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Assessing the Geomorphological Effects of Ungulate Exclosures on High Elevation Streams in the Valles Caldera National Preserve, New Mexico

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A catchment-scale approach to the connection between the terrestrial environment and aquatic ecosystems is important because of the context that it provides into stream form and function. A stream's physical and chemical qualities are often delineated, in large part, by the surrounding catchment's terrestrial properties and features, such as geology, soils, vegetation and topography (Van Horn *et al.* 2012). For example, losses in catchment vegetation can result in a change in terrestrial organic carbon inputs into streams (Bunn *et al.* 1999). Four important dimensions of connection exist when describing the physical, chemical, and energetic interactions between the stream itself and its surrounding catchment: 1) Longitudinal interactions that connect upstream and downstream, 2) lateral interactions which connect the terrestrial, riparian and aquatic portions of the catchment, 3) vertical interactions connecting groundwater and surface water, and 4) a temporal dimension, meaning these interactions are subject to a change in magnitude and scale over time on a catchment by catchment and stream by stream basis (Stanford *et al.* 2005).

Abiotic reach-scale stream traits, like nutrient cycling rates and stream geomorphology, coupled with in-stream and riparian biota, are heavily influenced by local catchment characteristics including parent geology, soil chemistry, and both natural and anthropogenic disturbance regimes (Van Horn *et al.* 2012). Ungulate grazing, for example, is a disturbance that can change in intensity both spatially and temporally, and have varying impacts on watershed dynamics (Nipper *et al*. 2013, Laine *et al.* 2015). Similarly, human activity and land-use practices, such as cattle grazing, can alter a watershed as well. Impacts can include physical and chemical effects to streams (Beschta *et al.* 2013, Hough-Snee *et al.* 2013, Batchelor *et al.* 2015). In the United States, nearly one million square kilometers of public land are used for livestock grazing (Batchelor *et al*. 2015). Included in this statistic is nearly 80% of the land under control by the Bureau of Land Management and 60% of land controlled by United States Forest Service (Batchelor *et al*. 2015). Livestock grazing on managed lands tend to convene in and around riparian areas, because of accessibility to water, as well as the availability of riparian forage (Kovalchik and Elmore 1992).

Ungulate grazing (livestock plus native ungulate wildlife) in and around small streams can have major impacts on stream geomorphology, water quality, and surrounding riparian areas. Hydrology is altered by high levels of ungulate grazing in and around streams, with non-grazed or lightly-grazed areas having a more porous quality, resulting in more consistent streamflow year-round (Belsky *et al.* 1999). In areas where high levels of ungulate grazing has occurred, the largest loss of water often becomes evaporation, meaning there is less overall water in the catchment for streams (Asner *et al.* 2004). Streambanks and channels are widened as a result of ungulate grazing, and recovery from disturbance is reduced in streambanks when grazers are present. Grazing also adversely affects riparian vegetation by altering native flora regimes, shifting from woody species to more grazing tolerant grasses. Increasing stocking rates of ungulates also causes greater soil compaction, resulting in less water infiltration (Ranganath *et al.* 2009). Additionally, nutrient inputs, like nitrogen and phosphorus from animal waste, can degrade water quality and cause eutrophication (Beschta *et al.* 2013, Van Horn *et al.* 2012). Decreased stream depth, increased width to depth ratios, decreased streambank angle, and bank retreat from stream edge results from in-stream grazing (Van Horn *et al.* 2012, Lucas *et al.* 2009, Chambers *et al.* 2004). These geomorphological effects can be seen in as little as two years of heavy grazing (Van Horn *et al.* 2012). Additional terrestrial changes, like decreases in native plant biodiversity and destruction of biotic soil crusts, can occur when grazers are present near streams (Batchelor *et al.* 2015).

The objectives of this study are (1) to compare the change, and differences in stream geomorphology both outside and inside of cattle and elk exclosures using spatial characteristics of stream cross-sections taken over a five year (2006-2010) period and (2) to incorporate light detection and ranging (LiDAR) data and stream survey data to create a United States Army Corps of Engineers' Hydrologic Engineering Centers River Analysis System (HEC-RAS) model of Rio San Antonio to examine hydrological differences both outside and inside of cattle and elk exclosures by monitoring overbanking behavior inside and outside of exclosed areas. We hypothesize that banks width to depth ratios will differ in areas when grazers are excluded compared to non-excluded areas, and that over-banking occurs more frequently in areas where grazers are excluded. We also hypothesize that streambanks will migrate and move through time.

Methods

SITE DESCRIPTION:

The Valles Caldera National Preserve (VALL) was established in 2000 in the Jemez Mountains of northern New Mexico (Figure 1). VALL is a 385 km^2 area that includes areas of grasslands, mixed conifer, spruce and ponderosa pine forests. Second and third order streams in VALL are generally low-grade, high-sinuosity streams located within grasslands with higher gradient headwater streams. Weather patterns include cold, wet winters, dry spring and early-summers, and monsoon-related moisture in mid- to late summer (Van Horn *et al.* 2010). The area is also home to wildlife, including elk, and contains designated areas used for cattle grazing. Historically, the land that the VALL now occupies was used for cattle grazing from the 1940s through present, and for sheep grazing, for several centuries before that. In 2009, an Environmental Assessment (EA) Record of Decision set the maximum grazing number of 4000 animal unit months (AUM), which is the amount of forage needed for one animal unit (adult cow and calf) for one month (equals 900 lbs. of dry forage). The EA developed for VALL also limited grazing utilization rates to no more than 40% in all areas (Anderson *et al.* 2010). The former ranch also had logging operations on many of its forested areas in the 20th Century, and allowed geothermal energy exploration in and around the area during the 1970s. These activities led to large areas of montane meadow to form following forest removal. Additionally, VALL has been subject to several severe fires in the recent past including the Cerro Grande fire in 2000 and the Las Conchas fire in 2011 (Dahm *et al.* 2015, Koklay *et al.* 2007). These fires severely impacted both terrestrial and aquatic environments within VALL.

The Jemez and San Antonio watersheds are the two main drainages that occur within VALL (Anderson *et al.* 2010), with the East Fork of the Jemez River (EF) and Rio San Antonio (RSA) being the largest streams that occur within these two drainages, respectively. The EF flow ranges from 0.22 m^3s^{-1} to 0.34 m^3s^{-1}, RSA flows range from 0.16 m^3s^{-1} to 0.48 m^3s^{-1}, and both EF and RSA are groundwater-fed third-order streams (Anderson *et al.* 2010). Jaramillo Creek (JC) is a first/second order spring-fed tributary to the EF with headwaters located within central VALL, and flows ranging from 0.02 m^3s^{-1} to 0.11 m^3s^{-1} (Anderson *et al.* 2010). The RSA lies in the northern portion of the VALL, EF located within the southeastern and southern portions of VALL, and JC extends east then south from central VALL (Figure 1).

In 2004, cattle and elk exclosures were installed for research purposes along on the RSA, JC, and EF (Van Horn *et al.* 2012). The exclosure fence gates were closed June 1, 2004. Each exclosure cluster is rectangular, 160 meters by 320 meters and encompasses about 300m of stream length (Van Horn *et al.* 2012). There are two types, of exclosures present at each exclosure cluster. The first exclosure type is a fenced area where elk and cattle are both excluded by a cyclone fence roughly 3 meters tall. The second type is a fenced area with a wire and post fence roughly 1.5 meters tall, which excludes cattle, but not elk. In total, there are three different exclosure locations on RSA, with one of each type of exclosure at each location. Two exclosure clusters with each type of exclosure are located on JC. EF has one exclosure with both exclosure types near the southern border of VALL. Each exclosure cluster except three are ordered from upstream to downstream: open upstream (no artificial impediments to grazing), elk exclosure, cattle exclosure, open downstream. Two exceptions are the central location on RSA, and the western-most location on RSA. The cattle exclosure precedes the elk exclosure at the western-most location. The ordering at the RSA central location is elk exclosure to cow exclosure; however, the elk and cattle exclosure are disconnected by about 1km. The other exception is the EF cluster, where the cattle exclosure precedes the elk exclosure.

Study design

FIELD METHODS:

In 2006, metal rebar were installed on opposite sides of streambank at sites inside and outside of the grazing exclosures at exclosure cluster locations on RSA, JC, and EF. The purpose of these installments was to document the geomorphology of the streams both temporally and spatially. Sites were established at exclosure clusters location: RSA west (SanA _LO), RSA central (SanA_MID), RSA east (SanA_UP), JC northern (Jaramillo_UP) JC southern (Jaramillo_LO), and East Fork (EF) (Figure 1). Site locations were divided further by exclosure environment type [cattle, elk, and open (control)]. Open environments consisted of areas of unimpeded grazing, cattle environments excluded cattle grazing, but not elk grazing, and elk environments excluded all grazing activity. Twelve rebar installments were placed in each exclosure and open site categorized by habitat type (pool, riffle, run). In total, 215 transects were established VALL-wide (SanA_MID only has 11 rebar installments). From most general location to most specific location, testing sites are divided as: Stream (three total) → exclosure cluster location (six total) → exclosure environment (three total) → habitat type (three total, twelve rebar installments per type).

We analyzed data gathered from these sites from 2006-2010. At each of the rebar installments, a taut string affixed to the bars at a constant height spanned the stream. Geomorphological cross-section data were taken using survey equipment by taking height of a measuring bar from the stream bottom to the string at 10 cm increments laterally along the string. Cross-sections were taken perpendicular to streamflow. Left and right demarcations were noted for bank boundaries, wetted stream edge, and vegetation boundaries.

Additionally, three exclosure clusters were sampled along RSA using survey equipment. At each exclosure cluster, an upstream, downstream, elk exclosure, and cattle exclosure site were sampled. Geomorphological samples were taken 70 stream-meters upstream of all exclosures, 70 stream-meters upstream from the downstream border of each exclosure type, and 70 stream-meters downstream from all exclosure types (Figure 2). Sampling was done August 29-30, 2015 (Appendix A). These geomorphological data were taken to supplement a hydrologic model of RSA.

