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Factors controlling streambank erosion and phosphorus loss in claypan watersheds

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Abstract: Although the targeting and efficacy of erosion management practices have improved, sediment and nutrients continue to degrade surface waters of North America. This study investigated the influence of land use, stream order, and season on streambank erosion and phosphorus (P) transport in the Central Claypan Region of northeast Missouri. The erosion pin method was used to measure bank erosion at 37 sites in Crooked Creek and Otter Creek watersheds from 2008 to 2011. At 18 of the sites, bank vegetation data were collected and watershed characteristics acquired. Bank erosion was highly variable with a mean linear erosion rate of 99 kg m^{-1} y^{-1} (66 lb ft^{-1} yr^{-1}) and a range of –31 to 490 kg m^{-1} y^{-1} (–20 to 330 lb ft^{-1} yr^{-1}); mean bank recession rate was 7.1 cm y^{-1} (2.8 in yr^{-1}) and ranged from –2.5 to 33 cm y^{-1} (–1 to 13 in yr^{-1}). Erosion rates were significantly greater in winter (December to March) than other seasons and were highly correlated to winter and annual stream discharge ($r \geq 0.95$, $p < 0.05$). Watershed-scale estimates showed that streambanks contributed an average of 83% of annual in-stream sediment and 67% of total P loss, clearly demonstrating the impact that bank erosion has on stream water quality in this region. Regression models developed using riparian vegetation and watershed variables accounted for up to 48% of the variability in streambank recession rates, but the models were insufficient for prediction purposes. Overall, the results indicated that current land use, bank vegetation, stream order, and watershed characteristics were not the primary controls on streambank erosion rates. Other factors, such as historic land use changes, stream channelization, and the damming of the Salt River have resulted in major alterations to stream hydrology and geomorphology in these watersheds that have not yet re-equilibrated. These overarching factors, in combination with season, continue to be the main factors controlling streambank erosion in these watersheds.

Key words: land use—phosphorus—riparian vegetation—stream order—streambank erosion—watershed characteristics

Sediment and nutrients are two of the most widespread and costly forms of water pollution in North America (Pimental et al. 1995; Ribaudo et al. 2011). In order to decrease sediment and nutrient loads in streams, their major sources must be identified. As in-field soil conservation practices have become more widespread and erosion from upland sources has decreased, some researchers suggest that the main source of eroded materials in streams is shifting from upland sources to the erosion of gullies and stream channels (Simon and Klimetz 2008; Wilson et al. 2008). There is a growing consensus in the literature that streambank erosion is almost always a significant source of stream sediment (Sekely et al. 2002; Simon and Rinaldi 2006; Wilson et al. 2008, 2014; Schilling et al. 2011; Willett et al. 2012) and in many instances it is the dominant source (Simon and Rinaldi 2006; Wilson et al. 2008, 2014; Schilling et al. 2011; Willett et al. 2012). Using bank erosion rates and upland erosion estimates, Willett et al. (2012) showed that 79% to 96% of the total in-stream sediment load was attributable to streambanks in two agricultural watersheds within the Central Claypan Region of Missouri (these watersheds are also the focus of this study). Wilson et al. (2008, 2014) used radionuclide tracers to track the sources of eroded sediment in five Conservation Effects Assessment Project (CEAP) watersheds. These studies found that during each runoff event, sediment transport was mostly from channel sources (54% to 80%) as opposed to eroded surface soils, and in all sites but one, the channel sources were dominated by collapsed bank material. Streambank erosion has also been shown to be an important source of carbon (C), nitrogen (N) (Willett et al. 2012), and phosphorus (P) (Zaimes et al. 2008a, 2008b) exported from agricultural watersheds. Willett et al. (2012) reported that an average of 23% of the total N exported annually from two Missouri claypan watersheds was derived from eroded streambanks. This is further supported in a two-year grazing study (Schwarte et al. 2011), which found that the major source of sediment and P in a pasture stream in Iowa was eroding streambanks, and not surface runoff or fecal deposition.

Natural resource managers have long used vegetation as a bank stabilization technique. However, whether riparian vegetation has stabilizing or destabilizing effects on streambanks depends on various characteristics of the plant community. Roots in the soil of streambanks have a stabilizing effect when they reinforce the soil matrix and remove water from the root zone (Simon and Collison 2002). Dense herbaceous vegetation has been shown to reduce bank erosion by decreasing soil moisture and protecting soils from freezing (Wynn and Mostaghimi 2006), two processes that increase erodibility, especially in winter months when bank erosion rates are typically greatest (Wolman 1959; Willett et al. 2012). However, plants may have a destabilizing effect and increase bank erosion when rainfall collects and infiltrates around the stems of plants (Simon and Collison 2002), or when banks are scoured by turbulent stream flows created by tree roots (McBride et al. 2007).

Watershed characteristics such as land use, drainage area, slope, sinuosity, and the pattern of the stream channel network all

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impact the discharge, velocity, and sediment load of streams. In watersheds with restrictive soil layers, subsurface flow and seepage have been recognized as significant processes that can further destabilize streambanks and increase bank erosion (Fox and Wilson 2010). In such watersheds, conservation practices that improve water infiltration will also increase subsurface flow and seepage, possibly leading to additional bank destabilization. Land disturbances, such as those that took place in many parts of the United States where natural vegetation was cleared for agricultural uses, cause greater discharge and velocities in streams (Schilling et al. 2010), resulting in destabilized streambanks becoming the main source of in-stream sediment (Simon and Kilmetz 2008; Belmont et al. 2011). Hydrologic alterations, such as stream channelization and dam construction, will also destabilize streambanks upstream of the alteration and increase bank erosion rates (Zaimes et al. 2004; Simon and Rinaldi 2006). These major land and hydrologic disturbances lead to destabilization of stream channels and initiate a process of channel evolution to reach a new quasi-equilibrium (Simon and Rinaldi 2006).

The Mark Twain Lake/Salt River watershed is in the Central Claypan Region, (Major Land Resource Area 113) within the Central Irregular Plains ecoregion (figure 1). The Central Claypan Region is characterized by loess overlying glacial drift; both parent materials have high clay content, resulting in a restrictive subsurface soil layer with exceptionally low hydraulic conductivity (Lerch et al. 2008). This naturally formed claypan periodically results in a perched water table following precipitation, resulting in runoff-prone soils and high soil erosion rates despite the gently rolling typography. Interflow is also a significant source of stream recharge in these watersheds, accounting for ~20% of stream recharge (Peters 2015). In Mark Twain Lake, the major source of public water in the region, sedimentation and turbidity are the most severe water quality problems (Lerch et al. 2008). Previous research in two Salt River subwatersheds indicated that streambanks were the major source of in-stream sediment and a significant source of total N transported from these watersheds (Willett et al. 2012), but the specific effects of several watershed characteristics (e.g., upstream land use, drainage area, and stream sinuosity) and bank vegetation were not previously investigated. The objectives of this study were to (1) investigate the effects and interactions of land use, stream order, and season on erosion rates from streams in two claypan watersheds over a four-year period; (2) estimate the total mass of sediment and total P contributed to streams by streambank erosion; and (3) investigate the impacts of bank vegetation cover and various watershed characteristics on bank recession.

