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Evaluation of Stream Bank Restoration to Improve Water Quality in a Semi-Arid Stream

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EVALUATION OF STREAM BANK RESTORATION TO IMPROVE WATER QUALITY IN A SEMI-ARID STREAM

by

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ABSTRACT

Evaluation of Stream Bank Restoration to Improve Water

Quality in a Semi-Arid Stream

by

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Humans have caused widespread watershed alterations including land cover shifts from native vegetation to agriculture crops and the introduction of livestock grazing. Specifically, these alterations have negatively impacted the water quality of Utah's Upper Sevier River where livestock grazing has led to accelerated bank erosion, loss of riparian vegetation, and subsequent stream temperature and turbidity increases due to reduce shading and increased fine sediment inputs. Utah has employed streambank restoration to address these degradations. The sort of streambank bioengineering performed by the state has been widely recommended and implemented, but its effects on water quality have rarely been quantified. Understanding the effectiveness of these actions is imperative as the state wishes to continue to implement this type of restoration. Thus, the objectives of this research were to quantitatively evaluate the effectiveness of completed streambank restoration near Hatch, UT and to test the ability of a simple sediment budget model to inform future streambank restoration.

First, I monitored stream temperature and turbidity upstream and downstream of restoration. I found that both stream temperature and turbidity increased downstream along the restored reach. I also used several stream temperature and turbidity thresholds to evaluate water quality—both biological standards for rainbow trout and those set by state governments. I found that turbidity frequently violated thresholds at both the upstream and downstream sites while temperature violated state standards at both sites but did not violate the biological thresholds. I was unable to conclude anything regarding the effectiveness of streambank restoration due to a lack of pre-restoration monitoring. I also completed a historical photo analysis (including quantification of uncertainty) to determine the effectiveness of streambank restoration on cut bank erosion rates. I found that streambank restoration dramatically decreased cut bank erosion to a non-zero rate thus reducing a major fine sediment source. My SIAM model was ultimately insufficient due to the dearth of available data. Streambank restoration successfully reduced cut bank erosion, and I recommended before-and-after monitoring for all future river restoration, suggested identifying and targeting upstream sources of fine sediment in addition to further downstream restoration, and pursuing more comprehensive predictive sediment modeling to inform future streambank restoration decisions.

(98 pages)

PUBLIC ABSTRACT

Evaluation of Stream Bank Restoration to Improve Water Quality in a Semi-Arid Stream

Johnathan Neenan

Human watershed activities such as converting land cover to agriculture and livestock grazing have negatively impacted stream water quality worldwide. One such case is Utah's Upper Sevier River where a loss of woody bank vegetation (reduced shading) and accelerated bank erosion (increased fine sediment inputs) has led to increased stream temperature and water turbidity. As a result, the state of Utah sought to improve water quality conditions using streambank restoration. While commonly recommended and performed, the effectiveness of this sort of restoration has rarely been quantified. Here, I evaluated a restored reach of the Upper Sevier River near Hatch, UT using continuous monitoring data and a historical photo analysis. As Utah wishes to continue performing this type of restoration in additional locations on the Upper Sevier River, I applied a simple sediment budget model to test its value in informing future streambank restoration decisions.

Continuous monitoring data at the upstream and downstream extent of restoration showed that both stream temperature and turbidity increased downstream along the restored reach. In addition, I found that stream temperature violated Utah's cold-water stream threshold at both sites but did not violate thresholds for rainbow trout. Turbidity violated state and biological thresholds at both sites. I was unable to conclude whether the streambank restoration directly altered water quality because I lacked monitoring data

before restoration occurred. Results of the historical aerial photo analysis showed that restoration practitioners were successful in reducing cut bank erosion. My use of SIAM as a simple sediment budget model proved insufficient due to poor data quality and quantity. Overall, streambank restoration was successful at reducing cut bank erosion, and I recommended monitoring future restoration before and after project completion, identifying and monitoring upstream sources of fine sediment, and pursuing more comprehensive sediment models to inform future streambank restoration.

