Ecosystem services of riparian areas: stream bank stability and avian habitat

by

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A thesis submitted to the graduate faculty

in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

Major: Ecology and Evolutionary Biology

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Iowa State University

Ames, Iowa

2009

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Chapter 1. General Introduction

Project description and introduction

Riparian areas are transitional zones between terrestrial and aquatic ecosystems (National Research Council 2002). Riparian areas investigated in this thesis are transitions between row-cropped uplands and lower order streams. Riparian areas can serve many functions such as filtering pollutants from overland flow, stabilizing stream banks, storing surface water and sediment, and maintaining biodiversity by providing wildlife habitat (Naiman et al. 1993; National Research Council 2002; Schultz et al. 2004). This thesis focuses on two of these functions, stream bank stability and wildlife habitat.

Since Euro-American settlement, much of the land in the U.S. Midwest has been converted from native plant communities to row crop or pasture agriculture. To maximize land in cultivation, riparian areas were often heavily manipulated and converted for agricultural use (Brinson et al. 1981; National Research Council 2002). Riparian areas were frequently converted to agriculture in Iowa and Missouri, but in Missouri, farmers have often left a thin, unmanaged strip of vegetation (5+ m) in the riparian areas (Herring et al. 2006). This dramatic change in plant communities, both spatially and temporally, has had dramatic effects on the ecology of these areas (Stauffer and Best 1980; Naiman and Decamps 1997). The removal of perennial vegetation and conversion to annual row-crop agriculture has dramatically reduced farmland biodiversity (Benton et al. 2003). It also has reduced soil organic matter, structure and porosity (Marquez et al. 1999; Tan 2009). Cultivation has also reduced infiltration which has increased overland flow (Bharati et al. 2002; Tan 2009). The removal of perennial vegetation in riparian areas also removes the structural functions associated with the vegetation, including potential reductions in stream bank stability and reduced amount and heterogeneity of wildlife habitat for many bird species.

Stream bank erosion is a natural process that can be exacerbated by changes in land use and increases in stream power (Zaimes et al. 2006, 2008; Clark et al. 1985). Sediment pollutes rivers, lakes, and reservoirs and costs billions of dollars annually to clean up (Holmes 1988). Other pollutants, such as phosphorus, commonly bind to sediment. Due to the sedimentation of surface waters in the US, conservation practices, such as riparian buffers, have been promoted with the primary goal of reducing contaminant loads. A secondary goal associated with these conservation practices is the creation of wildlife habitat.

The loss of habitat due to agricultural intensification has affected many bird species, especially grassland species that once flourished in the vast prairie that once covered Iowa. Riparian habitats can be critical to wildlife (Stauffer and Best 1980) as they provide diverse and complex terrestrial habitats, especially in areas dominated by intensive agriculture (Naiman et al. 1993).

This thesis describes two separate studies to assess ecosystem services provided by remnant or re-established riparian vegetation. In the first study, stream bank erosion was measured through the use of erosion pins on first and second order streams in northeast Missouri. We compared stream bank erosion on stream reaches with different riparian land use/vegetation. The vegetation in these riparian zones was naturally occurring and unmanaged. The second study compared avian communities within three different aged riparian buffers with those of the nearby matrix land cover types (row crop fields and intensively grazed pasture) through the use of bird surveys. The planted riparian buffers in this study were located in north-central Iowa and consisted of three zones; tree, shrub, and grass/forb. They were designed and managed to have high habitat heterogeneity.

Study locations

The study reported in chapter two that measuring stream bank erosion was conducted along first and second order streams located in two subwatersheds of the Mark Twain Lake/Salt River watershed in northeast Missouri. Crooked Creek (28,814 ha) and Otter Creek (26,709 ha) watersheds are subwatersheds with 4^{th} order streams (Strahler 1957). The major water quality concerns for the Mark Twain Lake/Salt River watershed are sedimentation and turbidity due to severe soil erosion from intensely cultivated land (Dames and Todd 2009). Mark Twain Lake and 63 km of the Salt River are designated as public drinking water supplies for a 12 county area (Missouri DNR 1986; Fletcher and Davis 2005).

Crooked and Otter Creek watersheds are representative of the intensively row cropped watersheds of the Claypan Prairie subregion of the Central Irregular Plains ecoregion (Chapman et al. 2002; Lerch and Blanchard 2003; Lerch et al. 2008). The land-use for the watersheds is dominated by cropland (corn [*Zea mays* L.], soybeans [*Glycine max* L. (Merr.)], winter wheat [*Triticum asetivumi*] and sorghum [*Sorghum bicolor*]), pasture, and forest in descending order (Lerch et al. 2008). The soils in this region have a well-developed claypan that ranges from 10-80 cm below the surface (Udawatta et al. 2004). The claypan causes low infiltration rates and high surface runoff. It also leads to a perched water table and lateral flow above the claypan (Jamison and Peters 1967; Blanco-Canqui et al. 2002).

Previous studies have been conducted in the same area to examine different aspects of the existing riparian vegetation. Herring et al. (2006) used a Geographic Information System (GIS) to assess the length, width, and vegetation type of existing riparian buffers in Crooked, Otter, and Long Branch Creek watersheds. Knight (2007) investigated the ability of natural forest buffers and grass filter strips along streams to buffer concentrated flow paths (CFPs). These two studies showed us the extent of the existing vegetation and the ability of two types of vegetation to buffer overland sources of sediment. This study uses some of the same sites and focuses on measuring stream bank erosion as a sediment source in these streams. Twenty-four sites were included in the study, including four different land-uses (crop, riparian forest, forest, and pasture) along first and second order streams.

The study reported in chapter three compares bird species diversity among a chronosequence of buffers and the surrounding matrix was conducted along Bear Creek and Long Dick (L.D.) Creek in north-central Iowa. Land-use surrounding these creeks is dominated by row crop agriculture (corn and soybeans) and pasture. Bear Creek (6,941 ha) and L.D. Creek (9,401 ha) watersheds are small drainage basins located within the Des Moines Lobe subregion of the Western Corn Belt Plains ecoregion (Griffith et al. 1994). In general, the topography of this area is flat to gently rolling.

Five sites were included in the study, including three different areas with riparian buffers, an area with intensively grazed riparian pasture, and a riparian area in row crops. The riparian buffer sites represent three different ages since planting: 14-18 years (hereafter 14+), 9 years, and 2 years. All buffers were composed of three zones; a managed tree zone adjacent to the stream followed by a shrub zone and a native grass/forb zone adjacent to the crop field (Schultz et al. 2004). The 14+ year-old buffer was planted to approximately 18 m on each side of the stream; however, there is a large center section between the original channel and a newly forming channel covered by native warm-season grass, forbs, and invading small willows. The combined width of this buffer is close to 100 m in some locations. The 9- and 2-year old buffers were planted to approximately 46 m on either side of the stream for a total width of 92 m. The oldest site is a contiguous section consisting of a downstream segment planted in 1990 and an upstream segment planted in 1994; thus, vegetation on the entire site was at least 14 years old at the time of the study. The intensively grazed pasture was dominated by short bluegrass (*Poa pratensis*), with one large silver maple and a few small shrubs present near the stream. The pasture site was bordered on the outside (upslope) edge by tall, unmowed grass at a distance of 25-100+ m from the stream. Within the row crop area, crops were planted as close as possible to the stream edge. As the stream was meandering, this left area on the inside of the meanders that could not be farmed (<2-30 m wide on each side of the stream) which were dominated by cool-season grasses and forbs. Bird surveys were conducted using 10 minute point counts with a 50 m recording radius (Ralph et al. 1995). Sites were surveyed eight times between May 15 and July 10, 2008.

Thesis organization

This thesis is arranged into four chapters. This first chapter is an introduction to the topics covered and the study areas. The second chapter is entitled "Stream bank sediment and phosphorus losses under different land-use practices in northeast Missouri", and has been prepared for submission to a peer-reviewed journal. It compares soil erosion and phosphorus losses under different land-uses on first and second order streams. The third chapter is entitled "Bird species diversity in riparian buffers, row crop fields, and grazed pastures within agriculturally dominated watersheds" and has been submitted to Agroforestry Systems. The final chapter presents general conclusions based on the results of the work conducted for this thesis.

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Chapter 2. Stream bank sediment and phosphorus losses under different land-use practices in northeast Missouri

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Introduction

Sediment is one of the most prevalent and significant pollutants in the surface waters of North America (Simon and Darby 1999). The annual costs of the physical, chemical, and biological damage due to sediment pollution in North America alone approaches US$16 billion (Osterkamp et al. 1998, Pons 2003). According to the US Environmental Protection Agency (USEPA), sediment and nutrients are two of the leading pollutants causing impairment in assessed rivers in the US (2000).

Excess suspended and bedded sediments (SABS) can degrade water quality for humans and other organisms. SABS increase turbidity which reduces sunlight penetration and dissolved oxygen production. It can damage fish gills as well as abrade fish eggs. It also fills in spaces between the gravel, cobble, and rocks of streambeds, thereby destroying habitat for benthic invertebrates and covering potential egg laying sites for fish. In addition, the water-storage capacity of reservoirs and lakes is reduced through sedimentation (Wesche and Isaak 1999; Clark et al. 1985), which shortens their lifespan and usefulness. Sedimentation can increase the cost of municipal water treatment by necessitating the removal of sediment and associated contaminants (Holmes 1988). SABS mitigation is estimated to cost between $458 million and $661 million annually in the US (Holmes 1988).

Nutrients, such as phosphorus, are some of the most significant contaminants associated with eroded soils (Clark et al. 1985). Because phosphorus often binds to sediment, specifically to smaller particles such as silts and clays (Sharpley and Smith 1990), reducing sediment movement to streams and other surface waters can also reduce phosphorus pollution of these waters. Phosphorus inputs to surface waters can accelerate eutrophication of water bodies (Sharpley et al. 2001). The USEPA (1996) has identified eutrophication as being one of the main causes of impaired surface water quality.

Of the 901,232 stream kilometers (of a total 5.6 million km) assessed in the US by the USEPA in 2004, 44% were identified as impaired or not supporting one or more of their designated uses (USEPA 2009). In Missouri in 2006, 53.2% of the 35,486 km of rivers and streams assessed were impaired. Some of the main causes of impairment of all of these streams in Missouri were SABS and organic enrichment (nutrients and low dissolved oxygen) (USEPA 2007a).

Sediment entering surface waters commonly originates from either upland or stream channel sources. Upland or surface erosion, such as sheet, rill, and gully erosion, can carry sediment overland to the streams. The other major potential source of sediment comes from stream bed or bank erosion (Clark et al. 1985; Walling 2005). Identifying the major source of sediment becomes important when trying to design and implement management practices to reduce sediment erosion and transport to streams and other surface waters (Walling 2005). Significant literature exists detailing the amount of surface erosion that can be delivered to streams, and both infield and riparian conservation practices exist that can dramatically reduce these inputs to streams (Dillaha et al. 1989; Lee et al. 1999, 2003; Dosskey et al. 2002). Much less is known about the mechanisms and quantity of sediment delivery from stream banks (Lawler et al. 1993, Zaimes et al. 2008). Stream bank erosion is a natural process but can be exacerbated by changes in land use and increases in runoff (Clark et al.

1985). It can be a major contributor of sediment to streams (Fox et al. 2007; Piercy and Wynn 2008). In the Midwestern United States, studies found that stream bank erosion contributed 31-44% of the sediment load to streams in Minnesota (Sekely et al. 2002), 45-80% of sediment load for Iowa rivers (Odgaard 1984, Wilson et al. 2008), and up to 50% in two Illinois streams (Wilkin and Hebel 1982). Although these studies estimate the percentage of sediment that enters streams from stream bank erosion, there is not yet a definitive way to quantify this contribution and much more work needs to be done to establish the role of bank erosion in the sedimentation of streams (Collins and Walling 2004). Fewer studies have estimated the stream total phosphorus load contributed by stream bank erosion (Sekley et al. 2002). Studies estimate that phosphorus contributions from stream bank erosion range from 7-10% to 55% in the United States (Sekley et al. 2002; Roseboom 1987). Laubel et al. (2003) found a similar range (15-40%) of phosphorus export from bank erosion in Denmark.

Stream bank stability is often decreased by the removal of stream-side perennial vegetation and stream bank trampling and entry by livestock. Vegetation directly protects soil surfaces by intercepting rainfall, promoting infiltration, and increasing resistance to erosion from flowing water (fluvial erosion) (Thorne 1990; Simon and Collison 2002). Vegetation also enhances stability of stream banks through root reinforcement (Abernethy and Rutherfurd 2000). Roots bind or restrain soil particles and create a strengthened and reinforced soil-root matrix (Gray and Leiser 1982; Simon and Collison 2002). Livestock grazing can reduce the vegetative cover and loosen and destabilize bank soils through trampling (Belsky et al. 1999; Trimble and Mendel 1995). Another major source of sediment in pasture streams comes from livestock access points to the stream and the erosion caused by livestock walking up and down the banks (Tufekcioglu 2006).

Many of the management practices suggested for controlling erosion and stabilizing stream banks include using vegetation such as riparian buffers or stream bank bioengineering (Bentrup 2008; Evette et al. 2009). Stream segments with woody vegetation, such as riparian forest buffers, generally have lower stream bank erosion when compared to those with herbaceous vegetation, row-crops, or grazed pastures (Micheli et al. 2004; Zaimes et al. 2004; Wynn and Mostaghimi 2006). Due to the high runoff potential of claypan soils, it has been assumed that the primary source of sediment in the streams of claypan watersheds comes from overland flow, and most studies have focused on mitigating upland erosion (Jamison et al. 1968; Ghidey and Alberts 1996). No previous studies have measured the amount of stream bank erosion in the Claypan Prairie region of northeastern Missouri (USEPA 2007b).

Accurately measuring stream bank erosion is difficult because of the need for a relatively fine resolution of measurement, the high spatial variability in erosion along banks, and the temporal variability in erosion, which can affect how frequently erosion is measured (Lawler 1993). Erosion pins are the most common technique for measuring stream bank erosion over short-time scales on lower order streams because they are cheap, easy to use, and suitable for a wide range of fluvial contexts (Lawler 1993). Some other methods for short-time scale measurements of stream bank erosion include terrestrial photogrammetry or Photo-Electronic Erosion Pin (PEEP) monitoring systems (Lawler 1993). The natural vegetation of the Claypan Prairie subregion of the Central Irregular Plains ecoregion included a mixture of tallgrass prairie, oak-hickory forests, savannas, and wetlands (Chapman et al.

2002; Ferguson 1995). Most of this natural vegetation has been cleared for agriculture (Lerch and Blanchard 2003). Herring et al. (2006) inventoried the width of perennial riparian vegetation along two streams within this ecoregion, Crooked and Otter Creeks, using a Geographic Information System (GIS). They found that for 1st order streams in the Crooked and Otter Creek watersheds, 85% and 76% of the stream length, respectively, had perennial vegetation to a distance of 15 m on each side of the stream. For the 2nd order streams, 93% and 88% of the stream length were buffered to 15 m. When the buffer width increased, the percent of stream length that was buffered decreased. For 1st order Crooked and Otter Creeks, 52% and 39% were buffered to 61m and similar numbers were observed on 2nd order streams. While this inventory identified the width of existing vegetation, it gave no indication of the quality of that vegetation or its ability to protect stream banks from erosion. Most of the vegetation was naturally occurring rather than planted as a designed buffer. Knight (2007) inventoried the number concentrated flow paths (CFPs) from crop fields that intersected remnant forests with or without adjacent grass filters and found that remnant forests dispersed the flow from 80% of the CFPs and grass filter strips dispersed 100% of CFPs. The CFPs that made it through the buffer to the stream all occurred in remnant forests without adjacent grass filters and with an average width of 13 m. His study found that the narrow remnant forests without grass filters can be ineffective at stopping concentrated flow of sediment from surface sources. However, he also noted that there seemed to be significant bank erosion along many of the streams surveyed. Because of the sediment problems in Mark Twain Lake, it is important to look at both surface and stream bank erosion. These two studies showed us the extent of the existing riparian vegetation in these watersheds and the ability of two types of vegetation to buffer overland sources of sediment. Zaimes et al.

(2008) found that there were significant differences in bank erosion in Iowa from different land uses with designed riparian buffers having the least erosion.

The overall goal of this study was to assess the effectiveness of different kinds of natural riparian vegetation to protect against stream bank erosion. Specific objectives were to quantify the amount of sediment and phosphorus contributed to the stream from stream bank erosion and to identify vegetative communities/land uses that provide the greatest protection from stream bank erosion. We hypothesized that deeper rooted riparian vegetation would provide greater protection against stream bank erosion than shallower rooted vegetation and that cattle would reduce stream bank stability. Results of this study will help gain a better understanding of the source of the sediment delivered Mark Twain Lake and to direct management to reduce these sediment loads.

Materials and Methods

Study area: The research was conducted in two sub-watersheds of the Mark Twain Lake/Salt River watershed in northeast Missouri (Figure 1). These same watersheds were the focus of studies by Herring et al. (2006) and Knight (2007). Crooked Creek (28,814 ha) and Otter Creek (26,709 ha) watersheds are sub-watersheds covering 4.8 percent and 4.6 percent of the Salt River watershed, respectively (Figure 1). Both streams are 4th order streams (Strahler 1957). The major water quality concerns for the Mark Twain Lake/Salt River watershed are sedimentation and turbidity due to severe soil erosion from intensely cultivated land (Dames and Todd 2009). Mark Twain Lake and 63 km of the Salt River are designated as public drinking water supplies for a 12 county area (Missouri DNR 1986; Fletcher and Davis 2005).

Crooked and Otter Creek watersheds are representative of the intensively row-cropped claypan watersheds of the Claypan Prairie subregion of the Central Irregular Plains ecoregion (Lerch and Blanchard 2003; Lerch et al. 2008, Chapman et al. 2002; USEPA 2007b). The land use in the Crooked Creek watershed is 56% cropland, 26.5% pasture, and 14.5% forest. The land use for the Otter Creek watershed is 64.6% cropland, 20.3% pasture, and 12.6% forest (Lerch et al. 2008). The primary crops grown in this region are corn [*Zea mays* L.], soybeans [*Glycine max* L. (Merr.)], winter wheat [*Triticum asetivumi*] and sorghum [*Sorghum bicolor*]), and the main livestock is beef cattle (Lerch et al. 2008). This region has gently rolling topography with well-developed claypan soils over glacial till (USEPA 2000). The depth to the claypan (argillic horizon) ranges from 10-80 cm (Udawatta et al. 2004). These soils have low infiltration rates and lead to a perched water table which creates lateral flow above the claypan (Jamison and Peters 1967; Blanco-Canqui et al. 2002). The dominant upland soils in the watersheds are Mexico silt loam and Lindley loam. Some of the dominant soils of the riparian zones are Armstrong loam, Leonard silt loam, and Arbela silt loam. The soils in this region are deep to very deep and are poorly drained (Watson 1979).

Riparian vegetation: Riparian land-uses compared in this study included riparian forest, forest, pasture, and crop. All sites had to have the same land-use on both sides of the stream for a length of at least 400 m to be included in this study. Riparian forests were defined as 15-30 m wide strips of naturally occurring trees and shrubs along both sides of the stream (Figure 2). Forests were defined as naturally occurring trees and shrubs with a width of greater than 30 m on both sides of the stream (Figure 3). Pasture sites were continuously grazed and had some scattered trees near the stream bank. The cattle had full access to the stream and the entire pasture throughout the grazing period, which was year-round in some cases (Figure 4). Crop sites were planted to either corn or soybeans in rotation. The crop sites usually had strips of grass, shrubs, and/or trees along the bank that were less than 15 m wide (Figure 5). Similar studies conducted in Iowa (Zaimes et al. 2008) had crop sites with corn or soybeans planted down to the edge of the stream bank, but most farmers in Missouri leave at least a narrow (5 m +) strip of naturally occurring grass, shrub, or tree vegetation along the banks (Herring et al. 2006). This study used some of the same study sites as Knight (2007) in assessing gully erosion.