Materials and Methods

Study Area. Study sites were established in Crooked Creek and Otter Creek watersheds located within the Salt River basin of northeastern Missouri (figure 1). The Salt River basin encompasses most of the central claypan areas in Missouri and was selected as a benchmark research watershed for the CEAP (Lerch et al. 2008). These watersheds were chosen because they have extensive agricultural land uses and are dominated by highly erodible claypan soils. Claypan soils have an argillic horizon ranging from about 0.1 m to 0.8 m (0.3 to 2.6 ft) below the surface that has a high clay content (>450 g kg^{-1} [>45%]) of smectitic mineralogy (Lerch et al. 2008). During precipitation events, water holding capacity is largely restricted to the soil layers above the claypan, which quickly saturate and generate surface runoff and lateral interflow above the claypan.

Experimental Design. The experimental design was a three-way factorial with main factors of land use, stream order, and season, and was described in detail by Willett et al. (2012). Crop, pasture, riparian forest, and forest sites were stratified across 1st, 2nd, and 3rd+ stream order stream reaches. The 3rd+ stream order category includes 3rd and 4th order streams because the total length of streams designated as 3rd order within these watersheds was too limited for this experimental design. Each land use and stream order pairing was replicated three times, with the exception of 3rd+ order crop treatment because only one such treatment reach could be found within the study area. During the four-year sampling period, three sites (1st order riparian forest, 1st order crop, and 2nd order forest) were abandoned due to landowner restrictions that caused some seasonal measurements to be missed. Replacement sites with the corresponding land uses and stream orders were established to prevent further loss of data for the affected treatments. Therefore, the data analysis included a total of 37 sites: 34 established in 2007 and 2008 plus three replacement sites established in 2010.

Field Measurements and Soil Collection. Pin measurements occurred in mid-March, early August, and late November as these dates are at the end of three designated seasons: Season 1 (winter)—December through March; Season 2 (spring/summer)—April through July; and Season 3 (summer/fall)—August through November. To correspond with the seasons defined above, a "year" was defined to be from December 1 to November 30 (e.g., 2008 data represent December 1, 2007, to November 30, 2008). Pin lengths recorded during seasonal measurements were subtracted from the lengths left exposed during the previous measurement date. Data from a total of 12 measurement dates were obtained from March of 2008 to November 2011, with erosion data from March of 2008 to November of 2009 reported in Willett et al. (2012).

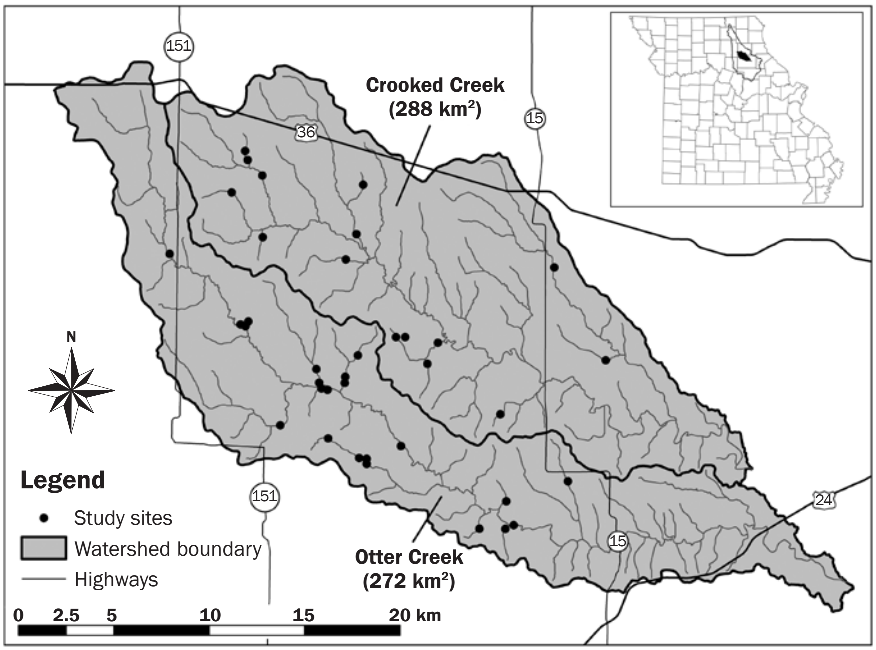
For determination of total P concentration, soil samples were collected from 50%, or a minimum of three pin plots, at each site, and samples were collected from each soil horizon. Soil P concentrations were analyzed using an alkaline oxidation method developed by Dick and Tabatabai (1977). Before analysis, samples were air dried and sieved through a 2 mm (0.08 in) screen. Alkaline oxidation of the samples was accomplished by digestion with a solution of sodium hypobromite (NaOBr), and the extracted P was quantified colorimetrically by a modified molybdate method (Peacher 2011). Average depth-weighted P concentrations were then calculated for each site.

Bank Recession and Linear Erosion Rates. Two calculations were used to represent the rate of loss or deposition of streambank materials: (1) bank recession rate (cm y^{-1}) and (2) linear erosion rate (kg m^{-1} y^{-1}). To calculate bank recession rate, the average pin length change was calculated for each pin plot; all pin plots for a given site were averaged and multiplied by the proportion of eroding bank length. Seasonal recession and linear rates were summed to acquire annual rates, and the rates were averaged for the four-year study period (2008 to 2011) to compute watershed-scale loads. Additional details of the bank erosion rate computations were reported by Willett et al. (2012).

Watershed-Scale Estimates of Sediment and Phosphorus Loads. Total stream length

Figure 1

Location of streambank erosion study sites in Crooked Creek and Otter Creek watersheds within the Salt River basin of northeastern Missouri, United States.



for each stream order in the study area (i.e., combined lengths of Crooked Creek and Otter Creek watersheds) was based on the National Hydrography Dataset (Dewald and Roth 1998). The total mass of in-stream sediment derived from streambanks in the study area was computed by multiplying the arithmetic mean linear erosion rates for each stream order by the total length of streams in that stream order. The annual P load from streambanks was estimated by multiplying the depth-weighted average P concentrations in bank soils by the estimated mass of streambank sediment for each stream order. Total P exported from the watershed was estimated from stream discharge data and P concentrations of water samples collected from Crooked and Otter creeks in a manner analogous to that described by Willett et al. (2012) for total N export. Total P concentrations in stream water were determined by acidic persulfate digestion of unfiltered samples in an autoclave (250°C [482°F] for 30 minutes) followed by detection of phosphate (PO_3^{4-}) by the molybdate method (Lerch et al. 2015). The data represent the combined total of all P forms (mineral P, dissolved and sediment-bound inorganic P, and organic P) in the stream samples, and concentrations were reported on an elemental basis. Automated samplers were used to collect flow-weighted samples during runoff events, and grab samples were collected under baseflow conditions. The number of samples collected annually ranged from 24 to 43 from Crooked Creek and 29 to 43 from Otter Creek. Phosphorus loss from streambanks was subtracted from total P export to estimate the proportion of P coming from other sources, such as overland erosion from crop and pasture land and fertilizer and livestock inputs. Given the fine textured bank soils across all the sites, streambank contributions to sediment bed load were considered negligible.