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INTRODUCTION

Stream temperature and turbidity are two important water quality constituents that have been degraded by land cover change and water use. Stream temperature is a water quality master variable that governs chemical reactions, biological growth, physical processes, and overall aquatic ecosystem health (Caissie, 2006). Humans have altered stream temperature regimes via riparian vegetation reduction (reduced shading), altered channel morphology, climate warming, and water withdrawals (Webb et al., 2008; Poole and Berman, 2001). Excessive turbidity is an ecological stressor that hinders fish foraging, clogs gills, reduces egg survival, alters invertebrate drift, and transports nutrients and heavy metals (Bilotta and Brazier, 2008). Turbidity, like temperature, has been severely altered by riparian vegetation removal in favor of river-encroaching row crop and grazing pasture. These actions have led to upland rill erosion and bank destabilization resulting in greatly increased fine sediment loads to streams (Syvitski and Kettner, 2011). In addition, streamflow alterations such as flow regulation can increase fine sediment fluxes (Hellawell, 1988).

Degradation of water quality constituents like temperature and turbidity are sometimes addressed via stream restoration (Agouridis et al., 2005). A common type of stream degradation that affects both temperature and turbidity is unnaturally accelerated cut bank erosion caused by riparian vegetation loss, upland vegetation alteration, streamflow management, and bank trampling (Zaimes et al., 2004; Trimble, 1994). Streambank restoration methods, applied to address these degradations by both public and private entities world-wide, range from directly addressing the source of degradation with mechanical intervention to methods that are process-based and holistic. Armoring

banks with rip-rap directly addresses bank degradation and seeks to stop erosion entirely. As a side-effect, bank armoring prohibits channel dynamics and degrades the instream habitat complexity that is necessary to support aquatic biota (Schmetterling et al., 2004). Process-based river restoration relies on the reestablishment of natural systems to correct anthropogenic degradation (Beechie et al., 2010). For example, recent research suggests that natural/artificial beaver dams and increased streamflow via water purchases can address human induced stream temperature degradation (Elmore et al., 2016; Weber et al., 2017; Majerova et al., 2015). Occupying the middle ground, “bioengineering” describes what is often a combination of methods including bank sloping, bank revegetation, cattle ex-closure, and riparian buffer strip installation (Alexander and Allan, 2006). Thus, bioengineering combines direct river engineering and process-based techniques and has been widely applied throughout Europe and the United States (Evette et al., 2009; Li and Eddleman, 2002).

While river restoration is widely implemented, most river restoration projects are not appropriately monitored nor are they quantitatively evaluated to determine their effectiveness. In the United States, Bash and Ryan (2002) found around half of Washington state restoration projects were monitored while Bernhardt et al. (2005) found only 10% of projects were monitored nationwide. In addition, of the small number of projects that were monitored, Bernhardt et al. (2007) found that just 59% used quantitative means to evaluate them. Indeed, the need to quantitatively evaluate the benefits of stream restoration is widely acknowledged and systematic steps to improve the process have been recommended (Kondolf, 1995; Galatowitsch, 2012). In practice, river restoration is not monitored for several reasons including lack of funding,

knowledge, labor, and time. Nonetheless, the need to monitor restoration has been recognized and groups such as Utah's Watershed Restoration Initiative, part of the Utah Department of Natural Resources, now include a monitoring category in their project funding prioritization framework (WRI, 2019).

Scale is another challenge confounding restoration goals. River restoration projects are frequently completed on a reach scale, rather than basin scale, due to cost, physical limitations, and shortsightedness. Research indicates that river health improvements (particularly water quality) cannot be addressed with such a spatially limited approach and that managers should think and act on the watershed scale (Bernhardt and Palmer, 2011; Wohl et al., 2005; Doyle and Douglas-Shields, 2012). However, Wohl et al. (2005) point out that a larger scale presents several challenges such as an increase in stakeholder numbers and diversity, varying river dynamics across the watershed, and difficulty predicting restoration outcomes. Restoration is costly and time consuming, and, when considering widespread implementation on the watershed scale, one might ask the useful question: how much river needs to be restored to achieve desired results?

This question is complex, and there are often many "correct" answers. Predictive simulation models can be used to alter system variables and evaluate the effects and as such, are a valuable tool for river restoration decision making. For example, one could simulate the impacts of restoration implementation at several scales which would address the nagging question of "how much is enough?" Indeed, many stream temperature models have been successfully applied to river restoration questions (Elmore et al., 2016; Dzara et al., 2019; Chen et al., 1998) While there are many models that are capable of

modeling sediment and, several have been applied to upland land management alternatives (Ullrich and Volk, 2009; Santhi et al., 2001), they are often complex and require intensive data inputs. A sediment budget model with fewer data requirements would make addressing questions of scale much more accessible to managers.

