5-2020

Carbon sequestration and weedy invasion in CRP fields

Destiny Danielle Magee
University of Northern Iowa

Recommended Citation

Magee, Destiny Danielle, "Carbon sequestration and weedy invasion in CRP fields" (2020). *Dissertations and Theses @ UNI*. 1030.
https://scholarworks.uni.edu/etd/1030

CARBON SEQUESTRATION AND WEEDY INVASION IN CRP FIELDS

An Abstract of a Thesis

Submitted

in Partial Fulfillment

of the Requirements for the Degree

Master of Science

Destiny Danielle Magee

University of Northern Iowa

May 2020

ABSTRACT

The Conservation Reserve Program (CRP) aims to provide ecosystem services in landscapes dominated by agriculture by converting previous agricultural fields into reconstructed prairies and wetlands. The main priorities of the program include reducing soil erosion and improving water quality, but the perennial vegetation in the fields can also provide other benefits such as increased carbon sequestration and biodiversity. These additional benefits of the program are not closely monitored, though these benefits are potentially equally valuable to the targeted program benefits. This study compared soil carbon in CRP fields and a subset of corn fields in order to determine the impact of reconstructed prairies on carbon sequestration in the landscape. Results showed that CRP fields decrease soil bulk density and increase carbon sequestration over time when compared to corn fields. This study also looked at the susceptibility of CRP fields to invasion by the weedy invasive species reed canary grass (*Phalaris arundincea*) and wild parsnip (*Pastinaca sativa*). CRP fields increase the biodiversity in the landscape, which can be beneficial to natives, but also to weedy species. Results show that increased grass species richness and grass percent cover decreased the chances of reed canary grass presence in a CRP field. Likewise, increased forb species richness and forb percent cover decreased the chances of *Pastinaca* being present in a field. This result suggests that increased biodiversity helps decrease the chances of weedy invasion in these CRP fields. Overall, conclusions show that CRP fields have the potential to bring ecosystem services to agricultural landscapes that are not targeted by the program.

CARBON SEQUESTRATION AND WEEDY INVASION IN CRP FIELDS

A Thesis

Submitted

in Partial Fulfillment

of the Requirements for the Degree

Master of Science

Destiny Danielle Magee

University of Northern Iowa

May 2020

This Study by: Destiny Danielle Magee

Entitled: CARBON SEQUESTRATION AND WEEDY INVASION IN CRP FIELDS

has been approved as meeting the thesis requirement for the

Degree of Master of Science in Biology

Date	Dr. Kenneth Elgersma, Chair, Thesis Committee
Date	Dr. Ai Wen, Thesis Committee Member
Date	Dr. Mark Myers, Thesis Committee Member
Date	Dr. Jennifer Waldron, Dean, Graduate College

ACKNOWLEDGEMENTS

I would like to express my great appreciation to my advisor Dr. Kenneth Elgersma and to Dr. Ai Wen for all their valuable and constructive suggestions and help during the planning and development of this research work. Your help was invaluable to completing this work. I would also like to thank Dr. Mark Myers for his help with revising this thesis and willingness to be on my committee, especially with short notice. I would also like to thank Kate Madson, Alec Glidden M.J. Lashbrook and Cody Welcher for their help with data collection; your help was greatly appreciated.

TABLE OF CONTENTS

PAGE

LIST OF TABLES

LIST OF FIGURES

PAGE

CHAPTER ONE

INTRODUCTION

The Conservation Reserve Program (CRP) was created in 1985 by the United States Department of Agriculture (USDA) as a way of mitigating the negative effects of agricultural production and improving environmental health. In exchange for a yearly-payment, farmers agree to take marginal or environmentally sensitive agricultural lands out of agricultural production and to plant them with beneficial plant species according to the guidelines of the conservation program they enroll in.

These contracts typically last between 10-15 years and have potential to be renewed. By planting and establishing long-term vegetation cover on environmentally sensitive or low producing crop land, the CRP program aims to bring a variety of beneficial ecosystem services to the agricultural landscape. Ecosystem services that are targeted by the program include, conserving soil, improving air and water quality and creating wildlife habitat.

The CRP program authorizes several different "conservation practices". The following conservation practices were assessed as a part of this research: CP-1, CP-2, CP-10, CP-4D, CP-23, CP-25, CP-33, CP-37 CP-38 and CP-42. Each conservation practice emphasizes a different ecosystem service or environmental benefit and consequently has different requirements and conditions as part of their enrollment in the CRP program (FSA 2017).

The CP-1 practice is the establishment of permanent introduced grasses and legumes. The goals of this program are to reduce soil erosion, increase water quality, and to create or improve wildlife habitat. Requirements of this program include that noxious weeds be controlled and that if the field is grazed or hayed during the contract, it must only be 1 out of every 3 years.

The CP-2 practice is the establishment of permanent native grasses. The goals of this program include to establish or maintain new existing cover to create habitat for wildlife as well as to prevent soil loss and maintain water quality. Requirements of being in this program include no grazing or haying through the life of the contract and that noxious weeds to be controlled.

The CP-10 practice is the maintenance of already established grasses or forbs and shrubs. The main priorities of this program are to create or maintain wildlife habitat as well as provide protection against soil erosion and increase water quality. Maintenance of this conservation program is only allowed if it is to increase the diversity or health of the field. Maintenance against noxious weeds is also required.

The CP-4D practice is the establishment of permanent wildlife habitat. The purpose of this conservation program is to provide shelter and food for wildlife, including upland and grassland birds.

The CP-23 practice is wetland restoration on floodplains. In order to be enrolled in this program the site must have at least 51% hydric soils that are located within the 100-year floodplain. In many cases, succession is allowed to occur in these fields, with vegetation reestablishing naturally. If it is concluded that diverse and native vegetation will not naturally reestablish within 3 years' time, the field will be seeded with beneficial species. The purpose of this conservation program is to provide habitat for waterfowl, grassland birds and other wetland species as well as filter sediment and nutrient runoff and to reduce downstream damages caused by flooding.

The CP-25 practice, restoration of rare and declining habitat, aims to restore the ecological function of endangered habitats, to provide habitat for wildlife and pollinators, and to reduce soil erosion. It requires that noxious weeds be controlled in the site.

The CP-33 practice is buffer habitat for upland birds. The goals of this program are to provide nesting habitat, cover and escape cover for upland birds around field edges. This practice is designed to be adjacent to crop fields and does not need to be a whole field itself. Vegetation can be seeded, or natural succession can be allowed to occur on the field margins. These habitat strips may not be mowed, grazed or hayed for the duration of the contract, nor can a food plot be established within it. It also may not be used for any agricultural purposes, such as turn arounds, lanes or storage.

The CP-37 practice is for duck nesting habitat. To be in this program the land must be a wetland outside of the 100-yr flood plain in one of the limited counties in MN, IA, ND, SD, MT in which it is available. A grassland buffer must be maintained around the wetland and no grazing is allowed.

The CP-38 practice is part of a special program within the CRP program, State Acres for Wildlife Enhancement (SAFE). This program aims to create vital habitat for high-priority wildlife species, such as those that are threatened or endangered as well as those that are considered to create economic benefits or that are highly valued.

The CP-42 practice aims to establish diverse pollinator habitat. The goals of this program are to create wildlife habitat, specifically focused on pollinators, protect soil productivity and to improve water quality. This program requires that there are at least nine pollinator friendly forbs established on the site and that noxious weeds are to be controlled. CP-42 sites must be at least 0.5 acres (0.2 ha).

These conservation practices represent most of the practices implemented through the CRP program. Many of these practices can bring with them other non-targeted benefits, some of which will be explored in this thesis. While millions of acres of agricultural land are enrolled in the CRP program each year, ecosystem services that are not directly targeted by the program are not widely monitored. The objective of this thesis is to determine if the CRP program is providing these valuable ecosystem services to the agricultural landscapes by exploring the benefits of carbon sequestration and increased biodiversity in CRP fields.

CHAPTER TWO

CARBON SEQUESTRATION IN CRP FIELDS

Introduction

Carbon dioxide has come to the center of international efforts to decrease the input and mitigate the effects of greenhouse gases accumulating in Earth's atmosphere as a result of human activity. A large reason for this is that carbon dioxide comprises around 60% of the total greenhouse gas emissions due to anthropogenic activities and it has a long atmospheric residence time, of over 100 years. It is therefore the most important contributor to anthropogenic climate change (Srivastava et al. 2012; Rastogi et al. 2002).

Many different drivers contribute to the increase of carbon dioxide in the atmosphere; the burning of fossil fuels and land use change are two of the main anthropogenic drivers. It is estimated that between 1750 and 2011, land use change has contributed over one-third of the carbon dioxide emissions to the environment (IPPC 2014). Land use change refers to the alteration of natural land cover (vegetation such as prairie or forest), for other uses such as agriculture. One area of the United States where extensive land use change has occurred is the tallgrass prairie region. This region consists of Midwestern states, including regions of ND, SD, NE, MN, IL, OH and IA, that have been extensively converted for agricultural purposes. In Iowa alone land use change has resulted in the loss of over 99.9% of the native tallgrass prairies (Samson and Knopf 1994).

Terrestrial systems hold about 50% more carbon than the carbon dioxide that resides in the atmosphere and over 75% of the carbon in terrestrial systems are held in soils, compared to 25% in the biomass of living organisms (McCarl et al. 2007; Fornara and Tilman 2008; Lal 2004). Some studies suggest various soils in the United States have lost 30-50% of the carbon that they held before large scale land conversion occurred for agricultural use (Baker et al. 2007; Haddaway et al. 2016; Kucharik et al. 2001).

Carbon is stored in soils through the process of carbon sequestration. Carbon sequestration is the process of vegetation taking carbon dioxide from the atmosphere during photosynthesis and fixating it into plant biomass. A portion of the carbon dioxide absorbed by plants during photosynthesis is stored underground in soils in the form of soil organic carbon. Soil organic carbon (SOC) is a component of soil organic matter (SOM), a mixture of organic compounds that are high in carbon content. Much of this SOM comes from dead litter that is deposited as dead plant material. This plant material is rich in organic carbon as it was used by plants to build up the plant's total biomass. Most of this dead litter that is deposited by plants is decomposed in the oxygen rich environment aboveground by microbes before the carbon can be deposited as SOC (Kell 2012).

Root biomass is a large contributor to SOC (Fornara and Tilman 2008; Farrar et al. 2003). This takes place in one of two ways; carbon is used to build root biomass and the carbon that is not used by the plant to build biomass is exuded through the plant's roots as mucilage and sloughed-off cells known as rhizodeposits. The root biomass of plants can be extensive, around 30-50% of carbon fixed during photosynthesis is used to build root biomass (Baker et al. 2007; Buyanovsky and Wagner 1997). While some deposited soil carbon is respired back into the atmosphere through microbial respiration, the rest is stored in the soil in the form of organic matter, such as humus. Soils are good at storing this SOC because they restrict oxygen availability; as a result, carbon that has been deposited by plant roots and dead litter tends to stay in the soil. However, recent evidence shows that a large amount of soil carbon has been lost from agricultural and natural landscapes due to accelerated soil respiration, leaching, and erosion (De Deyn et al. 2010; Lal 2004; Bellamy et al. 2005) all of which are increased in agricultural landscapes as plowing practices aerate the soil, eliminating anoxic conditions that slow down the decomposition of SOC.

While root biomass is an important regulator of carbon sequestration in soils, the properties of the soil itself have important implications for the ability of the soil to sequester carbon. One of these properties is bulk density. Bulk density indicates the level of compaction that a soil has experienced by measuring the mass per unit volume of the soil. Bulk density can influence how well root systems are able to infiltrate and grow in the soil and can influence the availability of resources such as water and nutrients to plants.

Consideration of bulk density is important when measuring or estimating carbon sequestration, as the amount of actual soil vs pore spaces in the soil has implications for how much carbon is being sequestered per unit volume. Kucharik et al. (2003) found that soil bulk density was 13% lower in tested CRP fields than in crop soils adjacent to the CRP fields. This is likely due at least in part to reduced compaction from heavy machinery and increased root penetration in soil.

If the input of carbon into soils exceeds the carbon lost to the atmosphere, SOC will increase over time and soil will act as a carbon “sink”, pulling CO_2 out of the atmosphere and sequestering it in soil, which has the potential to serve as long-term storage. Carbon is mostly sequestered into the soil from the exudates of plant root rhizomes, while soil aeration, such as tillage, leads to lost soil carbon due to resulting aerobic conditions that lead to accelerated SOM decomposition. As discussed earlier, nearly two times the amount of carbon that is stored in the atmosphere is stored in terrestrial soils, so even a small change in the input or output of carbon in soils can show a significant change in atmospheric carbon dioxide. Agricultural systems overall are a source of anthropogenic carbon dioxide, contributing up to 20% of total global emissions of CO_2 (Rahmat et al. 2012; Haile-Mariam et al. 2008).

The soil carbon pool that once existed under the tallgrass prairie ecosystem in the midwestern US is estimated to have decreased by 40% of its original size, due to land conversion for agriculture purposes. (Bernacchi et al. 2005; Donigian et al. 1994; Lal 2004). This is because agricultural practices have been further expanded in the tallgrass prairie region since their first cultivation. Advancements in farming technology such as genetically modified crops have contributed to increased farm sizes and intensified agricultural management (USDA NRCS 2006). Increased agricultural management can be seen in many Midwest states, such as Iowa, whose land cover is eighty-six percent devoted to row crop production. (USDA NRCS 2006).

The Conservation Reserve Program can mitigate many negative consequences of agricultural practices, including loss of soil carbon. Since the enactment of the Farm Bill of 1985, the United States Department of Agriculture (USDA) has overseen the conversion of millions of acres of agricultural land to grassland and wetland under the CRP program (Kucharik et al. 2003; Metting et al. 2001). Grasslands hold more than 10% of the terrestrial biomass carbon and between 10-30% of the total pool of global SOC (Schlesinger 1992; Scurlock and Hall 1998; Derner and Schuman 2007), giving them good potential to serve as a substantial global carbon sink.

While some carbon sequestration does occur on agricultural fields, the process is often slow and can take up to 100 or more years to return to soil carbon levels prior to land use change to agricultural production (Yang et al. 2019). The root biomass of common agricultural crops in the Midwest such as corn and soybeans do not have extensive root systems. Ordonez et al. (2018) found that in Iowa corn maximum root depth ranges from 2.9 to 5.2 ft and soybean ranges from 2.9 to 5.1 ft, although root depth declines with increasing soil fertility (Wilson and Tilman 1993). While soil conservation practices such as no-till agriculture conserve more carbon, Rastogi et al. (2002) found that no-till agricultural practices still lost 2.6 kg of carbon dioxide a day per hectare compared to 11 kg in intensive tilling practices. Thus, while no-till reduces the source strength, no-till fields remain a net carbon source and are not a carbon sink. Perennial vegetation, such as tallgrass prairies that once dominated the Midwest landscape have much more extensive root systems ranging anywhere from 2-15 feet in length. This gives them the potential to sequester more carbon than agricultural fields and potentially act as a carbon sink. The Conservation Reserve Program has many conservation practices that establish tallgrass prairie vegetation on agricultural land and would likely provide increased soil carbon storage as an additional benefit to the practice.

The second chapter of this thesis examined how multiple factors affect carbon sequestration in Iowa CRP Fields. Total soil carbon was analyzed including differences between the following factors: paired CRP-corn fields, time, conservation practice, carbon concentration and bulk density. Local climatic conditions and soil composition can have a significant effect on carbon sequestration (Poirier et al. 2009), therefore it is important to study the effect of CRP fields on carbon sequestration in various regions. This study will increase knowledge of how CRP conservation practices affect carbon sequestration in Iowa. I hypothesized that CRP fields would show an increase in carbon sequestration when compared with baseline carbon levels and when compared with paired agricultural fields.

The specific questions being asked in this research are:

1. Do CRP fields have higher total soil carbon when compared to adjacent corn fields?
2. Does total soil carbon increase in CRP fields over time?
3. Does total soil carbon differ between different CRP practices?

Methods

Study Area

All soil samples were taken from 34 CRP fields and 5 corn fields in Iowa in the summer of 2018 (Table 1). Fields were in 11 different counties including: Carrol, Guthrie, Grundy, Bremer, Butler, Mitchell, Story, Buchanan, Linn, Sac and Benton counties. There was little variation in temperature and precipitation among these sites. Across all 11 counties, average annual high temperature was 28.7 ° C, average annual low temperature was - 10.7° C and annual average precipitation was 90.4 centimeters.

Table 1: All CRP fields sampled; with conservation practice and number of years since CRP field establishment.

Field ID	CP Type	Years in CRP
IA.02.06	02	8 Years
IA.4D.07	4D	8 Years
IA.4D.17	4D	8 Years
IA.10.27	10	9 Years
IA.23.02	23	9 Years
IA.23.19	23	8 Years
IA.25.19	25	7 Years
IA.25.28	25	7 Years
IA.38.05	38	8 Years

(table continues)

Field ID	CP Type	Years in CRP
IA.38.16	38	6 Years
IA.38.17	38	9 Years
IA.38.23	38	8 Years
IA.38.29	38	9 Years
IA.42.03	42	6 Years
IA.42.05	42	6 Years
Moseley CRP	42	3 Years
Miller CRP	42	3 Years
Brindle CRP 1	42	3 Years
Brindle CRP 2	42	3 Years
Stone CRP	42	15 Years

Soil Sampling

Soil samples were taken from fourteen different CRP fields enrolled in various conservation practices, including CP-2, CP-10, CP-4D, CP-23, CP-25, CP-38 and CP-42. The CRP fields used in this analysis are a subset of 180 Iowa fields surveyed for a larger USGS project. Requirements for selecting these fields were that they were visible from a public road and therefore, could be visually assessed without entering private property. The collected soil samples were taken from CRP and crop field locations where landowners gave permission to collect soil samples.

