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Evaluation of Effectiveness of Streambank Stabilization Practices and Flood Impact on Cedar River, Nebraska

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EVALUATION OF EFFECTIVENESS OF STREAMBANK STABILIZATION PRACTICES AND FLOOD IMPACT ON CEDAR RIVER, NEBRASKA

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EVALUATION OF EFFECTIVENESS OF STREAMBANK STABILIZATION PRACTICES AND FLOOD IMPACT ON CEDAR RIVER, NEBRASKA

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Stream restoration has been a major environmental objective for preserving biodiversity, reducing loss of valuable cropland and improving water quality. The impact and efficiency of streambank stabilization practices has been simulated using erosion-prediction models; however, evaluation of the erosion-control practices to measure their efficiency is often neglected. This project monitored changes in fluvial geomorphology on the Cedar River in Nebraska and quantified the effectiveness of 18 sites with streambank stabilization practices. A flood event in 2010 due to dam failure acted as a major parameter in measuring the efficiency of the erosion control practices. The methodology included aerial imagery to measure streambank migration over time. Flow data, cross-sectional surveys, radius of curvature of meanders and vegetation density were documented to determine their influence on streambank erosion. Jetties were observed to be the most cost effective under moderate flow conditions, but three out of ten sites failed during the flood event. If the design of the jetties can be modified to provide additional stability without increased costs, the limitation of failures could be reduced. Erosion rates were significantly higher during the flood, especially immediately downstream of the failed dam. While NDVI was an effective tool in quantifying

vegetation density and eliminating visual bias, it did not significantly correlate to streambank erosion rates of this river. Additionally, radius of curvature of meander bends did not correlate with erosion rates before or during the flood event. Other factors like local site conditions, root depth and soil type may be more dominant in impacting the streambank in this stream system.

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CHAPTER 1 INTRODUCTION

1.1. Erosion and Environment

Streambank erosion is a natural process that is sometimes accelerated due to human-induced changes. As water depth increases, the applied shear stress on a streambank increases. When the applied shear stress exceeds critical shear stress of the soil material, erosion occurs. The streambank erosion rate is calculated using the following equation (Clark and Wynn, 2006; Hanson and Cook, 1997; Hanson, 1990a, 1990b; Partheniades, 1965):

$$\boldsymbol{\varepsilon = k_d\,(\tau_a - \tau_c)^a} \tag{1-1}$$

where Ɛ is the erosion rate (m/s), k_d is the erodibility coefficient (m^3/N.s), τ_a is the applied shear stress (Pa), τ_c is the critical shear stress (Pa) and a is an exponential coefficient typically assumed to be 1.

Applied shear stress is essentially a direct function of water depth and bed slope.

$$\boldsymbol{\tau_a = \rho.g.d.S} \tag{1-2}$$

where ρ is the density of water (kg/m^3), g is the gravitational acceleration (m/s^2), d is the depth of water (m) and S is the channel bed slope.

While many equations have been examined in the past, most modeling software still use the above equations to assess lateral streambank erosion. There are many factors that influence the erosion processes such as land use change, streamflow, soil composition, stream channelization and riparian vegetation. The influence of each parameter varies between stream systems (Peacher et al., 2018). Besides the physical phenomenon, it is important to consider local specific site conditions along with large scale factors like watershed characteristics and climatic conditions when evaluating or predicting erosion. Changes in these factors are sometimes accelerated by anthropogenic activities like land use change.

Although streambank erosion is a natural process, increased erosion may degrade and disrupt the ecological balance of the environment. Sediment is one of the most common pollutants, second only to pathogens in impairing 12.5% of the assessed U.S. streams (USEPA, 2018). As in-field conservation practices have increased, erosion from upland sources has declined, hence sometimes making streambank erosion a significant or dominant source of stream sediment in many streams (Peacher et al., 2018; Willett et al., 2012). In some watersheds, a major portion of sediment, as high as 85%, originates from streambank erosion. Streambank retreat can range from 1.5 to 1100 m/year (Clark and Wynn, 2006; Prosser et al., 2000; Simon et al., 2000; Wallbrink et al., 1998; Trimble, 1997). Previous studies have found that streambank erosion is one of the greatest causes of non-point source pollution (Ramirez-Avila et al., 2010; Rosgen, 2001). Accelerated rates of erosion causes loss of valuable cropland, increases turbidity of water thereby increasing water treatment costs (Buchanan et al., 2012). It also causes harm to aquatic life by clogging the gills of fish making it harder for the fish to breathe and thus sometimes leading to death. Sediment loss caused by erosion also reduces reservoir capacity and changes the cross-section of streams, thereby, increasing the risk of flood.

1.2. Streambank Behavior

Streambank migration is a natural process with erosion and deposition occurring simultaneously within a stream system. The stress level at which soil detachment begins is called critical shear stress (Clark and Wynn, 2006). Streambank erosion is categorized into three processes: subaerial processes, fluvial erosion and mass wasting (Purvis and Fox, 2016; Clark and Wynn, 2006; Couper, 2003; Couper and Maddock, 2001). Subaerial processes are comprised of preparatory weakening of the bank face by wetting-drying and freeze-thaw activities (Clark and Wynn, 2006; Couper and Maddock, 2001; Thorne, 1982). Fluvial erosion can be considered as a dominant process in most streams and is a result of applied shear stress, sinuosity, and stream flow (Purvis and Fox, 2016). It is safe to assume that increased subaerial processes will lead to increased fluvial erosion, which dominate the middle reaches of the stream (Couper and Maddock, 2001). The fluvial erosion then leads to undercutting and hence will lead to the formation of a critical bank. A bank is considered critical when bank toe and adjacent channel bed reach the maximum height and angle in terms of stability (Couper and Maddock, 2001; Simon et al., 2000). The critical banks will thus fail as a result of the fluvial undercutting and cause mass wasting, which is more dominant downstream of the river or stream (Couper and Maddock, 2001; Simon et al., 2000).

Research focusing on subaerial processes and fluvial erosion have documented the significant impact these processes can have in channel migration and increased risks of further erosion (Narasimhan et al., 2007; Couper, 2003; Couper and Maddock, 2001; Simon et al., 2000). However, more research involving documentation of the importance of mass wasting, which plays a very crucial role in the design of streambank stabilization procedures, is lacking (Purvis and Fox, 2016). Mass wasting produces a significant amount of sediment load in a very short period of time. Hence, the streambank is at a higher risk of being impaired in terms of sediment as a pollutant. Bank material after failure is delivered directly into the flow and either settles down as deposition or remains suspended (Simon et al., 2000). This sediment reduces reservoir capacity, increases turbidity and poses a threat to aquatic life. It is important to understand the dominant streambank-erosion processes occurring in a stream system in order to identify the appropriate technique to stabilize or restore the streambank and river system.

1.3. Streambank stabilization techniques

Multiple streambank stabilization practices have been designed to control the negative impacts of streambank erosion. There are many different ways to categorize streambank stabilization. Traditional methods of streambank stabilization in the United States include channelization and hard armoring (Li and Eddleman, 2002). Channelization involves reducing the formation of meanders and hence controls natural erosion processes (Li and Eddleman, 2002). However, the channelization process destroys the basic physical equilibrium over time and has adverse ecological impacts (Li and Eddleman, 2002). Hard armoring methods (also referred to as engineered or structural practices) like riprap, jetties, gabions, revetments and retaining walls, reinforce streambank shear strength (Li and Eddleman, 2002; Keown, 1983). These hard or engineered practices have been favored over time for provision of immediate protection of properties or infrastructure adjacent to streams, after the completion of construction (Li and Eddleman, 2002). However, the cost of most structural practices is higher and requires design consideration, and sometimes requires a compromise in ecological benefits of some natural stream components. Additionally, armoring banks and constraining active channels tends to result in bed scour as most of the stream power is disseminated to the river bed (Death et al., 2015).

Many researchers have found bioengineered techniques more sustainable and cost efficient in stabilizing a bank (Sudduth and Meyer, 2006; Li and Eddleman, 2002; Donat, 1995). The most common bioengineering technique is riparian vegetation or buffer strips. Riparian vegetation increases shear strength of soil due to root reinforcement (Donat, 1995). This technique is also considered one of the best practices in agricultural areas and often serves as a filter strip to trap sediment from agricultural runoff. Additionally, it also maintains an ecological balance by preserving wildlife habitat and adds aesthetic value to the stream. Different types of vegetation can be chosen depending on the desired surface roughness. Hydroseeding, erosion-control blankets, coconut fiber rolls, compost socks, and live fascines are other examples of bioengineering techniques (Sudduth and Meyer, 2006; Li and Eddleman, 2002; Donat, 1995).

1.4. Impact of flood events

Dams provide a range of benefits including flood control, water supply, hydropower, waste management, recreation, river navigation, and wildlife habitat (FEMA, 2018). Flood-control dams impound floodwater and release it downstream under controlled conditions, or, store or divert the water for other uses (FEMA, 2018). Even after careful consideration of climatic data and dam safety design, dams have previously failed due to one or more reasons such as structural failure, piping, overtopping, settlement and cracking, deliberate sabotage or inadequate maintenance, hence causing significant flooding (FEMA, 2018). The flood events can significantly disrupt the entire ecological balance of the soil, water and plant environment. A series of dam failures in the 1970s in West Virginia, Idaho and Georgia had a devastating impact on the economy, environment and social work, resulting in dam inspection and regulation (FEMA, 2018). A lot of research has been carried out using different models and tools to estimate flood risk in urban or rural areas (Wheater and Evans, 2009; Evans et al., 2004 a,b). However, few studies have focused on the impact of floods on streambank erosion and its correlation with bank stabilization. It is hypothesized that streambank stabilization practices are more apt to fail during flood events.

Altering the geomorphology of a stream system by anthropogenic activities might increase the flood risk. However, best management practices (BMP) such as riparian buffers will be able to control a lot of sediment loss during a flood event. It is important to understand the behavior of vegetation in flood events because it can turn out to be the most cost effective and long-term solution to controlling erosion during floods (Death et al., 2015; Urbanic, 2014; Gostner et al., 2013).

Riparian vegetation can also add to the benefits of streambank stability in terms of conserving the ecology of riverine habitat. It provides food to biological communities, improves the biodiversity and adds an aesthetic value to the riverine system. In fact, in some parts of Europe, the EU Water Framework Directive policy (European Commission, 2000) requires improving the riverine biological communities through more natural hydromorphology. Since riparian vegetation does not degrade the geomorphology of rivers, it is much preferred in stabilizing banks where deflection of flow is not required. Most of the time, structural practices will fix the location of the streambank and hence is more significant when used on critical banks. Floodplains can be protected from erosion with the help of bioengineering techniques like riparian vegetation. More research on successful stabilization techniques under extreme flood events can prove to be useful in making decisions for effective streambank stabilization design guides.

1.5. Importance of monitoring stabilized streams

Most of the time, streambank erosion predictions are made with the help of models like Bank Stability and Toe Erosion Model (BSTEM), and Soil and Water Assessment Tool (SWAT) (Mittelstet et al., 2017; Midgley et al., 2012). More than one-third of the rivers of the United States are listed as impaired or polluted (Bernhardt et al., 2005). Annually, the United States spends about a billion dollars on streambank stabilization (Heeren et al., 2012; Bernhardt et al., 2005; Lavendel, 2002). However, very little project monitoring has been conducted for restored reaches. According to a National River Restoration Science Synthesis (NRRS) database, 28% of the projects across the Southwest conducted some kind of monitoring for the projects, while nationwide project reports indicated only 10% monitoring activities (Follstad Shah et al., 2007; Bernhardt et al., 2005). Perhaps, the absence of research indicating important conclusions drawn from monitoring is the reason of negligence towards monitoring. It is necessary to include monitoring as a part of restoration projects in order to understand the efficiency and cost effectiveness of these projects.

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CHAPTER 2 QUANTIFYING EFFECTIVENESS OF STREAMBANK STABILIZATION PRACTICES ON CEDAR RIVER, NEBRASKA[1]

2.1. Abstract and Keywords

Abstract:

Excessive sediment is a major pollutant to surface waters worldwide. In some watersheds, streambanks are a significant source of this sediment, leading to the expenditure of billions of dollars in stabilization projects. Although costly streambank stabilization projects have been implemented worldwide, long-term monitoring to quantify their success is lacking. There is a critical need to document the long-term success of streambank restoration projects. The objectives of this research were to (1) quantify streambank retreat before and after the stabilization of 18 streambanks on the Cedar River in North Central Nebraska, USA; (2) assess the impact of a large flood event; and (3) determine the most cost-efficient stabilization practice. The stabilized streambanks included jetties (10), rock-toe protection (1), slope reduction/gravel bank (1), a retaining wall (1), rock vanes (2), and tree revetments (3). Streambank retreat and accumulation were quantified using aerial images from 1993 to 2016. Though streambank retreat has been significant throughout the study period, a breached dam in 2010 caused major flooding and streambank erosion on the Cedar River. This large-scale flood enabled us to quantify the effect of one extreme event and evaluate the effectiveness of the stabilized streambanks. With a 70% success rate, jetties were the

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most cost-efficient practice and yielded the most deposition. If minimal risk is unacceptable, a more costly yet immobile practice such as a gravel bank or retaining wall is recommended.