Riparian Vegetation Surveys. Vegetation surveys were conducted at 18 sites during the summer of 2010 (Peacher 2011). Eight bank top vegetation plots were established on a transect parallel to the stream channel at 50 m (164 ft) intervals and 1.5 m (4.9 ft) from the edge of the stream channel. At each vegetation plot, a ground/canopy cover survey, a shrub survey, and a tree survey were conducted. Bank face vegetation was also surveyed at the even numbered vegetation plots (i.e., 2nd, 4th, 6th, and 8th plots). Ground and canopy cover surveys were conducted with a 1 m² (10.8 ft²) frame, constructed from PVC pipe, and placed straddling the transect at each vegetation plot. Ground cover and canopy cover were recorded based on visual estimation of the proportion of each cover type within the PVC frame and assigned one of the following cover classes: <5%, 5% to 10%, 11% to 15%, 16% to 20%, 21% to 30%, 31% to 50%, 51% to 75%, or 76% to 100%. Applicable ground cover categories for bank top vegetation plots were tree, grass, and bare ground, and applicable canopy cover categories were tree and shrub. Percentage classes for each cover category were averaged by taking the middle number of the percentage class. For instance, the range 76% to 100% was translated into a value of 88%.

Shrub plots were established along the transect and positioned to include the ground/canopy vegetation plots. Shrub plots were 25 m² (269 ft²) (5 × 5 m [16 × 16 ft]), and only plants with a main trunk <5 cm (<2 in) in diameter at breast height (DBH) were included. Species and approximate number of stems for each shrub were recorded to calculate mean stem density (stems m^{-2}). Tree plots were established such that the shrub and ground/canopy vegetation plots were nested within them. Tree plots of 50 m² (538 ft²) (5 × 10 m [16 × 33 ft]) were parallel to the streambank and shared a common bank border with the smaller, nested plots. Only trees with a DBH >5 cm were included. Species and DBH were recorded for each tree, and the basal area (m² ha^{-1}) was calculated.

Bank face plots were 1 m (3.3 ft) wide and extended from the top of the bank to the bank toe. Estimated ground and canopy cover were recorded using the same classes as described above for the bank top plots. Tree or shrub ground cover was determined based on the proportion of bank face covered by tree trunks or shrub growth occurring within the plot and did not include canopy cover or over-hanging vegetation. Applicable ground cover classes for bank face plots were tree, shrub, grass, roots, and bare ground, and the only applicable canopy cover class for the bank faces was grass cover.

Watershed Characteristics. Watershed boundaries were determined for each of the 37 erosion pin sites, and five variables were investigated: drainage area (km²), average slope (%), sinuosity (m m^{-1}), and deforested surface cover (%). Watershed drainage areas were delineated using ArcSWAT (an extension of the Soil and Water Assessment Tool) and a 10 m (33 ft) digital elevation model (Missouri Spatial Data Information Service). Slope was computed for the longest flow path in each watershed as the difference in elevation between the lowest and highest points divided by the length. The total sinuosity was calculated by the length weighted average of the individual stream segments within each watershed using Hawth's Tools in ArcMap 9.2 (Esri Inc., Redlands, California). Land cover in each watershed was determined using 2006 data that included 14 separate land cover classes (National Land Cover Database). The deforested surface cover was calculated by summing the areas of development (low, medium, and high intensity), barren land, pasture/hay, and cultivated crops, and dividing that total by the watershed area. ArcMap 10 was used to determine the fraction of each land cover class in each watershed (Esri Inc., Redlands, California).

Statistical Analyses. Analysis of variance (ANOVA) models were used to analyze the main factors (land use, stream order, and season), and their interactions, on linear erosion and bank recession rates (Willett et al. 2012). The MIXED PROC procedure of SAS version 9.2 (SAS Institute Inc., Cary, North Carolina) was used to fit the respective three-way ANOVA models. The default covariance structure for SAS—variance components—was used, and since repeated measurements were made for each pin, the RANDOM statement was used with the site treatment as the random variable. This statement has the same effect as the REPEATED statement used to account for the decreased variability associated with repeated measurements from the same experimental unit. Because the data were unbalanced, least squares (LS) means of linear erosion and bank recession rates were used for statistical comparisons of the main factors. However, arithmetic means are reported for the annual data and used for computing watershed-scale estimates of sediment and P loads.

Bank recession and linear erosion rate data were not normally distributed, and log transformations were required. Bank recession rates were transformed using Y = log(recession rate + 4), and linear erosion rates using Y = log(linear erosion rate + 90). In both cases, the transformed data were deemed acceptable as inspection of residuals showed symmetrical distributions with only one outlier. To obtain LS mean annualized linear erosion and bank recession rates by land use and stream order, the seasonal data were summed to obtain annual rates for each site and year, and the log transformed data, as described above, were analyzed by two-way ANOVA with main factors of land use and stream order and sites as the random variable. Differences among years were determined for linear erosion and bank recession rates by analyzing the log transformed data as a one-way ANOVA with year as treatment and sites as the random variable. This analysis also provided the annual LS mean erosion and bank recession rates. Relationships between streambank depth-weighted soil P concentrations with land use and stream order were investigated using a two-way ANOVA, with land use and stream order as the main factors. These data were normally distributed and required no transformation for analysis. Again, LS means were used in the ANOVA because of the unbalanced nature of the data set.

A correlation matrix was created to test for covariance among the 18 vegetation and watershed parameters described above, with highly correlated variables (r > 0.80) presumed to violate the assumption of independence. The tree cover and bare ground variables for bank face vegetation were highly correlated (r = 0.81), and bank face tree cover was eliminated from the regression analyses. A best subset regression was performed using SAS to construct the best multiple linear regression models for predicting bank recession rates. The average bank recession rate for each site was used as the dependent variable, and the 17 vegetation and watershed characteristics described above were used as the independent variables. Models with the 10 highest coefficients of determination (R^2) values for all one, two, three, and four variable models were then saved. From these models, the top four models were chosen based on R^2 values. Assumptions for normality and equal variance were visually checked for each model using graphical methods. Additional details of the regression analyses were reported in Peacher (2011).

Results and Discussion

Site Characteristics. A total of 3,419 pins were installed in 250 pin plots across 37 sites, and over 37,000 individual pin measurements were recorded over the study. On average, each site had 92 pins, 7 pin plots, a bank height of 1.48 m (4.86 ft), and a bulk density of 1.41 g cm^{-3} (0.82 oz in^{-3}). The mean percentage eroding bank length per site was 53%, with a range of 15% to 100%. Zaimes et al. (2008a) and Schilling et al. (2011) reported that 9% to 54% of streambank lengths were designated as eroding for sites located in Iowa. Other studies conducted in the Midwest reported a range from 1% to 70% of bank length that was designated as eroding (Lyons et al. 2000; Raymond and Vondracek 2011). Simonson et al. (1994) suggested that streams in "excellent" condition have no more than 10% eroding bank length, streams in "good" condition between 10% and 25%, "fair" condition between 25% and 50%, and "poor" condition >50%. Thus, streambanks along the surveyed reaches within these two Missouri watersheds were generally in poor condition due to substantial erosion and lack of vegetation, demonstrating the general significance of streambank erosion for the study area.

