately two kilometers east of the Sheyenne National Grasslands (Latitude 46.526224; Longitude -97.132370) (Figure 1.1). This area is characterized by sandy soils that originated from an ancient river that made its way through the area and emptied into glacial Lake Agassiz. Historically the study site was primarily native tallgrass prairie, but was at one time plowed and converted into cropland. Hydrologic regimes at this site fluctuate greatly from year to year and no type of drainage has been applied. Because of this, farming has occurred only four times in the past 19 years, with the land remaining fallow on years it was not farmed (Dewey Lindgren, Personal Communication). This eventually led to a dominance of weedy species and robust invaders that were able to thrive on this fallow ground.

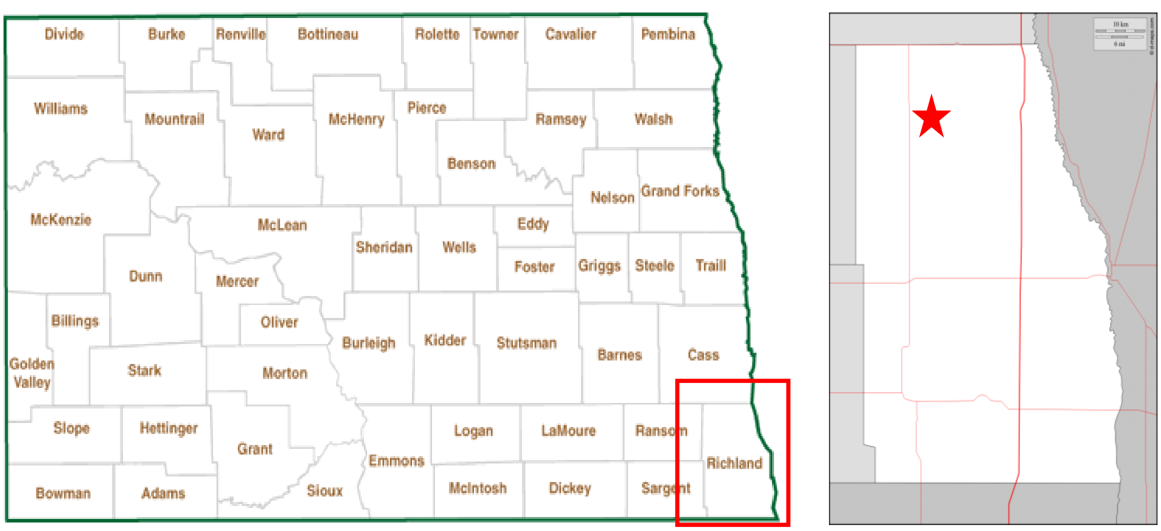

Figure 1.1: Location of the Ekre seedbank study in Richland County, southeastern North Dakota.

The major natural vegetation type in this area was tallgrass prairie, which was dominated by big bluestem (*Andropogon gerardii* Vitman), prairie sandreed (*Calamovilfa longifolia* (Hook.) Scribn.), sand bluestem (*Andropogon hallii* Hack.), needle-and-thread grass (*Hesperostipa comata* (Trin. & Rupr.) Barkworth), porcupine grass (*Hesperostipa spartea* (Trin.) Barkworth), sand dropseed (*Sporobolus cryptandrus* (Torr.) A. Gray), sideoats grama (*Bouteloua curtipendula* (Michx.) Torr.), prairie junegrass (*Koeleria macrantha* (Ledeb.) Schult.), Canada wildrye (*Elymus canadensis* L.), blue grama (*Bouteloua gracilis* (Willd. ex Kunth) Lag. ex Griffiths), and sedges (*Carex* spp.). Wetlands are typically dominated by woolly sedge (*Carex pellita* (Muhl.) Willd.), slough sedge (*Carex atheroides* Spreng.), fescue sedge (*Carex brevior* (Dewey) Mack., northern reedgrass (*Calamagrostis stricta* (Timm) Koeler), prairie cordgrass (*Spartina pectinata* (Bosc) Link), rushes (*Juncus & Elocharis* spp.), other sedges (*Carex* spp.), and forbs. (NRCS ESIS, ESD 2015).

Soils of the wetland are described as: 97.2% Aylmer-Rosewood-Serden complex, 0-9% slopes with minor components: Ulen, Hamar, Bantry and Venlo series; 1.7% Serden-Hamar complex, 0-15% slopes with minor components: Bantry, Alymer, Ulen, and Venlo series; and 1.1% Garborg loamy fine sand, 0-1% slope with minor components: Hamar, Delamere, Mantador, Ulen, and Venlo. Aylmer soils are derived from wind-worked sandy glaciofluvial deposits and classified as subirrigated sands. Rosewood soils are derived from sandy glaciofluvial deposits and classified as wet meadow. Serden soils are derived from sandy eolian deposits and are classified as sands. Hamar soils are derived from sandy glaciofluvial deposits and classified as wet meadow. Garborg soils are derived sandy glaciofluvial deposits and/or sandy glaciolacustrine deposits and are classified as subirrigated (NRCS WSS 2015).

The climate of this area is considered humid continental (cool summer) and has a wide fluctuation in annual and daily temperatures. Precipitation is highly variable in amount as well as annual totals. Thirty year averages for the area put precipitation at 47.2 cm during the April to October growing season from 1981-2010 (USDOC NOAA 2015). Rainfall total in 2013 were 53.2 cm, 6 cm above average, and in 2014, the total was 49.3 cm, 2.1 cm above average (NDAWN 2014).

Site Preparation

The study site was burned in the spring of 2013 to prepare the area for soybean planting and spraying. It was a dry spring so the wetland was able to be burned in its entirety. After the wetland was burned, approximately one week later the ground was prepared for planting. The site was run over with a salford vertical tillage tool (independently mounted discs, harrow, and rolling basket to firm the ground) three times, in order to remove any large hummocks or mounds that were present from decades of cropping, sitting fallow, and being inundated. Seeding of Roundup Ready soybeans immediately followed this process. Herbicide (Roundup PowerMax®, Monsanto Company, St. Louis, MO, 63167) was applied five times in 2013, once post burn before planting, 3 times in crop, and once again post-harvest. Application rates were consistent with label recommended dose for soybean application of 1.54 kg/ha.

Seedbank Sampling

One hundred eighty seedbank samples were taken in a systematic grid configuration twice over a one year span, once in early May 2013 before treatment and again in late May 2014 following one year of herbicide application (Figure 1.2). Sample sites from the first year were marked with a GPS unit (1 meter accuracy) so samples could be taken from the same spot the following year (Appendix A). The soil samples were taken in the spring of each year as soon as the weather allowed and conditions were acceptable for access. Soil samples were taken with a standard golf-hole cutter 10 cm in diameter and only the top 5 cm of each soil core were used. Thatch and debris were removed from the tops so only the top 5 cm of soil were included in each sample. These samples were then bagged individually and refrigerated at 3 ^{o}C for at least 1 month to incur dormancy in seeds that required it for maximum germination potential according to Perez et al. (1998).