Along with soil samples from CRP fields, soil samples were taken from five corn fields directly adjacent to five CRP fields in order to ensure as little difference in climate and soil type as possible; these fields are considered paired fields.

In each of the eighteen fields, four transects were laid down, in order to get a representative sample across the range of soil conditions with in the field 100 m or 25 m transects were laid down in a pattern (Figures:1,2,3) that best covered the entirety of the field. The patterns were chosen based on the size and shape of each field, 25 m transects (Figure 3) were used for smaller fields, mainly CP-42 fields as they are generally smaller than others due to program requirements. Sampling started 15 meters from the roadside to minimize edge effects. In each transect, soil samples were taken from two 1 m^2 quadrats that were 25 m apart from each other for a total of eight soil samples per field.

Soil samples were taken using a 4.8 cm diameter bulk density corer and the soil sample was taken from the top 15 cm of the soil. Before soil samples were taken, any litter present in the soil sample area was removed to ensure only mineral soil was included in the sample.

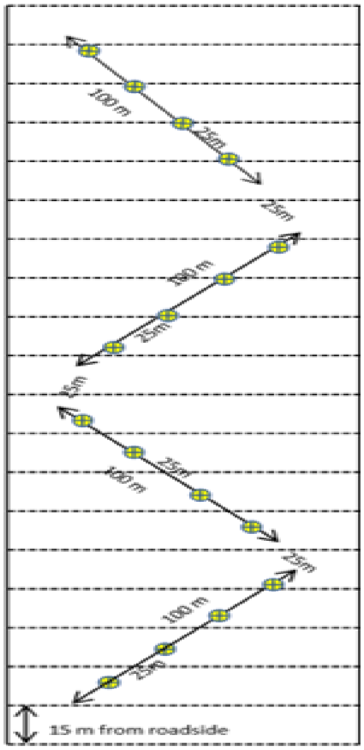


Figure 1: 100 m sampling design for field with unequal field margins (Yellow circles indicate quadrat positions).

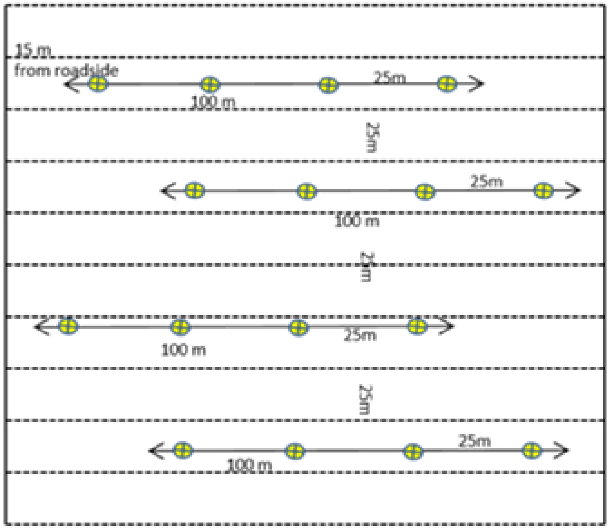


Figure 2: 100 m sampling design for uniform field with equal margins (Yellow circles indicate quadrat positions).

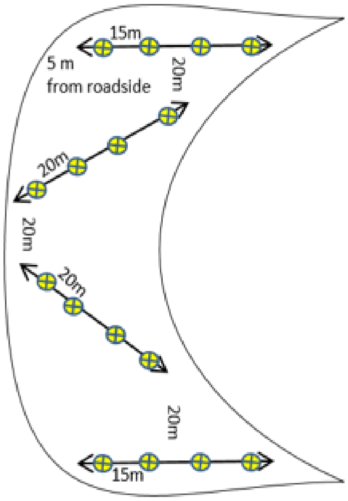


Figure 3: 60 m sampling design for smaller, irregular fields (Yellow circles indicate quadrat positions).

Soil Analysis

Soil samples were frozen immediately upon return from the field, always less than 12 hours from the time soil samples were taken. Samples were stored in the freezer to avoid loss of labile organic carbon, the portion of SOC that is readily broken down by microorganisms.

When soil samples were ready to be analyzed, they were thawed and weighed wet. A subsample was collected and weighed, then oven-dried at 105 ° C to constant mass to determine soil moisture of the bulk density core. This soil moisture was used to determine the dry soil mass of the bulk density core, allowing bulk density to be determined by dividing dry weight of each sample by the volume of the bulk density core. Each bulk density measurement from individual cores were averaged together with other samples from the same field, in order to reach an average bulk density for the entire field.

Organic carbon content was determined using the loss on ignition method (Ball 1964; Konen et al. 2002), a technique that measures organic carbon by combusting it out of the sample. A subset of between 5-10 g (dry soil mass) of the soil sample was weighed in a ceramic crucible prior to being put into an oven at 500 ° C for 200 min. The soil sample was then allowed to cool overnight before being weighed again. The soil weight obtained after organic carbon was burned off was subtracted from the dry soil weight in order to determine the total labile organic carbon in the soil sample.

Due to the difficulty of obtaining bulk density measurements, bulk density cores were not collected for 29 samples, so for these samples bulk density was instead estimated using the strong relationship observed in the data between Loss on Ignition measurements (LOI) and bulk density (BD) ($r = -0.827$):

$$BD = 1.832 - 0.0704 * LOI$$

Similar approaches have previously been used to infer missing bulk density measurements (Adams 1973; Luo et al. 2010).

Data Analysis

Because soils vary tremendously in inherent composition and carbon concentration, I compared the soil carbon concentration (%C) measured in each sample to the mean soil %C for each soil type. This was done by subtracting from the measured %C the mean soil %C for a given soil type, as reported through the USDA NRCS National Cooperative Soil Survey (USDA NRCS 2019).

This difference in soil carbon is expected to be equal to zero when no net gain or loss of soil carbon has occurred, while positive values indicate increased soil %C in the sample. To determine if the differences were significantly different from zero, they were analyzed in R version 3.5.2. (R Core Team 2019) using a linear mixed effects model, with “site” included as a random term in the model to account for non-independence of replicate cores from each farm. In addition, the effect of the grassland’s age was tested to determine if soil %C increased or decreased linearly over time.

In the subset of CRP with adjacent corn fields (paired fields), soil bulk density, carbon concentration, and total carbon content per unit area were compared between the paired fields. For these data, analysis was conducted as a split-plot design using a mixed effects model with the farm as a random term and habitat type as a fixed effect.

LOI % for CRP prairie fields that did not have a paired corn field were determined by obtaining the expected LOI baseline percent of that soil type using USDA web soil survey data. One LOI soil analysis from a CP-10 field was excluded from analysis, because the LOI percent was very high (more than 6 standard deviations from the mean, and nearly double the value of the next-highest data point); it was suspected that there was soil lost during analysis.

Results

Soil Organic Carbon Difference in CRP Fields

Baseline carbon, included to ensure that soil type was not driving any significant results, showed that CRP field soil content was higher than expected for each soil type (Figure 4), excluding the soil types found in CP-23 fields (1.8% lower than expected) and CP-25 fields (soil data unavailable from USDA soil survey).

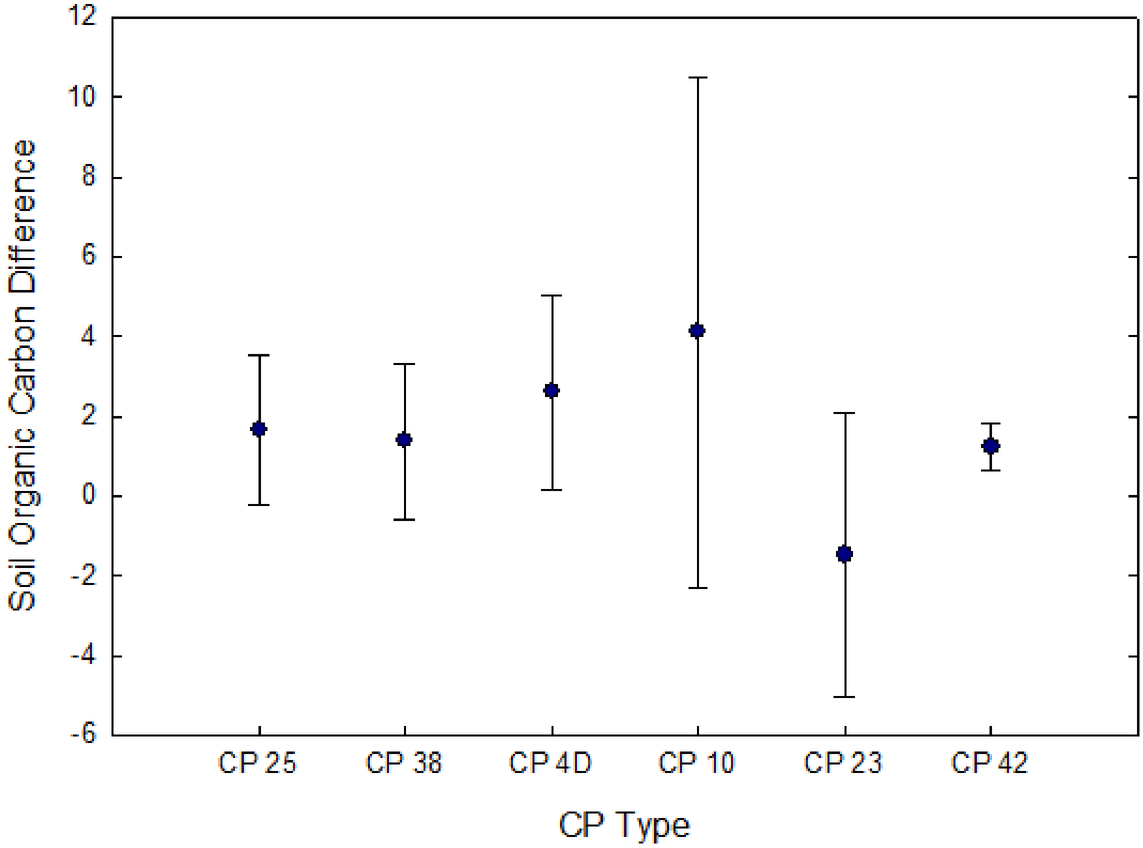


Figure 4: Average soil organic carbon (SOC) difference in LOI% between baseline SOC expected for soil type and SOC in CRP soil samples for each CRP program assessed.

Soil Carbon by CRP Age

CRP fields analyzed by age included fields aged 6,7,8,9 and 15 years. Carbon % averages (Figure 5) steadily increased with increasing CRP age. Across the timeline carbon % increased from 3.39% to 10.18%, an increase of 6.79% over the 9-year timeline. Bulk density averages (Figure 5) steadily declined with increasing CRP age, decreasing from 1.61 g dry soil/cm^3 to 1.10 g dry soil/cm^3, a decrease of0.51 g dry soil/cm^3. Total carbon content averages (Figure 5) showed a steady increase as the CRP field increased in age. CRP total soil carbon increased from 0.054 g C /cm^3 to 0.112 g C /cm^3 across the nine-year timeline. This implies a total carbon sequestration rate of 6.4 mg C/cm^3, or 0.96 kg C/m^2 in the top 15 cm of soil.

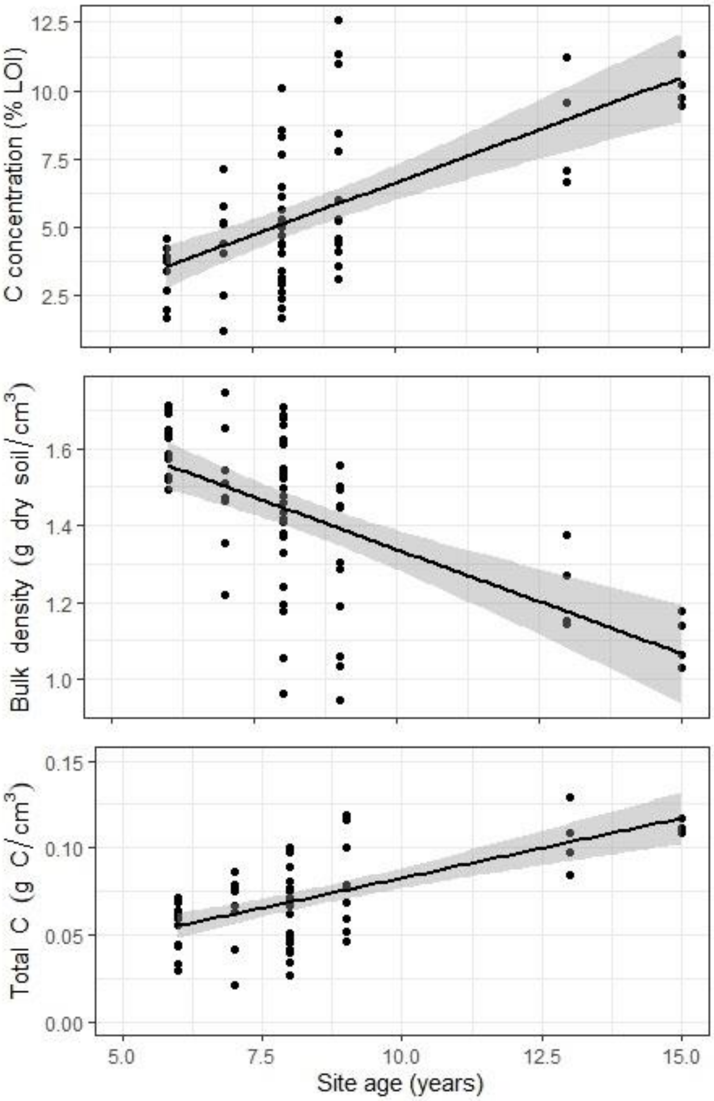


Figure 5: Difference in LOI% (top), bulk density (middle) and total carbon sequestered (bottom) by CRP age. Each point represents a single soil sample.

Carbon in Paired CRP and Corn Fields

CRP fields had significantly higher carbon concentration (%C) than adjacent corn fields (Table 2: $p<0.0001$), averaging 7.6% Loss On Ignition compared to 5.5% in corn fields (Figure 6). However, bulk density differences between the four paired corn and CRP fields showed a significant difference (Table 2: $p < 0.0001$) between the two different habitats as well (Figure 7).

Table 2: LOI%, bulk density and total carbon content significance between paired corn and CRP fields.

	LOI%		**Bulk Density**		**Total carbon content**	
	$F_{1,40}$	P	$F_{1,40}$	P	$F_{1,40}$	P
Site	57.12555	<.0001	372.5618	<.0001	96.62614	<.0001
Field Type	18.84875	0.0001	17.8052	0.0001	13.91953	0.0006

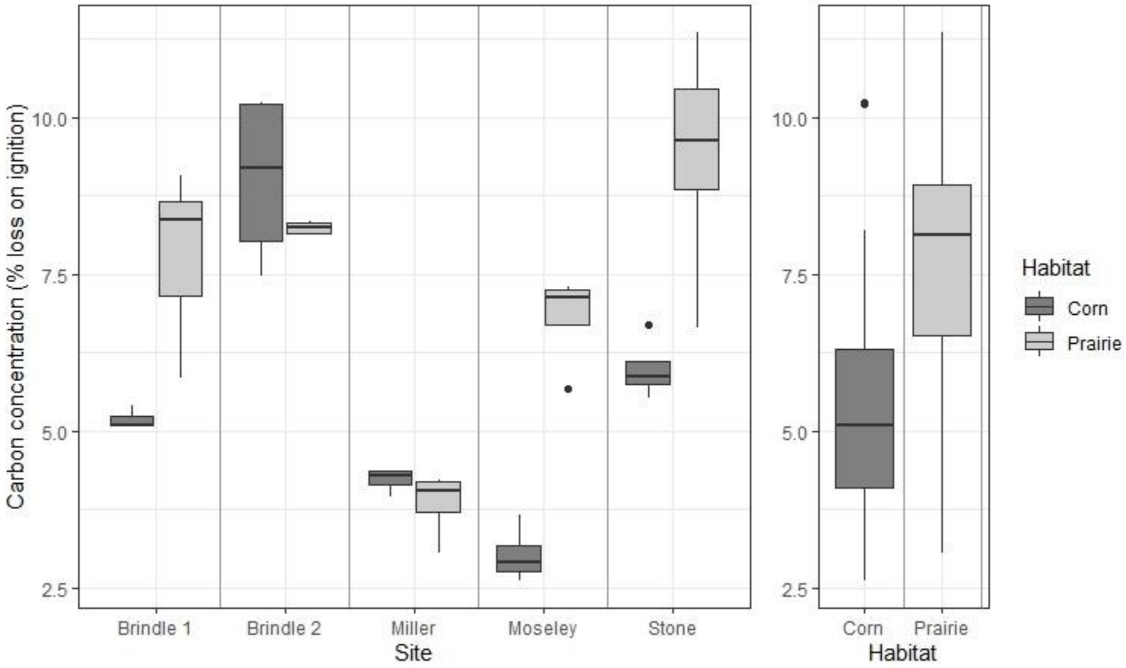


Figure 6: Carbon concentration between specific paired corn and CRP fields measured in LOI percent (left). Carbon concentration averages of all paired corn and CRP fields (habitat: prairie) measured in LOI percent (right).

The corn fields had a higher bulk density with an average of 1.51 g/cm^3 compared to 1.33 g/cm^3 for CRP fields (Figure 7). While corn fields had lower carbon concentration (%LOI), their higher bulk densities means that the total carbon content (g C/cm^3) was only slightly but still significantly ($p = 0.0006$) lower in corn (0.081 g C / cm^3) compared to CRP (0.098 g C / cm^3) fields (Figure 8).