Annual Bank Erosion and Recession Rates. Bank erosion was highly variable across sites and years, with an arithmetic mean linear erosion rate of 99 kg m^{-1} y^{-1} (66 lb ft^{-1} yr^{-1}) and a range from –31 to 490 kg m^{-1} y^{-1} (–20 to 330 lb ft^{-1} yr^{-1}). LS mean annual linear erosion rates ranged from a low of 30 kg m^{-1} y^{-1} (20 lb ft^{-1} yr^{-1}) in 2011 to a high of 110 kg m^{-1} y^{-1} (73 lb ft^{-1} yr^{-1}) in 2008 (figure 2a). The arithmetic mean bank recession rate across all sites and years was 7.1 cm y^{-1} (2.8 in yr^{-1}) and ranged from –2.5 to 33 cm y^{-1} (–1 to 13 in yr^{-1}). Annual recession data followed a similar pattern as the linear erosion rates, except that the highest LS mean recession rate was in 2009 (figure 2b). These bank recession rates were within the range reported for sites in Iowa (Zaimes et al. 2008a; Schilling et al. 2011).

Discharge data for Crooked Creek showed that 2008 to 2010 were three of the six wettest years, with the highest median daily discharge since 1980 (figure 3). Average daily discharge from 1980 to 2011 was 1.9 m^3 s^{-1} (67 ft^3 sec^{-1}). During this study, average daily discharge was >3 m^3 s^{-1} (>106 ft^3 sec^{-1}) from 2008 to 2010, but fell to 1.1 m^3 s^{-1} (39 ft^3 sec^{-1}) in 2011. Correlation analysis of annual linear erosion rates (arithmetic means) versus stream discharge (table 1) showed that Season 1 and total annual discharge were highly correlated to bank erosion ($r \geq 0.94$, $p < 0.05$) for each stream individually or for the total study area. Although Season 2 stream discharge accounted for the highest proportion of annual flow in every year except 2011, it was not significantly correlated to bank erosion rates. Overall, total annual discharge of the study area (i.e., combined Crooked Creek and Otter Creek discharges) was most highly correlated to bank erosion rates (table 1). These

Figure 2
Average (a) linear erosion and (b) bank recession rates from 2008 to 2011. Means followed by the same letter were not significantly different at $p = 0.05$.

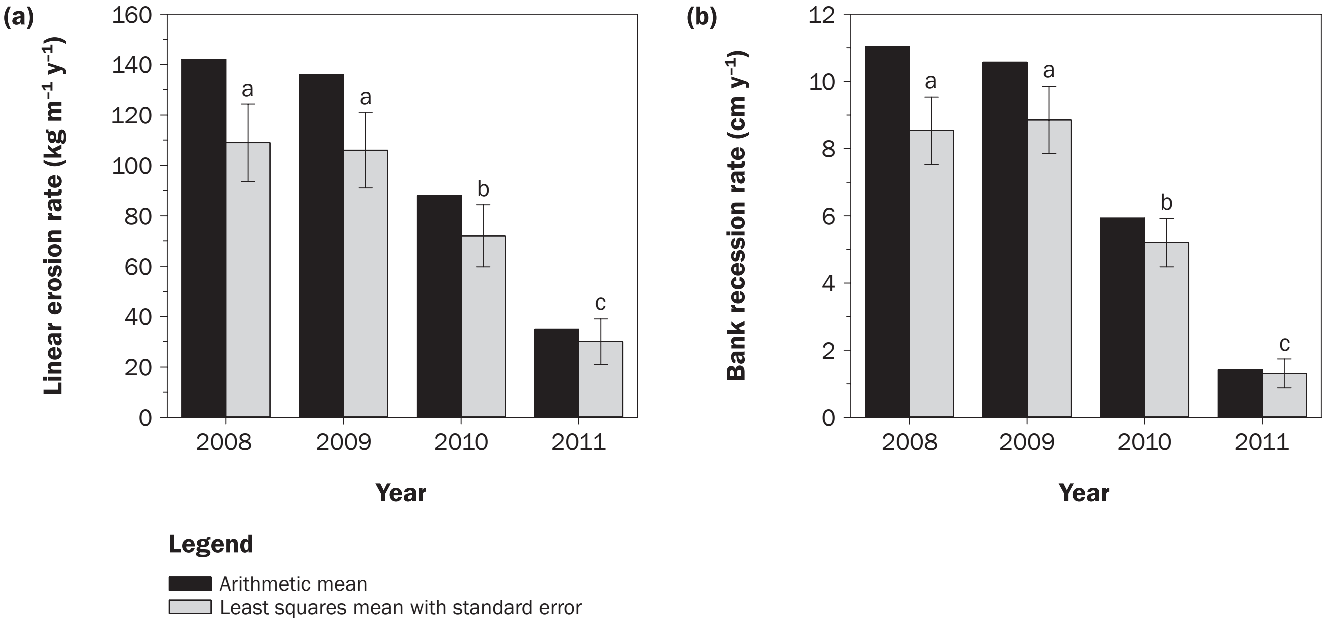
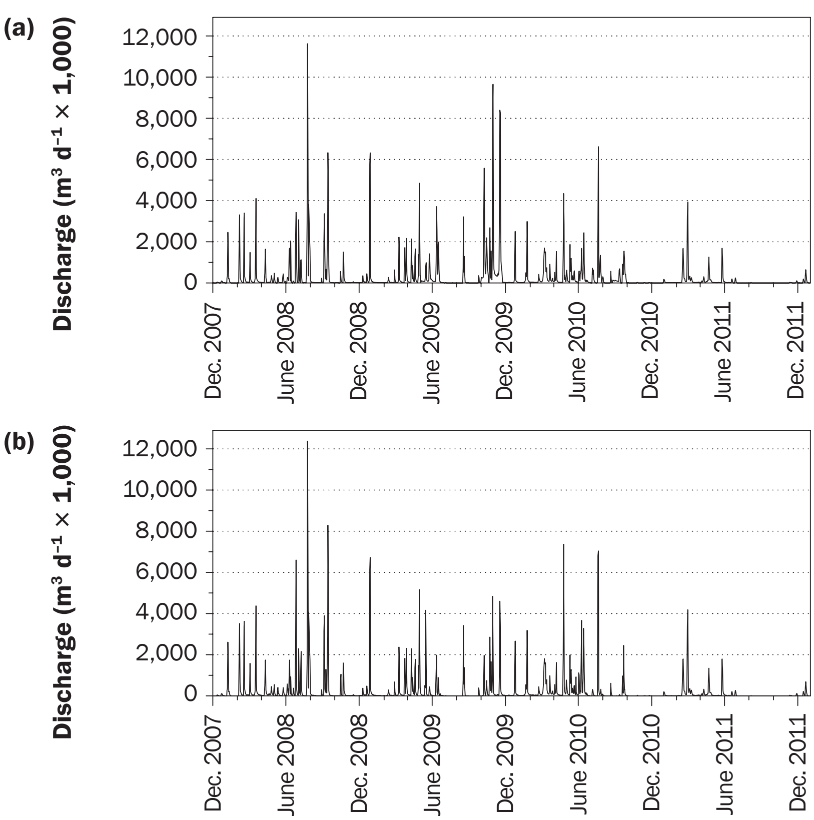


Figure 3
Average daily discharge for (a) Crooked Creek and (b) Otter Creek from December of 2007 through December of 2011.



relationships showed that the higher erosion and recession rates observed in 2008 and 2009 were associated with high stream discharge, especially during Season 1. Compared to 2008 and 2009, the lower erosion rates observed for 2010 were related to lower Season 1 and Season 3 discharges. The low erosion rates observed for 2011 were related to the much lower stream discharge throughout the entire year compared to the previous three years. Over a seven-year period, Palmer et al. (2014) also observed a strong relationship between bank recession rates and stream discharge in an Iowa watershed.