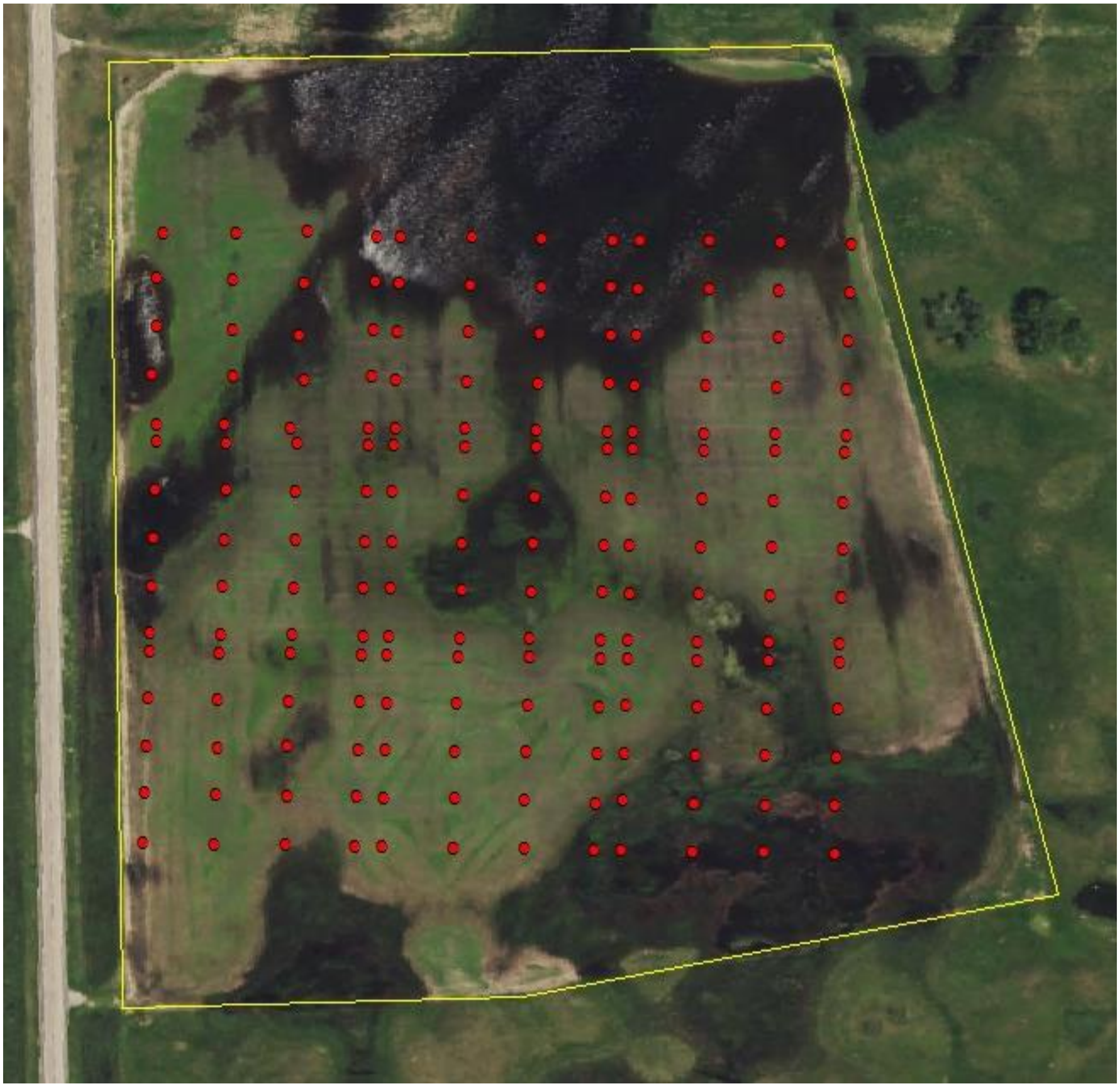

Figure 1.2: Map of sample locations for the Ekre seedbank study in southeastern North Dakota.

Seedbank Analysis

After being refrigerated, soil samples were processed following the procedures presented by Ter Heerdt et al. (1996). Ninety 26 x 26 x 6 cm square trays were prepared using a mixture of steam-sterilized soil and commercial potting mix (SunshineMix No. 1®, Sun Gro Horticulture, Bellevue, WA 98008) to a depth of 2.5 cm, then topped with 1.5 cm of steam-sterilized silica sand. Samples were individually washed through two different soil sieves, a coarse No. 5 (4 mm), and fine No. 70 (0.212 mm) to remove debris and root material that may influence germination results. After washing, two samples were poured together to create a “slurry” and added to the prepared trays of sand and potting mix to form a layer from 3-5 mm thick. Trays were then placed in the greenhouse and watered daily. Natural light was supplemented with halide lamps at 450 $\mu Em^{-2}\ s^{-1}$ from 6 am to 10 pm daily. Temperature was maintained between 20 and 30 ^{o}C for 8 to 9 weeks, the duration of this study. The study was conducted for this duration because Ter Heerdt et al. (1996) suggested 95 percent of all seedlings will emerge in 6 weeks. Seedlings were then counted and removed as they were identified, to allow further seedlings to emerge. Unknown seedlings were also removed and placed in separate trays to grow until they could be identified.

Data Analysis

Plant composition changes were analyzed by evaluating annual to non-annual seedlings and forb to graminoid seedling proportions. These counts were then transformed using the fourth root for total seedling counts, and an arcsine conversion for the proportion of annuals and proportion of forbs. After the conversion, each of these variables had an approximately normal distribution. Absolute counts were used to compare total seedlings between years. Total seedling counts per tray were extrapolated to the number of seeds per m^2 in a 5 cm layer of soil. Soil seedbank data were analyzed using one-way analysis of variance using SAS Enterprise Guide (SAS Institute Software 2013, Version 6.1, SAS Inc., Cary, NC) at the $P < 0.05$ significance level for total seedlings and plant composition changes.

Results

Seedling emergence was significantly different ($P < 0.05$) between pre and post herbicide application. The preliminary seedbank in 2013 resulted in 83,652 total seedlings. After one year of herbicide treatment, seedling count dropped to 31,835 in 2014. These results were converted to seedlings per square meter (Appendix B). In 2013, graminoids and forbs were nearly equally represented with 51.1 percent graminoid and 49.9 percent forb composition. In 2014 graminoid and forb composition were 41.6 and 58.4 percent respectively. This difference was not significant ($P < 0.05$). The biggest cause for this change in composition came from the loss of the annual graminoid *Cyperus erythrorhizos* from the seedbank. In both 2013 and 2014, the top five most abundant species accounted for 79 and 80 percent, respectively, of the total seedlings. In 2013 these species in order of abundance were *Cyperus erythrorhizos*, *Potentilla norvegica*, *Potentilla paradoxa*, *Eleocharis acicularis*, and *Juncus interior*. The 2014 results in order of abundance are *Lindernia dubia*, *Eleocharis acicularis*, *Juncus interior*, *Hypericum majus*, and *Potentilla norvegica*.

Due to the vast difference in total seedlings between 2013 and 2014, proportions of the total were used to calculate changes in plant composition before and after herbicide application. Plants that decreased the greatest proportionally after one year of glyphosate application were *Cyperus erythrorhizos*, *Potentilla paradoxa*, *Potentilla norvegica*, and *Cyperus squarrosus*. They decreased 30.5, 13.9, 13.1, and 4.1 percent respectively when 2014 proportions were compared to 2013. These plants all are mostly annual species so it is not surprising to see such drastic declines in their numbers. The greatest increasers after herbicide treatment were *Lindernia dubia*, *Elocharis acicularis*, *Juncus interior*, *Hypericum majus*, *Typha* spp., and *Veronica peregrina*. They increased 19.5, 16.8, 9.8, 6.8, 3.8, and 3.6 percent respectively in 2014 proportions versus 2013. Half of these plants are annuals, and the other half are perennial, showing a trend of increases in both.

Proportions of annual to non-annual seedlings did not change drastically between the years. In 2013, 43.5% of the species were annuals, and in 2014 this number slightly increased to 44.1%. These numbers are not significant ($P < 0.05$) and thus do not reflect any large change in the vegetation composition.

While some species had drastic increases or decreases after herbicide application, a majority did not. Of the 56 species found during this study, 35 increased or decreased by less than 1 percent, showing their populations remained relatively stable despite the application of herbicide for one year. Seven species that changed by this percentage were excluded from this statistic because only one seedling was found in either year.

Discussion

This study shows the impact decades of agriculture can have on a seedbank. Missing from our seedbank study were many of the native species usually found in most prairie pothole wetlands such as *Carex* spp., *Calamagrostis stricta*, *Spartina pectinata*, *Juncus* spp., and other native forbs (NRCS ESIS, ESD 2015). Our seedbank results support the conclusion that prolonged agricultural use leads to the fragmentation of wetlands in the landscape, and subsequent depletion of native species from the soil seedbank (Wienhold and van der Valk 1989; Kline 1997). The abundance of weedy, mudflat annuals in our seedbank is also consistent with the results of Galatowitsch and van der Valk (1996a), who also found emergent perennial species to be lacking in restored wetlands. Getting these emergent, wet meadow species reestablished in a wetland is difficult without direct intervention. Studies of the seed rain of other prairie pothole wetlands (Kettenring and Galatowitsch 2011) show similar results to this study. Numerous small wetlands (<1 km away) contain desirable native species (*Carex* spp.), but few seeds were found in the seedbank study suggesting the limited dispersal of these species. Incoming seed to these degraded wetlands often contain few, if any, of these natives, and, instead, is often dominated by invasive species, which often come to dominate restoration attempts as has been observed in many restored wetlands (Galatowitsch and van der Valk 1996a; 1996b; van der Valk 2013). Because of this, dependence upon the seedbank alone is not enough for restoration of previously observed vegetation in this area.

It should be noted there was an increase in *Typha* spp. plants as well as the emergence of reed canarygrass following herbicide application. These two species are known for their ability to produce dense, monotypic stands, which tend to suppress native species (Galatowitsch et al. 1999; Green and Galatowitsch 2001; Frieswyk et al. 2007). Reed canarygrass is able to form a dense network of rhizomes, and also produces a long lasting seedbank that can cause difficulties with restoration attempts (Adams and Galatowitsch 2008). The use of glyphosate helped to reduce aboveground distribution of both reed canarygrass and *Typha*, but reemergence from seemingly the same area was evident after 3-5 weeks, suggesting a vast network of roots and rhizomes belowground capable of coping with the effects of herbicide. After numerous treatments of glyphosate the density and distribution of both species had declined. In late summer 2014, while removing soil plugs for a transplant procedure, several *Typha* rhizomes were found inside of the removed soil plugs. Upon closer examination, they appeared to still be filled with carbohydrates, but were diminished in total size and appeared “shrunken.” This provides evidence about the vigor of *Typha* rhizomes, and how difficult they can be to eradicate once they are established in an area. Distribution of both reed canarygrass and *Typha* were reduced with multiple applications of herbicide, but *Typha* seemed to be more tolerant of the herbicide taking longer to senesce and producing new shoots sooner after spraying.