A substantial body of literature exists that refers to and recommends stream bank restoration techniques (Li and Eddleman, 2002), but no studies have specifically evaluated the water quality improvements of installing log and rock vanes, excluding grazing, sloping banks, and planting riparian vegetation in semi-arid streams like Utah's Sevier River. In northern Mississippi, Shields Jr (2008) found that watershed scale bank stabilization did not decrease suspended sediment yields. Another study in Mississippi found that streambank stabilization using bendway weirs considerably reduced phosphorus inputs from eroding banks (Hubbard et al., 2003). The dearth of studies on this topic is likely due to the lack of monitoring of completed restoration mentioned above. This research addresses these gaps to better understand the effects of streambank restoration on water quality in semi-arid streams. In addition, the use of aerial photography in evaluating lateral stream movement is not new (e.g. De Rose and Basher, 2011; Day et al., 2013; Brizga and Finlayson, 1990; Brooks and Brierley, 2002; Miller et al., 2014; Spiekermann et al., 2017). Often these studies have acknowledged the uncertainty associated with digitizing features in aerial photography, but these errors have only recently been quantified (Day et al., 2013; Spiekermann et al., 2017; De Rose and Basher, 2011). My results include uncertainty brackets around erosion findings.

In addition, this research adds to the small number of published SIAM applications. Grabowski and Gurnell (2016) used SIAM on a UK river to examine reach-

wise sediment supply by grain size. The stream was unable to move gravel-sized sediment and bed aggradation was mostly caused by upstream sand sources. Smith (2016) used SIAM to assess the overall aggradation/degradation trends of the Middle South Platte River and to assess the effects of hydraulic structures (e.g. bridges, diversions) on those trends. Hydraulic structures slowed streamflow considerably forcing sediment deposition. Similarly, USACE (2009) applied SIAM to a Michigan river to examine deposition trends and the effect of ponds on sediment transport. They also included a small, hypothetical analysis of bank erosion inputs and found that they had little impact on pond sedimentation. There are no published instances of SIAM's use in streambank restoration. Those projects had detailed hydraulic, geomorphic, and sediment characteristic data readily available or began the project with the intent and funding to collect the necessary data.

Overall, my research addressed the two common challenges in river restoration management described above: 1) the lack of monitoring and quantitative evaluation of river restoration and 2) the applicability of simple simulation models for river restoration decision making. To address the first challenge, I evaluated the effects of streambank restoration on stream temperature and turbidity/fine sediment fluxes using two methods: instream water quality/quantity monitoring and historical aerial photo analysis. To address the second challenge, I developed a ~140-kilometer hydraulic model using the U.S. Army Corp of Engineers' HEC-RAS to use Sediment Impact Analysis Methods (SIAM) module. SIAM is a sediment budget tool used for quick assessment of broad sediment balance trends (Brunner, 2016). I then evaluated whether the simple results SIAM provided were useful for informing river restoration decisions. This research is the

first to apply SIAM to streambank restoration and use it in a data-poor environment. The specific objectives of my research were:

1) Monitor a completed restoration project on the Sevier River near Hatch, UT to determine whether stream temperature, turbidity, and cut bank sediment fluxes improved along the restored reach and

2) Test the ability of a simple sediment budget model to inform streambank restoration practices.

STUDY AREA

Located in south-central Utah, the Sevier River flows from its headwaters on the Paunsaugunt Plateau to the intermittent and endorheic Sevier Lake located in the Sevier Desert (Figure 1). Sevier Basin elevation ranges from 1,400 meters at Sevier Lake to over 3,600 meters in the high mountains. Yearly precipitation varies from as little as 163 mm in dry valleys to 1000 mm at upper elevations (SRWUA, 2018). Three HUC-8 watersheds comprise the Sevier River: the Lower, Middle, and Upper Sevier basins. In the 312,000-hectare Upper Sevier watershed (Figure 1), extensive water development and agricultural land conversion have altered the basin and the river. These changes have negatively impacted stream health in a variety of ways including habitat alteration, increased phosphorus, turbidity, and stream temperature (USFS, 2004). As a result, the Upper Sevier River was listed as an impaired 3A cold water fishery under the Utah Division of Water Quality's 2004 Total Maximum Daily Load for suspended solids (USFS, 2004).