Keywords: cost-efficiency; monitoring; streambank retreat; streambank stabilization; water quality

2.2. Introduction

Sediment is a primary pollutant to surface water and a leading cause of water quality impairment [1]. Though erosion is a natural and even necessary process [2], the rate of erosion has been accelerated due to human activities such as farming and urbanization. In some watersheds, streambank erosion is the main source of sediment to rivers and streams [3]. The excess sediment affects the water chemistry, aquatic organisms, and the water clarity in our streams and reservoirs. Excess sediment is aesthetically displeasing, increases the cost of treating drinking water, decreases water clarity, and has an overall negative impact on the aquatic ecosystem [4–8]. The increased turbidity not only affects the water aesthetics but also reduces photosynthesis and organisms' ability to see. Siltation alters flow in streams and decreases the storage volume in our reservoirs, which in turn affects flooding, drinking water, and recreation.

Alluvial channels continue to adjust, change, and sometimes shift their location [9]. In some eroding stream reaches, the streambanks are stabilized using conventional or modified stabilization techniques. Such stabilization structures are typically implemented in an effort to lock the stream channel into a relatively fixed location and condition [9]. The channel bank infrastructure then alters the geomorphic processes and can lead to more erosion at the stabilized site as well as at locations upstream and downstream

[10,11]. When these channel migration processes are ignored, the erosion-control structures may become ineffective over time, causing failure during large flow events [2,9,10]. Rosgen [12] has restored and monitored over 48 km of rivers and has monitored the performance of various structures following major floods. The results contradicted the expected pattern, as many structures appeared to cause river instability.

The cost of stream restoration continues to increase, yet few resources are being allocated to evaluate and monitor the restoration practices [13]. Annually, the US spends $1 billion on stream restoration, the efforts of which are intended to improve the environmental health of a stream or river, generally by restoring a dynamically stable dimension, pattern, and profile [14,15]. One component of stream restoration is streambank stabilization, which is generally intended to maintain a stable channel dimension at the stabilization site. Although post-installation monitoring yields important information on the effectiveness of a stabilization practice, less than 10% of the projects include any assessment of outcomes [13]. Possible reasons include lack of funds (project ended) and poor project documentation. Even when monitoring occurs, inconsistencies in collection methods and limited reporting are common [16]. There is an urgent need to use evaluation data to update design and implementation methods to improve restoration approaches and increase the likelihood of success of the thousands of projects currently being planned and implemented [17].

Though several research papers have been written about monitoring stabilized streambanks, scant long-term monitoring of stabilized streambanks is relayed in the literature. In general, most previous research discusses the effectiveness of stabilization practices based on qualitative analysis or prediction-based models. Daly et al. [18] used

BSTEM to estimate the streambank retreat rate over a seven-year (2003–2010) study period at 10 sites on Barren Fork Creek, Oklahoma. Protected sites had less streambank retreat, ranging from 4.1 to 74.8 m per year, while unprotected sites had an estimated average streambank retreat of 49.2 m per year. Simon et al. [19] also used a BSTEM model to analyze reduction in streambank sediment loads and bank failures at the Big Sioux River, Lower Tombigbee River and Lake Tahoe Basin. The average annual streambank erosion in the Big Sioux River was reduced by 51% (503,000 m^3 to 243,000 m^3) after toe protection was added, while overall volume of eroded bank material was reduced by 87–100% [19]. Similarly, in the Lake Tahoe Basin, total streambank erosion was reduced by about 89%. In the lower Tombigbee River, the amount of lateral retreat and volume of failed material was reduced by about 500% (from 55,000 m^3 to 9500 m^3) [14].

Some project reports exist for stream assessment, but they consider general health of streams and measure sediment loads or fish habitat. Most techniques involve research on riparian vegetation or stream stabilization conducted using bioengineering techniques. No previous study has specifically focused on quantification of streambank erosion for monitoring of stabilization practices. This study evaluated 17 sites stabilized from 2000 to 2005 and one stabilized in 1950 on the Cedar River in Nebraska. These sites were stabilized to reduce surface-water degradation and sedimentation loading of the river system, improve the aquatic habitat through riparian buffers and increased vegetation, and reverse the loss of prime cropland and rangeland. No post-construction monitoring has been conducted for these sites to evaluate the effectiveness or the stability of the practices in mitigating the effects of erosion. For the 18 stabilized streambanks and 40 identified control streambanks, the objectives of this study were to (1) quantify streambank erosion from 1993–2016; (2) evaluate the impact of an extreme flood in 2010; and (3) determine the most cost-efficient technique.

2.3. Materials and Methods

2.3.1. Site Description

The Cedar River watershed is located in North Central Nebraska (Figure 2.1). The western half of the 3200 km^2 watershed is mainly grassland and sand dunes in the Sand Hills, whereas the eastern half is predominantly cropland. Streambank height on the Cedar River averages 2.1 m and ranges from 0.4 to 4.0 m [20]. Streambanks are typified by silty, vegetated low-lying banks or higher banks with a combination of sandy and silty soil (Figure 2.2).

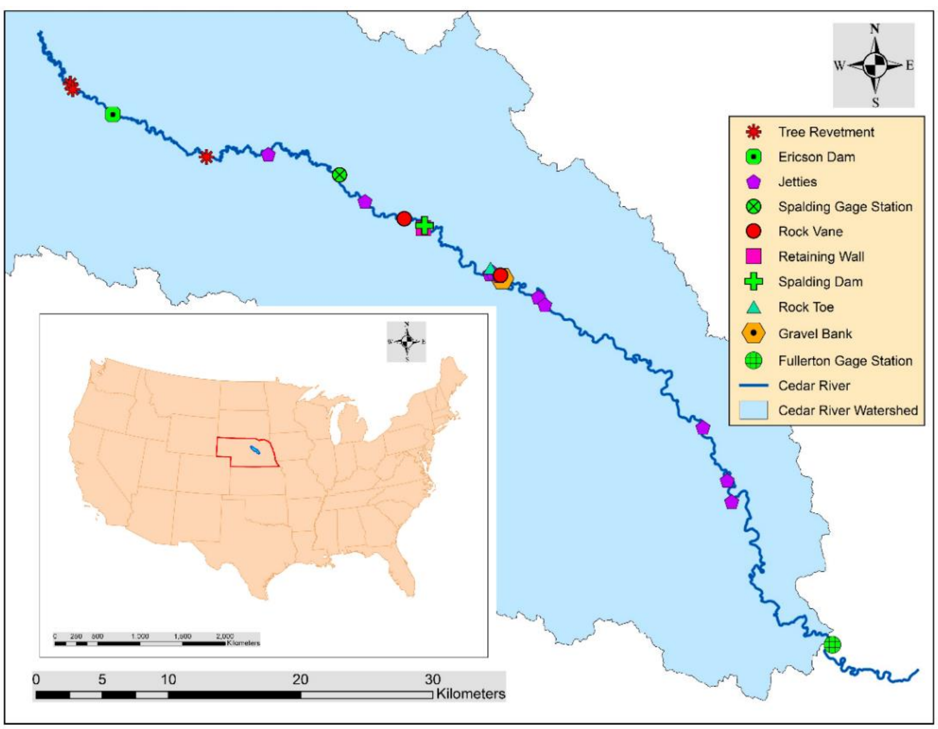


Figure 2.1 Locations of the dams, gage stations, and stabilized streambanks on the Cedar River in North Central Nebraska.

Figure 2.2 Typical streambank on Cedar River. Note the transition from sandy soil at the top to silty loam in the bottom.

Two reservoirs on the Cedar River, Ericson Lake, and Spalding Lake, are used primarily to generate electricity but also provide recreational benefits, as does the Cedar River. The 200 km long Cedar River begins at the confluence of the Big and Little Cedar Creeks in Eastern Garfield County [20]. Dams are located near Ericson Lake and Spalding Lake (Figure 2.1). There are two gage stations on the Cedar River, located near Spalding Lake and Fullerton, Nebraska. From 2006 to 2016, the average flow was 5.6 m^3 s^{-1} and 8.4 m^3 s^{-1} [21] at the Spalding and Fullerton gage stations, respectively. Heavy rains starting on 8 June 2010 led to a breach in the Ericson Dam. On the night of 13 June 2010, the spillway breached, causing major flooding downstream [22]. The flow peaked at 148.6 m^3 s^{-1} on 14 June 2010 at Spalding, an increase of 263% over the previous peak flow. On 15 June 2010, the flow peaked at 159.6 m^3 s^{-1} at Fullerton, an increase of 31% over the previous peak flow. The peak flow was not as dramatic at Fullerton since much of the flow was captured in Spalding Lake, which reached its capacity on 14 June. Excluding the flood event in 2010, average daily streamflow ranged from 2.3 to 40.9 $m^3 s^{-1}$ at Spalding and 0.9 to 122.4 $m^3 s^{-1}$ at Fullerton.

In the early 2000s, the Lower Loup Resource Conservation and Development, working with more than 100 landowners, received two grants to address streambank erosion and water quality on the Cedar River [23]. Using a Topcon HiPer V GPS [24], we visited and documented 17 of the 20 streambanks stabilized. During our tour of the sites, we identified an additional streambank stabilized in 1950. Six different stabilization techniques were implemented on the Cedar River: jetties, tree revetments, rock vanes, a rock toe, a retaining wall, and a gravel bank (Figure 2.1). These can be aggregated into two categories: bank protection and flow deflection. Tree revetments, the rock toe, the gravel bank, and the retaining wall were installed to protect the bank. The jetties and rock vanes were installed to deflect the flow. Two of the sites were located upstream of Ericson Dam, four of them between the Ericson and Spalding Dams and 12 downstream of Spalding Dam. These 18 sites were grouped based on the type of stabilization practice. Ten sites were stabilized with three to nine jetties, spaced approximately 30 m apart (Figure 2.3). Three sites were stabilized with tree revetments, which are cedar trees anchored along the streambanks to capture sediment in the leaves and roots and to stop further loss of sediment from the bank. Two sites were stabilized with rock vanes, which are linear structures that extend out in the stream channel from the streambank in an upstream direction. One streambank was stabilized with rock toe with reduced slope, one with a retaining wall and one with a gravel bank with reduced slope (Table 2.1). Cutting and filling of land is required to achieve a reduced slope to help stabilize the initially higher and non-uniformly sloped streambank. Though final costs are unknown, the proposed average costs per meter of streambank stabilized, obtained from original grants, were as follows: jetties: $26; cedar tree revetment: $72; rock vane: $205; rock toe: $179; retaining wall: $625; and gravel bank: $600.

(**a**)

(**b**)

Figure 2.3 Jetties installed in 1950 at Site 5 (a) and in 2000 at Site 4 (b). Note the sediment that has accumulated over time in front of the jetties installed in 1950.

Table 2.1 Detailed description of all sites.

Site	**Distance from Ericson Dam (km)**	**Technique (Number)**	**Length Stabilized (m)**	**Year Stabilized**	**Radius of Curvature (m)**	**Riparian Vegetation**	**No. of Control Sites**
Upstream of Ericson Dam							
1	5.4	Tree Revetment	57	2004	68	SG	2
2	4.9	Tree Revetment	105	2004	77	DG	2

Downstream of Ericson Dam							
3	12.5	Tree Revetment	88	2004	107	DG	2
4	22.5	Jetties (4)	106	2000	67	C	3
5	37.5	Jetties (3)	88	1950	87	SG	3
6	42.4	Rock Vanes (4)	98	2000	109	SG	2
Downstream of Spalding Dam							
7	45.2	Retaining Wall	31	2004	154	DG	3
8	55.4	Jetties (3)	127	2004	151	DG	2
9	55.9	Rock Toe	118	2000	112	DG	2
10	56.3	Jetties (4)	251	2000	196	C	3
11	56.9	Jetties (2)	63	2000	52	SG	1
12	57.1	Rock Vane	16	2000	92	SG	3
13	57.4	Gravel Bank	34	2000	143	SG	3
14	63.0	Jetties (3)	122	2004	114	SG	2
15	65.5	Jetties (4)	552	2004	247	SG	2
16	89.1	Jetties (3)	54	2000	156	C	1
17	95.8	Jetties (9)	207	2000	183	C	2
18	98.7	Jetties (8)	419	2000	178	C	2

Note: The abbreviations for riparian vegetation are SG = Sparse Grassland; DG = Dense Grassland; C = Cropland.

2.3.2. Lateral Streambank Erosion

The most common methods used to measure streambank erosion are erosion pins, aerial photography analysis, digital elevation models (DEMs), and repeat surveys [25]. Although aerial images are not as accurate as other methods, aerial photography is the only method that can be used to measure historical streambank retreat. Buchanan et al. [13], Pracheil [20], Mittelstet et al. [26], Heeren et al. [27], and Purvis and Fox [28] successfully analyzed aerial photography analysis to quantify the streambank erosion.