Season Effects on Streambank Erosion. Statistical analysis of the three-way factorial design for erosion rates and bank recession showed that season was the only significant main factor ($p < 0.05$) (table 2). Bank recession rates did not have any significant interactions. Erosion rates had three significant interactions, all involving stream order, which indicated an unambiguous interpretation of the season effect was not possible. However, the strong seasonal effect on bank recession and linear erosion rates was apparent, with Season 1 (December to March) consistently having the highest rates (figure 4). Linear erosion rate LS means (figure 4a) were 58 kg m^{-1} (39 lb ft^{-1}) for Season 1, 20 kg m^{-1} (13 lb ft^{-1}) for Season 2, and 4.7 kg m^{-1} (3.1 lb ft^{-1}) for Season 3. The LS mean bank

Table 1

Correlation of annual linear erosion rates to seasonal and total stream discharge for Crooked and Otter creeks.

Stream	Season 1	Season 2	Season 3	Total
Crooked Creek	**0.95***	0.66	**0.95**	**0.94**
Otter Creek	**0.95**	0.82	0.74	**0.96**
Crooked + Otter creeks	**0.95**	0.74	0.83	**0.99**

*Bold value indicates significance of the correlation coefficient, *r*, at $p < 0.05$.

recession rates (figure 4b) were 4.1 cm (1.6 in) for Season 1, 1.3 cm (0.51 in) for Season 2, and 0.34 cm (0.13 in) for Season 3. Note that seasonal bank recession and linear erosion rates were reported for four-month periods as opposed to the annual rates presented in the previous and subsequent sections. For all four years, statistical differences among seasonal recession and linear erosion rates were in the following order: Season 1 > Season 2 > Season 3. Season by stream order interactions supported this same trend, but Season 1 linear erosion rates of 3rd+ order streams were significantly greater than 1st order, indicating that the taller 3rd+ order banks were more susceptible to erosion during the winter months. Similar to that reported by Willett et al. (2012), the significant land use by stream order interaction for linear erosion rate was mainly due to site-specific factors as discussed below.

These results were consistent with Willett et al. (2012) who also reported much greater bank erosion rates in Season 1 compared to other seasons for this study area. Other studies have also shown that most bank erosion occurs during the winter and spring periods (Wolman 1959; Simon and Rinaldi 2000; Wynn et al. 2008; Tufekcioglu et al. 2012; Zaimes et al. 2008a). Wolman (1959) concluded that high winter erosion rates were largely caused by high flow events occurring during times when bank soils were already saturated. Consistent with the results presented here, Wolman (1959) noted that high flow events occurring in summer did not produce high erosion rates. Zaimes et al. (2006) noted that the highest magnitudes of bank erosion occurred in spring and early summer. Lawler et al. (1999) described the aforementioned factors along with decreased vegetative cover as factors destabilizing streambanks during the winter. Additional factors such as freeze-thaw cycles (Wynn et al. 2008) and bank seepage (Fox and Wilson 2010) have been found to decrease soil cohesion and bank stability. Air temperature often fluctuates above and below freezing in this region during winter and early spring (Missouri Mesonet 2016), and field observation of frozen and thawing banks in Season 1 indicated that freeze-thaw contributed to bank destabilization. Bank seepage was also frequently observed at the field sites during Season 1 measurements when seepage and interflow are expected to be most significant because of low evapotranspiration and high precipitation (Minshall and Jamison 1965; Peters 2015).

Land-Use Effects on Streambank Erosion. Neither linear erosion nor bank recession rates showed significant differences between land uses, in part due to the highly variable rates across the sites; coefficients of variation (CVs) for the land-use arithmetic means ranged from 81% to 129%. The LS mean linear erosion rates by adjacent land use were as follows: forest, 68 kg m^{-1} y^{-1} (46 lb ft^{-1} yr^{-1}); riparian forest, 72 kg m^{-1} y^{-1} (48 lb ft^{-1} yr^{-1}); crop, 81 kg m^{-1} y^{-1} (54 lb ft^{-1} yr^{-1}); and pasture, 93 kg m^{-1} y^{-1} (62 lb ft^{-1} yr^{-1}) (figure 5a). The LS mean bank recession rates for the different land uses (figure 5b) were as follows: pasture, 5 cm y^{-1} (2 in yr^{-1}); riparian forest, 5.4 cm y^{-1} (2.1 in yr^{-1}); forest, 5.7 cm y^{-1} (2.2 in yr^{-1}); and crop, 5.8 cm y^{-1} (2.2 in yr^{-1}). Although grazed pasture sites had the highest linear erosion rates, bank recession rates were very similar between land uses (figure 5). Factors such as stocking rates, stocking density, size of pasture, and length of rotations were not tracked or held constant or accounted for in this study, but were observed to vary over time. Previous results from this same study area also showed no significant land-use effect, but LS mean erosion rates were highest for the cropland treatments and lowest for the forest treatments (Willett et al. 2012).

There is no clear consensus in the literature on whether or not adjacent land use is a significant factor controlling streambank erosion. In general, forested riparian areas have less bank erosion compared to banks along crop and pasture lands, but the effect is often not significant because of the highly variable nature of bank erosion (Lyons et al. 2000; Zaimes et al. 2006; Schilling et al. 2010; Schwarte et al. 2011; Tufekcioglu et al. 2012; Willett et al. 2012). Many studies have shown the negative impacts that riparian grazing can have on streambank stability, and that fencing to exclude cattle from streams is an effective practice for reducing bank erosion (Owens et al. 1996; Belsky et al. 1999; Line et al. 2000; Magner et al. 2008; Tufekcioglu et al. 2012). In contrast, other studies indicate that vegetation directly on bank tops and faces may be more important to controlling erosion than adjacent land use within stream floodplains (Rosgen 2001; Simon and Collison 2002; McBride et al. 2007). Results from this study and Willett et al. (2012) suggest that factors other than adjacent land use were controlling bank erosion rates within these watersheds. Thus, the impact of factors, such as bank vegetation and watershed characteristics, were quantified and correlated to bank erosion (see below).