This research focused on a 140-kilometer reach of the Sevier River from Hatch, UT through Circleville Canyon (Figure 2). In this area, Utah's Watershed Restoration Initiative has funded river restoration to enhance fisheries by improving aquatic habitat and water quality. Restoration has included sloping vertical banks, installing log and rock vanes, grazing exclusions, and planting woody riparian vegetation to decrease fine sediment influxes and reduce high stream temperatures (WRI, 2016, 2009). Note that there were no successful woody riparian vegetation plantings in the reaches I studied. Most of the completed restoration was performed near the town of Hatch. The restoration occurred from 2006 to 2009 over 11 river kilometers, but several more kilometers have

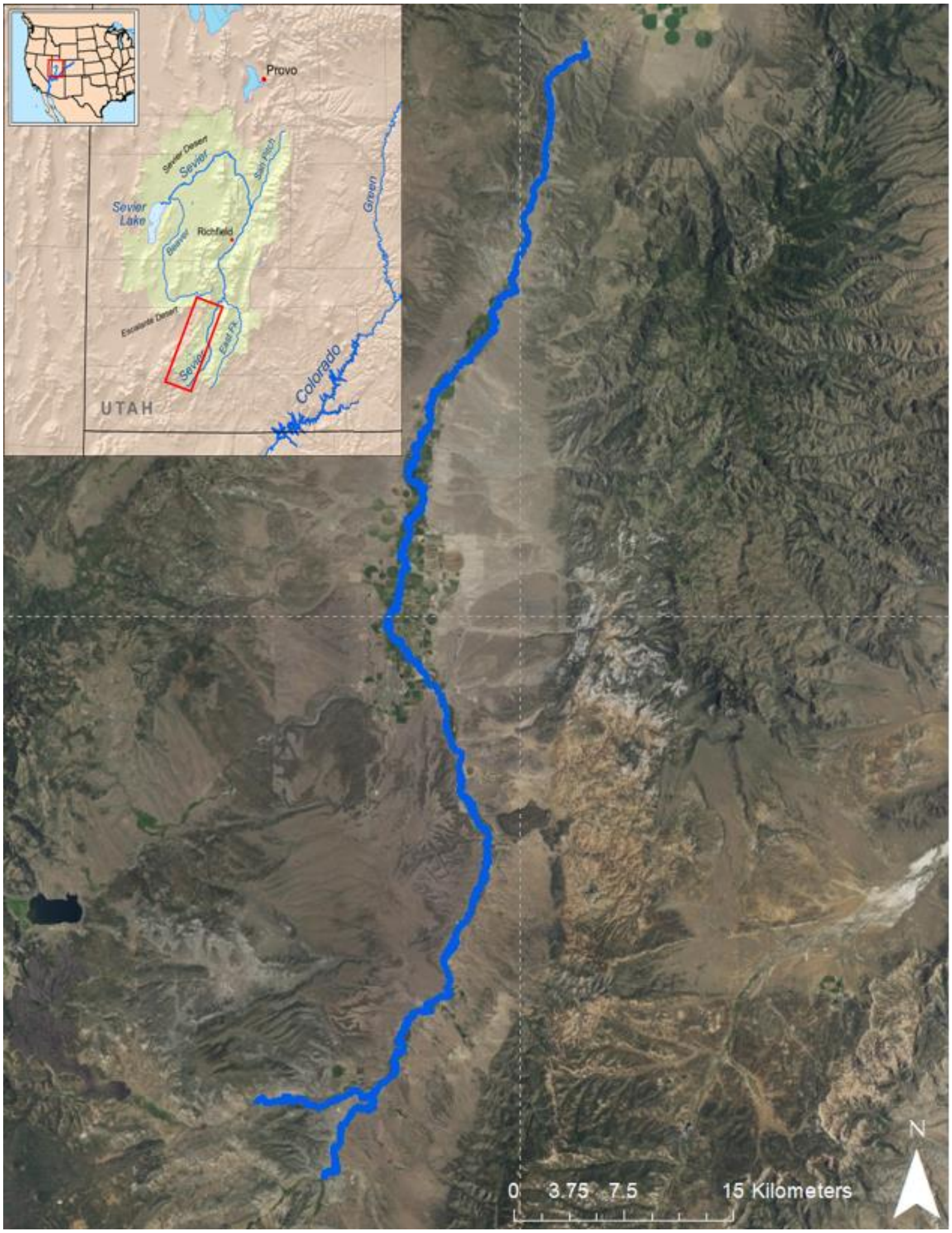

Figure 1. The Sevier River (with Mammoth and Asay Creeks at southern map extent) in Hatch and Panguitch Valleys and its location in Utah.