ANALYSIS METHODS:

Thalweg, stream width, and bank width were calculated for all 215 locations from the 2006-2010 dataset. Entries that were not labeled or that did not allow for complete calculations were not included and no entry was made for that spatial characteristic at that location. Bank width distribution curves for each exclosure environment, and for open areas were also created using ArcGIS and 2010 LiDAR data for microsite data comparison. Using (HEC-RAS), ArcGIS, and the hydrology extension HEC-GeoRAS, a distribution of bank widths from within all six exclosure clusters, as well as open areas on the RSA, JC, and EF, was digitized using a 2010 digital elevation model obtained from LiDAR data. Between 40 and 300 cross-sections were digitized per grazing exclosure and open environment on RSA, JC, and EF. Bank widths were calculated for each digitized cross-section.

We constructed a 16.3-km hydrologic model of San Antonio Creek from a 2010 digital elevation model obtained from LiDAR data. We used HEC-RAS, HEC-GeoRAS, a supplemental application to ArcGIS, and ArcGIS 10. Stream channel, banks, flow lines, and cross-sections were all digitized using HEC-GeoRAS inside ArcGIS. An interpolated cross-section was created every 2 m when needed for channel supplementation along the length on RSA between SanA_UP and SanA_LO exclosure clusters. In order to realistically assess high flows, a peak spring flow of 0.5 m^3s^{-1} was considered the upper end of flows (Anderson *et al.* 2010). LiDAR only obtains points from the water surface, and surrounding terrain, which in turn showed that the streambed on a HEC-RAS model would be the water surface points. We used geomorphological data surveyed from August 2015 to assess stream volume unaccounted for using our LiDAR-created DEM, and adjusted the flow event down 20% to 0.4m^3s^{-1} to compensate for the extra streambed that was not captured. We used an overbank manning's n =0.045, and a channel manning's n = 0.04. We also used a boundary condition of downstream normal flow, with a slope of 0.014 based on Anderson *et al. (*2010). We then ran the 0.4 m^3s^{-1} flow through the model in HEC-RAS and observed the overbanking behavior by importing the two dimensional overbanking data back into ArcGIS from HEC-RAS. We compared the number of overbanking cross-sections in each exclosure environment to number of over-banking cross-sections in open areas on a site-to-site basis.

STATISTICS:

Two-way analysis of variance (ANOVA) and main effects analysis post-hoc test were used to examine differences between bank width to thalweg depth on an inter- and intra-site basis and between year effects. A repeat/first measures ANOVA was also used to examine change in bank width to depth ratios on a yearly basis from 2006-2010. Linear regression was also used to examine the relationship between stream width to bank width. All significance is $\alpha<0.05$ unless noted otherwise.

Results

PHYSICAL STREAM CHARACTERISTICS

For all years measured, bank width was the parameter with the most spatial variation ranging up to meters to 13.7 meters (Table 1). Thalweg also varied widely across the five-year period (Table 1). EF, JC, and RSA bank widths fit within expected bank width values for 2006-2010 as calculated in ArcGIS by channel digitization (Figure 3). Bank width to thalweg depth ratios had normal distributions for all measured years.

Streams had significantly different ($p < 0.05$) bank width to depth ratios during four years (when data were available) when separated by stream on a yearly basis (Table 2), indicating that the EF, JC, and RSA have inherently different geomorphologies. The largest range of bank width to depth medians came in 2008 in exclosed areas, with the smallest amount of variance for a single year occurring in 200, in unexclosed areas (Figure 4). Regression analysis of stream width to bank width of all streams showed a significant ($p < 0.05$) positive relationship for the two parameters in all surveyed years (Figure 5).

Two-way ANOVAs were conducted to examine the effects of general stream regardless of exclosure locations (e.g., RSA, EF, JC) and/or exclosed or unexclosed environments on bank width to thalweg depth ratios (Table 3,4). There was a statistically significant interaction between the effects of stream and exclosure type in 2007 ($p = 0.05$). Simple main effects analysis indicated that bank width to thalweg depth ratios were statistically different in 2007 on EF between exclosed and unexclosed environments ($p = 0.014$).

Two way ANOVAs were conducted to examine site-by-site exclosure locations (e.g. SanA_Lo) and/or exclosed or unexclosed environments effects on bank width to thalweg depth ratios (Table 3,4). There was a statistically significant interaction between these effects in 2008

($p = 0.03$). Simple main effects analysis indicated that bank width to thalweg depth ratios were statistically different at the EF exclosure location between exclosed and unexclosed environments ($p = 0.05$)

Two way ANOVAs were conducted to examine exclosed or unexclosed environments and/or general stream effects on bank width (Table 3, 4). There were statistically significant interactions in 2008 ($p = 0.03$). Simple main effects analysis indicated differences in bank width measurements in 2008 on EF between exclosed and non-exclosed locations ($p = 0.05$).

A repeated measures ANOVA was conducted for all stream locations and environments for 2006-2010 (Table 5). Significant differences ($p < 0.05$) in bank width to depth ratios were shown when comparing multiple years. For example, 2006 bank width to depth ratio was significantly different than 2007 ($p < 0.005$).

A regression analysis was performed for all streams, and each year (2006-2010) showed a significant ($p < 0.005$) positive relationship for stream width to bank width (Figure 5).

Average bank movement was calculated for each transect (Figure 6). The East Fork cattle exclosure saw the most total movement with a movement of 64.58 cm the left bank alone. Data was not uniform across sites. A “start” value and a “finish” value was assigned for the first time bank measurements were observed and conversely on the last measurements observed. For example, some initial measurements could be in 2006 for a given transect, and final measurements taken in 2010. However, another transect could have the first measurement in 2006, and final in 2007, depending on available data. Only first and last data were taken into account as an initial and final position. Absolute value of total movement from 2006-2010 is not included.

HYDROLOGIC MODEL

The overbanking behavior of cross sections in our hydrologic model of RSA followed an elevational and upstream-to-downstream progressional pattern. At the eastern-most location, 100% of cross-sections showed overbanking. Moving downstream about five kilometers to the middle exclosure cluster, the percentage of overbanking cross-sections to non-overbanking cross-sections decreased to 65%. Finally, at the western-most exclosure, about 10 kilometers from the middle exclosure, overbanking to non-overbanking cross-sections decreased to around 50%. Non-exclosed cross-sections also followed the same pattern, decreasing in percentage of overbanking cross-sections to non-overbanking ones moving from upstream to downstream on the RSA. Cross-sectional views in both HEC-RAS and HEC-GeoRAS provide visual interpretation of overbanking on RSA (Figure 7).

Discussion

Grazing in and around streams causes geomorphological changes in streams (Timble *et al.* 1995). This study set out to investigate the effects of animal exclosures on stream geomorphology. Important to this study when comparing EF, JC, and RSA to one another was determining if they had pre-existing differences in geomorphologies. A way to manage these inherent differences in geomorphologies was to take the ratio of bank width to thalweg depth in order to normalize these streams to one another, and eliminate confounding effects like stream order. This allowed insight into the bank width to depth ratio of these streams, which is commonly observed when measuring the effects of grazing on stream geomorphology (Augustine *et al.* 1998).

In areas of heavy grazing, bank trampling will lead to streambank widening, while stream width may decrease or stay the same (Belsky *et al.* 1999). Since flow data are considered constant for each collection year, a wider bank with a narrower stream would be an indicator of bank degradation by grazing activity. A regression slope much greater than one would be a signal that grazer-induced impacts to stream geomorphology were possible. Our calculated regression slopes ranged from 0.55 to 1.07, indicating that as bank width increased, stream width did as well, but not in the proportions that would indicate grazer influence. This is evidence that factors other than grazers are influencing stream geomorphology in VALL.

Bank width to thalweg depth and bank width alone had significant differences between exclosed and unexclosed environments at the EF site ($p < 0.05$) in 2007 and 2008. Significant interactions occurred only at EF, and only in 2007 and 2008. It is likely that these significant changes are a result of factors other than biotic disturbance. More significant interactions seen on a particular stream or significant interactions in every year may have been a stronger indicator for grazer-driven disturbance.

Abiotic factors, like stream order, maximum peak flow events, elevation changes, stream ontogeny, velocity changes, and geological changes, are more likely driving stream geomorphology in VALL, as these typically have a large effect on stream physical characteristics (Brierley and Fryirs 2000). Additionally, in years since the data for this study was collected, disturbance like forest fire has had a major impact in VALL (Dahm *et al.* 2015). Large scale disturbances like forest fire have catchment-wide effects, and could mask or eliminate entirely any long-term evidence of grazer-caused geomorphological changes. High-energy floods coming through narrow streams can cause major geomorphological changes as well. These combined effects may make VALL a site where abiotic factors dominate biotic factors in geomorphological alterations. When normalized with a ratio like width to depth, it is still possible to see that exclosure sites still are different from one another, as they should be (Figure 6). Different stages in stream succession and stream order inherently lead to different geomorphologies (Vannote *et al.* 1980, Stanford *et al.* 2005). However, these exclosure sites are not different enough temporally or spatially to indicate that biotic disturbance is a driving geomorphological influence.

Bank movement follows a narrowing pattern over time with varying degrees of movement (Figure 6). However, there are still areas where the channel appears to shift in one direction or another. Additionally, in another instance at RSA Lo exclosure, distinct widening occurs in the ungulate exclosure. This suggests that excluding ungulates from grazing areas may be having an effect on bank widths by narrowing them, however, since change is seen in both exclosed and unexclosed environments, it’s more likely that the inconsistencies seen throughout the entire VALL are a result of abiotic disturbances. These data would conclude that natural disturbance processes are responsible for the bank changes seen.

