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Impacts of riparian buffer vegetation on soil quality physical parameters, 20-23 years after initial riparian buffer establishment

by

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Richard C. Schultz, Major Professor
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DEDICATION

"Essentially, all life depends upon the soil ... There can be no life without soil and no soil without life; they have evolved together." --- Charles E. Kellogg, USDA Yearbook of Agriculture, 1938

I dedicate this thesis to the past: to my grandparents, Everett and Flora Rice, who were early adopters of soil conservation practices on our family farm in northwest Iowa, and to my parents, Tom and Doris Rice, who didn't discourage me from getting dirty growing up, and tolerated my tree climbing and rock collecting.

I also dedicate this thesis to the present: to my husband Jeremy Long, who has stood by me for so many years, and has waited patiently for this thesis to come to fruition.

And finally, I dedicate this thesis to the future: to my daughters, Natalie and Nora Long. May they continue to have a sense of curiosity and wonder about the natural world.

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NOMENCLATURE

BMP	Best management practice
CRP	Conservation Reserve Program; when referring to a specific treatment in this thesis, 'CRP' also refers to the specific 10-year-old CRP planting sampled along Long Dick Creek in Story County, Iowa
CSG	Cool-season grass filter treatment
D_{130}	Tree diameter at 130 cm above the ground surface (commonly referred to as 'diameter at breast height' or 'DBH')
FOR	Naturally-occurring riparian forest treatment
LTG	Long-term grass treatment
MRB	Multi-species riparian buffer
POM	Particulate organic matter
POP	Poplar riparian forest buffer treatment, with cool-season grass understory
SMAF	Soil Management Assessment Framework
SOC	Soil organic carbon
SOM	Soil organic matter
SWG	Switchgrass filter treatment
WLOI	Weight loss-on-ignition

ABSTRACT

Multispecies riparian buffers (MRBs) are a conservation practice that protects water quality and prevents soil erosion by improving soil quality including aggregate stability, particulate organic matter (POM), and water infiltration. USDA Conservation Reserve Program MRB contracts last 10-15 years; evidence shows MRB soil quality may improve within 3-7 years, but there is little data on how MRB soils perform after contracts typically expire.

Slaked soil aggregate stability was measured in a 20-21 year-old MRB in central Iowa at the surface (0-15 cm) and used to calculate the indices mean weight diameter (MWD), geometric mean diameter (GMD), and percent water-stable macroaggregates (%WSA). The MRB contained zones of switchgrass (*Panicum virgatum* L. 'Cave-in-Rock'), hybrid poplar (*Populus spp.*), and cool-season grass; results were compared with an adjacent crop field, a formerly grazed pasture, and a natural riparian forest. Bulk density, total soil carbon, and POM were also measured.

A Cornell sprinkle infiltrometer was used to measure infiltration in a 10-year old tree and cool-season grass MRB, and in the switchgrass, cool-season grass and silver maple zones of the MRB used in the soil aggregate study. MRBs were compared with trafficked and non-trafficked crop interrows in a nearby crop field.

Perennial vegetation had greater MWD, GMD, %WSA, and total SOC compared to the crop field. Data collected in 1997 showed no significant differences in %WSA or MWD between switchgrass and crop field, but did between cool-season grasses and crop field. However, %WSA and MWD under switchgrass increased 45.8% and 120.5%,

respectively, since 1997; under cool-season grasses %WSA and MWD increased 17.9% and 34.3%, respectively, since 1997, but decreased by 37.0% and 35.2% under row crops.

Sixty-minute cumulative infiltration did not significantly differ among MRBs, but was greater than trafficked crop interrows, and was best explained by rainfall rate, bulk density, and initial saturation. Percent rooted vegetation was not a significant factor explaining infiltration in MRBs.

Results suggest MRBs do continue to positively impact selected soil physical parameters. Switchgrass may take longer to improve soil quality parameters in MRBs. Soil quality improvements depend upon the edaphic factors and the amount of disturbance on site.

CHAPTER 1. GENERAL INTRODUCTION

Thesis Organization

This research-based thesis follows the journal paper format. Chapter 1 includes the general introduction to the thesis, a brief review of the literature regarding soil quality, riparian buffers, and the Bear Creek watershed, and the goals and objectives of this study, which then will be followed by two complete manuscripts (Chapters 2 and 3), for which Leigh Ann Long is the primary author, with R.C. Schultz, T.M. Isenhart, and K.S. Hofmockel providing laboratory and field equipment, and assistance with study design and interpretation of results; R.C. Schulz and T.M. Isenhart were the PIs for the grant funding these studies. Chapter 2 explores changes in soil carbon fractions and soil aggregate stability within different riparian buffer vegetation zones over time, modified from a paper to be submitted to the journal *Agriculture, Ecosystems and Environment*. Chapter 3 is a journal article covering the capacity of water to infiltrate soils within different riparian buffer vegetation zones, to be submitted to the *Journal of the American Water Resources Association*. Chapter 4 contains a general discussion and conclusions, and ideas for future work. References for the content of each chapter are given at the end of the individual chapters.

Background

Intensive agriculture can increase soil erosion and transport of nutrients and agricultural chemicals to receiving waters. In the Midwestern USA, a large portion of the

native vegetation has been cleared for intensive agricultural purposes. In the state of Iowa, for example, 86% of the land is considered farmland (USDA Economic Research Service 2013). Of Iowa's native vegetation, 99% of the prairie and wetlands and 80% of the pre-European settlement forests have been converted to other uses (Bishop and van der Valk, 1982, Thomson and Hertel, 1981). While intensive production agriculture has produced the intended benefits of high-quality, low-cost food, feed, fuel in the forms of ethanol and biodiesel, and industrial raw materials, it has also had the unintended and undesirable consequences of increased soil erosion and increased sedimentation and nutrient loading of water bodies.

As native grasslands are tilled and converted to row-crop agriculture, soil organic matter (SOM) is lost. Tillage exposes organic matter-rich topsoil to the erosive forces of wind and water. Additionally, tillage stimulates microbial activity by increasing the oxygen content of the first few inches of the soil surface and placing plant residues in closer contact with moisture and the microbial community, which increases the mineralization of SOM. Tillage also destroys soil aggregates where SOM is sequestered, exposing that SOM to the now-more active microbial population (Six et al. 1998). In Iowa it is estimated the SOM content was reduced by mineralization 10 to 40% due to the cultivation of tall-grass prairies during European settlement in the mid-1800s (Russell et al., 2005). Additionally, annual agricultural crops have less belowground net primary production than native grasslands, and cannot keep pace with SOM losses (Guzman and Al-Kaisi, 2010; Sanford et al., 2012). As SOM is lost, other soil properties associated with SOM, such as soil aggregation and infiltration capacity, also decline.

Soil Quality

Soil properties such as SOM content, aggregation, and infiltration capacity are considered indicators of soil quality (Karlen et al., 1997). Soil quality is a complex concept, and the definition of 'soil quality' depends on the context of the soil and the background of the individuals assigning a definition (Blanco and Lal, 2008). The conceptual definition(s) of soil quality, and methods by which to assess it, are still evolving (Blanco and Lal, 2008), but soil quality can be broadly thought of as based on the capacity of a soil to perform or to function; it is dependent upon the inherent characteristics of the soil, and is relational to the specific function desired for that soil (Karlen et al., 1997). This could mean that a soil which has had its topsoil removed and now has a compacted clay subsurface horizon exposed could be considered to have 'high quality' for building, it would have 'low quality' if that same soil were expected to absorb and store water or produce a high-yielding crop. Also, because soil quality is dependent on the soil's inherent characteristics, expected high soil quality for a Mollisol in the midwestern U.S. will be different than for an Aridisol in the southwestern U.S.

Therefore, Karlen et al. (1997) more specifically defined soil quality as "the capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, to maintain or enhance water and air quality, and support human health and habitation" in order to balance multiple soil uses with goals for environmental quality. Other definitions of soil quality are similar to that written by Karlen et al (1997), and may mention the soil's ability to respond to

management, resist degradation, and produce economic goods and services (Blanco and Lal, 2008).

When examined as part of an ecosystem, soil quality assessments provide an effective method for evaluating direct and indirect environmental impacts of human management decisions (Karlen et al., 1997). Qualitative, holistic, and simple quantitative approaches to assessing soil quality may be more readily adopted by farmers and landowners, while more complex quantitative approaches will be more accepted by the scientific community. The USDA-NRCS Soil Quality Institute has developed a simple quantitative soil quality test kit that can be readily deployed in the field by landowners and natural resource conservation professionals (Sarrantonio et al., 1996; NRCS, 2001). Soil science professionals have used multivariate statistical methods such as principle component analysis to select representative soil quality indicators to create soil quality indices. Multivariate statistical analysis is also useful, as many of the soil properties that contribute to soil quality are highly correlated, and can reveal relationships not recognized when individual soil properties are analyzed separately (Bredja et al., 2000). The Soil Management Assessment Framework (SMAF) proposed by Andrews et al. (2004) is such an index that uses a minimum data set of soil quality indicators, interprets those indicators to provide a score value for each indicator, then integrates those scores into an index value.

Riparian Buffers

Perennial vegetation in riparian areas can act as a living filter to remove sediment and nutrients from uplands within the watershed, and buffer the impacts of that non-source

point pollution on the stream. The potential effectiveness of the living filter is a function of the soil quality within the riparian zone (Schultz et al., 2009).

Natural riparian zones acting as buffers in the USA have been studied, notably in Georgia (Lowrance et al., 1984), North Carolina (Cooper et al., 1987), and Illinois (Schoonover et al., 2005, 2006). Naturally vegetated soils in the flood plain often have high quality because of their depositional nature (Schultz et al., 2009; Ontl et al., 2013). In locations where perennial riparian vegetation has been replaced by agricultural practices, such as central Iowa, constructed buffers can be established by intentionally planting perennial woody and/or herbaceous vegetation between agricultural fields and streams in an attempt to recreate the ecosystem services of the original native vegetation and improve the soil quality. Future references in this thesis to 'riparian buffers' will refer to re-established perennial vegetation in the riparian zone.

Potential benefits of riparian buffer systems include reducing nutrient inputs to receiving waters through plant uptake and microbial interactions, removing sediment from overland flow, slowing flood waters, stabilizing stream banks against erosion, providing habitat for terrestrial and aquatic wildlife, providing harvestable crops, and enhancing recreational opportunities for landowners (Schultz et al., 2009). Riparian buffers also have the potential to re-sequester carbon in the soil (Udawatta and Jose, 2012). Converting land in riparian areas from agricultural uses to riparian buffers can have an additional economic benefit to the landowner, as this land may flood frequently, or may be subject to high water tables, and planting permanent vegetation is an alternative to losing agricultural crops.

Multi-species riparian buffer systems (also called riparian forest buffers) and grass filter strips have been accepted as 'best management practices' (BMPs) by the USDA to

mitigate soil erosion and water quality degradation from agricultural practices (USDA

NRCS, 2010a, 2010b). Landowners wishing to establish such buffers on their property

may qualify for government assistance, both technical and financial, through the USDA's

Conservation Reserve Program (CRP). While the goals of the CRP are mitigating soil

erosion and improving water quality, the USDA NRCS never developed a plan to

quantitatively evaluate whether the CRP accomplished these goals (Karlen et al., 1996).

Re-established riparian buffers are planted in three zones parallel to the stream

channel. The total width of a re-established riparian buffer generally ranges from 20 m

(Schultz et al., 1995) to 55 m (NRCS, 2006), but may be wider. A riparian forest buffer

can have variations on the three-zone design using a mixture of trees, shrubs, native grasses

and forbs, or nonnative cool-season grasses in zones of varying widths to better function

in specific settings and meet landowner objectives (Schultz et al., 2004).

The first zone, immediately adjacent to the stream, consists of four or five rows of

tree species that are suited to the soil moisture conditions present at the site. The

aboveground woody biomass provides a large C and N sink that should be systematically

removed to maintain the nutrient storage capacity of a buffer. Belowground, woody

vegetation stabilizes stream banks by providing root tension strength above the amount of

soil shear (Abernethy and Rutherfurd, 2001), and by increasing evapotranspiration, thereby

reducing the weight of the soil (Waldron and Dakessian, 1982). Commonly planted tree

species in riparian buffer systems in the upper Midwest include hybrid poplar (*Populus*

spp.), silver maple (*Acer saccharinum* L.), green ash (*Fraxnius pennsylvanica* Marsh.),

river birch (*Betula papyrifera*), sycamore (*Platanus occidentalis*), swamp white oak

(*Quercus bicolor*), willows (*Salix spp*.) and black walnut (*Juglans nigra* L.). On sites with

areas of drier soils, red oak (*Quercus rubra*) and bur oak (*Quercus macrocarpa*) can be planted (Schultz et al., 1997, 2009). Non-native cool-season grasses are planted for ground cover between tree rows. If the longevity of the tree component is of importance, then species diversity within the tree component of the buffer is critical; by mixing species within and between rows, the potential of large gaps in the buffer corridor resulting from insect or pathogen problems is reduced (Schultz et al., 2009).

The second zone consists of one to four rows of shrubs. Shrubs develop a perennial root system of intermediate density and depth, add diversity and wildlife habitat to the ecosystem, and help to slow floodwaters and trap flood related debris when the stream leaves its channel. Commonly planted shrub species in the upper Midwest include viburnums (*Viburnum spp.*), dogwoods (*Cornus spp.*), elderberry (*Sambucus canadensis* L.), hazelnut (*Corylus spp.*), wild plum (*Prunus americana*) and ninebark (*Physocarpus opulifolius* L.) (Schultz et al., 1997, 2009).

The third zone, farthest away from the stream channel and immediately adjacent to the agricultural field, is a strip of stiff-stemmed native grasses and forbs, the width of which will vary according to local site conditions (Dosskey et al., 2008), but is usually at least 7 m wide (Schultz et al., 2009; NRCS, 2010a). The stiff stems of the native grasses slow runoff flow, which drops sediment carried by runoff water just outside and within the buffer area. Sediment trapping efficiency generally increases as filter strip width increases, which decreases the field-to-buffer area ratio, but trapping efficiency is also affected by field slope, type of cultivation, rainfall amount and intensity, antecedent soil moisture, and field soil texture (Dosskey et al., 2008).

In addition, native grasses produce an extensive and deep root system, much of which is replaced annually, providing large amounts of organic matter to the soil. Forbs diversify the grass filter strip and provide food sources to support increased wildlife diversity. Warm-season grasses and forbs commonly planted in grass filter strips and as a part of riparian forest buffers in the upper Midwest include big bluestem (*Andropogon gerardii*), switchgrass (*Panicum virgatum* L.), Indiangrass (*Sorghastrum nutans*), purple coneflower [*Echinacea purpurea* (L.) Moench], and black-eyed Susan (*Rudbeckia hirta* L.) (Schultz et al., 2009).

All three zones work best with native plant species as they are more adapted to local pests and diseases and are more suited to the local wildlife. Buffer practices outlined here were developed for the Midwestern USA. Since soils, climate, plants, and agricultural practices are location specific, the plant species listed here may not be ideally suited to the conditions of another region, but analogous species could be substituted in this design. Riparian buffers, along with other conservation practices, should be established with careful consideration of landscape, capital investments, income, and environmental benefits. (Udawatta and Godsey, 2010).

The Bear Creek Watershed

One of the oldest and largest series of re-established riparian buffers exists in the Bear Creek watershed in central Iowa, USA (42◦ 11′ N, 93◦ 30′ W) (Schultz et al., 1995). Bear Creek is a third-order stream which is 34.8 km long; it has 27.8 km of major tributaries

that are primarily fed from artificial subsurface drainage outlets. Average discharge rates vary between 0.3-1.4 $m^3 sec^{-1}$ (Simpkins et al., 2002).

The Bear Creek watershed is 7,661 ha in size and lies entirely within the Des Moines Lobe, which is the depositional remnant of the late Wisconsin glaciation that advanced into Iowa approximately 14,000 years ago (Simpkins et al., 2002). The landscape is flat to undulating, with pothole wetlands. Prairie vegetation originally dominated the uplands, with the exception of forests that occurred along the lower end of the stream near its confluence with the South Skunk River (DeWitt, 1984). About 87% of the watershed is now used for intensive agriculture (Simpkins et al., 2002); agricultural activities by European settlers in the area began approximately 150 years ago. Soils within the watershed are well- to poorly-drained, and were formed in glacial till, local alluvium, or colluvium derived from till (DeWitt 1984).

Sixteen km of riparian buffer systems have been established on former agricultural soils on private land along the middle third of the stream reach since 1990, with the assistance of researchers at Iowa State University and the USDA Natural Resource Conservation Service. Riparian buffer systems were planted in 1990, 1994, 1995, 1997, 1998, 2000, and 2001 on sites that either had intensive row crop agriculture, generally a corn (*Zea mays* L.)-soybean (*Glycine max*) rotation, or intensive livestock grazing primarily by beef cattle on introduced cool-season grasses, down to the stream edge. These riparian buffers now range in age from 13-24 years. In 1998, The Bear Creek watershed was designated as a National Restoration Demonstration Watershed by the United States Environmental Protection Agency.

Design of the 24-year-old (1990) riparian buffer along a 1,000 m reach of Bear Creek and where much of the research described in chapters 2 and 3 of this thesis took place, is discussed in detail in Schultz et al. (1995), where trees were segregated into single-species blocks. A portion of the third zone of the 24-year-old riparian buffer was a monoculture planting of switchgrass, which has since been invaded by non-native cool-season grasses, primarily smooth bromegrass (*Bromus inermis*). Another portion of zone 3 was planted to non-native cool-season grasses, dominated by smooth bromegrass. Other areas of zone 3 in the 24-year-old buffer were planted to a mixture of native warm-season grass species. Plantings in 1994 and later at Bear Creek were not planted as monocultures, as research became more focused on overall ecosystem services and not as much on the capacity of the trees to eventually provide a biomass crop.

These riparian buffers have been extensively studied to quantify the effects of the re-established perennial riparian vegetation on above- and belowground biomass (Tufekciolgu et al., 1999, 2003), water infiltration into soil (Bharati et al., 2002), soil aggregation and carbon storage (Marquez et al., 1999, 2004), soil microbial biomass and respiration (Pickle, 1999; Tufekcioglu et al., 2001; Dornbush et al., 2008), greenhouse gas emissions (Kim et al., 2010), removal of nutrients and sediment from surface runoff (Lee et al., 1999, 2003), surface and groundwater hydrology and nutrient removal (Simpkins et al., 2002; Spear, 2003; Kult, 2008), and stream bank erosion (Zaimes et al., 2004).

Thesis Goals and Objectives

A definition of soil quality used in the context of the riparian buffer research within the Bear Creek watershed is 'the ability of a soil to sustain an active ecosystem that provides many important functions for reducing non-point source pollutants' (Raich et al., 2001). Based upon this definition, this thesis explores the following soil quality indicators in the surface soils of the various vegetation treatments in the 24-year-old riparian buffer and adjacent crop fields to 35 cm, and builds upon previous research conducted within the Bear Creek watershed as cited: (1) soil water infiltration (Bharati et al., 2002), (2) total soil organic matter and carbon (Pickle, 1999; Marquez et al., 1999, Kim et al., 2010), (3) soil particulate organic matter and carbon (Marquez et al., 1999), and (4) water-stable aggregates (Marquez et al., 2004).