There were a total of 24 treatment sites, all sites located on 1st and 2nd order streams (Strahler 1957). There were three replicates of each vegetation type on each stream order. The vegetation types were distributed throughout the two watersheds with 10 sites in the Crooked Creek watershed and 14 sites in the Otter Creek watershed (Figure 6). The two watersheds have similar area, land uses, and soil types and so were treated as one unit due to the strong similarities.

Measurement of stream bank erosion: Stream bank surveys were conducted to determine the location and length of severely to very severely eroding banks starting in March 2007 and continuing throughout the summer as sites were identified and permission received from landowners. Site locations were documented using GPS units (Trimble Juno ST using TerraSync Software version 3.01 and Trimble GeoExplorer3 using GeoExplorer Software, version 1.20 or Dell X51 with GlobalSat BC-337 Compact Flash GPS Receiver and Trac-Mate software Farm Works Software, Version 12.16). Each site had to fit the stated criteria: at least 400 m long, the same land use on both sides of the stream, and with riparian forest, forest, pasture or crop as the land use. Sites were initially assigned to a land use category based on aerial photographs but were reassigned to a different category, if necessary, once they were ground-truthed, often due to the width of riparian vegetation. More than the 24 reaches were initially surveyed, but some sites were later dropped because they were found to be unsuitable based on the established criteria.

Eroding banks were identified based on criteria established by the Natural Resources Conservation Service (USDA – NRCS 1998a, 1998b). Banks that were identified as being severely or very severely eroding possessed one or more of the following characteristics (Figure 7 a, b):

- 2/3 of the bank bare of vegetation or roots;
- less than 1/3 of the eroding surface area of banks protected by roots that extend to the baseflow elevation;
- severe vegetative overhang - bank underneath is eroded away;
- many exposed tree roots – but if covered by many smaller tree roots ¼ in to ½ in diameter then not severe;
- mature trees falling into stream annually (must be annually – a single tree fallen into the stream does not necessarily indicate severe erosion); and/or
- numerous slope failures apparent (if slump is stable – major slab resting at base of bank with permanent vegetation cover – then not severe).

GPS data points were taken at the upstream, middle, and downstream ends of eroding reaches (Figure 7a) as well as at debris dams, livestock access points and gullies. The number of livestock access points and gullies per km of bank was later determined from the GPS data using a GIS. Other field-based data collected included bank height at the same locations as the GPS points for the eroding banks, the location of migrating knickpoints, and the location of any other important sites such as large trash piles in or near the stream.

Following the field survey, the location and length of eroding and non-eroding bank lengths was determined using GPS data from the survey and the total eroding length for each site was calculated. A 20 percent subset of the eroded bank lengths at each site was randomly selected on which to measure stream bank erosion. Erosion was measured across the entire eroding bank section using pin plots consisting of 1-3 rows of erosion pins. Erosion pins are commonly used to measure stream bank erosion rates for many different types of bank materials and sizes of streams, especially over short time-scales (Lawler 1993). They are steel rods that are 762 mm long and 6.2 mm in diameter. Erosion pins were inserted perpendicular to the bank face with 10.2 cm exposed. The exposed pin length was spray painted with fluorescent pink or orange paint to aid in subsequent location. A fiberglass rod was installed on top of the bank on the downstream end of the site to aid in locating the start of the plot (Figure 7a). Pin plots were only installed in severely and very severely eroding banks because they are the major sources of sediment in the channel when compared to less severely eroding lengths (Zaimes et al. 2004, Beeson and Doyle, 1995).

The number of pin plots per site varied because of differences in the total lengths of eroding banks. Pin plots made up 20% of the eroding length for each site. The number of pin plots on a site varied with a minimum of three and a maximum of ten. The arrangement of pins within each pin plot was based on the bank height (Figure 8). Banks with heights less than 1 m received 1 row of pins at 1/2 bank height. Banks between 1 and 2 m tall received 2 rows of pins at 1/3 and 2/3 bank height. Banks over 2 m tall received 3 rows of pins at 1/4, 1/2, and 3/4 bank height (Figure 9). Horizontal spacing between pins was 2 m. The height and length of each pin plot was measured using scaled height poles and 50 m measuring tapes to determine the area of bank represented by each pin plot.

GPS coordinates of the downstream end of each pin plot were recorded to aid in locating the plots. Sites were established beginning in May 2007, and most of the sites were pinned by October 2007. The date of site pinning was dependent on having permission from the landowner and having completed a stream bank survey.

Pin monitoring commenced in November 2007. At each subsequent measuring date, pins were measured and reset to 10.2 cm if greater than this value and left alone if less than 10.2 cm was exposed (Hooke 1979). If less than 4 cm was visible, a new pin was installed next to the original one to reduce the likelihood that the pin would be buried by the time of the next measurement. If a pin appeared to have fallen out due to erosion, a value of 65 cm was used (Lawler 1993). Pins were measured in March 2008, August 2008, November 2008, and March 2009. Thus the erosion pin data represents four time periods: November 2007 – March 2008 (first winter); March 2008 – August 2008 (spring to mid-summer); August 2008 – November 2008 (mid-summer to autumn); and November 2008 – March 2009 (second winter). Figure 7b shows an example of an eroded pin and a ruler used for measuring the pins.

Soil sampling and analysis: Soil samples were collected over the summer and early fall of 2008 for use in determining soil bulk density, total phosphorus, and particle size analysis (PSA). Fifty percent or a minimum of three of the pin plots were sampled at each of the treatment sites. Samples were taken in one column from each of the major stratified bank layers (Odgaard 1984) just outside the plot boundaries or in between two of the columns of pins. Soil cores for bulk density were taken using sample rings of 7.5 cm diameter. A total of 404 soil samples were collected from the banks and kept separate for analysis. Additional samples were collected from each layer for use in phosphorus and PSA analysis, although the results for the latter will not be reported in this paper.

Bulk density samples were dried in an oven at 110°C for a minimum of 3 days, brought to room temperature in a desiccator, and weighed. Bulk density was calculated by dividing the dry soil weight by the volume of the soil core (Blake and Hate 1986). To obtain a depth-weighted average for each treatment site, bulk densities at each pin plot site were weighted based on the depth (height) of each stratified bank layer in proportion to the total bank height for each sampled pin plot and then averaged over the number of pin plots sampled on the treatment site. Total phosphorus was determined using an alkaline oxidation method developed by Dick and Tabatabai (1977). Soil samples were air dried and sieved through a 2 mm screen. Samples were digested with a sodium hypobromite solution and the extracted phosphorus was quantified colorimetrically by a modified molybdenum blue reaction (Murphy and Riley 1962).

Quantification of stream bank soil and phosphorus losses: Erosion pin measurements were used to calculate two aspects of stream bank erosion among land-uses, erosion rate and erosion activity. To determine the erosion rate, the difference between the most recent measurement and the past measurement was calculated. These values could be positive or negative with a positive value indicating erosion and a negative value indicating deposition. Erosion activity was calculated using the absolute values of the differences between erosion pin measurements. This gives an estimate of total soil movement on the bank, whether the pin measurements were negative (deposition) or positive (erosion) (Couper et al. 2002). The average erosion rate and activity were calculated for each pin plot. The average stream bank erosion rate (cm) and activity (cm) for a land-use were estimated by averaging the erosion rate and erosion activity for the pin plots in each specific land-use for each measuring period.

The total mass of stream bank soil loss for each site was calculated by using the average pin length per pin plot (erosion rate in m), the depth-weighted bulk density average (kg/m^3), and the total eroded area of each plot (m^2). Stream bank soil loss per unit length of stream bank was estimated by dividing the total stream bank soil loss for each treatment site by its total stream bank length including both banks (usually 800 m). Soil loss was then averaged across sites for each treatment. Soil loss was calculated using the following equations:

Average linear erosion rate (kg/m) per pin plot = (Average erosion rate (m) * Plot area (m^2) * Depth weighted bulk density (kg/m^3))/Pin plot length

Erosion rate (kg/m or tonnes/km) per length of stream reach = (Average linear erosion rate (kg/m) * total eroded length (m))/ (total length of stream reach)

Phosphorus losses were calculated by multiplying the total soil loss by the depth-weighted average phosphorus concentrations for each site. Mean phosphorus concentrations were then averaged across each treatment.

Stream discharge and air temperature: Discharge data for Crooked Creek was obtained from the US Geological Survey (2009) for the study period, November 2008 – March 2009. The data includes the average daily discharge from Crooked Creek near Paris, Missouri, which is a fourth order stream. We expect similar responses from the streams that we measured because of similar watershed size and shape and similar land-use. Air temperatures for Shelbina, Missouri were obtained from the National Climatic Data Center (NCDC 2009) for the entire period. The number of days in which the temperature never rose above freezing and the number of days in which the daily mean temperature was below freezing were calculated from the NCDC dataset.

Data analysis: Statistical analyses on the erosion rate, erosion activity, and soil loss data, and phosphorus data were run using the mixed effect model Proc MIXED (SAS Institute 2003). Comparisons were made for land use within each stream order and time period group, for time periods within each stream order and land use, and between the two stream orders as a whole. A Bonferroni adjustment was used on the significance levels. The Bonferroni-adjusted significance level for the land-use and time period analysis was $P = 0.1/6 = 0.0167$. The Bonferroni method is considered to be conservative (Westfall and Wolfinger 1997). The $P=0.1$ rather than $P=0.05$ was used due to the complexity of stream bank erosion and the many variables that can affect it.

Results

Erosion rates: Stream bank erosion rate varied throughout the year and between land uses but was not significantly related to land use. The only significant difference between land-uses on first order sites was observed during the first winter season (November 2007 – March 2008) where the crop sites were significantly greater than the riparian forest sites ($P=0.0104$) (Table 1). There were some seasonal trends observed within each land use. The highest average erosion rate generally was observed during the winter season (the end of November to mid-March) (Figure 10a). On the crop sites, the first winter had significantly higher average erosion (12.7 cm) than all other seasons (spring – mid-summer, -1 cm, $P= 0.001$;

mid-summer – autumn, 0.5 cm, $P<0.0001$; winter 2, 5.6 cm, $P=0.0021$). The second winter was also significantly higher than the mid-summer – autumn ($P=0.0104$). On the pasture sites, the mid-summer – autumn (0.6 cm) was significantly less than the first winter (5.6 cm, $P=0.0115$) and the second winter (7.3 cm, $P=0.0012$). On the riparian forest sites, the second winter (7.4 cm) was significantly greater than the mid-summer – autumn (0.7 cm, $P=0.0014$). On the forest sites, the second winter (5.8 cm) was once again higher than the mid-summer – autumn (0.6 cm, $P=0.0076$). During spring - mid-summer the average erosion rate on the crop sites was -0.9 cm indicating that deposition exceeded erosion. The yearly erosion rates were similar among the first order crop, pasture, and riparian forest sites (12.9, 12.5, and 12.8 cm respectively). The yearly erosion rate for the first order forest site was slightly lower at 10.2 cm. There were no significant differences when comparing first and second order streams.

On the second order sites (Figure 10b), average erosion rate was greatest on the forest sites across all seasons. The forest sites had significantly greater erosion than the crop and pasture sites during the spring – mid-summer (crop, $P=0.0038$; pasture, $P=0.0105$) and greater than all other sites during the second winter (crop, $P=0.0005$; pasture, $P=0.0006$; riparian forest, $P=0.0008$) (Table 2). When comparing seasonal erosion rate on each land use, many significant differences were found. On the crop sites, the first winter (6.3 cm) was significantly greater than the mid-summer – autumn (0.65 cm, $P=0.0055$). On the pasture sites, the erosion rate during the mid-summer – autumn (0.6 cm) was significantly less than the first winter (8.2 cm, $P=0.0007$) and the second winter (5.4 cm, $P=0.0126$). On the riparian forest sites, both winters were higher than the mid-summer – autumn (winter 1, 5.8 cm, $P=0.0048$; winter 2, 5.6 cm, $P=0.0048$. On the forest sites, the first winter (9.8 cm) and spring – mid-summer (8.1 cm) were significantly higher than the mid-summer – autumn (P=0.0004; P=0.0014). The second winter on the forest sites (17.9 cm) showed significantly greater erosion than the spring – mid-summer (8.1 cm, P=0.006) and mid-summer – autumn (1.3 cm, P<0.0001).

Erosion activity: Similar results were observed for erosion activity as for erosion rate (Figure 11a). On the first order sites, the only significant difference between land uses was observed during the first winter where the crop sites had significantly higher average erosion activity than the riparian forest sites (P=0.0115) (Table 1). Similar seasonal differences were observed when compared to erosion rate. On the first order crop sites, the first winter (14.0 cm) was significantly greater than all other seasons (spring – mid-summer, 2.1 cm, <0.0001; mid-summer – autumn, 1.9 cm, P=<0.0001; second winter, 6.5 cm, P=0.0069). In addition, the second winter was significantly greater than mid-summer – autumn (P=0.008). On the pasture sites, the mid-summer – autumn (2.3 cm) had significantly less erosion activity than the other seasons (winter 1, 7.2 cm, P=0.0146; spring – mid-summer, 7.4 cm, P=0.0034, second winter, 8.5 cm, P=0.0009). On the riparian forest sites, the mid-summer – autumn (1.4 cm) was significantly less than the spring – mid-summer (6.2 cm, P=0.0052) and the second winter (8.3 cm, P=0.0004). On the forest sites, the mid-summer – autumn (1.6 cm) was significantly less than the second winter (7.1 cm, P=0.0024). There were no significant differences when comparing first and second order streams.

On the second order sites, the pattern seen for erosion activity differed slightly from that seen for erosion rate (Figure 11b). Erosion activity was highest on the forest sites for all of the seasons as it was for erosion rate. Significant differences for land use were observed during the spring – mid-summer and second winter (Table 2). During the spring – mid-summer, the forest sites (9.9 cm) had significantly higher erosion activity than the crop sites (3.4 cm, P=0.0162). During the second winter, the forest sites (19.3 cm) had significantly higher erosion activity than all other sites (crop, 6.5 cm, P=0.0001; pasture, 6.2 cm, P=0.0001; riparian forest, 7.4 cm, P=0.0003). Many seasonal differences were observed. On the crop sites, the mid-summer – autumn (1.5 cm) had significantly less erosion activity than both winters (first winter, 7.3 cm, P=0.0055; second winter, 6.4 cm, P=0.0059). On the pasture sites, the first winter (8.9 cm) had significantly higher erosion activity than the spring – mid-summer (3.7 cm, P=0.0088) and mid-summer – autumn (2.1 cm, P=0.0018). The second winter (6.2 cm) also had significantly higher erosion activity than the mid-summer – autumn (P=0.0152). On the riparian forest sites, the mid-summer – autumn (1.1 cm) had significantly less erosion activity than the first winter (7.2 cm, P=0.0039) and second winter (7.4 cm, P=0.0007). On the forest sites, the mid-summer – autumn (2.2 cm) had significantly less erosion activity than all other sites (winter 1, 10.6 cm, P=0.0029; spring – mid-summer, 9.9 cm, P=0.0001; second winter, 19.3 cm, P<0.0001). The second winter also had significantly higher erosion activity than the first winter (P=0.0017) and spring – mid-summer (P=0.0017). The erosion rate and erosion activity for the sites was based on measurements of the erosion pins. The erosion rate data was used to calculate total soil and phosphorus losses on the sites.

Soil and Phosphorus Losses: Stream bank soil losses due to stream bank erosion also did not follow the expected trend. On the first order sites, soil loss was generally greatest the first winter, the spring – mid-summer, and the second winter and lowest in the mid-summer - autumn (Figure 12a). The values for soil loss are reported in tonnes/km of bank length. There were no significant differences between the land-uses during any season (Table 1). On the pasture, the second winter (98.3 tonnes/km) had significantly higher soil loss than the mid-summer – autumn (14.8 tonnes/km, P=0.0132). On the riparian forest sites, the second winter (123.1 tonnes/km) was also significantly higher than the mid-summer – autumn (11.9 tonnes/km, P=0.002). There were no significant differences when comparing first and second order streams.

On second order sites, soil loss was generally greatest in the two winters (Figure 12b). The only significant land use difference was observed during the second winter where the forest sites (226.3 tonnes/km) had higher soil loss than the riparian forest sites (29.4 tonnes/km, P=0.0063) (Table 2). Two significant seasonal differences were observed. On the crop sites, the second winter (84.9 tonnes/km) had higher soil loss than the spring – mid-summer (-11.8 tonnes/km, P=0.008). On the forest sites, the second winter was significantly greater than the mid-summer – autumn (17.1 tonnes/km, P<0.0001).

Phosphorus (P) losses were associated with soil loss because phosphorus binds to soil particles. Phosphorus losses were generally greatest during the first winter, spring – mid-summer, and the second winter on first order sites (Figure 13a). The values for P loss are reported in kg/km of bank length. On the first order sites, there were no significant differences between land uses (Table 1). The only significant seasonal differences were observed on the riparian forest sites where the mid-summer – autumn (3.9 kg/km) was significantly less than the spring – mid-summer (28.1 kg/km, P=0.0055) and the second winter (40.1 kg/km, P=0.0029). There were no significant differences when comparing first and second order streams.

On the second order sites, the two winters generally had the highest P loss (Figure 13b). There were no significant differences between land-use (Table 2). The only significant seasonal differences were observed on the crop sites. The first winter (44.2 kg/km) was significantly higher than the spring – mid summer (-5.7 kg/km, P=0.0049) and mid-summer – autumn (4.3 kg/km, P=0.0087).

Livestock access points and gullies

The number of livestock access points and gullies were grouped together because they were both areas of concentrated flow into the stream channels, with the assumption that concentrated flow paths (CFPs) observed on non-pasture sites were gullies. An average number was calculated per kilometer of stream bank for each land-use based on the original stream bank survey for each site (Table 4). The highest number of CFPs was observed on the first order pasture sites with an average of 31.7 CFPs/km. The smallest number of CFPs was observed on the second order riparian forest sites, 3.3 CFPs/km.

Stream discharge and air temperature: The spring – mid-summer had some of the largest and most frequent discharge events (143.3 m^3/s) (Figure 14). There were several smaller events in the first winter. There was one moderate sized event in both the mid-summer – autumn and the second winter (USGS 2009). During the first winter, there were 36 days that were below freezing throughout the entire day and 73 days in which the daily mean temperature was below freezing (NCDC 2009). During the second winter, there were 26 days below freezing throughout the entire day and 58 days in which the daily mean temperature was below freezing.

Discussion and Conclusions

Study results emphasize that the presence of a specific type or amount of riparian vegetation does not necessarily ensure bank stability because stream banks continue to adjust as fluvial processes and land use changes continue upstream and downstream (Abernethy and

Rutherfurd 2000). Stream and bank conditions at a location are rarely static, with erosion occuring at some locations along a stream and deposition along other locations. Changes in stream discharge, channelization increased sediment loading from overland or upstream sources, and other changes in the stream channel affect the erosion of the stream banks and bed (Clark et al. 1985). Alterations in land-use adjacent to streams can also affect stream bank erosion (Hooke 1980; Clark et al. 1985; Lyons et al. 2000; Zaimes et al. 2008). Because the sites were only approximately 400m long and land-uses were often different upstream and downstream of the sites, one might not expect the potential benefits of a specific land use to be equally effective in each setting.