The significant ($p < 0.05$) differences in width to depth ratios in the repeated measures ANOVA between years, like 2006 and 2007, show that the entire VALL is dynamic in nature. It allows insight, along with bank movement, into how the streams themselves are changing on a yearly basis. These data give further evidence that the VALL is a system driven by abiotic disturbance forces. Geomorphological changes both inside and outside of exclosure environments of a similar scale indicate again that biotic disturbance via ungulate grazing is not a significant driving force in VALL.

Data for this study were gathered two to six years after the establishment of the exclosures, which may not be enough time to observe exclosure-related geomorphology changes. With exclosures typically result in decreased width in streams, as well as increased depth and decreased width to depth ratios, recovery effects may not be seen for as long as 4-14 years (Ranganath *et al.* 2009). Also, grazing pressure is generally considered “light” (<40% forage utilization) and not uniform throughout VALL (Anderson *et al.* 2010). Some streams may be recovering at different rates than others as they may have been differentially affected by grazing. More heavily grazed areas are subject to quicker, and more noticeable recovery (Su *et al.* 2005).

Within our hydrologic model, we saw a high percentage of cross-sections overbanking versus not over-banking both inside and outside of exclosures in the higher elevation areas. Then, in downstream reaches, overbanking decreased at a similar rate in both open cross-sections and exclosed cross-sections in excluded areas (Figures 7,8). This may indicate a follow in the natural progression of a stream to widen as tributaries increasingly are incorporated and it moves further from its headwaters (Vannote *et al.* 1980). In our model, overbanking does not always occur uniformly on both sides of the stream. Cattle grazing on one side of RSA preferentially over another could have differential effects due to decreased plant cover on different reaches of RSA if overbanking occurs in a grazed area.

Removing ungulates from stream areas can have several beneficial results for overall stream health and can result in passive, or “hands-off”, restoration of the stream. Ungulate exclusion can result in a decrease of total in-stream nitrogen concentration from decreased animal waste input and can also decrease animal-borne disease vectors for in-stream wildlife (Hansen *et al.* 2013). Native plant species, including woody and herbaceous plants, can replace grazing-tolerant plant species. Over time this change in riparian plant species composition can help restore natural bank structure to the stream (Hough-Snee *et al.* 2013). Additionally, riparian biomass increases, and stream litter inputs via riparian sources increase in areas of animal exclosure (Van Horn *et al.* 2012). Fish populations can also recover as a result of increased bank stability, and native bird populations have seen increases following the removal of grazers from stream areas (Batchelor *et al.* 2015).

Applying passive restoration techniques to an area of formerly intensive grazing provides both in-stream and riparian benefits. Active restoration such as channel stabilization or planting native vegetation may still be necessary in some cases to recover from heavy grazing activity (Booth *et al.* 2012). However, benefits like the return of native in-stream species, riparian plant species, bird assemblages, and fish species have been seen from passive restoration efforts (Hough-Snee *et al.* 2013, Batchelor *et al.* 2015). When utilized even partially, passive restoration provides environmental benefits to the catchment. In the long-term, passive restoration may provide a stream with an opportunity for “self-correction” that active restoration may not offer. Examining the effects that passive stream restoration has in areas of heavy grazing may help to mitigate future costs and hardships by providing insight into when passive restoration is more practical or more economically viable than active restoration (Beschta *et al.* 2013).

One way to promote passive restoration is to exclude ungulates from areas of a stream that are targeted for restoration. Animal exclosure fences are one method of preventing ungulates from grazing in and around a stream and allowing passive restoration to proceed (Su *et al* 2005). Animal exclosure fences are a relatively inexpensive, easily removable, and can be made to be portable. They require routine maintenance depending on the level of livestock/ungulate activity, climatic factors, or disturbance events. An effective way to prevent native and domestic grazers from accessing a stream area for short- or long-term timeframes is fenced exclosures placed around areas of heavy grazing. (Su *et al.* 2005, Augustine *et al.* 1998). Additionally, stream and riparian monitoring prior to and following the installation of animal exclosures will help provide insight into long-term biotic and abiotic disturbance response, recovery, and exclosure effectiveness. Monitoring metrics may include soil compaction, riparian soil infiltration rates, carbon, nitrogen and phosphorus spiraling rates in the water column, water quality measurements like pH, alkalinity, and turbidity, in-stream and riparian biota sampling, stream physical characteristic surveying, and stream overbanking behavior.

Climate change introduces a new element into passive stream restoration by animal exclusion. Warming temperatures and a decrease in precipitation in the southwestern United States is expected over the next decades (Seager *et al.* 2007). Globally, grazing on managed lands has increased 600% in the last three-hundred years (Asner *et al.* 2004). Overgrazing in an area leads to decreased plant cover, increased susceptibility to wind erosion, and loss of soil nutrients (Su *et al.* 2005). Nearly 80% of streams and riparian areas in arid regions of the United States have already been damaged by livestock grazing (Belsky *et al.* 1999). An increase in the amount of grazing on managed lands combined with decreasing grazing habitat and increasing temperatures may cause more streams and riparian areas to become damaged, or exacerbate current damage. Stressors to stream ecosystems from a warming climate also include reduced in-stream flow as a result of decreased precipitation and increased water temperatures, which is already a major water quality concern (Beschta *et al.* 2013). These stressors are likely to worsen in the future, and active restoration may become more difficult and costly. The need for monitoring and assessing the effectiveness of passive restoration becomes an important task in order to provide informed management decisions to land managers in the future.

Implications

Although the conclusion of this project is that in this particular site, there do not appear to be significant effects to stream geomorphology due to biotic disturbance that does not discount the importance to consider passive restoration by animal exclusion as a viable restoration option. Indeed, VALL's collaborating organizations that are re-planting woody plants (e.g., willows) along streambanks always erect elk exclosure fences to protect the saplings from elk browsing (R. Parmenter, personal communication). This project was designed to lay out the benefits of passive restoration, along with providing a method of testing geomorphology which is repeatable on a spatial and temporal aspect. Streams are multifaceted and must be measured in a consistent and scientific manner in order to accurately diagnose irregularities in order to provide informed opinions on stream management. Monitoring a stream's biological, chemical, and physical parameters following the installation of an exclosure fence may provide valuable insight into cost-effectiveness of passive restoration verses active restoration. This study indicates that simply putting up an exclosure fence and not allowing grazing in and around a stream is not the only factor when considering passive restoration. Some factors of the exclosure itself must be taken into consideration as well: 1) Pre-treatment measurements will allow a baseline which all future measurements can be measured against; 2) The timing of the exclosure being established on the stream as well as timing of measurements is important to consider to obtain meaningful results; 3) The size of the exclosure can determine how much effect the exclosure has, be it reach-scale or catchment-scale and; 4) The placement of the exclosure must be carefully considered to maximize the desired effects (Sarr *et al.* 2002). In addition, all of a streams attributes, including ontogeny, physical, chemical, and biological factors must be examined and carefully considered before installing an exclosure. Upstream and downstream stream characteristics must be taken into consideration as well. Hard science must be considered carefully before installing an exclosure, rather than just having the desire to "fix" a catchment or reach.

In the VALL case, experimental exclosures were established over a wide range of stream channel sizes, across the entire preserve, to represent the variability of VALL stream systems. Measurements of riparian vegetation, stream water quality, and fish and aquatic invertebrate populations were begun immediately with exclosure construction, and streambank geomorphology measurements were started within two years; these exclosures should provide useful long-term data on stream ecosystem responses to ungulate grazing. Other exclosures constructed for woody riparian vegetation planting, with ongoing monitoring programs, will expand and compliment these data sets. Continued monitoring of biotic and abiotic variables should indicate the degree and rates of change in VALL streams with and without ungulate grazing activities.

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FIGURES:

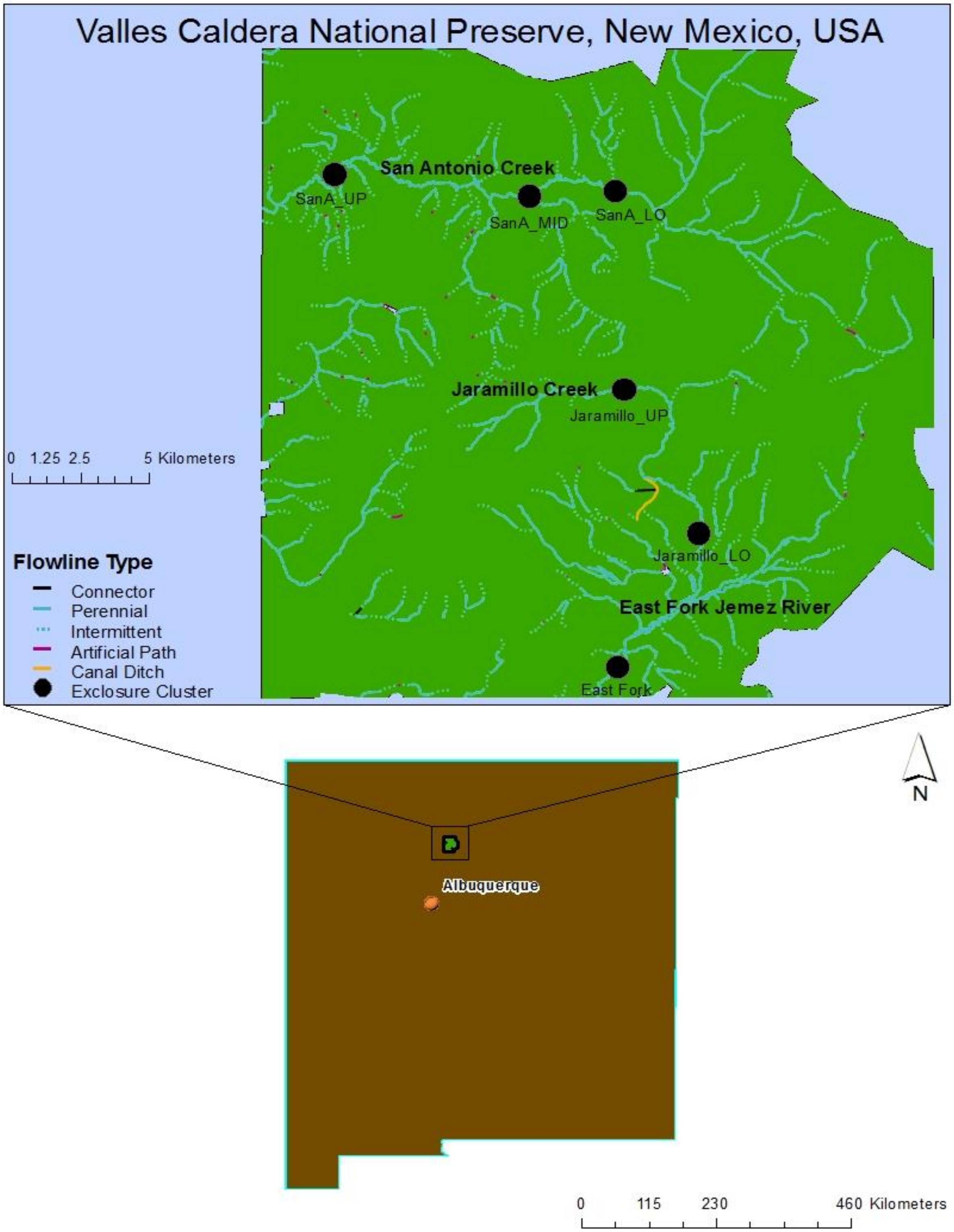


Figure 1. The Valles Caldera National Preserve, located in northern New Mexico.

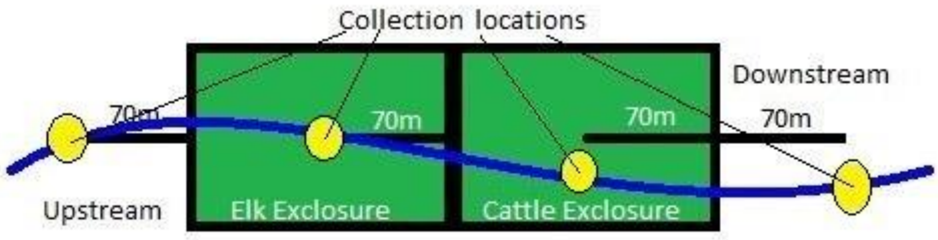


Figure 2. Diagram of collection locations for each research exclosure site.

Table 1. Descriptive statistics of bank width and thalweg depth for all streams, 2006-2010.

Descriptive Statistics

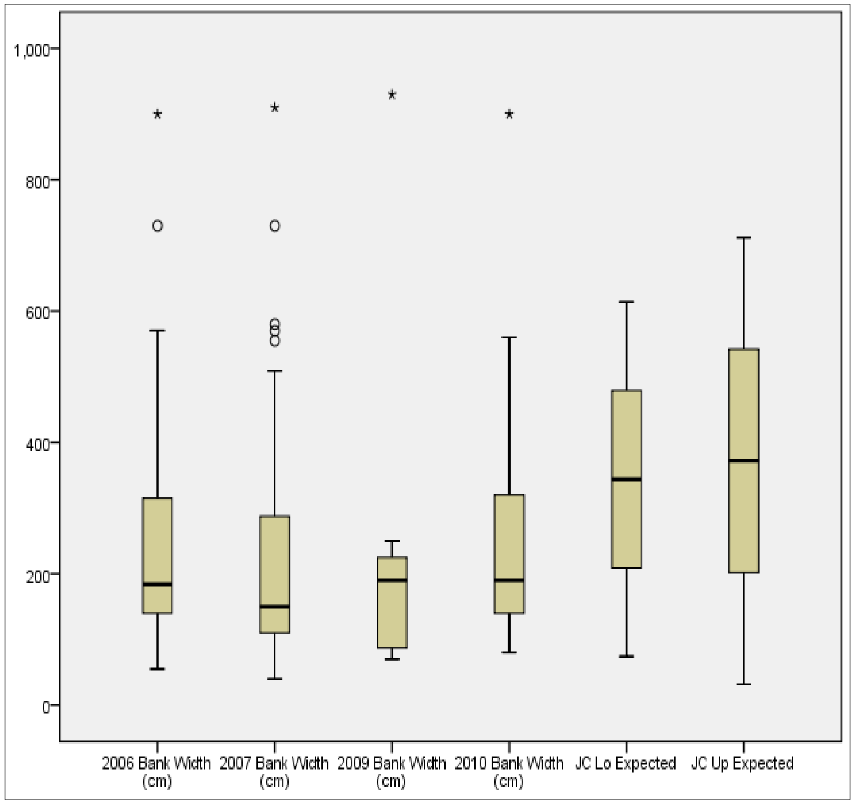
	N	Minimum	Maximum	Mean	Std. Deviation
Bank Width (cm)					
2006	104	55.0	1380.0	434.433	240.9162
2007	204	40.0	1410.0	386.250	244.7935
2008	70	177.0	1380.0	506.186	268.4861
2009	80	70.0	1300.0	488.275	264.3170
2010	106	80.0	1360.0	386.934	265.6551
Thalweg Depth (cm)					
2006	191	42.0	915.0	103.071	90.0955
2007	213	0.0	935.0	92.030	62.9669
2008	142	43.5	985.0	116.067	109.2052
2009	215	33.0	891.0	111.481	131.1288
2010	215	0.0	162.9	86.280	23.1246

Table 2. One-way ANOVA results comparing bank width to depth ratio 2006-2010. Bank width to thalweg depth ratios were compared for all locations per stream. Statistically significant differences ($p < 0.05$) indicate a difference between bank width to thalweg depth ratios between the two streams listed.

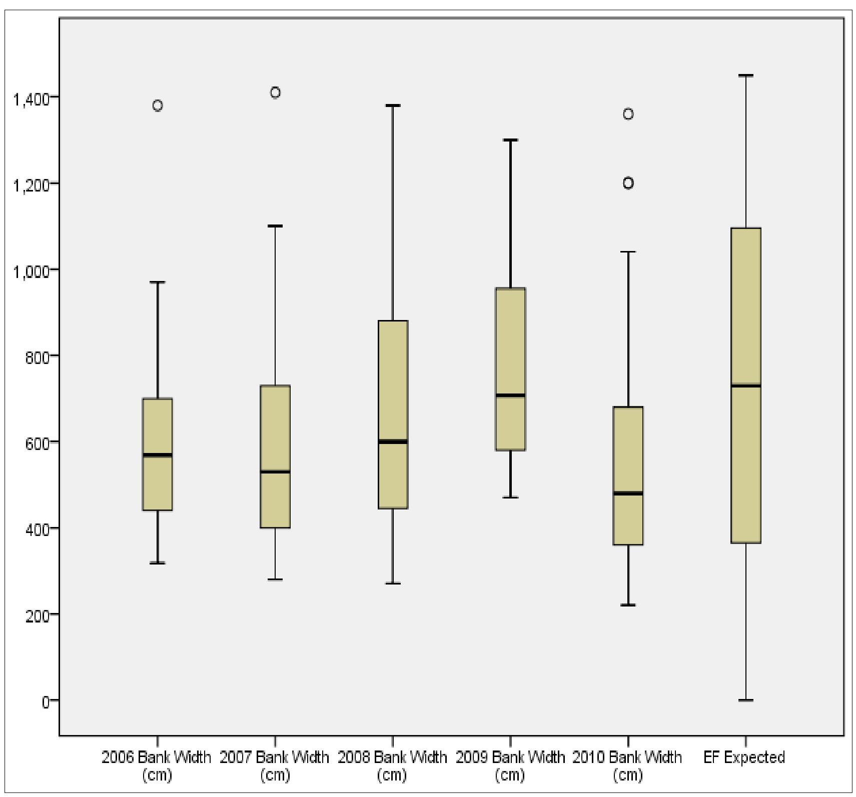
One-Way ANOVA

Width To Depth			Sig.	95% Confidence Interval Lower Bound	95% Confidence Interval Upper Bound
2006 Width to Depth Ratio	East Fork	Jaramillo	.000*	1.3668	3.1723
		San Antonio	.013*	.1803	1.8573
	Jaramillo	East Fork	.000*	-3.1723	-1.3668
		San Antonio	.000*	-1.9315	-.5700
2007 Width to Depth Ratio	East Fork	Jaramillo	.000*	1.9087	3.3211
		San Antonio	.000*	.5494	1.8953
	Jaramillo	East Fork	.000*	-3.3211	-1.9087
		San Antonio	.000*	-1.9127	-.8723
2009 Width to Depth Ratio	East Fork	Jaramillo	.000*	2.3399	8.4869
		San Antonio	.005*	.8573	5.7218
	Jaramillo	East Fork	.000*	-8.4869	-2.3399
		San Antonio	.045*	-4.2074	-.0403
2010 Width to Depth Ratio	East Fork	Jaramillo	.000*	1.5371	3.9316
		San Antonio	.000*	.8451	3.2563
	Jaramillo	East Fork	.000*	-3.9316	-1.5371
		San Antonio	.367	-1.8809	.5135

*. The mean difference is significant at the 0.05 level.


1,000
800
600
400
200
0
2006 Bank Width (cm)
2007 Bank Width (cm)
2009 Bank Width (cm)
2010 Bank Width (cm)
JC Lo Expected
JC Up Expected

a.


1,400
1,200
1,000
800
600
400
200
0
2006 Bank Width (cm)
2007 Bank Width (cm)
2008 Bank Width (cm)
2009 Bank Width (cm)
2010 Bank Width (cm)
EF Expected

b.

c.