Most of the original research cited was performed 6-8 years after the establishment of perennial vegetation, with the exception of the work done by Kim et al. (2010), which was performed 17 years after establishment. The work carried out in this thesis was performed 20-23 years after riparian buffer vegetation establishment. It is hypothesized that these soil quality parameters as well as the ecosystem services driven by these parameters, primarily the capacity to reduce non-point source water pollution, should continue to improve with the greater biodiversity and age of the vegetation.

A secondary goal of this research is to use the Soil Management Assessment Framework or another suitable soil quality index to evaluate the impact that riparian buffers may have had on soil quality in order to justify the return on investment of having riparian buffer land enrolled in CRP for the last 23 years. Most CRP contracts are 10-15 years long;

however, most carbon sequestration and/or soil quality studies on CRP land are performed either <10 years since CRP establishment (Gebhart et al., 1994), or is from upland sites, not riparian buffers or floodplain soils (Karlen et al., 1996; Kucharik, 2007). Soil quality indicator data from this long-term riparian buffer study should help to bridge this knowledge gap.

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CHAPTER 2. SOIL ORGANIC MATTER FRACTIONS AND AGGREGATE STABILITY UNDER RE-ESTABLISHED MULTI-SPECIES RIPARIAN BUFFER (MRB) VEGETATION

Modified from a paper to be submitted to *Agriculture, Ecosystems and Environment.*

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Abstract

Multispecies riparian buffers (MRBs) have been designed as a conservation practice that protects water quality and prevents soil erosion by improving soil quality. Contracts for MRB establishment under the USDA Conservation Reserve Program (CRP) are 10-15 years in length; there is evidence that soil quality improves under MRB vegetation within a few years, but may not continue to improve as time passes. In this study, the soil quality parameters of particulate organic matter (POM) and the slaked soil aggregate size-stability indices of mean weight diameter (MWD), geometric mean diameter (GMD), and percent water-stable macroaggregates (%WSA) were measured under switchgrass (*Panicum virgatum* L. 'Cave-in-Rock'), hybrid poplar (*Populus spp.*), and cool-season grass vegetation types within a 20-year-old MRB in central Iowa and contrasted with an adjacent crop field, a naturally occurring riparian forest, and a formerly

grazed pasture. All MRB vegetation types, the natural riparian forest, and the former pasture had greater MWD, GMD, %WSA, and total POM, and lower bulk density compared to the crop field. %WSA and MWD in the switchgrass increased 45.8% and 120.5%, respectively, in the past 14 years since it was last measured; under cool-season grasses %WSA and MWD increased 17.9% and 34.3%, respectively, but decreased by 37.0% and 35.2% in the crop field. Previous data collected in 1997 did not show significant differences in %WSA or MWD between the switchgrass and the crop field, but did between the cool-season grasses and crop field. These results suggest that native C_4 grasses such as switchgrass do take longer than C_3 grasses to improve soil quality parameters in MRBs, but improvements in soil quality are also dependent on the edaphic factors present on site.

Abbreviations: CRP, (USDA) Conservation Reserve Program; GMD, geometric mean diameter; MRB, multi-species riparian buffer; MWD, mean weight diameter; POM, particulate organic matter; SOC, soil organic carbon; SOM, soil organic matter; %WSA, percent water-stable aggregates (>0.25 mm).

Introduction

Multispecies riparian buffers (MRBs) are strips of any combination of trees, shrubs, and/or grasses planted parallel to stream channels, and are accepted best management practices for mitigating water quality issues from agricultural activities in upland areas (Schultz et al., 1995; Lowrance et al., 2002). Perennial vegetation in the riparian zone can improve several parameters related to soil quality, such as soil organic matter (SOM) and

particulate organic matter (POM) (Marquez et al., 1999; Tufekcioglu et al., 2003; Ontl et al., 2014) and soil aggregate stability (Marquez et al., 2004; Ontl et al., 2014).

SOM, especially biologically-active soil organic matter such as POM, is considered an important soil quality indicator variable because of its ability to stabilize soil particles, which minimizes erosion, increases infiltration and water-holding capacity, and reduces the negative environmental effects of pollutants. SOM will eventually be converted to more recalcitrant forms, which can contribute to carbon sequestration, an important response to human-induced climate change (Schlesinger and Andrews, 2000). However, changes in total SOM can be difficult to quantify in the short term (Cambardella et al., 2003; Ontl et al., 2014).

POM, defined as that part of SOM greater than 0.050 mm in size, has been identified as the organic matter fraction most sensitive to changes in soil management practices (Cambardella and Elliott, 1992). Changes within the POM pool can be identified even within the growing season (Marquez et al., 1999); the half-life of POM can range from days to a few years or decades, depending on whether or not the POM is protected within a soil aggregate (Wander, 2004).

Returning perennial vegetation to former agricultural fields is known to have positive impacts on SOM and SOC accrual; however, there is conflicting evidence as to how much SOC can accumulate after restoration. A centuries-long time frame may be necessary for total SOC concentrations to return to undisturbed levels (Jastrow, 1996). Also, controls on the rate and duration of C accrual among aggregate size classes, and within aggregates in the POM, silt, and clay pools (O'Brien and Jastrow, 2013) are not well understood (West and Six, 2007) and make total SOC accrual research challenging.

However, some changes in total SOC may be seen initially; Marquez (1999) noted that after five growing seasons, SOC in the top 35 cm of soil in a MRB had increased 8.5% under poplar and C_3 grasses, 3.2% under C_3 grasses alone, and 8.6% under switchgrass (a C_4 grass); POM C was also significantly greater than that found in the adjacent agricultural soil. McLauchlan et al. (2006) showed that in CRP fields in western Minnesota, the top 10 cm of soil accumulated SOC at a constant rate of 62.0 $g \cdot m^{-2} yr^{-1}$, regardless of whether the vegetation type was dominated by C_3 or C_4 grasses.

On the other hand, a study by Kucharik (2007) noted that rates of average SOC accumulation declined over time on a limited number of minimally managed CRP grasslands containing native and introduced C_3 and C_4 species on a variety of landscape positions on Mollisols in southern Wisconsin 16 years after establishment; CRP 4-5 years old had 0-5 cm SOC accrual rates of 79.7 g C m^{-2} y^{-1}, yet were not significantly different (LSD 0.05) than adjacent crop field soils; 10-16 year old CRP accrual rates were 24.7 g C m^{-2} y^{-1}, but were significantly different (LSD 0.05) compared with adjacent crop soils. Jastrow (1996) modeled that it would take 384 years to reach 99% equilibrium of native prairie SOC levels. It is entirely possible that soils may reach a steady-state saturation potential with respect to carbon (Stewart et al., 2007), or serial transient steady-states as the best-protected pools boost SOC accumulation into other less-protected pools (O'Brien and Jastrow, 2013).

Schlesinger and Andrews (2000) remark that soil carbon accumulation is typically driven by limits on litter decomposers (e.g., litter C:N ratio, temperature, and moisture), rather than large carbon inputs. Subsequently, Ontl (2013) used structural equation modeling of C cycling processes to reveal that existing soil properties which provide

habitat for litter decomposers indeed are the main drivers for change in soil C pools at the landscape scale rather than changes in root inputs and root-associated microbes. Floodplain soils, where MRBs are established, are particulary suited to accumluation of SOM if the soils contain clayey texture (Burke et al., 1989).

Soil aggregates are groups of primary soil particles (sand, silt, and clay) and organic matter that cohere more strongly to each other than to other surrounding particles. The means by which cohesion occurs may be chemical, biological, or physical in nature. Gaps between and within aggregates provide porosity in the soil necessary for water and air movement, as well as allowing for water and carbon storage, plant root growth, and habitat for soil flora and fauna. Quantification of soil aggregates is commonly done by physical methods such as sieving. The hierarchy of aggregate size fractions are usually classified as such: large macroaggregates are greater than 2 mm in diameter, small macroaggregates range from 0.25 to 2 mm in diameter, and microaggregates range in size from 0.053-0.25 mm in diameter. These size fractions exist among and within each other in the soil matrix. Larger macroaggregates are composed of smaller macroaggregates, which are in turn formed by microaggregates (Tisdall and Oades, 1982). Generally, macroaggregates are important in water filtration, root penetration, and soil aeration, while microaggregates are important for water holding capacity and long-term carbon storage.

At each level in the aggregate hierarchy, different mechanisms are more responsible for binding together the subunits; biochemical processes play a larger role in forming macroaggregates than inorganic chemical or physical processes. There is a two-way interaction between soil aggregates and POM and SOM; aggregates protect POM and SOM, but POM and SOM also binds soil particles together into aggregates and stabilize

them (Waters and Oades, 1991). Macroaggregate formation and stabilization in particular, is dependent on SOM content in moderately weathered soils (Waters and Oades, 1991; Six et al., 2000; Boix-Fayos et al., 2001). Physical and chemical processes play a larger role in forming small aggregates and in forming aggregates in soils with higher clay and oxide contents (Oades and Waters, 1991), and therefore fine-textured soils, as a general rule, have more stable aggregates (Boix-Fayos et al., 2001) and these smaller aggregates are more stable than larger ones (Tisdall and Oades, 1982).

Soil disturbance, notably tillage, breaks apart macroaggregates directly and can indirectly lead to aggregation destruction by exposing new soil to wet-dry and freeze-thaw cycles at the soil surface. It also allows organic matter that was physically protected to be exposed to new environments (i.e. temperature, moisture, and aeration) and communities of organisms (Six et al. 1998). However, by replanting perennial vegetation on formerly cultivated soils, such as when pasture, perennial bioenergy crops, or CRP practices such as MRBs are established, positive changes to soil aggregation occur in a short amount of time. In three years' time, Ontl (2013) found that soil aggregation changes were positive under a switchgrass monoculture. Jastrow (1996), using a combination of measurements across a prairie reconstruction chronosequence in Illinois and an exponential model, found that soil macroaggregates can reach 99% of equilibrium at 10.5 years after restoration in silt loam and silty clay loam soils. Guzman's study (2008) agrees with these findings; he found that across a chronosequence of large-scale native prairie restorations in central Iowa (Neal Smith NWR), the two most recent restoration sites (< 10 years) had significantly lower aggregate mean weight diameter (MWD) values compared to the longer established prairie restoration (>10 years) and nearby remnant prairie sites, and that the rate of increase in

aggregate MWD is much greater during the early years of prairie establishment. Therefore, macroaggregates could be expected to effectively reveal short-term responses of SOC dynamics to land management practices and duration of restoration (Puget et al., 2000). Given that CRP contracts for MRBs are 10-15 years in length (USDA-FSA, 2014) and may be reverted to cropland after that period of time, it is important to quantify whether soil aggregation and SOC accrual rates have reached their maximum potential in this amount of time.

Twenty to twenty-one years after establishment of a MRB system, we hypothesize that the rates of SOC accumulation will have declined within the MRB soils, while still observing increases in the total amount of SOC, with the greatest increases within the warm-season grass and woody zones of the MRB. We also hypothesize that the amount of POM, amount of macroaggregates, and total aggregate stability may have increased slightly from 7-8 years post-establishment (Marquez et al., 1999, 2004) to 20-21 years post-establishment, but also again, that the rate of gain will have declined.

The objectives of the study were: (i) to follow up on research presented by Marquez et al. (1999, 2004) and determine whether aggregate stability and total POM have increased, decreased, or remained constant in the last 15 years under perennial MRB vegetation, (ii) to determine the amount of POM associated with different-sized stable aggregates.

Materials and Methods

Site description

This study took place in 2010-2011 in perennial riparian plant communities and adjacent row crop fields located on private property in the Bear Creek watershed in central Iowa (Figure 2.1 and 2.2). Most of the perennial riparian plant communities are examples of standard practice conservation grass filters (USDA-NRCS, 1997) and reestablished riparian forest buffers planted in 1990 into areas which had previously been under long-term (> 80 years) row-crop agriculture (Schultz et al., 1995). Portions of these areas were also the focus of previous soil aggregation and POM research in 1996-1998, six to eight years post-buffer establishment (Marquez et al., 1999; 2004). The 20-year-old plant communities studied were: a cool-season grass filter (CSG) dominated by smooth brome (*Bromus inermis*), reed canarygrass (*Phalaris arundinacea*), and Kentucky bluegrass (*Poa pratensis* L.); a switchgrass (*Panicum virgatum* L. 'Cave-in-Rock') filter (SWG), heavily invaded by smooth brome in the 13 years since the Marquez et al., 2004 study; and a hybrid poplar (POP) riparian forest buffer (*Populus spp.*) with a cool-season grass understory dominated by smooth brome. An area east of the 20-year old MRB was also sampled, denoted as long-term grass (LTG); this area has been in grass since at least 1930 (Iowa State University Geographic Information Systems (GIS) Support and Research Facility, 1999) and used previously as a continuously grazed pasture until 1989, then was tilled once in 1994 and planted with a mixture of native warm-season grasses, dominated by big bluestem (*Andropogon gerardii)* and Indiangrass (*Sorghastrum nutans)*, with some encroachment by non-native cool-season species (primarily Kentucky bluegrass). The

adjacent row crop fields to the north (N. Crop) and the south (S. Crop) of the MRB are planted to a corn (*Zea mays* L.)-soybean (*Glycine max*) rotation; corn in even-numbered years, soybeans in odd-number years, and are chisel-plowed in the fall and spring. A naturally-occurring riparian forest (FOR) > 80 years old (Iowa State University Geographic Information Systems Support and Research Facility, 2014) and dominated by a silver maple (*Acer saccharinum*) overstory, 4.7 km downstream from the Bear Creek MRB, was also studied.

Soil sampling and sample preparation

Three 150 m^2 replicate soil sampling plots were established within each vegetation type on the alluvial floodplain; a 1 m border around each replicate plot was established and not sampled to minimize edge effects. Table 2.1 summarizes the parent material, drainage class, and U.S. Soil Taxonomy for each soil series used in this study.

Plots were established in areas mapped primarily as Coland (Cumulic Endoaquoll) clay loam, 0-2% slope, formed in alluvium. Other soils present within the study are Cylinder (Aquic Hapludoll) loam, 0-2% slope, formed in loamy sediments over sand and gravel, present in the north row crop field; and Clarion (Typic Hapludoll) loam, 2-6% slope, formed in glacial till; and Webster (Typic Endoaquoll) clay loam, 0-2% slope, formed in glacial till or local alluvium derived from till in the SWG and south row crop fields. The soils of the FOR plots are mapped as a Spillville-Coland complex. All soil series within this study are Mollisols in the mesic precipitation regime, are of mixed mineralogy and are superactive (DeWitt, 1984).

Soil samples were collected in November 2010 all plots excepting the LTG and FOR plots to a depth of 15 cm, using a 3.175 cm diameter push probe. Five cores per plot were randomly sampled and composited to create one sample per plot. All plots, including the LTG and FOR plots, were resampled in May, July, and November 2011. Samples were placed in sealed plastic bags and kept cool until transported to the laboratory, where they were held at 4°C until further processing could occur. In the laboratory, the total mass of the soil sample was recorded and passed through an 8-mm sieve. Plant fragments larger than 8 mm were discarded; large woody roots and rocks which did not pass the 8-mm sieve were weighed and subtracted from the total sample. A portion of the 8-mm sieved sample was set aside to air-dry, while a second portion of the 8-mm sieved sample was passed through a 2-mm sieve before being air-dried.

Soil analysis

Soil characteristics determined were bulk density, particle size distribution, field soil moisture, slaked aggregate stability, and total POM. Bulk density was estimated using the oven-dried soil mass and the volume of the field sample (Grossman and Reinsch, 2002). Field soil moisture was determined gravimetrically on a 15-30 g 8-mm sieved subsample. Particle size distribution (sand, silt, and clay) was measured on the 2-mm air dry fraction using the pipette method accompanied by organic matter destruction with 30% H_2O_2 (Gee and Bauder, 1986). Slaked aggregate stability was determined on a 100 g subsample of the 8-mm sieved air-dry soil by wet-sieving following the protocol reported by Marquez et al. (2004). Aggregates were physically separated into four size fractions:

(i)   large macroaggregates greater than 2 mm in diameter,

(ii)   small macroaggregates between 0.25-2 mm in diameter,

(iii)   microaggregates between 0.053-0.25 mm in diameter, and

(iv)   the mineral fraction (silt+clay) less than 0.053 mm in diameter.

After wet sieving, all fractions were dried at 65°C and weighed. To correct for the influence of sand on the mass of each aggregate-size fraction (Marquez et al., 2004), each aggregate-size fraction was thoroughly mixed, and a subsample of the aggregates (approximately 5-10 g) was quantitatively removed. This subsample was dispersed in 50 mL 5 g L^{-1} sodium metaphosphate solution, shaken overnight on a reciprocal shaker, passed through a 0.053 mm sieve and rinsed until the rinsate was clear. Sand and POM retained by the 0.053 mm sieve was backwashed into an aluminum weighing dish, dried at 65°C overnight, and weighed. Aggregate-size fractions were corrected for sand content by subtracting the sand fraction from the total dry mass of the whole aggregate fraction. Water-stable macroaggregate (> 250 mm) percentage (WSA%) (USDA-ARS and NRCS, 2001) was calculated with equation 2.1:

$$WSA\ (\%) = \left(\frac{(weight\ of\ dry\ aggregates-sand)}{(weight\ of\ dry\ soil-sand)}\right) * 100 \qquad [2.1]$$

WSA% assumes that large macroaggregates (>2 mm) are not fragmented into small macroaggregates (0.25-2 mm) during the sieving process (Marquez et al., 2004).

Geometric mean diameter (GMD) (Kemper and Rosenau, 1986) of the slaked aggregates was also determined, using equation 2.2:

$$GMD = \exp \left[\sum_{i=1}^{n} w_i \, ln\overline{x}_i / \sum_{i=1}^{n} w_i \right] \qquad [2.2]$$

where $\overline{x}_i$ is the mean particle diameter (mm) of size fraction i, w_i is the proportion of the whole soil in the given fraction i corrected for sand content, and n is the number of sieves used. The GMD was used as an index of aggregate size distribution rather than the mean weight diameter, as aggregate size distribution in most soils are approximately log-normal (Kemper and Rosenau, 1986), and use of the MWD as an index is questionable with the aggregate-size distribution is nonsymmetrical (Six et al., 2000b). However, MWD was calculated in order to make comparisons from the Marquez et al. (2004) study from these same soils. Mean weight diameter is calculated using formula 2.3 (Kemper and Rosenau, 1986).

$$MWD = \sum_{i=1}^{n} \overline{x}_i \, w_i \qquad [2.3]$$

Total (free + intra-aggregate) POM associated with each aggregate-size fraction was determined by first removing and discarding plant fragments (> 2 mm) found in the dried sand+POM from each aggregate-size fraction subsample; these plant fragments are not identified as POM or SOM (Cambardella et al., 2001). The dried sand+POM fraction was placed in a muffle furnace for 4 hours at 450°C to determine total POM via the weight loss-on-ignition (WLOI) method for each aggregate size fraction and summed to estimate total POM in the soil (Cambardella et al., 2001). Total POM associated with each aggregate fraction was calculated using the following equation:

$$POM \left(\frac{mg}{g\ soil}\right) = \frac{[(Sand+POM\ weight\ at\ 65°C)-(Sand+POM\ weight\ at\ 450°C]*1000}{weight\ of\ aggregate\ fraction\ at\ 55°C} \qquad [2.4]$$

Total POM was also expressed as a fraction of the whole soil as follows:

$$POM\left(\frac{mg}{g\ soil}\right) * P_{(aggregate\ fraction)} \qquad [2.5]$$

where $P_{(aggregate\ fraction)}$ is the proportion of whole soil mass comprised in the aggregate fraction of interest.