For this study, National Agricultural Imagery Program (NAIP) aerial photographs, acquired at 1 m spatial resolution, were used to measure the average lateral streambank retreat. High-resolution images were available for 1993, 1999, 2003, 2006, 2009, 2010, 2012, 2014, and 2016. Using ArcGIS 10.3 (Esri, Redlands, CA, USA), we measured the lateral streambank retreat for multiple time periods: pre-stabilization (1993–1999/2003), post-stabilization (1999/2003–2009), flood (2009–2010), and post-flood (2010–2016) (Figure 2.4). The lateral streambank erosion rate was calculated by dividing the measured eroded area by the length of the streambank. For example, for Site 11 (Figure 2.4), considering the calculation for the flood period (2009–2010), the length of the streambank in 2009 was 66.8 m. The streambank erosion was measured to be 152.9 m^2 over a period of one year; hence, the total annual lateral streambank erosion rate was 152.9 $m^2/(66.8\ m \times 1\ year) = 2.3\ m^2\ m^{-1}\ year^{-1}$. The streambank erosion rate was calculated as the eroded area per meter of streambank, while deposition was measured as the gain or addition of sediment on the streambank, both in $m^2\ m^{-1}\ year^{-1}$. Deposition was measured for several of the stabilized streambanks, often occurring just upstream of the jetty (Figure 2.3). The net streambank erosion for each reach was calculated for each time period by subtracting total accumulation from total erosion. To determine the success of each stabilized practice, we compared lateral streambank retreat for the pre-stabilization and post-stabilization time periods and to control sites. We used the Mann–Whitney test, a non-parametric test that could indicate whether the population medians of the two groups differed. Statistics were calculated using Minitab 17 Statistical Software (TIBCO Software Inc., Palo Alto, CA, USA), [29].

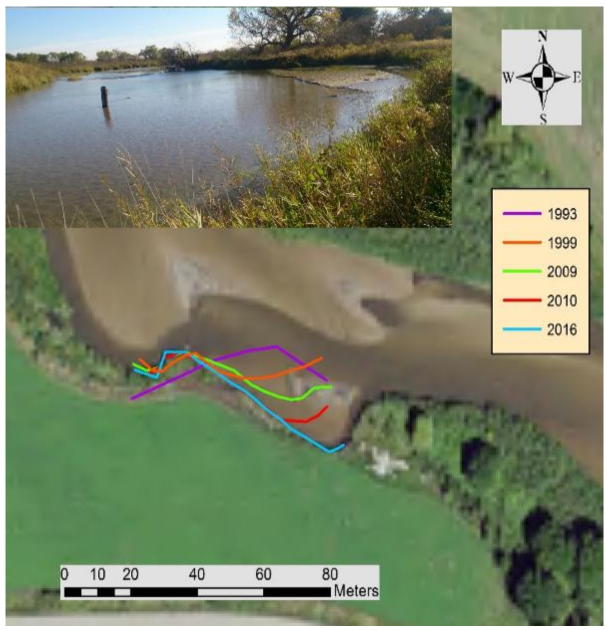
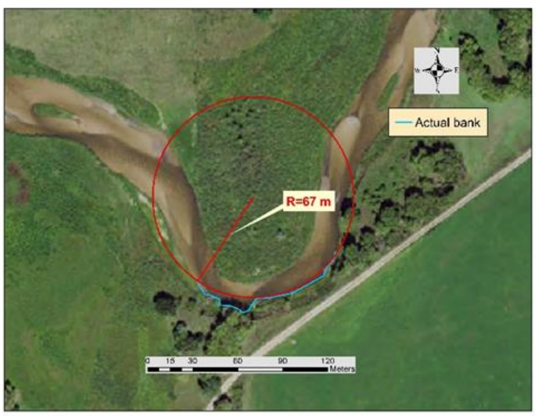
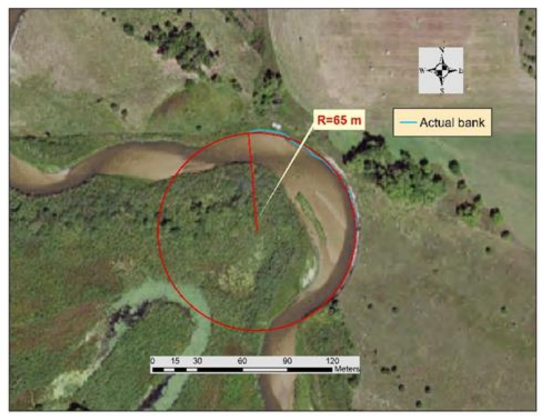


Figure 2.4 Site 11 in 2016. This site had two jetties that failed (insert). The flow direction is from left to right. The streambank retreat was measured by drawing a polyline where the streambank was for each aerial image. The area between the polylines was then calculated for each period.

2.3.3. Identification of Control Sites

Since flow characteristics during the pre-stabilization and post-stabilization periods varied, control sites were used to compare streambank erosion for similar streamflow. Control sites were identified based on the following criteria: no dam located between the stabilized streambank and control site since the dams alter streamflow and sediment transport; the sites had a similar radius of curvature; and the sites had similar riparian vegetation. Vegetation was labeled as sparse grassland, dense grassland, or cropland. The radius of curvature was measured by drawing a circle that would coincide with the shape of curvature of the stabilized bank. Since the streambanks were uneven and it was difficult to measure the precise radius of curvature, the inflection point at the beginning of the site was used to measure the radius from the center of the circle. Figure 2.5 illustrates the radius of curvature for Site 4 and the control site in 2016. Values for radius of curvature in Table 2.1 were calculated using the 1993 aerial images.





(**a**) (**b**)

Figure 2.5 The left image (a) shows the radius of curvature of the stabilized bank (Site 4) in 2016; the right image (b) shows the radius of curvature of one of the three control sites for Site 4 (The radius was 64 m in 1993). Note that both sites have cropland on the outer bank and dense trees on the inside bank. The flow direction is from left to right.

One to three control sites were identified for each stabilized streambank. The 40 identified control sites were selected in the near vicinity of the actual stabilized bank to ensure similar streamflow. Streambank retreat, deposition, and net streambank erosion were calculated for each of the control sites using the same methodology as that for the stabilized streambanks. Total erosion, total accumulation, and total retreat during all four periods (pre-stabilization, post-stabilization, flood, and post-flood) on the control sites represent the natural conditions of the streambank.

2.4. Results and Discussion

To determine the success of the implemented practices, the streambank erosion was calculated for 18 stabilized banks and 40 control streambanks. The objectives of this study were to (1) quantify streambank erosion rates from 1993 to 2016; (2) evaluate the impact of the flood in 2010; and (3) determine the most cost-efficient practice.

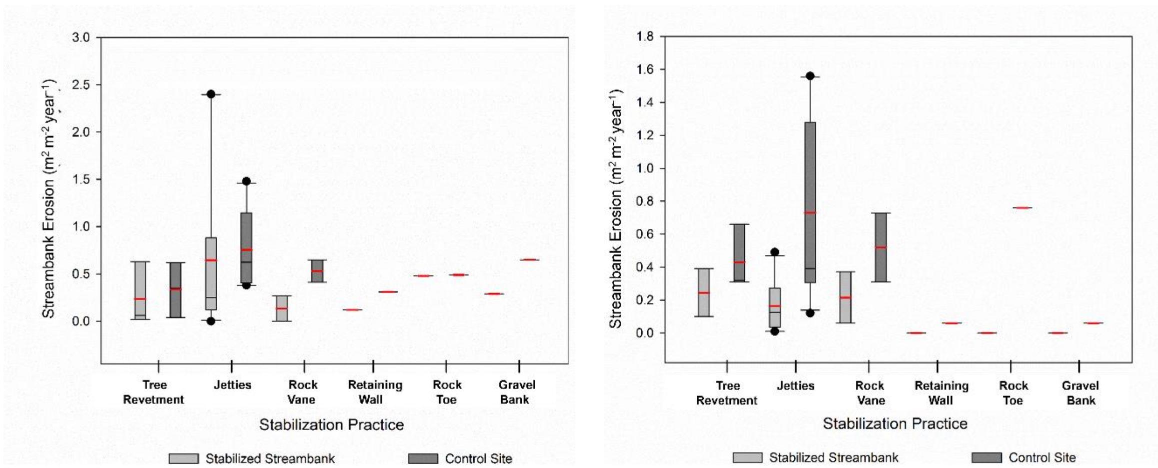
2.4.1. Prestabilization and Poststabilization

The average streambank erosion rate for the 18 stabilized streambanks before (1993–1999/2003) and after stabilization (1999/2003–2009) was 0.45 m^2 m^{-1} $year^{-1}$ and 0.16 m^2 m^{-1} $year^{-1}$, respectively. Assuming an average bank height of 2.1 m, the average total volume is 0.95 m^3 m^{-1} $year^{-1}$ and 0.34 m^3 m^{-1} $year^{-1}$. Since the data were not normally distributed, a nonparametric test was used to determine if the pre-stabilization and post-stabilization medians were significantly different. With a p-value of 0.34, the medians (0.22 m^2 m^{-1} $year^{-1}$ and 0.13 m^2 m^{-1} $year^{-1}$) were not significantly different. This may be attributed to the difference in streamflow between the two time periods; however, this supposition could not be tested since streamflow data prior to 2006 were not available from either gage station.

The average streambank erosion rate for the jetties was 0.65 m^2 m^{-1} $year^{-1}$ pre-stabilization and was significantly different from post-stabilization (0.16 m^2 m^{-1} $year^{-1}$). The streambank erosion rate for the retaining wall, rock toe, and gravel bank was zero following streambank stabilization (Figure 2.6). The tree revetments and rock vanes were less successful, with a small amount of erosion occurring. Similar to what we discovered, Brown [30] found that 27% of the banks protected with rootwad revetment (similar to cedar tree revetment) partially or fully failed to protect the streambank and that outflanking occurred at 25% of the rock vanes assessed. He also found that riprap (similar to the rock toe and gravel bank) fully protected the streambanks. Two years after installation, Buchanan et al. [13] found that 28% of the bank vanes (which are vanes that do not go across the entire river width but protect only the bank at the desired location) failed and that 36% were impaired. One limitation to comparing differences in streambank erosion rates among the practices observed within the present study is the limited number of observations for several of the practices.

To distinguish the impact of varied streamflow between the pre-stabilization and post-stabilization periods, the streambank erosion rate was compared with the 40 control sites (Table 2.2). Although we intended to identify control sites that mimicked our study sites, the median streambank erosion was significantly greater for the control sites (0.55 m^2 m^{-1} $year^{-1}$) than the stabilized streambanks (0.22 m^2 m^{-1} $year^{-1}$) during the pre-stabilization period with a *p*-value of 0.0042. The average streambank erosion rate pre-stabilization for the stabilized streambanks was 0.45 m^2 m^{-1} $year^{-1}$ with a range of 0–2.4 m^2 m^{-1} $year^{-1}$ compared with 0.63 m^2 m^{-1} $year^{-1}$ and a range of 0.04–1.5 m^2 m^{-1} $year^{-1}$ for the control sites. The median streambank erosion rate during the post-stabilization period between the stabilized and control streambanks were significantly different (a *p*-value of 0.0002) with median streambank erosion rates of 0.13 m^2 m^{-1} $year^{-1}$ and 0.38 m^2 m^{-1} $year^{-1}$ for the stabilized and control sites, respectively. The jetties were successful with the median erosion for the stabilized streambanks and controls sites, being 0.13 m^2 m^{-1} $year^{-1}$ and

0.39 m^2 m^{-1} $year^{-1}$, respectively. The erosion was significantly different with a *p*-value of 0.0036. The more expensive practices (rock toe, retaining wall, and gravel bank) were all successful and completely stabilized the streambanks.



(**a**) (**b**)

Figure 2.6 Boxplots illustrating the streambank erosion rates for the pre-stabilization (a) and post-stabilization (b) periods for the stabilized streambanks and control sites. Red lines indicate the mean erosion rates.

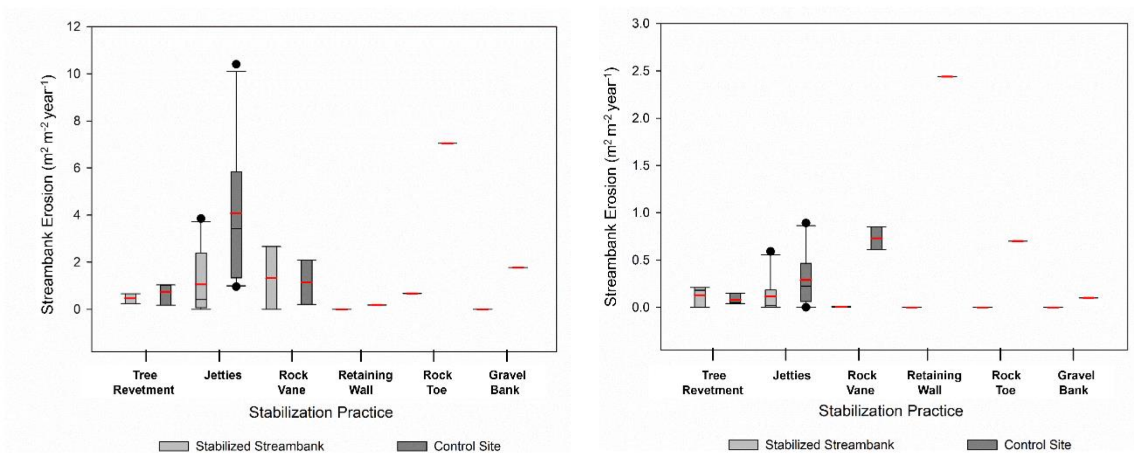
Table 2.2 Calculated streambank erosion rate for each site compared with the corresponding control site(s). Each control site value is an average of erosion of one to three identified control sites. All units are m^2 m^{-1} yr^{-1}.