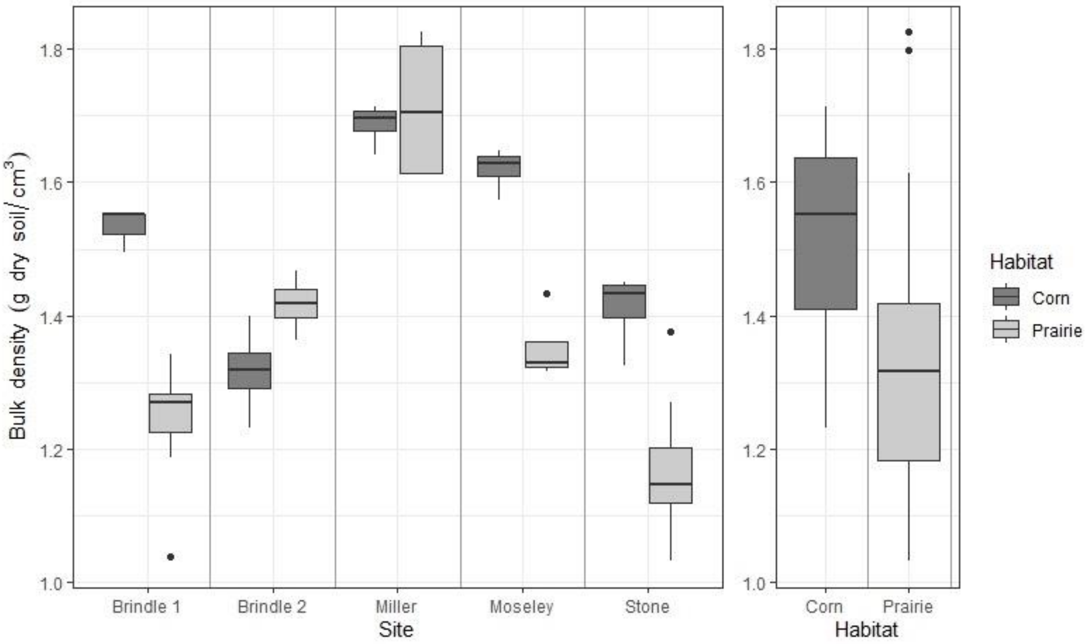


Figure 7: Bulk density between specific paired corn and CRP fields (left). Average bulk density between all paired corn and CRP (habitat: prairie) fields (right).

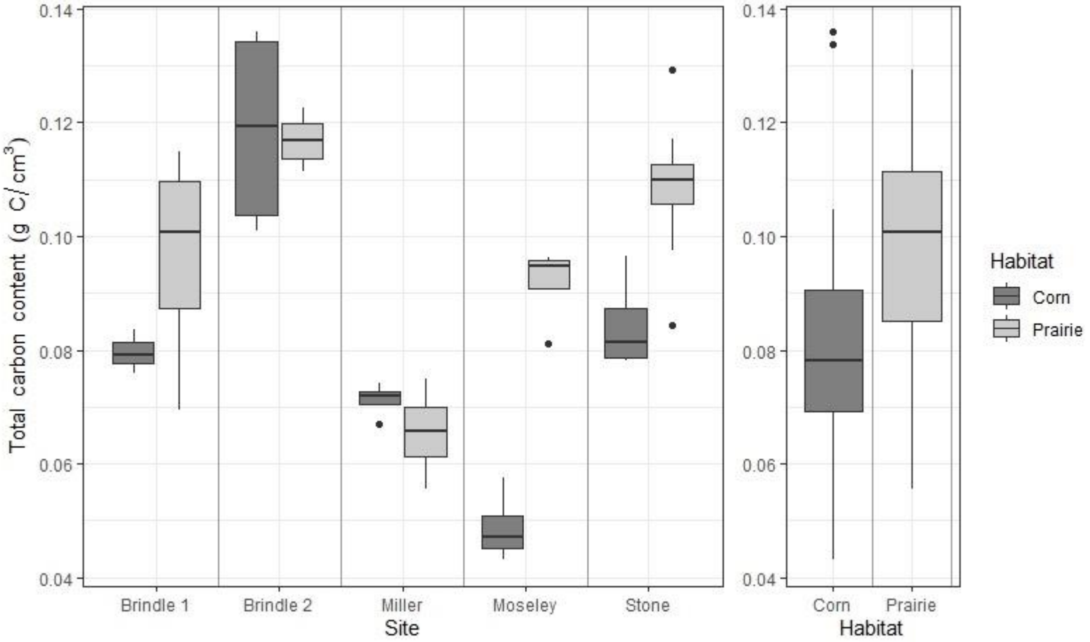


Figure 8: Total soil carbon between specific paired corn and CRP fields (left). Total soil carbon between all paired corn and CRP (habitat: prairie) fields (right).

Three of the five paired CRP fields had significantly higher total carbon content than their adjacent paired corn fields (paired t-tests), while the other two CRP fields had a statistically insignificant lower total carbon content than adjacent corn fields (Figure 6).

Total carbon content was significantly ($p = 0.032$) higher at Brindle 1 prairie (0.097 g C / cm^3) compared to Brindle 1 corn (0.080 g C / cm^3) , Stone prairie (0.108 g C / cm^3) and Stone corn (0.084 g C / cm^3) were also significantly different ($p = 0.004$) and Moseley prairie (0.092 g C / cm^3) was significantly different ($p=0.0001$) from Moseley corn (0.049 g C / cm^3). The two paired corn-CRP fields where corn showed a higher LOI percent were not significantly different (Brindle 2: $p=0.848$, Miller: $p=0.268$).

Soil Carbon by CRP Type

When comparing LOI percent by CRP type, CP-23 had the highest LOI percent (8.97%) and CP-25 had the lowest LOI percent (3.25%). Bulk density by CRP type was significantly different ($p=0.0061$); bulk density was highest in CP-42 fields (1.59 g C /cm^3) and the lowest in CP-23 (1.20 g C /cm^3) with a difference of up to 0.39 g/cm^3 between CRP types that were analyzed (Table 3). Taking bulk density into consideration the lowest total soil carbon content was found in CP-42, the youngest program (0.063 g C /cm^3). The highest total carbon content was found in CP-10 (0.095 g C/cm^3); these fields are likely older than the CRP contract, as this program is used for enrolling existing grasslands. While age was not a variable included in this analysis, the site age is similar and not significantly different among specific conservation practice groups ($F_{6,53}$= 1.3101; $p=0.2689$).

Table 3: Soil carbon concentration, bulk density and total soil carbon found by CP practice type.

CP Practice	**Carbon concentration (LOI%)**	**Bulk Density (g dry soil/cm^3**	**Total carbon (g C/cm^3)**
10	5.533481	1.356977	0.6883433
2	5.076899	1.419403	0.06941173
23	7.148702	1.224452	0.07926510
25	4.831820	1.468102	0.06298927
38	4.276120	1.534753	0.06298927
42	4.601478	1.519883	0.06748453
4D	4.886987	1.449668	0.06560547

Discussion

While carbon sequestration is not an ecosystem service directly targeted by CRP fields, they have the potential to sequester large amounts of carbon as the land is converted from agricultural production to management for perennial vegetation cover (Gebhart et al. 1994; Li et al. 2017). The purpose of this study was to determine if CRP fields provide increased carbon sequestration in agricultural landscapes. Total soil carbon was significantly higher in CRP fields when compared to adjacent corn fields, indicating that CRP has an increased carbon sequestration rate over agricultural fields. CRP fields also showed increased total soil carbon when compared to expected soil carbon content of individual soil types. This indicates that, not only do CRP fields provide more carbon sequestration than agricultural fields, but they also provide increased carbon sequestration than is expected for a given soil type.

The difference in soil carbon concentration in CRP fields compared to baseline soil carbon concentration, based on the USDA web soil survey showed a trend of increased soil carbon concentration in CRP fields, confirming that soil type was not a driver of soil carbon results. The only CRP type where soil carbon was below what was expected for the given soil type was CP-23. CP-23 is a wetland program and as such, fields enrolled in this program are required to be in a 100-yr floodplain with at least 50% of the field containing hydric soils. Wet, hydric soils are anoxic and therefore are capable of storing soil carbon for longer periods of time; it is likely that these soils would lose a significant amount of carbon after mechanical disturbances that aerated the soil, and would take longer to return to expected soil carbon levels for that soil type. Since these were formerly agricultural fields, this is likely showing that there are still effects from aeration of the soil through plowing practices releasing trapped soil carbon.

Total soil carbon, which is the important response variable when it comes to climate change was found by determining the carbon concentration (LOI%) within a given density, or mass per volume of soil. When total soil carbon was compared between paired CRP and corn fields, there were mixed results and high variability between carbon concentration and bulk density results. Averaged across all fields, there was significantly more total carbon in CRP fields than in adjacent corn fields, providing strong evidence for increased C sequestration in CRP fields. Still, while three of the five CRP fields analyzed had significantly higher total soil carbon, the two CRP fields that had lower total soil carbon were not statistically different from their paired corn fields.

Soils have long been known for being inherently variable (Mader 1963) and our results confirm that better estimates would be obtained by sampling more sites. Soil bulk density and soil carbon content is highly correlated with soil texture and aeration as well as cultivation and history (Murphy et al. 2006; Shah et al. 2017). Therefore, these results likely depend on time since prairie was established, soil type, and history of management on both fields, explaining how we could see a variance in comparisons between paired corn-prairie fields. Furthermore, evidence of past agricultural management, including higher bulk density and lower percent soil carbon, can be seen in restored prairie soils more than 50 years after the disturbance (Kucharik et al. 2006). Because past management histories for our sites were unknown, this may have contributed to the observed variability.

When the paired fields were averaged for each variable (LOI%, bulk density and total soil carbon) the results were significant between the two habitat types. Carbon concentration was 40% higher in CRP fields than corn fields; even when taking the significantly lower bulk density of CRP fields into account, total soil carbon (g C/cm^3) was slightly, but still significantly higher in CRP fields. This result shows that CRP fields can give increased carbon sequestration over crop fields in comparable soil types in Iowa. Total soil carbon in Iowa CRP fields also showed a steady increase over time, consistent with findings that after the end of agricultural disturbance, total soil carbon steadily increases (Murphy et al. 2006). However, since these CRP contracts are typically between 10-15 years, the question remains if these benefits will last once the contracts expire. Between 2013 and 2018, approximately 4.4 million hectares of land in CRP expired (FSA 2012) and higher corn and soybean prices could motivate farmers to return CRP land to crop production (Du et al. 2008; Secchi et al. 2009). Ruan and Robertson (2013) found indications that the conversion of CRP land can not only increase emissions of CO_2 but can cause the loss of the CRP land's total greenhouse mitigation ability. This can be further exacerbated by the type of tillage that is used to convert a field. Using conventional tillage practices can cost around 8 years' worth of lost carbon sequestration after a single time (Ruan and Robertson 2013).

CRP fields enrolled under different conservation practices showed some small, nonsignificant differences in total soil carbon between practices. While it is difficult to draw definitive conclusions from this result because of limited data, it is possible that different CRP practices which have different vegetation types could lead to differences in carbon sequestration. It has been observed that differences in functional diversity may affect carbon sequestration rates over a time scale of decades (Ampleman et al. 2013).

While total soil carbon was the focus of this research as it is the most important factor in soil carbon sequestration, bulk density results in themselves are important as they have indications for soil health. When bulk density is increased, the porosity of the soil decreases, less pores in the soil hinders the movement of water, nutrients and microorganism activity, all of which have impacts on soil productivity and processes (Duiker 2005).

Bulk density results from the paired CRP and corn fields, showed that CRP fields significantly decrease soil bulk density when compared to corn fields in similar soil types and climate in Iowa. Bulk density also decreases in CRP fields with increasing age, indicating that as CRP fields age the soil becomes less compact with more air pores in the soil, likely due to less compaction from agricultural equipment and increased root penetration of the soil as seen in other studies (Murphy et al. 2006).

Bulk density was also slightly different between CRP types; the highest bulk density was found in CP-42 (pollinator habitat) fields and the lowest was found in CP-23 (wetland habitat). The highest bulk density was likely found in the CP-42 practice as it was the youngest average field age between practices analyzed and likely had the most recent mechanical disturbance due to agricultural practices, such as plowing. Bulk density has been shown to increase under continual agricultural management which causes soil compaction by farm equipment and depletion of soil organic matter decreasing the amount of soil porosity, therefore increasing bulk density (Murphy et al. 2004). Conclusions about whether the specific CRP enrollment program a field is enrolled in makes a significant difference in the bulk density of the soil cannot be made due to the large number of different CRP practices and the limited amount of data for each practice. However, Murphy et al. (2006) found a difference in bulk density between warm-season, cool-season and warm-season native fields as well as the previous agricultural management at the site, suggesting that true differences may exist among CRP practices.

Overall, the results show that CRP fields in Iowa create additional ecosystem services outside of their targeted conservation goals. CRP fields not only increase total soil carbon, an important sink for atmospheric carbon dioxide that is contributing to global climate change, they also decrease soil bulk density and thereby improve soil health.

CHAPTER THREE

WEEDY INVASION IN CRP FIELDS

Invasive Species

Flowering plants have been successful colonizers throughout history as they have the potential to travel long distances via wind, ocean currents and animal vectors and have evolved seeds that are well-adapted to surviving hardships involved in long distance traveling (Ridley 1990; Novak and Mack 2001). However, humans have become the most important mode of long-distance transportation for plants (Ridley 1990; Novak and Mack 2001). Due to anthropogenic activities, globalization has facilitated the movement of countless plant species across natural geographic barriers and many plants have established successful populations in new geographic areas (Theoharides and Dukes 2007; Milton 2004; van Kleunen et al. 2015).

Many species transported to novel environments never establish in the new range; however, a small number become established and naturalized near the site of release but do not colonize new areas (Mack 1996). A subset of these established species create viable permanent populations that spread over large new areas (Mack and Lonsdale 2001). When these species cause serious human, economic, or ecological harm they are considered invasive species (Tang et al. 2019; Mack and Lonsdale 2001).

The spread of these invasive species cause severe economic and ecological consequences. In the United States alone, invasive species cause approximately $120 billion in damages and control costs a year (Pimentel et al. 2005; Lodge et al. 2006), around $26 billion in control and damage are incurred annually by the agricultural sector from weedy plant species (Pimentel et al. 2005). Invasive species are also the second leading cause of biodiversity loss behind habitat loss (Germain et al. 2019). In fact, nearly one-half of species listed as threatened or endangered under the United States Endangered Species Act are considered to be threatened or endangered primarily because of an invasive species (Wilcove et al. 1998).

Invasive plant species can create debilitating consequences as they have the potential to alter ecosystems. (DAntonio and Vitousek 1992; Vitousek 1990; Hobbs and Mooney 1986; Braithwaite and Lonsdale 1987). There are several ways that invasive plant species can have negative effects on an ecosystem; they can alter the hydrology, nutrient cycling, soil formation and change the natural disturbance regime (Vitousek 1990). They commonly outcompete native species for resources, including reducing the amount of water, sunlight, space and nutrients or food that are available to the native species (Vitousek 1990) and by causing diseases that native species have not been exposed to. Native species and invasive species can also hybridize (Vila et al. 2000), altering the genetic makeup and reducing the gene pool of the native species. This problem is likely to get worse as global temperatures rise and precipitation patterns change as a result of climate change (Ricciardi 2000; Dukes and Mooney 1999), invasive species will likely be enabled to establish and invade new areas.

Invasive plants commonly share several key life history characteristics. Many plant invaders are asexually reproducing species (Amsellem et al. 2001; Price and Jain 1981), while sexually reproducing species tend to have high reproduction rates with smaller seed size, large seed yields, and higher rates of seed germination and growth

(Lavergne and Molofsky 2007). They often have short generation times, allowing them to outnumber native species quickly after a disturbance. They also tend to have few or no natural predators, parasites, or diseases in the environment they are introduced to (Wolfe 2002; Torchin et al. 2003). These species are commonly generalist and tolerant of many different environmental conditions, showing ecological plasticity (Maron et al. 2004). , Many invasive plants are pioneer or colonizer species, the first species to recolonize disturbed or degraded areas. While many invasive species share one or more of these characteristics, it is not necessary for a species to possess all these qualities to be an invasive species.

Many species with these characteristics, such as those with small seeds and a high potential for rapid growth, depend on the exposure of resource rich areas with little competition to successfully establish a population (Burke and Grime 1996). These conditions along with the characteristics of a “good colonizer” (Bazzaz 1986) allows weeds to take advantage when these conditions suddenly appear in the landscape (Burke and Grime 1996). Seeds of these invasive species are commonly transported via vehicles, livestock, and clothing as well as within commercial seed lots (Mack and Lonsdale 2001; Muenscher 1987) such as those used to plant CRP fields. Invasive plants may also be transported intentionally for uses such as crops, ornamentals and for erosion control activities (Mack and Lonsdale 2001).

In the United States nearly 500 species of native and non-native plants have become weedy pests in agricultural landscapes (Pimentel et al. 1989) and approximately 73% of these are non-native species (Pimentel et al. 2000). Disturbance regimes create open spaces for these invaders to colonize, especially in agricultural ecosystems. The amount of disturbance in an area directly affects the potential success of invaders because it decreases the ability of native plants to compete (Burke and Grime 1996; Lonsdale 1999).

Intensive agricultural systems, such as those in the Midwestern U.S., can increase weed propagation. In fact, as the management of a field is intensified, the greater the potential is for habitats adjacent to those fields to become a source of weeds (Boatman et al. 1994). One of the reasons for this is that frequent disturbance and fragmentation promote plant invasions (Robertson et al. 1994; Clark and Ji 1995; Cavers and Harper 1967; Crawley et al. 1986). Another reason for this is that agricultural disturbance regimes are often associated with an increase in nutrient availability in the surrounding landscape. In habitats with these characteristics of disturbance and increased resources, successful plant invasions are more likely to occur (Rejmanek 1989; Burke and Grime 1996).