Statistical analyses

Data within treatments were initially tested for sampling date effects with one-way analysis of variance (ANOVA). When no significant differences between sampling dates were found, data were pooled across all sampling dates and tested with ANOVA, using Tukey-Kramer HSD *post-hoc* tests to compare means between treatments. Results were considered significant if $p \leq 0.05$. All statistics were performed using the JMP Pro v. 11.0.0 statistical software package (SAS Institute, Inc., Cary, NC, USA).

Results and Discussion

Texture

Texture data were pooled for all sampling dates (Table 2.2). Although efforts were made to site the plots on similar soils, clay percentage was significantly higher in the poplar ($p \leq 0.0138$) than any of the other vegetation types, with the exception of the cool-season

grass ($p = 0.0544$). Cool-season grass, long-term grass, and switchgrass had similar clay

contents, which were significantly higher than the clay contents in the north and south crop

fields and the naturally-occurring riparian forest. This may be due either to the inclusions

of soil series with smaller clay contents in the crop fields, or due to losses of clay from

erosion during the long history of tillage in the crop field. The lesser amount of clay present

in the forest is due to the complex combination of alluvial sandy sediments mixed with

clayey alluvium in the Spillville-Coland soil complex present.

Bulk density

One-way ANOVA tests showed that bulk density did not vary among sampling

dates, so data were pooled for analysis (Table 2.2). Bulk density was highest in both crop

fields and was significantly higher ($p < 0.0001$) than any of the other MRB vegetation

types, as expected. Tillage destroys soil structure and allows soil particles to pack tighter

together. The MRB vegetation types had bulk densities in the order of switchgrass > forest

$\geq$ poplar $\geq$ cool-season grass > warm-season grass. The switchgrass vegetation plot is only

7 m wide and has received heavy foot traffic during the 20 years of establishment, although

pains were taken to not sample within obvious traffic paths. This may explain the

significant differences between the bulk density in the switchgrass plot and the other MRB

vegetation types. The poplar and cool-season grass have received moderate amounts of

foot traffic on a monthly basis during the last twenty years, while the warm-season grass

plot is seldom visited.

Soil carbon and nitrogen

Table 2.3 presents the total soil carbon and nitrogen present in the surface 0-15 cm. The hybrid poplar has significantly higher total soil carbon ($p < 0.0245$) and total soil nitrogen ($p < 0.0016$) than any of the other treatments. Cool-season grass, long-term grass, forest, and switchgrass were not significantly different than one another in total carbon or nitrogen content, but all had significantly more total carbon and nitrogen than the row crop treatments ($p < 0.0001$).

Aggregate size and stability indices

No significant differences between sampling dates within vegetation types was seen, so aggregate data were pooled for analysis. Regardless of whether GMD or MWD is used as an index of slaked aggregate size stability distribution (Table 2.4), the order of aggregate size is cool-season grasses > warm-season grasses ≥ poplar = switchgrass ≥ forest > N. Crop = S. Crop.

When %WSA is used as an index of aggregate stability, the order of slaked water-stable macroaggregates is cool-season grasses ≥ poplar = warm-season grasses ≥ switchgrass = forest > N. Crop = S. Crop (Table 2.5).

There are clear differences in the amount of large macroaggregates (LM, >2 mm) and small macroaggregates (SM, 0.25-2mm) (Figure 2.3). The cool-season grasses have the largest percentage of LM (45.0%); this is significantly different ($p < 0.0001$) than the percentage contained within the warm-season grasses (34.8%). The percentage of LM in the switchgrass (26%) is neither significantly less than that in the warm-season grass nor

greater than that in the poplar (24.7%). Poplar and forest (21.5%) LM are significantly larger (p < 0.0001) than the row crop soils (S. Crop, 1.3%; N. Crop, 1.0%).

The small macroaggregate (0.25-2 mm) mean percentages contain more overlap (Table 2.5). Overall, SM dominate the balance of macroaggregates in the row crop soils, and outrank LM in the forest, poplar, and switchgrass soils by 2.3:1, 2.1:1, and 1.7:1, respectively. SM are approximately equally balanced with LM in the warm-season grass soils, and occur in a ratio of 0.8 to 1 with the LM in the cool-season grass soils.

When compared with the previously collected data by Marquez et al. (2004), the general trends for %WSA (Figure 2.4) are slight increases in the naturally-occurring forest (16.1%) and cool-season grasses (17.9%), moderate increases in the switchgrass (45.8%), and moderate decreases in the row crop soils (-37.0%). Increases in MWD (Figure 2.5) were seen in the forest (7.5%), cool-season grass (34.3%), and switchgrass (120.5%) soils, and decreases in MWD were seen in the row crop soils (-35.2%). The row crop soils continue to receive disturbance at least twice annually, which destroys any gains in aggregate size or stability accrued during the 6 month period of no disturbance. The forest has existed for > 80 years with minimal disturbance, and aggregate turnover rates are expected to be relatively stable by this time.

Marquez et al (2004) noted that the %WSA and MWD under seven years of switchgrass growth was not significantly different than that found under the row crop soils; however, after 20-21 years of perennial vegetation, a significant increase in both %WSA and MWD can be seen (Tables 2.3 and 2.4). The significant gains in MWD and %WSA under the switchgrass soils could be due to the switch in vegetation type from C_4 to C_3 grasses. As previously noted, the switchgrass filter has been heavily invaded by smooth

brome since the late 1990's (Figure 2.6). Increases in water-stable large macroaggregates under the cool-season grasses between establishment in 1990 and sampling in 1997 were thought to be as a result of greater fine root inputs and decreased C:N ratios of the root inputs of the C_3 grasses (Tufekcioglu et al., 2003) and increased microbial biomass (Pickle, 1999) in the soils under the C_3 grasses compared with the smaller fine root inputs and decreased C:N ratios in the root inputs by the switchgrass, a C_4 species, along with lower microbial biomass in the soils under the switchgrass.

The long-term grass plots, that now contain a mixture of non-native C3 grasses and reintroduced C4 grasses as of 1994, although not sampled in 1997, now have a larger %WSA (Table 2.4) and aggregates of larger MWD (Table 2.5) than the current switchgrass plots and larger than the 1997 cool-season grasses, but not nearly as large as the current aggregates under the cool-season grasses. If the increase in aggregate size and stability is driven primarily by SOC content, then these results agrees with a study undertaken by Corre et al (1999), who noted that it took 16-18 years for C_4 grasses to regain the same levels of SOC as found under C_3 grasses originally occurring in the same location. However, when looking at the SOC content, there is no significant difference between the long-term grass, switchgrass, and cool-season grass plots (Table 2.3). Kim et al. (2010) found this also when sampling in some of these plots in 2006-2007; this suggests that total SOC content in the grass portion of MRBs may be nearing a steady-state or transient steady-state at this time. Total POM (Figure 2.8) was greatest under long-term grass, which may be driving the aggregate dynamics more than total SOC.

Total POM associated with aggregate size fractions

There were no clear significant differences in total (free+intraggregate) POM between sample dates within vegetation type, with the exception of the N. Crop field having greater total POM in July 2011 than in either Nov. 2010 ($p < 0.0326$) or Nov. 2011 ($p < 0.0410$), and the cool-season grasses having greater total POM in Nov. 2011 than in Nov. 2010 ($p < 0.0499$). In spite of these few significant differences, all data were pooled for analysis. Total (free + intra-aggregate) POM on a mg POM g^{-1} soil basis (Figure 2.7) ranked in order from greatest to least: long-term grass = cool-season grass ≥ forest ≥ switchgrass = poplar > S. Crop = N. Crop. On an aggregate proportion basis (Figure 2.8), total POM ranked in order from greatest to least: cool-season grass ≥ long-term grass = forest ≥ poplar ≥ switchgrass ≥ S. Crop > N. Crop.

There were no significant differences in total LM POM concentration (mg POM g^{-1} soil) between any of the vegetation types, but trends do show all MRB vegetation having greater LM-associated POM than the row crop fields (Figure 2.7). However, when expressed as a proportion of aggregates isolated from each vegetation type (Figure 2.8), the total amounts of LM POM are significantly different in the order of cool-season grasses ≥ long-term grasses = poplar = switchgrass = forest ≥ S. Crop ≥ N. Crop. Aggregate size fractions can harbor different microbial communities and microbial activities. Bach and Hofmockel (2014) found that wet-sieved macroaggregates (>1 mm) had greater aggregate potential enzyme activity than smaller aggregate fractions, particularly for the C-cycling enzymes cellobiohydrolase and β-glucosidase. Macroaggregates found under corn had higher potential enzyme activity for these same enzymes than the aggregates found under reestablished upland prairie, suggesting that macroaggregates under corn would not be as

rich in labile C content. However, we did not find clear evidence of this in LM aggregates in the Bear Creek MRB on a mg POM g^{-1} soil basis.

Total POM concentration (mg POM g^{-1} soil) associated with the SM fraction ranked in greatest to least in the order of warm-season grasses > cool-season grasses ≥ forest ≥ poplar ≥ switchgrass ≥ S. Crop ≥ N. Crop (Figure 2.7). When weighted by proportion of SM aggregates, SM POM ranked in order of warm-season grasses ≥ cool-season grass ≥ forest ≥ poplar ≥ switchgrass ≥ S. Crop ≥ N. Crop (Figure 2.8). Although SM POM under poplar is less concentrated, poplar have the most SM of any of the vegetation types, thus more POM is present in the SM fraction. Because SM dominate the macroaggregate fraction under row crops (Figure 2.2), much of the POM present in these row crop soils is contained in the SM fraction, indicating SM are important aggregate size fractions for retaining POM in cultivated soils. This agrees with the results of Ontl et al. (2015), which found under continous corn grown on soils, intraggregate SM POM was the largest POM fraction.

Total POM concentration associated with the microaggregate fraction (mg POM g^{-1} soil) ranked from greatest to least in the order of cool-season grasses ≥ forest ≥ poplar = warm-season grasses ≥ switchgrass = N. Crop = S. Crop (Figure 2.7). When weighted by the proportion of microaggregates, micro POM ranked from greatest to least in the order of cool-season grasses ≥ forest ≥ mixed grasses ≥ N. Crop ≥ poplar = S. Crop = switchgrass (Figure 2.8). POM associated with microaggregates may be more stable, which may explain why few differences were seen in the POM associated with the microaggregate size class on a mg POM g^{-1} soil basis. We did see a large amount of free+intra-aggregate POM under the natural riparian forest, indicating time may be playing a role in the amount of

micro POM present, as there are not significantly more microaggregates present under the natural forest vegetation. Microaggregate POM is least in the the switchgrass (0.085 g POM kg^{-1} soil); POM may still be transitioning from larger sizes to microaggregate size due to higher C:N ratios present in the POM and the higher amount of coarse roots present under switchgrass (Marquez, 2001). Data published by Ontl et al. (2015), which showed regardless of annual (corn or triticale/sorghum) or perennial cropping system (3-year-old switchgrass), there were no differences in the amounts of intra-microaggregate or fine-free POM isolated from the microaggregate sized fraction; it is possible that changes occur in the microaggregate-associated POM over an decadal time frame, as supported by our findings.

Further exploration of the LOI method showed high coefficients of variation for the data, indicating that the LOI method is not sensitive enough to changes in POM between aggregate size classes, and may overestimate the POM-C content (Pribyl, 2010). A more quantitative, but time-consuming method, to isolate POM and quantify POM-C from aggregates would be to use a density fractionation method similar to that used by Cambardella and Elliott (1992) and Ontl et al. (2015).

Conclusions

Any perennial vegetation present, whether native or non-native, natural or reintroduced improves soil aggregate stability, total SOC, and total POM, primarily through lack of disturbance by tillage. Multi-species riparian buffers show initial improvements in soil quality parameters after only 7 years, but have the capacity to

continue to improve the soil quality parameters of soil aggregate stability and particulate organic matter twenty years after establishment, even though the length of a CRP contract for an MRB is commonly 10-15 years. Native warm-season grasses may take longer to establish and improve soil quality, although the other benefits of native warm-season grasses in the short-term, such as improved erosion control (Lee et al., 1998) and wildlife habitat (Berges et al., 2010) should not be overlooked. Additionally, the success of a particular MRB will be dictated by the soil properties already present on site, such as soil texture and bulk density.

After 20-21 years, cool-season grasses have slightly higher %WSA, larger GMD and MWD and greater SOC than the long-term grass; however, total POM is slightly higher in the long-term grass than in the cool-season grass. All these parameters were greater than that found under switchgrass. However, it was noted that soils under the former switchgrass buffer have made significant increases in the amount of water-stable aggregates present in the past 14 years; whereas only seven years after MRB establishment (Marquez et al., 1999; 2004) it was evident that soils under switchgrass were not significantly different in aggregate stability than the adjacent crop fields.

Trees in MRBs, whether naturally-occurring or reintroduced, seem to favor the production of small macroaggregates over large macroaggregates, but the implications of this finding are unclear. However, the total %WSA under hybrid poplar with a cool-season grass understory tends to be greater than under natural forest, and is similar to cool-season grasses alone. This may be mostly a function of clay percentage, which is significantly less in the natural forest versus the hybrid poplar or cool-season grass types.

Directions for future work

Total POM-C will be more accurately assessed by determining the total C contained within the silt+clay fraction extracted from whole soil samples and subtracting that from the total C contained within the whole soil sample, according to the protocol used by Marquez et al. (1999). Total POM within aggregate size fractions could be further refined into intra-aggregate POM and free POM by density fractionation, as according to the method used by Ontl et al. (2015). A more accurate representation of the aggregate size stability distribution could be obtained if the Total Soil Stability Index (Marquez et al, 2004) is used; this would require an additional test of capillary-wetted aggregate stability.

A secondary goal of this research is to use the Soil Management Assessment Framework or another suitable soil quality index to evaluate the impact that riparian buffers may have had on soil quality in order to justify the return on investment of having riparian buffer land enrolled in CRP for the last 23 years. Most CRP contracts are 10-15 years long; however, most carbon sequestration and/or soil quality studies on CRP land are performed either <10 years since CRP establishment (Gebhart et al., 1994), or is from upland sites, not riparian buffers or floodplain soils (Karlen et al., 1996; Kucharik, 2007). The soil quality indicator data obtained from this study should help to bridge this knowledge gap.

Table 2.1. Description of soil series contained within the study sites.

Soil Series	Parent material	Drainage class	Taxonomic class
Clarion	Glacial till	Moderately well-drained	Fine-loamy Typic Hapludolls
Coland	Alluvium	Poorly drained	Fine-loamy Cumulic Endoaquolls
Cylinder	Alluvium over glacial outwash	Somewhat poorly drained	Fine-loamy over sandy or sandy-skeletal, Aquic Hapludolls
Spillville-Coland complex[1]	Alluvium	Somewhat poorly drained; poorly drained	Fine-loamy Cumulic Hapludolls
Webster	Glacial till or local alluvium	Poorly drained	Fine-loamy Typic Endoaquolls

Table 2.2. Pooled mean (± SE) surface soil (0-15 cm) physical properties data collected within multispecies riparian buffer (MRB) vegetation zones and two adjacent row crop fields. CSG = cool-season grass, SWG = switchgrass, LTG = warm-season grass, N. Crop and S. Crop = row crop fields. Lower case letters indicate significant differences between vegetation types at the $p < 0.05$ level, using one-way ANOVA with comparison of means with a *post-hoc* Tukey-Kramer HSD test.

Vegetation Type	n	Bulk Density (g cm^{-3})	Sand (%)	Silt (%)	Clay (%)	Texture
N. Crop	12	1.33 ± 0.02a	61.5 ± 0.6	25.0 ± 0.6	13.5 ± 0.4d	Sandy Loam
S. Crop	12	1.33 ± 0.03a	58.4 ± 1.2	25.3 ± 1.2	16.4 ± 0.5cd	Sandy Loam
Hybrid Poplar	12	1.12 ± 0.03bc	38.5 ± 0.8	38.3 ± 0.7	23.2 ± 0.9a	Loam
Forest	9	1.13 ± 0.03bc	53.1 ± 1.7	31.0 ± 1.2	15.9 ± 0.7cd	Sandy Loam
CSG	12	1.11 ± 0.02bc	36.4 ± 1.4	43.3 ± 1.3	20.3 ± 0.4ab	Loam
LTG	9	1.05 ± 0.02c	41.2 ± 1.3	39.3 ± 1.2	19.5 ± 1.1b	Loam
SWG	12	1.19 ± 0.03b	49.2 ± 1.4	31.9 ± 1.4	19.0 ± 0.7bc	Loam

Table 2.3. Total soil carbon and nitrogen (0-15 cm) present in 20-year-old multispecies riparian buffer (MRB) vegetation zones, two long-term perennial areas (Forest and LTG) and two adjacent row crop fields (N. Crop and S. Crop). CSG = cool-season grass, SWG = switchgrass, LTG = long-term (>80 years) grass. Values are expressed as pooled means for all sampling dates ± SE. Lower case letters indicate significant differences between vegetation types at the $p < 0.05$ level, using one-way ANOVA with comparison of means with a *post-hoc* Tukey-Kramer HSD test.

Vegetation Type	Total C	Total N
	g kg^{-1} soil	
N. Crop	16.3 (0.3)c	1.6 (0.0)c
S. Crop	18.4 (0.4)c	1.8 (0.0)c
Hybrid Poplar	38.8 (2.2)a	3.8 (0.2)a
Forest	30.6 (1.1)b	2.6 (0.1)b
CSG	33.3 (1.0)b	3.1 (0.1)
LTG	30.9 (1.2)b	2.9 (0.1)b
SWG	29.4 (1.1)b	2.7 (0.1)b

Table 2.4. Slaked aggregate size distribution, expressed as mean weight diameter (MWD) and geometric mean diameter (GMD), of 0-15 cm soils expressed on a sand-free basis, present in 20-year-old multispecies riparian buffer (MRB) vegetation zones, two long-term perennial areas (Forest and LTG) and two adjacent row crop fields (N. Crop and S. Crop) collected in Nov. 2010, May 2011, July 2011, and Nov. 2011. CSG = cool-season grass, SWG = switchgrass, LTG = long-term (>80 years) grass. Values are expressed as pooled means for all sampling dates ± SE. Lower case letters indicate significant differences between vegetation types at the $p < 0.05$ level, using one-way ANOVA with comparison of means with a *post-hoc* Tukey-Kramer HSD test.

Vegetation Type	n	MWD (mm)	GMD (mm)
N. Crop	12	0.46 (0.03)d	0.22 (0.01)d
S. Crop	12	0.46 (0.03)d	0.22 (0.01)d
Hybrid Poplar	12	1.85 (0.11)bc	0.95 (0.07)bc
Forest	9	1.57 (0.14)c	0.73 (0.06)c
CSG	12	2.66 (0.09)a	1.33 (0.08)a
LTG	9	2.20 (0.10)b	0.99 (0.07)b
SWG	12	1.83 (0.10)bc	0.82 (0.05)bc

Table 2.5. Slaked aggregate size distribution expressed as percent total water-stable macroaggregates (> 0.25 mm) (%WSA), percent large macroaggregates (> 2 mm) (% LM), and small macroaggregates (0.25-2 mm) (% SM) of 0-15 cm soils present in 20-year-old multispecies riparian buffer (MRB) vegetation zones, two long-term perennial areas (Forest and LTG) and two adjacent row crop fields (N. Crop and S. Crop) collected in Nov. 2010, May 2011, July 2011, and Nov. 2011, expressed on a sand-free basis. CSG = cool-season grass, SWG = switchgrass, LTG = long-term (>80 years) grass. Values are expressed as pooled means for all sampling dates ± SE. Lower case letters indicate significant differences between vegetation types at the $p < 0.05$ level, using one-way ANOVA with comparison of means with a *post-hoc* Tukey-Kramer HSD test.