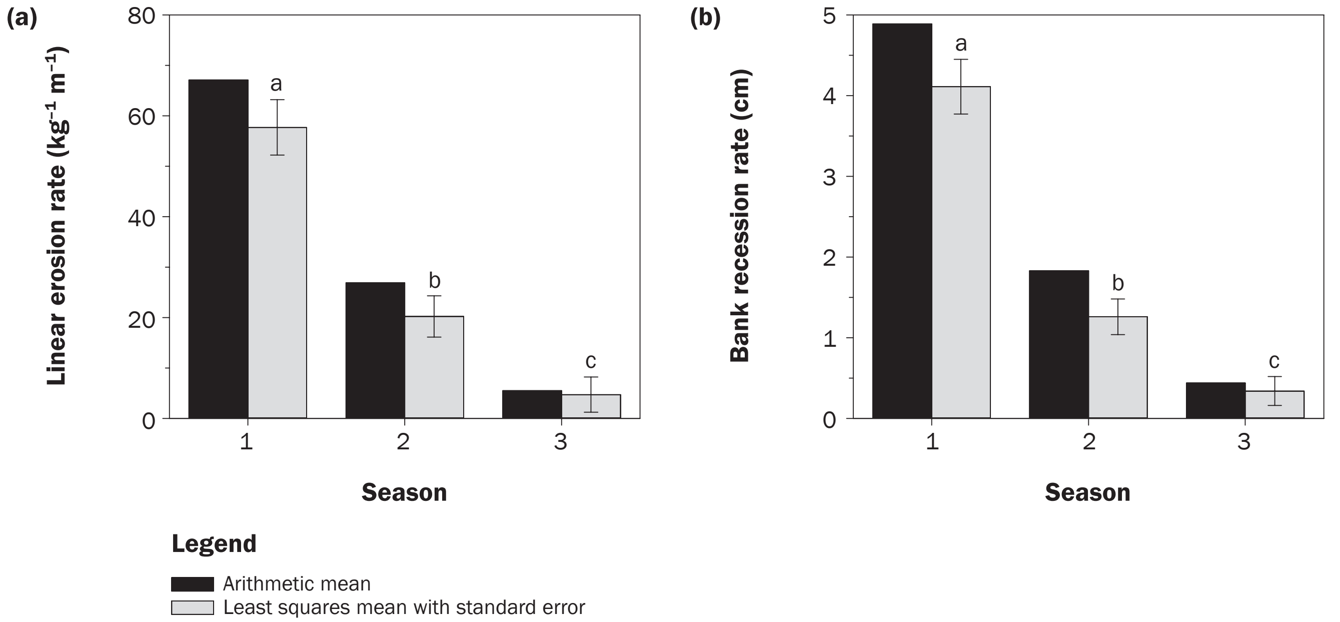
Stream Order Effects on Streambank Erosion. Third and fourth order streams

Table 2

Significance of main factors and interactions among main factors on linear erosion and bank recession rates.

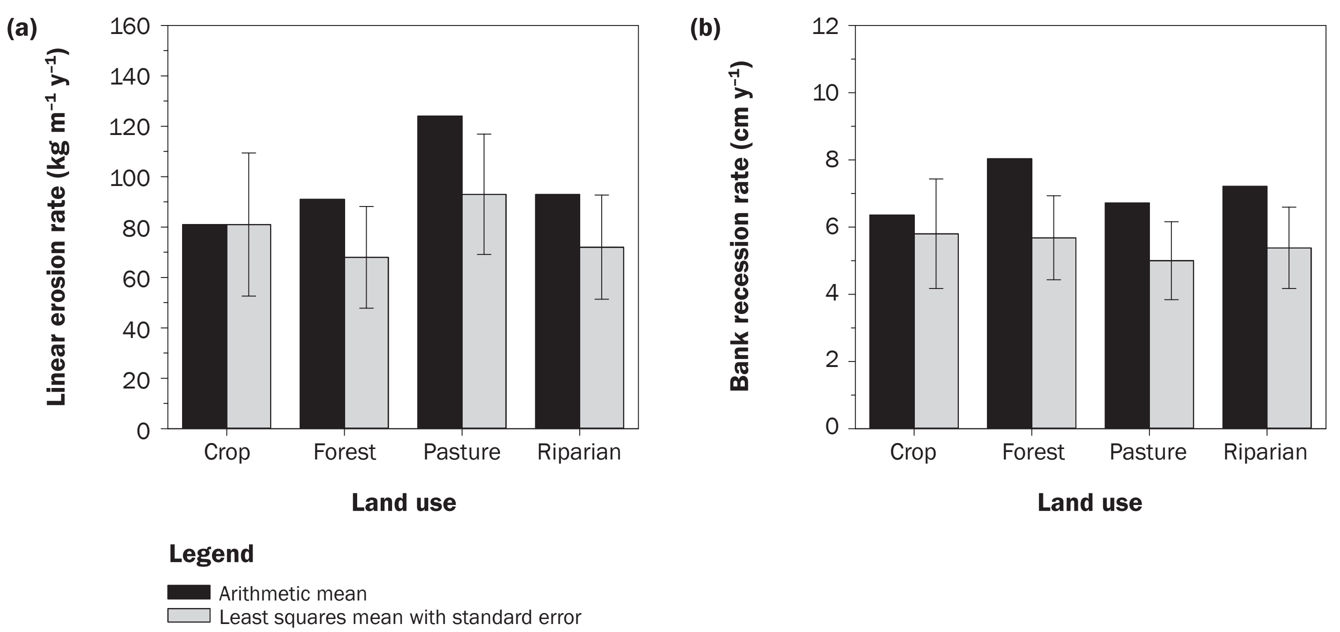
Effect	Degrees of freedom	Linear erosion rate *p* value	Bank recession rate *p* value
Land use	3	0.9780	0.6874
Stream order	2	0.7572	0.9139
Season	2	<0.0001	<0.0001
Land use × stream order	6	0.0204	0.0608
Land use × season	6	0.1519	0.3482
Stream order × season	4	0.0059	0.3740
Land use × stream order × season	12	0.0003	0.1386

Figure 4
Average (a) linear erosion and (b) bank recession rates by season. Least squares means with a different letter were significantly different at p = 0.05.



had higher linear erosion rates than lower order streams, but there were no significant differences between stream orders for sediment bank recession rates or linear erosion rates (table 2; figure 6). Similar to the land use means, average bank recession and linear erosion rates by stream order were highly variable, with CVs ranging from 47% to 103%. The LS mean linear erosion rates by stream order were 67 kg m^{-1} y^{-1} (45 lb ft^{-1} yr^{-1}) for 1st order, 70 kg m^{-1} y^{-1} (47 lb ft^{-1} yr^{-1}) for 2nd order, and 100 kg m^{-1} y^{-1} (67 lb ft^{-1} yr^{-1}) for 3rd+ order (figure 6a). For bank recession rates, LS means by stream order were 5.2 cm y^{-1} (2 in yr^{-1}) for 1st order, 5.2 cm y^{-1} (2 in yr^{-1}) for 2nd order, and 6 cm y^{-1} (2.4 in yr^{-1}) for 3rd+ order (figure 6b). Since the bank recession rates were similar

Figure 5
Average (a) linear erosion and (b) bank recession rates by land use.



among the stream orders (a calculation that only takes into account average pin length change and percentage eroding length), it can be concluded that the difference in linear erosion rates was caused by the greater average bank height of 3rd+ order streams. The results presented here were consistent with Willett et al. (2012) and provided further support that streambank erosion and recession were independent of stream order.

Bank Vegetation and Watershed Characteristics. Results of the regression analyses identified bank top trees, bank face trees, and bank face roots as significant factors affecting streambank erosion, as these variables were included in all four of the highest correlated models (table 3). Watershed variables were included in three of the four highest correlated models, indicating the importance of watershed characteristics for understanding bank erosion processes. The selected bank vegetation and upstream watershed characteristics explained 43% to 48% of the observed variation in bank erosion. Despite the high variation in erosion rates for this study, the regression models demonstrated that bank vegetation and watershed characteristics were significant factors affecting bank erosion. However, the low R^2 values of the models showed that these factors alone insufficiently captured the observed variation in bank erosion and would not be useful for predictive purposes. Peacher (2011) used the bank vegetation data described in this study to compute two modified versions of the Bank Erosion Hazard Index (BEHI) (Rosgen 2001), and correlated the BEHI scores to observed recession and linear erosion rates of the 18 sites. Results showed that BEHI site scores encompassed a narrow range (25 to 33 for one procedure and 28 to 38 for another) and were poorly correlated to observed erosion rates. BEHI scores categorized all sites as moderate to extreme erosion hazard despite average rates of 11 to 340 kg m^{-1} y^{-1} (230 lb ft^{-1} yr^{-1}) across sites. Thus, bank vegetation metrics could not capture the sources of variation underlying the observed erosion rates in this study and have limited utility for predicting streambank erosion in these watersheds.