It is possible that our application rate of glyphosate was not high enough to efficiently kill hybrid cattail in this study. Solberg and Higgins (1993) reported nearly 100% extermination rates of hybrid cattail dominated wetlands in South Dakota that were sprayed with 3.4 kg/ha glyphosate from a fixed wing aircraft. They reported effects of this treatment lasted up to 4 years if water levels were consistent to prevent its reseeding. Comes and Kelly (1989) reported 3.4 and 4.4 kg/ha were effective at controlling cattail, but a rate of 2.2 kg/ha was insufficient for significant reduction. These reported rates are all more potent than label recommended application for soybeans and may lead to the differences in control between their studies and ours.

Timing of herbicide application is important for control of reed canarygrass. A combination of spring burning followed by early application of herbicide is often the most common treatment for limiting its spread. However, this has been shown to not be the most effective method for its control. Applying herbicide in late August or late September was more effective at reducing reed canarygrass biomass than an application in late May (Adams and Galatowitsch 2006). A possible explanation for this increased herbicide effectiveness is increased rhizome mortality from a seasonal carbohydrate flux as the plant begins to senesce after the growing season, as also observed in Hemp Dogbane (Becker and Fawcett 1998). Adams and Galatowitsch (2006) also found that burning did not reduce existing reed canarygrass plants, but resulted in the reduction of its seedbank, therefore limiting recolonization after herbicide application. This is supported by our results due to the lack of reed canarygrass present in the seedbank and its severely limited distribution in the study site.

Our study gives evidence that planting Roundup Ready soybeans for one year and using multiple rounds of glyphosate for one year to prepare an area before starting a restoration project will help significantly reduce the seedbank of the site, and lead to less competition for planted native seed. Previous knowledge of an area is helpful prior to starting a restoration project to determine the extent of invasion and degradation in an area. Dense stands of reed canarygrass or *Typha* spp. may take an additional year of herbicide application to decrease their abundance and provide a better site for native planting.

This study also provides evidence that prolonged agriculture can significantly alter a seedbank from that of natural wetlands. The lack of native perennial vegetation in our seedbank is congruent with results found in O'Connell et al. (2013), where perennial vegetation was greatly reduced in cropland wetlands. Our study also shows that agricultural practices, not just the hydric drainage often associated along with it, can significantly alter the seedbanks over time. Tilling and the use of herbicide, both regular practices in modern agriculture, have been shown to reduce the seedbank of annual weeds (Schreiber 1992). Tilling alone has also been shown to have significant impacts on perennial, rhizomatous weeds (Farkas 2002; Gendron and Wilson 2007). Because of this, it may be suggested that regular tilling and herbicide application in hydric cropland can work to eliminate native plants by breaking up their rhizomes and preventing them from producing seed, thereby depleting them from the seedbank.

According to the US Fish and Wildlife Service (Karen Smith Personal, Communication), three years of Roundup Ready cropping is recommended to fully prepare an upland area for restoration. The Natural Resources Conservation Service Wahpeton office, North Dakota (Steven

Cole, Personal Communication) and Rowe (2010) also recommend at least 2-3 year of soybean-corn rotations for maximum weed control and to help offset preparation costs by leasing to local farmers. The final year of crop production on a restoration site should be soybeans. Rowe (2010) suggested this because soybeans do not leave furrows present in the soil and leave only a light layer of residue on the soil surface that can help planted seeds bind to the soil surface. Time constraints only allowed one year of soybean preparation, which this study shows can still have a significant impact on the present seedbank. If there were an additional 2-3 years of cropping with herbicide applications, the seedbank may be even more depleted that what was shown in this study.

Management Implications

While the difference in plant composition was not significantly different between the years, this study shows individual species may be greatly impacted by only a single season of herbicide, allowing desirable plants to potentially fill the holes left in the community. The complete loss of *Cyperus erythrorhizos* from the seedbank was surprising because of the quantity found the previous year, but inspection of the study site after herbicide reflected the same results seen in the greenhouse. The increase of annual species such as *Lindernia dubia* and *Hypericum majus* does not come as a surprise because these annual plants produce large quantities of seed yearly. This may be beneficial from a management perspective because these annual species can often be suppressed and displaced by seeded native perennial plants and mowing (Török et al. 2012). These annual weeds may act like a cover crop by limiting the amount of exposed soil available for other weeds to establish (Hartwig and Ammon 2002). This is especially relevant at the Ekre wetland because the surrounding ditches and pastures have large quantities of reed canarygrass and *Typha* spp. Regular management will have to be conducted to monitor and, if deemed necessary, provide action to prevent the encroachment of invaders.

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PAPER 2. WETLAND RESTORATION TECHNIQUES AND THEIR ASSOCIATED COSTS FOR SOUTHEASTERN NORTH DAKOTA

Abstract

Wetland restoration is often a time consuming and expensive process with results that can be highly variable. Many different methods exist for the restoration of wetlands, but each situation is unique and no one method is guaranteed to be effective. In many cases, funding is the limiting factor to the restoration process, and thus limits what may be done when better options are available. Restoration of vegetation is often the quickest aspect of a wetland to be restored, and preempts the return of other wetland processes such as nutrient cycling and establishment of wetland soils. In this experiment, we utilized three different restoration methods: (1) seed only; (2) seed and hay mulch; and (3) seed, hay mulch, and transplanted wetland soil plugs. We also looked at the estimated costs associated with implementing each of these methods and evaluated them at the per hectare level. Cost per hectare increased with the increase in restoration efforts. Treatment costs are as follows: seed only was estimated at $1,963/ha; seed and hay mulch at $2,342/ha; and seed, hay mulch, and soil plugs at $5,145/ha. Future research will be done on these constructed plots to help analyze the cost/benefit ratio of these different restoration combinations and their resemblance in form and function to nearby "reference" wetlands.

Introduction

Wetland restoration and management in the United States has undergone several advancements since its inception in the 1970's (Mitsch and Gosselink 2007). Scientists have also identified wetland qualities that can be used to assess their functioning in the environment. Many early restorations focused on restoring hydrological regimes to an area, believing that vegetation would restore itself. While this is true to an extent, the type of vegetation that reestablished on its own was often very different than that of “natural”, reference wetlands in the area (Galatowitsch and van der Valk 1996a). In their study, Galatowitsch and van der Valk (1996a) found that entire guilds of plants occurring in natural wetlands were missing from these hydrologically restored wetlands, and a potential cause for this discrepancy was the native species’ poor dispersal ability. After years of cropping and drainage, seed banks can be depleted to the point where naturally occurring wetland species are eliminated (Wienhold and van der Valk 1989). Without the recolonization of these restored wetlands by native species, the resulting open areas are prone to invasion.

Invasion of restored wetlands by unwanted plant species is an ever growing issue impacting restoration attempts across the United States. In the Prairie Pothole Region (PPR), some common hindrances to current restoration attempts are *Phalaris arundinacea* (reed canarygrass) and *Typha* spp. (cattails). These species are effective in quickly establishing themselves in recently disturbed sites, such as those created with the restoration of water to a wetland (Lindig-Cisneros and Zedler 2002). Reed canarygrass is of great concern due to its ability to prolifically spread its seed across the landscape, and the ability of this seed to produce vigorous seedlings that can rapidly establish themselves on moist soil (Wasser 1982). Once established, it is then able to rapidly dominate an area by vegetative reproduction and create dense colonies that shade out other, generally native species attempting to become established (Maurer and Zedler 2002). Because of this, it is important to rapidly establish vegetation cover on restoration sites to prevent the establishment of species such as reed canarygrass. Linding-Cisneros and Zedler (2002) suggest using a species-rich seed mix when seeding sites so a more complex vegetative canopy can establish to resist invasion, planting multiple times or using soil plugs, and including broad-leaved plants in the seed mixture to work in unison to combat reed canarygrass in the system and ensure it remains a minor component, when present.

Typha spp. add a similar, yet different, problem to restoration attempts in our region. There are three species of cattail found in the PPR, *Typha angustifolia* (narrowleaf cattail), *Typha latifolia* (broadleaf cattail), and their hybrid *Typha* × *glauca* (hybrid cattail). Broadleaf cattail is a native to this region, and is often found in deep marsh areas of natural wetlands (Stewart and Kantrud 1972). Narrowleaf cattail is of an unknown origin but is believed to have arrived here sometime in the past 200 years, and now is often found in wetlands where broadleaf is present (Smith 1987). This species can persist in both the shallow and deep marshes of wetlands, but tends to tolerate more variable water depths than broadleaf. These two species do have the potential to become invasive, but are much less likely to do so than their offspring. Hybrid cattail was once believed to be sterile, like most hybrids, but recently it is assumed to be only partially sterile (Travis et al. 2010). This hybrid cattail does not take over an area as rapidly as reed canarygrass does, as shown by Green and Galatowitsch (2001). Their study, however, was short term (4 months) and did not look at the impacts it can have on the community composition as time goes on. It has been shown to grow in a wider variety of water conditions than either parent (Waters and Shay 1990; 1992). Hybrid cattail influences native plant communities by producing an abundance of slowly decaying litter each season which can act to change soil characteristics such as nitrogen mineralization and soil organic matter (Farrer and Goldberg 2009). This buildup of litter then works against native species not only by changing soil characteristics, but also by producing a dense litter mat that shades out, or physically blocks native seedling emergence. Farrer and Goldberg (2009) showed a 98% reduction of light penetration through cattail litter buildup. Because of the influence this litter accumulation can have on the plant community, it is important to prevent such accumulation. Elimination of live plants is not enough to prevent the replacing of natives. Two commonly used approaches to reduce litter levels are prescribed fire and harvest. Prescribed fire is a cheap, efficient way to remove this layer but can be counterproductive if it's used at the wrong time or incorrect intensity (Thompson and Shay 1985). Harvesting is another option for litter removal, but is often costly and time-consuming on large scales making it an infeasible option (Larkin et al. 2012).