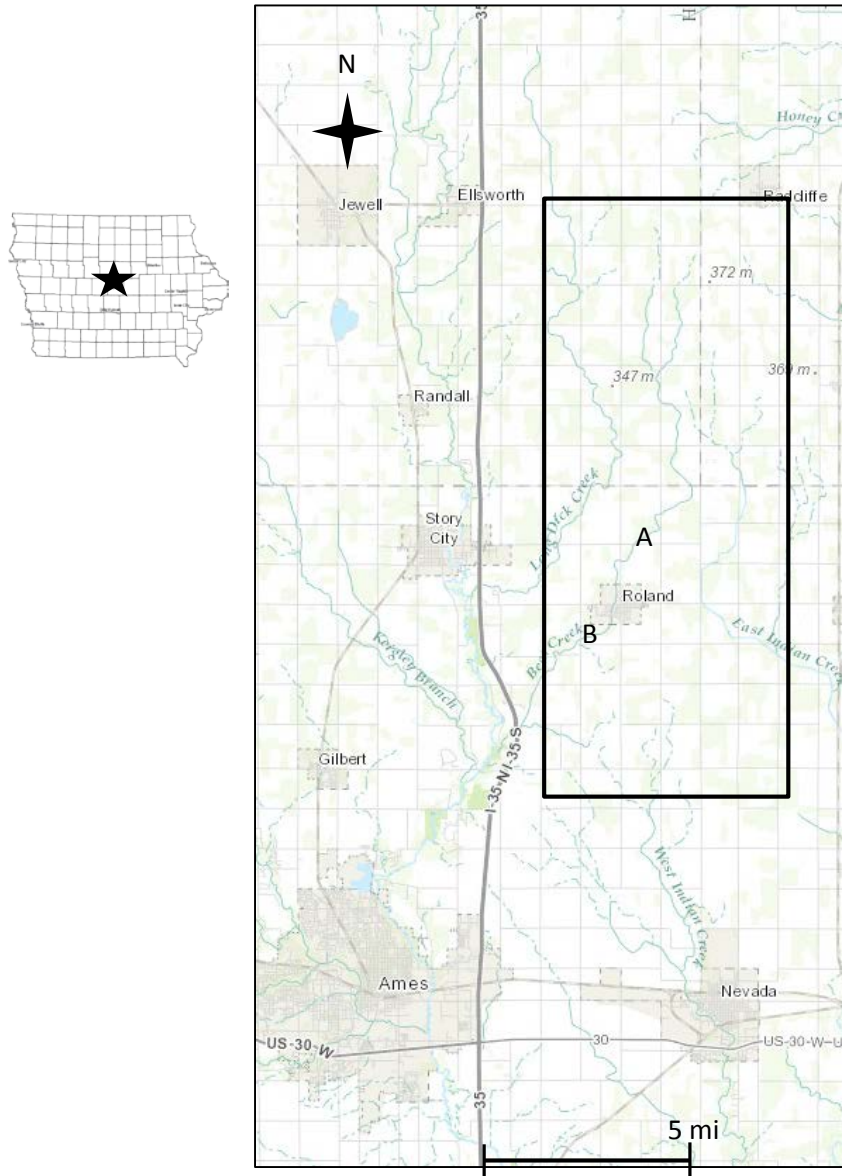| Vegetation Type	%WSA	% LM	% SM
N. Crop	28.9 (2.6)c	1.0 (0.2)d	28.7 (2.4)de
S. Crop	28.4 (2.7)c	1.3 (0.3)d	26.1 (2.7)e
Hybrid Poplar	77.4 (1.5)ab	24.7 (2.7)c	52.7 (2.0)a
Forest	69.2 (2.2)b	21.5 (4.5)c	50.0 (3.7)ab
CSG	78.9 (1.2)a	45.0 (2.1)a	33.8 (1.5)de
LTG	72.4 (1.6)ab	36.5 (2.6)b	36.7 (1.4)cd
SWG	69.8 (1.6)b	26.0 (2.2)bc	43.8 (1.2)bc

Figure 2.1. Location of Bear Creek watershed in Iowa, USA. Location A contains the warm-season, cool-season, switchgrass, poplar, and crop field plots (Detailed view in Figure 2.2). Location B contains the natural riparian forest plots.

Figure 2.2. Locations of (A) S. Crop, (B) switchgrass, (C) poplar, (D) cool-season grass, (E) N. Crop, and (F) long-term (> 80 years) grass plots in, or adjacent to, the multi-species riparian buffer (MRB) established in 1990. Triplicate plots established were 150 m² in size, but dimensions varied according to the width of the vegetation. Photo credit: 2013 natural cover aerial photo from Story County, Iowa Assessors Office.
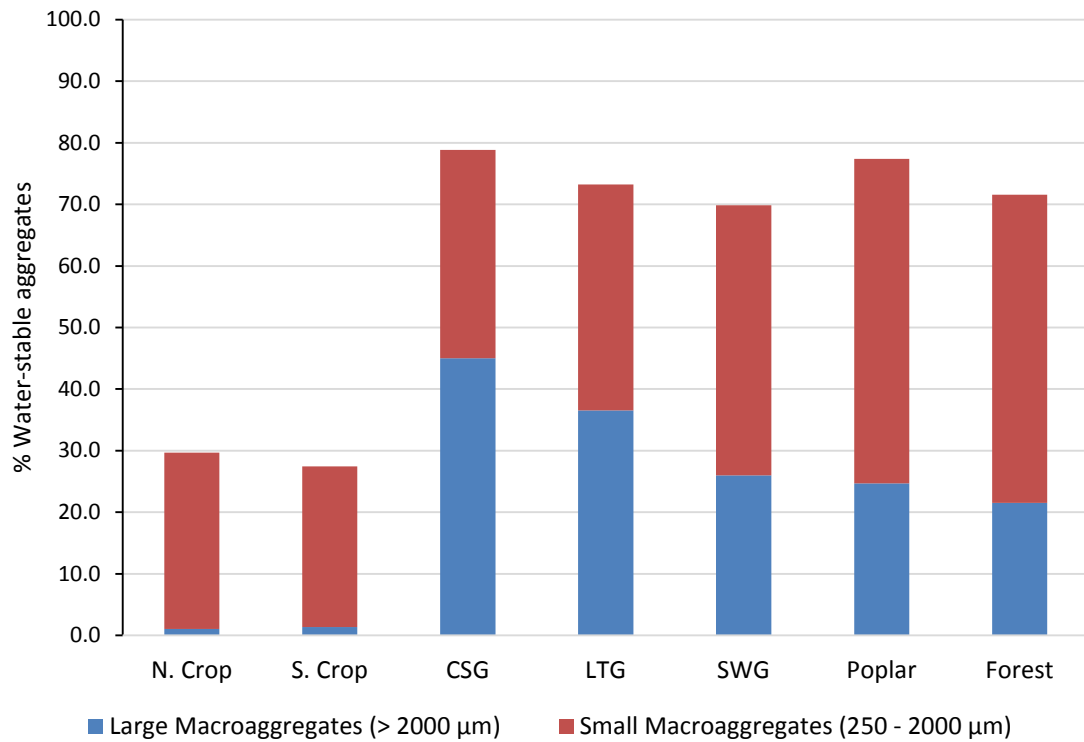
Figure 2.3. Water-stable large and small macroaggregates in various multi-species riparian buffer (MRB) vegetation zones and adjacent crop fields (N. Crop and S. Crop). Aggregate data are expressed on a sand-free basis. CSG = cool-season grass, SWG = switchgrass, LTG = long-term (>80 year) grass.
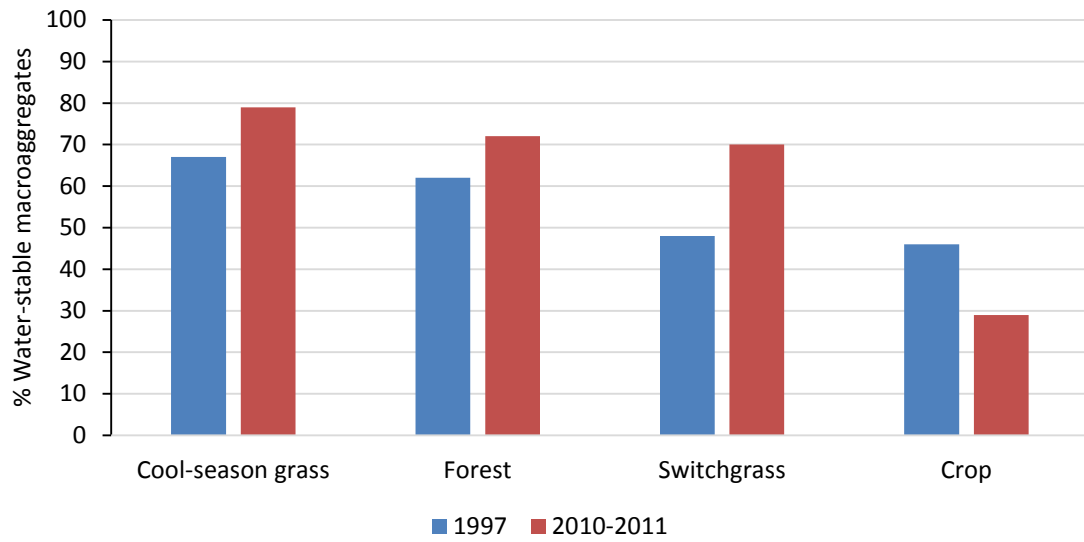
Figure 2.4. Comparison of water-stable macroaggregate percentage (> 0.25 mm) between 1997 and 2010-2011; 7 and 21 years post-MRB establishment.
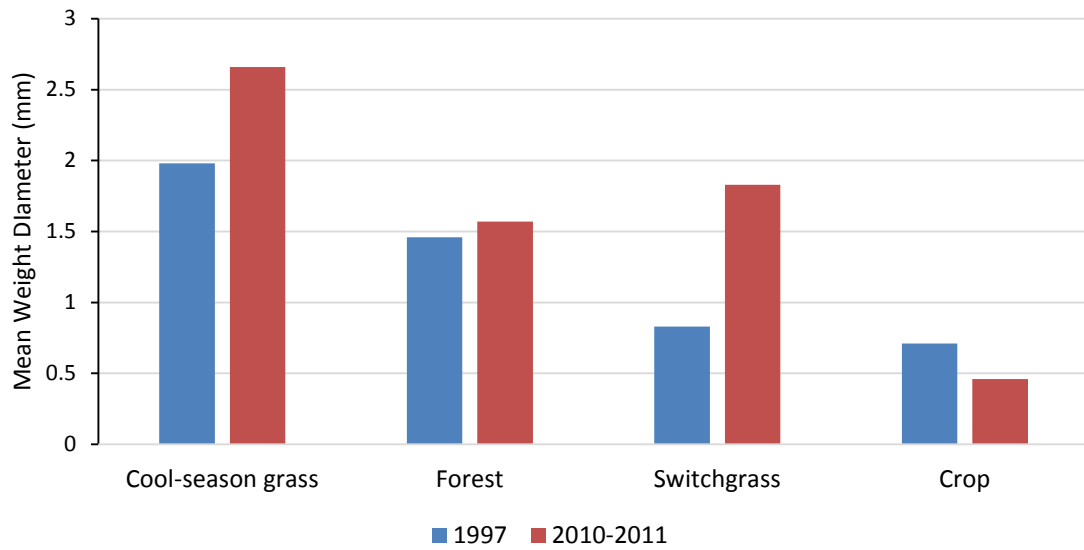

Figure 2.5. Comparison of mean weight diameter (MWD) of aggregates between 1997 and 2010-2011; 7 and 21 years post-MRB establishment.
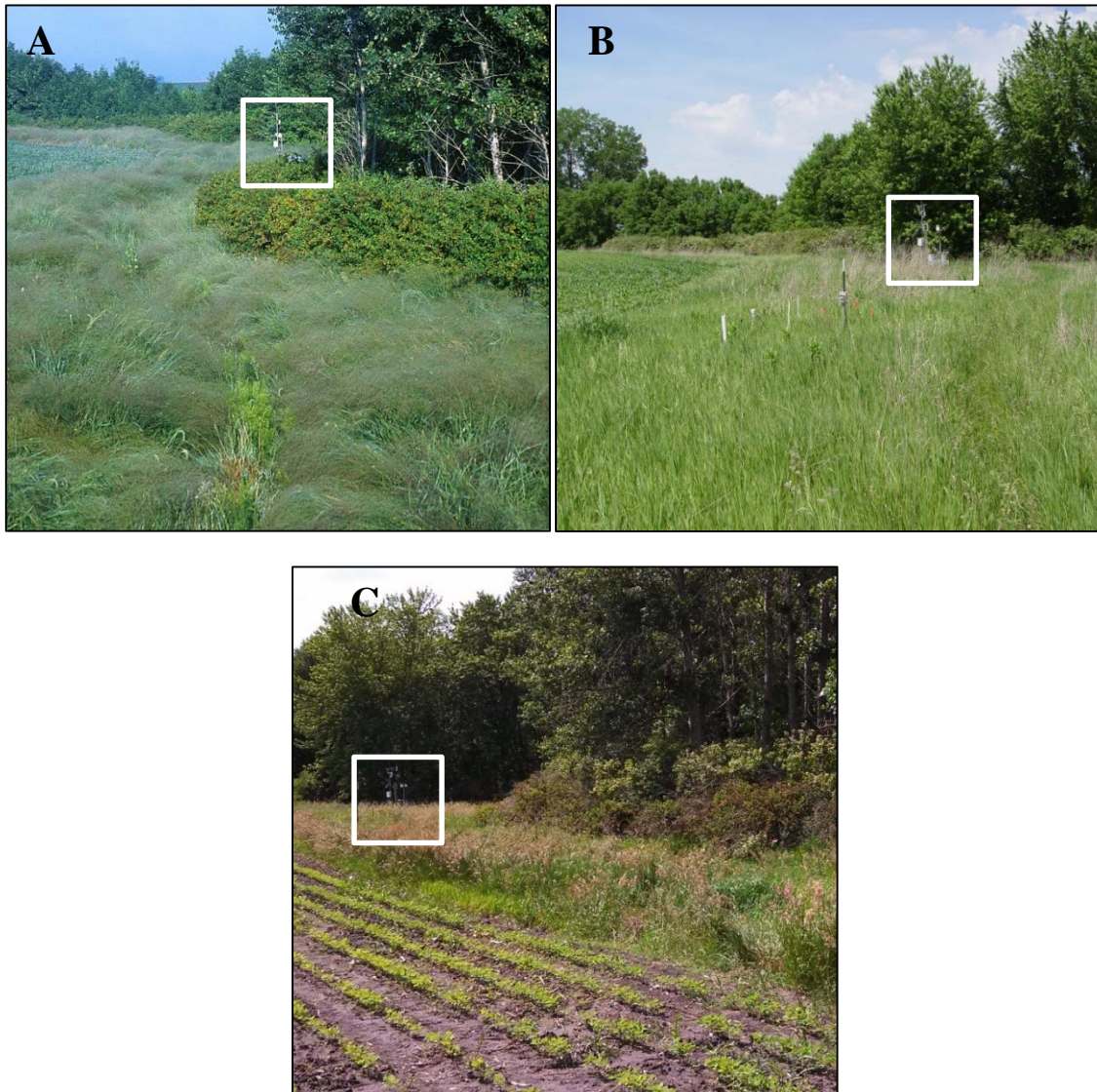
Figure 2.6. Switchgrass (*Panicum virgatum* L. 'Cave-in-Rock') stand decline over ~15 years. Note the weather station in the background (white box). Photo A (photo credit: R. Schultz) was taken in the late 1990's-early 2000's; the switchgrass stand is the predominant plant species present. Photo B (photo credit: D.G. Kim) was taken June 1, 2006; cool-season grass species are now predominant, but switchgrass is still evident to a small degree by the presence of dormant switchgrass stalks. Photo C was taken July 3, 2013; no clear evidence of the presence of switchgrass remains in this photo.

Figure 2.7. Total (free+intra-aggregate) particulate organic matter (POM) concentrations associated with aggregate size classes isolated from 0-15 cm soil samples collected in Nov. 2010, and May, July, and Nov. 2011 in various multi-species riparian buffer (MRB) vegetation zones and adjacent crop fields (N. Crop and S. Crop). Mean values are expressed as a mass of POM per mass unit of whole soil within each aggregate size class. CSG = cool-season grass, LTG = long-term (>80 year) grass, SWG = switchgrass.
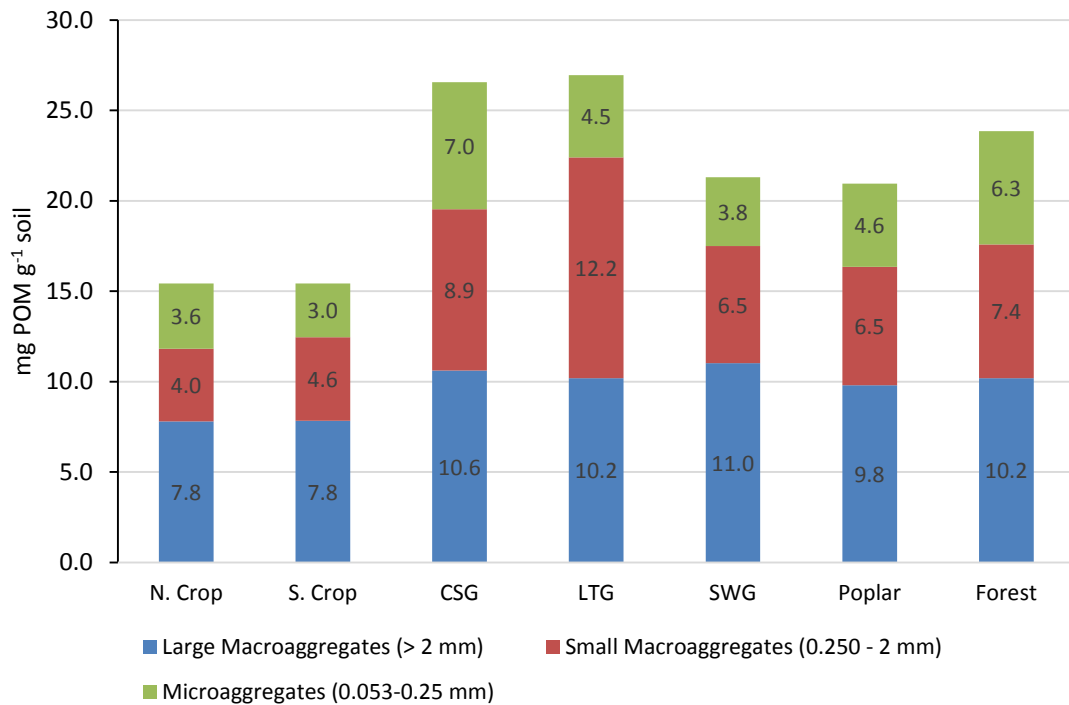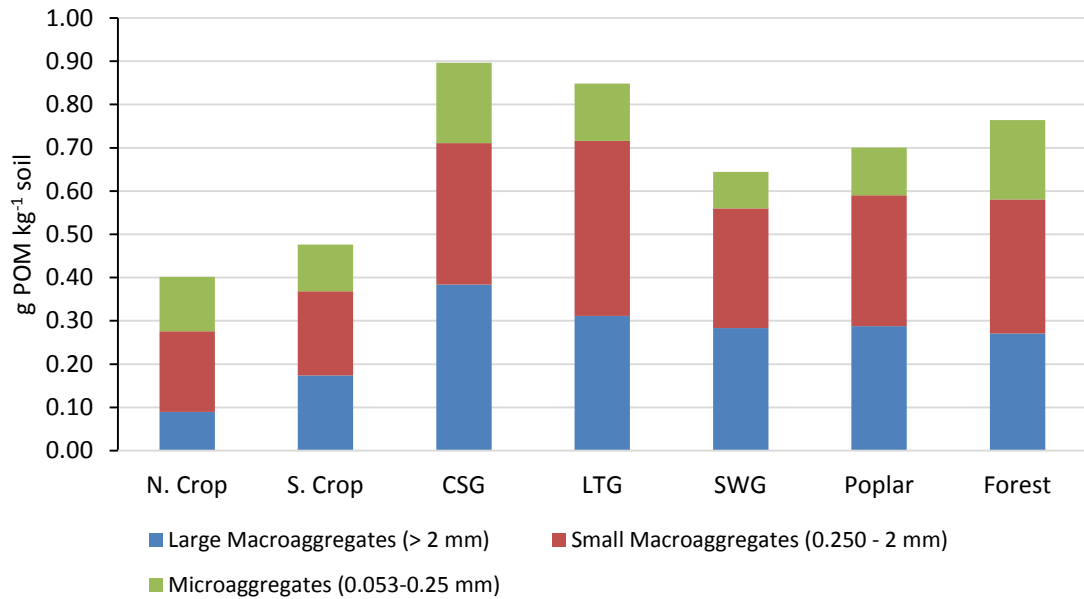
Figure 2.8. Total (free+intra-aggregate) particulate organic matter (POM) associated with aggregate size classes and expressed as a mass isolated from 0-15 cm soil samples collected in Nov. 2010, and May, July, and Nov. 2011 in various multi-species riparian buffer (MRB) vegetation zones and adjacent crop fields (N. Crop and S. Crop). Values are expressed as mass of POM per mass of whole soil without sand removal isolated from 100 g of soil. CSG = cool-season grass, LTG = long-term (> 80 year) grass, SWG = switchgrass.