When looking at the results in this study, there was no trend in erosion based on land use for either first or second order streams. There was, however, a general trend in seasonal erosion where erosion was generally highest in the two winters followed by the spring – mid-summer and lowest in the mid-summer - autumn. The yearly erosion rates in this study (8 – 26.9 cm) were within the ranges found in other studies (0-150) (Hagerty et al. 1981; Lawler 1993; Zaimes 2004; Palmer 2008).

Previous studies have found that streams adjacent to continuously grazed pastures (Lawler 1993; Zaimes et al. 2008) and row crop fields (Zaimes et al. 2008) generally have higher erosion rates than forested sites (Lawler 1993) or those with planted riparian buffers (Zaimes et al. 2008). In contrast, and in agreement with results in this study, a recent study conducted in central Iowa (Palmer 2008) found results more similar to the results of this study. That study found that forested sites had the highest average erosion and the lowest was on the grazed pasture sites. However, the forest sites in the Palmer study were at the lower end of the watershed and so may have been influenced by other factors as well. We found that there was high variability in erosion rates between pin plots on sites from the same treatment and even within one pin plot. This variability has been observed in other studies as well (Lawler 1993; Zaimes et al. 2008). Yearly soil losses per unit length (33.5 – 124.2 tonnes/km/yr) were in the range reported by Zaimes et al. (2008) of 5-304 tonnes/km/year. Yearly phosphorus losses per unit length (19.5 – 73.3 kg/km/yr) were also in the range of phosphorus losses that these authors reported (2 – 123 kg/km/yr).

Soil losses and phosphorus losses were calculated using average bank heights and the total eroding length of each of the sites. Bank heights for each pin plot ranged from 0.5 m to 2.75 m (average heights 0.75 – 1.90 m; Table 3). The lowest average bank height for the sites was on the first order crop sites. Many of the first order sites looked more like gullies that extended into the crop field (Figure 15). As would be expected, second order sites generally had taller banks than the first order sites. In the context of the channel evolution model (Schumm et al. 1984), the streams in this study are either downcutting and incising (Stage II) or widening with bank collapse (Stage III). Most of the streams are widening, especially on the second order streams, although there are a few first order streams in the upper reaches that are still downcutting, evidenced by the presence of knickpoints. Most of the banks were vertical and more vegetation grew on the tops of the banks than on the bank faces (Figure 7a). This, combined with the stage in channel evolution, indicates that the banks are not stable now and that riparian vegetation may not be controlling stream bank erosion.

Erosion rates are estimates of the total soil lost from stream bank erosion. There were no land-use trends on the first order sites, but on the second order sites the forest sites consistently had the highest erosion rates. Palmer (2008) found that areas with high sinuosity located directly downstream of channelized reaches were subject to higher erosion rates and longer eroding bank lengths than locations higher in the watershed. Since forested sections were less likely to be channelized, they may have had higher erosion rates when downstream of a channelized reach (Palmer 2008). Erosion activity is used to indicate the amount of soil movement on a bank, whether erosion or deposition. The erosion activity followed the same general seasonal trend as was observed for erosion rate. Due to the seasonal nature of erosion observed, subaerial processes such as freezing and thawing cycles and related swelling/shrinking changes may be playing a bigger role in stream bank erosion than fluvial processes. In several past studies, researchers have observed that stream bank erosion was greatest during the winter months, which they attributed to freezing and thawing of the streambanks and higher stream bank soil moisture during this time (Wolman 1959; Lawler 1986, Simon et al. 2000; Couper 2003). Wolman (1959) found the lowest erosion rates during the summer months. Subaerial processes are thought to dominate the erosion processes in the upper reaches of rivers and streams (Lawler 1995; Couper and Maddock 2001). Few studies have been conducted on subaerial erosion due to their seasonal nature and the difficulty associated with separating such processes from fluvial erosion (Abernethy and Rutherfurd 1998).

Soils with high silt-clay contents have been found to be more resistant to fluvial entrainment (Thorne and Tovey 1981). However, they are more susceptible to freeze-thaw or needle ice development which reduces soil strength and makes them less resistant to fluvial erosion during winter months (Couper 2003). The type of vegetation on the banks affects the occurrence of freeze-thaw cycles. A study by Wynn and Mostaghimi (2006) found that banks with herbaceous cover had fewer freeze-thaw cycles than forested banks because the herbaceous vegetation provided some insulation to the soil surface. The annual herbaceous vegetation on the banks in this study was dead during the winter months and may not provide sufficient insulation. The higher number of freeze-thaw cycles for banks with deciduous trees indicates that the trees provided little protection for the banks against freeze thaw cycling during the winter months (Wynn and Mostaghimi 2006). The number of days below freezing obtained from the NCDC is indicative of the freeze-thaw cycling occurring. While these values report air temperature rather than soil temperature, they do have an affect on the likelihood of freeze-thaw occurring in the banks. Close to a third of the days during the winter months never rose above freezing. Close to half of the winter days had an average air temperature below freezing. These data, combined with the high erosion rates during the winter months, supports the conclusion that freeze-thaw cycling may be promoting higher stream bank erosion during the winter. Freeze-thaw cycling results in deposition at the base of banks which is then carried away by higher discharges in the late winter and early spring (Couper 2003).

In the warmer months, evapotranspiration by vegetation along the banks removes water from the banks. Drier banks are generally more stable than wet banks (Thorne 1990). Trees may be able to affect the antecedent moisture conditions throughout the bank because tree roots have deeper root systems than herbaceous vegetation and can obtain water from a larger volume of soil (Abernethy and Rutherfurd 1998; Wynn and Mostaghimi 2006). Evapotranspiration would have been less in winter because of minimal transpiration and so the bank soils would likely have higher antecedent moisture due to precipitation events during these months. Many of the banks observed in this study were lacking perennial vegetation on the bank face and so most of the effects of vegetation on the banks would be due to the vegetation on the top of the banks. The lack of vegetation on the banks during the winter months also leaves the banks more exposed and vulnerable to scour and freeze-thaw (Thorne 1990).

The largest discharge events occurred during the summer months. These high discharges, however, did not coincide with the largest erosion rates. Simon et al. (2000) found that prolonged wet periods rather than peak storm events were more likely to induce bank failures due to an increase in soil unit weight and a decrease in matric suction. Matric suction is an apparent cohesion caused by the attraction of water molecules towards each other in the space between unsaturated soil particles (Ward and Robinson 1990). The role of vegetation in enhancing or reducing the stability of banks is complex (Thorne 1990). While vegetation can increase matric suction by extracting water from the soil via the root system (Thorne 1990), the main effect of vegetation on the stability of stream banks is due to root reinforcement (Gray and Leiser 1982; Thorne 1990; Abernethy and Rutherfurd 1998). However, root reinforcement is reduced when banks are saturated (Pollen 2007). This means that when banks are at risk of failure due to reduced matric suction and increased soil weight, root reinforcement may be low. If the banks had high antecedent moisture during the winter months, the late winter and early spring precipitation events may have caused bank failure.

Other sources of sediment that directly enter streams, but were not measured in this study, are gullies (Figure 16) and cattle paths (Figure 17 a, b), grouped into concentrated flow paths (CFPs). The average number of CFPs per land use did not follow any specific patterns. However, CFPs were found on every study site, indicating that their presence is influencing the discharge and sediment load of the streams. A future study could measure gully erosion through the use of erosion pins, although they probably would be unsuitable for cattle paths because the cattle would trample the pins. Using a similar study design, Palmer (2008) found that there were numerous gullies that brought additional water and sediment to the streams on his study sites in central Iowa. Zaimes (2004) compared the number of gullies on different riparian land uses and measured gully erosion within pastures in Iowa through using erosion pins. He found that constructed riparian forest buffers and grass filters had the fewest number of gullies followed by pastures in which cattle were excluded from the streams. He determined that the gullies on the continuously grazed pasture sites had a total erosion rate of approximately 16.5 cm/yr. In a study conducted within many of the same sites as this study, Knight (2007) estimated the amount of soil loss from gullies that flowed into stream channels. He found that 20% of the gullies that met the edge of natural riparian forests extended through to the stream. Nine gullies on his sites extended from the crop field down to the stream and it was estimated that these gullies accounted for 97 tonnes of soil loss if assuming that 100% of the soil made it to the stream. Finally, another study in Iowa measured suspended sediment and phosphorus in surface runoff from stream bank critical source areas (cattle access and loafing areas) (Figure 17 a, b) and found that they accounted for up to 72% and 55% of the total sediment and phosphorus loads, respectively (Tufekcioglu 2006). In a watershed scale study, Piest and Bowie (1974) found that twenty percent of the sediment in four Iowa watersheds was estimated to be contributed by gully erosion. Sediment is also stored and moved as part of the stream bed. The erosion and deposition of sediment on stream beds is very difficult to measure and was not measured in this study; however it is part of the sediment delivery process in a stream.

It is evident that there are many sources of sediment entering these streams. Further work needs to be done to determine the source areas of the sediment, possibly through sediment source tracking using naturally occurring radionuclides (^{7}Be and ^{210}Pb) as tracers (Walling 2005). It might also be useful to look at the erosion pin data to determine the location on the bank face where erosion and deposition were taking place seasonally; top, middle, or bottom of the bank.

Particle size analysis is being run on the soil samples collected in this study to determine the percentage of sand, silt, and clay which affect the erodibility of the soils. A more detailed bank survey conducted within one season would eliminate uncertainties due to seasonal vegetation. In addition, more frequent pin measurements would help to determine when erosion and deposition were occurring during the year and may clarify which erosion processes are dominant during different times of year. A survey of the vegetation along the banks to determine factors such as the type, density, size, and health of the vegetation would be useful because these factors control the vegetations influence on bank stability (Thorne 1990).

While stream bank erosion is affected by the presence or absence of riparian vegetation, the relationship between land use (vegetation) and stream bank erosion was not evident in this study. There was, however, an apparent seasonal trend in which the winter months experienced the highest erosion. This seasonal trend was likely due to subaerial processes such as freeze-thaw cycling that cause the soil to become loosened and then get carried away by fluvial processes. Stream bank erosion is a complex process and is affected by many variables including soil type, upland land use, upstream/downstream land use, vegetation, discharge, bank soil moisture, and stage of channel evolution. In order to reduce the sedimentation of Mark Twain Lake, entire watersheds need to be evaluated to determine where management practices would be most beneficial and cost-effective (Secchi et al. 2008) while keeping in mind that many of the streams are still evolving to adapt to upland and upstream changes in land use.

Acknowledgements

This research has been funded through the University of Missouri Center for Agroforestry. We thank the landowners who allowed us access to their property. Thank you to the many students who assisted with field and lab work. We would also like to thank the USDA-ARS at the University of Missouri for their assistance in locating landowners and collecting field data.

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Table 1. Significant differences between different seasons and land-uses on 1st order sites. The P values had Bonferroni adjusted significance levels.

Erosion Rate

Season	**Difference**	**P value**
Winter 1	1st crop > 1st riparian	0.0104
Land-use		
1st Crop	winter 1 > spring - mid-summer	0.001
	winter 1 > mid-summer - autumn	<.0001
	winter 1 > winter 2	0.0021
	winter 2 > mid-summer - autumn	0.0104
1st Pasture	winter 1 > mid-summer - autumn	0.0115
	winter 2 > mid-summer - autumn	0.0012
1st Riparian	winter 2 > mid-summer - autumn	0.0014
1st Forest	winter 2 > mid-summer - autumn	0.0076

Erosion Activity

Season	**Difference**	**P value**
Winter 1	1st crop > 1st riparian	0.0115
Land-use		
1st Crop	winter 1 > spring - mid-summer	<.0001
	winter 1 > mid-summer - autumn	<.0001
	winter 1 > winter 2	0.0069
	winter 2 > mid-summer - autumn	0.008
1st Pasture	winter 1 > mid-summer - autumn	0.0146
	spring - mid-summer > mid-summer - autumn	0.0034
	winter 2 > mid-summer - autumn	0.0009
1st Riparian	spring - mid-summer > mid-summer - autumn	0.0052
	winter 2 > mid-summer - autumn	0.0004
1st Forest	winter 2 > mid-summer - autumn	0.0024

Soil Loss

Season	**Difference**	**P value**
	no significant differences	
Land-use		
1st Pasture	winter 2 > mid-summer - autumn	0.0132
1st Riparian	winter 2 > mid-summer - autumn	0.002

Phosphorus Loss

Season	**Difference**	**P value**
	no significant differences	
Land-use		
1st Riparian	spring - mid-summer > mid-summer - autumn	0.0055
	winter 2 > mid-summer - autumn	0.0029

Table 2. Significant differences between different seasons and land-uses on 2nd order sites.

Erosion Rate

Season	Difference	P value
Spring - early summer	2nd forest > 2nd crop	0.0038
	2nd forest > 2nd pasture	0.0105
Winter 2	2nd forest > 2nd crop	0.0005
	2nd forest > 2nd pasture	0.0006
	2nd forest > 2nd riparian	0.0008
Land Use		
2nd Crop	winter 1 > late summer - autumn	0.0055
2nd Pasture	winter 1 > spring - early summer	0.0103
	winter 1 > late summer - autumn	0.0007
	winter 4 > late summer - autumn	0.0126
2nd Riparian	winter 1 > late summer - autumn	0.0048
	winter 2 > late summer - autumn	0.0048
2nd Forest	winter 1 > late summer - autumn	0.0004
	spring - early summer > late summer - autumn	0.0014
	winter 2 > spring - early summer	0.006
	winter 2 > late summer - autumn	<.0001

Erosion Activity

Season	Difference	P value
Spring - early summer	2nd forest > 2nd crop	0.0162
Winter 2	2nd forest > 2nd crop	0.0001
	2nd forest > 2nd pasture	0.0001
	2nd forest > 2nd riparian	0.0003
Land Use		
2nd Crop	winter 1 > late summer - autumn	0.0055
	winter 2 > late summer - autumn	0.0059
2nd Pasture	winter 1 > spring - early summer	0.0088
	winter 1 > late summer - autumn	0.0018
	winter 2 > late summer - autumn	0.0152
2nd Riparian	winter 1 > late summer - autumn	0.0039
	winter 2 > late summer - autumn	0.0007
2nd Forest	winter 1 > late summer - autumn	0.0029
	spring - early summer > late summer - autumn	0.0001
	winter 2 > winter 1	0.0017
	winter 2 > spring - early summer	0.0017
	winter 2 > late summer - autumn	<.0001

Soil Loss

Season	Difference	P value
Winter 2	2nd forest > 2nd riparian	0.0063
Land Use		
2nd Crop	winter 2 > spring - early summer	0.008
2nd Forest	winter 2 > late summer - autumn	<.0001

Phosphorus Loss

Season	Difference	P value
	no significant differences	
Land Use		
2nd Crop	winter 1 > spring - early summer	0.0049
	winter 1 > late summer - autumn	0.0087
	winter 2 > spring - early summer	0.0018
	winter 2 > late summer - autumn	0.0049
2nd Pasture	winter 1 > spring - early summer	0.0109
	winter 1 > late summer - autumn	0.0073
2nd Forest	winter 2 > late summer - autumn	0.0027

Table 3. Soil and phosphorus losses from stream bank erosion under different land uses on 1^{st} and 2^{nd} order streams in Missouri.

Riparian Land Use & Stream Order	Site Length	Severe and Very Severe Stream Bank					Average Stream Bank Soil Loss	Stream Bank Soil Phosphorus Concentrations	Stream Bank Soil Phosphorus Losses
	Total	Length	% of Total Site Length	Ave. Height	Eroded Area	Bulk Density	Yearly, Per Unit Length		Per Unit Length
	(m)	(m)	(%)	(m)	(m2)	(g/cm3)	(tonnes/km/yr or kg/m/yr)	(mg/kg)	(kg/km/yr)
1st Crop 1	800	330	41.3	0.84	277.2	1.36		614.7	
1st Crop 2	800	188	23.5	0.88	165.4	1.37		352.4	
1st Crop 3	800	365	45.6	0.85	310.3	1.38	62.7	432.1	31.4
1st Forest 1	680	318	46.8	0.75	238.5	1.40		382.3	
1st Forest 2	600	223	37.2	1.43	318.9	1.52		197.2	
1st Forest 3	800	488	61.0	1.44	702.7	1.49	90.6	260.9	19.5
1st Pasture 1	800	567	70.9	1.65	935.6	1.40		286.1	
1st Pasture 2	800	185.8	23.2	1.24	230.4	1.36		359.6	
1st Pasture 3	800	580	72.5	1.90	1102.0	1.48	50.8	311.0	62.6
1st Riparian 1	800	573	71.6	1.67	956.9	1.43		315.6	
1st Riparian 2	800	689	86.1	1.18	813.0	1.30		356.8	
1st Riparian 3	800	665	83.1	1.43	951.0	1.48	54.3	327.0	71.0
2nd Crop 1	800	800	100.0	1.46	1168.0	1.26		503.0	
2nd Crop 2	800	459	57.4	1.87	858.3	1.40		377.5	
2nd Crop3	800	549	68.6	1.56	856.4	1.30	33.5	385.8	58.5
2nd Forest 1	800	467	58.4	1.71	798.6	1.63		208.2	
2nd Forest 2	800	321	40.1	1.53	491.1	1.47		248.9	
2nd Forest 3	1200	279	23.3	1.78	496.6	1.48	82.3	263.0	73.3
2nd Pasture 1	800	780	97.5	1.76	1372.8	1.45		381.7	
2nd Pasture 2	950	145	15.3	1.75	253.8	1.36		534.4	
2nd Pasture 3	800	294	36.8	1.21	355.7	1.34	124.2	412.3	50.5
2nd Riparian 1	800	186	23.3	1.08	200.9	1.49		314.2	
2nd Riparian 2	800	178	22.3	1.30	231.4	1.48		406.5	
2nd Riparian 3	800	364	45.5	1.00	362.2	1.36	50.3	314.7	17.4

Table 4. Number of gullies and livestock access points grouped into the number of concentrated flow paths (CFPs) per km of stream bank for each land use.