Very few restoration studies have been completed to date that include detailed explanations of costs. Cost of a restoration project is very important to consider, and often dictates whether a project is initiated or not. Studies on restoration cost estimates often fail to include all costs associated with the project. In an estimate of small (0.4 ha), seasonal wetland restorations, Zentner et al. (2003) noted their estimates excluded the additional costs of land acquisition; planning, permitting, and engineering; and monitoring and maintenance. Despite missing these important, often expensive pieces, they estimated restoration costs from $12,000 to $42,000/acre ($30,000 to $104,000/ha). These costs included native reseeding and land preparation only. Wetland construction costs were also evaluated on private hydric cropland in Missouri. These costs were estimated to range from $50 to $200/acre ($125 to $500/ha) (Prato et al. 1995). However, these prices have not been adjusted for inflation and only included the costs associated with earth work, water control structures, and grass seeding for erosion control. Estimates of mitigation projects also vary. In a survey of public works agencies, Zentner et al. (2003) found estimations for the creation/restoration/enhancement for mitigation ranges from $50,000 to $500,000/acre ($124,000 to $1,240,000/ha). These costs are highly dependent upon project design and location, and also differ greatly when completed by public vs. non-profit agencies (Zentner et al. 2003).

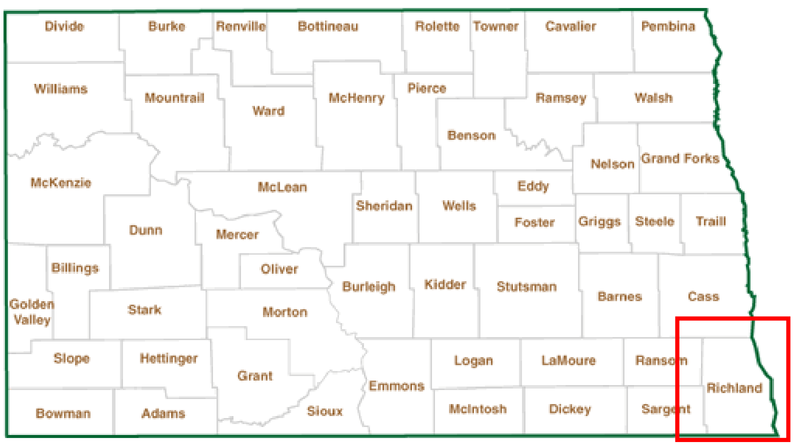
The use of native hay transfer to aid native seed establishment is becoming a frequently utilized practice in restoration attempts. Its use has been shown to increase species diversity and helps to suppress weedy species which can inhibit establishment of natives (Török et al. 2012). Hay mulch works to aid establishment in many ways. It prevents soil desiccation (Fowler 1988), buffers the soil from large temperature fluctuations (Foster and Gross 1998), may act as a physical barrier to wind dispersed weeds preventing them from making soil contact (Wedin and Tilman 1993), and also may exhibit some allelopathic effects (Ruprecht et al. 2010). In addition to the inhibition of weedy species early in the restoration, it can also act as an additional seed source to complement plantings and those found in the seedbank (Klimkowska et al. 2010).

In this study we introduce potential methods to reclaim wetlands invaded and/or completely dominated by invasive species such as cattail and reed canarygrass. The wetland we utilized for this project already contained invasive species when the project began. We also look at estimated total costs associated with the three different treatment methods outlined in this study: (1) seed only; (2) seed and native hay mulch; and (3) seed, native hay mulch, and transplanted vegetation plugs. We also address future research that can be done at these sites to estimate the cost efficiency of each treatment, and estimate the rate of spread of transplanted wet meadow soil plugs.

Methods

Study Site

This study was conducted on an 18.86 ha area located on the Albert Ekre Grassland Preserve in Richland County, North Dakota, approximately two kilometers east of the Sheyenne National Grasslands (Latitude 46.526224; Longitude -97.132370) (Figure 2.1). This area is characterized by sandy soils that originated from an ancient river that made its way through the area and emptied into glacial Lake Agassiz. Historically the study site was primarily native tallgrass prairie, but was at one time plowed and converted into cropland. Hydrologic regimes at this site fluctuate greatly from year to year and no type of drainage has been applied. Because of this, farming has occurred only four times in the past 19 years, with the land remaining fallow on years it was not farmed (Dewey Lindgren, personal communication). This eventually led to a dominance of weedy species and robust invaders that were able to thrive on this fallow ground.



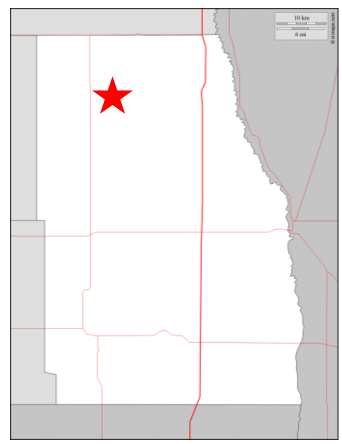


Figure 2.1: Location of the Ekre wetland restoration project in Richland County, southeastern North Dakota.

The major natural vegetation type in this area was tallgrass prairie, which was dominated by big bluestem (*Andropogon gerardii* Vitman), prairie sandreed (*Calamovilfa longifolia* (Hook.) Scribn.), sand bluestem (*Andropogon hallii* Hack.), needle-and-thread grass (*Hesperostipa comata* (Trin. and Rupr.) Barkworth), porcupine grass (*Hesperostipa spartea* (Trin.) Barkworth), sand dropseed (*Sporobolus cryptandrus* (Torr.) A. Gray), sideoats grama (*Bouteloua curtipendula* (Michx.) Torr.), prairie junegrass (*Koeleria macrantha* (Ledeb.) Schult.), Canada wildrye (*Elymus canadensis* L.), blue grama (*Bouteloua gracilis* (Willd. ex Kunth) Lag. ex Griffiths), and sedges (*Carex* spp.). Wetlands are typically dominated by woolly sedge (*Carex pellita* (Muhl.) Willd.), slough sedge (*Carex atheroides* Spreng.), fescue sedge (*Carex brevior* (Dewey) Mack., northern reedgrass (*Calamagrostis stricta* (Timm) Koeler), prairie cordgrass (*Spartina pectinata* (Bosc) Link), rushes, (*Juncus and Elocharis* spp.), other sedges (*Carex* spp.), and forbs. (NRCS ESIS, ESD 2015).

Soils of the wetland are described as: 97.2% Aylmer-Rosewood-Serden complex, 0-9% slopes with minor components: Ulen, Hamar, Bantry and Venlo series; 1.7% Serden-Hamar complex, 0-15% slopes with minor components: Bantry, Alymer, Ulen, and Venlo series; and 1.1% Garborg loamy fine sand, 0-1% slope with minor components: Hamar, Delamere, Mantador, Ulen, and Venlo. Aylmer soils are derived from wind-worked sandy glaciofluvial deposits and classified as subirrigated sands. Rosewood soils are derived from sandy glaciofluvial deposits and classified as wet meadow. Serden soils are derived from sandy eolian deposits and are classified as sands. Hamar soils are derived from sandy glaciofluvial deposits and classified as wet meadow. Garborg soils are derived sandy glaciofluvial deposits and/or sandy glaciolacustrine deposits and are classified as subirrigated (NRCS WSS 2015).

The climate of this area is considered humid continental (cool summer) and has a wide fluctuation in annual and daily temperatures. Precipitation is highly variable in amount as well as annual totals. Thirty year averages for the area put precipitation at 47.2 cm through the April to October growing season from 1981-2010 (USDOC NOAA 2015). Rainfall totals in 2013 were 53.2 cm, 6 cm above average, and in 2014, totals were 49.3 cm, 2.1 cm above average (NDAWN 2014).

Treatment Design

As soon as the study site was identified, a randomized complete block design experiment was set up. Approximately 11.04 of 18.86 ha were included in this experiment, and this area was divided into 9 equal plots, arranged in a 3 by 3 sequence. Each plot was approximately 1.23 ha in size. Three treatments were utilized, and one was randomly assigned to each block of each row.

These treatments were: seed only; seed and mulch; and seed, mulch, and transplanted soil plugs (Figure 2.2).

Seed Only	Seed/ Mulch	Seed/ Mulch/ Plugs
Seed/ Mulch	Seed Only	Seed/ Mulch/ Plugs
Seed/ Mulch/ Plugs	Seed/ Mulch	Seed Only

Figure 2.2: Experimental design for the Ekre wetland restoration in southeastern North Dakota.