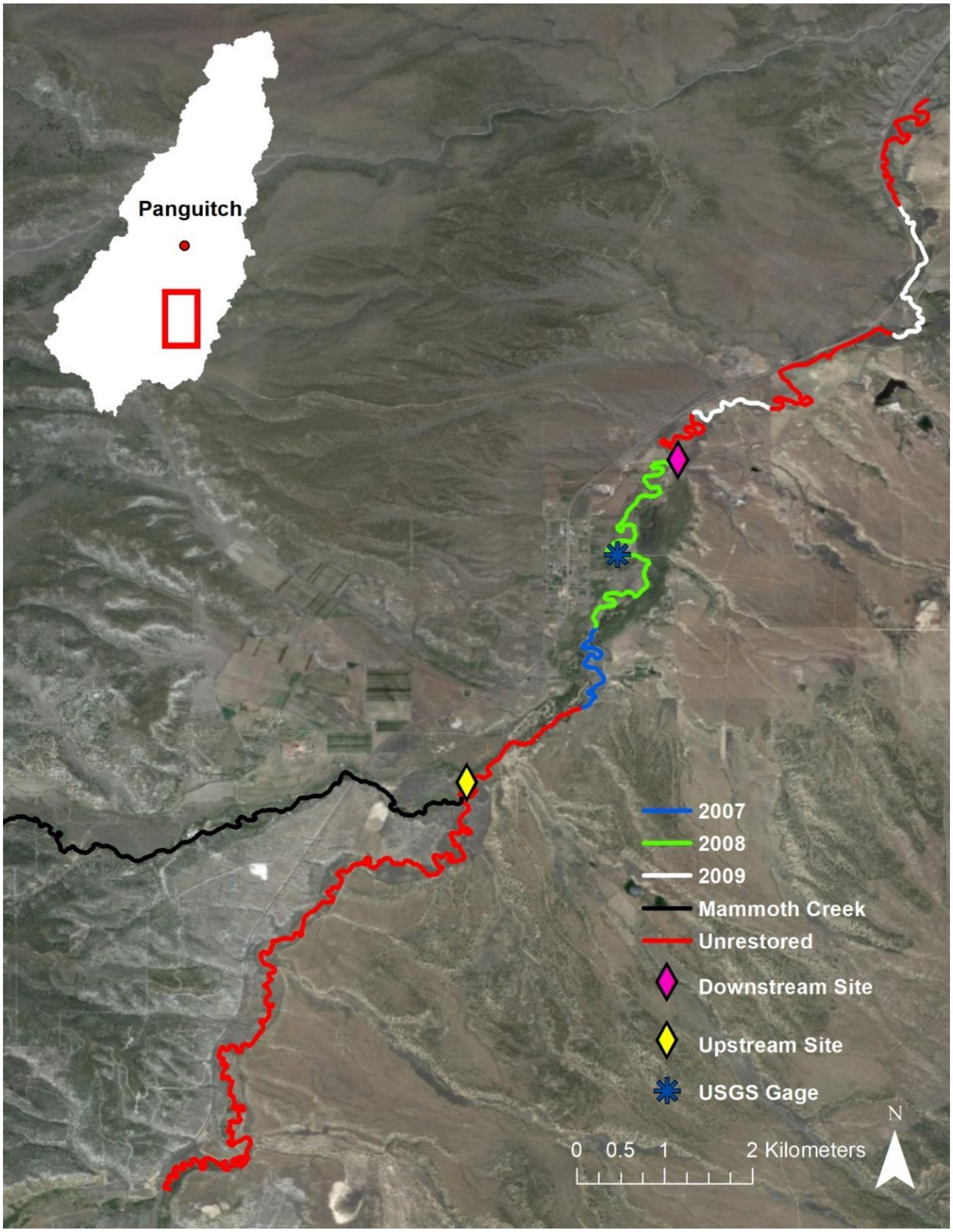

Figure 2. The Sever River and its location in the Upper Sevier Watershed. Continuous monitoring sites are indicated by diamonds and the colored reaches indicate restoration status and, if restored, when.