Order	Covertype	# CFPs/km
1st	crop	7.9
1st	pasture	31.7
1st	riparian	15.8
1st	forest	4.8
2nd	crop	11.7
2nd	pasture	9.5
2nd	riparian	3.3
2nd	forest	9.9

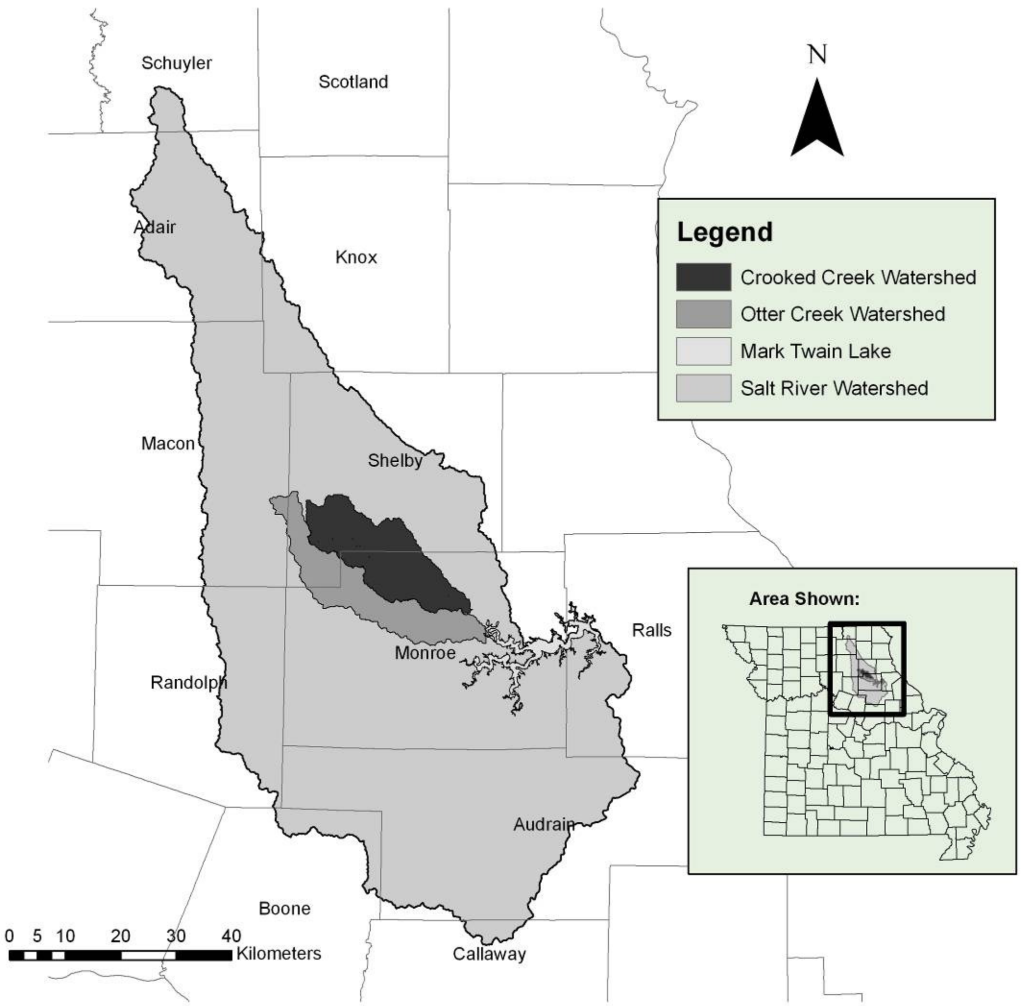


Figure 1. Map showing the location of Crooked and Otter Creek watersheds in the larger Salt River watershed. The streams feed into Mark Twain Lake.

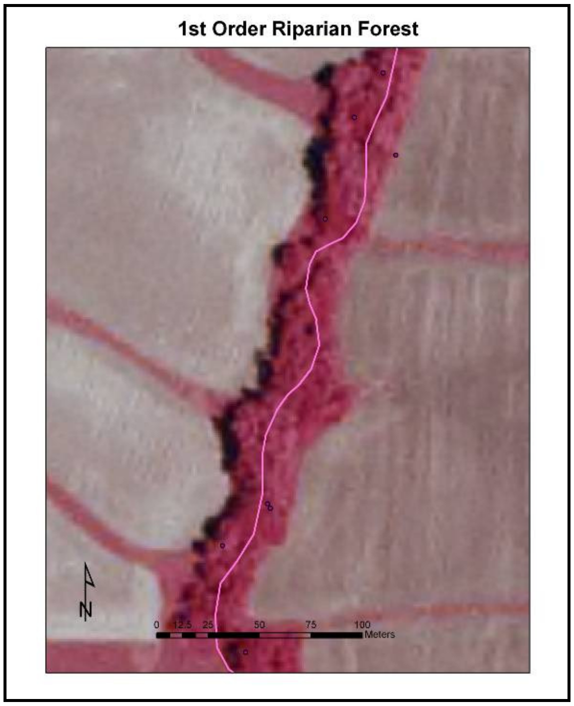


Figure 2. First order riparian forest site with a width of 15 – 30 m on both sides of the stream.

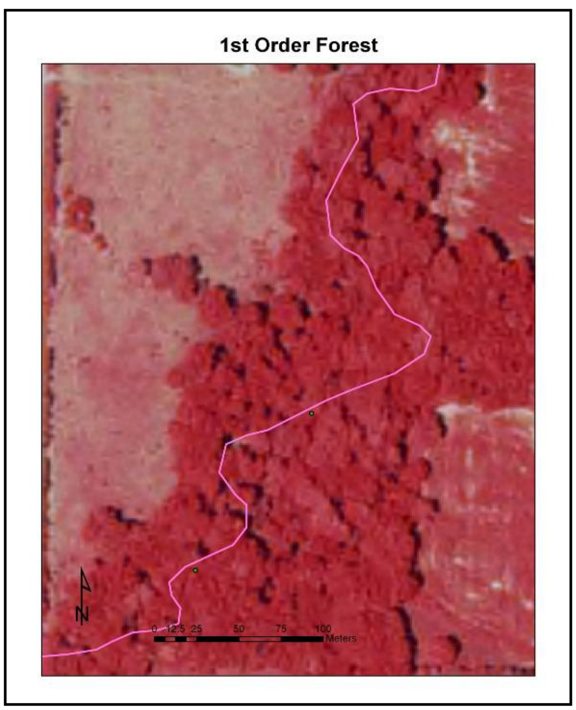


Figure 3. First order forest site with a width of 30+ m on both sides of the stream.

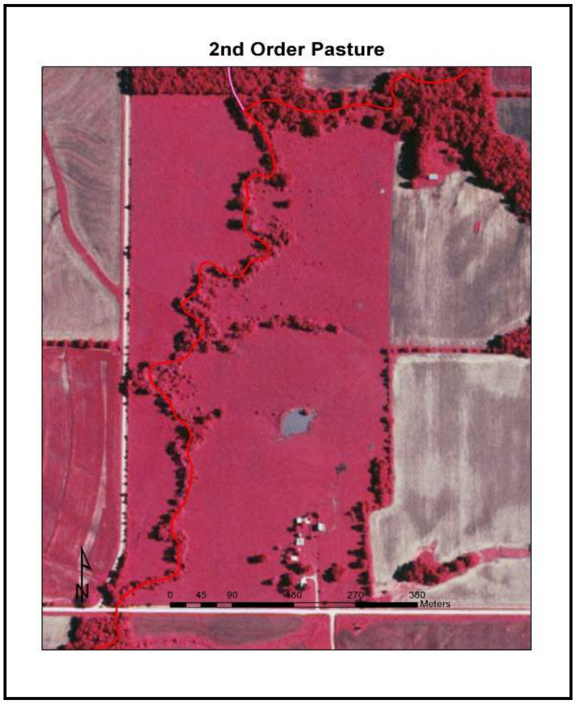


Figure 4. Second order pasture site with scattered trees near the bank. Cattle had full access to the stream.

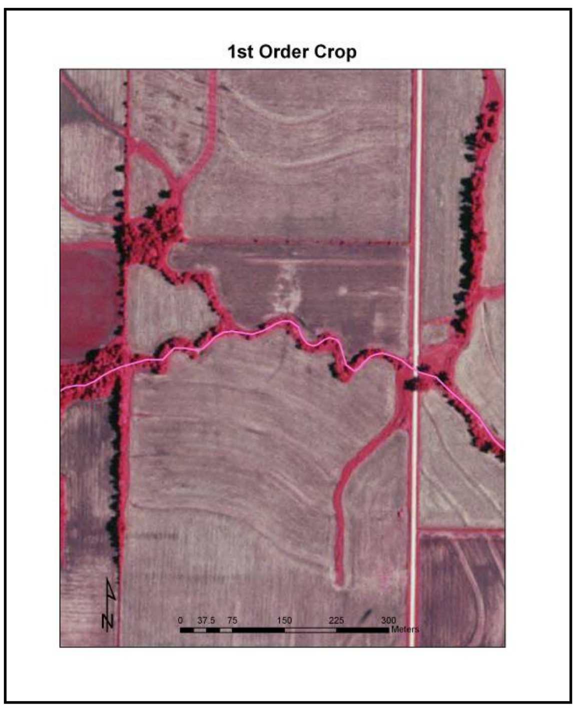


Figure 5. First order crop site with thin strip of perennial vegetation such as small trees, shrubs, and grass along the channel.

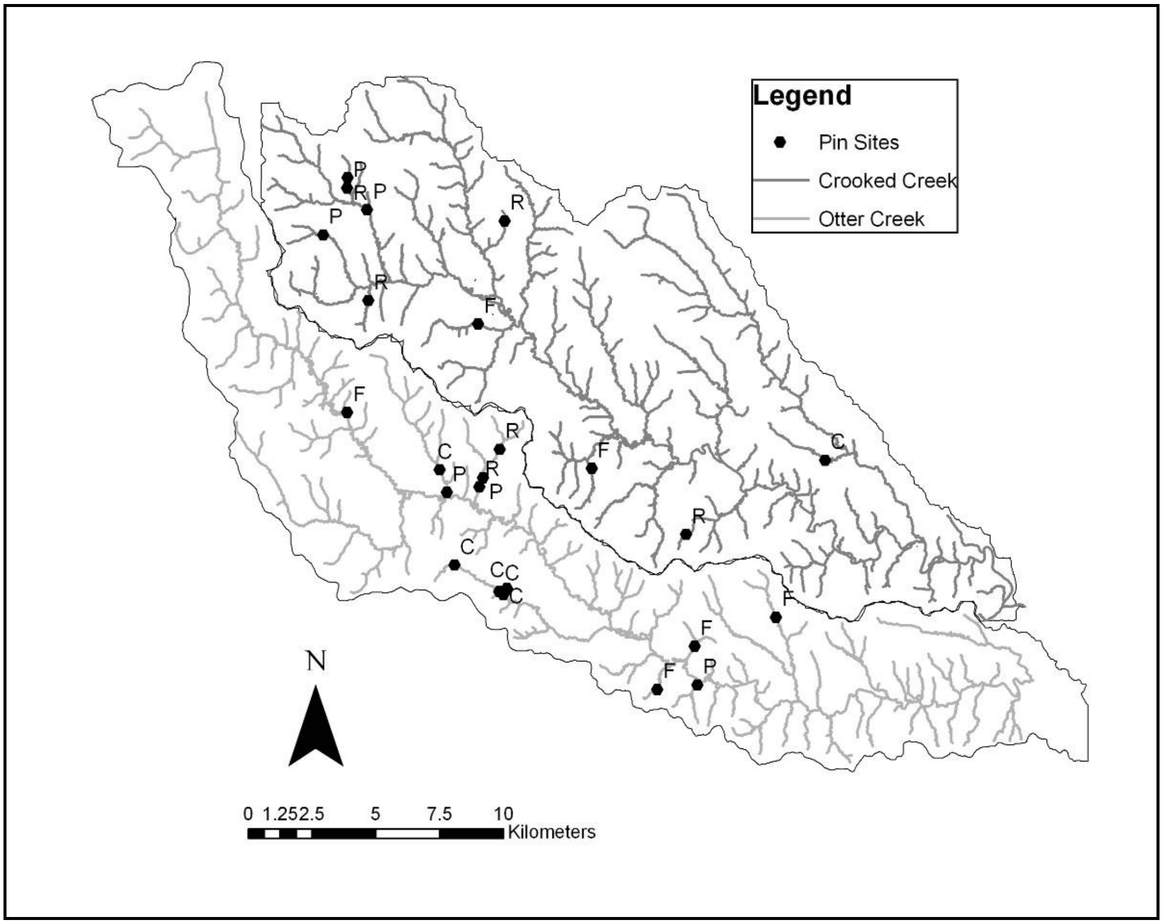


Figure 6. Map showing the location of the study sites within Crooked and Otter Creek watersheds. The sites are labeled by their land use: C = crop, P = pasture, R = riparian forest, and F = forest.

(A)

(B)

Figure 7. Photos of severely to very severely eroding banks. The orange pole (A) represents the downstream end of a pin plot. The ruler (B) is 30 cm long and is used when measuring pins.

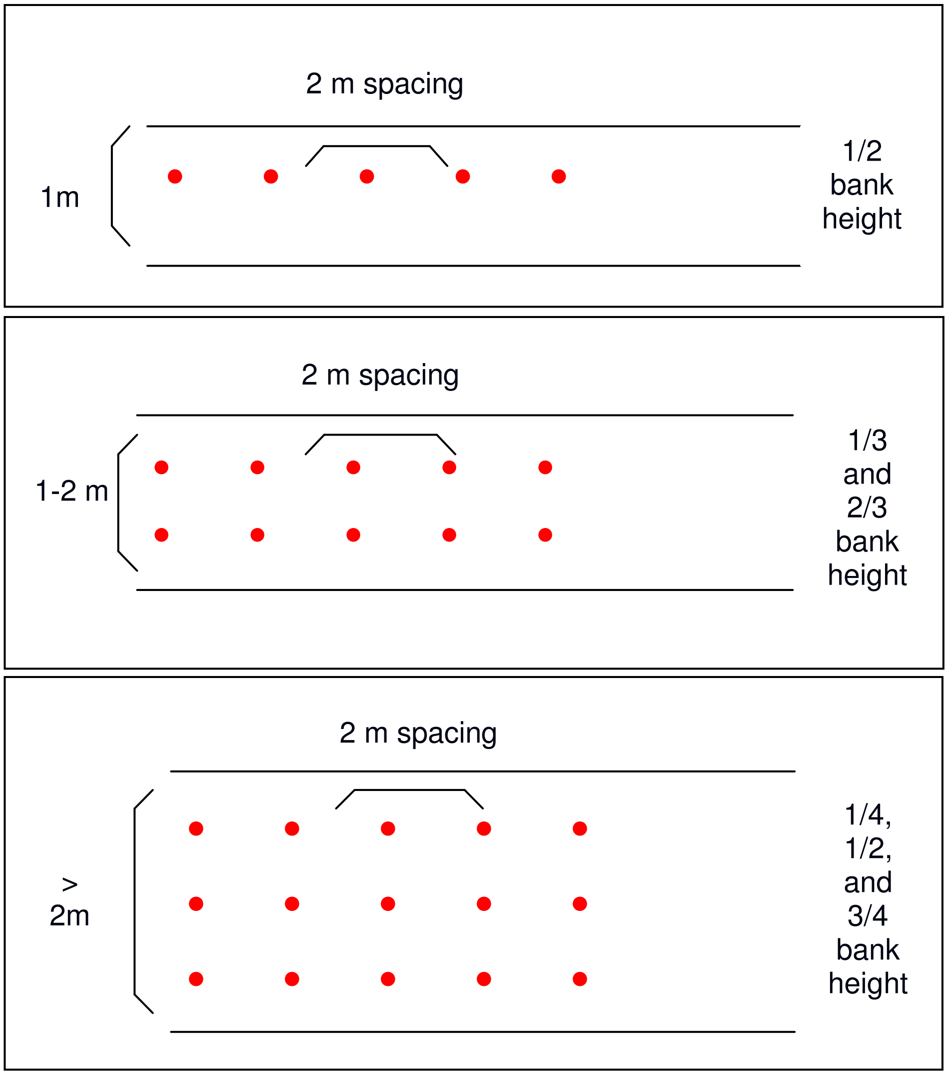
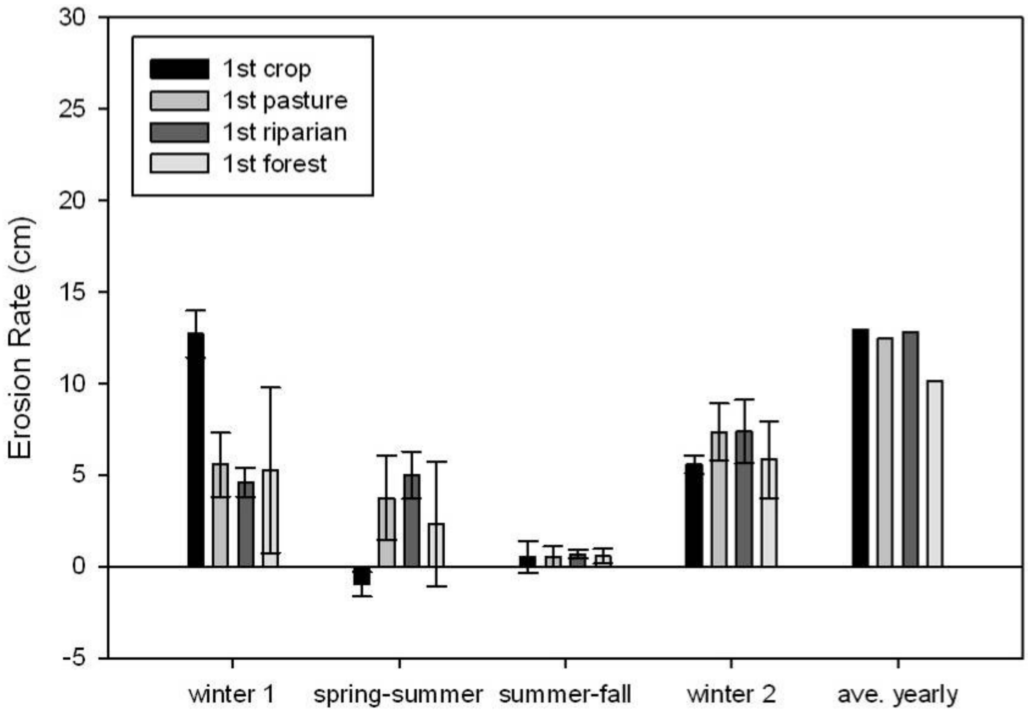


Figure 8. Diagrams of erosion pin placement on eroding stream banks. Pins are in 1, 2, or 3 rows depending on bank height.

Figure 9. Photo of erosion pin placement on a stream that is between 1-2 m tall.