Site Preparation

The study site was burned in the spring of 2013 to prepare the area for soybean planting and spraying with glyphosate. It was a dry spring so the wetland was able to be burned in its entirety. After the wetland was burned, approximately one week later the ground was prepared for planting. The site was run over with a salford vertical tillage tool (independently mounted discs, harrow, and rolling basket to firm the ground) three times, in order to remove any large hummocks or mounds that were present from decades of cropping, sitting fallow, and being inundated. Seeding of Roundup Ready soybeans immediately followed this process.

Soybean Seeding and Glyphosate Applications

During the summer of 2013 the wetland was planted with Roundup Ready soybeans and treated with 3 applications of glyphosate (Roundup PowerMax®, Monsanto Company, St. Louis, MO, 63167) at equal intervals for the duration of the growing season. Application rates were consistent with recommended use of 1.54 kg/ha. Any areas that were not sprayed during crop application of glyphosate were completed by ATV to ensure maximum coverage (primarily fence-lines and unplanted areas). One application was also applied pre-planting and post-harvest for five total in the summer of 2013. Two additional treatments were applied in early summer 2014 prior to seeding of native vegetation for seven applications overall.

Native Vegetation Seeding

Native seeding was completed during late July to mid-August 2014. The area was seeded two separate times due to the different size seed found in the separate mixes (Appendix C). Once was with larger grass-like seed and a second subsequently with the much smaller sedge and rush-like seed. Seeding rates were 8.03 kg/ha for the larger mix and 1.51 kg/ha for the smaller, as recommended by the supplier. Each seed mix contained species that could survive the spectrum of hydrological conditions present, semi-permanently inundated to upland, allowing the same mix to be seeded across the entire wetland. The reason seeding was completed so late in the season was twofold; the primary reason was water levels in the wetland. High water levels left over 50% of the wetland area inaccessible by tractor and prevented the possibility of planting. Secondary reasons for the late planting had to do with the timing of herbicide application. One treatment of glyphosate was applied in early June and again in early July 2014 with 4 weeks between sprayings. We waited 1 to 2 weeks in-between the final spraying and seeding.

Soil Vegetation Transplant

Soil vegetation plugs were obtained from “reference” wet meadow areas identified by the US Forest Service from the Sheyenne National Grasslands in Richland County, ND, and other privately owned wet meadows near to the study site. Sites were visited and desirable species for transplant were identified and targeted. Any areas with invasive or undesired plants were excluded from the sampling areas. Cores were taken using a standard golf-hole cutter 10 cm in diameter to a depth of at least 15 cm so as to obtain root material. Three of the nine treatment plots were randomly assigned to receive the plugs, and 1,122 plugs were transplanted into each for a total of 3,366 across the entire site in a 33 by 34 plug configuration covering 4,224 m^2 per treatment (Figure 2.3). These cores were transplanted in a square-like grid following the cardinal directions with 2 m separating columns East-West, and 2 m separating rows North-South. Transplanting began at the geographic center of each plot and 16 additional rows were added to the North and to the South for a total of 33 rows. A one meter East-West offset was included between each row to form an isometric dot pattern. Soil was removed to allow transplant utilizing the same technique to obtain the plugs and the excess soil plug was scattered nearby.

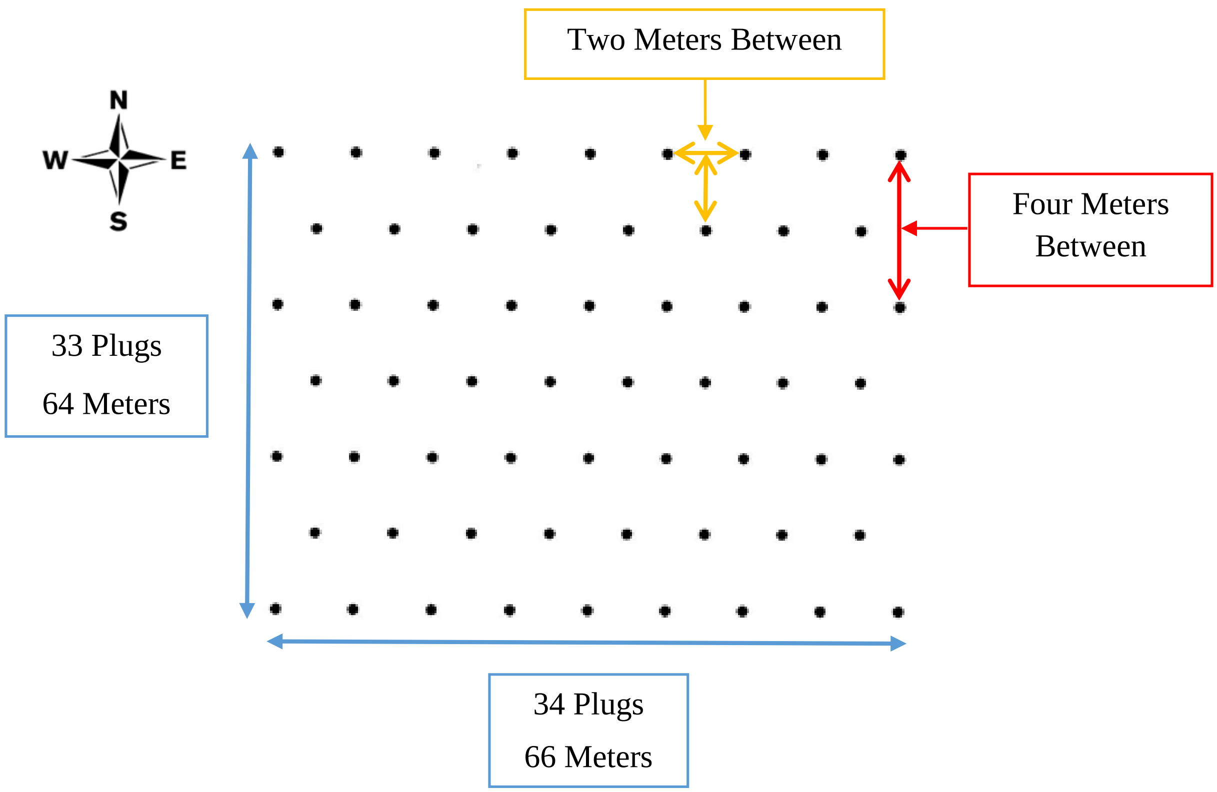


Figure 2.3: Treatment design for the soil vegetation transplant at the Ekre wetland restoration in southeastern North Dakota.

Hay Transfer

Hay used in the hay transfer procedure was collected in late July 2014 from nearby "reference" wet meadow areas located in the Sheyenne National Grasslands and other privately owned wet meadows near to the study site. These sites were assessed and selected prior to cutting of the hay to ensure the quality of the hay and to confirm it contained desirable species. Hay was collected at this time for maximum seed quantity and quality. Once cut and allowed to dry for approximately a week, the hay was bailed into large round bales with net wrapping to prevent potential seed loss. In early September, hay bales were put through a bale shredder, and the resulting hay mulch was spread out in a layer approximately 5-7 cm thick as suggested by Klimkowska et al. (2010). There was variation in mulch depth across the plots due to uneven vegetation heights catching mulch, but it was spread to maximize uniformity.

Results

Results were calculated to determine the total costs of each treatment per acre, and then extrapolated to total per hectare. The price for each treatment type was estimated to be approximately $1,143/ ha, $379/ha, and $2,803/ha for seeding, hay mulching, and soil transplant respectively. Associated costs are broken down and discussed further in Table 1. Additional costs that are included in totals for each treatment are costs/ha for burning, additional herbicide application applied before native seed was planted, and soybean planting and related costs. These estimated costs come out to approximately $102, $242, and $476/ha, respectively. Labor rates are assessed at $12.00 per hour. Final estimate for cost per treatment per hectare is as follows: $1,963 for native seeding only; $2,342 for native seed and hay mulch; and $5,145 for native seed, hay mulch, and transplanted soil plugs.

Table 1: Costs associated with the seeding, hay mulch, and soil transplant treatments applied to the wetland restoration in southeastern North Dakota.