These results, combined with the lack of significant land-use or stream order effects on bank erosion, indicated that one or more large-scale factors have affected the geomorphology of these two watersheds and exert control on streambank erosion rates. For example, the two sites with the highest observed erosion rates illustrated the likely effects of stream channelization on bank erosion. These sites, a 1st order pasture and a 2nd order forest, had average linear erosion rates of 260 and 340 kg m^{-1} y^{-1} (170 and 230 lb ft^{-1} yr^{-1}), respectively, from 2008 to 2011. The pasture site had adequate ground cover, mainly tall fescue (*Festuca arundinacea*), and did not appear to be overgrazed. Like all the pasture sites in the study, cattle had open access to the stream channel. The forest site had the largest continuous forested area of any in the study. However, the sites drain to a channelized section of lower Otter Creek, and the observed high bank erosion rates of these sites were at least partially attributable to the increased slope and discharge velocity caused by the downstream channelization.

Postsettlement clearing of forested watersheds for agricultural production is another key factor affecting bank erosion in these watersheds. Large-scale land clearing for agricultural production in the Salt River basin began in the mid-1800s and accelerated through the early 20th century (Lerch et al. 2008). By the 1930s, this major land clearance destabilized the hydrologic and geomorphic conditions of these landscapes, filling the valleys and stream channels with sediment from the eroded uplands. Although upland conservation practices have greatly improved over time, watersheds in this region have not re-equilibrated following this massive land disturbance (Fitzpatrick et al. 1999; Trimble 1999). This eroded sediment remains in the valleys and channels of Midwest streams, creating a disconnect between current land uses and observed bank erosion rates. An additional factor was the creation of Mark Twain Lake, located at the outlet to these watersheds, which raised the base level of the streams by 15 to 30 m (49 to 98 ft). The reservoir has impeded stream incision, and when combined with low gradient sediment-filled valleys, created the existing trapezoidal shaped (wide and low) channels that dominate in these watersheds today. Based on our field observations, these large-scale disturbances control the observed variability in bank erosion, and stream channels within the study area and the Salt River basin remain largely at Stage IV in channel evolution (Willett et al. 2012).

Watershed-Scale Transport of Sediment and Phosphorus. The total amount of in-stream sediment contributed from streambanks within the watersheds was an average of 135,000 Mg y^{-1} (149,000 tn yr^{-1}; table 4) or 240 Mg km^{-2} y^{-1} (690 tn mi^{-2} yr^{-1}) on an area basis. Schilling et al. (2011) reported annual average streambank erosion rates of 149 Mg km^{-2} y^{-1} (425 tn mi^{-2} yr^{-1}) for two watersheds in southern Iowa. First-order streams represent 52% of the bank length in the study area and contributed 49% of the annual sediment load (65,000 Mg y^{-1} [71,700 tn yr^{-1}]). Second-order streams lost an average of 31,100 Mg y^{-1} (34,300 tn yr^{-1}), and 3rd+ order streams lost an average of 38,800 Mg y^{-1} (42,800 tn yr^{-1}). Annual estimated sediment loss from streambanks was 193,000 Mg (213,000 tn) in 2008; 184,000 Mg (203,000 tn) in 2009; 116,000 Mg (128,000 tn) in 2010; and 46,000 Mg (50,700 tn) in 2011. Using the average annual linear erosion rate and standard deviation (SD) from this study (135,000 ± 68,000 Mg y^{-1} [149,000 ± 75,000 tn yr^{-1}]) and the estimated annual overland erosion rate (28,000 ± 19,000 Mg y^{-1} [30,900 ± 20,900 tn yr^{-1}]) from Willett et al. (2012), we estimated that streambanks accounted for 81% to 88% ($\overline{\chi}$ = 83%) of the total in-stream sediment within the Crooked Creek and Otter Creek watersheds. These estimates were within the range of that previously reported for these watersheds (Willett et al. 2012) and for Goodwater Creek watershed, a nearby claypan soil watershed in northeast Missouri (Wilson et al. 2014). Among five CEAP benchmark watersheds in Georgia, Iowa, Mississippi, and Oklahoma, Wilson et al. (2008) estimated stream channels contributed 46% to 82% of the suspended sediment, with collapsed bank material being the dominant sediment source in four of the five watersheds. The results of this study are in strong agreement with recent studies demonstrating the importance of bank erosion as the critical source of stream sediment in agricultural watersheds.

Adjacent land use and stream order did not significantly affect bank soil P concentrations. The LS mean bank soil P concentration for all sites was 372 ± 16 (standard error [SE]) mg kg^{-1} [ppm]). Average soil P concentrations (LS means) by stream order ranged from 346 mg kg^{-1} for 1st order streams to 404 mg kg^{-1} for 3rd+ order streams. Across land uses, average soil P concentrations ranged from a low of 319 mg kg^{-1} for forest sites to a high of 398 mg kg^{-1} for crop sites. These soil P concentrations were at the lower end of those reported in the literature for sev-

Figure 6

Average (a) linear erosion and (b) bank recession rates by stream order.

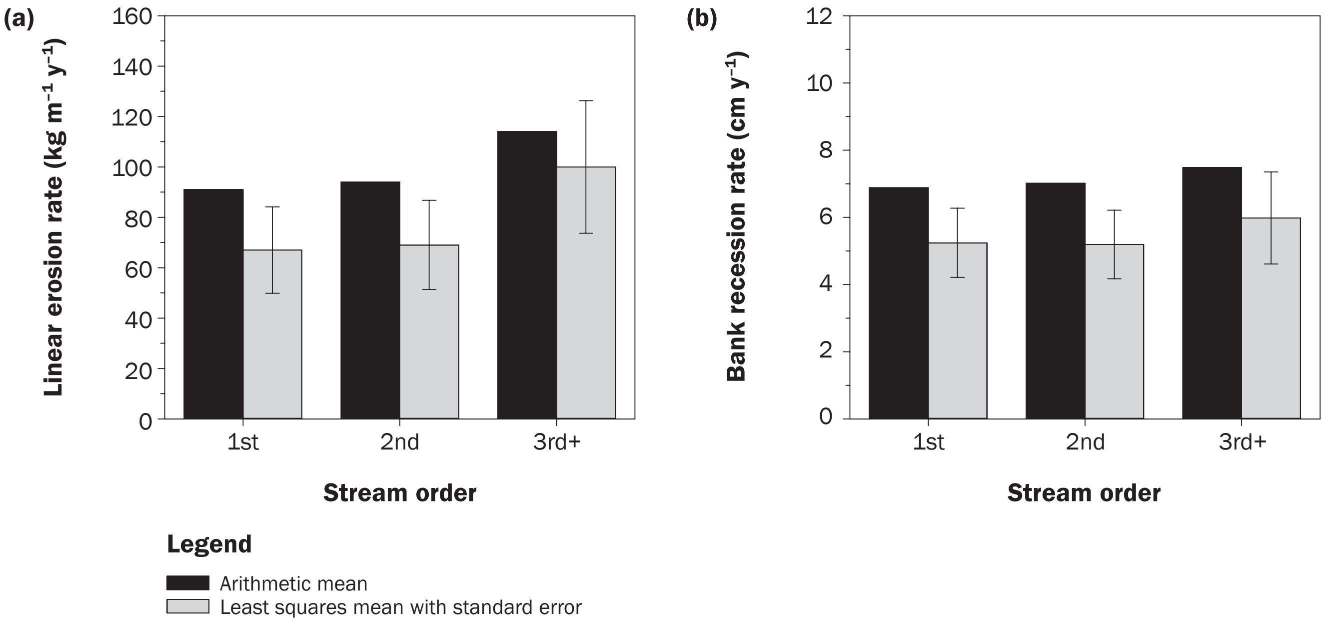


Table 3

Four-variable multiple linear regression models.