Site	Prestabilization		Poststabilization		Flood		Post-Flood	
	Stabilized Banks	Control Site	Stabilized Banks	Control Site	Stabilized Bank	Control Site	Stabilized Bank	Control Site
1	0.06	0.62	0.24	0.66	0.24	0.17	0.18	0.05
2	0.02	0.04	0.1	0.32	0.66	1.03	0	0.04
3	0.63	0.35	0.39	0.31	0.49	1.02	0.21	0.15
4	0.27	0.65	0.06	0.31	2.66	2.09	0.01	0.61
5	0	0.39	0.04	0.33	0.29	0.96	0	0.15
6	0	0.85	0.24	0.43	0.61	2.71	0	0.08

7	0.12	0.31	0	0.32	0	0.18	0	2.44
8	0.12	1.29	0.28	0.45	0	10.4	0	0.42
9	0.48	0.49	0	0.76	0.67	7.05	0	0.7
10	0.23	0.38	0.12	1.21	0.57	5.18	0.16	0.07
11	0.39	0.77	0.27	1.56	2.29	7.32	0.25	0
12	0	0.41	0.37	0.73	0	0.2	0	0.85
13	0.29	0.65	0	0.06	0	1.77	0	0.1
14	0.21	1.48	0.01	1.21	0.09	5.35	0.03	0.03
15	0.12	0.41	0.02	0.33	3.85	1.34	0.59	0.89
16	2.4	1.1	0.13	0.12	0	1.31	0	0.2
17	2.35	0.47	0.49	0.3	0.5	3.23	0	0.25
18	0.38	0.6	0.13	0.92	0.32	3.61	0.01	0.27

2.4.2. Flood Impact

The flood in 2010 caused a significant quantity of erosion, especially at the control sites (Figure 2.7). The average erosion rate during the flood was 0.74 $m^2\ m^{-1}$ and 3.1 $m^2\ m^{-1}$ for the stabilized streambanks and control sites, respectively. The medians, 0.40 $m^2\ m^{-1}$ and 1.93 $m^2\ m^{-1}$, were significantly different with a p-value of 0.0013. Four sites downstream of Spalding failed because their structure lost functionality during the flood: Sites 10, 16, and 18, which have jetties, and Site 12, which has a rock vane. Even with some of the jetties failing at Sites 10, 16, and 18 (Figure 2.3), the average streambank erosion rates for jetties for the stabilized streambanks and the control sites were significantly different (0.007) with a median erosion of 0.41 $m^2\ m^{-1}$ and 3.42 $m^2\ m^{-1}$, respectively. Minimal erosion occurred at the top of the streambank at Site 9 (rock toe). No erosion occurred on the streambanks stabilized with the retaining wall and gravel bank. For comparison, Miller and Kochel [31] conducted an assessment of 26 stream restoration projects across North Carolina and almost 30% of the structures underwent partial or total damage and lost their functionality. Though these were stream restoration projects, calculated results mainly represented the erosion control practices. Spatially, the damage was done at 10 out of 16 sites and amongst those sites, cross vanes and double wings exhibited the greatest damage out of six different types of practices: cross vanes, rock vanes, j-hook, double wings, log vanes, and rootwads [31].



(**a**) (**b**)

Figure 2.7 Streambank erosion rate for the flood period (2009–2010) (a) and post-flood period (2010–2016) (b) for the stabilized streambanks and control sites.

Previously, Dragićević et al. [32] evaluated flood erosion rates using aerial photography on the Kolubara River, which experienced a flood wave in May 2014, and found that the land loss in 2014 was three times larger than in 2013. Comparable to that study, the average erosion rate for the control sites was five times larger during the flood in 2010 (3.05 m^2 m^{-1} $year^{-1}$) compared to the time period 2001–2009 (0.57 m^2 m^{-1} $year^{-1}$).

We also compared the erosion upstream of Ericson Dam (2 sites) to the 4 sites downstream of the breached dam and to the 10 sites downstream of Spalding Dam. For the stabilized streambanks, the medians were not significantly different from the upstream sites (0.45 m^2 m^{-1} $year^{-1}$), those downstream of the Ericson Dam (0.55 m^2 m^{-1} $year^{-1}$), or those downstream of Lake Spalding (0.21 m^2 m^{-1} $year^{-1}$). Since the lack of a significant difference may be attributed to the types of practices installed at the three locations, we evaluated the erosion at the control sites. The median streambank erosion rate was 0.60 m^2 m^{-1} $year^{-1}$, 1.56 m^2 m^{-1} $year^{-1}$, and 3.42 m^2 m^{-1} $year^{-1}$ upstream of Lake Ericson, downstream of Lake Ericson, and downstream of Lake Spalding, respectively. The erosion immediately downstream of the breached dam at the control sites was surprisingly less than the erosion below Spalding Dam. This may be a function of a larger number of control sites or possibly the differences in streambank and channel characteristics. Comparing the stabilized streambanks and the control sites, the median streambank erosion downstream of Lake Ericson was not significantly different, with a *p*-value of 0.19, but was significantly different downstream of Spalding Dam (a *p*-value of 0.0029).

2.4.3. Post-Flood Impact

The flood influenced the stabilized streambanks in an unexpected manner. We expected more erosion during the post-flood period than the pre-flood period (post-stabilization) due to the disturbance caused by the extreme flow event. For this project, however, excluding Sites 10, 14, and 15 (all jetties), the post-flood erosion was much lower than the pre-flood erosion. The median post-flood streambank erosion rate was 0.0 m^2 m^{-1} $year^{-1}$ for the stabilized streambanks and 0.17 m^2 m^{-1} $year^{-1}$ for the control sites, significantly different with a *p*-value of 0.001. The average erosion rate was 0.08 m^2 m^{-1} $year^{-1}$ and 0.41 m^2 m^{-1} $year^{-1}$, respectively. The median pre-flood (0.13 m^2 m^{-1} $year^{-1}$)

and post-flood (0.0 m^2 m^{-1} $year^{-1}$) streambank erosion rate was significantly different for the stabilized streambanks, with a p-value of 0.03. The same was true for the control sites with median pre-flood and post-flood erosion of 0.38 m^2 m^{-1} $year^{-1}$ and 0.18 m^2 m^{-1} $year^{-1}$ (a p-value of 0.026). Part of this may be explained by the difference in flow. At the Spalding gage station, the average flow pre-flood (2006–2010) and post-flood (2010–2016) were both 5.5 m^3 s^{-1}, although the standard deviation and maximum flow were higher in the pre-flood period (2.97 m^3 s^{-1} and 40.9 m^3 s^{-1}) than the post-flood period (2.11 m^3 s^{-1} and 26.0 m^3 s^{-1}).

2.4.4. Deposition

The average deposition for the stabilized streambanks and control sites for the pre-stabilization period was 0.15 m^2 m^{-1} $year^{-1}$ and 0.14 m^2 m^{-1} $year^{-1}$, respectively. During the post-stabilization period, the average deposition was 0.24 m^2 m^{-1} $year^{-1}$ and 0.09 m^2 m^{-1} $year^{-1}$ for the stabilized streambanks and control sites, respectively. Based on a Mann–Whitney test, the medians (0.14 m^2 m^{-1} $year^{-1}$ and 0.05 m^2 m^{-1} $year^{-1}$) were not significantly different, with a p-value of 0.07. For post-stabilization, the deposition for the jetties was significantly different (a p-value of 0.007) with a median deposition of 0.19 m^2 m^{-1} $year^{-1}$ and 0.03 m^2 m^{-1} $year^{-1}$ for the stabilized and control sites, respectively. This is not surprising since a function of jetties is to increase the sediment deposition upstream of each jetty.

The control site for jetties had less accumulation (0.51 m^2 m^{-1} $year^{-1}$) than the stabilized banks (0.99 m^2 m^{-1} $year^{-1}$) during the flood, which shows a very efficient sediment capture. However, it is believed that excessive sediment deposition causes failure of some structures because it leads to a loss of channel capacity and subsequent change in the stable dimension, pattern and profile of the river [7].

Not surprisingly, during the flood, the high deposition rates were upstream of the breached dam and downstream of Lake Spalding. Between Lake Ericson and Lake Spalding, there was very little deposition. The average deposition at Sites 3 to 6 downstream of Lake Ericson was 0.23 m^2 m^{-1} $year^{-1}$ and 0.34 m^2 m^{-1} $year^{-1}$ for the stabilized and control sites, respectively. Downstream of Lake Spalding, the deposition was much higher, 0.83 m^2 m^{-1} $year^{-1}$ and 1.23 m^2 m^{-1} $year^{-1}$ for the stabilized and control sites, respectively.

2.4.5. Cost-Effectiveness

Although a limitation of the cost-efficiency analysis is the number of sites for some of the practices, the analysis still provides insight into the efficiency of the various practices. Though the exact installation cost per site is unknown, the approximate cost per meter of installation are as follows: jetties: $26; cedar tree revetments: $72; rock vanes: $205; rock toe: $179; retaining wall: $625; and gravel bank: $600 (Table 2.3). Cost efficiency is calculated as the reduction in streambank erosion per dollar spent; hence, the larger reduction per dollar spent, the higher the efficiency. To calculate the efficiency, we compared the streambank erosion rate for the post-stabilization period from 1999/2003 to 2009 and 1999/2003 to 2016 for jetties, rock vanes, tree revetments, rock toe, and the retaining wall/gravel bank to the control sites using the following equation:

$$\text{cost effectiveness} = ([(CS - SB)/CS] \times 100)/\text{cost of installation} \quad (2\text{-}1)$$

where CS is the average streambank erosion at the control sites, and SB is the average streambank erosion at the stabilized streambanks.

Table 2.3 Percent reduction in streambank erosion rate and cost efficiency for each stabilization practice for the post-stabilization period (1999/2003–2009) and the period from 1999/2003–2016, which included the flood and post-flood periods.

Stabilization Practice	Average Cost Per Meter	(1999/2003–2009) % Reduction in Erosion Rate	(1999/2003–2009) Cost Efficiency (%)	(1999/2003–2016) % Reduction in Erosion Rate	(1999/2003–2016) Cost Efficiency (%)
Jetties	26	76.9	2.99	73.7	2.86
Tree Revetments	72	43.5	0.61	28.8	0.40
Rock Vane	205	47.2	0.23	66.1	0.32
Rock toe	179	100	0.56	96.7	0.54
Retaining Wall	625	100	0.16	100	0.16
Gravel Bank	600	100	0.17	100	0.17

This comparison assumes that the streambank erosion rate would have been the same at the study sites as the control sites if the streambanks were not stabilized. The number of sites stabilized by each practice is not the same, and some practices, like rock vanes, have very few sites. Hence, to overcome the limitations of our data set, calculations were conducted using the average reduction in erosion per meter per dollar spent. The calculations showed jetties to be the most cost-efficient technique, with reductions of 2.99% and 2.86% per dollar spent per meter for the periods of 1999/2003 to 2009 and 1999/2003 to 2016, respectively. Other than the two expensive stabilization practices, rock vanes were the least cost-efficient, with reductions of 0.23% and 0.32% for the two periods. Rock toe and tree revetments were similar in cost-efficiency, averaging 0.50%

(tree revetments) and 0.55% (rock toe) for the two periods. Reductions per dollar spent for the retaining wall/gravel bank were 0.16% and 0.17% for the two periods. With the retaining wall/gravel bank costing around $600 per meter of stabilized streambank, one needs to ask if the cost is justified. The jetties, at a cost of only $26 per foot, reduced the streambank erosion rates significantly compared with the control sites, though 3 of the 10 failed. Based on these results, jetties are successful at stabilizing the streambank at minimal cost, though at some risk, and thus should be installed if some risk can be taken. If it is imperative that the streambank not fail (to protect infrastructure, for example), a retaining wall or gravel bank should be used since their likeliness to fail is minimal.