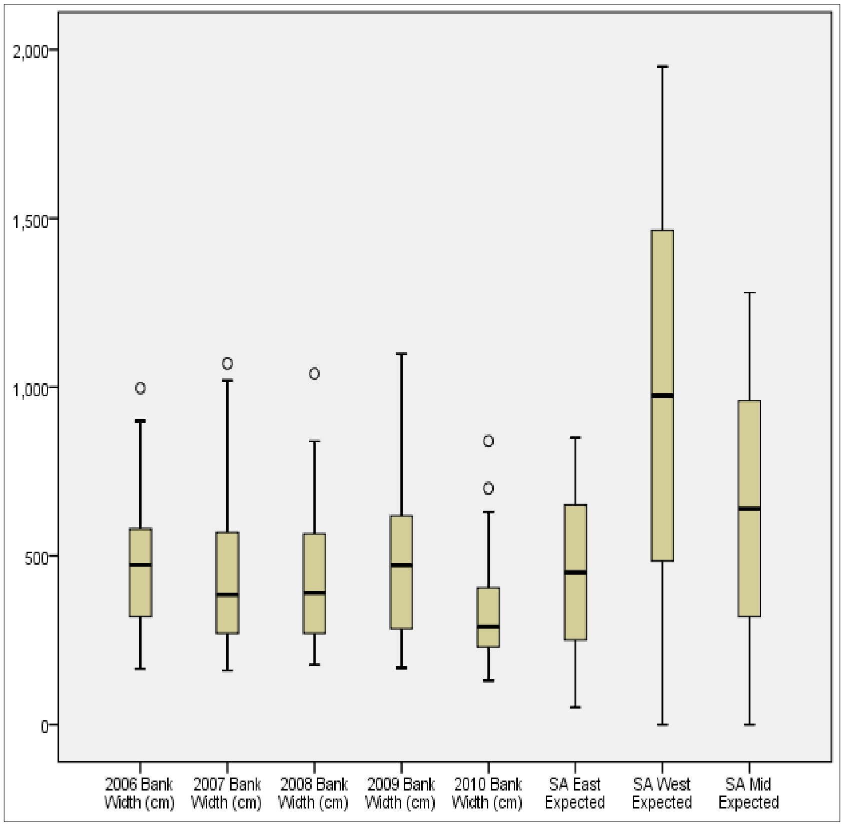


Figure 3. Measured bank widths by year (2006-2010) and predicted bank widths by exclosure location as calculated in ArcGIS. a. Jaramillo Creek b. East Fork Jemez c. Rio San Antonio. No significant differences between any groups. Asterisks and circles denote outliers.

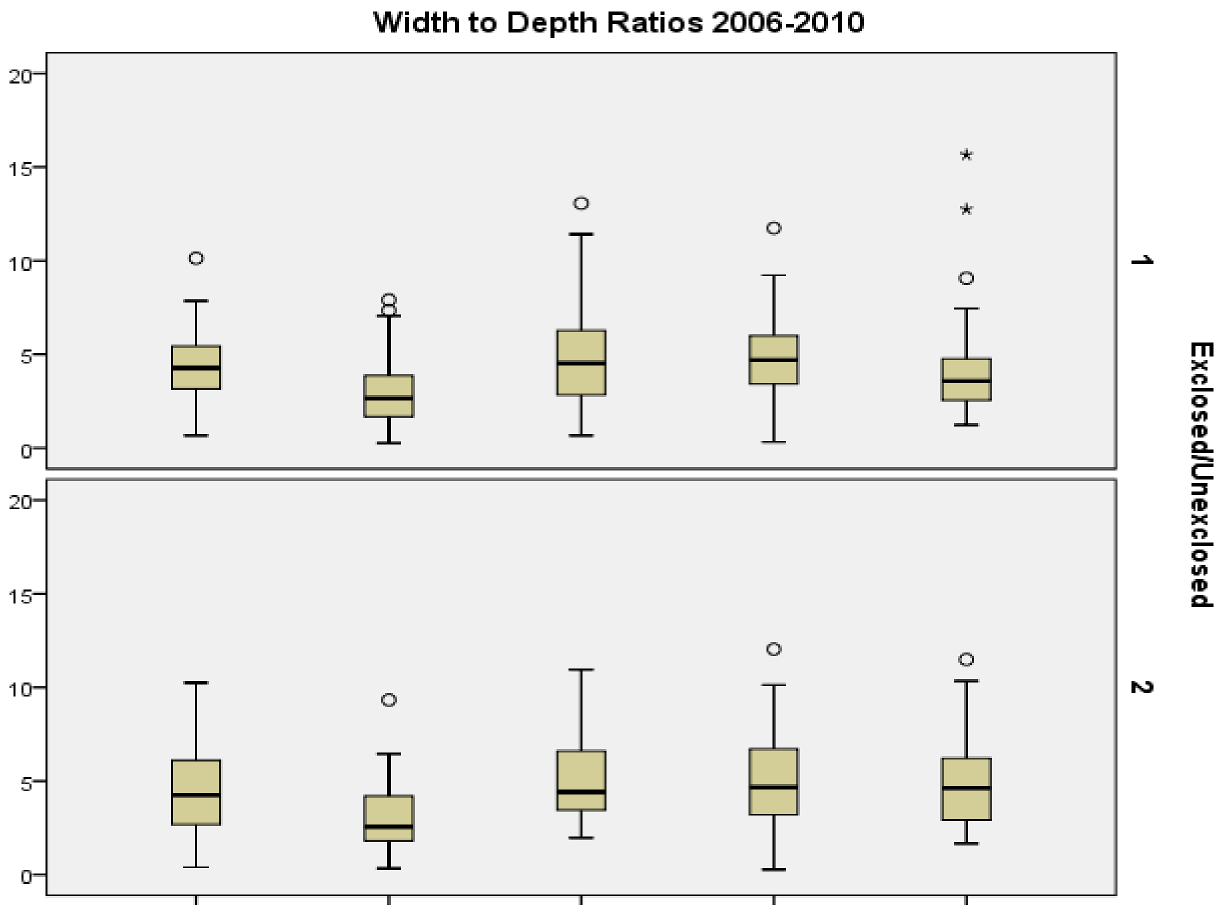
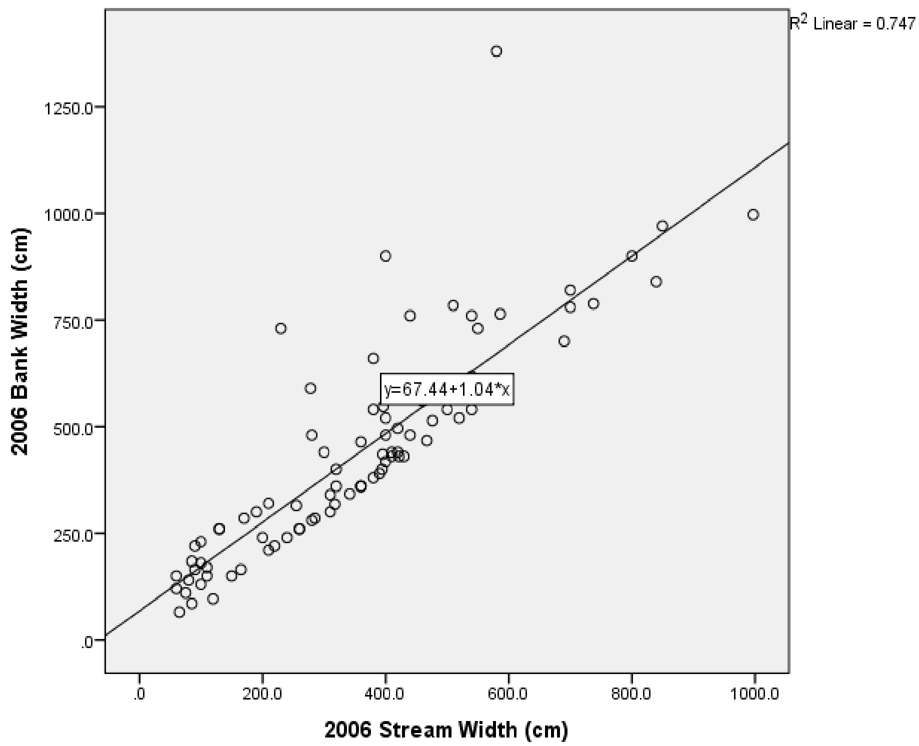
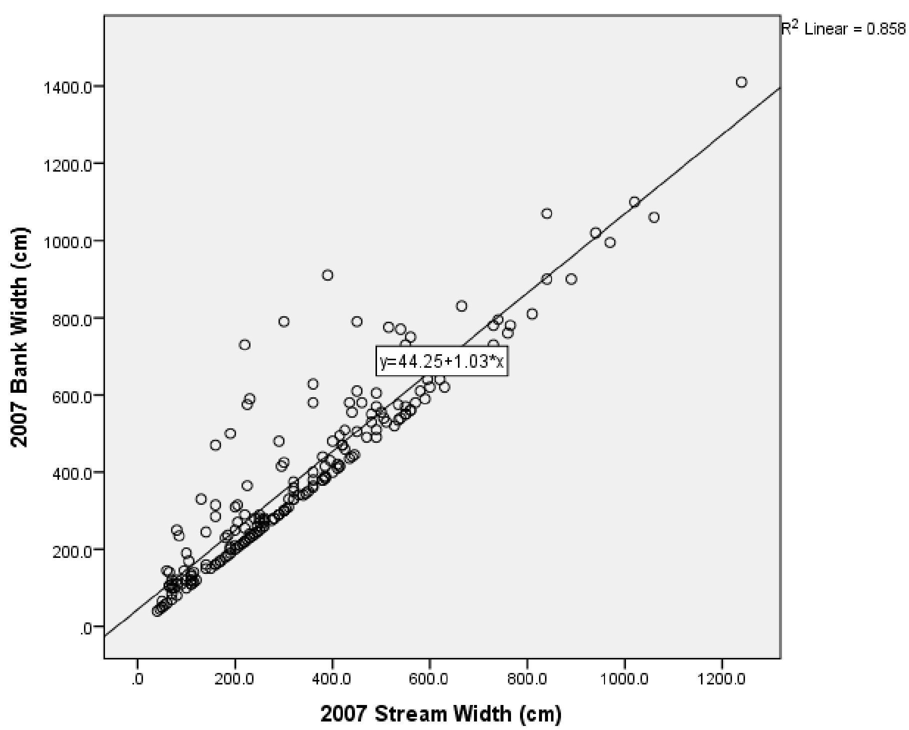
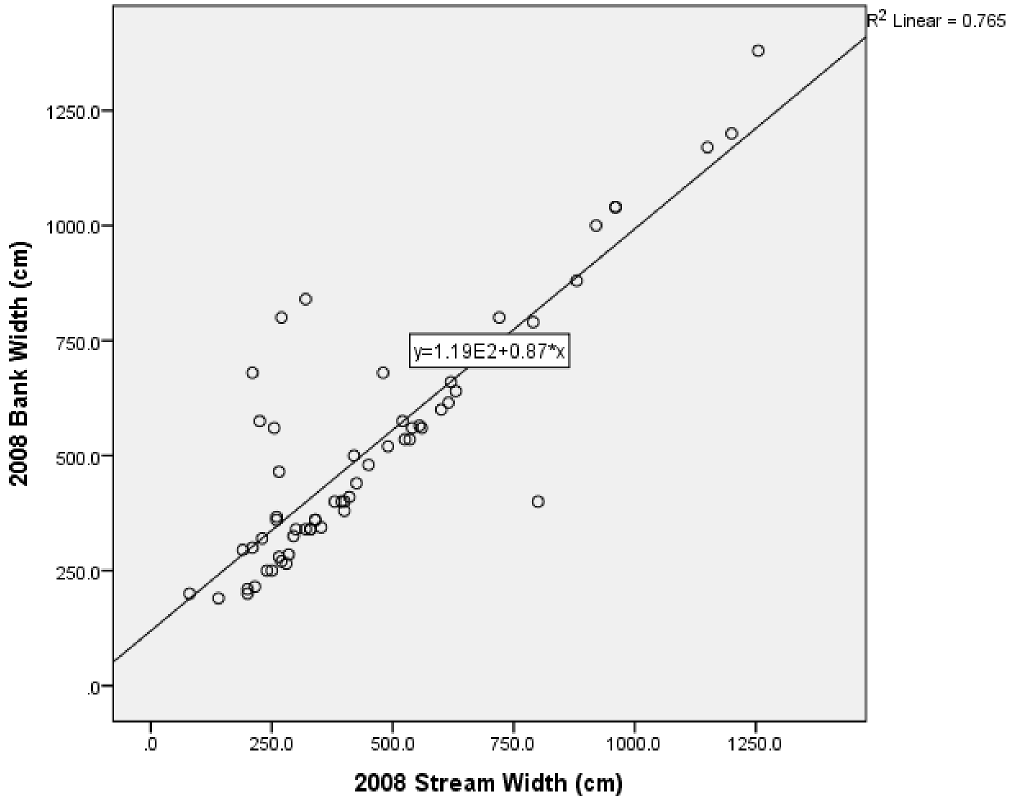