CRP fields bring many ecosystem services, including biodiversity to agricultural landscapes that otherwise consist mostly of large crop monocultures. CRP sites are seeded and planted with native and beneficial grasses, forbs and legumes creating valuable native habitat. While these fields can be beneficial to native species, they can also create a lot of potential habitat for weedy species since the intensely managed agricultural landscape has many factors that benefit their establishment. Therefore, CRP fields should be assessed not only for the environmental benefits they bring, but also for any potential negative environmental effects, such as promoting the invasion and spread of weeds. Two weeds of particular concern in Midwestern CRP fields are *Phalaris arundincea* and *Pastinaca sativa* .

Reed Canary Grass

Reed canary grass (*Phalaris arundincea*, hereafter "*Phalaris*") is a grass that has hairless stems with bluish green tapering leaves. The panicles flower from May to mid-June and are green to purple in color, fading to beige as the seeds ripen.

Phalaris is believed to be native to all temperate regions in the northern hemisphere. However, in North America a genotype of *Phalaris* has emerged that is considered a noxious weed species. Unlike its native counterpart, this noxious genotype creates single species monocultures. The origin of this invasive genotype has not been positively identified; it is a cryptogenic species (Carlton 1996; Galatowitsch et al. 1999). There are a few different theories as to how it became a noxious weed. One hypothesis with some support is that it was cultivated in Europe as a forage crop for cattle, brought to the United States where it invaded native ecosystems (Merigliano and Lesica 1998; Paveglio and Kilbride 2000; Lavergne and Molofsky 2004, 2007; Kercher et al. 2006). Another theory is that this cultivated version was brought from Europe and then hybridized with our native *Phalaris* in the United States.

Phalaris Biology. The invasive genotype of *Phalaris* is a cool-season, C3 perennial grass (Kephart and Buxton 1993; Carlton 1996). *Phalaris* plants are self-sterile and therefore sexually reproduce largely through cross-pollination (Ostrem 1987, 1988a), but it also grows asexually along dense rhizomes causing it to form dense sod-forming stands or clumps, rather than individual stems (Galatowitsch et al. 1999). These rhizomes store carbohydrates and allow for early, rapid growth of *Phalaris* in the early spring, with tillers sprouting from the rhizomes (Carlton 1996; Tamura and Moriyama 2001). *Phalaris* rapidly develops aboveground biomass to photosynthesize, therefore shading out many competitors (He et al. 2010).

Phalaris has strong reproduction capability through sexual and asexual means, and combined with its ability to grow rapidly, it frequently forms dense stands in wetlands (He et al. 2010). While it is most productive in wet environments, it is adapted to a large range in soil moisture (Galatowitsch et al. 1999; Zeiders and Sherwood 1985) and can even tolerate drought conditions (Lavergne and Molofsky 2004). These adaptations allow *Phalaris* to survive and compete in a gradient of soil moisture conditions including wet meadows and mesic grasslands.

Phalaris has a very high annual seed yield (Baltensperger and Kalton 1958; Ostrem 1988b). These seeds express dormancy and are therefore important components in seed banks (Odland 1997; Odland and del Moral 2002), allowing them to take advantage of newly disturbed environments. Newly disturbed areas allow seed germination of *Phalaris* as it requires light and grows best in moist soils (Vose 1962; Landgraff and Junttila 1979; Lindig-Cisneros and Zedler 2001, 2002) with the highest germination success in soils that are water saturated (Coops and Van der Velde 1995; Kellogg et al. 2003). Many of these characteristics have allowed *Phalaris* to displace native wetland species and form large monocultures. It is considered one of the top noxious weeds invading wetlands in the Midwestern U.S. (He et al. 2010; Jakubowski et al. 2011). Kercher and Zedler (2004) found that in a wet-prairie grassland, fertilization and disturbance, both common in an agricultural landscape, increased both the biomass and frequency of *Phalaris*. Disturbance was found to decrease the total biomass of resident wet-prairie species, thereby increasing the amount of available light and space allowing *Phalaris* to expand within wet-prairie (Kercher and Zedler 2004).

Wild Parsnip

Wild parsnip (*Pastinaca sativa*, hereafter referred to as *Pastinaca*) is an herbaceous biennial plant that sometimes behaves as a monocarpic perennial; it dies once it has flowered and set seed (Baskin and Baskin 1979; Gleason and Cronquist 1991). Its native range is throughout Europe and temperate Asia (Averill and Ditommaso 2007) and it is the only species of *Pastinaca* in the United States, where it can be found in 45 of the 50 U.S. states (excluding Alabama, Florida, Georgia, Mississippi, and Hawaii) (Averill and Ditommaso 2007). *Pastinaca* is believed to have been cultivated in early Europe and widely grown there as a crop by the 16thth century (Averill and Ditommaso 2007). *Pastinaca* was introduced to North America by European settlers and was considered common by the early 17th century. It is believed that this cultivated *Pastinaca* escaped and reverted to its wild form (Averill and Ditommaso 2007).

Pastinaca Biology.

Pastinaca has a deep, thick taproot that can reach depths of up to 1.5 m (Gleason and Cronquist 1991). *Pastinaca* growth starts as short rosettes that have pinnately compound alternately arranged leaves, that are around 15 cm in length (Lorenzi and Jeffery 1987). It remains in this stage for approximately two years, at which time the plants grow tall, stout stems (Baskin and Baskin 1979). Inflorescences consist of large, compound umbels that are between 10-20 cm in width. Flowers have yellow petals and umbels on each individual plant consist of both male and hermaphroditic flowers (Nitao and Zangerl 1987) with mechanisms that prevent cross-pollination of flowers on the same plant (Cruden and Hermann-Parker 1977). Each flower produces two flattened seeds (Gleason and Cronquist 1991); *Pastinaca* reproduces only by seed; it does not reproduce vegetatively (Hendrix and Trapp 1992) However, *Pastinaca* plants create a large amount of seed that can stay viable in seed banks for years, contributing to its ability to persist and reproduce (Schaefer 2015).

Pastinaca has been declared a noxious weed in Ohio and Minnesota (USDA NRCS 2006; MN DNR 2020) and invasive in Kentucky, Nebraska, Tennessee and Wisconsin (Averill and Ditommaso 2007; Haragan 2015). It can cause phytophotodermatitis in humans if bare skin is exposed to the sap full of furanocoumarins, which cause a photoreaction when exposed to sunlight (Averill and Ditommaso 2007). Other concerns are that *Pastinaca* invasion reduces the quality of agricultural forage crops (MDA 2020), as they can harm cattle when ingested and they may also decrease habitat quality for honeybees (*Apis mellifera*). Honeybees do not use or pollinate *Pastinaca* and it is outcompeting and displacing other important plants that are used by honeybees (Averill and Ditommaso 2007).

While *Pastinaca* grows best in moist, nutrient rich soils, it can survive well under poor soil conditions (Gleason and Cronquist 1991; Averill and Ditommaso 2007). In fact, it has been observed that under drought conditions, *Pastinaca* growth increased, while perennial grass growth decreased (Sternberg et al. 1999). It is commonly found in road ditches as well as other disturbed habitats and along patch edges. While it is slow to establish, once it does it quickly spreads and can modify habitats from dry to wet-moist soil conditions. It has also been found to invade native prairies (MN DNR 2020). With its ability to invade a range of habitat conditions and its prevalence along roadsides, *Pastinaca* has the ability and opportunity to invade CRP fields.

This chapter will further explore the weedy invasion of *Phalaris* and *Pastinaca* within CRP sites in the Midwestern agricultural landscape. While *Phalaris* invasion in wetlands has been well explored in scientific literature, exploring its potential to invade and degrade CRP fields is lacking in the scientific literature. The invasive potential of *Pastinaca* on the other hand, is widely lacking in the literature, let alone its potential to invade CRP fields. Agricultural practices commonly create prime conditions and opportunities within the surrounding landscape for invasives such as *Phalaris* and *Pastinaca* to establish and thrive. Therefore, they have potential to invade and degrade CRP fields.

The specific questions that are being asked in this research are:

1. Do *Phalaris* and *Pastinaca* invade specific CRP practices more frequently?
2. Does the invasion success of *Phalaris* and *Pastinaca* change with the age of the CRP establishment?

3. Do disturbance or vegetation characteristics (grass & forb richness and % cover, % bare ground) predict invasion of *Phalaris* and *Pastinaca* in CRP fields?

Methods

Study Area

Edge-of-field surveys were used to visually assess 1,793 fields from 2016-2017 in 14 states(IA, MO, MN, CO, ID, KS, MT, ND, NE, OK, OR, SD, TX and WA) as part of a larger USGS project, I personally surveyed fields in IA, MN and MO but all surveyors used identical protocols and data collection forms. The fields were enrolled in the following CRP programs: CP-1, CP-2, CP-10, CP-4D, CP-23, CP-25, CP-33, CP-37 CP-38 and CP-42. Fields were randomly chosen from a subset of fields enrolled in the CRP program. Requirements for the subset of CRP fields included were that they were within at least 25 m of an existing road centerline (in order to account for the width of the road and adjacent right of ways) and that the fields be at least 2.0 hectares in size (an exception was made for CP-42, which required a minimum of 0.8 hectare, in order to reflect the smaller acreage typical of these fields.

At each edge-of-field survey, each forb and grass species visible from the edge of the field was recorded as well as each species' percent cover; finally, overall percent cover of forb, grass, shrub, trees and bare ground was determined for the entire field. For overall cover estimates and each individual species estimate, percentages were rounded to the nearest 5%. The presence of shrub and trees was very rare in areas where the CRP fields were selected to carry out further analysis (see “Delineation of Distribution Range” below). Therefore, the percent cover of shrub and trees were not included in the following statistical analyses.

Validation

In order to validate data collected from roadside surveys, in-field surveys were done in a subset of 68 fields (28 in Iowa, 19 in Colorado and 21 in Idaho), including fields enrolled in the following programs: CP-2, CP-10, CP-4D, CP-23, CP-25, CP-38 and CP-42. At each location, a regular roadside survey was first conducted, followed by an in-field survey.

In-field surveys consisted of detailed sampling of vegetation in the field from four 1 m^2 quadrats spaced 25 m (or 6 m) apart along a 100 m (or 25 m) transect. A total of four transects were sampled from each field, with 25 m (or 6 m) between each transect. Transects were organized in a pattern that best represented the vegetation in the entire field (Figures:9,10,11) .

Transects started 15 m (or 5 m) from the edge of the field, in order to minimize any edge effect. In each quadrat the percentage of each species was estimated to the nearest 1%, as was the overall percentage of forbs, grasses, shrubs, trees and bare ground. In total there were 16 vegetation samples per field.

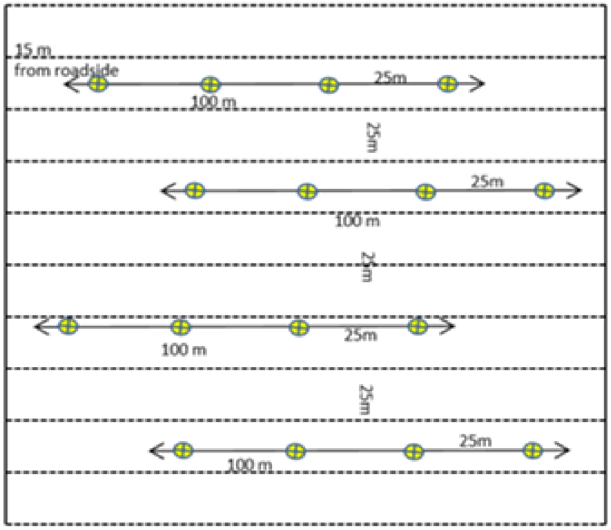


Figure 9: 100 m sampling design for uniform field with equal margins.

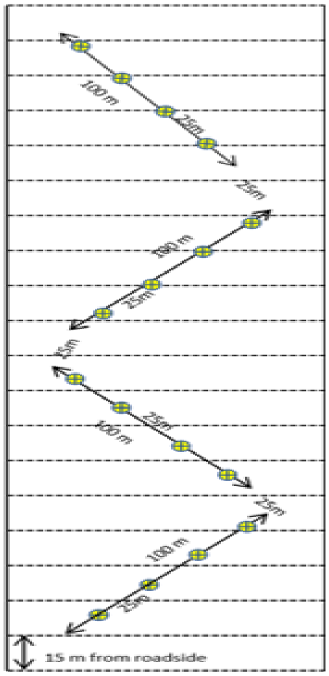


Figure 10: 100 m sampling design for field with unequal margins.

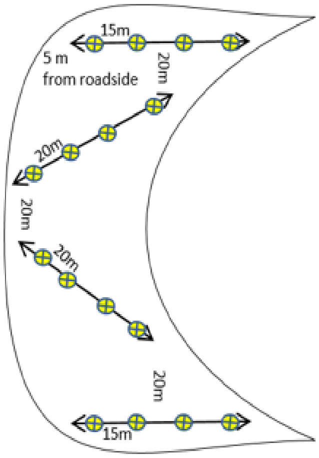


Figure 11: 60 m sampling design for smaller and irregular shaped fields.

Data Analysis

Delineation of Distribution Range. In this study I focus on local vegetation factors that may influence the presence/absence of a plant species within its potential distribution range. However, the presence/absence of a species is also limited by its geographic range, caused by dispersal limitation or climate and geological factors, which is beyond the scope of this study. In order to outline the potential distribution range for *Phalaris* and *Pastinaca*, I used the “minimum convex polygon” method in ArcGIS to delineate the range where *Phalaris* and *Pastinaca* was distributed (Figures 16 and 22).

The CRP sites outside each species’ distribution range were not included in the following analysis for each species (i.e. the absence of each species in these sites was considered to be due to climate or geological factors, not local factors).

Generalized Linear Regression. The CRP fields surveyed within the created minimum convex polygons were further analyzed in R version 3.5.2 (R Core Team 2019) using a logistic regression to assess the influence of several factors on the presence/absence of *Phalaris* and *Pastinaca*. The presence/absence of *Phalaris* and *Pastinaca* was used as a response variable in the model, while forb richness, grass richness, total grass cover, total forb cover, total bare ground, and site expiration year were used as predictor variables. In the *Phalaris* analysis, the presence of *Phalaris* was removed from grass species richness count and the coverage of *Phalaris* was removed from the total percent grass cover; similarly, in the *Pastinaca* analysis, the presence of *Pastinaca* was removed from the total forb species count and its coverage was not counted toward the total forb cover.

Results

Validation

Grass coverage estimated from the roadside was higher than the mean grass coverage observed from within the field (Figures 12 and 15; roadside-infield = 13.415.0%); this was especially true in Colorado and Idaho but less so in Iowa where the majority of *Phalaris* and *Pastinaca* were found. Forb coverage was also higher in roadside estimates than from within the field measurements (Figure 13; roadside-infield = 1.913.1%) especially in Colorado. Grass coverage and forb coverage estimated from roadside surveys had a strong positive correlation with the grass and forb coverage measured in the in-field surveys (Figure 14 and 15; grass coverage p-value=1.07e-11, R^2=0.5059; forb coverage p-value<2e-16, R^2=0.7174). The strong correlations between in-field and roadside estimates suggest that roadside estimates provide reasonably similar estimates as more traditional in-field measures.

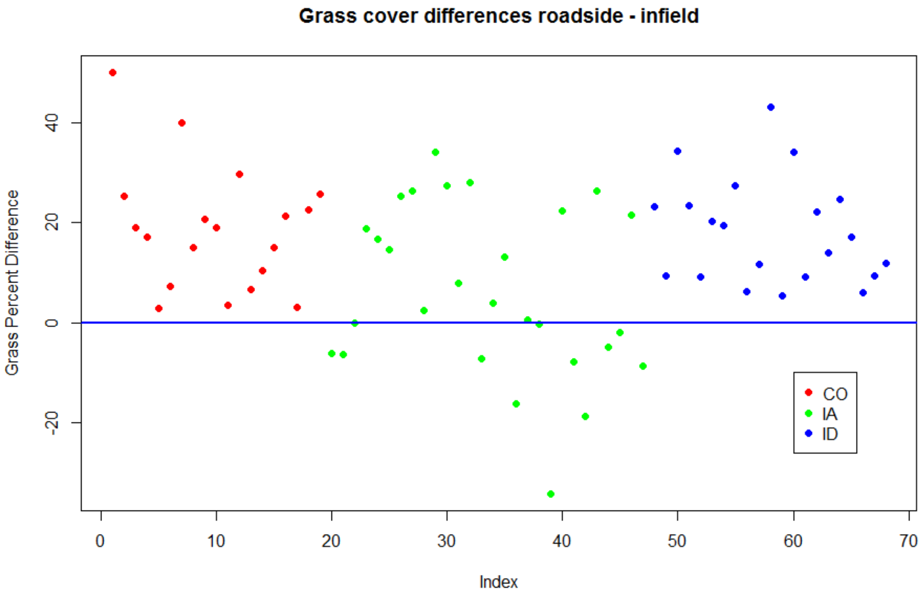


Figure 12: Differences between roadside and in-field grass cover estimate.