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CHAPTER 3. CHANGES IN INFILTRATION RATES IN RE-ESTABLISHED MULTISPECIES RIPARIAN BUFFERS AND ADJACENT CROP FIELDS OVER TIME

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Abstract

Multispecies riparian buffers (MRBs) are designed to intercept surface runoff from adjacent row crop fields. The rate of soil water infiltration within the MRB dictates the length of time surface runoff and associated contaminants will interact either with the surface vegetation or the root zone of a riparian buffer system. A Cornell sprinkle infiltrometer was used to determine soil infiltration rates within 10-year-old and 23-year old riparian buffer vegetation and an adjacent row crop field with trafficked and untrafficked interrows. Sixty-minute cumulative infiltration did not vary significantly between MRB vegetation types, but was significantly higher than the trafficked interrow in the crop field. Cumulative infiltration and field-saturated infiltration rate are explained by the rainfall rate and the soil parameters bulk density and percent initial saturation. Perennial vegetation in the form of an MRB has positive impacts on water infiltration into soils.

Introduction

Multispecies riparian buffers (MRBs) are strips of any combination of trees, shrubs, and/or grasses planted parallel to stream channels, and are accepted best management practices for mitigating water quality issues from agricultural activities in upland areas (Schultz et al., 1995; Lowrance et al., 2002). The most important factor controlling the effectiveness of a MRB for the removal of non-point source pollutants is the hydrology present of the site.

Perennial vegetation, whether in the riparian zone or in the uplands, can improve soil structure and hydraulic properties by increasing the number and size of macropores (>75 µm diameter) (Arshad et al., 1996; Yunusa et al. 2002; Seobi et al., 2005), by improving aggregate stability (Arshad et al., 1996; Marquez et al., 2004), and by building organic matter (Marquez et al., 1999; Tufekcioglu et al., 2003). These improved soil and hydraulic properties, combined with the deeper and more extensive root systems (Tufekciolgu et al., 1999; Asbjornsen et al., 2007), greater water use and rainfall interception (Brye et al., 2000, Anderson et al., 2009), and longer growing seasons of perennial vegetation compared with annual crops contribute to positive impacts on hydrologic regulation. This is indicated by reduced surface runoff (Hernandez et al., 2013), increased soil water infiltration (Bharati et al., 2002; Schoonover et al. 2006; Guzman and Al-Kaisi, 2011), and increased saturated hydraulic conductivity (Udawatta et al., 2006, 2008) in perennially vegetated areas.

Infiltration capacity dictates the partitioning of water entering a MRB into surface runoff and subsurface flow, and affects the proportions of each. It will also dictate the

length of time surface runoff-associated sediment and contaminants will interact either with the surface vegetation or the root zone of a MRB. Allowing surface runoff to infiltrate into a MRB will enable the associated dissolved contaminants to come into contact with the root zone and microbial community present underneath the perennial vegetation. Nitrate present in surface runoff can undergo denitrification to nitrogen gas (Correll and Weller, 1989), with only a small chance that transformation to nitrous oxide will occur (Isenhart et al., 2009). Herbicides (eg., atrazine) can interact with the root zone of the riparian buffer to undergo degradation (Lin et al., 2004). However, it has also been suggested that excessive infiltration rates, such as those associated with the macropores created by anecic earthworm species, have been associated with rapid leaching of certain non–adsorbed chemicals, such as nitrate, to groundwater (Li and Ghodrati, 1995; Shipitalo et al., 2000; Shuster et al., 2003). Very high earthworm activity has been anecdotally observed within young silver maple MRBs (Bharati, 1997).

Previous infiltration work specifically in MRBs used in this study has been carried out by Bharati et al. (2002) using double ring infiltrometers. It was found that perennial vegetation in six-year old MRBs increased water infiltration in MRBs five times as compared to row-cropped or pastured sites. In southern Illinois, USA, Schoonover et al. (2006) found that 30-year old stands of giant cane (*Arundinaria gigantea* (Walt.) Chapm.), used in place of a grass filter strip, and natural forest buffers of the same age exhibited relatively high mean infiltration rates; 83 cm h^{-1} and 50 cm h^{-1}, respectively. This is greater than the rates measured by Bharati et al. (2002), and may be due in part to differences in age since establishment.

A riparian forest buffer with an intact understory or associated with a grass filter strip provides greater water quality mitigation than woody vegetation alone can provide. Lin et al. (2004) encouraged incorporation of switchgrass, tall fescue and smooth bromegrass into riparian forest buffers for bioremediation of atrazine and nitrate. Daniels and Gilliam (1996) found that 6 m wide grass filter strips and grass filter strips plus forested areas reduced sediment load from sheet and rill erosion from row crop fields by 60-90%, but where concentrated runoff moved from crop fields through forested riparian areas with little ground cover, nutrient and sediment loads were reduced very little. Knight et al. (2010) found that 20% of the narrow naturally-occurring riparian forests assessed in northeastern Missouri had concentrated flow paths that connected the row crop field to the stream, and where the riparian forest was bordered by a USDA-NRCS approved grass filter strip, concentrated flow entering the grass filter strip was prevented from reaching the riparian forest 100% of the time.

Existing riparian forest buffers may have the understory shaded out due to canopy density. The unmanaged riparian forests studied by Knight et al. (2010) had an average of 767 trees ha^{-1} (avg. dbh = 16.4 cm), which is higher than the USDA-NRCS suggested 222 trees ha^{-1} (avg. dbh = 20-30 cm) to 550 trees ha^{-1} (avg. dbh = 5-16 cm) in a riparian buffer (USDA-NRCS, 2014). High-quality forest remnants have a ground cover of shade-tolerant perennial plants during much of the season, particularly in spring when runoff is high (Mabry et al., 2008), but over-grazed natural remnants and re-established riparian buffers may not.

Soil macroaggregates (>250 µm), which also help to create macropores, are usually the least stable aggregate size fraction. Unstable soil aggregates can be disrupted,

particularly under rapid wetting events (Gale et al., 2000). Additionally, soils not protected by plant litter may have fine particles detached by raindrop energy. These products can create soil crusts or plug soil pores, thereby restricting infiltration rates (Arshad et al., 1996). Soil freezing also reduces erosion resistance for soils; therefore, an herbaceous ground cover may provide more added protection against soil weakening due to freeze/thaw cycling than deciduous woody vegetation and litter alone (Wynn and Mostaghimi, 2006).

The overall goals of this study were: (1) determine if water infiltration rates vary with MRB vegetation type and ground cover, (2) determine whether infiltration rates have increased, decreased, or remained constant over time since the establishment of the MRB, and (3) to explore the use of a portable infiltrometer for use in MRBs where use of traditional infiltration equipment is logistically difficult. To accomplish this, soil infiltration rates were determined using a Cornell sprinkle infiltrometer in riparian buffers and adjacent row crop fields composed of different types of vegetation. Visual surveys were performed to determine understory and overstory vegetation composition, and soil samples were collected to determine antecedent soil moisture, soil texture, and aggregate stability.

Materials and Methods

Site descriptions

Infiltration plots were established in 2012 in 22-year old MRB vegetation used in the infiltration research undertaken by Bharati et al. (2002), located along Bear Creek in

Story County, Iowa, USA (Figure 3.1). Detailed information about the design of this MRB has been published by Schultz et al. (1995). Three replicate infiltration plots were laid out 11 m apart on the following vegetation types: switchgrass (*Panicum virgatum* L. 'Cave-in-Rock') invaded by smooth brome (*Bromus inermis*) and other cool-season exotic species; cool-season grass, predominantly smooth brome, reed canarygrass (*Phalaris arundinacea*), and Kentucky bluegrass (*Poa pratensis* L.); and silver maple (*Acer saccharinum)* vegetation zones of the MRB. Within the silver maple plot, plots were only 9.1 m apart, in order to achieve three replications within that vegetation type.

Infiltration plots were also established in the row crop fields adjacent to the riparian buffer in trafficked and untrafficked interrows. Trafficked and untrafficked interrows were visually identified after crop planting. The field was planted to a corn (*Zea mays* L.)-soybean (*Glycine max*) rotation; the crop was corn in 2012 and soybeans in 2013. Infiltration plot areas received tillage in the fall after infiltration measurements were performed.

Additional infiltration plots were established on a site in the Long Dick Creek watershed, 2.8 km from the Bear Creek MRB, hereafter referred to as the 'CRP' site (Figure 3.1). This site had been continuously used as pasture land since at least the 1930s (Iowa State University Geographic Information Systems Support and Research Facility, 2014), but was placed in the Conservation Reserve Program in 2003 (personal communication with the landowner). A MRB consisting of black walnut *(Juglans nigra),* green ash (*Fraxinus pennsylvanica)* and honey locust (*Gleditsia triacanthos)* was planted directly into the pasture sod, dominated by Kentucky bluegrass and timothy (*Phleum pratense*).

Soils within the study area are predominately mapped as Coland (Cumulic Endoaquoll) clay loam, 0-2% slope, formed in alluvium. Other soils present within the study are Cylinder (Aquic Hapludoll) loam, 0-2% slope, formed in loamy sediments over sand and gravel; Clarion (Typic Hapludoll) loam, 2-6% slope, formed in glacial till; and Webster (Typic Endoaquoll) clay loam, 0-2% slope, formed in glacial till or local alluvium derived from till (DeWitt, 1984).

Infiltration measurements

Field-saturated infiltration was measured in July 2012, Sept. 2012, and late June/early July 2013 with a Cornell sprinkle infiltrometer (Cornell University, Ithaca, NY) (Ogden et al., 1997) (Figure 3.2). The Cornell sprinkle infiltrometer system consists of a portable rainfall simulator moderated by a Mariotte tube to provide constant head. The infiltrometer is placed onto a single 24.1 cm diameter ring inserted into the ground to 7.5 cm, and allows for application of simulated rainfall at a wide range of predetermined rates. This system was chosen over more standard infiltration equipment (eg. double-ring infiltrometers) because it wets the soil in a more natural manner and eliminates soil slaking as a result of instantaneous ponding and reduces unnaturally high contributions of macropore flow under ponded conditions. Additionally, it was chosen for the ease of use by an individual operator, conservation of water, and ability to be operated in small spaces.

The individual Cornell sprinkler infiltrometers were calibrated each morning to deliver a target rate of 30 cm hr^{-1} of simulated rainfall by measuring the drop in the water column within the infiltrometer in centimeters over a two minute period. Actual rainfall rates for each measurement period were calculated, as the simulated rainfall rates varied

under field conditions, due to temperature variations in the water and shifting of simulator tubes during transport of the infiltrometer to the measurement plot. The source water used in the infiltration measurements was local potable water, and was assumed to not cause excessive dispersion of soil particles over natural rainfall (Bohl Bormann et al., 2010).

Simulated rainfall rate was calculated by dividing the difference of the water column at the beginning and end of the simulation by the time elapsed. Runoff rate (ro_t, cm min^{-1}) was calculated with the following equation (van Es and Schindelbeck, undated):

$$ro_t = \frac{V_t}{(457.30 * t)}$$

[3.1]

Where: 457.30 (cm^2) is the area of the ring,

t = time interval (min), and

V_t = volume of water (mL) during time interval t

Infiltration rates (i_t) were determined by the difference between the rainfall rate and runoff rate. Field-saturated infiltrability (i_{fs}) reflects the steady-state infiltration capacity of the soil after wet-up, and is based on the data collected at the end of the measurement period, or whenever steady-state runoff conditions occur. Runoff occurs when the soil has reached field saturation; therefore, calculating field-saturated infiltrability allows comparison of sites with different initial soil moisture content. Since the infiltration apparatus uses a single ring, the measured infiltration rate needs to be adjusted for three-dimensional flow at the bottom of the ring. The adjustment factor used was 0.8 (van Es and Schindelbeck, undated), based on the loam soil texture present in the plots that

generated runoff (Table 3.1), and data from Reynolds and Elrick (1990) who used numerical modeling to estimate the effects of three-dimensional flow at the bottom of the ring.

Sixty-minute cumulative infiltration (i_c) was calculated by summing the field-saturated infiltrability (i_{fs}) for each time interval for the entire measurement period and normalizing that result to reflect a 60-min time period. For plots that never produced runoff, a minimum 60-min cumulative infiltration was estimated by multiplying the rate of water applied per minute by 60 min.

Soil sampling

At each infiltration plot, prior to performing the infiltration measurements, three 7.5 cm long by 3.175 cm diameter soil cores were pulled equidistantly from the soil surrounding the infiltration ring with a push probe, composited and brought back to the laboratory. The total weight of the composite sample was recorded. Composite samples were then sieved through an 8 mm sieve; rocks larger than 8 mm and vegetation were removed. A subsample of the composite soil sample was weighed and dried at 105°C to determine antecedent gravimetric soil moisture (Gardner, 1986) and calculate soil bulk density (Blake and Hartge, 1986). Gravimetric soil moisture and bulk density were used to determine water-filled pore space, or percent saturation (Linn and Doran, 1984). Water-stable macroaggregate (>0.25 mm) percentage (%WSA) and soil texture via the pipet method were determined on soil samples taken from the surface 15 cm in 2010-2012 for a companion study (this thesis, Chapter 2).

Vegetation surveys

In July 2013, a visual assessment of the ground cover within the infiltration plot was completed prior to setting up the infiltration equipment. The percent ground cover and shrub canopy cover in a 1 m^2 area surrounding the infiltration ring was determined and divided into the following categories: warm-season grasses, cool-season grasses, leguminous forbs, non-leguminous forbs, woody shrubs, litter (herbaceous and woody), standing dead woody material, and bare ground. Cover for each of the categories was assigned an average percent range, designated as follows: 0%, 0.5%, 3%, 16%, 38%, 63%, 85%, and 96%. Ground cover percentage was calculated by summing the percentages of each category, but excluding the percentage of bare ground. Rooted ground cover was estimated by summing the percentages of living ground cover, excluding the categories of litter and standing dead woody material.

In the silver maple plots, the basal area was determined in late 2012 using a 10x factor wedge prism. Measurement of tree diameter at 130 cm above the ground (D_{130}) was recorded for all countable trees.

Statistical analysis

Percent clay, %WSA, antecedent soil moisture, and % rooted ground cover were analyzed with one-way analysis of variance (ANOVA). Sixty-minute actual and estimated cumulative infiltration was analyzed with a standard least-squares model, with vegetation, % saturation, and rainfall rate as the effects parameters in a full factorial design in the model. Since % saturation is dependent on antecedent moisture and is calculated using bulk density, it was used as a proxy for sampling date and bulk density in the least-squares

model.   Tukey-Kramer HSD *post-hoc* tests were used to compare means between vegetation types. Results were considered significant if $p \leq 0.05$. For plots where runoff volume exceeded infiltration capacity, truncated linear regression, fitted with a quadratic equation, was used to describe the response of i_{fs} to % soil saturation and rainfall rate. All statistics were performed using the JMP Pro v. 11.0.0 statistical software package (SAS Institute, Inc., Cary, NC, USA).

Results and Discussion

Soil properties

Table 3.1 summarizes soil texture and the amount of water-stable aggregates in the surface 15 cm of soil, collected as part of a companion study (this thesis, Chapter 2). Samples from the silver maple and CRP plots were collected in Nov. 2012. Clay % differs significantly between the CRP and the other vegetation types ($p < 0.0001$). The clay % under silver maple, cool-season grasses, and switchgrass are significantly greater than that found under the crop ($p < 0.008$, $p < 0.0001$, and $p < 0.0003$, respectively). Water-stable macroaggregate percentage (%WSA, < 0.25 mm) was significantly higher than the row crop soil for all MRB types.

Bulk density in the surface 7.5 cm of the soil did not differ significantly ($p \leq 0.05$) among sample dates. Figure 3.3 shows the pooled data from all three sample dates; plots within trafficked interrows have significantly higher bulk densities than all riparian vegetation types ($p \leq 0.002$); the difference between trafficked and non-trafficked interrows was slightly insignificant ($p \leq 0.056$). The CRP plots had significantly lower

bulk densities than any of the other plots ($p \leq 0.006$). There were no other significant differences detected among any of the other vegetation types. Trends in bulk density may be explained by the fact that the other MRB vegetation plots receive moderate foot traffic at least monthly, but the CRP plots are now seldom visited by humans either on foot or by vehicles since establishment in 2003. It is known that most of the increase in bulk density occurs in the first few trips across a soil (Cambi et al., 2015).

Antecedent soil moisture does vary among dates, as expected (Table 3.2). Soil moisture was lowest during July 2012 for all vegetation types except for the silver maple and switchgrass. By Sept. 2012, soil moisture had increased significantly in the untrafficked and trafficked portions of the row crop field ($p \leq 0.001$ and $p = 0.003$, respectively); however, in the perennial vegetation, there was not a statistically significant increase. By late June/early July 2013, antecedent soil moisture was significantly greater ($p \leq 0.021$) than it had been in July 2012 in all vegetation types except for silver maple.