	Summer 2013	Summer 2014
Burning Total[a]	$ 1,900.00	$ -
Burning-Man Hours	$ 960.00	$ -
Burning-Preparation	$ 440.00[b]	$ -
Burning-Fire Crew Rental	$ 500.00[c]	$ -
Herbicide Application Total	$ 4,134.00	$ 1,704.00
Herbicide-Spot Spraying Hours	$ 864.00	$ 504.00
Herbicide-Soybean Applications	$ 1,920.00[d]	$ -
Herbicide-Equipment Rental[e]	$ 600.00	$ 600.00
Herbicide-Spot Spraying Herbicide	$ 750.00	$ 600.00
Planting Total[f]	$ 5,776.00	$ 18,504.00
Planting-Yield Totals	$ 4,036.50[g]	$ -
Planting-Soybean Harvest Cost	$ 1,200.00	$ -
Planting-Soybean Land Prep	$ 840.00	$ -
Planting-Seed Drill & Tractor Rental	$ 856.00	$ 3,504.00[h]
Planting-Seed Cost	$ 2,880.00	$ 15,000.00
Mulching Total	$ -	$ 1,140.00
Mulching-Hours	$ -	$ 640.00[i]
Mulching-Bale Shredder Rental	$ -	$ 500.00
Soil Vegetation Transplant Total	$ -	$ 3,552.00
Transplant-Hours	$ -	$ 2,952.00[j]
Transplant-Equipment Rental	$ -	$ 600.00
Mowing Total	$ 2,560.00	$ 640.00
Mowing-Tractor & Mower Rental[k]	$ 2,560.00	$ 640.00
Native Hay Bale Total	$ -	$ 1,650.00[l]
Total Cost	$ 14,370.00	$ 27,190.00
Total Revenue	$ 4,036.50	
Grand Total		**$ 37,523.50**

[a]Approximately forty-six acres burned.
[b]Creation of a disked fireguard around the wetland prior to burning.
[c]Rental of a 3 man fire crew from McCleod, ND.
[d]Three applications of herbicide with all associated costs included.
[e]Cost for renting an ATV for large scale spot spraying operations.
[f]Approximately forty acres planted with soybeans and native seed.
[g]Revenue for harvested soybeans.
[h]Drill and tractor rental costs based on hourly rates for both.
[i]Tractor rental costs for total hours.
[j]Total cost for all man hours involved.
[k]Total for tractor and mower rental hours.
[l]Fifty native hay bales at $33 each.

Discussion

As predicted, total cost goes up as treatment level increases. The more labor and time intensive the treatment, the more it costs per hectare to restore. Since only one year of Roundup Ready cropping was utilized in this study due to time constraints, our cropping costs are less than would be predicted if 2-3 years were utilized. However, if the land is leased for cropping, this could lower the costs by removing cropping costs entirely and possibly saving money depending upon the terms of the lease. At this time we cannot make a certain conclusion as to the effectiveness of any treatment, but we predict the most intensive treatment (i.e. Seed, Mulch, and Plugs) will contain a higher diversity of desirable plant species, based on initial observations at the conclusion of work on the treatment sites in Fall 2014.

The estimates we make here are likely to vary greatly with location of the restoration which influences the choice of native seed mix. Pay rates may also be different than those utilized in this study. Hourly wages were estimated at $12.00 per hour, and were completed by a graduate student. These rates are unlikely to carry over when work is completed by professionals. Seeding methods may also vary greatly between restorations. Hydroseeding often is a more expensive process than regular drill seeding, and the transplant of live root material is more expensive than either method individually. Zentner et al. (2003) estimated hydroseeding for their restoration to cost $4,000/acre ($9,900/ha), and transplanting of live plants to be $8,500 to $34,000/acre ($21,000 to $84,000/ha). However, these costs took into account a variety of transplant "plug" sizes at 0.6 to 0.9 meter planting intervals. The drastic difference in transplanting cost compared to our study comes mainly from the cost of obtaining plugs. Soil plugs in our study were donated versus having to be purchased, as in Zentner's study at $6.00 per plug. While our results may differ from these other studies, we offer one estimate as to overall project costs. Contingent upon the situation, project costs may differ drastically from those offered here.

The use of hay mulch as a source of seed is a commonly used practice. Studies done in Europe (Kiehl et al. 2006; Edwards et al. 2007; Klimkowska et al. 2010), Canada (Desserud and Naeth 2011), and the United States (Gates 1962) have supported its use as a beneficial source of native seed. The type of hay transferred is also an important factor. The use of either dried or wet, fresh cut hay has also been utilized. In an evaluation of many restoration projects across Europe, Kiehl et al. (2010) found many studies that utilized wet hay transfer instead of dry. This wet hay has potentially higher seed content and is suggested to be much more promising in restoration than using dry hay (Rasran et al. 2006). Kiehl et al. (2010) suggested the use of dry hay as a seed source is limited because the cutting, turning, and swathing necessary to dry hay will reduce the seed content considerably. The results of our study have yet to be determined, but dry hay has been shown to aid in the establishment and development of seeded native species (Kirmer et al. 2012; Török et al. 2012), and also have no impact on their establishment (Bakker et al. 2003).

The transplant of wetland soil as a restoration technique is a relatively new method which is gaining popularity. Transplant of the seed bank and root material in the soil gives these species a jump start on establishment. This is especially useful for species that have limited dispersal and/or colonize slowly. Fraser and Kindscher (2001), found sod transplanting to be an effective method for vegetative restoration with species that spread quickly (*Eleocharis macrostachya*) and those that have limited reproduction from seed alone (*Spartina pectinata*). These transplanted rhizomes are able to rapidly spread once they have been replanted. Many plant species have been shown to reproduce rapidly through clonal growth (Clevering and van Guilik

1997; Brewer et al. 1998), and these growth rates have been suggested to continue for decades (Frenkel and Boss 1988). Fraser and Kindscher (2001), found their transplanted vegetation plugs to have exponential growth over a four year period.

Consideration of which species to transplant is also important. While some wet meadow species can naturally disperse from adjacent wetlands, important species such as *Carex* spp. and *Spartina pectinata* have been shown to have limited dispersal and poor reproduction from seed (Schütz 2000). Budelsky and Galatowitsch (1999), found that *Carex* spp. seeds have a specific set of conditions that are necessary for germination to occur, and even when those are met, germination rates can still be at best approximately 50%. Because of this, restoration of naturally occurring wet meadow vegetation is difficult to complete. Galatowitsch and van der Valk (1996b) found that after restoration had occurred, reference like wet prairie and sedge meadow zones had failed to appear on many of their sites. The depletion of these species from the landscape, and possibly the seedbank, following degradation is evident in their analysis. These species are important to the wet prairie/meadow medley, so they need to be restored on the landscape. Fraser and Kindscher (2005) recommend planting many small (20 cm diameter) soil plugs to reestablish difficult wet meadow species such as *Spartina pectinata*, and to maximize area in the restoration. Their study showed promising results with plugs becoming established and increasing in area, although densities and stem counts were still less than reference populations.

The removal of soil for transplanting can have significant impacts on the original site from which the soil is taken. This removal can leave holes in the landscape mosaic which may be utilized by rapidly spreading exotic species to establish a foothold in the landscape, or create an area that is prone to soil erosion (Davis and Short 1997). Few, if any, studies have been done on the impacts of soil plug removal on donor sites, suggesting this is an area where more research may be needed. One way to alleviate these effects is to salvage vegetation from native areas destined for impact, and utilizing the vegetation in mitigation restorations for better native establishment (Fraser and Kindscher 2001).

The economics of hydric cropland to wetland conversion is a topic that has not been extensively examined. One possible reason for this is the difficulty of estimating the value of services provided by wetlands. Costanza et al. (1997) completed an estimate of the value of wetland services, but not many studies have been completed for individual wetlands. Wetland functioning, and therefore value, may vary drastically across a small landscape, further complicating this valuation process. A geoeconomic study of the conversion of hydric cropland to wetland in Missouri was completed by Prato et al. (1995), and showed as long as benefits of the restoration remain high, it is more profitable to the landowner than continued cropping. The only benefits they acknowledged of restoring these wetlands, however, was money that landowner could make from leasing the land to waterfowl hunters. Hammack and Brown (1974) looked at the value of wetlands by putting a monetary value on the waterfowl produced from them, and the loss of revenue from waterfowl if these wetlands were to disappear from the landscape.

Management Implications

What is often lacking in many restoration attempts is a post management plan. We look to conclude this study by offering insight as to what we believe to be an effective management strategy to monitor the progress, and determine if intervention is necessary to ensure its continued success. No large scale follow-up study should be done for at least five years post restoration to allow the plant community to stabilize. However, the site should be visited regularly to ensure that undesirable species are prevented from dominating the trials and data should be gathered to track species establishment. Direct intervention and control may need to be used if it is found invasive species are encroaching. As there are pockets of woody species found along the edges of the wetland, it may be necessary to implement a burn plan and burn the wetland frequently (1-2 years) to prevent their encroachment after seeded species are established.

We suggest starting in summer of 2016 to complete an analysis of the treatment areas. These early samplings (< 5 years post restoration) can give vital information on early succession in restored wetlands, and allow monitoring for future restorations. With these data it may be possible to predict trends in future projects and provide information on when/if intervention is needed to keep a project on track. We predict the first few years post restoration will result in the area being covered with many annual weeds prior to perennial grasses and grass-likes establishing, as discovered by Rowe (2010). For sampling protocol, we recommend starting the center of each plot and moving out 2 m from center in the four cardinal directions for each additional point, and taking modified Daubenmire readings with a .25 m^2 quadrat utilizing the six cover classes. Each treatment will have 17 readings, four from each length away from center and one from the center to determine species composition and cover (Figure 2.4). These results can then be analyzed with a standard ANOVA to determine if there are significant differences between composition and cover across treatments.