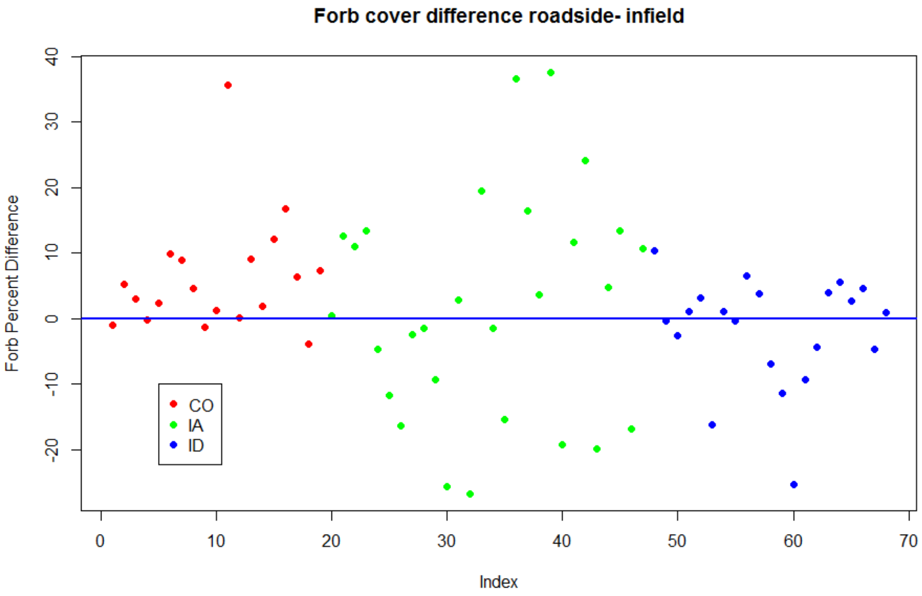


Figure 13: Differences between roadside and in-field cover estimate.

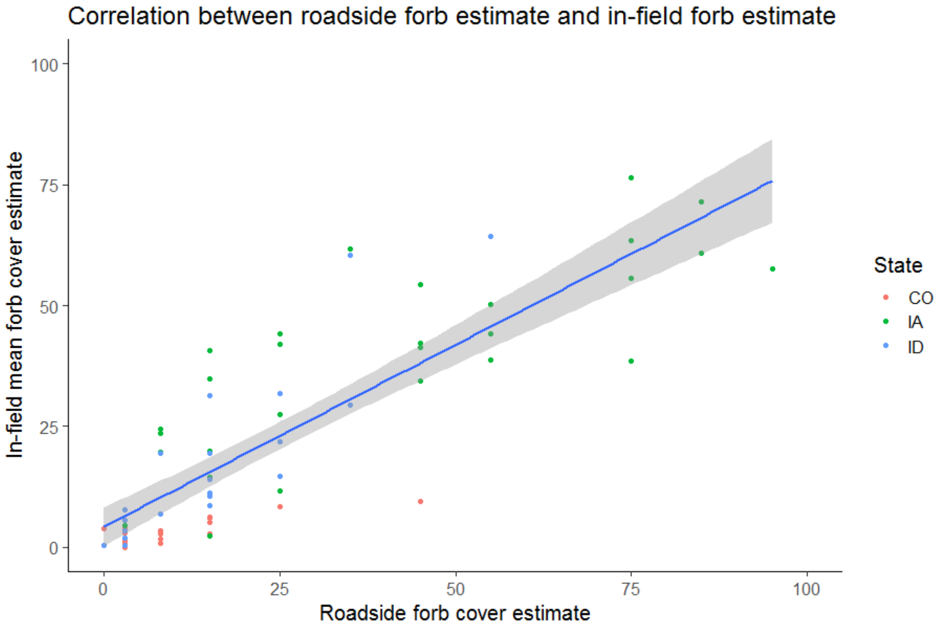


Figure 14: Correlation of forb coverage between in-field survey and roadside survey.

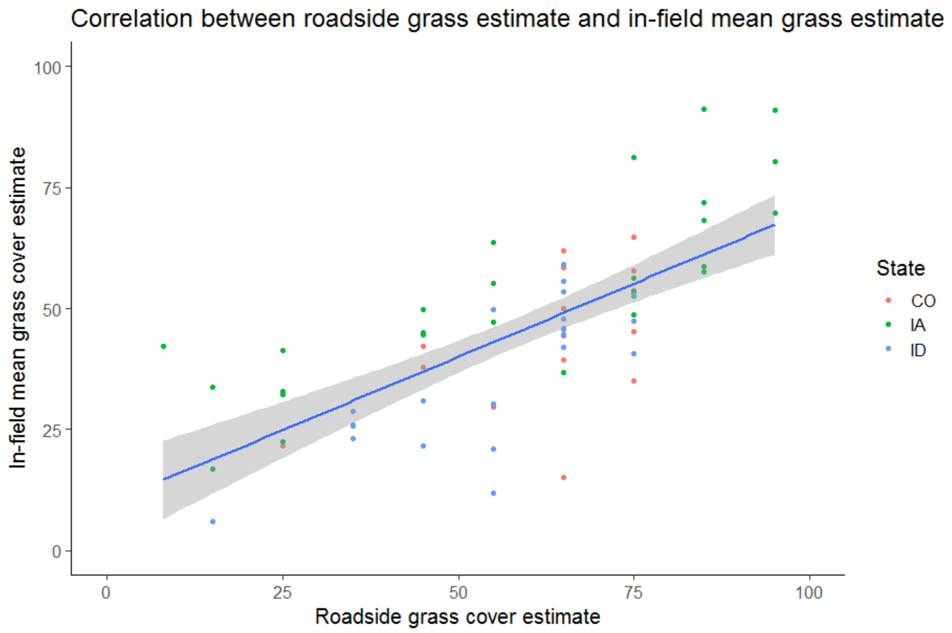


Figure 15: Correlation of grass coverage between in-field survey and roadside survey.

Phalaris

In its delineated range, *Phalaris* was present in 149 fields and absent in 484 (Figure 16; Table 5). *Phalaris* was present in all CRP conservation practices (Table 6) surveyed; and in 9 out of 14 states (Table 5). While *Phalaris* was present in OR, ID, and WA, for the purposes of this study I limited the analysis to climatically similar Midwestern US (MN, MO, ND, SD and IA) and did not include the geographically disjunct population of the Pacific Northwest. *Phalaris* range distribution within the created MCP showed *Phalaris* range in CRP fields covering portions of Minnesota, Iowa, North Dakota, South Dakota and Missouri (Figure 16).

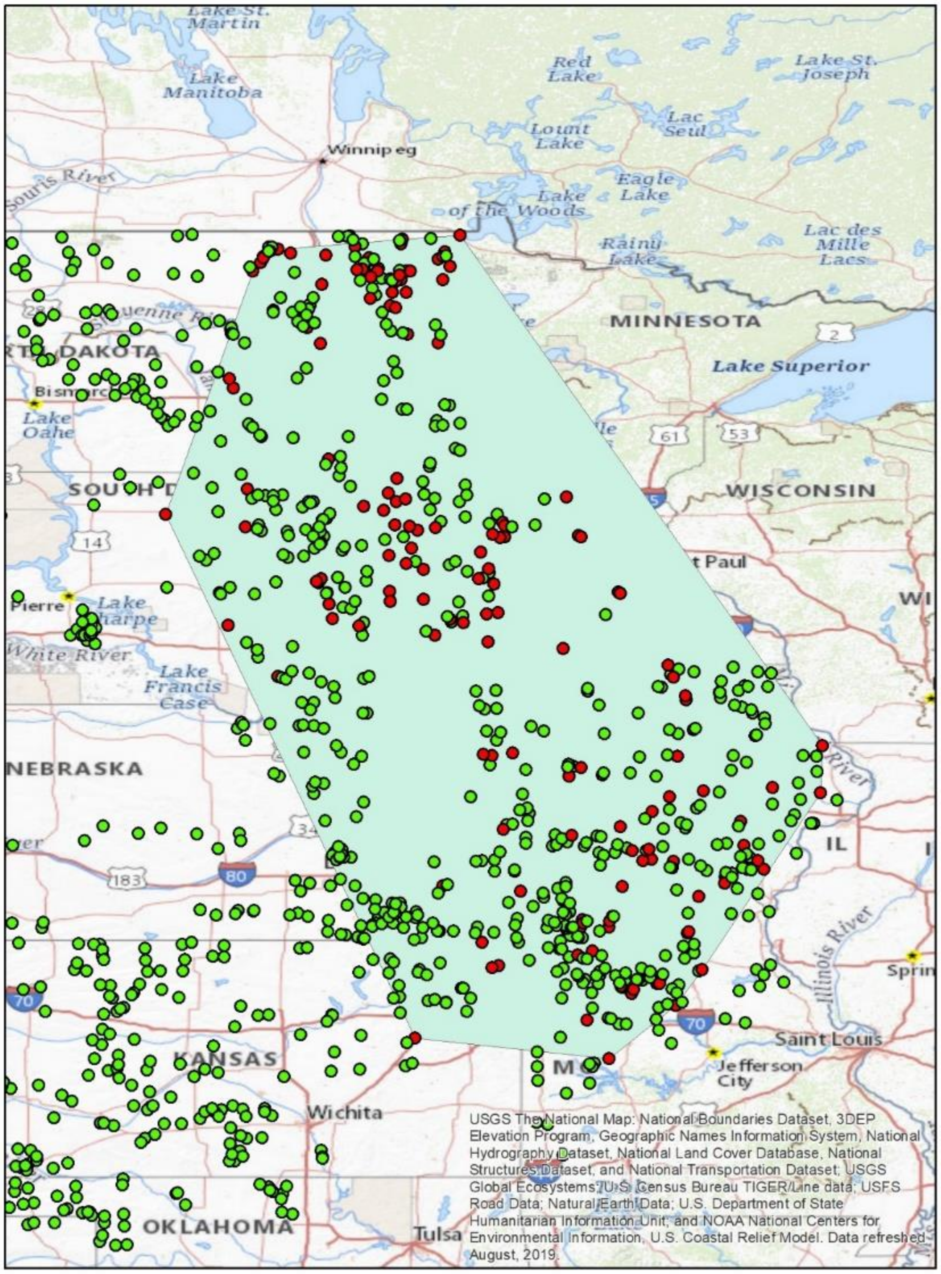


Figure 16: Minimum convex polygon for the distribution of *Phalaris* based on *Phalaris* presence (red circles) and absence (green circles) in CRP fields.

Phalaris presence was positively and significantly influenced by the type of CRP practice (Table 4; p=0.0004), due to higher presence in CP-23 (wetland practice) CRP fields. No other CP types were significantly different from one another (Tukey's HSD post-hoc tests, p > 0.05; Figure 17; Table 6).

Table 4: ANOVA results of logistic regression analysis testing the effect of local vegetation factors on *Phalaris* presence.

Source	**DF**	**Deviance**	**Resid. Df**	**Resid. Dev**	**Pr(>Chi)**
Intercept			632	690.86	
CP type	8	40.504	624	650.36	2.58E-06
Forb richness	1	2.677	623	647.68	0.101826
Grass richness	1	13.703	622	633.98	0.0002141
Grass cover	1	52.211	621	581.77	4.98E-13
Forb cover	1	45.617	620	536.15	1.44E-11
Bare ground	1	61.866	619	474.28	3.68E-15
Field age	1	0.044	618	474.24	0.8331895

Table 5: *Phalaris* presence, absence and prevalence in all 14 states surveyed.

State	**Absence**	**Presence**	**Prevalence**
CO	118	0	0%
IA	145	33	18.5%
ID	115	1	0.86%
KS	174	1	0.57%
MN	80	71	47%
MO	123	22	15.2%
MT	55	0	0%
ND	164	15	8.4%
NE	155	0	0%
OK	79	0	0%
OR	82	1	1.2%
SD	138	13	8.6%
TX	101	0	0%
WA	63	5	7.4%

Table 6: *Phalaris* presence, absence and prevalence in each CP program surveyed within delineated range; numbers (top) indicate the conservation practice (CP) number.

	1	**2**	**10**	**23**	**25**	**33**	**37**	**38**	**42**	**4D**
Absence	65	67	58	36	71	34	0	75	40	38
Presence	14	14	14	39	21	6	0	14	9	18
Prevalence (%)	17.7	17.3	19.4	52.0	22.8	15.0	0	7.7	18.3	32.1

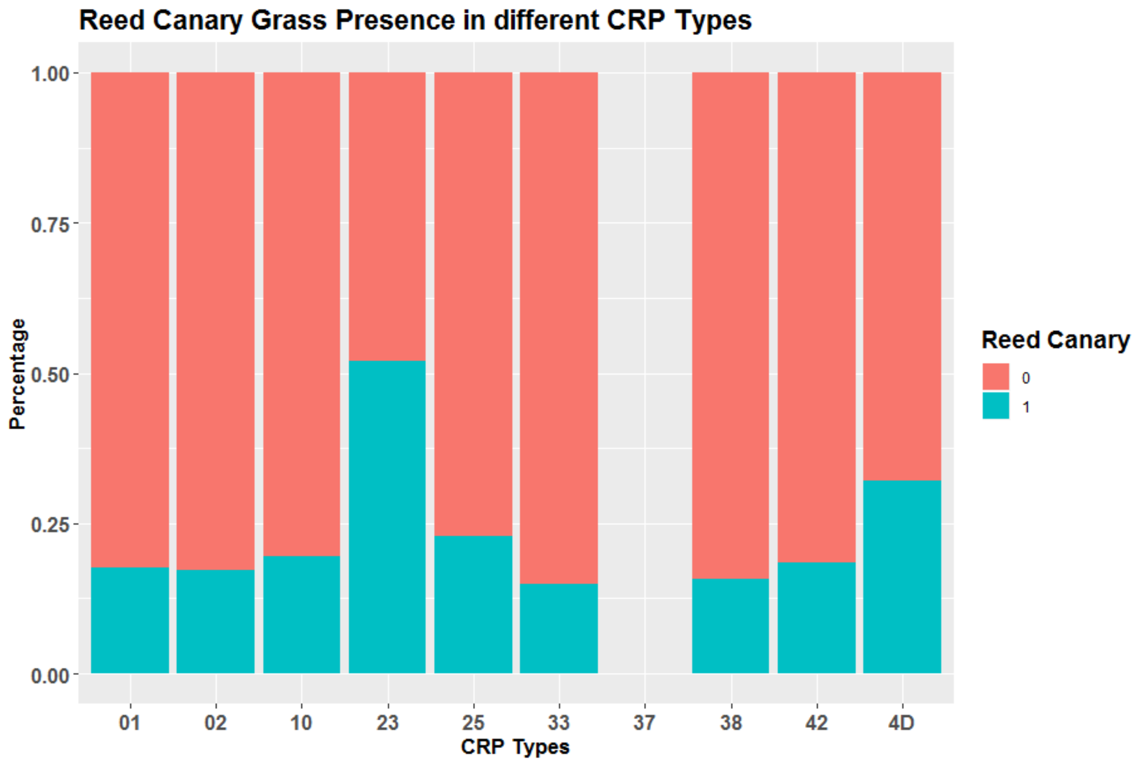


Figure 17: *Phalaris* presence in CRP fields based on CRP program type the field has been enrolled in. 1 (blue) indicates *Phalaris* presence, 0 (red) indicates *Phalaris* absence.

The number of years the CRP field had been enrolled in the program had no significant effect (p=0.8332) on the presence of *Phalaris* in the field. CRP fields with higher grass species richness (p=0.0128; Figure 18) and higher percent grass cover (p < 2e-16; Figure 19) had a lower likelihood of *Phalaris* presence. Greater forb species richness had no significant effect on the presence of *Phalaris* in CRP fields (p=0.848) and greater forb cover in the CRP fields decreased the chances of *Phalaris* being present (p=1.63e-07; Figure 20). Contrary to expectations, the amount of bare soil present showed a significant negative relationship with *Phalaris* presence (p=3.00e-12; Figure

21), fields with more bare soil areas were less likely to have *Phalaris* present. This result could be a correlation between the presence of *Phalaris* filling in bare soil areas and not the cause of *Phalaris* presence.

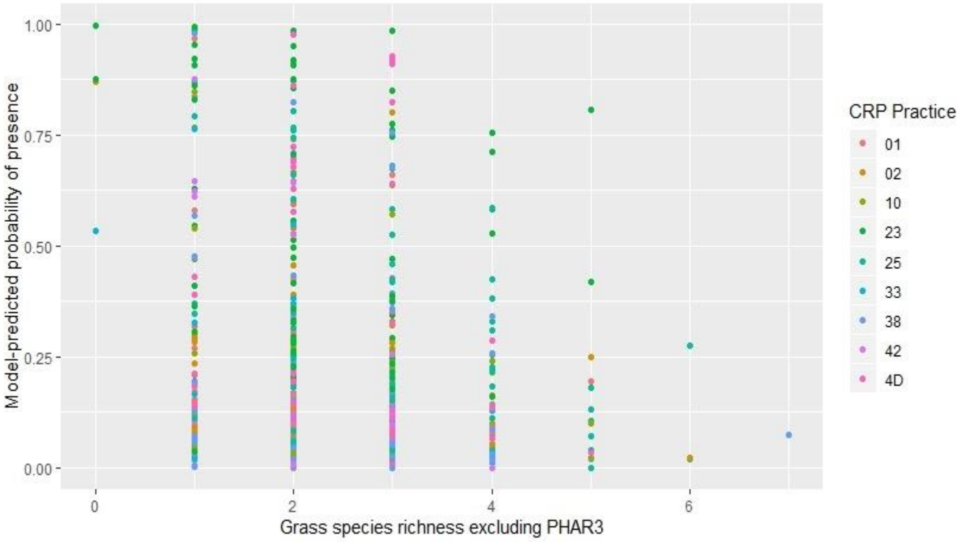


Figure 18: Relationship between the presence of *Phalaris* and grass species richness (excluding *Phalaris*) by CP type.