During all three sampling periods, precipitation was well below normal levels, although April and May 2013 had higher than normal precipitation, which recharged the soil moisture profile going into the late June/early July 2013 sample date (Figure 3.4). Plant growth stage, canopy cover, and air temperature and humidity will all affect the amount of evapotranspiration (ET) occurring, and thus alter the antecedent soil moisture. In July 2012, corn was at stage VT, which is complete canopy closure with tassel formation (Figure 3.5A); corn can uptake on average 0.20-0.25 inches of water per day at this stage (Abendroth et al., 2011). In September 2012, the corn had been harvested (Figure 3.5B) and therefore no further ET was occurring, while ET was still occurring in the perennial vegetation. In late June/early July 2013, soybeans were predominately in the V1 growth

stage (Figure 3.5C); at this emergence stage, soybeans can uptake 0.10-0.15 inches of water per day (Licht, 2014).

Antecedent soil moisture also varies by vegetation type within sampling date (Table 3.2). At all three sampling dates, CRP had the greatest antecedent soil moisture. During July 2012 and 2013, the row crop vegetation, both trafficked and untrafficked interrows, had the least antecedent soil moisture. This was not the case in Sept. 2012, again likely because of the lack of actively growing vegetation and increased ground cover from corn stover remaining after harvest. There are no clear trends among the other perennial vegetation types within sampling date.

In July 2012, there was a statistically significant difference in antecedent soil moisture between vegetation treatments as determined by one-way ANOVA ($F(5,12) = 20.758$, $p < 0.001$). A Tukey HSD *post-hoc* test revealed that the CRP vegetation had statistically significantly ($p < 0.049$) higher antecedent soil moisture than all other vegetation treatments except silver maple (Table 3.2). Silver maple had statistically significantly ($p < 0.049$) higher antecedent soil moisture than the row crop treatments, but not higher than any of the other perennial vegetation besides the CRP treatment in July 2012.

In Sept. 2012, there was a statistically significant difference in antecedent soil moisture between vegetation treatments as determined by one-way ANOVA ($F(5,13) = 15.013$, $p < 0.001$). A Tukey HSD *post-hoc* test revealed that CRP vegetation had statistically significant ($p < 0.021$) higher antecedent soil moisture than all other vegetation treatments (Table 3.2). There were no statistically significant differences between the other vegetation types, but the cool-season grass treatment had slightly more antecedent soil

moisture than the other vegetation types. Antecedent soil moisture for the silver maple and switchgrass vegetation types was highly variable among the replications in Sept. 2012.

In June/July 2013, there was a statistically significant difference in antecedent soil moisture between vegetation treatments as determined by one-way ANOVA ($F(6,14) = 9.569$, $p < 0.001$). A Tukey HSD *post-hoc* test revealed that CRP vegetation had statistically significantly ($p < 0.015$) higher antecedent soil moisture than silver maple and the row crop field in both trafficked and untrafficked interrows (Table 3.2). However, CRP vegetation did not have statistically significantly higher antecedent moisture than the cool-season grass or switchgrass vegetation types. Switchgrass ($p < 0.006$) and cool-season grasses ($p < 0.023$) had statistically significantly higher antecedent soil moisture than trafficked and untrafficked interrows, but not any of the other perennial vegetation treatments. The silver maple was not significantly different than the row crop treatments in June/July 2013.

Vegetation surveys

Ground cover within the switchgrass infiltration plots is no longer characterized by switchgrass, or by any other species of warm-season grass, as indicated by Table 3.3. A comparison of photographs taken at the site over approximately 15 years from the late 1990's-early 2000's until 2013 does indicate the decline of the switchgrass monoculture (Figure 3.6).

There is a significant difference between % rooted ground cover between the MRB vegetation and the row crop field ($p < 0.0012$). While there are trends in % rooted ground cover among MRB vegetation types, there were no significant differences ($p < 0.4258$). The young CRP plot (100.7%) and both grass plots (switchgrass = 99.3%, cool-season grass = 97.3%) contained more rooted ground cover than the silver maple plot (77.8%), indicating that tree shading is slightly impacting the amount of live rooted ground cover present. This is confirmed by the tree mensuration data; basal area in the silver maple zone was 55.1 m^2 ha^{-1}, and mean D_{130} ($\pm$SE) for all trees counted within the prism plot (n=28) was 24.0$\pm$1.0 cm. This is more than the recommended stand density of 222 trees ha^{-1} (avg. dbh = 20-30 cm) in a riparian forest buffer (USDA-NRCS, 2014), indicating that some thinning ought to occur to open up the forest canopy to increase the amount of live ground cover.

Infiltration

Runoff was not achieved during some infiltration trials, regardless of the amount of rainfall applied (Table 3.4). In these instances, a minimum 60-min cumulative

infiltration was estimated by multiplying the rate of water applied per minute by 60 min. Actual 60-min cumulative infiltration rates may therefore be much greater than reported.

For those plots which did generate runoff, steady-state runoff was determined to occur at approximately 30 minutes after the start of runoff in most plots in this study, thus mean ro_t, i_t, and i_{fs} were calculated from data collected between minutes 30 and 60 of the infiltration trial.

General trends for actual and estimated 60-min cumulative infiltration for the three sample dates in the MRB vegetation (Figure 3.7) showed the least 60-min cumulative infiltration occurring in July 2013 and the most occurring in Sept. 2012, with the exception occurring in the silver maple. In the silver maple plots, the least 60-min cumulative infiltration occurred in July 2012, and Sept. 2012 and July 2013 had similar 60-min cumulative infiltration amounts. This trend is different for the trafficked and untrafficked interrows, in which cumulative infiltration was highest in July 2012, and similar in Sept. 2012 and July 2013. This may be driven by antecedent soil moisture, which was significantly less in July 2012.

The least-squares model (Figure 3.8) showed there were significant differences in actual and estimated 60-minute cumulative infiltration among vegetation types ($p <$ 0.0001), in the order of CRP $\geq$ cool-season grass = switchgrass = silver maple $\geq$ non-trafficked interrow $\geq$ trafficked interrow. All MRB vegetation types had significantly greater cumulative infiltration than the trafficked interrow ($p \leq$ 0.0002). Cumulative infiltration in the non-trafficked interrow was not significantly different than the trafficked interrow ($p =$ 0.3502) or the MRB vegetation ($p \geq$ 0.2305), with the exception of the CRP ($p =$ 0.0410). Rainfall rate explained differences in cumulative infiltration ($p <$ 0.0001,

but not % saturation ($p = 0.1540$). This may be because % saturation does not take pore size into account, and it is known that infiltration is driven primarily by macropores. The combined interaction of all factors is slightly insignificant ($p = 0.0935$).

When average field-saturated infiltrability (i_{fs}) is regressed against % saturation in the soil, there is a slight negative quadratic correlation (Figure 3.9); that is, when % saturation goes up, the infiltration rate goes down. Pores in the soil are already holding water, thus additional water cannot enter the soil and is more likely to run off. Conversely, when i_{fs} is regressed against rainfall rate (Figure 3.10), there is a slight positive correlation; when rainfall rate increases, i_{fs} also increases. However, fit of the line is improved from $r^2 = 0.233$ to 0.586 when points from the trafficked interrow are removed, indicating that bulk density may drive i_{fs} more than rainfall rate at a certain point. Bulk density is negatively correlated with both 60-min cumulative infiltration (Figure 3.11) and i_{fs} (Figure 3.12); as bulk density increases, infiltrability and cumulative infiltration decreases. An additional confounding variable might be the stability, or lack thereof, of soil aggregates. High-intensity rainfall may slake unstable soil aggregates creating a surface seal which slows infiltration. The row crop soils have more unstable aggregates than any of the MRB soils (Table 3.1).

Conclusions

Perennial vegetation in the riparian zone, regardless of whether it is a grass, forb, shrub, or tree, does appear to have positive impacts on the amount of water infiltration into soil. However, it is the underlying soil factors of bulk density, porosity, aggregate stability,

and initial water content that drive the infiltration process, factors which can be influenced by vegetation.

Various methods of measuring infiltration make comparisons between research studies challenging. Only one publication was found (Zwirtes et al., 2013) which compared the use of the Cornell sprinkle infiltrometer with the double ring infiltrometer in native forests, pastures, and no-till row crop soils in Brazil. Infiltration rates as estimated, as no runoff was observed, were 30 cm hr^{-1} for the Cornell sprinkle infiltrometer in the native forest. Infiltration as measured with the double ring infiltrometer was 142.8 cm hr^{-1} in the Zwirtes et al. (2013) study. Therefore, statistical comparisons are not able to be made between this study and the earlier infiltration study carried out by Bharati et al. (2002), although measurements were made within the same plots.

Figure 3.13 does show general trends between 1995 and 2012-2013, though runoff was only achieved in 4 of 9 trials in the silver maple and cool-season vegetation zones, and 5 out of 9 trials in the switchgrass zones; actual 60-min cumulative infiltration may be much larger than what is reported for 2012-2013. Cumulative infiltration for silver maple is trending downward as the buffer ages. This may be due to a loss of macropores from a loss in water-stable aggregates. Although %WSA was not significantly different between silver maple (%WSA = 62.3±7.2, n = 2) and cool-season grass (%WSA = 78.9±1.2, n = 12); variability within the silver maple was high. Means may have separated themselves given more %WSA samples collected within the silver maple plots. Natural forest understory vegetation, even when sparse, can promote high infiltration rates; Schoonover et al. 2005 noted that a 30-year-old mixed deciduous riparian forest with a sparse rooted

ground cover, and a discontinuous 1.7 cm thick litter layer covering 18% of the area, had mean cumulative infiltration rates of 50 cm h^{-1}.

Periodic maintenance of riparian forest buffers to allow an adequate stocking rate to permit understory growth to occur may be necessary, as we noted a decrease in infiltration capacity under silver maple which has lost its grass understory in exchange for a more diverse understory of forbs. Bare ground may possibly lead to the destruction of soil aggregates through raindrop impacts and lessen infiltration capacity. Litter cover in the silver maple plots was even less than this at 10.8%, but was highly variable among the three plots (SD = 8.9%), as was the amount of bare ground (23.3% ± 12.7) and live rooted vegetation (77.8% ± 11.9). Time of year also dictated the ground cover present in the silver maple; preliminary measurements taken in November 2012 found few forbs present (8.9%) and more litter (85.0%), but measurements taken in July 2013 found many more forbs (54.7%) and less litter (10.8%). Further work remains to be done exploring forest understory structure in relation to infiltration capacity, POM and aggregate stability.

Various methods of measuring infiltration make comparisons between research studies challenging. Therefore, statistical comparisons were not able to be made to the earlier infiltration study carried out by Bharati et al. (2002), although measurements were made in the same plots as this earlier study. Cumulative infiltration for switchgrass and cool-season grasses were similar from 1995 to 2012-13, but cumulative infiltration under silver maple was drastically reduced. This may be due to a loss of macropores from a loss in water-stable aggregates. Although mean %WSA was not significantly different between silver maple (%WSA = 62.3±7.2, n = 2) and cool-season grass (%WSA = 78.9±1.2, n = 12), cool-season grass does have higher %WSA. Means may have separated themselves if

more %WSA samples had been collected within the silver maple plots, as variability is high in the silver maple.

Perennial vegetation can create soil conditions favorable to high infiltration rates, higher than can be reasonably measured with the Cornell sprinkle infiltrometer. The maximum linear water holding capacity for the Cornell sprinkle infiltrometer is approximately 45 cm; other studies in perennial vegetation measured 60-min infiltration rates greater than this (Schoonover et al., 2006, Zwirtes et al., 2013). It is not advisable to refill the Cornell sprinkle infiltrometer as that would alter the constant pressure head applied to the infiltrating water. A different measurement device, capable of supplying more water, might be more suitable for these cases (such as a double-ring infiltrometer with a large water supply tank). The need for larger amounts of water for infiltration measurements is offset by the challenge of accessibility in areas of perennial vegetation, such as in riparian forest buffers. Future work could also incorporate a side-by-side comparison of a standard double-ring infiltrometer with a Cornell spring infiltrometer.

These results come from a single study with few replications, and do not allow a general statement to be made about different types of vegetation in MRBs. Results from this particular study, however, do demonstrate infiltration capacity was greater in the MRB than in the row crop soils that are impacted by wheel traffic and are generally better in MRBs than row crop soils not impacted by wheel traffic.

Table 3.1.. Soil texture and water-stable aggregate (>0.25 mm) percentage, 0-15 cm. Values are expressed as mean ± SE. Lower case letters indicate significant differences between vegetation types at the $p < 0.05$ level, using one-way ANOVA with comparison of means with a *post-hoc* Tukey-Kramer HSD test.

Vegetation Type	n	Soil Texture			Soil Texture class	Water-stable aggregates (%)
		Sand (%)	Silt (%)	Clay (%)		
Untrafficked interrow Trafficked interrow	24	59.9 (0.7)[1]	25.2 (0.6)	14.9 (0.4)c	Sandy Loam	28.6 (1.8)b
CRP	3	20.6 (4.9)	46.1 (2.0)	33.3 (3.4)a	Clay Loam	78.3 (1.7)a
Silver maple	2	36.5 (9.6)	42.1 (5.6)	21.5 (4.1)b	Loam	62.3 (7.2)a
Cool-season grass	12	36.4 (1.4)	43.3 (1.3)	20.3 (0.4)b	Loam	78.9 (1.2)a
Switchgrass	12	49.2 (1.4)	31.9 (1.4)	19.0 (0.7)b	Loam	69.8 (1.6)a

[1]Soil texture and water-stable aggregate values for the untrafficked and trafficked interrows were pooled, as differentiation between trafficked and untrafficked interrows was not made at the time of sampling for soil texture.

Table 3.2. Mean antecedent gravimetric soil moisture (g/g) in the surface 7.5 cm of the soil. Within each column, lower-case letters indicate statistically significant differences between vegetation types during the same sampling date using a one-way ANOVA with a post-hoc Tukey test ($\alpha = 0.05$). Within each row, upper-case letters indicate statistically significant differences between sampling dates within the same vegetation type using a one-way ANOVA with a post-hoc Tukey test ($\alpha = 0.05$).

Vegetation Type	July 2012	Sept. 2012	July 2013
Untrafficked interrow	0.08 (0.01)^{bC}	0.16 (0.00)^{aB}	0.16 (0.01)^{aC}
Trafficked interrow	0.09 (0.01)^{bC}	0.15 (0.00)^{aB}	0.16 (0.01)^{aC}
CRP	0.22 (0.01)^{bA}	0.27 (0.01)^{abA}	0.30 (0.02)^{aA}
Silver maple[1]	0.21 (0.02)^{aAB}	0.14 (0.02)^{aB}	0.20 (0.02)^{aBC}
Cool-season grass	0.16 (0.02)^{bB}	0.20 (0.01)^{abB}	0.26 (0.03)^{aA}
Switchgrass	0.16 (0.00)^{bB}	0.15 (0.02)^{bB}	0.28 (0.02)^{aAB}

[1]Normality test on data failed ($P < 0.050$). Kruskal-Wallis One-Way ANOVA on Ranks ($H = 2.489$ with 2 degrees of freedom. P(exact)= 0.339).

Table 3.3. Average percent ground cover and shrub canopy cover in a 1 m^2 area within each vegetation type. The 'Total Ground Cover' category is the sum of all categories except 'Bare Ground'. The 'Rooted Ground Cover' category is the sum of all categories except 'Litter', 'Standing Dead Woody Material', and 'Bare Ground'. Mean values from 3 plots collected in June/July 2013.

Vegetation Type	Warm-Season Grasses (%)	Cool-Season Grasses (%)	Leguminous Forbs (%)	Non-Leguminous Forbs (%)	Live Woody Shrubs (%)	Litter (%)	Standing Dead Woody Material (%)	Bare Ground (%)	Total Ground Cover (%)	Rooted Ground Cover (%)
Untrafficked interrow	0.0	0.2	16.0	0.2	0.0	2.2	0.0	88.7	18.5	16.3
Trafficked interrow	0.0	0.5	16.0	0.2	0.0	2.2	0.0	85.0	18.8	16.7
CRP	0.2	88.7	0.2	11.7	0.0	21.3	0.2	5.7	122.2	100.7
Silver maple	2.0	14.7	0.0	54.7	6.5	10.8	0.2	23.3	88.8	77.8
Cool-season grass	0.0	96.0	0.0	1.3	0.0	23.0	0.0	0.2	120.3	97.3
Switchgrass	0.5	92.3	5.3	1.2	0.0	21.3	0.0	5.7	120.7	99.3

Table 3.4. The number of infiltration trials performed on each vegetation type, and the number of times runoff actually occurred.

Vegetation Type	n	Times runoff occurred
Trafficked interrow	10	10
Untrafficked interrow	9	8
Switchgrass	9	5
Silver maple	9	4
Cool-season grass	9	4
CRP	9	0
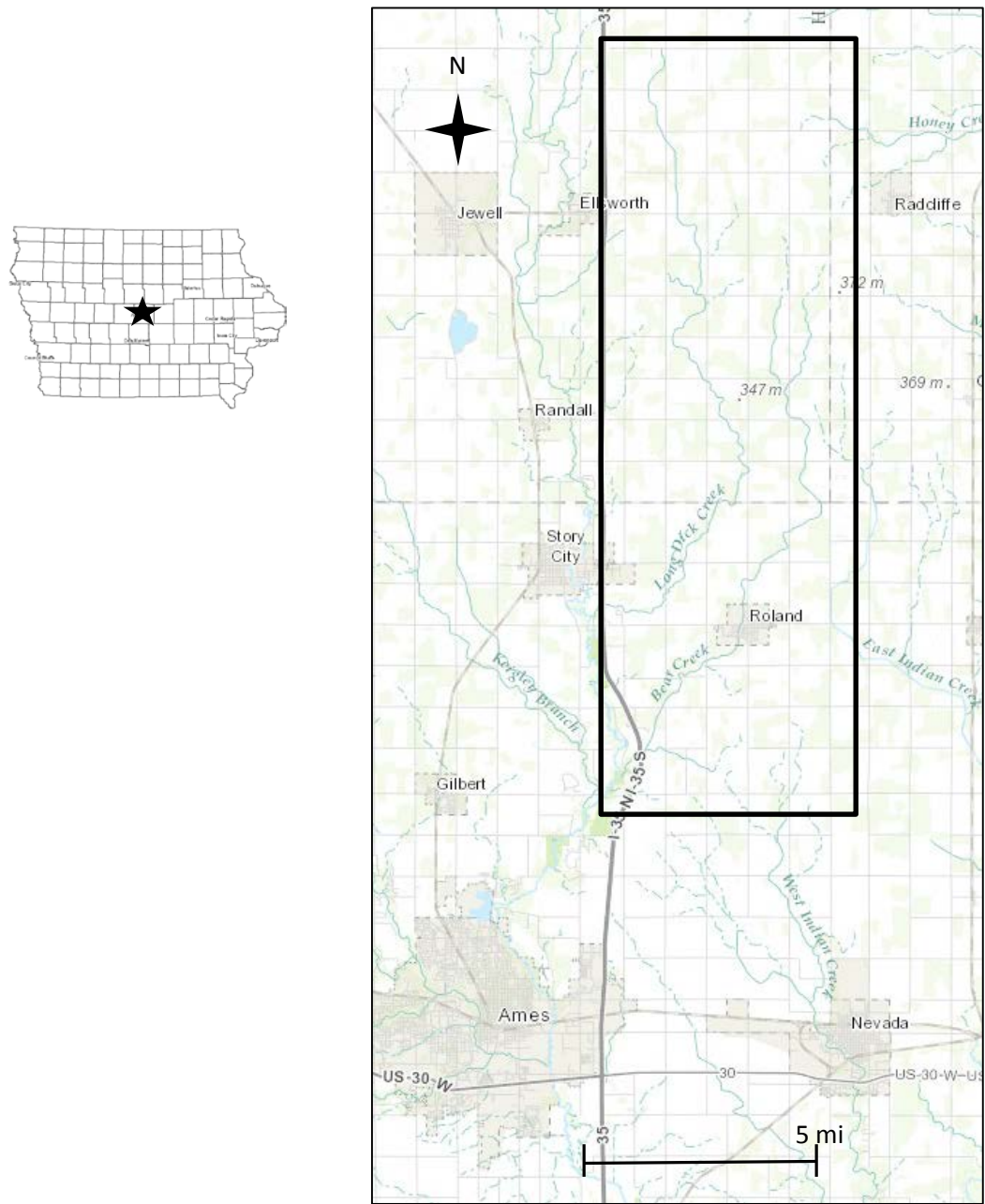
Figure 3.1. Location of Bear Creek and Long Dick Creek watersheds in Iowa, USA.