(A)



(B)

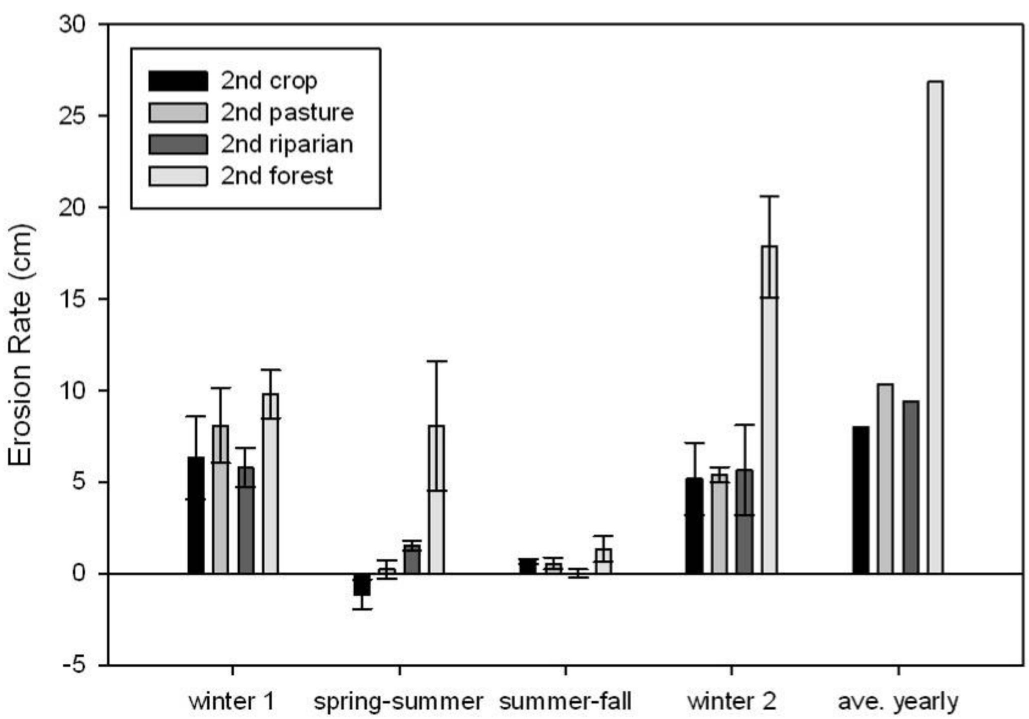
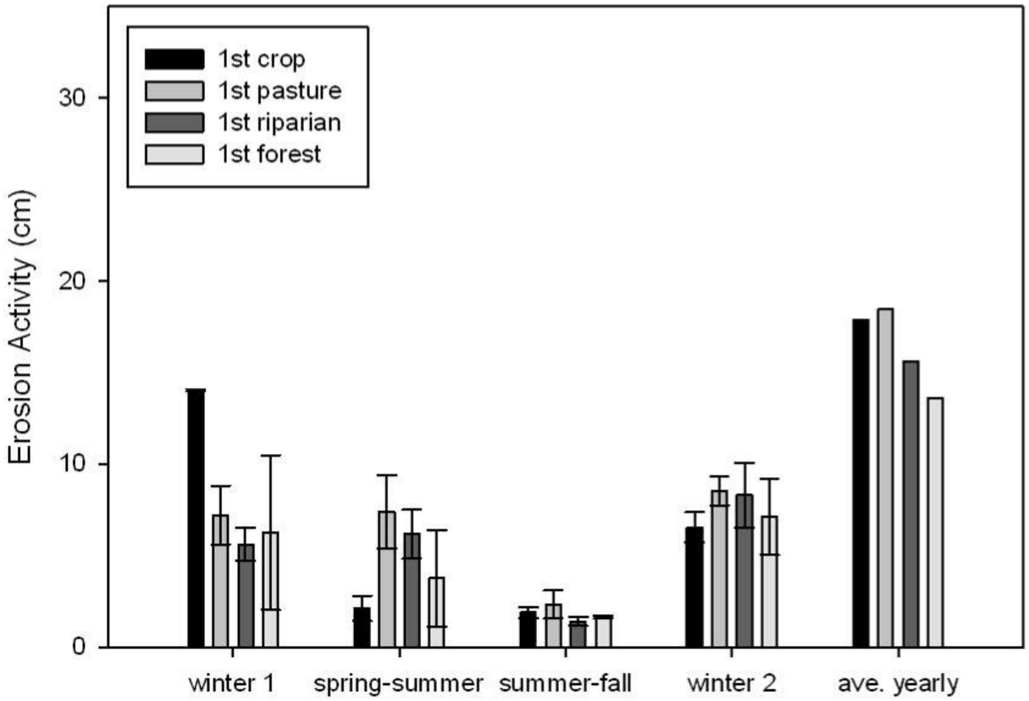


Figure 10. Mean stream bank erosion rates under different land uses on (A) 1st and (B) 2nd order streams.

(A)



(B)

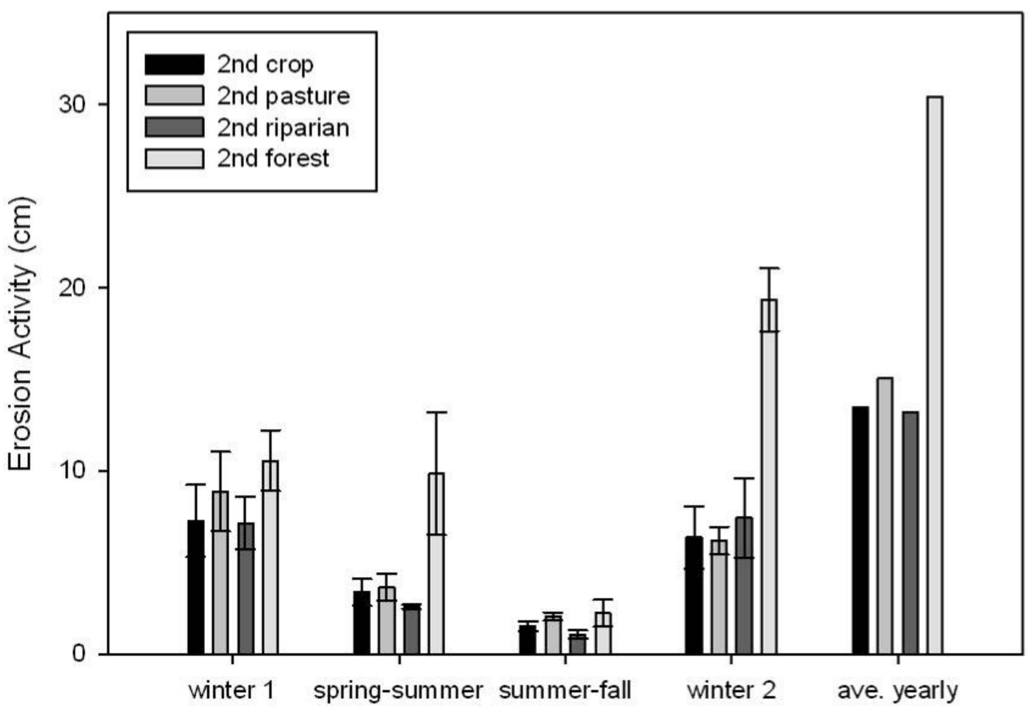
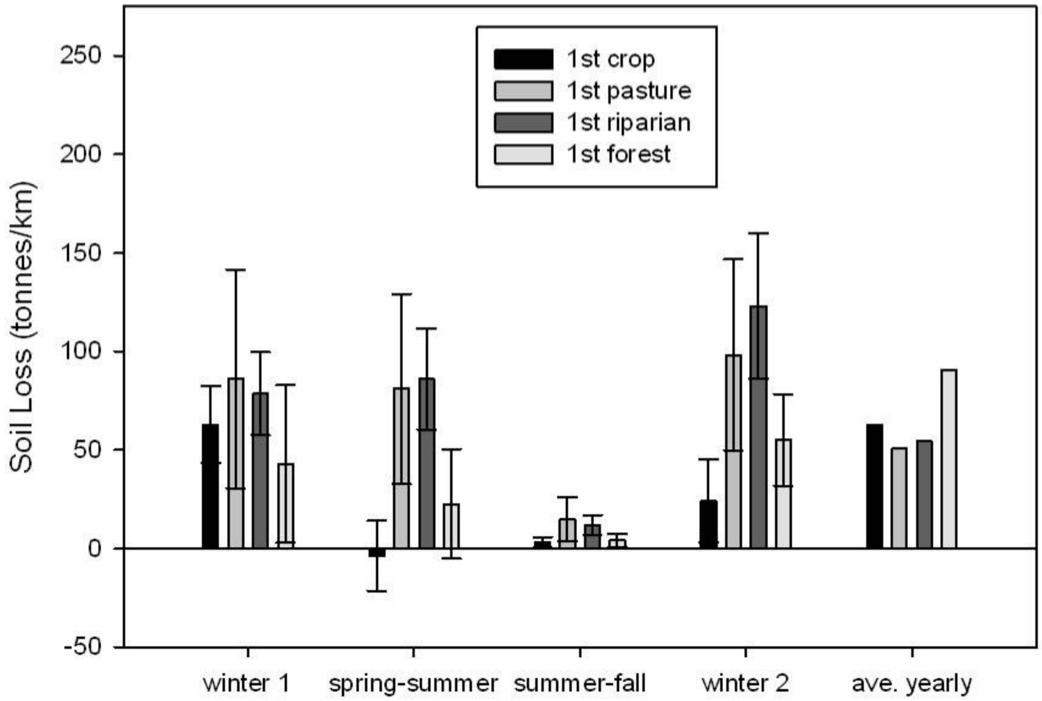


Figure 11. Mean stream bank erosion activity under different land uses on (A) 1^{st} and (B) 2^{nd} order streams.

(A)



(B)

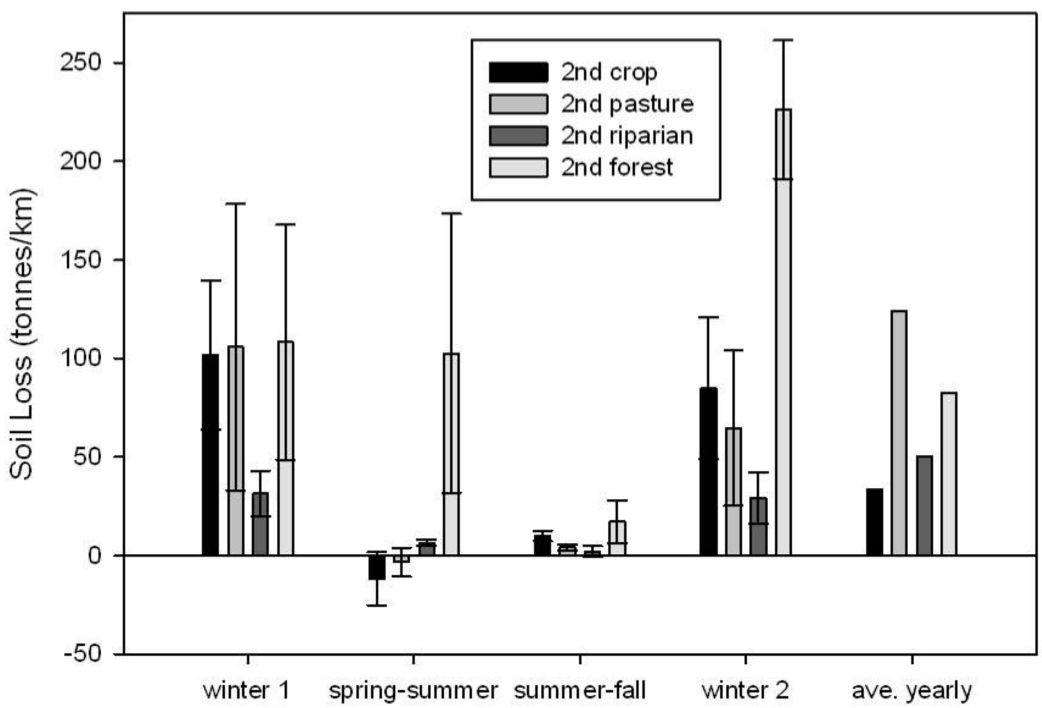
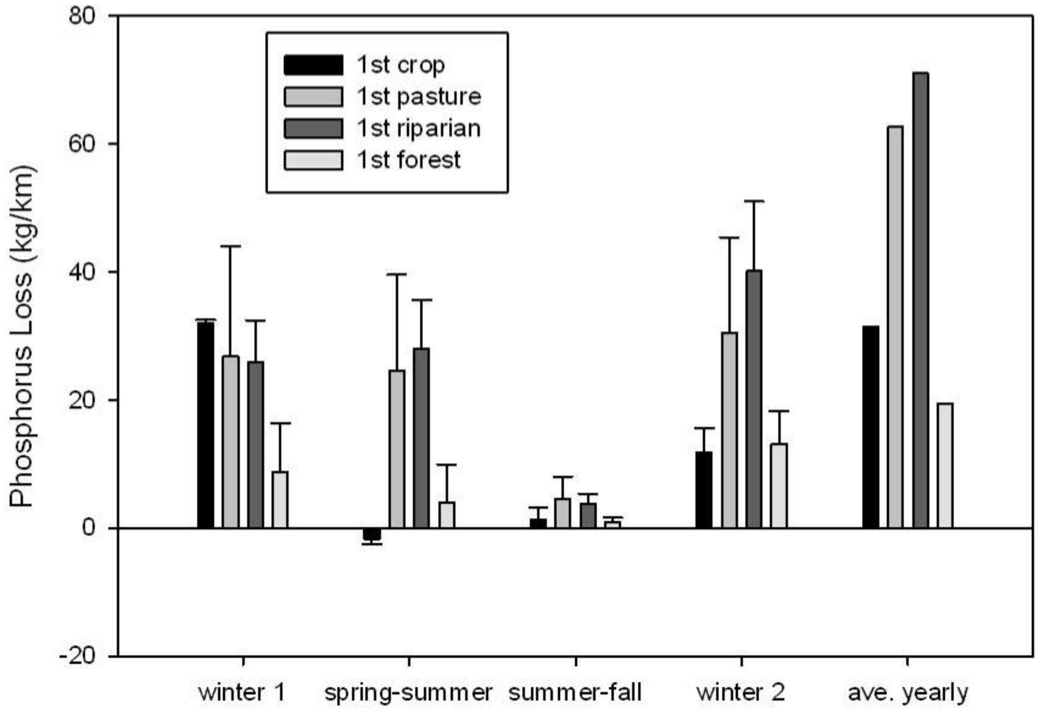


Figure 12. Mean soil loss due to stream bank erosion under different land uses on (A) 1^{st} and (B) 2^{nd} order streams.

(A)



(B)

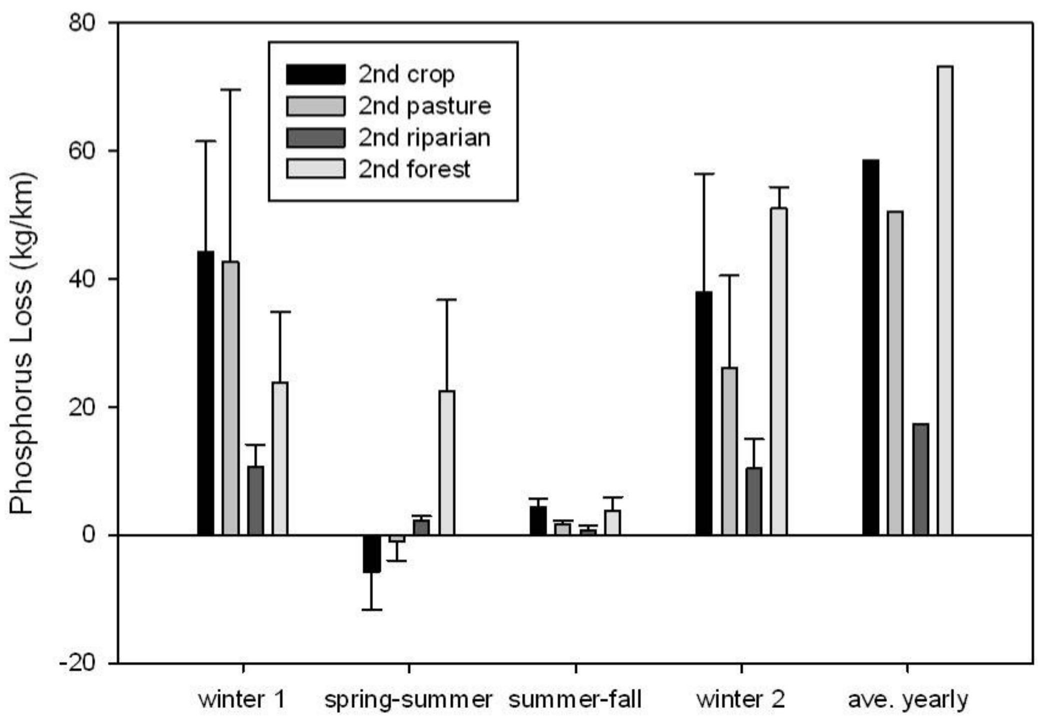


Figure 13. Mean soil phosphorus losses due to stream bank erosion under different land uses on (A) 1st and (B) 2nd order streams.

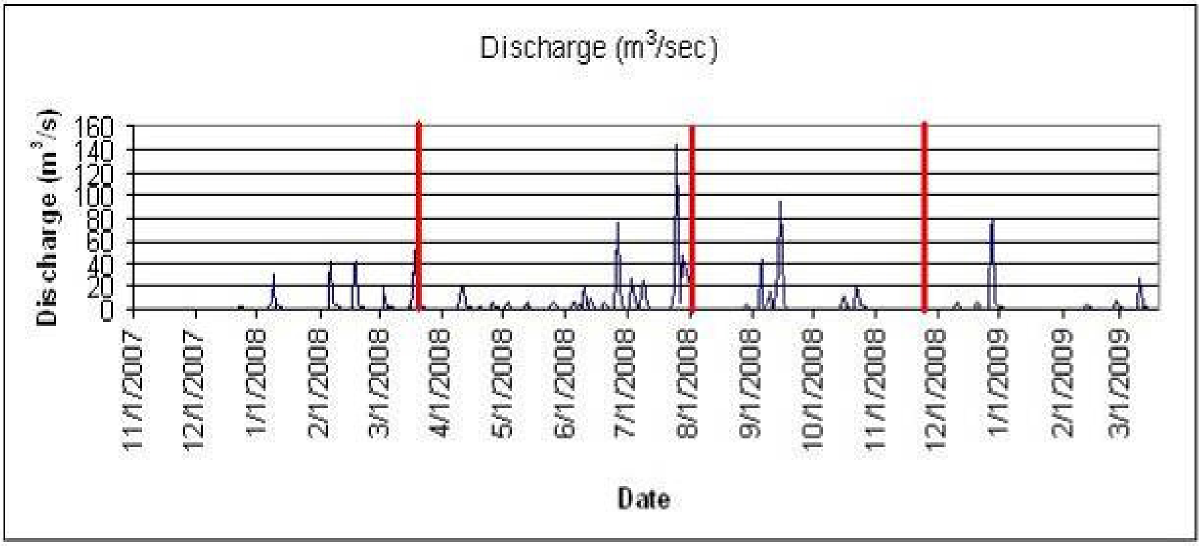


Figure 14. Daily discharge of Crooked Creek near Paris, Missouri from November 2007 to March 2009. The red lines divide the data into the four time periods of measurement.

Figure 15. Photo of eroding stream banks on a first order crop site with low banks and overhanging vegetation.

Figure 16. An incised gully entering a stream on a first order riparian forest site.

(A)

(B)

Figure 17. Photos of pasture sites where (A) shows where cattle access the stream and the destruction they cause and (B) shows a path that cattle have trampled down to the stream and the circled figure is of a dead cow near the stream.

Chapter 3. Bird species diversity in riparian buffers, row crop fields, and grazed pastures within agriculturally dominated watersheds

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Abstract: In response to concern about the loss of ecosystem services once provided by natural riparian systems, state and federal agencies have established incentive programs for landowners to convert sensitive lands from agricultural to conservation uses. Enhancement of wildlife habitat, while identified as a function of such systems, has often been of secondary importance to soil conservation and water quality objectives. Though greatly important, little consideration has been given to how specific species will respond to the design and management of riparian buffers or other conservation lands. This study compared avian communities within a chronosequence of riparian buffers established on previously cropped or pastured land with those of the nearby matrix land cover types (row crop fields and an intensively grazed pasture). The riparian buffers consisted of native grasses, forbs, and woody vegetation established at three different times (2, 9, and 14+ years prior to survey). At each site, 10 min point counts for breeding birds were conducted using 50 m fixed radius plots, which were visited eight times between May 15 and July 10, 2008. A total of 54 bird species were observed over all of the study sites. The re-established riparian buffers in this study had higher bird abundance, richness, and diversity than the crop and pasture sites. These results suggest that re-establishing native riparian vegetation in areas of intensive agriculture will provide habitat for a broad suite of bird species, but that specific species will reflect successional stage, horizontal and vertical vegetative structure, and

compositional diversity of the buffer vegetation. These results emphasize the importance of matching buffer design and management to species requirements if the objectives are to attract specific target species or species groups.

Keywords: bird habitat, riparian forest buffer, Iowa, point count, species richness

Introduction

Since Euro-American settlement, large proportions of wildlife habitat in the U.S. Midwest have been converted from native plant communities to agricultural fields. Economic incentives and technological advances have led to continued agricultural intensification and reduction of native vegetation and habitat in the last 65 years (Paarlberg and Paarlberg 2000; Anderson 2009). This is especially true in Iowa, which has the smallest percentage of its original natural habitat remaining out of all 50 states (Cosner 2001). As a result, heterogeneity in habitat structure, in time and space, has been replaced with homogeneity and vast monocultures, which has reduced plant and animal biodiversity (Benton et al. 2003). To maximize land in cultivation, riparian areas in Iowa and across the Midwest have also been heavily manipulated and converted to agricultural use (Brinson et al. 1981; National Research Council 2002; Deschenes et al. 2003). As a result, many of the ecosystem functions of natural riparian areas have been lost, including water quality functions and provision of wildlife habitat (Schultz et al. 2004).