since been restored in the Hatch area. Sevier River restoration managers estimate 85% of potential Hatch Valley restoration sites have been restored (as of 2018) and progress on the remaining 15% is unlikely due to unwilling landowners (Nic Braithwaite and Wally Dodds, personal communication, 2018). Since completing restoration in Hatch Valley, restoration focus has turned to improving Circleville Canyon’s brown and rainbow trout (Salmo trutta and Oncorhynchus mykiss) sport fishery. The canyon’s fishery has been impacted by poor water quality delivered from the upstream, unrestored Panguitch Valley reach. Though considerable restoration has been completed in Hatch Valley, little has been done to quantitatively evaluate its effectiveness and managers continue to implement stream bank and riparian restoration projects without a reliable estimation of its effectiveness.

METHODS

The general structure of my methods and their connection to my research objectives is shown in Figure 3. This section begins by discussing the methods I used to collect and analyze instream field data. I then discuss the methods I used to quantify sediment fluxes and build the HEC-RAS and SIAM models. I used the historical aerial photo analysis results in both restoration evaluation and SIAM development.

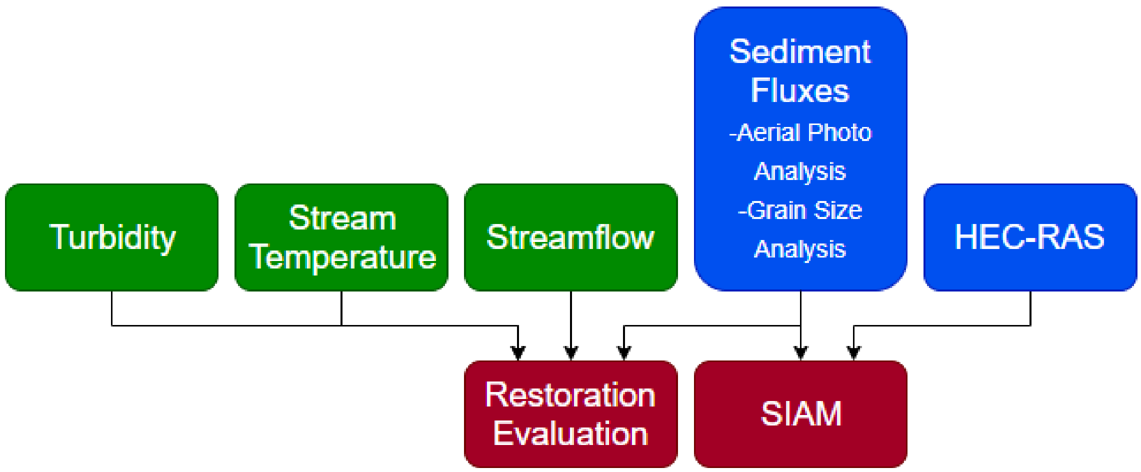

Figure 3. Conceptual model showing the connections between research objectives (red), field monitoring (green), and lab work (blue).

Water Quality Monitoring Near Hatch, UT

To quantify the effects of river restoration, I monitored hourly Sevier River stream temperature, turbidity and streamflow (Table 1) at sites upstream and downstream of a seven kilometer restored reach near Hatch following the approach shown in Figure 4. I also made periodic flow measurements using a portable flow meter. The turbidity sensor was connected to a Campbell Scientific CR200X datalogger. I constructed a semi-permanent monitoring structure at both monitoring sites that consisted of a 6-foot, 1.5-

inch galvanized steel post to which I attached a solar panel and a waterproof enclosure containing the datalogger and battery. I reinforced the structure with rocks to help withstand high streamflow and its inevitable use as a cow scratching post. An 8-foot, 4-inch diameter ABS tube ran from the datalogger enclosure to the water and housed the turbidity sensor. I placed one level logger in the water and attached a second to the steel post to measure and correct for barometric pressure. The under-water level logger was attached via zip ties to rebar. I collected data from July 28, 2017 to October 14, 2018.

Table 1. Field monitoring constituents and collection instruments with resolution and accuracy.