Figure 3.2. Cornell Sprinkle Infiltrometer
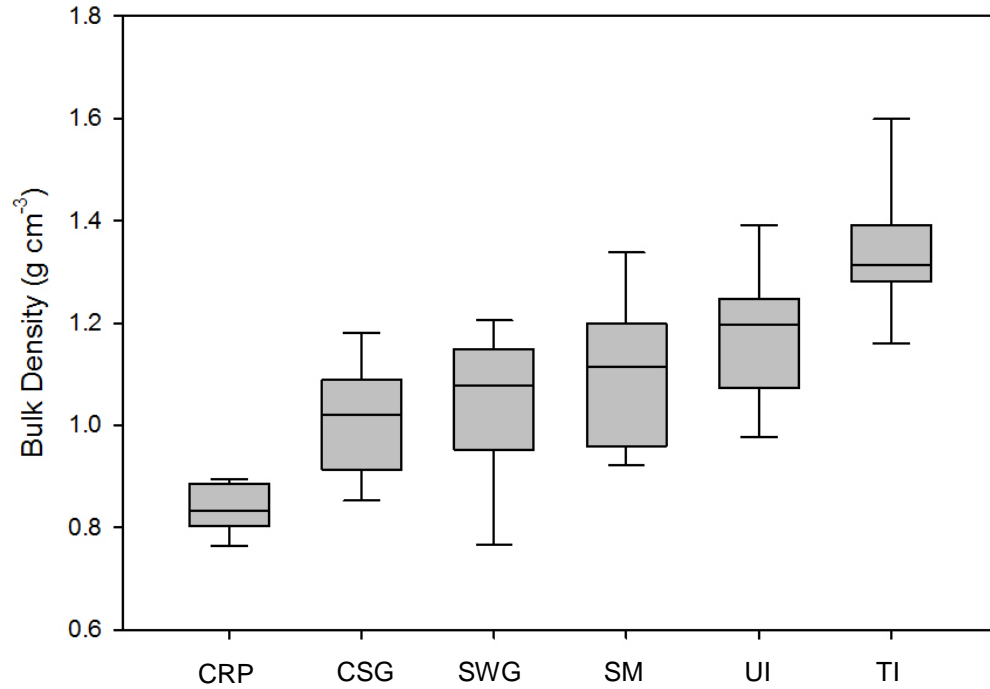
Figure 3.3. Mean (n=9) bulk density in the surface 7.5 cm of soil for four MRB vegetation types and two row crop treatments. Vegetation types: 10-year old CRP (CRP), cool-season grass (CSG), switchgrass (SWG), silver maple (SM), row crop untrafficked interrow (UI), row crop trafficked interrow (TI).
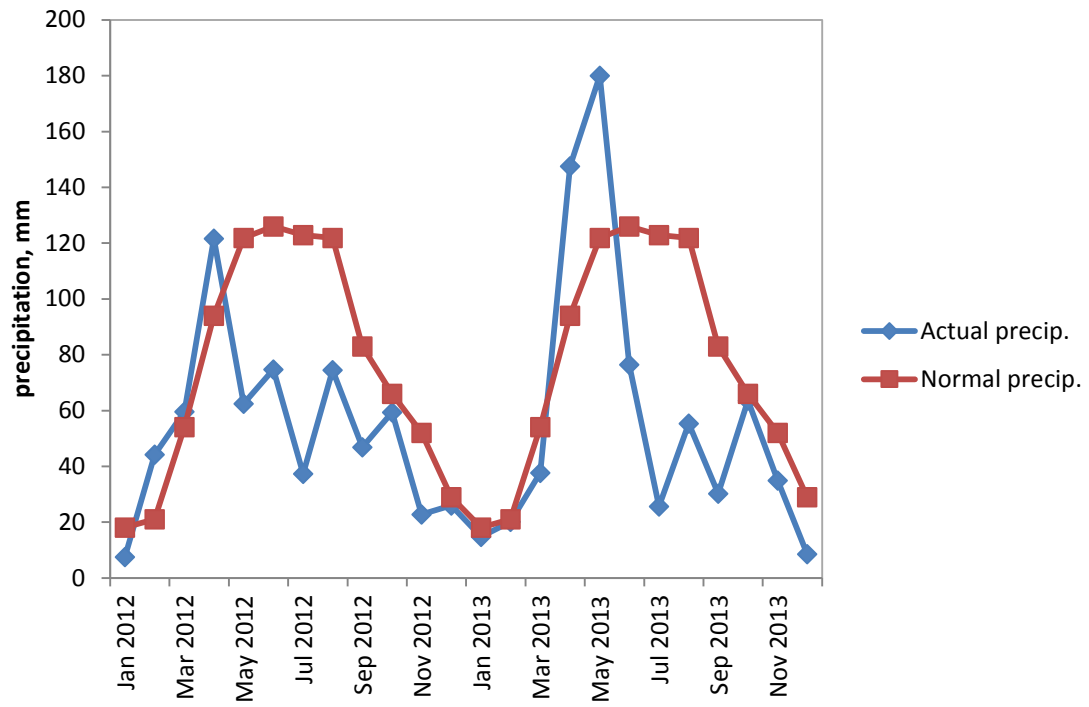
Figure 3.4. Actual monthly precipitation values in 2012-2013 for Ames, IA compared to normal precipitation.