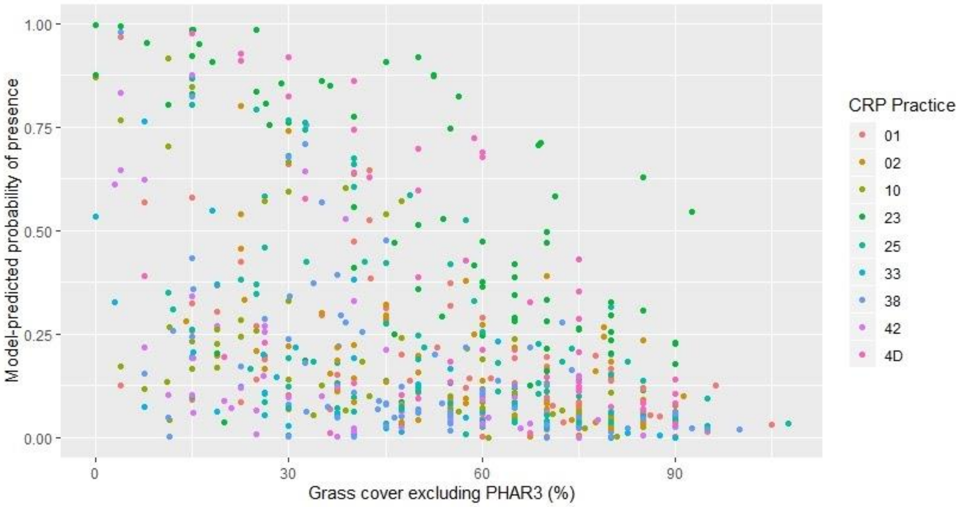


Figure 19: Relationship between the presence of *Phalaris* and percent grass cover (excluding *Phalaris*) by CP type.

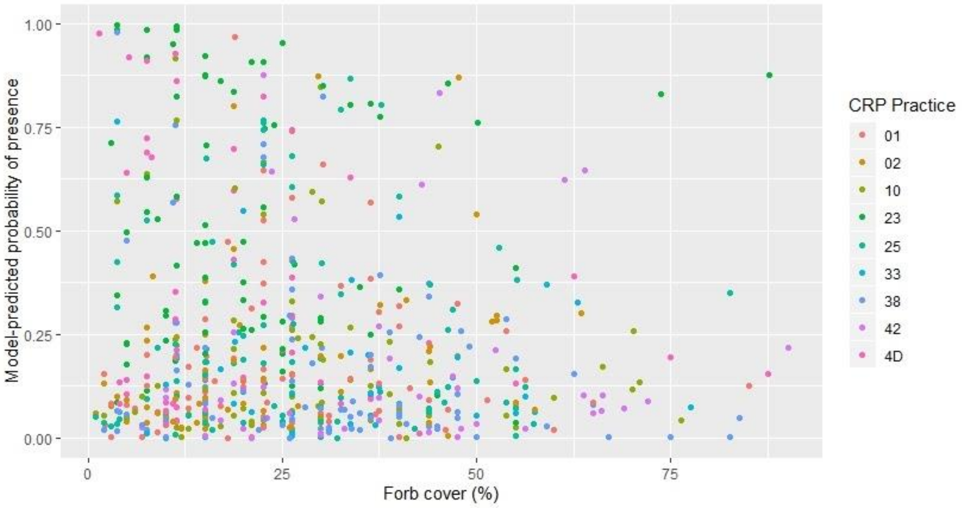


Figure 20: Relationship between presence of *Phalaris* and percent forb cover by CP type.

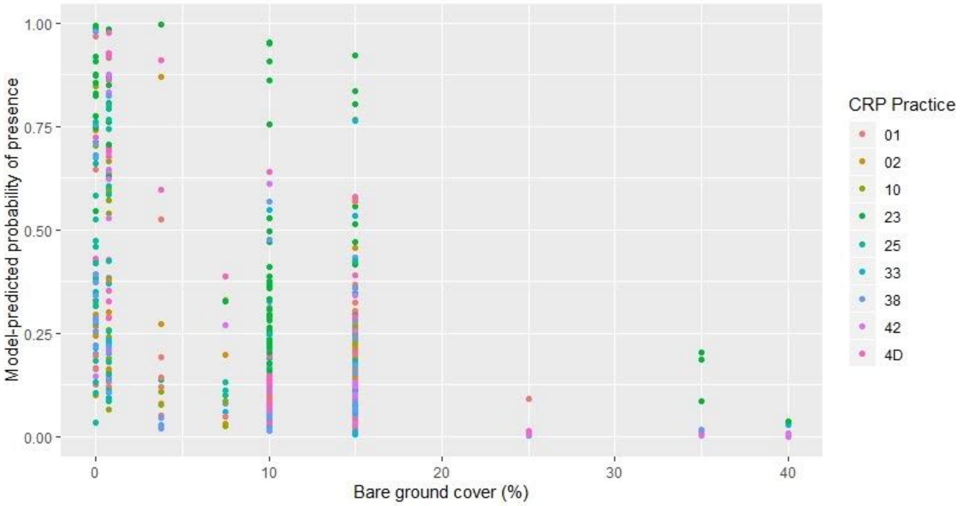
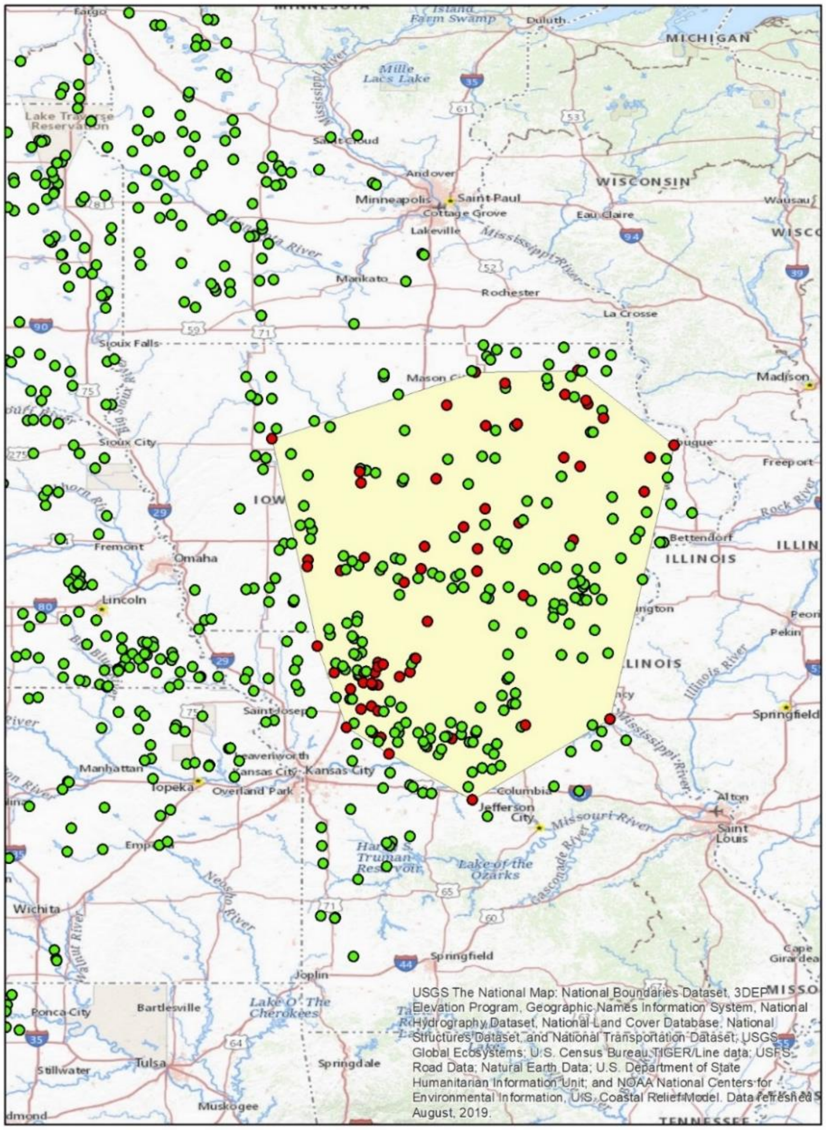


Figure 21: Relationship between presence of *Phalaris* and percent bare ground cover by CP type.

Pastinaca

Pastinaca was present in 3 out of 14 states: Iowa, Minnesota and Missouri (Figure 22; Table 7). Within the delineated range of *Pastinaca*, the species was present in 64 fields and absent from 265 fields (Figure 22; Table 7). *Pastinaca* was found in at least one field of each conservation practice and its presence did not vary among CP types (Tables 8 & 9). The number of years the CRP field had been enrolled in the CRP program was not a significant predictor of the presence of *Pastinaca* in the field (p=0.9993).



USGS The National Map: National Boundaries Dataset, 3DEP Elevation Program, Geographic Names Information System, National Hydrography Dataset, National Land Cover Database, National Structures Dataset, and National Transportation Dataset; USGS Global Ecosystems; U.S. Census Bureau TIGER/Line data; USFS Road Data; Natural Earth Data; U.S. Department of State Humanitarian Information Unit; and NOAA National Centers for Environmental Information, U.S. Coastal Relief Model. Data refreshed August, 2019.

Figure 22: Minimum convex polygon for the distribution of *Pastinaca* based on *Pastinaca* presence (red circles) and absence (green circles) in CRP fields.

Table 7: *Pastinaca* presence, absence and prevalence in all 14 states surveyed.

State	Absence	Presence	Prevalence
CO	118	0	0%
IA	143	35	19.7%
ID	116	0	0%
KS	175	0	0%
MN	149	2	1.3%
MO	118	27	18.6%
MT	55	0	0%
ND	179	0	0%
NE	155	0	0%
OK	79	0	0%
OR	83	0	0%
SD	151	0	0%
TX	101	0	0%
WA	68	0	0%

Table 8: *Pastinaca* presence, absence and prevalence in each CP program surveyed; numbers (top) indicate the conservation practice (CP) number.

	1	2	10	23	25	33	37	38	42	4D
Absence	246	265	250	110	146	86	47	175	153	212
Presence	10	7	5	2	12	5	2	7	9	19
Prevalence (%)	3.9	2.6	2.0	1.8%	7.6	5.5	4.1	3.8	5.6	8.2

Table 9: ANOVA results of logistic regression analysis testing the effect of local vegetation factors on *Pastinaca* presence.

Source	Df	Deviance	Resid. Df	Resid. Dev	Pr(>Chi)
Intercept			225	265.56	
CP type	8	12.14	217	253.42	0.1450652
Forb richness	1	0.3249	216	253.1	0.5686727
Grass richness	1	1.3331	215	251.76	0.2482538
Grass cover	1	9.1992	214	242.56	0.0024212
Forb cover	1	11.745	213	230.82	0.0006101
Bare ground	1	5.3959	212	225.42	0.0201843
Field age	1	0.0001	211	225.42	0.9931492

Pastinaca presence was not significantly correlated with either forb or grass species richness. Increased forb cover (excluding *Pastinaca*) significantly reduced *Pastinaca* presence (p=0.0006; Figure 23); but higher grass cover increased *Pastinaca* presence (p=0.002; Figure 24). Percent bare soil found in a CRP field (Figure 25) had a significant effect of the presence of *Pastinaca* (p=0.02); contrary to expectations CRP fields that had higher bare soil areas were less likely to have *Pastinaca* present. This result could be a correlation between the presence of *Pastinaca* filling in bare soil areas and not the cause of *Pastinaca* presence.

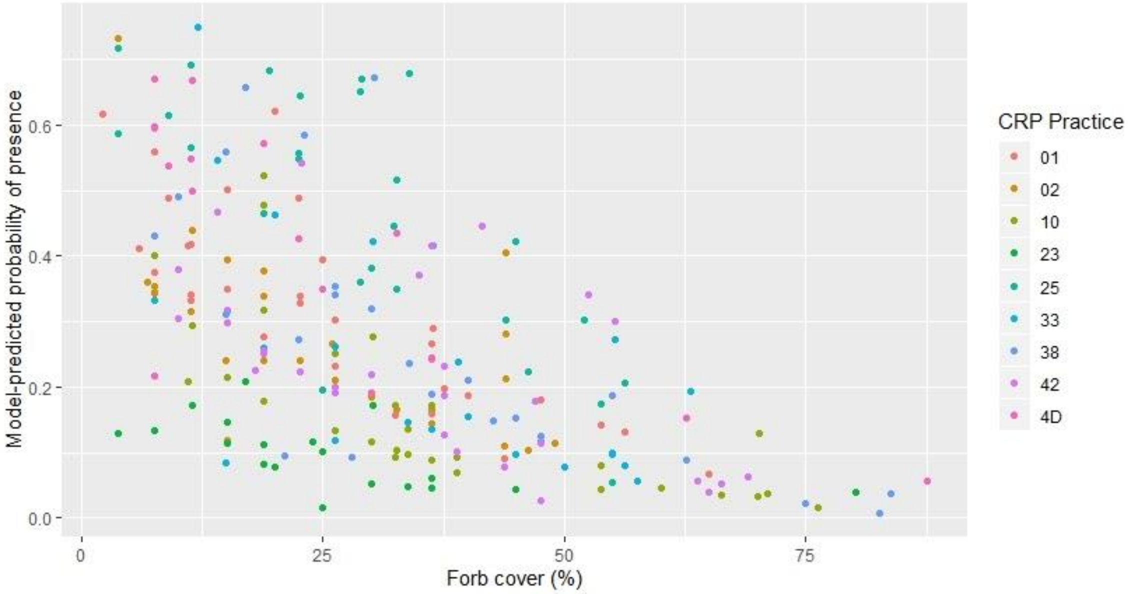


Figure 23: Relationship between presence of *Pastinaca* and percent forb cover (excluding *Pastinaca*).

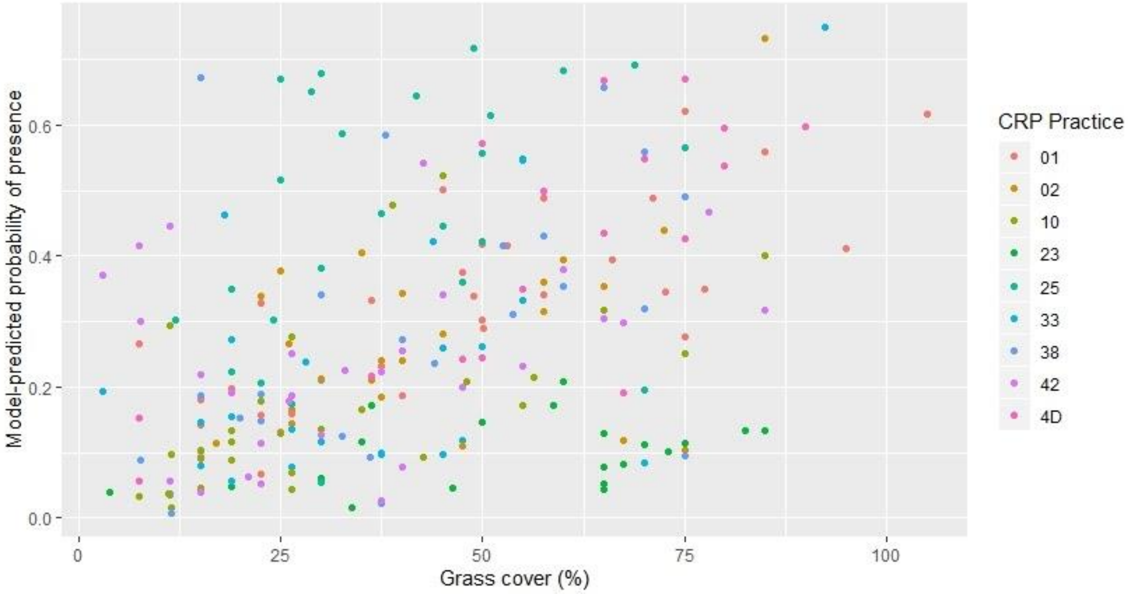


Figure 24: Relationship between presence of *Pastinaca* and percent grass cover by CP type.

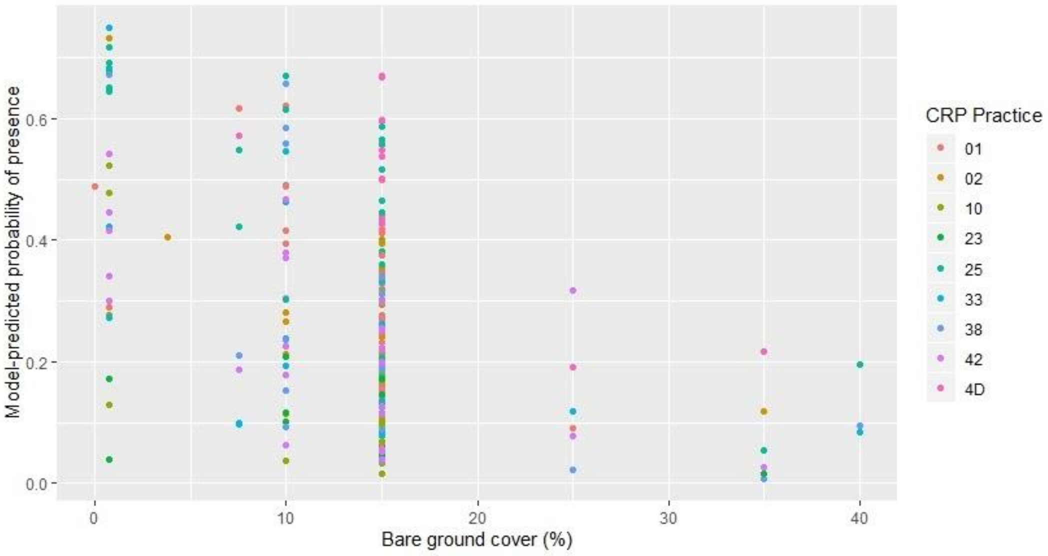


Figure 25: Relationship between presence of *Pastinaca* and percent bare ground cover by CP type.