2.5. Conclusions

Streambank erosion has increased due to land use change and urbanization. Multiple practices are used to stabilize streambanks, but there is a need to find the most efficient. In 1950 and from 2000 to 2004, 18 streambanks were stabilized on the Cedar River using six techniques: jetties (10), rock-toe protection (1), slope reduction/gravel bank (1), retaining wall (1), rock vanes (2) and tree revetment (3). Using historical aerial images, we documented the streambank erosion and deposition for 18 stabilized streambanks and 40 control points for four periods of time: pre-stabilization, post-stabilization, flood, and post-flood. We found that stabilized banks were more efficient than similar control sites at controlling erosion. Comparing the six erosion-control practices to one another allowed us to identify which was the most efficient and cost-effective. We found that the structurally designed stabilization practices, such as the retaining wall and gravel bank, demonstrated immovability even during the flood in 2010 but required a large investment ($600 per meter). Other practices such as jetties, rock toe, and tree revetments require negligible capital investments but have higher rates of failure. We also found that rock vanes were the least effective, failing during the flood. Although rock vanes were the least effective, our study had a small sample size (two) with which to compare. As mentioned previously, the limitation of this study was the small number of sites for several of the stabilization practices.

For this project, 3 out of 10 sites stabilized with jetties failed during the flood; however, that equates to a 70% success rate at a minimal cost ($26 per meter of stream). In terms of cost efficiency, jetties should be preferred since their cost is much lower, materials are easily available, and they are most efficient at capturing sediment. In conclusion, jetties were the most cost-efficient technique on the Cedar River and should be used if minimal failure of risk is acceptable. The jetty installed in 1950 demonstrates that jetties can endure and be successful for extended periods of time. Our study indicated that structurally designed practices such as retaining walls might be worth the investment only if failure cannot be risked. Because of the ease of installation, affordability, and stability, agricultural producers and others may prefer using jetties.

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Conflicts of Interest: The authors declare no conflict of interest.

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CHAPTER 3 IMPACT OF AN EXTREME FLOOD EVENT ON STREAMBANK RETREAT ON CEDAR RIVER, NEBRASKA

3.1. Abstract and Keywords

Abstract

Streambank migration is a natural process categorized by simultaneous erosion and accumulation processes. Some of the factors influencing these processes are streamflow, soil type, land-use change and riparian vegetation. Extreme flood events can cause failure of hydraulic structures, high amounts of sediment yield, and loss of valuable land to streambank erosion. Little to no literature documenting the impact of a single extreme flood on streambank processes has been published to date. The objectives of this study were to (i) evaluate the impact of an extreme flood event in 2010 on streambank retreat on 45 km of the Cedar River relative to the average annual retreat from 2006 to 2016; (ii) quantify the changes in streambank retreat for each kilometer downstream of the breached dam; and (iii) evaluate the impact of riparian vegetation and radius of curvature on streambank retreat. The average annual streambank retreat was calculated during the pre-flood (2006-2010), flood (2010) and post-flood (2010-2016) periods for each kilometer downstream of the dam using aerial images and ArcGIS. The average annual streambank retreat during the flood period, 2820 $m^2\ km^{-1}\ yr^{-1}$, was significantly higher than pre-flood and post-flood periods, 576 and 384 $m^2\ km^{-1}\ yr^{-1}$, respectively. The flood peaked at approximately three times greater than then the next highest recorded discharge, with a return period of 2000 years. Riparian vegetation was quantified using Normalized Difference Vegetation Index (NDVI) to evaluate its impact on streambank erosion for the pre-flood and flood periods. Neither NDVI nor radius of curvature

correlated well with erosion, indicating that other factors such as soil composition and root depth and density may be driving streambank erosion for this river.

Keywords: Flood impact, Riparian vegetation, Streambank retreat, Sediment loading, Erosion

3.2. Introduction

Streambanks undergo continuous erosion and deposition. Streambank erosion along the outside of a meander and deposition inside the bend is a part of natural channel migration (Kondolf, 1996). However, accelerated rate of streambank erosion caused by land use or climate change can cause negative environmental impacts. Geomorphologic characteristics of the river basin, soil composition, flow characteristics and vegetation are the main drivers of streambank erosion (Dragićević et al., 2017; Petrovszki et al., 2014; Viktoria and K.I.S.S., 2011; Zaimes et al., 2004). Physical impacts of increased streambank erosion include accelerated sediment yields, changes in stream form, channel instability and associated stream type changes (Rosgen, 2001). Changes in stream properties can be natural or human induced. It is important to understand the behavior of streambanks relative to extreme flow events to protect loss of valuable land and preserve the aquatic habitat.

Previous data indicated that floods and droughts have caused more damage in the U.S than any other disaster (Lins and Slack, 1999). Public concern has since increased because of the damage to valuable property and increased media coverage (Hall et al., 2014). Previously, studies have been conducted to estimate the future flood risk and have stated that the frequency of floods is increasing around the world (European Commission DG, 2016; Hall et al., 2014; Lins and Slack, 1999; Rogger et al., 2017). It is difficult to trace the precise reason of flood events due to its episodic nature and classification as a natural disaster. According to Hall et al. (2014), there are three major potential drivers of floods: climate change, land use change and hydraulic structures. Previous studies have investigated the relationship between climatic variation and occurrence of flood events (Hall et al., 2014; Lins and Slack, 1999; Rogger et al., 2017). However, human modifications such as channel straightening, streambank protection and flow regulation force the river in an encapsulated state and hence, increase the risk of floods (Vandenberghe et al., 2012). Studies have also shown a strong correlation in land cover and flood events, explaining that urbanization or loss of vegetation increases the intensity of runoff (Evrard et al., 2007; Wheater and Evans, 2009). Additionally, it is also discussed that agricultural practices may intensify floods because of the changes in flow paths, flow velocities, water storage, flow connectivity and concentration times (Rogger et al., 2017). Hydraulic structures serve as a major barrier to natural flow of water and hence may increase flood risk (Hall et al., 2014). Additionally, hydraulic structures may undergo failure due to one or more reasons including natural events or structural deficiencies hence causing sudden peaks in flows (Federal Emergency Management Agency, 2017). According to a 2016 report by National Inventory of Dams, out of more than 90,850 dams in the US, 14,726 dams fall under "high" or "significant" hazard potential threat to property, environment, and life (Federal Emergency Management Agency, 2017).

Observational data from flood events is very important in understanding flood regime changes and to formulate better flood management strategies (Hall et al., 2014). There is a lack of specific data related to quantifying erosion relative to flood events. Previously, Matsumoto et al., (2016) evaluated erosional and sedimentary features resulting from flooding near a breached levee. According to the European Commission DG (2016), structural changes caused by flooding may impact river ecology as a whole, even more than the direct impact of a flood. However, research indicating the impact of single flood event on streambank erosion is lacking.

The impact of elevated streamflow can be minimized by implementing flood control structures, providing riparian vegetation or stabilizing the streambank. From the records collected for river and riparian restoration projects, riparian management projects rank only sixth in terms of allocated cumulative costs despite being the most common project intent category across the southwest US, transcending 13 other categories such as water-quality management, streambank stabilization and streamflow modification (Follstad Shah et al., 2007). For regular flows, riparian vegetation serves as one of the most efficient solutions for erosion control. Zaimes et al., (2004) found that using riparian forest buffers on the 11 km stream reach in Central Iowa could reduce streambank soil loss by 72%. Vegetated banks decrease turbulence and increase soil moisture making the streambank less susceptible to erosion (Kuehn, 2015). Apart from erosion control, riparian vegetation also helps to conserve river ecology by providing habitat and food to wildlife, increases roughness factor, and adds an aesthetic value. Investigators can understand potential impacts of altering the flow regime and floodplain by connecting hydrogeomorphic models with vegetation patterns (Auble et al., 1994; Larsen et al., 2006).

This study evaluated the impact of flooding on the Cedar River in Nebraska caused by a breached dam at Ericson Reservoir in June 2010. No monitoring has been conducted to evaluate the impact of the flood on the streambank erosion. Furthermore, no assessment of vegetated banks has been made to evaluate the resistance to erosion due to presence of green cover under normal or extreme flow. The evaluation of erosion rates, discharge rates and presence of vegetation was conducted for this project using remote sensing data. Three periods were defined as pre-flood (2006-2009), flood (2009-2010) and post-flood (2010-2016). The objectives of this study were to (i) evaluate the impact of the extreme flood event in 2010 on streambank retreat on 45 km of Cedar River relative to the average annual retreat from 2006 to 2016; (ii) quantify the changes in streambank retreat for each km downstream of the breached dam; and (iii) evaluate the impact of riparian vegetation and radius of curvature on streambank retreat.

3.3. Methods and Materials

3.3.1. Site description

The Cedar River, located in Northcentral Nebraska, originates in the Nebraska Sand Hills and flows approximately 200 km before merging with the Loup River. The western half of the 3,200 square km watershed is located in the Sand Hills while the eastern half is predominantly cropland. The streambank soil composition is mainly typified by silty, vegetated low-lying banks or higher banks with a combination of sandy and silty soil. There are two dams on the river; one near Ericson, Nebraska and the second located 45 km downstream near Spalding, Nebraska. Both serve the purpose of hydroelectrical generation and recreational benefits. This study focuses on the 45 km reach between Ericson and Spalding Dams (Figure 3.1).

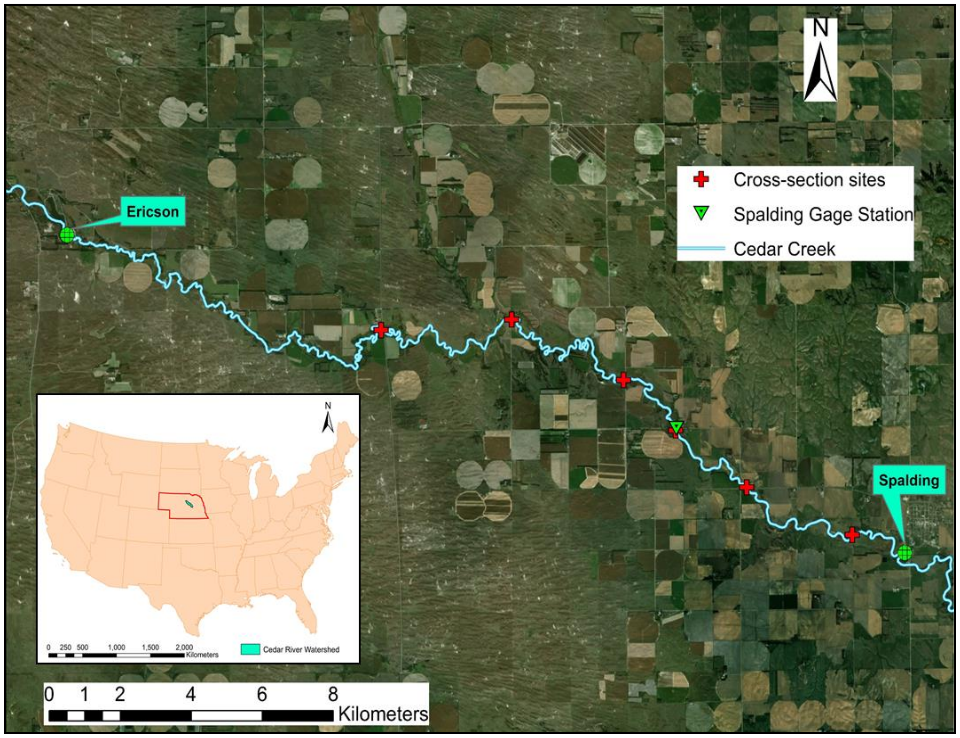

Figure 3.1 The location of our study site located on a 45 km stretch of Cedar River in Northcentral Nebraska from Ericson to Spalding dams. The streamflow is west to east.

Historical erosion in the Cedar River watershed reduced the storage capacity of the two reservoirs. This led to two dredging projects in the 1990's that cost over four million dollars. Much of the sedimentation is believed to originate from streambank failures along Cedar River. The Environmental Trust and Nebraska Department of Environmental Quality contributed $213,000 and $444,000, respectively to stabilize 32 streambanks on the Cedar River to improve the surface water quality by reducing streambank erosion (Dave and Mittelstet, 2017).

3.3.2. Streamflow data and flood event

Streambank erosion is strongly influenced by stream discharge and stream power. Stream discharge from 1944 to 2017 was available at the stream gage near Spalding (DNR, 2018). On June 14, 2010, heavy rains led to the failure of Ericson dam. The breached dam increased the peak flow and created major destruction downstream of the dam (Figure 3.2). The flow peaked at 190.4 $m^3\ s^{-1}$, nearly three times greater than the next highest recorded discharge (Figure 3.3). Stream discharge was used to calculate the stream power and return period for the flood and pre- and post-flood periods.

Figure 3.2 Flooding on Cedar River due to dam failure on June 14, 2010. Pictures by: Nebraska Department of Natural Resources.