Figure 3.5. Crop growth stage and presence or absence of canopy cover in July 2012, when the row crop field was in corn (A), in Sept. 2012 following corn harvest (B), and July 2013, when the row crop field was in soybeans (C). Photo credits: Angela Stone (A), Richard Schultz (B).
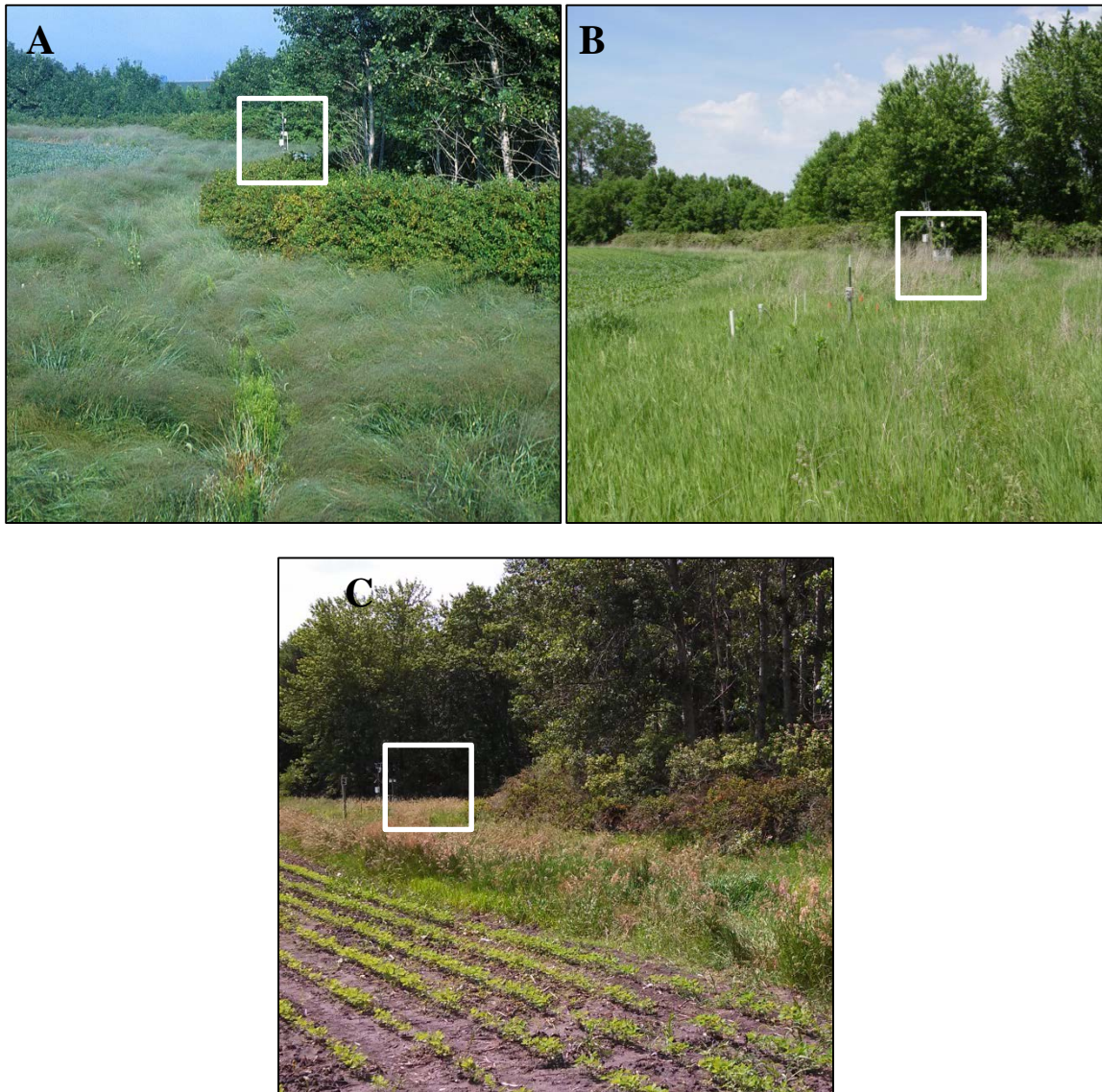
Figure 3.6. Switchgrass (*Panicum virgatum* L. 'Cave-in-Rock') stand decline over ~15 years. Note the weather station in the background (highlighted by the white box). Photo A (photo credit: R. Schultz) was taken in the late 1990's-early 2000's; the switchgrass stand is the predominant plant species present. Photo B (photo credit: D.G. Kim) was taken June 1, 2006; cool-season grass species are now predominant, but switchgrass is still evident to a small degree by the presence of dormant switchgrass stalks. Photo C was taken July 3, 2013; no clear evidence of switchgrass presence remains in this photo, and only a trace of switchgrass was evident in the visual vegetation surveys (Table 3.3).
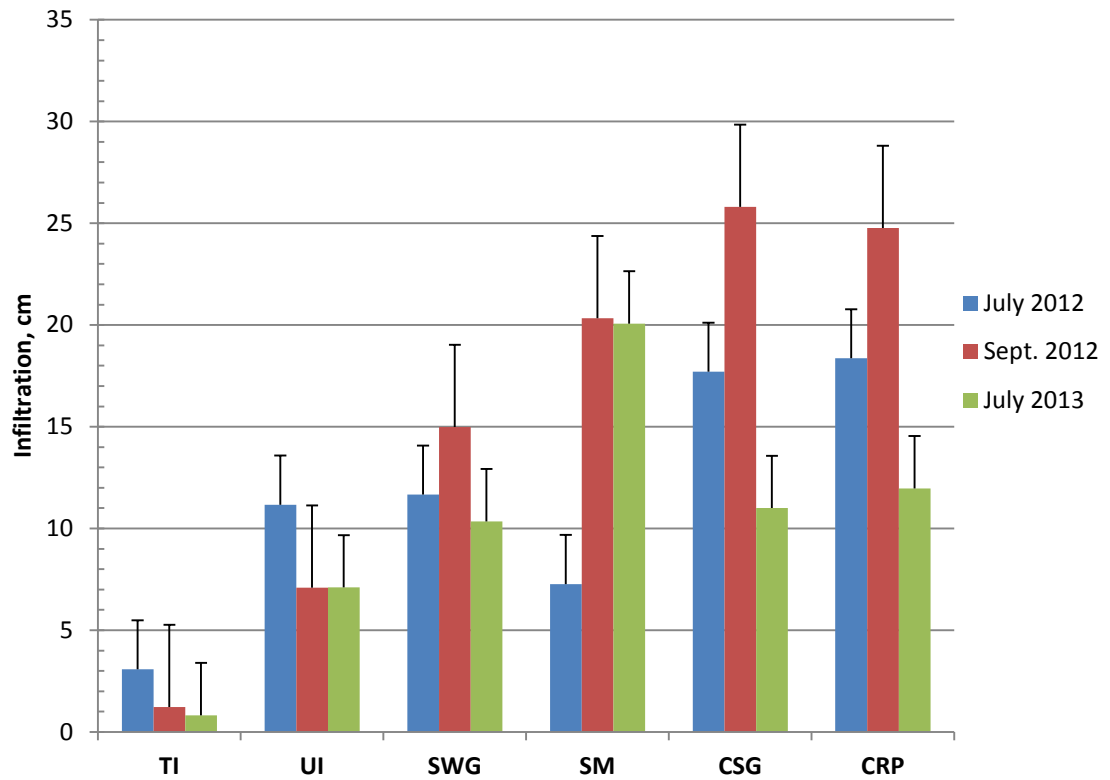
Figure 3.7. Mean (n=3, except for Sept. 2012 TI and UI, where n=4) actual and estimated 60-min cumulative infiltration for four MRB vegetation types and two row crop treatments. The error bars indicate standard error. Vegetation types: Trafficked interrow (TI), untrafficked interrow (UI), switchgrass (SWG), cool-season grass (CSG), 10-year old CRP (CRP), silver maple (SM).
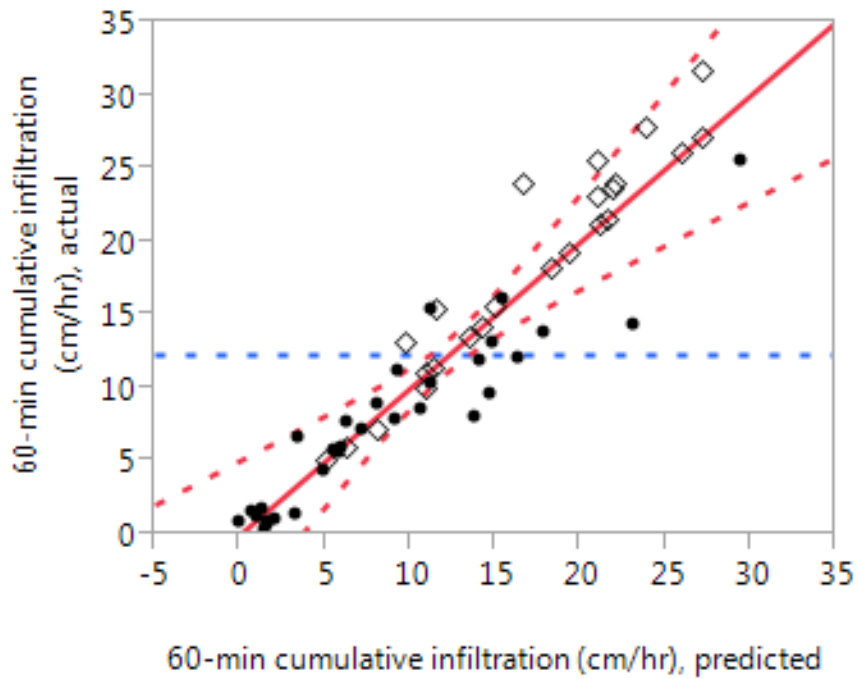
Figure 3.8. Actual versus predicted values derived from fitting a full-factorial least-squares equation to the actual (solid black dots) and estimated (open diamonds) 60 minute cumulative infiltration across all vegetation types, rainfall rates, and % saturation. $p <$ 0.0001, adjusted $r^2 = 0.82$, RMSE = 3.5658, n = 55.
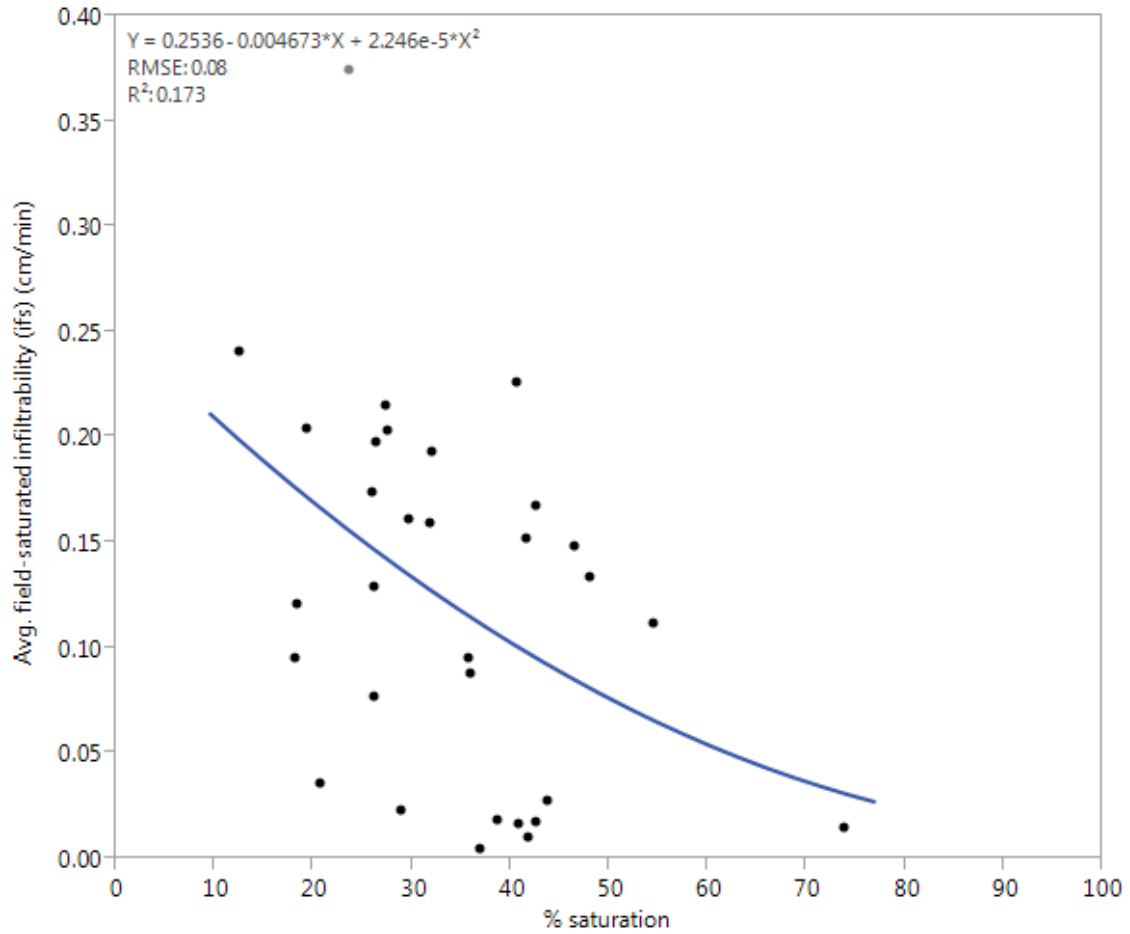
Figure 3.9. Regression of average steady-state field-saturated infiltrability (i_{fs}) (cm/min) versus percent saturation in the soil. Values are included only for plots in which runoff occurred (n=32).
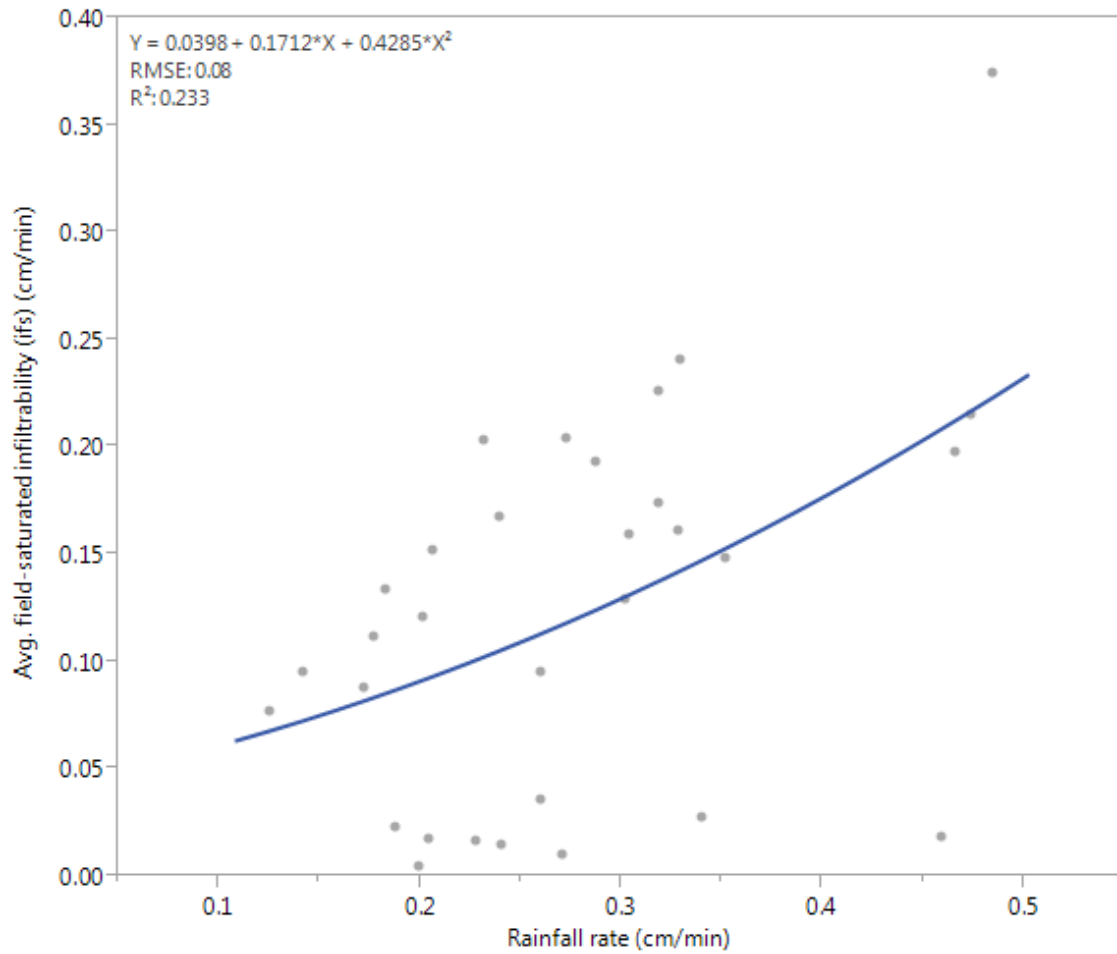
Figure 3.10. Regression of average steady-state field-saturated infiltrability (i_{fs}) (cm/min) versus rainfall rate (cm/min) applied. Values are included only for plots in which runoff occurred (n=33).
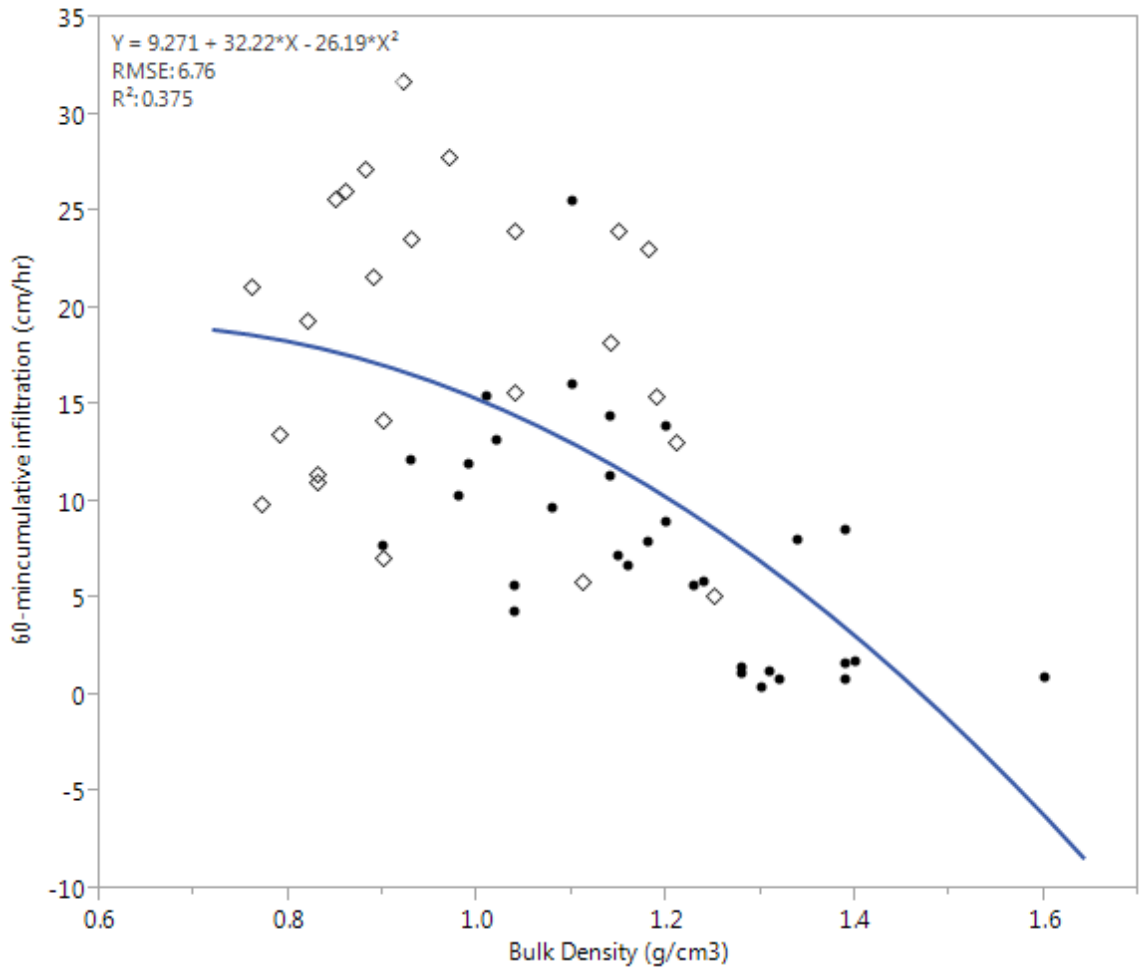
Figure 3.11. Regression of actual (solid black dots) and estimated (open diamonds) 60-min cumulative infiltration (cm min^{-1}) versus bulk density (g cm^{-3}).
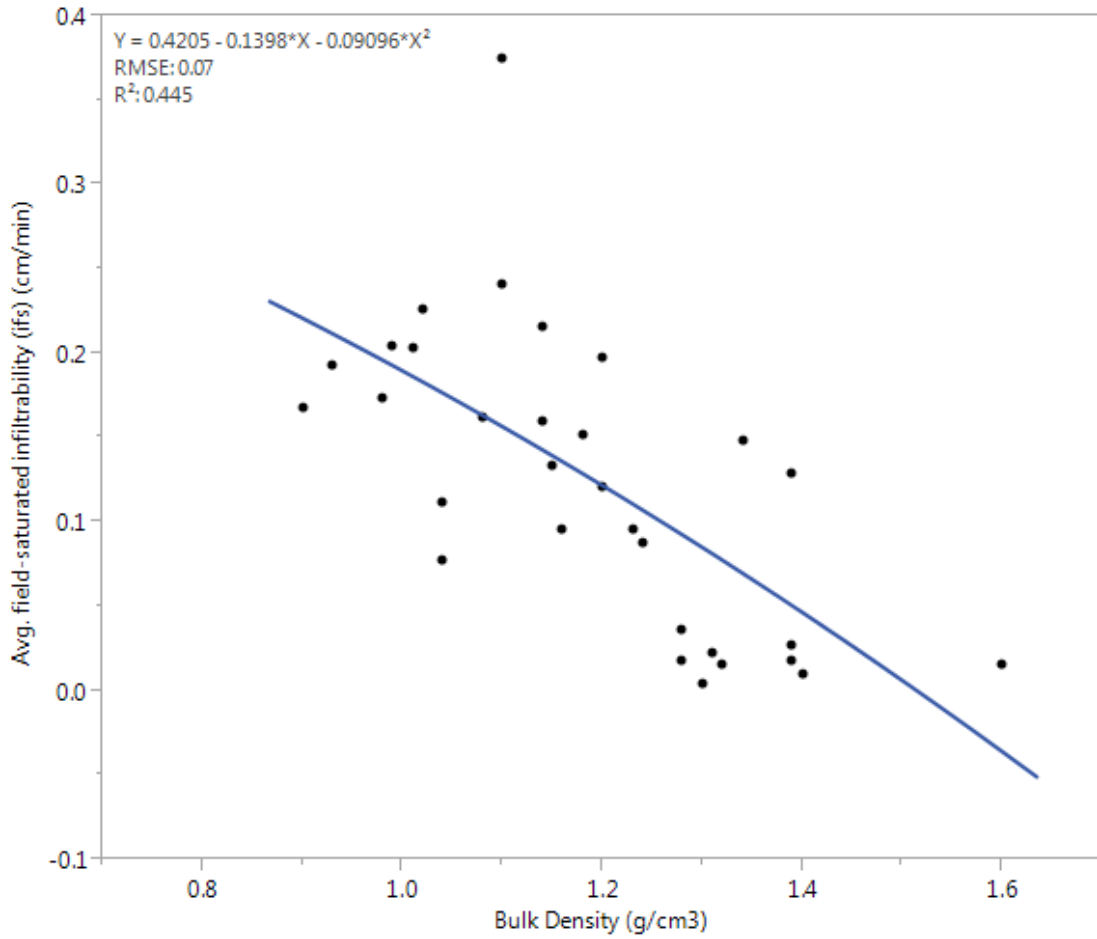
Figure 3.12. Regression of average steady-state field-saturated infiltrability (i_{fs}) (cm/min) versus percent saturation in the soil. Values are included only for plots in which runoff occurred (n=32).
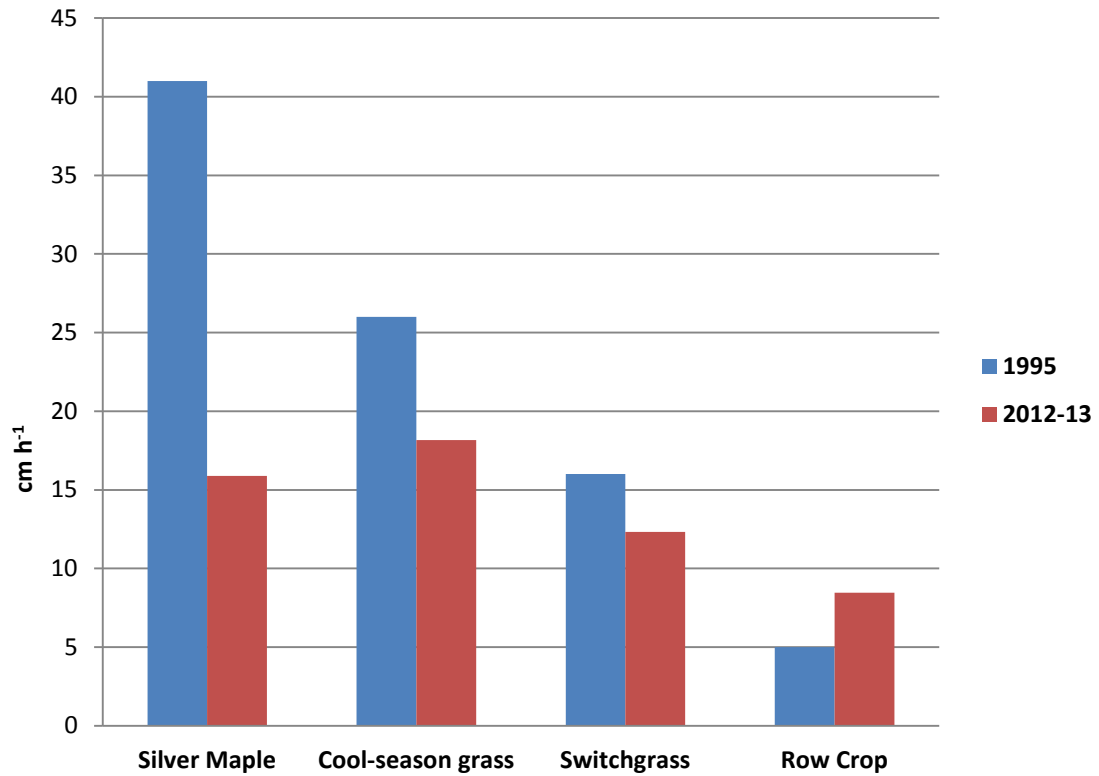
Figure 3.13. Comparison of 60-minute cumulative infiltration measured in 1995 in the Bear Creek MRB silver maple, cool-season grass, and switchgrass zones, and adjacent row crop field by Bharati et al. (2002), using double-ring infiltrometers, and in 2012-2013 using Cornell Sprinkle infiltrometers.

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CHAPTER 4. GENERAL DISCUSSION AND CONCLUSIONS

Any perennial vegetation present, whether native or non-native; natural or reintroduced; grass, tree, or forb; improves soil aggregate stability, total SOC and POM, and water infiltration, primarily through lack of disturbance by tillage. However, disturbance through heavy foot traffic in perennial vegetation can have negative impacts on soil quality parameters, though not as negative as semi-annual tillage. The underlying soil factors of soil texture, bulk density, porosity, and initial water content are linked and drive the soil aggregation, carbon sequestration, and infiltration processes. Vegetation is influenced by the underlying soil factors, but vegetation type (whether C_3 or C_4 type) may affect the rate at which carbon is sequestered in the soil and thus the soil aggregation processes.

Multi-species riparian buffers show initial improvements in soil quality parameters after only 7 years, but have the capacity to continue to improve the soil quality parameters of soil aggregate stability and particulate organic matter even twenty years after establishment, even though the length of a CRP contract for an MRB is commonly 10-15 years. Native warm-season grasses may take longer to establish and improve soil quality, although the other benefits of native warm-season grasses in the short-term, such as improved erosion control (Lee et al., 1998) and wildlife habitat (Berges et al., 2010) should not be overlooked.

After 20-21 years, cool-season grasses have slightly higher %WSA, larger GMD and MWD and more total POM than the warm-season grasses. All these parameters were greater than that found under switchgrass. However, it was noted that soils under the

former switchgrass buffer have made significant increases in the amount of water-stable aggregates present in the past 14 years; whereas only seven years after MRB establishment (Marquez et al., 1999; 2004) it was evident that soils under switchgrass were not significantly different in aggregate stability than the adjacent crop fields.

Trees in MRBs, whether naturally-occurring or reintroduced, seem to favor the production of small macroaggregates over large macroaggregates, but the implications of this finding are unclear. However, the %WSA under hybrid poplar with a cool-season grass understory tends to be greater than under natural forest, and is similar to cool-season grasses alone. Periodic maintenance of riparian forest buffers to allow an adequate stocking rate to permit understory growth to occur may be necessary, as we noted a decrease in infiltration capacity under silver maple which has lost its grass understory in exchange for a more diverse understory of forbs. Further work remains to be done exploring forest understory structure in relation to infiltration capacity, POM and aggregate stability.

Various methods of measuring infiltration make comparisons between research studies challenging. Therefore, statistical comparisons were not able to be made to the earlier infiltration study carried out by Bharati et al. (2002), although measurements were made in the same plots as this earlier study. Cumulative infiltration for switchgrass and cool-season grasses were similar from 1995 to 2012-13, but cumulative infiltration under silver maple was drastically reduced. This may be due to a loss of macropores from a loss in water-stable aggregates. Although mean %WSA was not significantly different between silver maple (%WSA = 62.3±7.2, n = 2) and cool-season grass (%WSA = 78.9±1.2, n = 12), cool-season grass does have higher %WSA. Means may have separated themselves if

more %WSA samples had been collected within the silver maple plots, as variability is high in the silver maple.

Natural forest understory vegetation, even when sparse, can promote high infiltration rates. Schoonover et al. 2005 noted that a 30-year-old mixed deciduous riparian forest with a sparse rooted ground cover and a discontinuous 1.7 cm thick litter layer covering 18% of the area, had mean cumulative infiltration rates of 50 cm h^{-1}.

Bare ground may possibly lead to the destruction of soil aggregates through raindrop impacts and lessen infiltration capacity. Litter cover in the silver maple plots was even less than this at 10.8%, but was highly variable among the three plots (SD = 8.9%), as was the amount of bare ground (23.3% ± 12.7) and live rooted vegetation (77.8% ± 11.9). Time of year also dictated the ground cover present in the silver maple; preliminary measurements taken in November 2012 found few forbs present (8.9%) and more litter (85.0%), but measurements taken in July 2013 found many more forbs (54.7%) and less litter (10.8%).

Perennial vegetation can create soil conditions favorable to high infiltration rates, higher than can be reasonably measured with the Cornell sprinkle infiltrometer. The maximum linear water holding capacity for the Cornell sprinkle infiltrometer is approximately 45 cm; other studies in perennial vegetation measured 60-min infiltration rates greater than this (Schoonover et al., 2006; Zwirtes et al., 2013). The need for larger amounts of water for infiltration measurements is offset by the challenge of accessibility in areas of perennial vegetation, such as in riparian forest buffers.

These results come from a single study with few replications, and do not allow a general statement to be made about different types of vegetation in MRBs. Results from

this particular study, however, do demonstrate that the soil quality parameters of soil aggregate stability, particulate organic matter, and infiltration capacity was greater in the MRB than in adjacent row crop soils.

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