Such habitat loss and fragmentation in the Midwest U.S. has led to a reduction in available habitat for many wildlife species, with many species of greatest conservation need associated with the habitats with the greatest areal reduction. For example, loss of habitat has caused populations of many grassland dependent bird species to decline, especially in areas that were once covered with extensive prairie (Best et al. 1995, Murphy 2003; Brennan and Kuvlesky 2005), with remaining species occupying anthropogenic habitats that did not exist prior to agricultural conversion (Vickery et al. 1999). Riparian habitats can also be important wildlife habitat (Stauffer and Best 1980) as natural riparian zones provide some of the most diverse and complex terrestrial habitats, especially in areas dominated by intensive agriculture (Naiman et al. 1993).

In response to concern about the loss of ecosystem services such as maintenance of water quality and provision of wildlife habitat once provided by these natural systems, state and federal agencies have established incentive programs for landowners to convert sensitive lands from agricultural to conservation uses. One program widely adopted within the U.S. Midwest is the United States Department of Agriculture Conservation Reserve Program (CRP), in which landowners with eligible land are provided rental payments and cost-share assistance to establish long-term, resource-conserving covers. Initial areas were enrolled in the General Conservation Reserve Program which typically enrolled whole fields. In contrast, the Continuous Sign-up of the CRP (CCRP), authorized in 1996, is geared to small tracts of land, not whole fields, and focuses on many of the “buffer” practices. Under the CCRP, landowners with previously cropped or pastured land within riparian zones have the option of establishing perennial cool- or warm-season grass mixes (Filter Strip, Conservation Practice Standard Code 393), or mixtures of woody vegetation and grasses (Riparian Forest Buffer, Conservation Practice Standard Code 391). Since establishment of the CRP in the 1985 Farm Bill, over 130,000 ha of riparian area in Iowa have been re-established to perennial vegetation under these two practice standards. Recent trends however, have shown lower enrollments into such programs and, in some cases, reversion of the land use back to agricultural production (Secchi et al. 2008), resulting from annual rental rates for most land enrolled in the CRP not being competitive with current crop prices. This trend may well continue as over 242,000 hectares currently enrolled in CRP in Iowa will have contracts expire between 2008 and 2012 (USDA – FSA 2009), with a significant proportion of this area currently enrolled in the general CRP. Such land use conversion could further reduce critical wildlife habitat within these regions of intensive agriculture, increasing the importance of those areas enrolled in CCRP.

The overall goals of the CCRP are to improve the quality of water, control soil erosion, and enhance wildlife habitat. To achieve these objectives, significant flexibility is allowed with respect to species selection, and the establishment and management of the site. Enhancement of wildlife habitat, while identified as a function of such systems, has often been of secondary importance to soil conservation and water quality objectives. Though greatly important, little consideration has been given to how specific species or guilds of species will respond to the design and management of CRP and other conservation lands. Here we focus on the wildlife response within a chronosequence of riparian buffers re-established within two watersheds in Central Iowa. We seek to understand how differences in their design and management impact habitat use by wildlife species or guilds of species, in this case breeding birds. The goal of this research is to inform the design of riparian buffers to achieve ecological services beyond the typical soil and water conservation. We achieve this through a comparison of breeding bird composition at five sites, including a chronosequence of riparian buffers, nearby row crop fields, and an intensively grazed pasture along Bear Creek and Long Dick Creek in north-central Iowa, USA.

Conservation practices implemented within the Bear Creek National Restoration Demonstration Watershed provide an ideal opportunity for assessing avian response to riparian buffer establishment. This project represents one of the longest-term assessments of ecosystem services provided by the incorporation of continuous living cover and perennials within an intensively agricultural watershed in the Midwest (Schultz et al. 2004). Since initiation in 1990, this project has grown to include nearly 16 km of riparian buffer with over 12 cooperating landowners. All of the buffers have been established under standards required by the state or federal incentive program utilized by the landowners, and the watershed represents a microcosm for the nearly 130,000 ha of riparian conservation buffers established in Iowa under the CRP.

This study compared avian communities within a chronosequence of riparian buffers established predominantly for water quality functions, with those of the nearby matrix land cover types (row crop fields and an intensively grazed pasture) with the goal to understand how buffer design and management impacts habitat use by breeding birds. These results were compared to findings from similar studies conducted within these same riparian areas in 1994, 1997, and 1999. Comparison of data from the 2008 survey to past surveys provides insight on how bird communities on the same or similarly-aged sites changed over time as the re-established buffers matured and increased in plant species diversity and complexity. We hypothesized that bird species diversity would be related to the complexity of the habitat (e.g. foliage height diversity), which is a function of buffer design and age. We also hypothesized that guilds of species would exhibit affinities for different habitat, which are also a function of buffer design and age. While the study design strictly limits the conclusions from this study to the sites and watersheds assessed, we believe that the inferences can be used to inform the design and management of similar conservation buffer systems within the region to enhance avian or other wildlife habitat.

Materials and Methods

Description of the study sites: The study was conducted along Bear Creek and Long Dick Creek of Hamilton and Story counties in north-central Iowa. Land use surrounding these creeks is dominated by agriculture, primarily row crop farming (corn [*Zea mays* L.] and soybeans [*Glycine max* L. (Merr.)]) and pasture. Bear Creek (6,941 ha) and Long Dick Creek (9,401 ha) watersheds are small drainage basins located within the Des Moines Lobe subregion of the Western Corn Belt Plains ecoregion (Griffith et al. 1994). In general, the topography of this area is flat to gently rolling.

Five sites were included in the study, including three different areas with riparian buffers, an area with intensively grazed pasture, and an area in row crops (Figure 1). All buffer sites were established under standards required by the state or federal program utilized (generally Conservation Practice Standard Code 391 – Riparian Forest Buffer). As such, prior to establishment, these sites were within row crop or pasture agriculture. The riparian buffer sites represent three different ages since planting: 14-18 years (hereafter 14+), 9 years, and 2 years. All buffers were composed of three zones; a managed tree zone adjacent to the stream followed by a shrub zone and a native grass/forb zone adjacent to the crop field (Schultz et al. 2004). The 14+ year-old buffer was planted to approximately 18 m on each side of the stream; however, there is a large center section between the original channel and a newly forming channel covered by native warm-season grass, forbs, and invading small willows. The combined width of this buffer is close to 100 m in some locations. The 9- and 2-year old buffers were planted to approximately 46 m on either side of the stream for a total width of 92 m. The tree zone of all of the buffers includes three or more of the following species: silver maple (*Acer saccharinum* L.), green ash (*Fraxinus pennsylvanica* Marsh.), black walnut (*Juglans nigra* L.), willow (*Salix* spp), cottonwood hybrids (*Populus* spp., e.g., *Populus* clone NC-5326, a designated clone of the US Forest Service Northern Research Station), red oak (*Quercus rubra* L.), bur oak (*Quercus macrocarpa* Michx.), and swamp white oak (*Quercus bicolor* Willd.). Shrub species include chokecherry (*Prunus virginiana* L.), Nanking cherry (*Prunus tomentosa* Thunb.), wild plum (*Prunus americana* Marsh.), red osier dogwood (*Cornus stolonifera* Michx.), and ninebark (*Physocarpus opulifolius* Max.). The grass zones consist of mixtures of several native warm season grasses and up to 15 native forb species. The oldest site is a contiguous section consisting of a downstream segment planted in 1990 and an upstream segment planted in 1994; thus, vegetation on the entire site was at least 14 years old at the time of the study.

The intensively grazed pasture had a width ranging from 50 m to 200 m and was dominated by short bluegrass (*Poa pratensis*), with sparse residual woody vegetation present near the stream. The pasture site was bordered on the outside (upslope) edge by tall, unmowed grass at a distance of 25-100+ m from the stream. Within the row crop area, crops were planted as close as possible to the stream edge. As the stream was meandering, this left areas on the inside of the meanders that could not be farmed (<2-30 m wide on each side of the stream) which were dominated by cool-season grasses and forbs. Example images of a buffer, the pasture, and the crop site are shown in Figure 1.

Bird surveys: All sites were surveyed for breeding birds eight times between May 15 and July 10, 2008 using 10 minute point counts with a 50 m recording radius (Ralph et al. 1995). At each site, three-to-seven non-overlapping point-count locations were placed randomly along each stream reach and extended across the stream on both sides. A total of 21 plots were surveyed with the number of plots per site based on its length and complexity (7 plots on the 14+ year-old buffer, 4 plots on the 9 year-old buffer, 4 plots on the 2 year-old buffer, 3 plots on the crop site, and 3 plots on the pasture site). Plot centers were at least 100 m apart to ensure non-overlapping plots. Surveys began at sunrise and ended by 9:30 a.m. each day. Data were not collected on mornings with high winds (>10 mph) or rain because these factors could affect bird activity. All birds seen or heard within each 10 minute point-count period were recorded. Non-resident migrant birds observed were recorded but were not included in data analysis. Aerial foragers were included because they were utilizing the habitat to search for prey (Best et al. 1995). In an effort to prevent multiple recording of the same individual on any given survey day, the location of each observed individual was recorded on a diagram of each plot and notations were made if a bird flew from its initial location during the count period. To reduce temporal bias, the order in which the sites were surveyed was rotated between days. No attempt was made to establish whether the birds were nesting within the plots; hence, plots may have provided food or resting locations in lieu of nesting habitat. Similar methodologies were used for surveys conducted in 1994, 1997, and 1999, with some differences in specific sites and plot locations within sites resulting from buffer establishment. Birds are often grouped into guilds based on their use of similar resources in a similar manner (Ehrlich et al. 1988). Birds in this study were grouped based on major habitat type such as grassland, forest, edge, shrub, and open country (Table 2).

Habitat characterization: Vegetation sampling was conducted on all of the 21 survey locations between September 5 and 15, 2008. Prior to field sampling, each plot was divided into subplots based on dominant vegetation type as recorded in a Geographic Information System (GIS) (ArcGIS 9.2 2006) from aerial images. Subplot vegetation categories were tree, grass/forb, shrub, shrub/grass, tree/shrub, and crop. The number of subplots per plot was a representation of the heterogeneity of the plots. Data collected within each subplot included canopy height, percent canopy cover (using a spherical densiometer (Peak and Thompson 2006)), total number of trees, shrub density, percent grass/forb cover, and dominant tree, shrub, grass, and forb species. Canopy cover was estimated for the subplots with trees or shrubs. Shrubs were defined as woody vegetation at least 0.3 m tall and with a dbh < 5 cm. GIS was also used to estimate the percent cover of each dominant vegetation type comprising survey plots.

Data analysis: To compare avian use of the re-established riparian buffers to cropped fields or pastures, total bird abundance, species richness, and Shannon-Wiener Diversity were calculated for each of the survey plots (Margules and Usher 1981; Dunn 2004; Milne et al. 2007). Total bird abundance was calculated for each of the plots by summing the maximum number of individuals of each species observed across all eight surveys conducted at each plot. Abundance was then averaged across plots for each site to obtain a site-based estimate. Species richness was calculated by counting the total number of species observed across all surveys conducted at each point count plot; plots were then averaged to obtain a site-based estimate. The Shannon-Wiener Diversity Index (H') (Shannon 1948) combines species richness and the relative abundances of species observed for an integrated measure of bird response (Molles 2008). The log of richness and abundance was computed to reduce the variance in the data. Then a generalized linear model (Proc GLM) and Tukey's test were run on the richness, abundance and diversity data to test for differences among the five sites with a significance level of $P<0.05$ (SAS Institute 2003). There has been a recent trend in avian studies to use distance methodology for estimates of bird density and abundance (Rosenstock et al. 2002). We chose not to use distance sampling because we did not estimate bird density on the sites but instead determined presence/absence of species. Additional limitations for distance sampling were that buffer sites were of limited width and that we did not have sufficient observations for many species for accurate density estimates (Bart et al. 2004). Toms et al. (2006) determined that point counts provide a reasonable estimate of abundance, which was one of the data analyses conducted in this study.

Regression analysis was used to compare the number of species from the 2008 survey to those from the previous surveys conducted at the study sites. Regression analysis was performed using SigmaPlot 11 (2008). Due to spatial limitations associated with buffer site design, several individual sample plots were located in close proximity (100 m between plot centers). Although we made attempts to reduce or eliminate the double counting of individuals, individual birds may possibly have flown between and been counted at more than one site. Any impact this may have had on treatment differences would likely be limited to the chronosequence of buffer ages, however, due to their proximity, and not between buffer and crop/pasture sites. Yet, given the issue of replication, caution should be used in extrapolating results to other areas.

Non-metric multidimensional scaling (NMS) was used to analyze bird community response among sites utilizing bird species abundance data. Prior to analysis, rare bird species (those observed only one time at one plot) were removed from the bird community data set. PC-ORD (McCune and Mefford 1999) was used to perform the NMS analysis with the Bray-Curtis distance measure. Preliminary NMS analyses were run with as many as six ordination axes, but the final analysis was run with only two axes. Additional dimensions provided only small reductions in stress. The specifications for the final NMS run included two axes, a random starting configuration, and one run with real data. This solution had a final stress of 14.91 and a final instability of 0.0045 based on 21 iterations of randomized data. The amount of variation explained by the two axes was high ($r^2 = 0.87$) and the amount of stress in the final solution was satisfactory for ecological community data (McCune and Grace 2002). Vegetation variables were graphed as vectors (arrows) on the NMS plots. Vectors indicate the direction and relative strength of correlations among bird species and vegetation in ordination space. A Monte-Carlo randomization was used to evaluate the test statistic (McCune and Grace 2002).

Total species richness for the buffer sites in this survey was compared to total species richness from the 1994, 1997, and 1999 surveys conducted in the same area. Past surveys did not analyze species abundance or diversity. They only measured total species richness and point counts were 10-15 minutes in length, with the number of point count plots varying between the survey years. In 1994, 17 plots were surveyed; in 1997, 21 plots were surveyed; in 1999, 19 plots were surveyed; and in 2008, 21 plots were surveyed. Some of the sites from the 2008 survey were different than those in the past surveys, although all sites were planted in a similar design. In addition, the same survey plots were not used from year to year, potentially affecting specific vegetation within a plot and/or bird use. A polynomial regression was used to characterize the relationship between buffer age (the number of years since the sites were planted) and the number of bird species observed at each site using the data from all four surveys.

Results

In total, 2255 individuals from 54 bird species were observed across all surveys and locations during sampling in 2008. Five of the observed species are listed as species of regional importance by Partners in Flight (PIF 2008), including the Dickcissel (*Spiza americana*), Northern Flicker (*Colaptes auratus*), Brown Thrasher (*Toxostoma rufum*), Western Meadowlark (*Sturnella neglecta*), and Sedge Wren (*Cistothorus platensis*). The Iowa Wildlife Action Plan (IDNR 2006) lists two of the same species as being breeding bird species of greatest conservation need in Iowa (Sedge Wren and Dickcissel) as well as three additional species (Least Flycatcher (*Empidonax minimus*), Field Sparrow (*Spizella pusilla*), and Bobolink (*Dolichonyx oryzivorus*)) observed in our studies.

The highest average bird abundance was found in the 2 year-old buffer (42.8 birds) and the lowest was in the crop site (24.7 birds) (Figure 2). The buffer sites did not have significantly different total bird abundance when compared to each other, but differed significantly from pasture (14+ yr, P=0.019; 9 yr, P= 0.044; 2 yr, P=0.004) and crop sites (14+ yr, P=0.007; 9 yr, P=0.019; 2 yr, P=0.002).

Average species richness across all survey dates was highest in the 14+ year-old buffer (20.9) and lowest in the crop site (10.7) (Figure 2). The 14+ year old buffer had significantly greater richness than the crop site (14+ yr, P=0.009), but none of the buffers were significantly different from each other. The buffers and crop site were not significantly different from the pasture site.

The Shannon-Wiener Diversity Index was also highest in the 14+ year old buffer (2.82) and lowest in the crop site (2.09) (Figure 2). Diversity did not significantly differ between the different ages of buffers. None of the buffers were significantly more diverse than the pasture site. All buffer sites had significantly higher diversity than the crop site (14+ yr, $P<0.001$; 9 yr, $P=0.033$; 2 yr, $P=0.003$).

The 14+ year-old buffer had the most vertical and horizontal stratification of all of the sites, with a greater variety of tree and shrub sizes (Figures 4 and 5). This buffer attracted a greater number of bird species known to be associated with forest and edge habitats (i.e., Baltimore Oriole (*Icterus galbula*), Eastern Phoebe (*Sayornis phoebe*), Least Flycatcher, and Red-eyed Vireo (*Vireo olivaceus*) (Ehrlich et al. 1988). The 9 year-old buffer did not yet have many large trees and as a result attracted many bird species known to respond to shrub and edge habitat, such as the Common Yellowthroat (*Geothlypis trichas*), American Goldfinch (*Carduelis tristis*), and Gray Catbird (*Dumetella carolinensis*). The 2 year-old buffer was dominated by grasses but contained some large trees and shrubs along the stream edge that predated buffer establishment. Primarily grassland birds, such as the Savanna Sparrow (*Passerculus sandwichensis*), Dickcissel, and Western Meadowlark, were found within this buffer (Table 2).

NMS analyses identified 14 species that were strongly correlated ($r > 0.50$; Table 1) with Axis 1 and three with Axis 2. Bobolink, Killdeer (*Charadrius vociferous*), Savannah Sparrow, and Western Meadowlark abundance were all positively correlated with Axis 1. The abundance of American Robin (*Turdus migratorius*), Brown-headed Cowbird (*Molothrus ater*), Common Yellowthroat, Eastern Phoebe, Gray Catbird, House Wren (*Troglodytes aedon*), Least Flycatcher, Northern Cardinal (*Cardinalis cardinalis*), Song Sparrow (*Melospiza melodia*), and Yellow Warbler (*Dendroica petechia*) were all negatively correlated with this axis. American Robin, Canada Goose (*Branta canadensis*), and Cliff

Swallow (*Petrochelidon pyrrhonota*) were all negatively correlated with Axis 2. Other bird species were only weakly associated with Axis 1 or 2.

The NMS revealed a distinct clustering of bird species (Figure 3). Axis 1 represents a gradient with Grassland species (i.e., Dickcissel, Sedge Wren, Western Meadowlark, Savanna Sparrow and Bobolink) clustered together on one end and most woodland and edge species clustered on the other. Ten vegetation variables were significantly correlated with the bird community result. Grassland bird species were associated with plots without trees (NoTree), the lowest shrub density category (ShrubD0), and the percent of the plot in grass (% Area Grass). The maximum tree density (MaxTD), percent of the area with trees (%AreaTre), percent of the area with shrubs (%AreaSh), maximum canopy height (MaxCH), and the number of subplots for vegetation surveys (#Subplots) all aligned with the edge and woodland species.

Analysis of site NMS (Figure 3) revealed clustering among plots within given site types, with the exception of the 14+ year-old buffer (Buf14) at the bottom of the NMS graph. This plot was dominated by grass and shrubs, similar to the younger buffers, whereas the other plots located in this buffer had a greater proportion of trees. Vegetation variable vectors were consistent with the plots (Figure 3) as illustrated by grass variables having strong associations with grass dominated sites and tree/shrub variables having strong associations with the sites having a greater percentage of trees and shrubs.