Discussion

The purpose of CRP is to improve ecosystem services on sensitive agricultural lands. These agricultural landscapes create conditions favorable for weedy species, through large and frequent disturbances, transportation of invasive seeds on cattle and farm equipment and increased nutrient availability. These factors increase the chances that CRP fields can be invaded by weedy species that can have many negative economic and ecological consequences. This research aimed to determine what vegetation factors influence the presence of *Phalaris* and *Pastinaca* in CRP fields using a rapid roadside assessment. The close correspondence between our roadside assessment is a useful technique to rapidly survey vegetation in large numbers of CRP fields.

Using this method, we found that grass species richness, forb and grass cover were all significant factors that reduced the presence of *Phalaris* in CRP fields. For *Pastinaca*, forb and grass cover were the important predictors. Elton (1958) first proposed that more diverse communities are less susceptible to invasion, because of greater niche and resource utilization, and these results as well as many others (Lonsdale 1999; Fargione and Tilman 2005; Tilman 1997) uphold the hypothesis that higher species richness and percent cover is expected to reduce invasibility of a community. Forb species richness, however, did not have a significant effect on the presence of *Phalaris* or *Pastinaca*, suggesting that there are more factors than overall diversity affecting the resistance of a community to resistance.

Overall, the conservation practice did not have a significant effect on the presence of *Pastinaca*, and only CP-23 fields were significantly more likely to have *Phalaris* present. *Phalaris* is commonly found in wetland and riparian habitats, so its association with CP-23 is not a surprising result.

The less bare soil areas there were in a CRP field, the more likely it was to have *Phalaris* and *Pastinaca*. While this result was unexpected, one way this relationship might be explained is from the large monocultures that *Phalaris* creates when it establishes in a favorable area, decreasing bare soil areas. However, *Pastinaca* does not typically form large dense stands that reduce bare ground cover like *Phalaris* does, so while this explanation is possible, it seems less likely in the case of *Pastinaca*. This explanation could indicate that less bare ground areas are in response to the invasion of these species and not a factor causing the invasion.

Time since establishment was not a factor in the presence of *Phalaris* or *Pastinaca* in CRP fields. This result could indicate that disturbances on the site prior to and during reestablishment of perennial vegetation at the site created space and opportunity for these two species to colonize the site early in establishment. This is further supported by the fact that invasive species are expected to begin growth earlier than native species and that climate change may be increasing this phenomenon (Dickson et al. 2012). Considering CRP fields typically have a contract duration of 10-15 years, it leads to the question of how beneficial CRP fields are for native vegetation that may take longer to establish than "pioneer" invasive species.

Overall conclusions suggest that increased plant cover, and to some extent diversity can lead to a decreased chance of invasion into CRP fields. It appears that the presence of *Phalaris* and *Pastinaca* was most significantly affected by the cover of the functional group (forb or grass) that they belong to. The importance of functional groups in invasion has been identified as a potential important factor in invasion before (Lanta and Leps 2007). The results also suggest that these weedy species invade CRP fields early in the establishment of the field, as the chance of *Phalaris* and *Pastinaca* presence was not significantly affected through the time the CRP fields are under contract. This suggests efforts should be made early during vegetation establishment to prevent invasion by these weedy species.

CHAPTER FOUR

CONCLUSION

This research took an in-depth look at two potential ecosystem services provided by fields enrolled in the CRP, carbon sequestration and weed invasion resistance. These services are not directly targeted by the CRP program, but are potentially valuable services. Different variables that affected these services included time, species richness and cover. We also found that the use of roadside surveys is a rapid and useful way to estimate percent forb and grass cover and is a worthwhile method of monitoring CRP fields. Another interesting finding is that although different conservation practices follow different requirements, they generally are not a significant factor in either carbon sequestration or weedy invasion.

Time since the establishment of CRP had a significant effect on carbon sequestration in CRP fields, increasing total soil carbon with increased time. This indicates that if carbon sequestration is a conservation goal of a CRP field, keeping the field in CRP for longer periods of time will be more beneficial to soil carbon sequestration. Interestingly, time had no significant effect on weedy invasion in CRP fields, this indicates that CRP fields should be monitored early in establishment so preventative actions can be taken against it.

We also found that increased species richness and cover by the same functional group (grass or forb) had a significant impact on the presence of *Phalaris* and *Pastinaca*. *Phalaris* was less likely to be present in fields with high grass species richness and cover and *Pastinaca* was less likely to be present in fields with high forb species richness and cover. This indicates that seeding and establishing diverse forb and grass species in a CRP field will decrease the chance of weedy invasion, potentially keeping CRP fields from becoming a source of weeds that can possibly invade agricultural fields.

When it comes to carbon sequestration and increased weedy invasion resistance, the specific conservation practice that the CRP field is enrolled in doesn’t seem to make a difference in the Midwest, with the exception of *Phalaris* invasion in CP-23 fields, which should be closely monitored for *Phalaris* invasion as they are likely to create monocultures. Although these programs have different conservation goals with different seeding and management requirements, carbon sequestration and weedy invasion are not significantly affected by these differences. Roadside surveys were validated using in-field surveys, where the same measurements were recorded from both the roadside and in the field. Results from these validated (in-field surveys) fields indicate that roadside surveys are a faster and sufficient replacement for in-field measurements when it comes to estimating species richness and cover.

In conclusion, CRP fields bring many environmental benefits to agricultural landscapes, along with the many services the program aims to provide; they also provide benefits that are not targeted by the program, including carbon sequestration. However, it is important to monitor these CRP fields for potential issues as they can become a potential source of invasive species. Road-side surveys are a sufficient way to monitor these CRP fields for vegetative variables such as forb and grass species cover and percentage estimations of weedy invaders and can help in improving the state of these CRP fields.

REFERENCES

Adams WA. 1973. The effect of organic matter on the bulk and true densities of some uncultivated podzolic soils. *Journal of Soil Science*, 24(1), 10–17.

Ampleman MD, Crawford KM, Fike DA. 2013. Differential soil organic carbon storage at forb- and grass-dominated plant communities, 33 years after tallgrass prairie restoration. *Plant and Soil*, 374(1-2), 899–913.

Amsellem L, Noyer J-L, Hossaert-Mckey M. 2001. Evidence for a switch in the reproductive biology of *Rubus alceifolius* (Rosaceae) towards apomixis, between its native range and its area of introduction. *American Journal of Botany*, 88(12), 2243–2251.

Averill KM, Ditommaso A. 2007. Wild Parsnip (*Pasinaca sativa*): A troublesome species of increasing concern. *Weed Technology*, 21(1), 279–287.

Baker HG. 1974. The evolution of weeds. *Annual Review of Ecology and Systematics*, 5(1), 1–24.

Baker JM, Ochsner TE, Venterea RT, Griffis TJ. 2007. Tillage and soil carbon sequestration—What do we really know? *Agriculture, Ecosystems & Environment*, 118(1-4), 1–5.

Ball DF. 1964. Loss-on-ignition as an estimate of organic matter and organic carbon in non-calcareous soils. *Soil Science*, 15(84–92).

Baltensperger AA, Kalton RR. 1958. Variability in Reed Canarygrass, *Phalaris Arundinacea L*. I. agronomic characteristics 1. *Agronomy Journal*, 50(11), 659–663.

Baskin JM, Baskin CM. 1979. Studies on the autecology and population biology of the weedy monocarpic perennial, *Pastinaca Sativa*. *The Journal of Ecology*, 67(2), 601-610.

Bazzaz FA. 1986. Life history of colonizing plants: Some demographic, genetic, and physiological features. In: Ecology of biological invasions of North America and Hawaii. New York, NY: Springer. 96–110.

Bellamy PH, Loveland PJ, Bradley RI, Lark RM, Kirk GJD. 2005. Carbon losses from all soils across England and Wales 1978–2003. *Nature*, 437(7056), 245–248.

Bernacchi CJ, Hollinger SE, Meyers T. 2005. The conversion of the corn/soybean ecosystem to no-till agriculture may result in a carbon sink. *Global Change Biology*, 0(0).

Boatman ND, Rew LJ, Theaker AJ, Froudwillams RJ. 1994. The impact of nitrogen fertilizers on field margin flora. *British Crop Protection Council Monograph Series*. 58, 209-215.

Braithwaite RW, Lonsdale WM. 1987. The rarity of *Sminthopsis virginiae*, (Marsupialia: Dasyuridae) in relation to natural and unnatural habitats. *Conservation Biology*, 1(4), 341–343.

Brown RL, Peet RK. 2003. Diversity and invasibility of southern Appalachian plant communities. *Ecology*, 84(1), 32–39.

Burke MJW, Grime JP. 1996. An experimental study of plant community invasibility. *Ecology*, 77(3), 776–790.

Buyanovsky GA, Wagner GH. 1997. In: Paul, E.A., et al. (Eds.), Soil organic matter in temperate ecosystems: Long-term experiments in North America. Boca Raton, FL: CRC Press. 73–83.

Caplan JS, Wheaton CN, Mozdzer TJ. 2014. Belowground advantages in construction cost facilitate a cryptic plant invasion. *AoB PLANTS*, 6.

Carlson IT, Oram RN, Surprenant J. 2015. Reed Canarygrass and other *Phalaris* species. *Cool-Season Forage Grasses Agronomy Monographs*, 34, 569–604.

Carlton JT. 1996. Biological invasions and cryptogenic species. *Ecology*, 77(6), 1653–1655.

Cavers PB, Harper JL. 1967. Studies in the dynamics of plant populations: I. The fate of seed and transplants introduced into various habitats. *The Journal of Ecology*, 55(1), 59.

Clark JS, Ji Y. 1995. Fecundity and dispersal in plant populations: Implications for structure and diversity. *The American Naturalist*, 146(1), 72–111.

Clark JS, Fastie C, Hurtt G, Jackson ST, Johnson C, King GA, Lewis M, Lynch J, Pacala S, Prentice C. 1998. Reids paradox of rapid plant migration. *BioScience*, 48(1), 13–24.

Coops H, Van der Velde G. 1995. Seed dispersal, germination and seedling growth of six helophyte species in relation to water-level zonation. *Freshwater Biology*, 34(1), 13–20.

Crawley MJ, Kornberg H, Lawton JH, Usher MB, Southwood R, O'Connor RJ, Gibbs A. 1986. The population biology of invaders [and discussion]. Philosophical Transactions of the Royal Society of London. *Series B, Biological Sciences*, 314(1167), 711-31.

Cruden RW, Hermann-Parker SM. 1977. Temporal dioecism: An alternative to dioecism. *Evolution*, 31(4), 863-866.

Dantonio CM, Vitousek PM. 1992. Biological invasions by exotic grasses, the grass/fire cycle, and global Change. *Annual Review of Ecology and Systematics*, 23(1), 63–87.

Davis MA, Pelsor M. 2001. Experimental support for a resource-based mechanistic model of invasibility. *Ecology letters*, 4(5), 421-428.

De Deyn GB, Shiel RS, Ostle NJ, Mcnamara NP, Oakley S, Young I, Freeman C, Fenner N, Quirk H, Bardgett RD. 2010. Additional carbon sequestration benefits of grassland diversity restoration. *Journal of Applied Ecology*, 48(3), 600–608.

Derner JD, Schuman GE. 2007. Carbon sequestration and rangelands: A synthesis of land management and precipitation effects. *Journal of Soil and Water Conservation*, 62(2), 77-85.

Dickson TL, Hopwood JL, Wilsey BJ. 2012. Do priority effects benefit invasive plants more than native plants? An experiment with six grassland species. *Biological Invasions*, 14(12), 2617–2624.

Donigian AS Jr., Barnwell TO, Jackson RB, Patwardhan AS, Weinreich KB, Rowell AL, Chinnaswamy RV, Cole CV. 1994. Assessment of alternative management practices and policies affecting soil carbon in agroecosystems of the central United States. Publication No. EPA/600/R-94/067. U.S. Environmental Protection Agency, Athens, Georgia.

Du X, Hennessy D, Edwards WA. 2008. Does a rising biofuels tide raise all boats? A study of cash rent determinants for Iowa farmland under hay and pasture. *Journal of Agricultural & Food Industrial Organization*, 6(2).

Duiker SW. 2005. Effects of soil compaction. Penn State Extension. Available at: https://extension.psu.edu/effects-of-soil-compaction

Dukes JS, Mooney HA. 1999. Does global change increase the success of biological invaders? *Trends in Ecology & Evolution*, 14(4), 135–139.

Dunstan PK, Johnson CR. 2006. Linking richness, community variability, and invasion resistance with patch size. *Ecology*, 87(11), 2842–2850.

Elton CS. 1958. The ecology of invasions by animals and plants. University of Chicago Press 2000 ed. Chicago: U of Chicago.

Fargione JE, Tilman D. 2005. Diversity decreases invasion via both sampling and complementarity effects. *Ecology Letters*, 8(6), 604-611.

Farrar J, Hawes M, Jones D, Lindow S. 2003. How roots control the flux of carbon to the rhizosphere. *Ecology*, 84(4), 827–837.

Fornara DA, Tilman D. 2008. Plant functional composition influences rates of soil carbon and nitrogen accumulation. *Journal of Ecology*, 96(2), 314–322.

Franklin J, Maarel E. 2012. Vegetation ecology. Chichester: Wiley-Blackwell.

[FSA] Farm Service Agency. 2012. CRP contract summary and statistics. Available at: http://www.fsa.usda.gov/FSA/webapp?area=home&subject=copr&topic=rns-css.

[FSA] Farm Service Agency. 2017. Programs and services: CRP practices. Available at: https://www.fsa.usda.gov/programs-and-services/conservation-programs/crp-practices-library/index

Galatowitsch SM, Anderson NO, Ascher PD. 1999. Invasiveness in wetland plants in temperate North America. *Wetlands*, 19(4), 733–755.

Gebhart, DL, Johnson HB, Mayeux HS, Polley HW. 1994. The CRP increases soil organic carbon. *Journal of Soil and Water Conservation*, 49(5), 488-492.

Germain RM, Jones NT, Grainger TN. 2019. Cryptic dispersal networks sha pe biodiversity in an invaded landscape. *Ecology*, 100(8).

Gleason HA, Cronquist A. 1991. Manual of vascular plants of Northeastern United States and adjacent Canada. Bronx, NY: New York Botanical Garden, 2007.

Haddaway NR, Hedlund K, Jackson LE, Kätterer T, Lugato E, Thomsen IK, Jørgensen HB, Isberg PE. 2016. How does tillage intensity affect soil organic carbon? A systematic review protocol. *Environmental Evidence*, 5(1).

Haile-Mariam S, Collins HP, Higgins SS. 2008. Greenhouse gas fluxes from an irrigated Sweet Corn (*Zea mays L.*)-Potato (*Solanum tuberosum L.*) Rotation. *Journal of Environmental Quality*, 37(3), 759–771.

Haragan PD. 2015. Weeds of Kentucky and adjacent states a field guide. Lexington: The University Press of Kentucky.

He Z, Bentley LP, Holaday AS. 2010. Greater seasonal carbon gain across a broad temperature range contributes to the invasive potential of *Phalaris arundinacea* (Poaceae; reed canary grass) over the native sedge *Carex stricta* (Cyperaceae). *American Journal of Botany*, 98(1), 20–30.

Hejda M, Pyšek P, Jarošík V. 2009. Impact of invasive plants on the species richness, diversity and composition of invaded communities. *Journal of Ecology*, 97(3), 393–403.

Hendrix SD, Trapp EJ. 1992. Population demography of *Pastinaca Sativa* (Apiaceae): Effects of seed mass on emergence, survival, and recruitment. *American Journal of Botany*, 79(4),365–375.

Hobbs RJ, Mooney HA. 1986. Community changes following shrub invasion of grassland. *Oecologia*, 70(4),508–513.

[IPCC] Pachauri RK, Allen MR, Barros VR, Broome J, Cramer W, Christ R, Church JA, Clarke L, Dahe Q, Dasgupta P, Dubash NK. 2014. Climate change 2014: Synthesis report. Contribution of Working Groups I, II and III to the fifth assessment report of the Intergovernmental Panel on Climate Change. Ipcc.

Jakubowski AR, Casler MD, Jackson RD. 2011. Has selection for improved agronomic traits made Reed Canarygrass invasive? *PLoS ONE*, 6(10).

Jakubowski AR, Casler MD, Jackson RD. 2010. Landscape context predicts Reed Canarygrass invasion: Implications for management. *Wetlands*, 30(4), 685–692.

Kell DB. 2012. Large-scale sequestration of atmospheric carbon via plant roots in natural and agricultural ecosystems: why and how. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 367(1595), 1589–1597.

Kellogg CH, Bridgham SD, Leicht SA. 2003. Effects of water level, shade and time on germination and growth of freshwater marsh plants along a simulated successional gradient. *Journal of Ecology*, 91(2),274–282.

Kephart KD, Buxton DR. 1993. Forage quality responses of C3 and C4 perennial grasses to shade. *Crop Science*, 33(4),831–837.

Kercher SM, Herr-Turoff A, Zedler JB. 2006. Understanding invasion as a process: The case of *Phalaris arundinacea* in wet prairies. *Biological Invasions*, 9(6), 657–665.

Kercher SM, Zedler JB. 2004. Multiple disturbances accelerate invasion of reed canary grass (*Phalaris arundinacea L.*) in a mesocosm study. *Oecologia*, 138(3), 455–464.

Konen ME, Jacons PM, Burras CL, Talage BJ, Mason JA. 2002. Equations for predicting soil organic carbon using loss-on-ignition for North Central U.S. soils. *Soil Science Society of America Journal*, 66(6), 1878-1881.