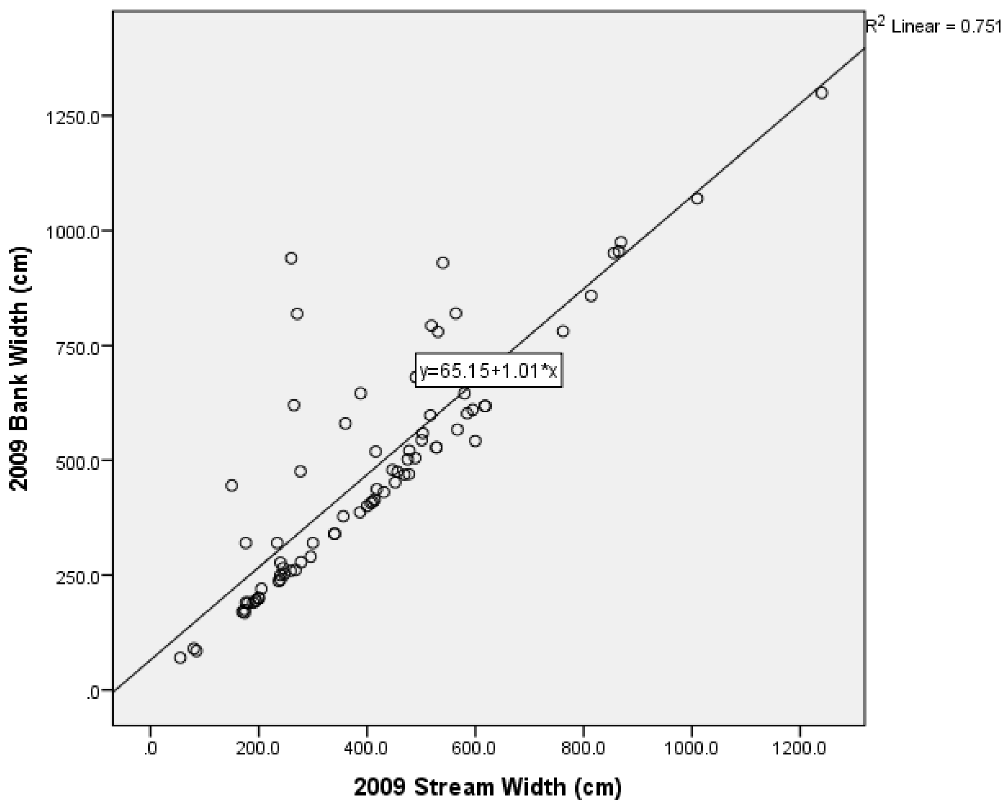
Figure 4. Bank width to depth ratios for unexclosed and exclosed locations for all streams, 2006-2010. No significant differences between groups. 1. Exclosed 2. Unexclosed. No significant differences between groups.


R2 Linear = 0.747
y=67.44+1.04*x
2006 Bank Width (cm)
2006 Stream Width (cm)
1250.0
1000.0
750.0
500.0
250.0
.0
200.0
400.0
600.0
800.0
1000.0


R2 Linear = 0.858
y=44.25+1.03*x
2007 Bank Width (cm)
2007 Stream Width (cm)
1400.0
1200.0
1000.0
800.0
600.0
400.0
200.0
.0
1200.0



R2 Linear = 0.765
y=1.19E2+0.87*x
2008 Bank Width (cm)
2008 Stream Width (cm)
.0
250.0
500.0
750.0
1000.0
1250.0



R2 Linear = 0.751
y=65.15+1.01*x
2009 Bank Width (cm)
2009 Stream Width (cm)
.0
250.0
500.0
750.0
1000.0
1250.0
200.0
400.0
600.0
800.0
1200.0

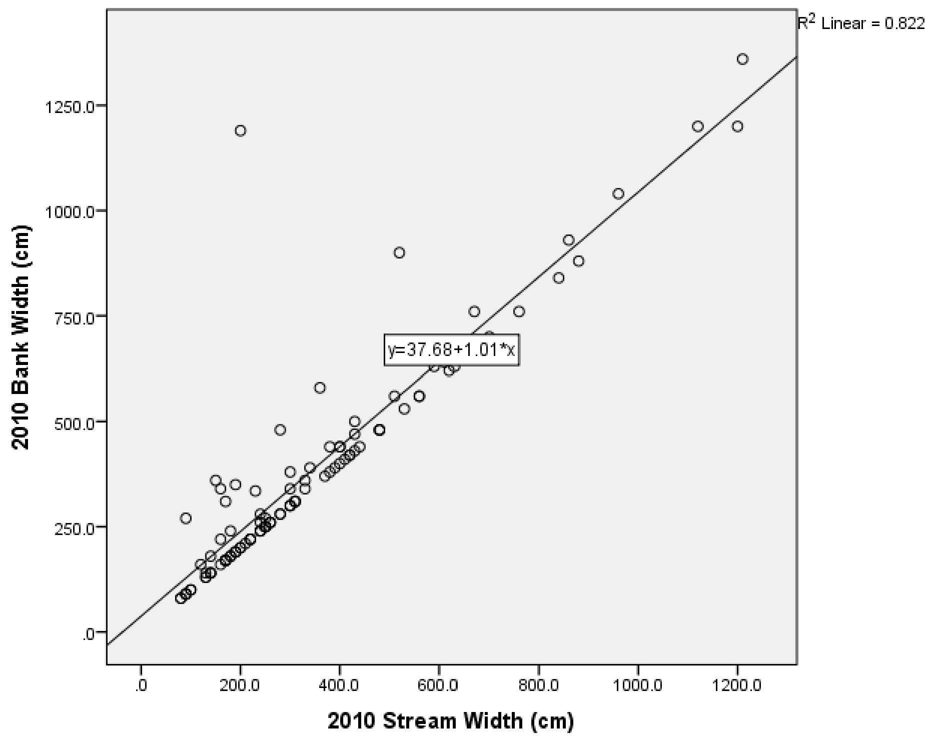


Figure 5. Regression analysis of stream width to bank width for 2006-2010. All regression relationships shown are significant ($p < 0.005$).