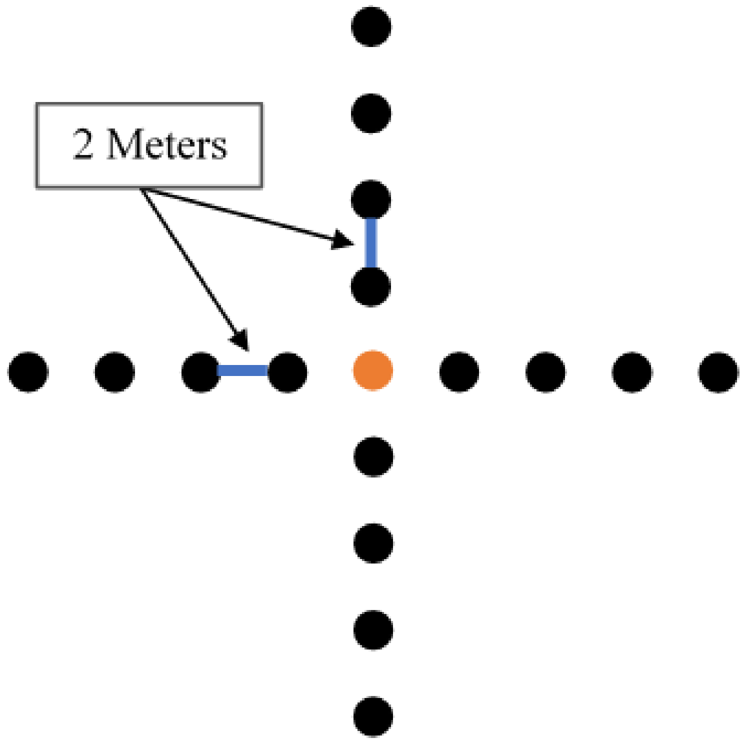


Figure 2.4: Vegetation sampling design for future studies to be completed on the Ekre wetland restoration in southeastern North Dakota.

In order to address the “success” or “failure” of this project, it will also be necessary to locate nearby “reference” condition wetlands that can be used to compare both species diversity and cover to that of our restored wetland. Modified Daubenmire quadrats also may be used at randomly selected sites around chosen wetlands, ensuring there are enough readings relative to its size. Once all samples have been collected, the restored and “reference” wetlands can be compared. We recommend using Non-metric Multidimensional Scaling (NMS) to compare each treatment to “reference” conditions to see if any of the utilized restoration methods shows a significant trend towards “reference” condition.

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APPENDIX A. GPS COORDINATES OF SEEDBANK SAMPLING LOCATIONS FOR SEEDBANK PAPER

Point Name*	X	Y	Point Name	X	Y
1-1-1-1	643074.5247	5154378.634	3-3-1-1	643280.161	5154379.051
1-1-1-2	643106.1265	5154379.158	3-3-1-2	643310.0704	5154379.759
1-1-1-3	643136.5359	5154381.434	3-3-1-3	643340.773	5154379.374
1-1-1-4	643166.6061	5154378.587	3-3-1-4	643371.4756	5154378.99
1-1-2-1	643073.0608	5154352.917	3-3-2-1	643280.0517	5154351.254
1-1-2-2	643105.2337	5154352.01	3-3-2-2	643310.7281	5154351.98
1-1-2-3	643135.6507	5154350.728	3-3-2-3	643340.6639	5154351.577
1-1-2-4	643166.3008	5154352.565	3-3-2-4	643371.3667	5154351.193
1-1-3-1	643073.5771	5154324.58	3-3-3-1	643279.916	5154324.569
1-1-3-2	643105.745	5154323.895	3-3-3-2	643310.6189	5154324.184
1-1-3-3	643135.2654	5154321.591	3-3-3-3	643341.2955	5154324.91
1-1-3-4	643166.7278	5154324.781	3-3-3-4	643371.2315	5154324.507
1-1-4-1	643071.6052	5154297.628	3-3-4-1	643279.8066	5154296.772
1-1-4-2	643106.3859	5154296.783	3-3-4-2	643310.4833	5154297.498
1-1-4-3	643136.8588	5154296.392	3-3-4-3	643341.1864	5154297.113
1-1-4-4	643166.2901	5154297.866	3-3-4-4	643371.1225	5154296.711
1-1-5-1	643074.7979	5154269.687	3-3-5-1	643279.6709	5154270.086
1-1-5-2	643103.7954	5154270.039	3-3-5-2	643310.3741	5154269.701
1-1-5-3	643132.2823	5154269.268	3-3-5-3	643341.051	5154270.428
1-1-5-4	643166.114	5154269.623	3-3-5-4	643371.7541	5154270.043
2-2-1-1	643177.3428	5154378.841	4-2-1-1	643075.0342	5154259.687
2-2-1-2	643208.0191	5154379.567	4-2-1-2	643104.9705	5154259.283
2-2-1-3	643237.9548	5154379.164	4-2-1-3	643135.6474	5154260.008
2-2-1-4	643268.6574	5154378.779	4-2-1-4	643166.3505	5154259.623
2-2-2-1	643177.2067	5154352.156	4-2-2-1	643074.9239	5154231.891
2-2-2-2	643207.9094	5154351.77	4-2-2-2	643105.6009	5154232.616
2-2-2-3	643238.6122	5154351.385	4-2-2-3	643135.5373	5154232.212
2-2-2-4	643268.5217	5154352.093	4-2-2-4	643166.2143	5154232.937
2-2-3-1	643177.0969	5154324.359	4-2-3-1	643074.7873	5154205.205
2-2-3-2	643207.7734	5154325.085	4-2-3-2	643105.4907	5154204.819
2-2-3-3	643238.4763	5154324.7	4-2-3-3	643136.1679	5154205.544
2-2-3-4	643269.1792	5154324.314	4-2-3-4	643166.1044	5154205.141
2-2-4-1	643176.9608	5154297.674	4-2-4-1	643074.677	5154177.408
2-2-4-2	643207.6637	5154297.288	4-2-4-2	643105.3543	5154178.134
2-2-4-3	643238.3668	5154296.903	4-2-4-3	643136.0578	5154177.748
2-2-4-4	643269.0435	5154297.629	4-2-4-4	643165.9682	5154178.455
2-2-5-1	643177.6178	5154269.895	4-2-5-1	643074.5403	5154150.723
2-2-5-2	643207.5278	5154270.603	4-2-5-2	643105.244	5154150.337
2-2-5-3	643238.2309	5154270.217	4-2-5-3	643135.9215	5154151.062
2-2-5-4	643268.934	5154269.832	4-2-5-4	643166.6252	5154150.677

Point Name	X	Y
5-1-1-1	643177.0874	5154259.877
5-1-1-2	643207.7906	5154259.491
5-1-1-3	643238.4675	5154260.217
5-1-1-4	643269.1707	5154259.832
5-1-2-1	643176.9513	5154233.191
5-1-2-2	643207.6546	5154232.806
5-1-2-3	643238.358	5154232.42
5-1-2-4	643269.035	5154233.146
5-1-3-1	643177.6084	5154205.413
5-1-3-2	643207.5449	5154205.009
5-1-3-3	643238.2221	5154205.735
5-1-3-4	643268.9256	5154205.35
5-1-4-1	643177.4722	5154178.727
5-1-4-2	643208.1758	5154178.342
5-1-4-3	643238.1125	5154177.938
5-1-4-4	643268.7898	5154178.664
5-1-5-1	643177.3623	5154150.931
5-1-5-2	643208.0661	5154150.545
5-1-5-3	643237.9766	5154151.253
5-3-5-4	643268.6803	5154150.868
6-3-1-1	643279.9077	5154260.086
6-3-1-2	643310.6109	5154259.701
6-3-1-3	643341.2878	5154260.428
6-3-1-4	643371.2241	5154260.025
6-3-2-1	643279.7983	5154232.289
6-3-2-2	643310.4753	5154233.016
6-3-2-3	643341.1787	5154232.631
6-3-2-4	643371.1151	5154232.228
6-3-3-1	643279.6626	5154205.604
6-3-3-2	643310.3661	5154205.219
6-3-3-3	643341.0432	5154205.945
6-3-3-4	643371.7467	5154205.561
6-3-4-1	643280.3201	5154177.825
6-3-4-2	643310.2305	5154178.533
6-3-4-3	643340.9341	5154178.149
6-3-4-4	643371.6378	5154177.764
6-3-5-1	643280.1844	5154151.14
6-3-5-2	643310.1212	5154150.737
6-3-5-3	643340.7987	5154151.463
6-3-5-4	643371.5024	5154151.079
7-3-1-1	643074.7767	5154140.723
7-3-1-2	643104.7135	5154140.318
7-3-1-3	643135.391	5154141.044