Monitoring Constituent	Instrument (Resolution, Accuracy)
Flow	Onset HOBO U20L-01 Level Logger (0.21 cm, ±1 cm) (one in water, one out of water to measure barometric pressure)
	Hach FH950 portable flow meter
Turbidity	FTS DTS-12 (0.01 NTU, ±0.1 NTU)
Temperature	Onset HOBO U20L-01 Level Logger (0.1 ˚C, ±0.44 ˚C)
	Onset HOBO U22-001 (0.02 ˚C, ±0.21 ˚C)

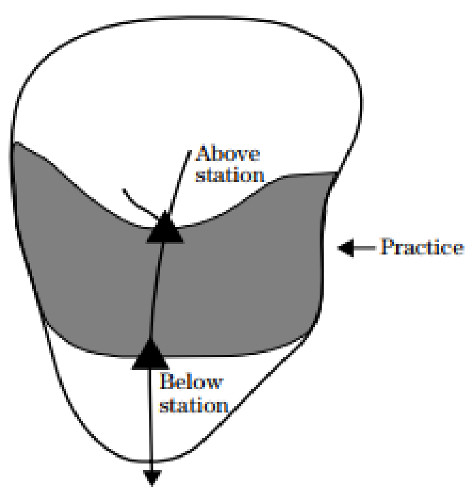

Figure 4. Upstream-downstream monitoring where triangles represent continuous sampling sites and the shaded area indicates completed restoration (NRCS, 2003).

Water Quality Analysis

Stage-Discharge

I developed a stage-discharge rating curve using periodical in-stream flow measurements and continuous stage data. I offset barometric pressure using an out-of-water pressure transducer and Onsets's HOBOware software. I used the built-in R tool "nls()" to perform a nonlinear regression to develop the relationship between discharge and stage following the standard power function described in Equation A1. I used root mean square error (RMSE) to quantify regression fit. RMSE is a commonly used measure of the difference between observed and modeled data where a value of 0 represents a perfect fit. Singh et al. (2005) suggested that RMSE values less than half of the measured data's standard deviation are considered low. I used the regression to calculate discharge from the stage time series. Unfortunately, I was unable to establish acceptable rating curves (Figure A5, Figure A6, Table A4) so any further analyses involving streamflow were completed using data from the previously mentioned nearby USGS gage.

Stream Temperature

I evaluated temperature using several state and biological standards. Utah's class 3A standard specifies a maximum stream temperature of 20 °C (Utah DEQ, 2019). Unfortunately, guidelines for assessing whether a stream is in violation of this standard are unclear (Utah DWQ, 2018). For example, it is not clear whether a single value in excess of the threshold is considered an impairment or if more (and how many) are necessary. Of course, acute exceedances (e.g. one or two hours) of a fish's thermal tolerance impact their health differently than prolonged warm periods. I tested whether daily temperature maximums, means, and minimums exceeded 20 °C using R package

“StreamThermal” (Tsang et al., 2016). I also calculated both 7-day running average for daily maximum (7DADM) and daily average (7DADA) stream temperature. These metrics are more biologically relevant than examining hourly or daily maxima/means (Nelitz et al., 2007; Sullivan et al., 2000). Both metrics are used by Oregon DEQ, Idaho DEQ, and Colorado Water Quality Control Division to assess stream temperatures (Colorado WQCC, 2011; Oregon DEQ, 2004; Idaho DEQ, 2011). I also assessed whether stream temperature violated biological thresholds of a relevant sport fish. I used rainbow trout thresholds of 19 °C and 24 °C for maximum weekly average temperature and maximum temperature for short exposure (24 hours) survival, respectively (Brungs and Jones, 1977). In addition, I evaluated the maximum 7DADM threshold of 22 °C set for the rainbow trout set by Feldhaus et al. (2010). Idaho DEQ uses the same 22 °C maximum 7DADM threshold for “cold water” stream which are largely salmonid habitat that includes rainbow trout (Idaho DEQ, 2011). Results show June through August stream temperatures were the only months with elevated stream temperatures near the cited thresholds.