The pasture was dominated by bluegrass. Many of the birds found on the pasture were grassland birds (i.e., Western Meadowlark, Bobolink). The crop site had a narrow strip of grass and small shrubs (<2 – 30 m on each side of the channel) that could comprise potential habitat for birds. The rest of the site was composed of corn that had not yet reached maturity. Most of the birds using the site (i.e., Red-winged Blackbird (*Agelaius phoeniceus*), Song Sparrow, and Common Yellowthroat) were concentrated in the thin grass strip. The only bird species consistently observed in the crop field was Killdeer.

A total of 42 bird species was observed on the 14+ year-old, 27 species on the 9 year-old, 28 species on the 2 year-old riparian buffer sites, respectively; 23 species were found in the pasture and 14 species were found in the row crop site. When comparing the bird survey conducted in 2008 study with past studies (Table 2), as the age of the buffer increased there was a general trend of an increase in the number of bird species observed at a decreasing rate (Figure 4). The polynomial regression plot shows that 64.2 % of the variance is described by the age of the buffer (P=0.041).

Discussion

These results support the assumption that wildlife habitat, as manifested by an increase in bird diversity, can be enhanced by re-establishing riparian buffers within the highly modified and fragmented landscape of north-central Iowa. However, these results also demonstrate the importance of identifying a focal or target species or group of species as the first step in habitat restoration, with their requirements guiding the design (Miller and Hobbs 2007). The data followed a trend where the re-established riparian buffers included in this study had higher bird abundance, richness, and diversity than the crop and pasture sites, apparently responding to the greater habitat heterogeneity. However, these differences were not significant when comparing the buffer and pasture sites for species richness and diversity. Because eight species found in the buffers are listed as species of regional importance by PIF (2008) or breeding birds of greatest conservation need (Iowa DNR 2006), similar sites could be important for preservation of these species in Iowa. However, this study was limited to presence/absence of a species, and additional research on reproductive success is warranted. Additional studies documenting changes in bird species abundance, richness, and diversity as the riparian perennial plant systems mature would also be useful in directing management recommendations.

In general, total bird abundance, species richness, and the Shannon-Wiener Diversity Index were greatest in the riparian buffers (Figure 2), likely because they provide habitat that is more suitable to more bird species than do row crop fields or pastures. There were differences in the bird species using the different aged buffers, even though the bird analyses were not significantly different from each other. Because habitat use differs among species, the different aged buffers might have appealed to different species (Best et al., 1995) due to variations in habitat structure of the combinations of trees, shrubs, and grass.

The row crop site we surveyed had the lowest habitat value for the overall bird community (Figure 2). Best et al. (2001) found that few species are residents of row crop fields; many birds use them only for food and are more likely to do so when associated with an adjacent grass or wooded habitat. In this study, the strip of grass with a few shrubs in the meander belt adjacent to the row crop fields provided some habitat and may have increased the use of the row crop fields. Most of the birds on the crop site were concentrated in the strip of grass, which provided only a limited area of habitat to support the bird species found there. Best et al. (1997) found 1.4 to 10.5 times greater abundance of birds in CRP when compared to row crop fields but similar species richness.

The crop and pasture sites had less suitable habitat for many bird species, presumably due to a lack of habitat structure in the form of trees, shrubs, or tall grass. The pasture site had few places to perch and no tall grass. Many of the birds found on the pasture site were species known to prefer open landscapes. Some birds made use of the only large tree in the pasture. Others moved periodically from the tall grass that was planted adjacent to one side of the pasture. They did not have much horizontal or vertical stratification and so could not support as many different species as could the buffers.

The study sites varied in terms of width of riparian vegetation and degree of horizontal and vertical stratification. In general, bird species richness and abundance increased with amount and width of woody vegetation, which is similar to trends found by other researchers (Stauffer and Best 1980; Deschenes et al. 2003). Martin et al. (2006) found that riparian areas with a tree layer were species rich and were dominated by small-bodied insectivores while areas that had been cleared of the tree layer were dominated by a few generalist-foragers. Furthermore, different bird species exhibit affinities for different habitats (Dinsmore et al. 1984) and have varying vegetation structure requirements (Benton et al. 2003), which explains why certain species were more common on some sites rather than others (Table 2). The narrow width (total widths between approximately 36 and 100 m) of most of the buffers often attracts edge species, reflecting the large proportion of edge habitat (and Thompson 2006). Peak and Thompson (2006) studied sites with riparian widths of 55-95 m and considered all of the sites to be edge habitats. These narrow habitat patches generally do not meet the minimum habitat requirements for area of forest for most area-sensitive species, likely explaining their absence (Best et al. 1995).

The variation in habitat patches created by re-established riparian buffers may be able to provide habitat for many species, including grassland bird species, when designed appropriately. Grassland birds have experienced large population declines across the Midwest due to loss of habitat (Murphy 2003). Because so little native grassland habitat remains, established grassland habitats (such as riparian buffers) can be effective for the conservation of some grassland species (Vickery et al. 1999). Many obligate grassland species were observed on at least one of the buffer sites (Table 2), including Sedge Wren, Savannah Sparrow, Dickcissel, and Western Meadowlark (Vickery et al. 1999). Several facultative grassland birds were also observed: Canada Goose, Mallard (*Anas platyrhynchos*), Blue-winged Teal (*Anas discors*), Ring-necked Pheasant (*Phasianus colchicus*), Killdeer, Mourning Dove (*Zenaida macroura*), Eastern Kingbird (*Tyrannus tyrannus*), Common Yellowthroat, Red-winged Blackbird, and Brown-headed Cowbird (Vickery et al. 1999). This indicates that the buffers were able to provide habitat for many grassland species, including three species of regional concern (Dickcissel, Sedge Wren, and Western Meadowlark) (PIF 2008).

Regression analysis illustrates that a large number of species initially colonized the sites, but as the sites aged, the rate of adding new species declined (Figure 4). Total species richness for these sites appears to have reached an asymptote at around 42 species. The observed trend is likely the result of the buffers not being wide enough or large enough to provide adequate habitat for some forest species (Rosenzweig 1995; Peak and Thompson 2006) or having too many trees/shrubs for some grassland species (Best et al. 1995). Farley et al. (1994) found that the complexity of the bird assemblage generally increases with the age of the site, but had not yet observed maximum species richness on their sites. Peak and Thompson (2006) reported an average species richness of 25 species/site on narrow buffers (55-95 m). Based on their findings, the highest species richness in this study of 21.9 on the oldest buffer may be near the maximum number of species supported by a buffer strip of this width.

Results of this and similar studies can provide useful recommendations for the design and establishment of riparian buffers to enhance avian habitat within intensively modified agricultural landscapes. As a general rule, buffer design should enhance the resources required by the focal species or group. If the management goals are to attract specific target species or species groups, then their habitat requirements guide the design of the buffer (Henningson and Best 2005; Miller and Hobbs 2007). Features of the surrounding matrix can also strongly influence the bird assemblages within the riparian buffers (Saab 1999; Miller and Hobbs 2007). Agricultural land with interspersed natural habitat can potentially attract nest predators, brood parasites, and exotic species (Best et al. 1997; Saab 1999; Deschenes et al. 2003) However, riparian buffers may support higher bird diversity and therefore may also attract desirable species such as insectivores (Deschenes et al. 2003; Table 2). Our results show that species often viewed as being negative (i.e., Red-winged Blackbird and Brown-headed Cowbird) and desirable species (i.e. Least Flycatcher and Tree Swallow) used the buffers.

Reinstating heterogeneity in the monoculture dominated landscape of north-central Iowa is important in order to conserve and enhance biodiversity (Fisher et al. 2006). We have shown that riparian buffers can support a wide variety of bird species based on the vegetation structure of the site, but that the spatial distribution of different habitats also affects species use (Saab 1999; Fisher et al. 2006). Habitat connectivity is important because patches that are closer to each other will have higher bird species richness than patches that are isolated (Saab 1999; Fisher et al. 2006). By increasing the amount of riparian buffers on the landscape, connectivity would increase, which could also increase the number of birds using the buffers.

Conclusions

In response to water quality and other concerns, state and federal agencies have established incentive programs for landowners to convert sensitive riparian lands from agricultural uses to perennial vegetation. The resulting establishment of perennial plant systems composed of trees, shrubs, and/or grass provides an opportunity to increase habitat for many bird species, including those identified as being of greatest conservation need. However, while riparian buffer design criteria identify wildlife habitat as an important function of such systems, little concern is given for the species- or guild-specific responses to buffer design. This study illustrates that wildlife habitat, as manifested by an increase in bird diversity, can be enhanced by re-establishing riparian buffers within the highly modified and fragmented landscape of north-central Iowa. However, this study also emphasizes the importance of matching buffer design and management to species requirements if the management goals are to attract specific target species or species groups. While the study design strictly limits the conclusions from this study to the sites and watersheds assessed, we believe that the inferences can be used to inform the design and management of similar conservation buffer systems within the region to enhance avian or other wildlife habitat. This study was limited to presence/absence of a species, and additional research on reproductive success is warranted. Additional studies documenting changes in bird species abundance, richness, and diversity as the riparian perennial plant systems mature would also be useful in directing management recommendations.

Conservation practices, including riparian buffers, provide many benefits that affect more than just the landowners who establish them. Riparian buffers can improve water quality (Schultz et al. 2004), aesthetics, and, as this study has shown, provide habitat for many bird species. When compared to the surrounding agricultural matrix, buffers increase landscape heterogeneity and provide habitat to a broader suite of breeding birds than found in agricultural lands alone. With such a large percentage of the Midwestern U.S. landscape in row crop agriculture, it is important to understand the relationship between landscape structure and avian diversity in such modified systems.

Acknowledgements

This research has been funded through the University of Missouri Center for Agroforestry. We thank the landowners who allowed us access to their property and Jaymi LeBrun for her assistance in setting up the bird survey.

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Table 1. Observed bird species and their correlations with axes from non-metric multi-dimensional scaling (NMS) ordination

Bird Species	**Code**	**Axis 1, r**	**Axis 2, r**
Canada Goose (*Branta canadensis*)	CAGO	-0.219	-0.578
Mallard (*Anas platyrhynchos*)	MALL	0.464	0.373
Blue-winged Teal (*Anas discors*)	BWTE	-0.193	-0.215
Ring-necked Pheasant (*Phasianus colchicus*)	RNEP	-0.419	0.164
Great Blue Heron (*Ardea herodias*)	GBHE	0.024	0.309
Killdeer (*Charadrius vociferous*)	KILL	0.829	-0.034
Mourning Dove (*Zenaida macroura*)	MODO	-0.243	-0.107
Chimney Swift (*Chaetura pelagica*)	CHSW	-0.172	-0.127
Eastern Wood-Pewee (*Contopus virens*)	EAWP	-0.405	0.214
Least Flycatcher (*Empidonax minimus*)	LEFL	-0.556	0.204
Eastern Phoebe (*Sayornis phoebe*)	EAPH	-0.632	-0.064
Great-crested Flycatcher (*Myiarchus crinitus*)	GCFL	-0.413	-0.004
Eastern Kingbird (*Tyrannus tyrannus*)	EAKI	0.388	-0.202
Warbling Vireo (*Vireo gilvus*)	WAVI	-0.479	-0.028
Red-eyed Vireo (*Vireo olivaceus*)	REVI	-0.413	-0.004
Blue Jay (*Cyanocitta cristata)*	BLJA	-0.36	0.206
American Crow (*Corvus brachyrhynchos*)	AMCR	-0.403	0.311
Tree Swallow (*Tachycineta bicolor*)	TRES	0.187	-0.375
Northern Rough-winged Swallow (*Stelgidopteryx serripennis*)	NRWS	0.442	-0.232
Cliff Swallow (*Petrochelidon pyrrhonota*)	CLSW	0.184	-0.719
Barn Swallow (*Hirundo rustica*)	BARS	0.23	-0.059
House Wren (*Troglodytes aedon*)	HOWR	-0.649	0.155
Sedge Wren (*Cistothorus platensis*)*	SEWR	0.107	-0.286
American Robin (*Turdus migratorius*)	AMRO	-0.586	-0.435
Gray Catbird (*Dumetella carolinensis*)	GRCA	-0.657	0.021
Brown Thrasher (*Toxostoma rufum*)*	BRTH	-0.2	0.323
Cedar Waxwing (*Bombycilla cedrorum*)	CEDW	-0.249	0.208
Yellow Warbler (*Dendroica petechia*)	YWAR	-0.734	0.164
Common Yellowthroat (*Geothlypis trichas*)	COYE	-0.567	-0.199

* Indicates regionally important species (PIF, 2008)

Table 1. Continued

Bird Species		**Axis 1, r**	**Axis 2, r**
Chipping Sparrow (*Spizella passerine*)	CHSP	-0.332	-0.07
Field Sparrow (*Spizella pusilla*)	FISP	-0.174	-0.066
Savannah Sparrow (*Passerculus sandwichensis*)	SAVS	0.759	0.001
Song Sparrow (*Melospiza melodia*)	SOSP	-0.614	0.053
Northern Cardinal (*Cardinalis cardinalis*)	NOCA	-0.526	-0.19
Rose-breasted Grosbeak (*Pheucticus ludovicianus*)	RBGR	-0.42	0.076
Indigo Bunting (*Passerina cyanea*)	INBU	-0.369	-0.266
Dickcissel (*Spiza americana*)*	DICK	0.025	-0.348
Bobolink (*Dolichonyx oryzivorus*)	BOBO	0.65	-0.215
Red-winged Blackbird (*Agelaius phoeniceus*)	RWBL	0.441	0.002
Western Meadowlark (*Sturnella neglecta*)*	WEME	0.65	-0.01
Common Grackle (*Quiscalus quiscula*)	COGR	-0.093	-0.266
Brown-headed Cowbird (*Molothrus ater*)	BHCO	-0.75	0.188
Baltimore Oriole (*Icterus galbula*)	BAOR	-0.497	-0.253
American Goldfinch (*Carduelis tristis*)	AMGO	-0.473	-0.548

* Indicates regionally important species (PIF, 2008)

Table 2. Bird species observed during bird surveys on riparian buffers, row crop fields, and pastures in 1994, 1997, 1999, and 2008 along Bear and Long Dick Creeks and the primary habitats where these species are found (Ehrlich et al. 1988).

Site		Pasture			Crop				Buffer 06	Buffer 97 or 99			Buffer 90/94			
Year site was sampled		**97**	**99**	**08**	**94**	**97**	**99**	**08**	**08**	**97**	**99**	**08**	**94**	**97**	**99**	**08**
Species	**Habitat**															
Canada Goose	freshwater marsh, meadow								X							X
Wood Duck	wooded swamp, flooded forest														X	X
Mallard	shallow pond, lake, marsh			X				X		X	X				X	
Blue-winged Teal	prairie potholes, marsh, pond									X	X		X	X		X
Northern Bobwhite	tall grassland, brushy fields														X	
Ring-necked Pheasant	open country, cultivated area					X	X		X	X		X	X	X	X	X
Great Blue Heron	freshwater marsh, meadow			X											X	X
Red-tailed Hawk	woodland and open country		X					X			X				X	
Killdeer	field, meadow, pasture	X		X	X	X	X	X	X	X	X	X	X	X		
Rock Pigeon	cities, towns, rural						X			X					X	
Mourning Dove	open woodland, ag. with few trees	X				X	X	X	X	X		X	X	X	X	X
Yellow-billed Cuckoo	open wood w/dense undergrowth														X	
Great Horned Owl	conif. or decid. forest										X				X	
Chimney Swift	woodland, near human habitat								X							X
Belted Kingfisher	along watercourses												X	X	X	X
Red-headed Woodpecker	decid. woodland, open with trees														X	
Downy Woodpecker	decid./mixed forest								X							
Northern Flicker	below tree line, open ground															X
Eastern Wood-pewee	decid./mixed forest, edge														X	X
Least Flycatcher	open decid./mixed woodland															X
Eastern Phoebe	open and riparian woodland								X			X	X	X		X
Great-crested Flycatcher	decid. forest edge, woodland															X

Table 2. Continued

Site		Pasture			Crop				Buffer 06	Buffer 97 or 99			Buffer 90/94			
Year site was sampled		**97**	**99**	**08**	**94**	**97**	**99**	**08**	**08**	**97**	**99**	**08**	**94**	**97**	**99**	**08**
Species	**Habitat**															
Eastern Kingbird	farmland, open woodland			X			X		X	X				X	X	X
Loggerhead Shrike	open fields and woodlands													X		
Warbling Vireo	open decid./mixed woodland											X				X
Red-eyed Vireo	decid. forest and woodland															X
Blue Jay	decid./mixed forest, open woodland								X	X			X	X	X	X
American Crow	open woodland/farmland			X			X			X		X		X	X	X
Purple Martin	open country, rural, esp. near water												X	X		
Tree Swallow	open, woodland edge near water	X		X						X				X		X
N. Rough-winged Swallow	open country near water			X				X	X			X				X
Cliff Swallow	open country near water		X	X			X			X		X		X	X	X
Barn Swallow	open country near water		X	X			X	X	X			X	X	X	X	X
Black-capped Chickadee	decid./mixed or riparian woodland												X	X	X	
White-breasted Nuthatch	decid./mixed forest, edge															X
House Wren	open woodland								X			X			X	X
Sedge Wren	wet meadow, dry marsh			X		X			X	X		X	X		X	
Eastern Bluebird	forest edge, open w/scattered trees			X												
American Robin	open with trees, generalist			X	X	X	X	X	X	X		X	X	X	X	X
Gray Catbird	dense brush, shrubland								X	X		X		X	X	X
Brown Thrasher	brush and shrubland			X				X		X		X	X	X	X	X
European Starling	habitat generalist	X					X						X	X	X	X
Cedar Waxwing	woodland, forest edge															X
Yellow Warbler	habitat generalist											X	X	X	X	X

Table 2. Continued

Site		Pasture			Crop				Buffer 06	Buffer 97 or 99			Buffer 90/94			
Year site was sampled		**97**	**99**	**08**	**94**	**97**	**99**	**08**	**08**	**97**	**99**	**08**	**94**	**97**	**99**	**08**
Species	**Habitat**															
Yellow-rumped Warbler	conif./mixed forest											X		X		
Common Yellowthroat	overgrown field, hedgerow			X		X	X	X	X			X	X	X	X	X
Eastern Towhee	forest edge, riparian thickets														X	
Chipping Sparrow	open forest, forest edge		X				X					X	X	X	X	X
Field Sparrow	old field, brush, forest edge				X	X			X	X			X		X	X
Vesper Sparrow	grassland, prairie, savanna				X	X				X			X	X		
Savannah Sparrow	grassland, meadow			X						X			X		X	
Song sparrow	dense veg. along water	X	X	X		X	X	X	X	X		X	X	X	X	X
Northern Cardinal	thicket, dense shrub, residential								X			X			X	X
Rose-breasted Grosbeak	decid. forest, woodland								X						X	X
Indigo Bunting	decid. forest edge, clearing								X			X				X
Dickcissel	grassland, meadow								X	X			X	X	X	
Bobolink	tall grass, prairie, grain field			X									X			
Red-winged Blackbird	marsh, shruby field	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
Eastern Meadowlark	grassland, savanna, fields									X			X	X		
Western Meadowlark	grassland, savanna, pasture		X	X					X						X	
Common Grackle	partly open w/scattered trees		X	X	X	X		X	X	X		X	X	X	X	X
Brown-headed Cowbird	woodland/edge, grassland	X		X	X	X		X	X			X	X	X	X	X
Baltimore Oriole	open/riparian woodland			X					X			X			X	X
House Finch	open woodland, urban											X				
American Goldfinch	open, scattered tree/brush		X	X		X	X	X	X	X	X	X	X	X	X	X
House Sparrow	cultivated lands, near human habitat	X					X							X	X	
Total # of spp.		8	9	23	7	13	16	14	28	24	26	27	28	32	42	42
Ave. # of spp.		5	6	8	4.1	9	9.3	7	14.6	14.5	13.8	12.6	18	23	20	20

Figure 1. Representative photos from the study area; the left photo is the 14+ aged buffer, the middle photo is of the pasture site, and the right photo is the crop site with grass inside the stream meanders.