Model	Variable	Parameter estimate	*p*-value of *t*	R^2*	*p*-value of F
1	TG	−0.24	0.16	0.48	0.06
	BTG	−0.21	0.06		
	BRG	0.19	0.06		
	DSUR	−0.30	0.17		
2	TG	−0.22	0.23	0.45	0.08
	BTG	−0.22	0.05		
	BRG	0.23	0.02		
	S	2.68	0.27		
3	TG	−0.27	0.11	0.45	0.08
	TC	0.04	0.28		
	BTG	−0.22	0.05		
	BRG	0.18	0.09		
4	TG	−0.41	0.07	0.43	0.10
	BTG	−0.25	0.03		
	BRG	0.24	0.02		
	DA	0.00	0.38		

Notes: TG = tree ground cover, bank top. BTG = tree ground cover, bank face. BRG = root ground cover, bank face. DSUR = deforested surface cover. S = average watershed slope. TC = tree canopy, bank top. DA = watershed drainage area.

*Coefficient of determination.

eral studies from around the United States and Europe (Fox et al. 2016), but within the range of 246 to 555 mg P kg^{-1} for sites in southern Iowa (Zaimes 2008b; Tufekcioglu et al. 2012).

Combining the erosion and soil P concentration data, streambanks contributed an average of 58 Mg P y^{-1} (64 tn P yr^{-1}) (table 4) or 104 kg P km^{-2} y^{-1} (590 lb P mi^{-2} yr^{-1}). Of the total P from streambanks, 44% originated from 1st order streams, 24% from 2nd order streams, and 32% from 3rd+ order streams (table 4). Soil P loss rates from streambanks ranged from 38 g m^{-1} y^{-1} (0.41 oz ft^{-1} yr^{-1}) for 1st order streams to 49 g m^{-1} y^{-1} (0.53 oz ft^{-1} yr^{-1}) for 3rd+ order streams (table 4), and were within the range reported by Tufekcioglu et al. (2012). During the course of this study, total P transport was monitored at sites near the outlets to Crooked Creek and Otter Creek watersheds (Lerch et al. 2008). Combined total P transported from the two watersheds ranged from 16 Mg y^{-1} (18 tn yr^{-1}) in 2011 to 120 Mg y^{-1} (132 tn yr^{-1}) in 2008 and 2009, and streambanks contributed an estimated 20 to 80 Mg P y^{-1} (22 to 88 tn P yr^{-1}) (table 5). Thus, P loads from streambanks represented 55% to 100% ($\bar{\chi}$ = 67%) of the estimated P transported from the study area, and indicated that streambanks were the predominant source of P in Crooked and Otter creeks. Collectively, other sources, such as fertilizer and manure P and eroded soils from upland cropped or pasture fields, accounted for less P load than that derived from streambanks. The observed P loads in this study fell within the range reported in several other studies, but P contributions from streambanks vary widely between watersheds, ranging from 6% to 100% of the exported P (Fox et al. 2016). In addition, Willett et al. (2012) reported that streambanks accounted for 23% of the total

N and were also a major source of C transported from this study area. Robertson and Saad (2011) estimated total P transport in the Salt River basin to be >98 kg km^{-2} y^{-1} (>560 lb mi^{-2} yr^{-2}) using the SPARROW model, a regression-based mass balance model that does not account for the contribution from streambanks. However, the observed total P transport from the study area was within the predicted range of the SPARROW model from 2008 to 2010. This finding implies that contributions from upland agricultural sources (i.e., fertilizer and manure P) were overestimated by SPARROW.

Summary and Conclusions

More than 3,400 erosion pins were measured three times per year from 2008 to 2011 at a total of 37 sites across the study area to assess the effects of adjacent land use, stream order, and season on streambank erosion. LS mean annual linear erosion rates for the study area ranged from 30 kg m^{-1} y^{-1} (20 lb ft^{-1} yr^{-1}) in 2011 to 110 kg m^{-1} y^{-1} (73 lb ft^{-1} yr^{-1}) in 2008, rates that were within the range reported in the literature. Although interactions precluded unambiguous interpretation of the season effect, it was clear that Season 1 (December to March) was the period with the greatest bank erosion rates. Stream discharge during Season 1 was found to be highly correlated to annual linear erosion rates, demonstrating that winter runoff events were most erosive. As previously found for this study area, land use and stream order were not significant factors affecting bank erosion. Moreover, bank vegetation and upstream watershed characteristics explained <50% of the observed variation in bank erosion. These results indicated a disconnect between observed bank erosion and the current land use, bank vegetation, and watershed characteristics. Postsettlement land clearance, stream channelization, and impeded stream incision due to a downstream reservoir have resulted in wide, meandering stream channels that are susceptible to bank erosion and indicative of Stage IV channel evolution. These large-scale, historic factors in combination with variations in Season 1 stream discharge were the main factors controlling bank erosion in these watersheds. Within the study area, streambanks accounted for an average of 83% of the annual in-stream sediment and 67% of the annual total P transported. Combined with known contributions to total N and total C transport, it is clear that streambanks are a major source of sediment and nutrient contamination in these streams. These findings support the need for implementation of management practices to reduce streambank erosion to improve water quality in streams of the central claypan areas of Missouri.

Table 4
Streambank sediment and phosphorus (P) transport in Crooked Creek and Otter Creek watersheds.

Stream order	Bank length (km)	Linear erosion rate (kg m^{-1} y^{-1})	Annual soil loss (Mg y^{-1})	P loss rate (g m^{-1} y^{-1})	P loss mass (Mg y^{-1})
1st	704	92 ± 7*	65,000 ± 17,600*	38 ± 6*	27 ± 2
2nd	323	96 ± 7	31,100 ± 7,500	42 ± 8	14 ± 1
3rd+	338	115 ± 9	38,800 ± 9,200	49 ± 8	17 ± 1
Totals/means	1,365	99 ± 10†	135,000 ± 6,000	42 ± 4†	58 ± 2

*Arithmetic mean ± standard error.

†Length-weighted average ± standard error.

Table 5
Mass balance of total phosphorus (P) transport in Crooked Creek and Otter Creek watersheds.

Year	Total P transport (Mg y^{-1})	Streambank erosion P (Mg y^{-1})	Other P sources (Mg y^{-1})*	Total P from streambanks (%)
2008	120	80	40	67
2009	120	80	40	67
2010	93	51	42	55
2011	16	20	−4	100
Average	87	58	29	67

*Computed as difference between total P transport and streambank erosion P.

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Disclaimer

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