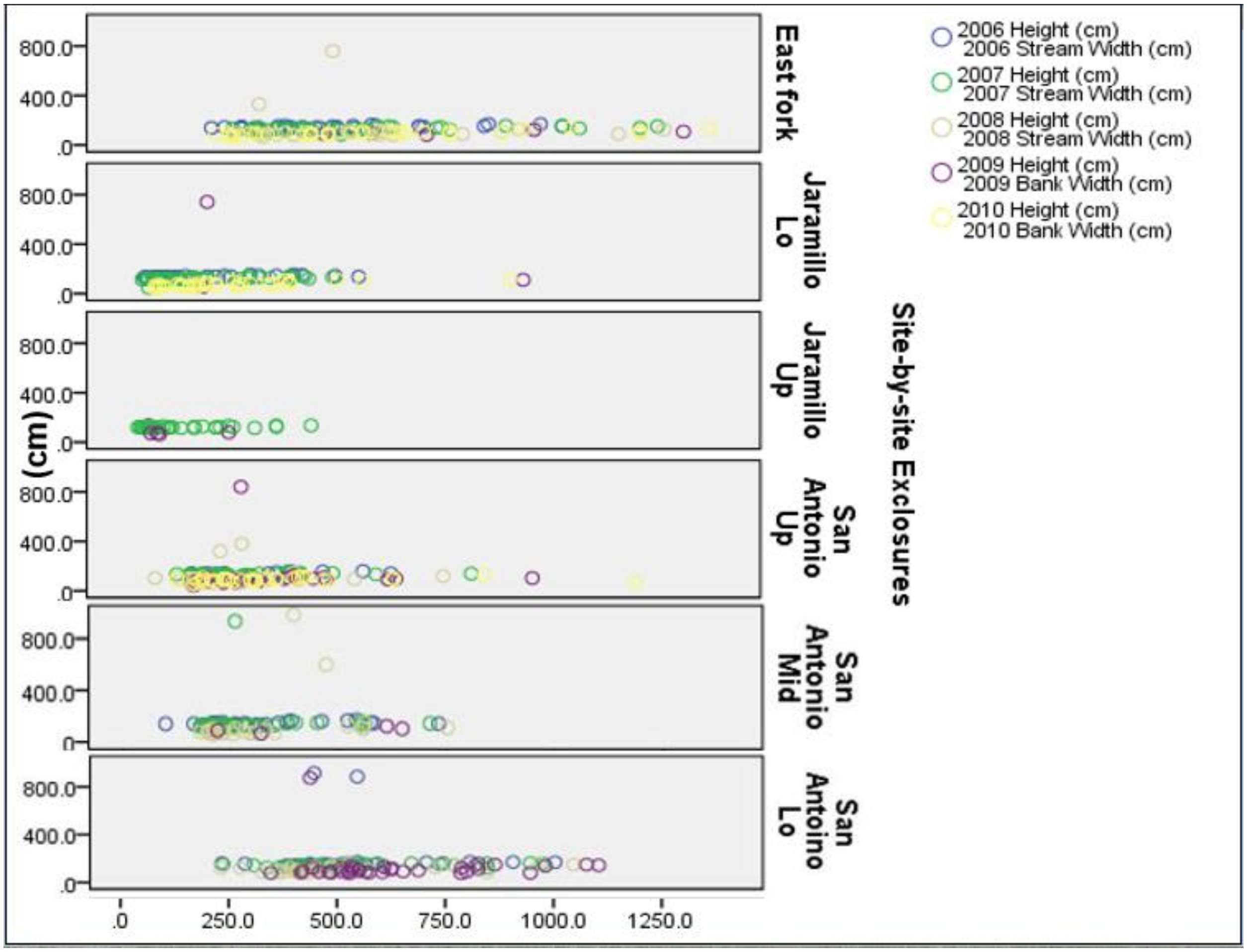


Figure 5. Bank width to depth ratios by exclosure locaiton 2006-2010. There are no significant differences between exclosure locations.

Table 3. Results of two-way ANOVAs with bank width to thalweg depth. Significant values ($p < 0.05$) are listed for two-way ANOVA, and for simple main analysis post-hoc test, which identifies where significant interactions occur.

Two-way ANOVA Results

Bank Width to Depth Ratio	Main Effects Analysis		ANOVA sig.	Main Effects sig.
2007	East Fork	Gen-Stream Ex/Non-Ex	0.05	0.014
	East Fork	Stream Ex/Non-Ex	0.03	0.005

Two-way ANOVA Results

Bank Width	Main Effects Analysis		ANOVA sig.	Main Effects sig.
2008	East Fork	Gen-Stream Ex/Non-Ex	0.05	0.03

Table 4. Values from two-way ANOVAs that produced significant results seen in Table 3.

2007 Width to Depth Ratio

Stream			Mean Difference	Std. Error	Sig.	95% Confidence Interval for Difference	
						Lower Bound	Upper Bound
East Fork	Exclosed	Unexclosed	-1.323*	.533	.014	-2.374	-.272
Jaramillo	Exclosed	Unexclosed	-.120	.353	.734	-.816	.576
San Antonio	Exclosed	Unexclosed	.182	.297	.541	-.404	.768

2007 Width to Depth Ratio

Site-by-site Exclosures			Mean Difference	Std. Error	Sig.	95% Confidence Interval for Difference	
						Lower Bound	Upper Bound
East Fork	Unexclosed	Exclosed	-1.323*	.466	.005	-2.243	-.403
Jaramillo Lo	Unexclosed	Exclosed	-.653	.435	.135	-1.512	.205
Jaramillo Up	Unexclosed	Exclosed	.407	.439	.355	-.458	1.272
San Antonio Up	Unexclosed	Exclosed	.142	.442	.748	-.730	1.014
San Antonio Lo	Unexclosed	Exclosed	.560	.439	.203	-.305	1.425
San Antonio Mid	Unexclosed	Exclosed	-.163	.473	.731	-1.096	.770

2008 Bank Width (cm)

Stream			Mean Difference	Std. Error	Sig.	95% Confidence Interval for Difference	
						Lower Bound	Upper Bound
East Fork	Exclosed	Unexclosed	-256.500*	115.583	.030	-487.268	-25.732
San Antonio	Exclosed	Unexclosed	36.286	72.893	.620	-109.249	181.821

Table 5. Results and values calculated from a repeated measures ANOVA for 2006-2010. Significant values ($p < 0.05$) indicate a between-year difference in bank width to thalweg depth ratios.

Repeated Measures ANOVA

Measure WD:

Year		Mean Difference	Std. Error	Sig.	95% Confidence Interval for Difference	
					Lower Bound	Upper Bound
2006	2007	.878*	.042	.000	.679	1.077
	2008	-.245	.224	1.000	-1.317	.826
	2009	-.414	.170	.588	-1.226	.397
	2010	-.239	.115	.925	-.790	.311
2007	2006	-.878*	.042	.000	-1.077	-.679
	2008	-1.123*	.208	.029	-2.115	-.131
	2009	-1.292*	.173	.007	-2.116	-.468
	2010	-1.117*	.141	.005	-1.790	-.444
2008	2006	.245	.224	1.000	-.826	1.317
	2007	1.123*	.208	.029	.131	2.115
	2009	-.169	.103	1.000	-.660	.322
	2010	.006	.220	1.000	-1.042	1.054
2009	2006	.414	.170	.588	-.397	1.226
	2007	1.292*	.173	.007	.468	2.116
	2008	.169	.103	1.000	-.322	.660
	2010	.175	.147	1.000	-.526	.876
2010	2006	.239	.115	.925	-.311	.790
	2007	1.117*	.141	.005	.444	1.790
	2008	-.006	.220	1.000	-1.054	1.042
	2009	-.175	.147	1.000	-.876	.526