(A)

(B)

(C)

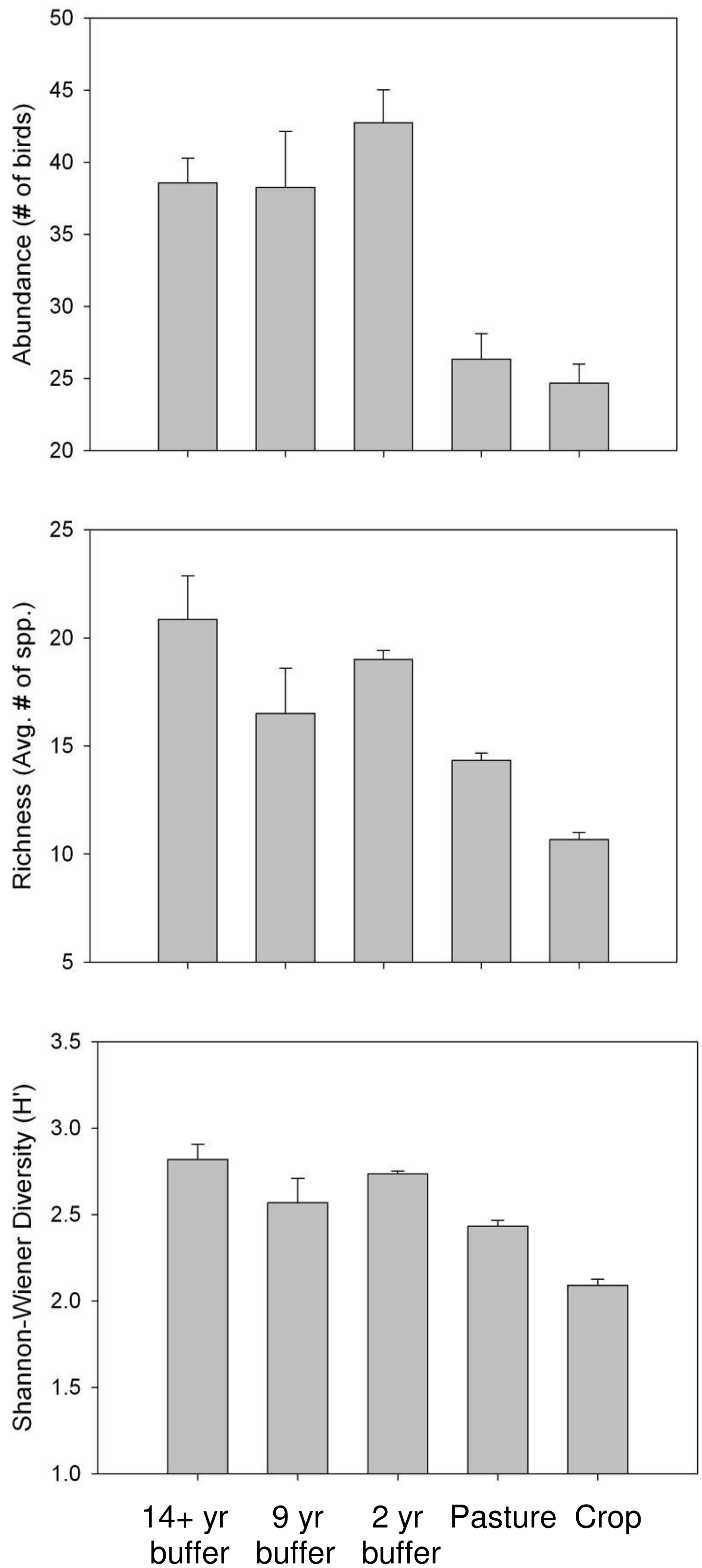

Figure 2. Differences in bird (A) abundance, (B) richness, and (C) diversity across the 5 study sites. The error bars show standard error (SE).

(A)

(B)

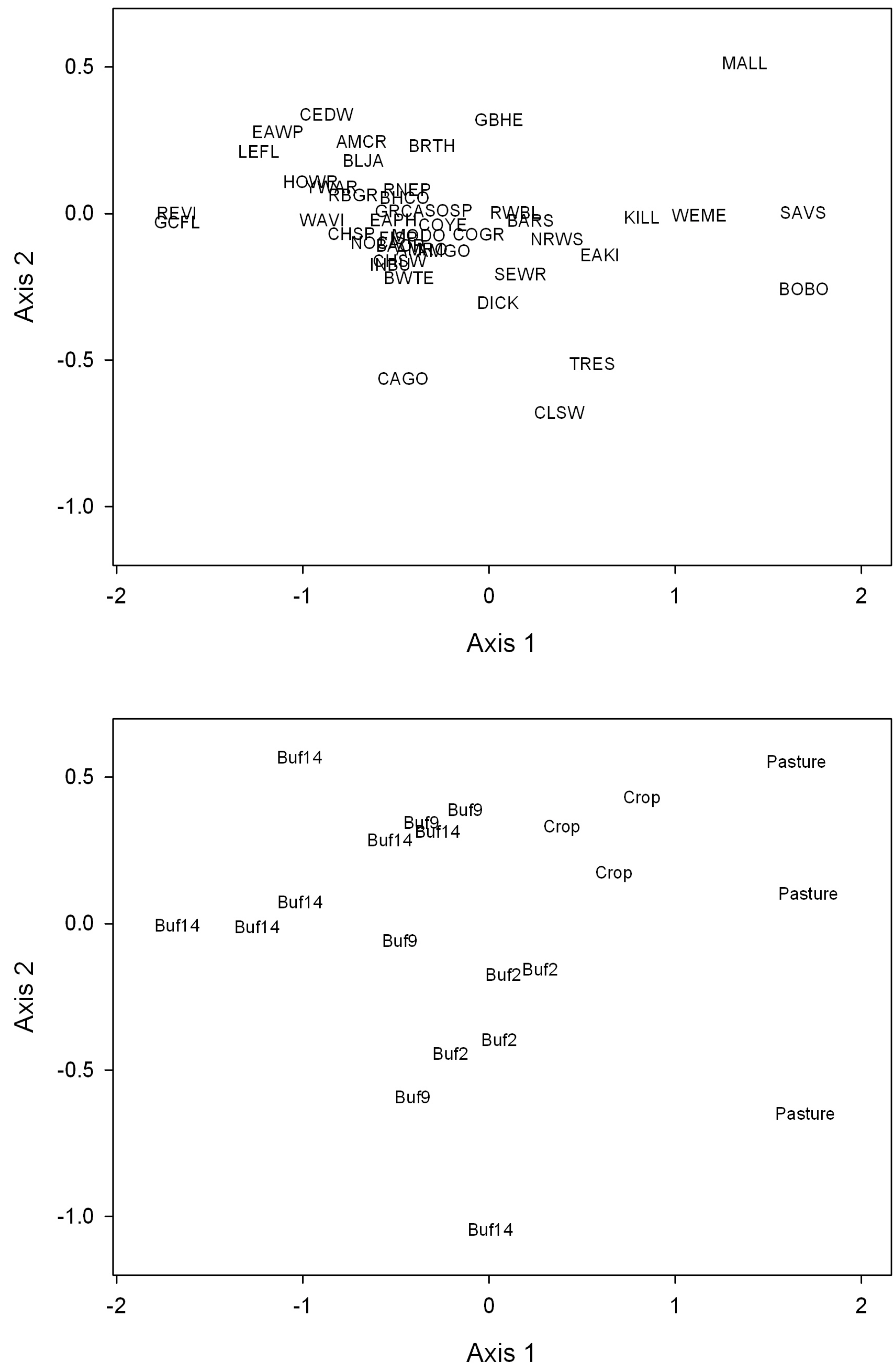

Figure 3. Non-metric multidimensional scaling (NMS) results illustrating (A) variation in bird community composition across all sites (species codes are listed in Table 1), (B) variation and clustering among bird survey plots, and (C) associated vegetation variables. Lengths and directions of the vectors indicate strengths and directions of the correlations. MaxTD = Maximum tree density, %AreaTre = % of the plot with trees, WillowSh = small diameter willows, ShrubD0 = the smallest shrub category, MixGrass = prairie-like mixture, MaxCH = Maximum canopy height, and %AreaSh = % of the plot with shrubs.

(C)

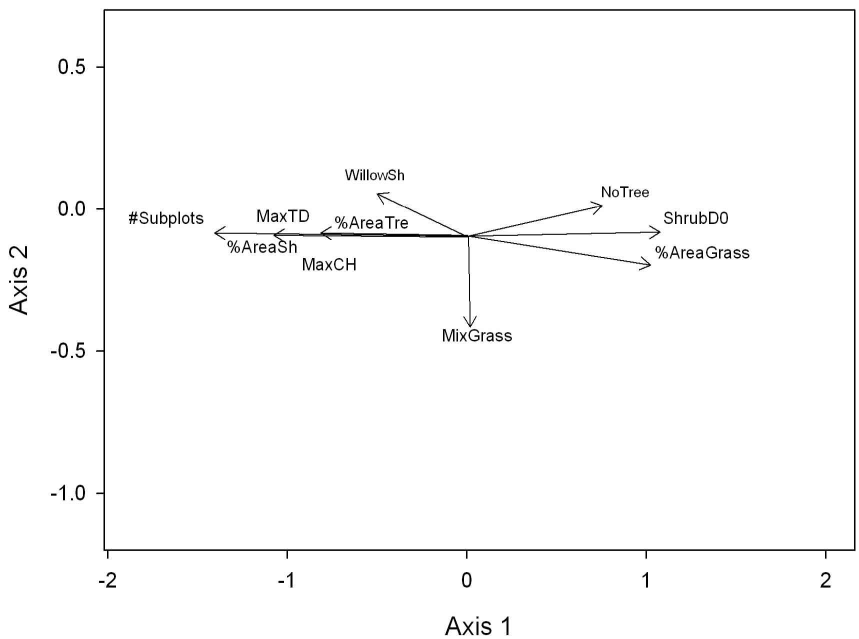

Figure 3. Continued

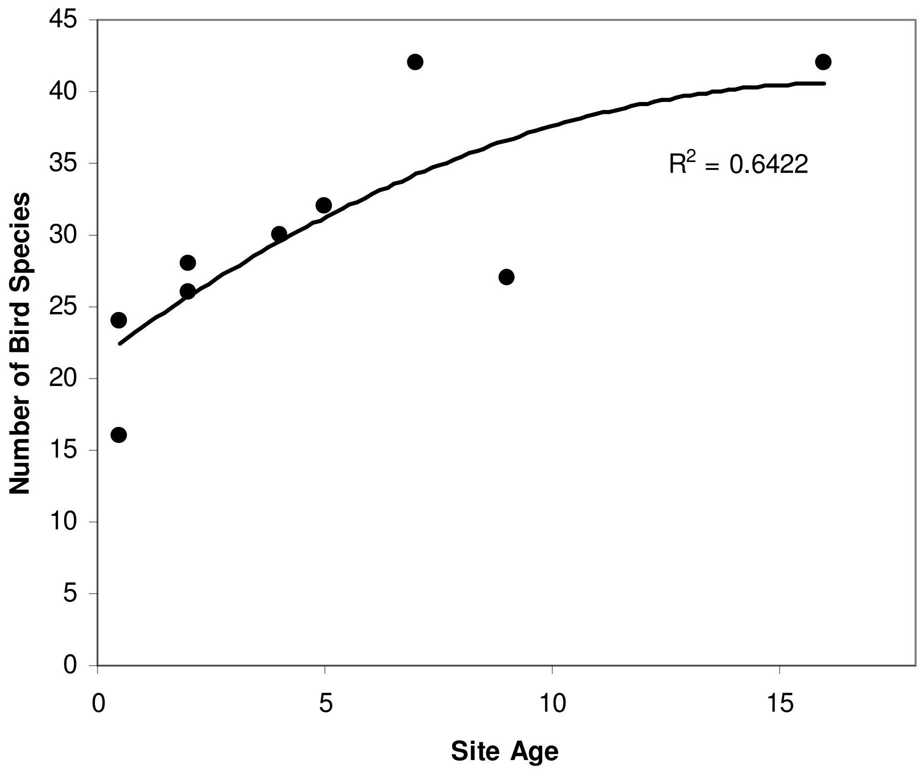

Figure 4. Polynomial regression illustrating the combined total species richness within designed riparian buffers across all surveys. Site age is the age since establishment at the time the survey was conducted.

Chapter 4. General Conclusions

Riparian areas can provide many ecosystem services if they are managed properly. Two of these services, stream bank stability and wildlife habitat, were assessed in this research. The stream bank study was conducted in northeast Missouri and the bird survey was conducted in north-central Iowa. Riparian vegetation in Missouri and Iowa is usually left unmanaged and often is cut down to create more room for farm land. However, the riparian areas in this study included designed riparian buffers in Iowa and remnant forests in Missouri. The designed buffers had a minimum width of 18 m on either side of the stream, had high structural diversity, and were managed. The remnant forests had a minimum width of 5+ m on either side of the stream, had much less structural diversity, and were unmanaged.

Mark Twain Lake in northeast Missouri is impaired by the waters that flow into it and contaminants they may carry. Mark Twain Lake is a major source of drinking water for numerous counties surrounding it (Fletcher and Davis 2005). One of the major water quality concerns for the Mark Twain Lake and the associated Salt River watershed are sedimentation (Dames and Todd 2009). Crooked and Otter Creek watersheds, both subwatersheds in the Salt River watershed, drain primarily agricultural land and have two main sources of sediment, upland or surface erosion and stream bank erosion (Clark et al. 1985). They are found in the Claypan Prairie subregion of the Central Irregular Plains ecoregion (USEPA 2007). The soils in the Claypan Prairie have low infiltration rates and high surface runoff resulting in surface erosion (Blanco-Canqui et al. 2002). Streams in this region are adjusting to changes in upland land-use and changes in the channel upstream and downstream following the channel evolution model (Schumm et al. 1984). Phosphorus binds to sediment and so is associated with soil erosion. Phosphorus inputs to surface waters can accelerate eutrophication which has been identified as one of the main causes of impaired surface water quality (USEPA 1996, Sharpley et al. 2001). No studies have been done in this area to determine how much sediment might be coming from bank erosion. Therefore, the objectives of this study were to use erosion pins to quantify the amount of sediment and phosphorus contributed to the streams from stream bank erosion and to identify vegetative communities/land uses that provided the greatest protection from stream bank erosion. Seasonal erosion measurements were compared to identify if any seasonal patterns in erosion could be observed.

Stream bank erosion is often influenced by the vegetation along the banks, although, that relationship was not evident in this study. The relationship between land use and stream bank erosion is very complex and is affected by soil type, adjacent upland land use, upstream/downstream land use, riparian vegetation, discharge, bank soil moisture, and stage of channel evolution. Our observations suggest that the streams are widening and a few are still downcutting, which suggests that land use may not have as much direct impact on reducing stream bank erosion as when the banks are at a more stable stage. Land use in this case has the indirect effect of contributing more rapidly delivered and more volume of surface runoff because of the clay pan nature of the soils and their response to intense row-crop agriculture. It is assumed that until the channels have adjusted to the greater volume of surface runoff, naturally occurring riparian vegetation may have a reduced effect and bank stability. There was, however, an apparent seasonal trend in stream bank erosion. The highest erosion occurred during the winter months and the lowest during the mid-summer – autumn months. This could be due to freeze-thaw cycling which reduces soil strength and makes the soils less resistant for fluvial erosion (Couper 2003). One way of increasing stream bank stability in pastures includes eliminating cattle from the stream banks and allowing plants to re-establish (Zaimes et al. 2008). In order to stabilize severely and very severely eroding banks in channels that are still actively moving through the steps of the channel evolution model, bank reshaping, structural measures (rock rip-rap, geotextile fabric, tree revetment, etc.), and/or bioengineering (willow stakes, poles, or fascines, brush mattresses, etc.) may be required (USDA-NRCS 1996). In order to be able to reduce the sedimentation of Mark Twain Lake, entire watersheds would need to be evaluated to determine where management practices would be most beneficial and cost-effective (Secchi et al. 2008).

Bird use of riparian areas varies widely and is affected by the amount, quality, type, and width of riparian vegetation. Most of the original Iowa habitat has been converted from prairie to row crop agriculture (corn and soybeans). Perennial riparian habitats, which can be critical to wildlife (Stauffer and Best 1980), have also been simplified and converted to annual row crops due to agricultural intensification. One management practice often used to improve water quality and habitat in riparian areas is the riparian buffer. In this study, the planted buffers consisted of three zones; tree, shrub, and grass/forb. They were managed and were designed to have high habitat heterogeneity. Bird species abundance, richness, and diversity between three different aged riparian buffers (2, 9 and 14 + years) were compared to the bird species observed on sites representing the crop and pasture matrix of this landscape

The buffers had higher bird abundance, richness, and diversity than the crop sites, although the difference was not always significant. These results suggest that re-establishing riparian buffers in highly modified and fragmented landscapes can increase bird diversity.

Different species were attracted to the different aged buffer sites, indicating that the dominant type of vegetation (tree, shrub, or grass) and the height of the vegetation impacts which species may be attracted to the site. This information could be used when designing future buffers if one of the design goals is to attract specific species or a wide range of species. Grassland species were often observed on the younger buffers while woodland and edge species were more frequently observed on the older buffers.

Incentive programs have been established by state and federal agencies to convert sensitive riparian lands from agricultural use to perennial vegetation. The results of the bird survey in this study support the establishment of designed riparian buffers including trees, shrubs, and/or grass to increase habitat for many bird species in highly modified landscapes such as the agriculturally dominated Midwest. However, the results of the stream bank erosion study did not show that natural perennial vegetation (trees and shrubs) was reducing stream bank erosion in northeast Missouri. In Iowa, Zaimes et al. (2008) found that designed riparian buffers had the least amount of erosion when compared to row crop fields and pastures. Future research could be conducted on designed riparian buffers or native forest buffers with grass filters in the same region to determine if more designed and managed riparian vegetation could reduce stream bank erosion.

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Acknowledgements

I would like to thank the University of Missouri Center for Agroforestry for financial support for this project. Thank you to all of the staff, faculty, and students in the Department of Natural Resource Ecology and Management who have supported and encouraged me during my time at ISU. I greatly appreciate my committee members, Dick Schultz, Tom Isenhart, and Lisa Schulte, for their guidance and encouragement.

Thanks to all of the individuals who assisted with field work or provided general support: Kris Knight, Keegan Kult, Jason Palmer, Mustafa Tufekcioglu, Tyler Harms, Eric Kilburg, Nathan Unsworth, Leigh Ann Long, Cammy Willett, Aaron Rector, and all other student employees who have worked as part of the buffer research team through the years. Thanks to the other graduate students who have become wonderful friends and supporters. I am grateful to my family for their encouragement, advice, and support. Special thanks to my husband, Cameron, and my pets, Winston, Tess, Boris and Nika who are all important for making life enjoyable and helping me to be fascinated with the world around us.