Point Name	X	Y
7-3-1-4	643166.0948	5154140.658
7-3-2-1	643074.6401	5154114.037
7-3-2-2	643104.577	5154113.633
7-3-2-3	643135.2809	5154113.247
7-3-2-4	643165.9586	5154113.973
7-3-3-1	643074.5297	5154086.24
7-3-3-2	643105.2337	5154085.854
7-3-3-3	643135.1445	5154086.562
7-3-3-4	643165.8486	5154086.176
7-3-4-1	643074.393	5154059.555
7-3-4-2	643105.0972	5154059.169
7-3-4-3	643135.8014	5154058.783
7-3-4-4	643165.7123	5154059.49
7-3-5-1	643074.2826	5154031.758
7-3-5-2	643104.9869	5154031.372
7-3-5-3	643135.665	5154032.098
7-3-5-4	643165.6024	5154031.694
8-2-1-1	643176.8319	5154140.912
8-2-1-2	643207.5357	5154140.527
8-2-1-3	643238.2132	5154141.252
8-2-1-4	643268.917	5154140.867
8-2-2-1	643177.489	5154113.134
8-2-2-2	643207.3997	5154113.841
8-2-2-3	643238.1036	5154113.456
8-2-2-4	643268.8075	5154113.071
8-2-3-1	643177.3528	5154086.448
8-2-3-2	643207.2899	5154086.044
8-2-3-3	643237.9677	5154086.77
8-2-3-4	643268.6717	5154086.385
8-2-4-1	643177.2428	5154058.651
8-2-4-2	643207.9208	5154059.377
8-2-4-3	643237.858	5154058.974
8-2-4-4	643268.5622	5154058.589
8-2-5-1	643177.1066	5154031.966
8-2-5-2	643207.811	5154031.58
8-2-5-3	643238.489	5154032.306
8-2-5-4	643268.4264	5154031.903
9-1-1-1	643280.4211	5154141.14
9-1-1-2	643310.358	5154140.737
9-1-1-3	643341.0355	5154141.463
9-1-1-4	643371.7393	5154141.078
9-1-2-1	643280.2854	5154114.454
9-1-2-2	643310.9893	5154114.069

Point Name	X	Y
9-1-2-3	643340.9263	5154113.666
9-1-2-4	643371.604	5154114.393
9-1-3-1	643280.1759	5154086.657
9-1-3-2	643310.88	5154086.273
9-1-3-3	643340.7909	5154086.981
9-1-3-4	643371.495	5154086.596
9-1-4-1	643280.0402	5154059.972

Point Name	X	Y
9-1-4-2	643310.7444	5154059.587
9-1-4-3	643341.4486	5154059.202
9-1-4-4	643371.3596	5154059.911
9-1-5-1	643279.9307	5154032.175
9-1-5-2	643310.6351	5154031.79
9-1-5-3	643341.3131	5154032.517
9-1-5-4	643372.0175	5154032.132

*All point coordinates are in UTM NAD 1983 zone 14N in meters.

APPENDIX B. SPECIES LIST AND SEEDLING COUNTS FOR SEEDBANK PAPER

Species Name[a]	2013[b]	2014[b]	Origin	Lifespan	Growth
Alisma gramineum	38	1	Native	Perennial	Forb
Amaranthus albus	17	0	Introduced	Annual	Forb
Amaranthus retroflexus	3	0	Native	Annual	Forb
Ambrosia artemisiifolia	2	0	Native	Annual	Forb
Ammannia robusta	141	690	Native	Annual	Forb
Androsace occidentalis	0	2	Native	Annual	Forb
Arctium lappa	0	1	Introduced	Biennial	Forb
Artemisia biennis	1	0	Introduced	Annual, Biennial	Forb
Berteroa incana	87	33	Introduced	All	Forb
Bidens cernua	1	0	Native	Annual	Forb
Calamagrostis stricta	0	1	Native	Perennial	Graminoid
Carex spp.[c]	263	179	Native	Perennial	Graminoid
Chenopodium album	16	29	Introduced	Annual	Forb
Chenopodium rubrum	1	1	Native	Annual	Forb
Cirsium arvense	4	0	Introduced	Perennial	Forb
Conyza canadensis	1	17	Native	Annual	Forb
Cyperus acuminatus	637	0	Native	Annual	Graminoid
Cyperus erythrorhizos	18,064	0	Native	Annual	Graminoid
Cyperus squarrosus	2,432	0	Native	Annual	Graminoid
Echinochloa crus-galli	34	5	Introduced	Annual	Graminoid
Eleocharis acicularis	4,574	5,519	Native	Perennial	Graminoid
Eleocharis macrostachya	1	1	Native	Perennial	Graminoid
Eleocharis obtusa	49	28	Native	Annual	Graminoid
Epilobium ciliatum	81	11	Native	Perennial	Forb
Gratiola neglecta	4	6	Native	Annual	Forb
Hordeum jubatum	12	10	Native	Perennial	Graminoid
Hypericum majus	498	1,722	Native	Annual	Forb
Iva annua	1	0	Native	Annual	Forb
Juncus interior	3,707	3,613	Native	Perennial	Graminoid
Lindernia dubia	3,034	5,536	Native	Annual	Forb
Lycopus americanus	386	61	Native	Perennial	Forb
Melilotus officinalis	3	1	Introduced	All	Forb
Nepeta cataria	0	1	Introduced	Perennial	Forb
Oenothera biennis	2	0	Native	Biennial	Forb
Panicum miliaceum	88	0	Introduced	Annual	Graminoid
Panicum virgatum	2	2	Native	Perennial	Graminoid
Phalaris arundinacea	0	4	Introduced	Perennial	Graminoid
Plantago major	0	4	Introduced	Perennial	Forb
Polygonum pensylvanicum	862	519	Native	Annual	Forb
Polygonum ramosissimum	14	0	Native	Annual	Forb
Potentilla norvegica	12,038	1,639	Native	Annual, Biennial	Forb
Potentilla paradoxa	8,230	0	Native	Annual, Perennial	Forb

Species Name[a]	2013[b]	2014[b]	Origin	Lifespan	Growth
Ranunculus longirostris	0	42	Native	Perennial	Forb
Ranunculus pensylvanicus	0	1	Native	Annual, Perennial	Forb
Ranunculus sceleratus	82	104	Native	Annual	Forb
Rorippa palustris	218	189	Native	All	Forb
Rotala ramosior	438	229	Native	Annual	Forb
Rumex crispus	572	28	Introduced	Perennial	Forb
Salicornia rubra	16	250	Native	Annual	Forb
Schoenoplectus acutus	397	0	Native	Perennial	Graminoid
Sonchus arvensis	3	1	Introduced	Perennial	Forb
Symphyotrichum lanceolatum	15	0	Native	Perennial	Forb
Typha spp.	931	1,215	Introduced	Perennial	Forb
Verbena bracteata	1	0	Native	All	Forb
Verbena hastata	1,149	1	Native	Perennial	Forb
Veronica peregrina	23	821	Native	Annual	Forb
Seedling Totals	**59,172**	**22,519**			

[a]Species names are from The PLANTS Database: USDA, NRCS. 2015. The PLANTS Database (http://plants.usda.gov, 9 March 2015). National Plant Data Team, Greensboro, NC 27401-4901 USA.
[b]Seedling totals per m^2
[c]Multiple sedge species possible.

APPENDIX C. SPECIES LIST AND PERCENT COMPOSITION OF DRILLED NATIVE SEED MIXES FOR RESTORATION COSTS PAPER

Large Seed Mix

Variety Name	**Scientific Name[a]**	**Percent Mix**
Certified Rosana Western Wheatgrass	*Pascopyrum smithii*	4.91%
Certified Revenue Slender Wheatgrass	*Elymus trachycaulus*	4.84%
Bison Big Bluestem	*Andropogon gerardii*	20.34%
Pierre Sideoats Grama	*Bouteloua curtipendula*	11.42%
Bad River Blue Grama	*Bouteloua gracilis*	0.62%
Goshen Prairie Sandreed	*Calamovilfa longifolia*	8.30%
Certified Mandan Canada Wildrye	*Elymus canadensis*	5.75%
Sunburst Switchgrass	*Panicum virgatum*	7.10%
Itasca Little Bluestem	*Schizachyrium scoparium*	6.12%
Tomahawk Indiangrass	*Sorghastrum nutans*	9.44%
Red River Prairie Cordgrass	*Spartina pectinata*	4.82%
Needle and Thread	*Hesperostipa comata*	4.14%

Small Seed Mix

Variety Name	**Scientific Name[a]**	**Percent Mix**
MN Native Purple Prairie Clover	*Dalea purpurea*	47.60%
American Sloughgrass	*Beckmannia syzigachne*	5.29%
AK Native Canada Bluejoint	*Calamagrostis canadensis*	0.60%
SD Native Slimstem Reedgrass	*Calamagrostis stricta*	2.69%
IA Native Prairie Sedge	*Carex prairea*	9.72%
IA Native Plains Oval Sedge	*Carex brevior*	0.34%
SD Native Pale Sedge	*Carex pallescens*	3.97%
WI Native Porcupine Sedge	*Carex hystericina*	4.64%
SD Native Smoothcone Sedge	*Carex laeviconica*	1.04%
MN Native Woolly Sedge	*Carex pellita*	0.56%
Brown Fox Sedge	*Carex vulpinoidea*	3.90%
OR Native Creeping Spike Rush	*Eleocharis fallax*	3.00%
SD Native Reed Manna Grass	*Glyceria maxima*	2.13%
Prairie Junegrass	*Koeleria macrantha*	0.29%
MN Native Green Muhly	*Muhlenbergia ramulosa*	0.45%
SD Native Pale Bulrush	*Scirpus pallidus*	0.32%
SD Native Three Square Bulrush	*Schoenoplectus pungens*	6.35%
SD Native Sand Dropseed	*Sporobolus cryptandrus*	0.62%

[a]Species names are from The PLANTS Database: USDA, NRCS. 2015. The PLANTS Database (http://plants.usda.gov, 9 March 2015). National Plant Data Team, Greensboro, NC 27401-4901 USA.