Kucharik CJ, Brye KR, Norman JM, Foley JA, Gower ST, Bundy LG. 2001. Measurements and modeling of carbon and nitrogen cycling in agroecosystems of Southern Wisconsin: Potential for SOC sequestration during the next 50 years. *Ecosystems*, 4(3), 237–258.

Kucharik CJ, Roth JA, Nabielski RT. 2003. Statistical assessment of a paired-site approach for verification of carbon and nitrogen sequestration on Wisconsin Conservation Reserve Program land. *Journal of Soil and Water Conservation*, 58(1), 58-66.

Kucharik CJ, Fayram NJ, Cahill KN. 2006. A paired study of prairie carbon stocks, fluxes, and phenology: Comparing the world's oldest prairie restoration with an adjacent remnant. *Global Change Biology*, 12(1), 122–139.

Lal R. 2004. Soil carbon sequestration impacts on global climate change and food security. *Science*, 304(5677), 1623–1627.

Landgraff A, Junttila O. 1979. Germination and dormancy of Reed Canary-Grass seeds (*Phalaris arundinacea*). *Physiologia Plantarum*, 45(1), 96–102.

Lanta V, Leps J. 2007. Effects of species and functional group richness on production in two fertility environments: An experiment with communities of perennial plants. *Acta Oecologica*, 32(1), 93-103.

Lavergne S, Molofsky J. 2004. Reed Canary Grass (*Phalaris arundinacea*) as a biological model in the study of plant invasions. *Critical Reviews in Plant Sciences*, 23(5), 415–429.

Lavergne S, Molofsky J. 2007. Increased genetic variation and evolutionary potential drive the success of an invasive grass. *Proceedings of the National Academy of Sciences*, 104(10), 3883–3888.

Levine JM. 2000. Species diversity and biological invasions: Relating local process to community pattern. *Science*, 288(5467), 852–854.

Lewis DQ, Pope RO. 2001. An overview and management plan of Iowa's non-native, invasive, terrestrial forbs. *Journal of the Iowa Academy of Science: JIAS.* 108(4), 116-123.

Li C, Fultz LM, Moore-Kucera J, Acosta-Martínez V, Horita J, Strauss R, Zak J, Calderón F, Weindorf D. 2017. Soil carbon sequestration potential in semi-arid grasslands in the conservation reserve program. *Geoderma*, 294, 80–90.

Lindig-Cisneros R, Zedler J. 2001. Effect of light on seed germination in *Phalaris arundinacea* L.(reed canary grass). *Plant Ecology*, 155(1), 75-78.

Lindig-Cisneros R, Zedler JB. 2002. Relationships between canopy complexity and germination microsites for *Phalaris arundinacea* L. *Oecologia*, 133.2, 159-167.

Lodge DM, Williams S, MacIsaac HJ, Hayes KR, Leung B, Reichard S, Mack RN, Moyle PB, Smith M, Andow DA, Carlton JT, McMichael A. 2006. Biological invasions: Recommendations for U.S. policy and management. *Ecological Applications*, 16(6), 2035-054.

Lonsdale WM. 1999. Global patterns of plant invasions and the concept of invasibility. *Ecology*, 80(5), 1522-536.

Lorenzi HJ, Jeffery LS. 1987. Weeds of the United States and their control. New York: Van Nostrand Reinhold.

Luo Z, Wang E, Sun OJ. 2010. Can no-tillage stimulate carbon sequestration in agricultural soils? A meta-analysis of paired experiments. *Agriculture, Ecosystems & Environment*, 139(1-2), 224-231.

Mader DL. 1963. Soil variability—A serious problem in soil-site studies in the Northeast. *Soil Science Society of America Journal*, 27(6), 707-709.

Mack RN. 1991. The commercial seed trade: An early disperser of weeds in the United States. *Economic Botany*, 2, 257-73.

Mack RN. 1996. Predicting the identity and fate of plant invaders: Emergent and emerging approaches. *Biological Conservation*, 78(1-2), 107-21.

Mack RN, Lonsdale WM. 2001. Humans as global plant dispersers: Getting more than we bargained for. Current introductions of species for aesthetic purposes present the largest single challenge for predicting which plant immigrants will become future pests. *BioScience*, 51(2), 95-102.

Maron JL, Vilà M, Bommarco R, Elmendorf S, Beardsley P. 2004. Rapid evolution of an invasive plant. *Ecological Monographs*, 74(2), 261-80.

McCarl BA, Metting FB, Rice CW. 2007. Soil carbon sequestration. *Climatic Change*, 80(1-2), 1-3.

Merigliano MF, Lesica P. 1998. The native status of Reed Canarygrass (*Phalaris arundinacea L.*) in the inland northwest, USA. *Natural Areas Journal*, 18(3), 223-230.

Metting F, Smith B, Amthor J, Izaurralde L. 2001. Science needs and new technology for increasing soil carbon sequestration. *Climatic Change*, 51(1), 11-34.

Milton S. 2004. Grasses as invasive alien plants in South Africa. *South African Journal of Science*, 100, 69-75.

[MDA] Minnesota Department of Agriculture. Wild Parsnip. 2020. Available at: https://www.mda.state.mn.us/plants/pestmanagement/weedcontrol/noxiouslist/wildparsnip

[MDNR] Minnesota Department of Natural Resources. 2020. Wild Parsnip (*Pastinaca sativa*). Invasive terrestrial plants. Available at: http://www.dnr.state.mn.us/invasives/terrestrialplants/herbaceous/wildparsnip.html

Muenscher WCL. 1987. Weeds. 1st Pbk. ed. Ithaca, N.Y.: Comstock Pub. Associates.

Murphy CA, Foster BL, Ramspott ME, Price KP. 2004. Grassland management effects on soil bulk density. *Transactions of the Kansas Academy of Science*, 107(1-2), 45-54.

Murphy CA, Foster B., Ramspott ME, Price, KP. 2006. Effects of cultivation history and current grassland management on soil quality in northeastern Kansas. *Journal of Soil and Water Conservation*, 61(2), 75-84.

Myhre G, Shindell D, Bréon FM, Collins W, Fuglestvedt J, Huang J, Koch D, Lamarque JF, Lee D, Mendoza B, Nakajima T, Robock A, Stephens G, Takemura T, Zhang H. 2013. Anthropogenic and natural radiative forcing. In: Climate change 2013: The physical science basis. Contribution of working group I to the fifth assessment report of the intergovernmental panel on climate change [Stocker TF, Qin D, Plattner GK, Tignor M, Allen SK, Boschung J, Nauels A, Xia Y, Bex V, Midgley PM, (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Nitao JK, Zangerl AR. 1987. Floral development and chemical defense allocation in wild parsnip (Pastinaca sativa). *Ecology*, 68(3), 521-529.

Novak SJ, Mack RN. 2001. Tracing plant introduction and spread: Genetic evidence from *Bromus Tectorum* (Cheatgrass). Introductions of the invasive grass *Bromus Tectorum* worldwide were broadly similar and closely tied to patterns of European human immigration. *BioScience*, 51(2), 114-22.

Odland A. 1997. Development of vegetation in created wetlands in western Norway. *Aquatic Botany*, 59(1-2), 45-62.

Odland A, Del Moral R. 2002. Thirteen years of wetland vegetation succession following a permanent drawdown, Myrkdalen Lake, Norway. *Plant Ecology*, 162(2), 185-198.

Ordonez RA, Castellano MJ, Hatfield JL, Helmers MJ, Licht MA, Liebman M, Dietzel R, Martinez-Feria R, Iqbal J, Puntel LA, Cordova SC. 2018. Maize and soybean root front velocity and maximum depth in Iowa, USA. *Field Crops Research*, 215, 122–131.

Ostrem L. 1987. Studies on genetic variation in reed canarygrass, *Phalaris arundinacea* L. I. Alkaloid type and concentration. *Hereditas*, 107(2):235–248.

Ostrem L. 1988a. Studies on genetic variation in reed canarygrass, *Phalaris arundinacea* L. II. Forage yield and quality. *Hereditas*, 108(1), 103-13.

Ostrem LIV. 1988b. Studies on genetic variation in reed canary grass, *Phalaris arundinacea* L.: III. Seed yield and seed yield components. *Hereditas*, 108(2), 159-168.

Paul EA, Elliott ET, Paustian K, Cole CV. 1997. Soil organic matter in temperate agroecosystems: long-term experiments in North America. Boca Raton, FL: CRC Press.

Paveglio FL, Kilbride KM. 2000. Response of vegetation to control of reed canarygrass in seasonally managed wetlands of southwestern Washington. *Wildlife Society Bulletin*, pp.730-740.

Pimentel D, Hunter MS, Lagro JA, Efroymson RA, Landers JC, Mervis FT, Mccarthy CA, Boyd AE. 1989. Benefits and risks of genetic engineering in agriculture. *BioScience*, 39(9), 606-14.

Pimentel DL, Lach L, Zunigs R, Morrison D. 2000. Environmental and economic costs associated with non-indigenous species in the United States. *Bioscience*, 50(1), 53-65.

Pimentel D, Zuniga R, Morrison D. 2005. Update on the environmental and economic costs associated with alien-invasive species in the United States. *Ecological Economics*, 52(3), 273-288.

Poirier V, Angers DA, Rochette P, Chantigny MH, Ziadi N, Tremblay G, Fortin J. 2009. Interactive effects of tillage and mineral fertilization on soil carbon profiles. *Soil Science Society of America Journal*, 73(1), 255-61.

Price SC, SK Jain. 1981. Are inbreeders better colonizers? *Oecologia*, 49(2), 283-286.

Pysek P, Pergl J, Essl F, Lenzner B, Dawson W, Kreft H, Weigelt P, Winter M, Kartesz J, Nishino M, Antonova L, Barcelona J, Cabezas F, Cardenas D, Cardenas-Toro J, Castano, N, Chacon-Madrigal E, Chatelain C, Dullinger S, van Kleunen M. 2017. Naturalized alien flora of the world: Species diversity, taxonomic and phylogenetic patterns, geographic distribution and global hotspots of plant invasion. *Preslia*, 89(3), 203-274.

R Core Team (2019). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Rahmat S, Fernando LK, Banuwa IS, Buchari H, Utomo M. 2012. Carbon storage and carbon dioxide emission as influenced by long-term conservation tillage and nitrogen fertilization in corn-soybean rotation. *Jurnal Tanah Tropika*, 17(1), 75-84.

Rastogi M, Singh S, Pathak H. 2002. Emission of carbon dioxide from soil. *Current Science*, 82(5), 510-517.

Rejmanek M. 1989. Invasibility of plant communities. In: Drake JA, Mooney HA, Castri F, Groves RH, Kruger FJ Rejmánek M, Williamson M, editors. Biological invasions. A global perspective. Chinchester: John Wiley & Sons. 369–388.

Ricciardi A. 2000. Toward a global information system for invasive species. *BioScience*, 50(3) 239-45.

Ridley HN. 1990. The dispersal of plants throughout the world. Koenigstein, W. Germany: Otto Koeltz Science Publishers.

Robertson DJ, Robertson MC, Tague T. 1994. Colonization dynamics of four exotic plants in a Northern Piedmont natural area. *Bulletin of the Torrey Botanical Club*, 121(2), 107-118.

Robinson GR, Quinn JF, Stanton ML. 1995. Invasibility of experimental habitat islands in a California winter annual grassland. *Ecology*, 76(3), 786-94.

Ruan L, Robertson GP. 2013. Initial nitrous oxide, carbon dioxide, and methane costs of converting conservation reserve program grassland to row crops under no-till vs. conventional tillage. *Global Change Biology*, 19(8), 2478-489.

Samson F, Knopf F. 1994. Prairie conservation in North America. *BioScience*, 44(6), 418-21.

Schaefer K. 2015. Wild Parsnip - a weed to watch. Small Farm Sustainability. Available at: http://www.extension.iastate.edu/smallfarms/wild-parsnip-weed-watch.

Schlesinger WH. 1992. Biogeochemistry: An analysis of global change. San Diego, CA: Academic Press.

Scurlock JMO, Hall DO. 1998. The global carbon sink: A grassland perspective. *Global Change Biology*, 4(2), 229-33.

Secchi S, Gassman P, Williams W, Babcock J. 2009. Corn-based ethanol production and environmental quality: A case of Iowa and the conservation reserve program. *Environmental Management*, 44(4), 732-44.

Shah A, Tanveer N, Shahzad M, Yang B, Fahad G, Ali S, Bukhari S, Tung M, Hafeez S, and Souliyanonh A. 2017. Soil compaction effects on soil health and crop productivity: An overview. *Environmental Science and Pollution Research*, 24(11), 10056-10067.

Spyreas G, Wilm B, Plocher W, Ketzner A, Matthews E, Ellis D, Heske M. 2010. Biological consequences of invasion by reed canary grass (*Phalaris Arundinacea*). *Biological Invasions*, 12(5), 1253-267.

Srivastava P, Kumar A, Behera SK, Sharma YK, Singh N. 2012. Soil carbon sequestration: An innovative strategy for reducing atmospheric carbon dioxide concentration. *Biodiversity and Conservation*, 21(5), 1343–1358.

Stein BA, Flack SR. 1996. America’s least wanted: Alien species invasions of U.S. ecosystems. Arlington, Virginia: The Nature Conservancy.

Sternberg M, Brown V, Masters K, Clarke G. 1999. Plant community dynamics in a calcareous grassland under climate change manipulations. *Plant Ecology*, 143(1), 29-37.

Stohlgren TJ, Binkley D, Chong GW, Kalkhan MA, Schell LD, Bull KA, Otsuki Y, Newman G, Bashkin M, Son Y. 1999. Exotic plant species invade hot spots of native plant diversity. *Ecological Monographs*, 69(1), 25–46.

Tamura Y, Moriyama M. 2001. Nonstructural carbohydrate reserves in roots and the ability of temperate perennial grasses to overwinter in early growth stages. *Plant Production Science*, 4(1), 56-61.

Tang J, Jinhua L, Hui L, Fuping L, Baoqian L. 2019. Potential distribution of an invasive pest, *Euplatypus Parallelus*, in China as predicted by maxent. *Pest Management Science*, 75(6), 1630-637.

Tilman D. 1997. Community invasibility, recruitment limitation, and grassland biodiversity. *Ecology,* 78(1), 81-92.

Theoharides KA, Dukes JS. 2007. Plant invasion across space and time: Factors affecting nonindigenous species success during four stages of invasion. *New Phytologist*, 176(2), 256-73.

Torchin ME, Lafferty KD, Dobson AP, Mckenzie VJ, Kuris AM. 2003. Introduced species and their missing parasites. *Nature*, 421(6923), 628-630.

[USDA NRCS] United States Department of Agriculture, Natural Resources Conservation Service. 2006. Plants profile for *Pastinaca sativa* (wild parsnip). [accessed 2020 Jan 11]. https://plants.usda.gov/core/profile?symbol=PASA2

[USDA NRCS] United States Department of Agriculture, Natural Resources Conservation Service. 2019. Web Soil Survey. Available at: https://websoilsurvey.sc.egov.usda.gov/.

Van Kleunen M, Dawson E, Winter P, Kreft W, Kartesz W, Antonova N, Cabezas B, Cárdenas-Toro C, Chacón C, Ebel C, Fuentes F, Henderson G, Kupriyanov I, Meerman M, Moser M, Patzelt N, Baptiste P, Schulze P, Shu S, Velayos T, Pyšek W. 2015. Global exchange and accumulation of non-native plants. *Nature*, 525(7567), 100-3.

Van Kleunen M, Essl F, Pergl J, Brundu G, Carboni M, Dullinger S, Early R, González-Moreno P, Groom QJ, Hulme PE, Kueffer C, Kuhn I, Maguas C, Maurel N, Novoa A, Parepa M, Pysek P, Seebens H, Tanner R, Touza-Montero JM, Verbrugge LNH, Weber E, Dawson W, Weigelt HKP, Winter M, Klonner G, Talluto M. 2018. The changing role of ornamental horticulture in alien plant invasions. *Biological Reviews*, 93(3), 1421–1437.

Vila M, Weber E, Antonio C. 2000. Conservation implications of invasion by plant hybridization. *Biological Invasions*, 2(3), 207-17.

Vitousek PM, Walker LR, Whiteaker LD, Mueller-Dombois D, Matson PA. 1987. Biological invasion by *Myrica Faya* alters ecosystem development in Hawaii. *Science*, 4828: 802-804.

Vitousek PM. 1990. Biological invasions and ecosystem processes: Towards an integration of population biology and ecosystem studies. *Oikos*, 57(1), 7-13.

Vose PB. 1962. Delayed germination in reed canary-grass *Phalaris arundinacea* L. *Annals of Botany*, 26(2), 197-206.

Watson AK. 1985. Integrated management of leafy spurge. In Leafy spurge, ed. No. 3. Champaign, IL: Weed Science Society of America.

Wilcove DS, Rothstein D, Dubow J, Phillips A, Losos E. 1998. Quantifying threats to imperiled species in the United States. *BioScience*, 48(8), 607-15.

Wilson SD, Tilman D. 1993. Plant competition and resource availability in response to disturbance and fertilization. *Ecology*, 74(2), 599-611.

Wolfe LM. 2002. Why alien invaders succeed: Support for the escape-from-enemy hypothesis. *The American Naturalist*, 160(6), 705-11.

Yang Y, Tilman D, Furey G, Lehman C. 2019. Soil carbon sequestration accelerated by restoration of grassland biodiversity. *Nature Communications*, 10(1), 718.

Zeiders KE, Sherwood RT. 1985. Environmental interactions among Reed Canarygrass genotypes for nutritive value, height, and disease severity 1. *Agronomy Journal*, 77(1), 94-98.