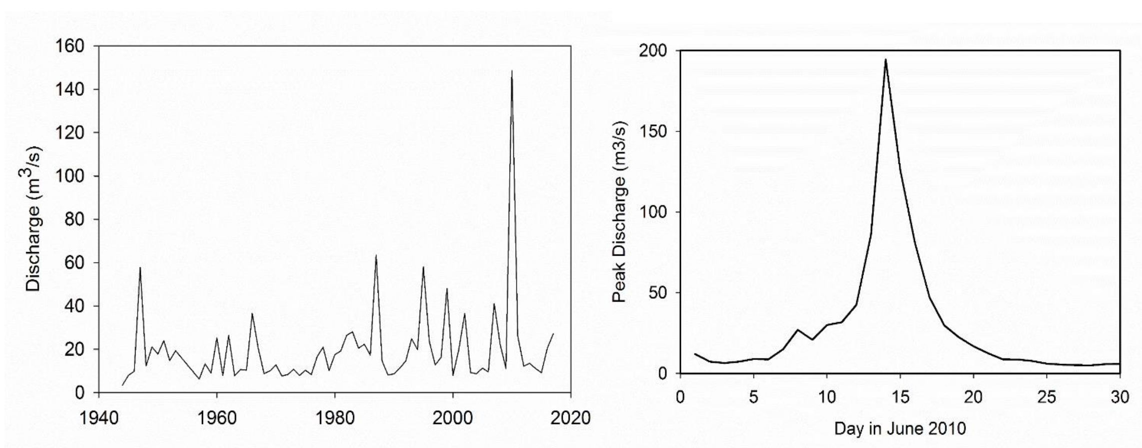

Figure 3.3 Stream flow with daily average discharge near Spalding from 1944 to 2018 (left) and the flood event with daily peaks caused by the breached dam in June 2010 (right).

Stream power is the rate of potential energy expenditure per unit length, expressed in terms of kg m s^{-3} or W m^{-1} (Larsen et al., 2006). Stream power can be used to represent the effects of time varying flow in modelling to account for bank erodibility, curvature and other factors for migration calculations (Larsen et al., 2006; Larsen and Anderson, 2002; Larsen and Greco, 2002). For this study, reach-averaged stream power for each period (pre-flood, flood and post-flood) was calculated to determine its relationship with erosion rates for each period. Stream power was calculated as a product of specific weight of water, discharge and bed slope using the following equation (Larsen et al., 2006):

$$\Omega = \rho_w \, g \, Q \, S \qquad (3\text{-}1)$$

where Ω is the stream power (W m^{-1} of channel length), ρ_w is the water density (kg m^{-3}), g is the gravitational acceleration (m s^{-2}), Q is the stream discharge (m^3 s^{-1}) and S is the slope of river bed (m/m). Based on the average slope of Cedar River (0.07%), the stream power is approximately 7.5 times the stream discharge (Figure 3.4).

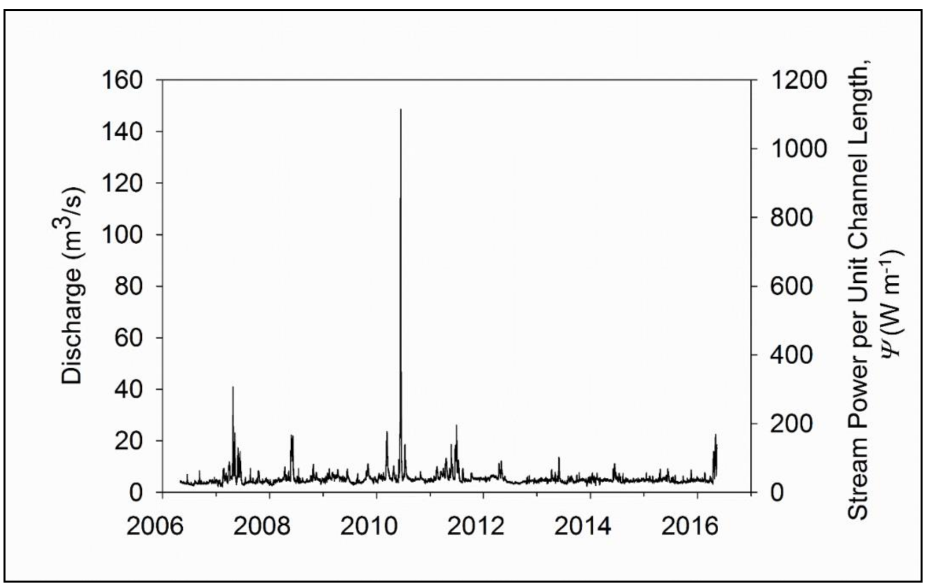

Figure 3.4 The relationship of stream discharge and stream power from 2006 to 2016 indicating the peak in year 2010 for the flood event.

For the return period calculation, years 1944 to 2017, excluding unavailable years 1954-1956 and 2010, were used for the flood analysis with daily data using Log-Person type III distribution. Flow was ranked maximum to minimum for average daily peak flow, leaving out the flood maximum since it was an unnatural event. Our objective was to determine the likelihood of a 190.4 m^3 s^{-1} flow event occurring under natural conditions. Recurrence intervals from 1.01 to 10,000 years were used to document corresponding discharge using log-Pearson Analysis III as described by following equation (U.S. Water Resources Council, 1982):

$$\log(QTr) = avg(\log Q) + [K\ (Tr*Cs)] * \sigma \log Q \quad (3\text{-}2)$$

The return period for the flood was about 2000 years (Figure 3.5). The probability of occurrence of a 190.4 m^3 s^{-1} flow event under natural conditions would only be 0.0005 each year.

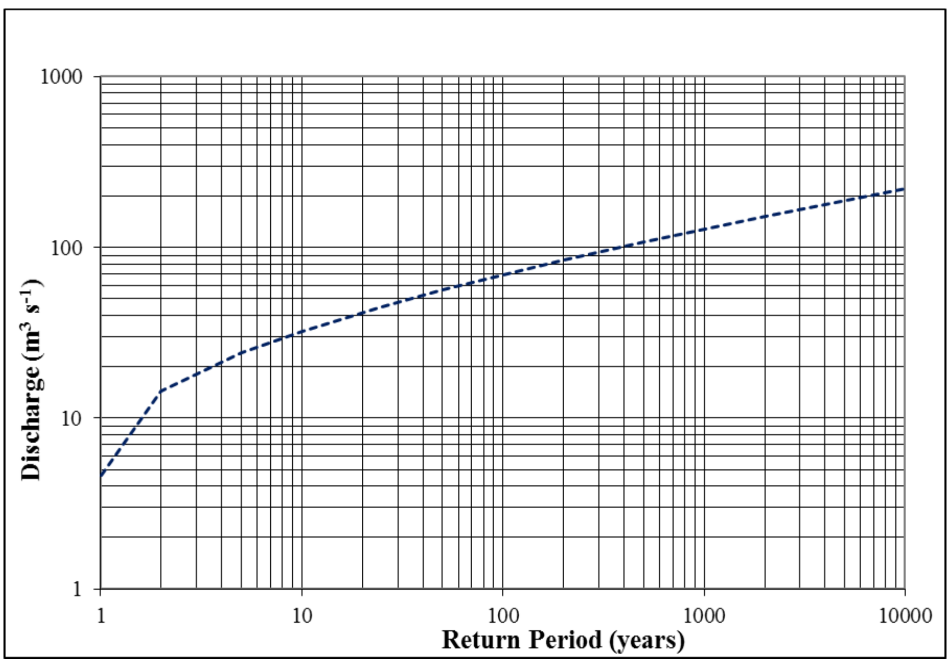

Figure 3.5 Return period for stream discharge using log Pearson analysis III for water year 1944 to 2017.

3.3.3. Streambank data collection

In August 2017, we conducted a total of 19 cross-sectional surveys at five sites on Cedar River (Figure 3.1) using a Topcon Hiper V GPS (Topcon Positioning Systems, Inc., 2018). The cross sections at each site, located approximately 25 m apart, were used to calculate the height of the critical banks. At each site, soil samples were collected from the streambank with a cylindrical soil sampler of 5.4 cm diameter. The 15 collected samples were dried and then the bulk density calculated. Though the sites were spatially distributed along Cedar River, the number of sites was limited due to landowner permission and accessibility.

3.3.4. Streambank retreat analysis

Streambank retreat was estimated using aerial imagery analysis. Currently, the most accurate methods to measure streambank retreat include erosion pins for in situ methods and high precision digital elevation models for the analytical methods. However, bank pins cannot be used to quantify historical erosion rates and LIDAR data was not available for this region. Since no other monitoring has been conducted for this river, aerial image analysis was used to quantify lateral streambank retreat. The method of aerial imagery analysis has been used in many research projects successfully especially in assessment of historical erosion rates (Heeren et al., 2012; Dave and Mittelstet, 2017; Purvis and Fox, 2016). Aerial images from National Agricultural Imagery Program (NAIP) were acquired at 1 m spatial resolution. Since all the streambanks were stabilized by 2005 under Cedar River Corridor Project I and II, we chose the study period 2006 to 2016 to eliminate the influence of the stabilization practices. Images from the years 2006, 2009, 2010 and 2016 were used to measure the difference in streambank retreat. The years 2006-2009 were defined as the pre-flood period. The 2009 and 2010 images were used to measure streambank retreat during the flood event. Although the flood event only lasted for five days in 2010, it was not possible to acquire images from those exact days. However, the 2010 image was collected shortly after the flood on June 28, 2010 and hence could be used as an appropriate estimate for the flood event. The 2010 and 2016 images were used to calculate the streambank retreat for the post-flood period. These images were added as layers in ArcGIS 10.4.1. Polylines were created illustrating the location of each streambank in 2006, 2009, 2010 and 2016. After projecting these lines on each other, erosion was quantified based on the idea that when the streambank lines move away from the stream compared to previous year's polyline, it has undergone erosion and if it has moved inside towards the stream, deposition has occurred. Streambank retreat calculations were conducted for each km on each side of the streambank by measuring the area of the polygon formed due to the shift in the streambank using the measuring tool in ArcGIS.

Deposition can be measured in the same way, but it is challenging to quantify with the use of just aerial images. The deposition may be loose sediment, which might not be detected if it is underwater. In addition, when the water level decreases it may expose a part of channel bed near the bank, creating an illusion of deposition. It is also not possible to determine the depth of deposition using only aerial imagery. Due to each of these challenges, deposition was not quantified in this study.

3.3.5. Streambank erosion and mass

In addition to streambank retreat, it is also important to determine the volume and mass of eroded sediment originating from the streambanks. To calculate the volume and mass of eroded streambank, a Monte Carlo analysis was conducted using @Risk (Palisade, 2016) and streambank height and bulk density. Due to our small sample of streambank height and bulk density measurements, it was important to calculate a range of sediment mass and volume to account for uncertainty. Using the software, 100 simulations with 500 iterations were performed. The inputs of bulk density and bank heights were accounted for uncertainty analysis and was functioned to give a direct output of sediment loads which was also accounted as at-risk output. Sediment load was calculated using streambank retreat, bank heights and bulk density using equation 3:

$$SL = EA * H * \rho_s \tag{3-3}$$

where SL is sediment loading (kg yr^{-1}), EA is eroded area (m^2 yr^{-1}), H is bank height (m) and ρ_s is bulk density of soil (kg m^{-3}).

3.3.6 Riparian vegetation and radius of curvature

To account for the impact of vegetation on streambank erosion during the pre-flood and flood periods, 19 vegetated and 18 barely vegetated meanders were identified visually using the aerial images. Stream bank protection was categorized as protected for dense vegetation or unprotected for sparse or no vegetation. Protected meanders consisted either of thick grass, dense trees or a combination of both. To remove the bias, an NDVI (Normalized Difference Vegetation Index) layer was generated in ArcGIS and compared with the visually-identified meanders. The NDVI layer with greener cover and higher value represent presence of vegetation while red color and lower value resembles water or bare soil (Figure 3.6).

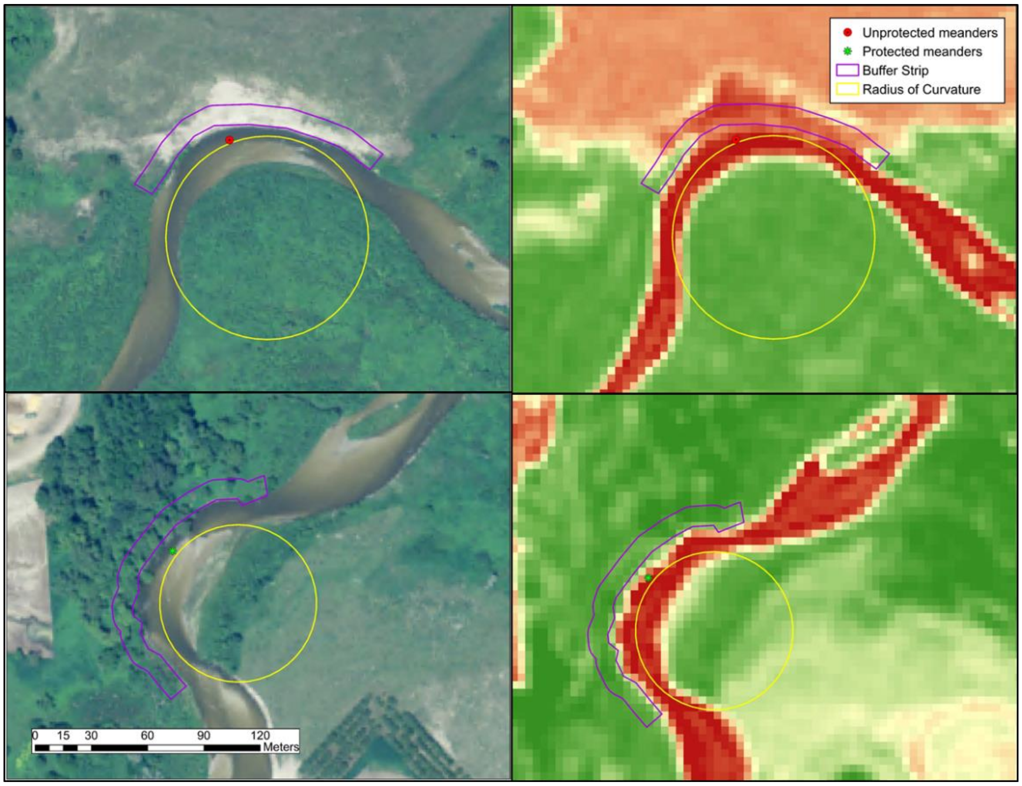

Figure 3.6 Unprotected and protected banks from top to bottom with aerial imagery on the left and NDVI on the right for the year 2009. Polygons on the outside of meanders are selected buffer strips with 10.7 m width. The yellow circles illustrate the radius of curvature.