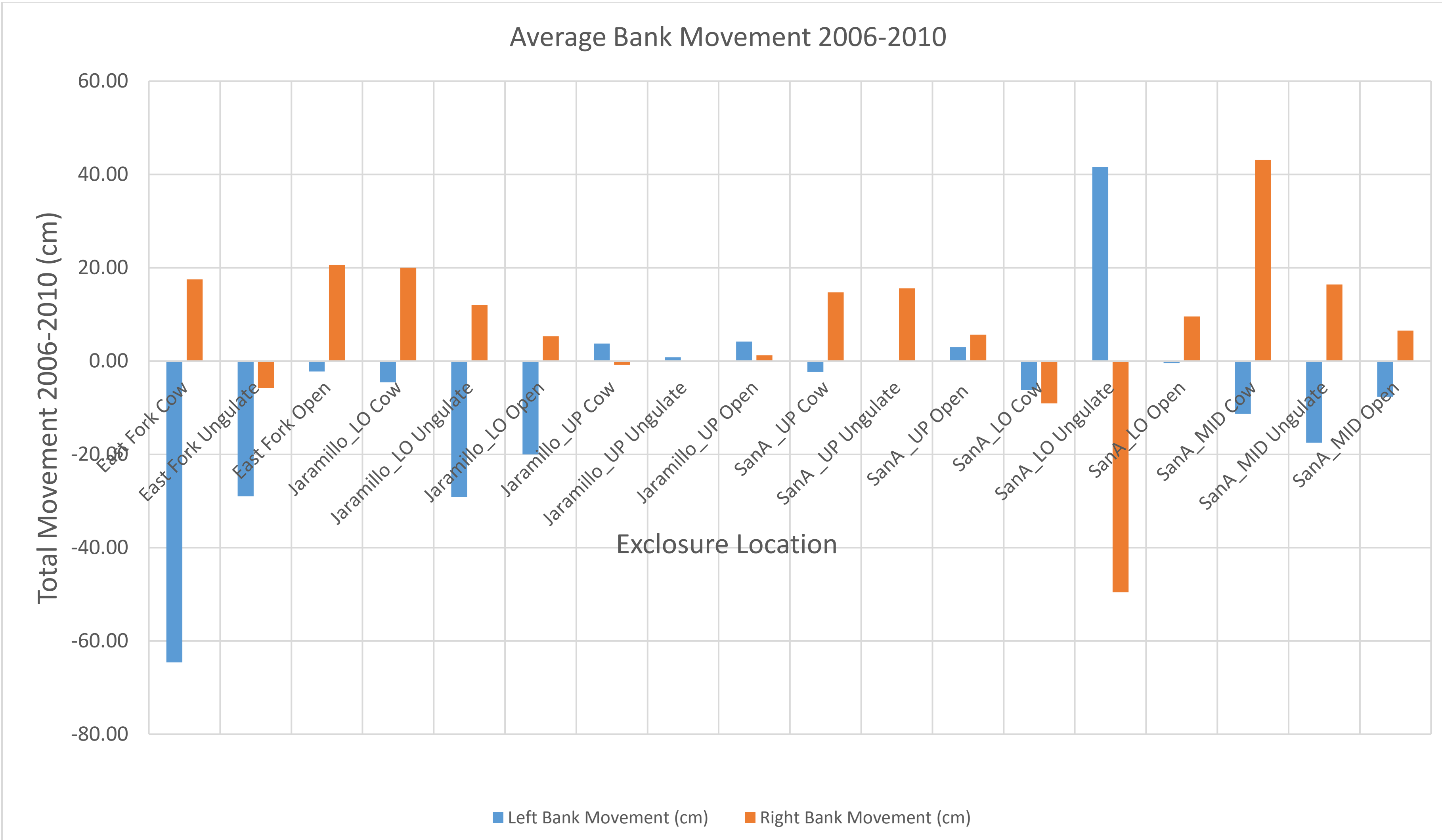
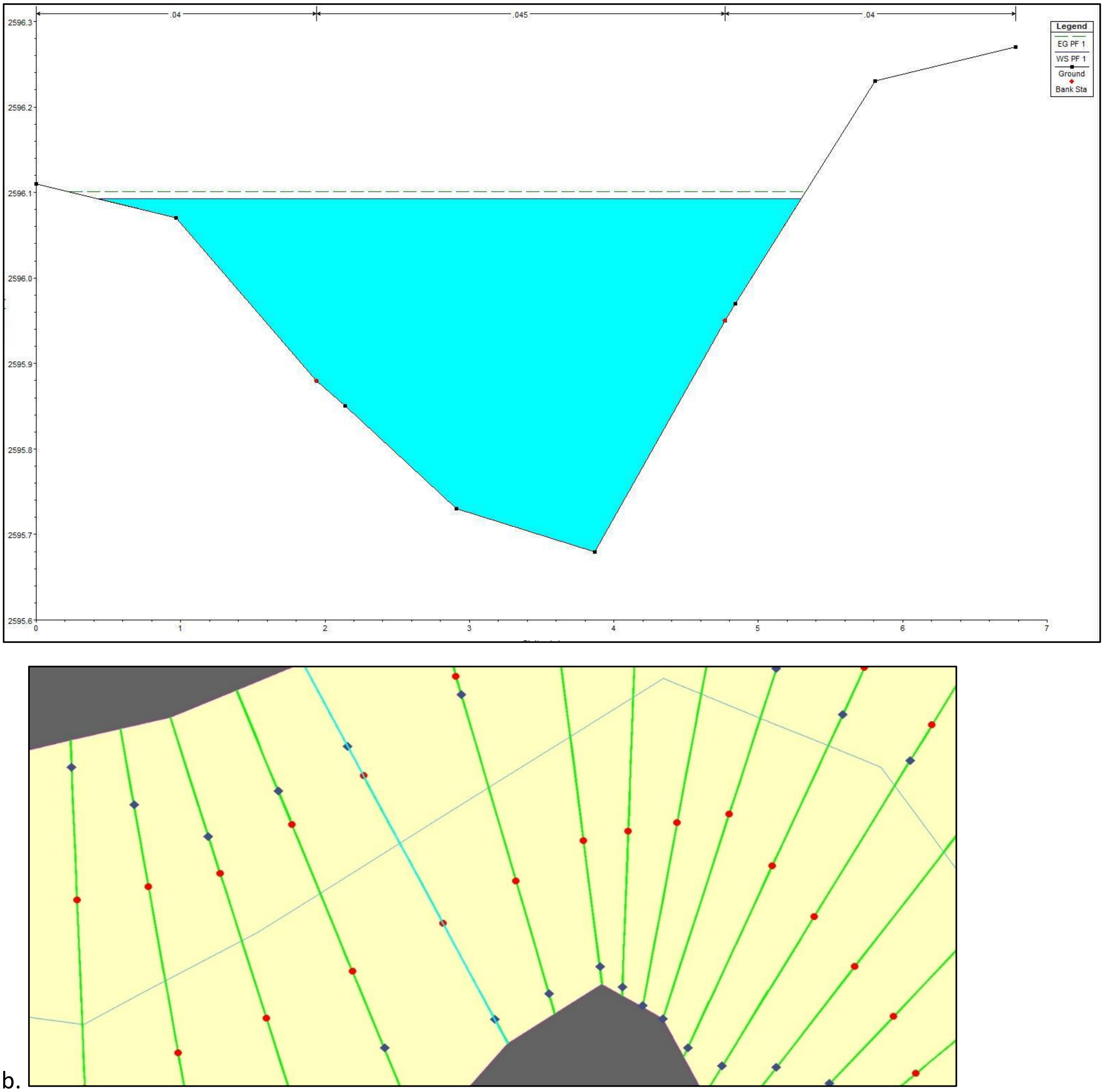


Figure 6. Averaged left and right bank movements through time (2006-2010). Negative values indicate a bank movement to the right, away from the rebar post where measurement starts, positive values indicate a movement to the left, away from the rebar where measurement starts.

a.



b.

Figure 7. A depiction of the same cross-section (light blue) in both HEC-RAS (a) and HEC-GeoRAS (b) with 0.4m^2 s^{-1} flow. In HEC-GeoRAS, the thin blue line represents the stream center line while the yellow lines depict cross-sections. The circles represent bank lines on each cross-section, with diamonds depicting the water's extent. Diamonds outside of circles represents overbanking.

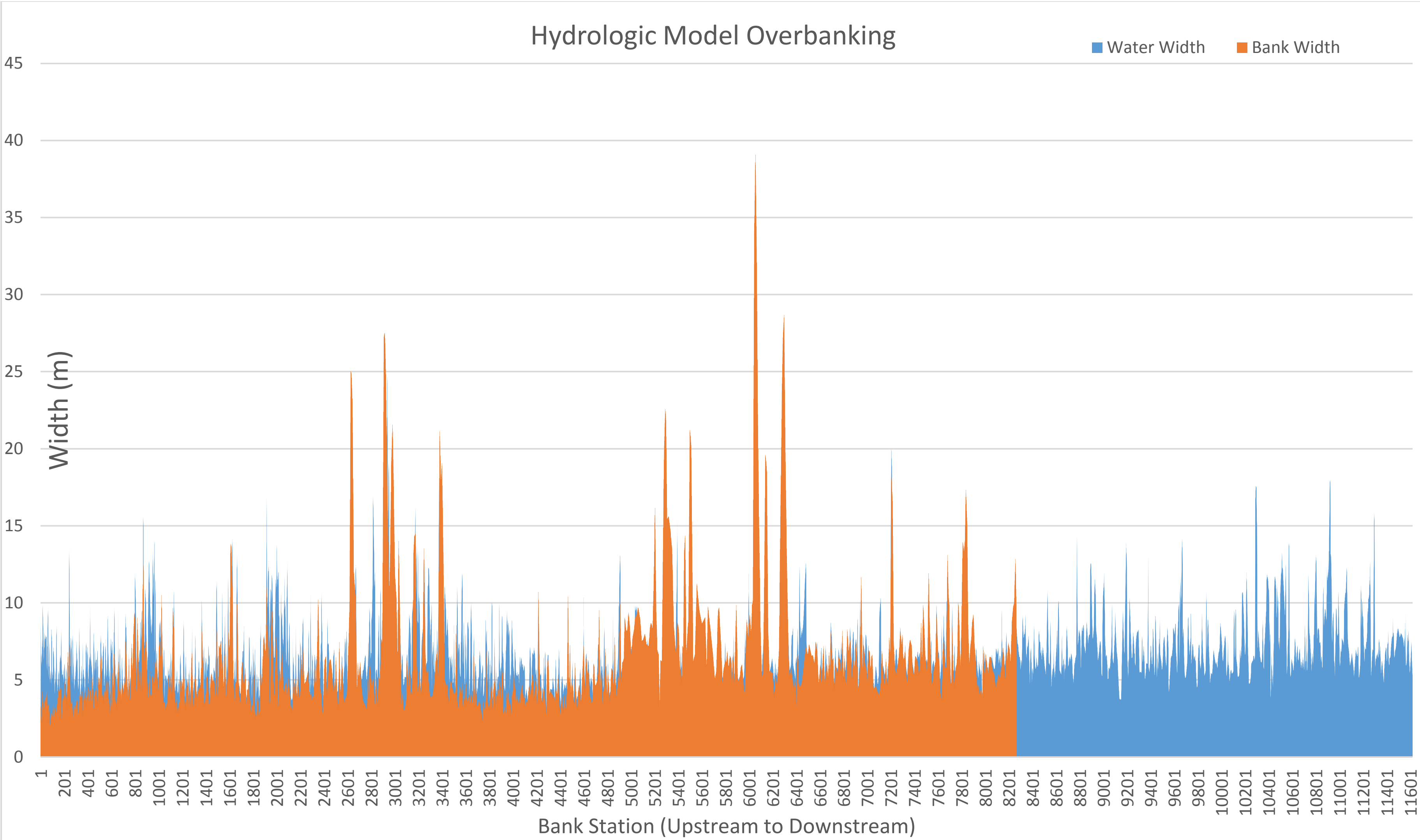


Figure 8. Data output from HEC-RAS model. Water widths are indicated by blue, and bank widths by orange. Any place where the blue graph is higher than the orange graph is an area where water width is wider than bank width, and overbanking is occurring. Upstream is left on the graph, downstream is right

Appendix A:

San Antonio Creek Exclosures: The eastern-most exclosures are located at 35°58’9.25” N, 106°30’13.94” W. The central exclosures are located at 35°58’3.47” N, 106°32’9.52” W. The western-most exclosures are located at 35°58’26.38” N, 106°35’45.50” W.