For the year 2009, aerial image from June 29, 2009 was used to analyze the vegetation and retreat on 37 meanders selected on Cedar River between Lake Ericson and Spalding Dam. For each meander, riparian protection, NDVI, radius of curvature and erosion rates were recorded.

A number of studies have previously tried to aid in design of a buffer strip based on factors like function of strip, slope, land use, precipitation, vegetation type, or more (Dosskey et al., 2011). According to NRCS Conservation Practice Standard codes 393 and 391 (NRCS 2016; NRCS 2010), minimum riparian forest buffer strip width was 35 feet (10.7 m) and that for a grass filter strip for nutrient removal was 20-30 feet (Fox et al., 2005). Ultimately in this study, for the purpose of streambank stabilization, any streambank with vegetation width significantly less than 30-35 feet was not accounted as 'protected'. The NDVI values were calculated using KOMPSAT satellite images with a 4 m resolution (Satellite Imaging Corporation, 2018). The average NDVI for each buffer was recorded along with minimum and maximum values of NDVI to compare the NDVI to the visually-identified vegetated streambanks. The NDVI values were correlated with the erosion rates to quantify a relationship between vegetation and erosion for Cedar River.

3.4. Results and Discussion

3.4.1. Streambank erosion

Streambank retreat was calculated on the 45 km river for each 1 kilometer interval downstream of Ericson Dam in each of the three time periods: pre-flood (2006-2009) flood (2009-2010) and post-flood (2010-2016). The first calculation began 500 m downstream of the dam since the erosion just below the dam was too massive to quantify (Figure 3.7).

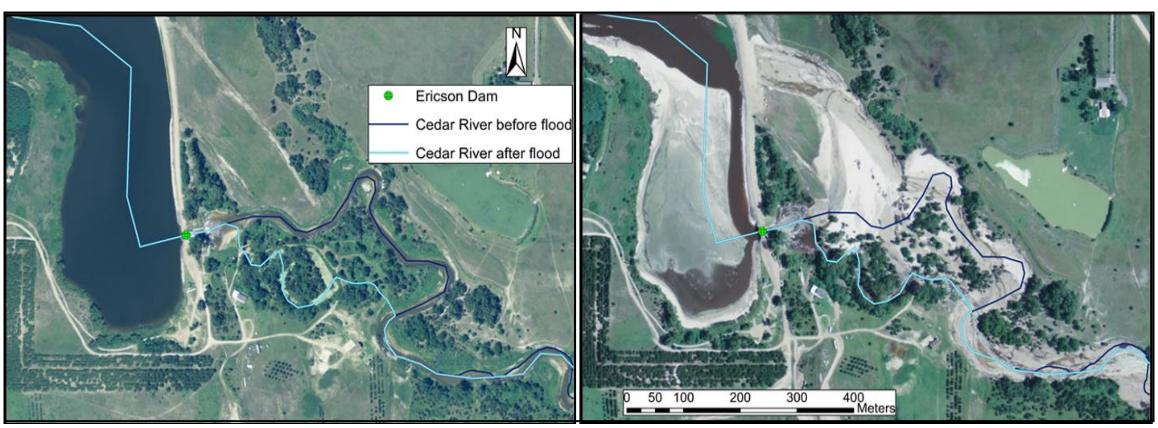

Figure 3.7 Aerial images illustrating the location of Cedar River just below Ericson Dam before the flood event in 2009 (left) and after the flood event in 2010 (right).

The streambank retreat ranged from 212.7 – 1145.2 m^2 km^{-1} yr^{-1} for the pre-flood period, 910.0 – 13171.2 m^2 km^{-1} yr^{-1} during flood period, and 0 – 820.0 m^2 km^{-1} yr^{-1} for the post-flood period (Figure 3.8). The average annual streambank retreat was 575.9, 2819.7 and 383.6 m^2 km^{-1} yr^{-1} for pre-flood, flood and post-flood periods respectively. In Minitab (Minitab, 2018), an ANOVA was used to determine if the streambank retreat was significantly greater for the flood period and upstream vs downstream. To compare the streambank retreat longitudinally, the retreat for each 5 km reach was summed and an ANOVA conducted to compare the retreat for each of the other 5 km reaches. Streambank retreat in first five km during the flood period was significantly greater than each of the downstream five km reaches, most likely due to the sediment-deprived water from Lake Ericson. Statistical analysis showed that erosion rates in first five km of the river did not significantly vary from the rates of remaining downstream intervals for any other period or reach. The pre-flood and post-flood erosion rates were also not significantly different from each other. However, statistics confirmed that overall streambank retreat for the flood period was significantly greater than for the pre- and post-flood retreat rates, reaffirming the significant impact of the flood on streambank retreat (Figure 3.8). The total streambank retreat for the flood period was 124,064 m^2 compared to 76,015 and 101,278 m^2 for the three years pre-flood and six years post flood. From 2006 to 2016, over 40% of the streambank erosion occurred during this one event thus demonstrating the impact that one extreme flood event can have on streambank retreat and the geomorphology of a stream system.

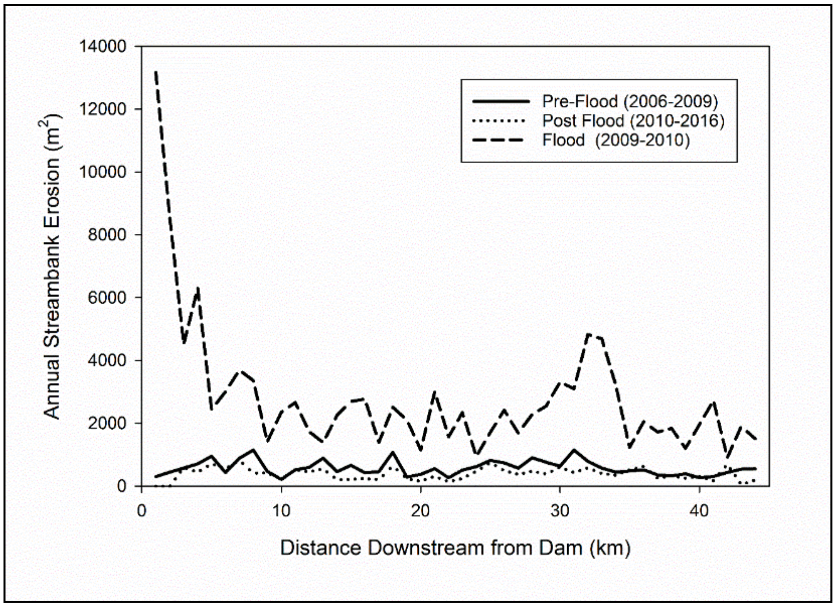

Figure 3.8 Annual streambank retreat rates for pre-flood, flood and post flood periods for each 5-km reach of Cedar River. Distance 0 is 500 m downstream of Ericson Dam.

Using the Monte Carlo analysis, a distribution fit was evaluated for the streambank heights and bulk density. Bulk density had a normal distribution with an Anderson-Darling (AD) value of 0.59 as the best fit with a mean 1.37 g cm^{-3} and a standard deviation of 0.23 g cm^{-3}. Bank height had a lognormal distribution with an AD value of 0.27, a mean of 1.52 m and a standard deviation of 0.63 m as the best fit. The p-value was 0.10 and 0.61 for bulk density and bank height, respectively. The Monte Carlo simulation resulted in a range of 86.6 g cm^{-2} to 385.3 g cm^{-2} for bank height*bulk density. This value, when multiplied with erosion rates, yielded a sediment load of 11.1 to 49.3 kg m^{-1} yr^{-1} for pre-flood, 54.2 to 241 kg m^{-1} yr^{-1} for flood and 7.48 to 32.9 kg m^{-1} yr^{-1} for post-flood periods. In comparison, a study on Crooked and Otter Creeks in Missouri measured average and maximum erosion rates of 99 and 490 kg m^{-1} yr^{-1} (Peacher et al., 2018). A study on the Barren Fork Creek in Oklahoma reported reached-weighted streambank erosion of 42 kg m^{-1} yr^{-1} (Mittelstet et al., 2017).

Though there has been a lot of streambank erosion on Cedar River, specifically during the flood period, not all the sediment reaches Lake Spalding. Much of the sediment is deposited in the river downstream from where it eroded. Due to limitations of aerial images, deposition rates were not calculated. Though deposition can easily be observed, measuring historical accumulated depth with aerial imagery is not possible. Figure 3.9, from a 2010 aerial image, illustrates the large quantity of sediment deposited during the flood period. For this study, erosion was typically observed at critical banks and deposition was observed on the corresponding opposite bank. However, during the floods, as water levels increased, aerial images could not identify the deposition at banks distinctively. In the post-flood periods, when the water receded, a significant amount of sediment deposition was observed on the river, which might have been brought by the flood event. However, since the depth of sediment deposit is unknown, there is a possibility of overestimating or underestimating total accumulation. Because of the bias and uncertainty of aerial imagery, deposition was not quantified.

Figure 3.9 A reach on Cedar River illustrating the deposition of sediment in post-flood period (right) compared to pre-flood period (left).

3.4.2. Stream power

Stream power evaluated using bed slope and stream discharge has been used previously in multiple studies to evaluate the impact on sediment transport (Begin, 1981; Hickin and Nanson, 1984; Leopold et al., 1965; Sklar and Dietrich, 2004). The stream power and discharge are directly proportional as shown in Figure 3.4. It was observed that with a significant change in stream power, the erosion rates also changed significantly. Although, the post-flood period had lower stream power than that of pre-flood, the erosion rate for the post-flood period was higher. However, the stream power was only 0.5 W m^{-1} higher and hence it was not a significant change. In similar conditions, other factors like soil type and vegetation might dominate rates of erosion. Based on the discharge and bed slope, the average stream power for the pre-flood, flood and post-flood periods was 36.8, 53.8 and 37.3 kg m s^{-3} respectively. Corresponding average annual erosion rates per km were 575.9, 2819.7 and 383.6 m^2 km^{-1} yr^{-1}. (Table 3.1).

Table 3.1 Stream power and annual erosion rates for pre-flood (2006-2009), flood (2010) and post flood (2010-2016) periods.

	Stream Power (W m^{-1})			Streambank Retreat (m^2 km^{-1} yr^{-1})		
Statistic	**Pre-flood**	**Flood**	**Post flood**	**Pre-Flood**	**Flood**	**Post Flood**
Average	36.8	53.8	37.3	575.9	2,819.7	383.6
Median	31.9	38.2	33.9	527.7	2,342.7	391.9
Min	15.8	25.0	18.4	212.7	910.0	0
Max	285.7	1034.4	181.4	1,145.2	13,171.2	820.0

3.4.3. Vegetation Impact

In evaluating the impact of vegetation on streambank retreat, we found unexpected results (Figure 3.10). Out of 37 randomly selected meanders, there were 18 meanders that were completely unprotected, ranging from no erosion to 5.49 $m^2m^{-1}yr^{-1}$ of erosion during the flood period. Nineteen meanders were identified that were completely protected with dense vegetation and/or trees. All the protected meanders showed little to significant amount of erosion ranging from 0.28 to 8.90 $m^2m^{-1}yr^{-1}$ during the flood period. For the pre-flood period, the erosion ranged from 0.06 to 1.77, and 0.08 to 1.59 $m^2m^{-1}yr^{-1}$ for unprotected and protected banks respectively. Statistical analysis shows that protected banks in the pre-flood period with an average of 0.83 $m^2m^{-1}yr^{-1}$ erosion was not significantly different than the unprotected banks with an average of 0.74 $m^2m^{-1}yr^{-1}$ with a p-value of 0.55. For flood period, average erosion rate for protected (2.54 $m^2m^{-1}yr^{-1}$) and unprotected banks (2.10 $m^2m^{-1}yr^{-1}$) also did not show a significant difference with a p-value of 0.52. Though not significantly different, we did expect the protected streambanks to have less erosion than the unprotected streambanks. This conclusion has been found in other studies such as Peacher et al., 2018 and Willett et al., 2012 where they concluded that other factors controlled streambank erosion.

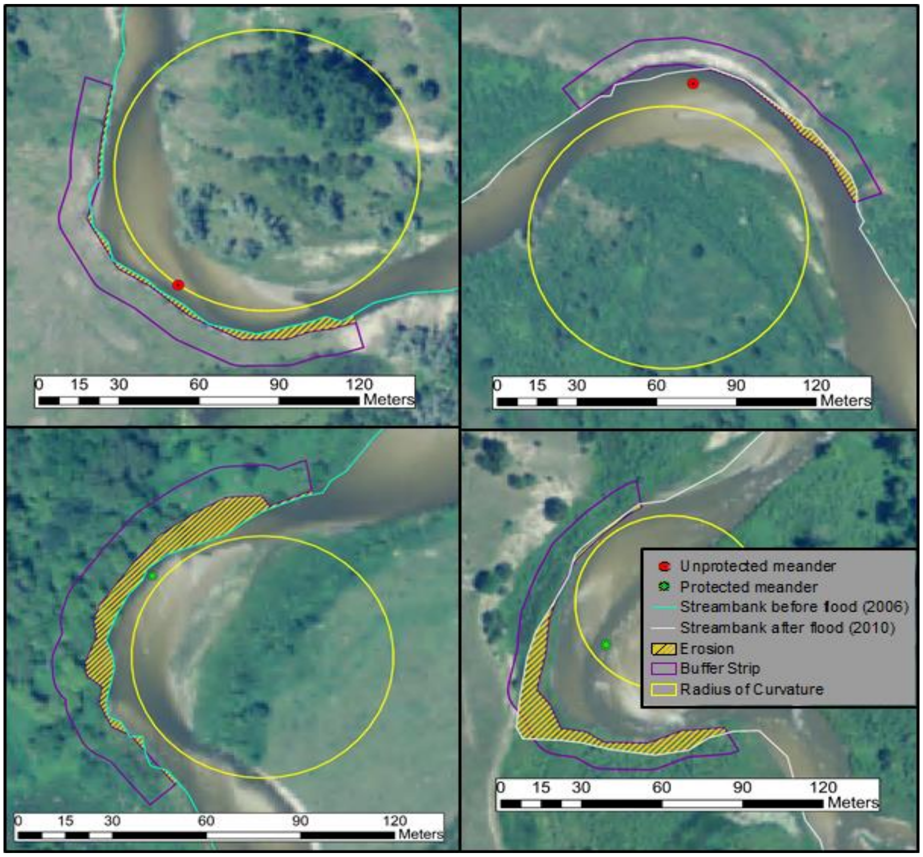

Figure 3.10 Illustration of unprotected streambanks with no erosion and protected streambanks with significant erosion. The two images on the left illustrate erosion during the pre-flood period and on the images on the right illustrate the erosion during the flood period. The 2009 aerial image is used as background for all images.

The average NDVI for the protected streambanks was 0.42, (range: 0.14 to 0.66) and 0.28 (range: -0.02 to 0.45) for the unprotected. There was a significant difference with a p-value of 0.019. This shows that NDVI can be used to aid in the identification of protected streambanks and to avoid visual bias. Relationship between NDVI and erosion did not show any signs of correlation during flood or pre-flood period. For example, during the flood period, out of the top five streambanks with maximum erosion, three were protected banks with NDVI values of 0.49, 0.21 and 0.15.

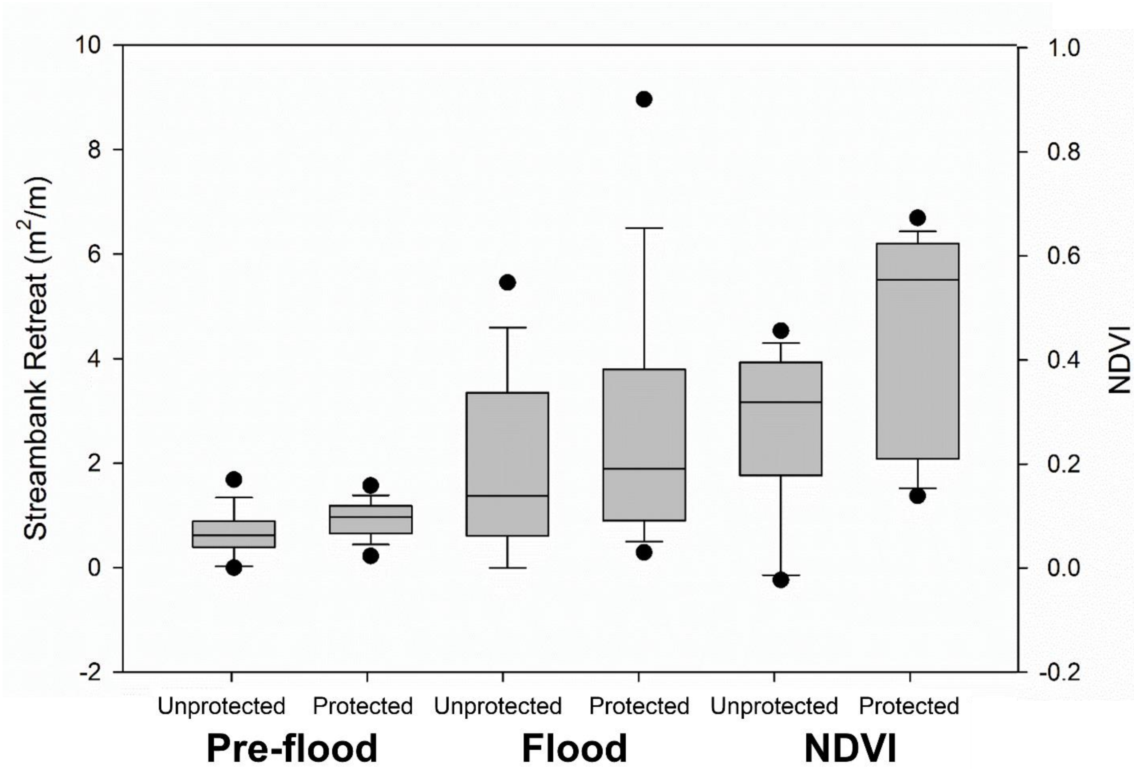

Figure 3.11 The relationship between streambank protection by vegetation, Normalized Difference Vegetation Index (NDVI) and average annual streambank retreat. The first four whisker plots from left, show the relationship with streambank retreat, and the last two show the relationship with NDVI.

3.4.4. Radius of Curvature

Radius of curvature was recorded for all 37 meanders of the stream and was plotted against the average annual streambank retreat for the pre-flood and flood periods. The relationship between radius of curvature and streambank retreat yielded a poor correlation. For example, during the flood period a reach with a radius of curvature of 28 m had only 0.31 m^2 of bank retreat while a reach with a radius of curvature of 68 m had 6.57 m^2 of bank retreat. Although not extremely common, a study by Hudson and Kessel, (2000) indicated the non-linear and non-declining relationship between streambank migration and meander-bend curvature (radius to width ratio), forming an envelope curve with peak migration near the center of the graph. Previously, Daly et al., (2015) has recorded data for radius of curvature and average annual retreat for about 20 meander bends, concluding the large variability in the erosion data and hence failing to record a strong declining correlation.

The poor correlation between streambank erosion, vegetation and radius of curvature signifies that other variables are more relevant in influencing streambank erosion on our study site. Some potential factors for this river include soil erodibility, root depth and type, and bank height. The absence of historical data for each of the factors makes it challenging for us to trace a dominant contributing factor in erosion of this river. Though beyond the scope of this study, further work is needed to document the soil composition of the 37 streambanks as well as the type of vegetation and its corresponding root depth and density.

3.5. Conclusion

This research evaluated a stretch of 45 km of Cedar River in Nebraska flowing from Ericson Dam to Spalding Dam. After the failure of Ericson Dam in 2010, a 2000-year flood event was recorded, causing significant streambank erosion with the average erosion rate nearly five times the pre-flood average erosion rate. The single flood event in 2010 caused over 40% of the total erosion during our study period (2006-2016).

To assess the impact of floods on streambank erosion, we evaluated factors related to stream power, radius of curvature, and riparian protection. There was notable change in erosion during the extreme flood period compared to the periods before and after flood, however, other factors affecting the erosion rates were not conclusive. The strongest correlation to streambank erosion was with stream power. Other factors such as radius of curvature and riparian protection showed a poor correlation with streambank erosion during the extreme flood period, indicating that there are additional factors contributing to the erosion rates.

We used NDVI as a method to quantify riparian vegetation. Although, NDVI did not correlate with erosion, it did correlate with visual analysis of vegetation density and therefore may be used in future studies to accurately represent vegetation conditions. Because of the lack of historical local soil type and streambank cross-section data, we cannot conclude that riparian vegetation failed at controlling erosion. However, we can suggest that it may be beneficial to consider root strength of vegetation and soil properties when evaluating mass failure of critical banks. Radius of curvature of 37 meanders was compared to corresponding streambank retreat in pre-flood and flood periods. A poor correlation was observed between radius of curvature and average annual streambank retreat for both periods, with and without the influence of vegetation.

Evaluation of specific site conditions for each factor contributing to geomorphic processes is very important before making any changes or drawing conclusions for the entire system. This research did not identify a significant impact of radius of curvature and riparian vegetation in controlling streambank erosion during the flood event. Although a significant peak was observed during the flood, causing devastating streambank erosion and failure, radius of curvature and vegetation were not the predominant factors in causing streambank erosion. Further evaluation of other factors impacting streambank erosion may help in making better stream stabilization or restoration decisions for this river.

Acknowledgments

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APPENDIX A: BANK HEIGHTS OF CRITICAL BANKS OF MEANDERS AND BULK DENSITY OF SOIL SAMPLES

Site Number	Bank height (m)	Bulk Density (g cm^{-3})
1	7.87	1.31
	5.63	1.61
	5.91	1.19
	2.94	
2	4.32	1.51
	4.38	1.59
	4.41	1.53
	4.96	
	3.24	
	7.32	
	6.02	
	3.88	
3	6.05	1.07
	3.63	1.59
	5.69	1.43
	5.07	1.26
	14.64	
	6.52	
4	4.71	1.46
	4.61	0.821
	6.18	
	3.09	
	2.90	
	4.44	
	4.12	
	2.92	
5	4.23	1.12
	3.50	1.44
	2.81	1.59
	7.55	
	4.75	
	2.03	

APPENDIX B: SITE CHARACTERISTICS OF UNPROTECTED MEANDERS ON CEDAR RIVER

Meander No.	**Radius of Curvature in 2006 (m)**	**Average NDVI**	**Erosion during Pre flood period (m^2 m^{-1} yr^{-1})**	**Erosion during Flood period (m^2 m^{-1} yr^{-1})**
1	32.1	0.43	0.071	0
4	30.9	0.301	0.486	1.62
5	56.3	0.42	0.872	4.47
6	38.1	0.258	1.77	1.36
7	40.7	0.328	0.056	2.64
8	58.8	0.385	0.34	1.37
9	52.6	0.45	1.15	0.535
12	49.3	0.41	1.42	4.4
14	47.5	0.35	0.483	3.32
20	53.8	0.277	0.484	0.949
21	34.2	0.384	0.59	3.35
25	44.5	0.055	1.61	3.25
28	52.3	-0.013	0.8	0.63
30	32.1	-0.023	0.6	5.49
31	66.5	0.158	0.994	1.59
34	39.9	0.297	0.851	0
35	58.1	0.181	0.255	0.983
37	46.2	0.351	0.489	1.82

APPENDIX C: DETAILS OF SELECTED PROTECTED MEANDERS OF THE RIVER

T=trees; G=Grass

Meander No.	Vegetation Type	Radius of Curvature (m)	Average NDVI	Erosion during Preflood period (m^2 m^{-1} yr^{-1})	Erosion during Flood period (m^2 m^{-1} yr^{-1})
2	T	41.9	0.489	0.998	8.90
3	T	37.4	0.621	0.718	2.06
10	T,G	41.7	0.665	1.21	3.77
11	G	45.7	0.547	0.549	1.39
13	G	30.2	0.637	0.661	1.74
15	G	27.4	0.64	0.625	1.37
16	G	37.5	0.599	1.17	0.615
17	T,G	42.5	0.564	0.651	0.614
18	T,G	47.4	0.617	1.19	1.88
19	T	30.5	0.585	0.645	3.94
22	G	80.0	0.570	0.085	2.34
23	G	28.1	0.207	1.16	0.312
24	G	47.2	0.172	0.854	0.506
26	G	55.3	0.235	0.266	0.889
27	T,G	29.4	0.137	0.849	1.99
29	G	42.2	0.217	1.41	2.59
32	G	67.8	0.206	1.59	6.58

33	G	35.2	0.151	0.871	6.47
36	T	59.2	0.211	0.271	0.284