Turbidity

I quantitively evaluated turbidity in three ways. First, I compared upstream and downstream daily mean turbidity to one another directly. Because rivers flow downstream and bring with them their upstream contents, I offset the initial hourly time series by five hours to account for travel time. I examined several peak turbidity events and velocity measurements to determine the stream’s travel time. Second, I calculated the fraction of the time period above biological turbidity thresholds for rainbow trout. Rowe et al., (2003) determined two turbidity thresholds where juvenile rainbow trout no longer

size-selected prey. These were 20 NTU for Chironomidae larvae and 160 NTU for Deleatidium. Rainbow trout are strongly size selective so an inability to choose larger prey would likely lead to decreased growth and poorer population health (Rowe et al., 2003). Last, I evaluated the absolute, instantaneous 50 NTU threshold recommended by the Idaho DEQ (Paul et al., 2008). They established this and other water quality thresholds with respect to aquatic biota and in particular, salmonids. I performed each analysis for the full period of observation and separately across each meteorological season. I quantified threshold violations by calculating the fraction of daily means above the threshold in each period. Note that I did not directly examine Utah's turbidity standard of 10 NTUs above background levels because natural background levels vary with geology, land cover, and hydrologic conditions that change seasonally so their quantification is ambiguous and difficult to apply (Paul et al., 2008). Due to its limited temporal and spatial scales, this study does not attempt to rigorously quantify these background levels, but this data could nonetheless inform a broader effort to do so.

Sediment Flux Analysis

Effect of Streambank Restoration on Cut Bank Erosion

My field monitoring approach employed an upstream-downstream (control-impact) monitoring method described by NRCS (2003). The primary assumption of this approach is that any measured water quality changes are a direct result of the river restoration rather than a function of potentially confounding, unaccounted for variables. For example, it is possible that turbidity naturally increases along my reach of interest, and, restoration simply dampened the degradation-related increase. Without knowing the background conditions, upstream-downstream monitoring results might suggest either no

improvement or worsening water quality when, in fact, the restoration may have produced an improvement—a type II error. One solution is to supplement control-impact monitoring with a before-after analysis (i.e. a BACI design) to identify pre-restoration and background trends. Unfortunately, as the Sevier restoration was previously performed and no new restoration was implemented over the course of my research, I could not perform employ a before-after analysis. To supplement my field analysis, I examined historical aerial photographs.

I performed a historical aerial photo analysis to determine the volume of cut bank sediment eroded over several time periods and spatial locations pre- and post-restoration. Rather than conduct a full channel migration analysis, I examined only the sources of sediment targeted by restoration practitioners. These turbidity-relevant sediment sources are called cut banks and are often found on outside bends. This analysis directly examined the efforts of restoration practitioners.

I acquired aerial imagery for the Sevier River in both Hatch and Panguitch Valleys for five years (Table 2). I chose these years for two reasons: 1) they bracket both a major flood in 2005 restoration and the restoration which occurred from 2007 to 2009) and 2) photo availability and resolution. I examined the following time periods: 1974 – 2004, 2004 – 2006, 2006 – 2009, and 2009 – 2014 (Table 2). In addition to the analysis in Hatch, I performed the same analysis for a reach in Panguitch Valley that has never been restored. Though each photo was previously georeferenced, I verified this by identifying permanent structures (e.g. boulders, bridges, etc.) and measuring whether these points of reference moved substantially between photographs. No georeferencing changes were necessary.

Table 2. Sevier River aerial photo year, source, resolution, restoration status, and color properties C = Color, CI = Color Infrared, all photos available at raster.utah.gov. NAPP = National Aerial Photography Program, NAIP = National Agriculture Imagery Program.

Restoration Status	Year	Source	Resolution (m)	Attributes
Pre-	1974	NAPP	1	CI
Pre-	2004	NAIP	1	C
Pre-	2006	NAIP	1	C/CI
During	2009	NAIP	1	C
Post-	2014	Google	0.5	C

Streamflow, especially peak flow, is a major driver of cut bank erosion. Because of this, it is important that there were no major hydrologic differences between the compared time periods. Figure 5 demonstrates the similarity of these periods with regards to annual peak and total streamflow for the Sevier River near Hatch, UT. The largest flow during this period occurred during 2005 (1990 cfs). Photo dates bracketed this event so it did not influence pre- to post-restoration comparisons and could be examined individually (or combined with the pre-restoration time period if need be). I calculated peak flow recurrences for 1974 to 2014 and found that the post-restoration period experienced both 10-year (1200 cfs) and 4-year (760 cfs) floods during the five-year period. This flood record for a five-year period is not uncommon in the pre-restoration time period (see: mid-90s, mid-80s) so these eras are likely hydrologically similar.

I used ArcMap to digitize cut banks along the stream as polylines for each photo year. After digitizing cut banks, I connected neighboring polylines to create 2D polygons that represented lateral cut bank movement over the time period (e.g. 2009 to 2014). In

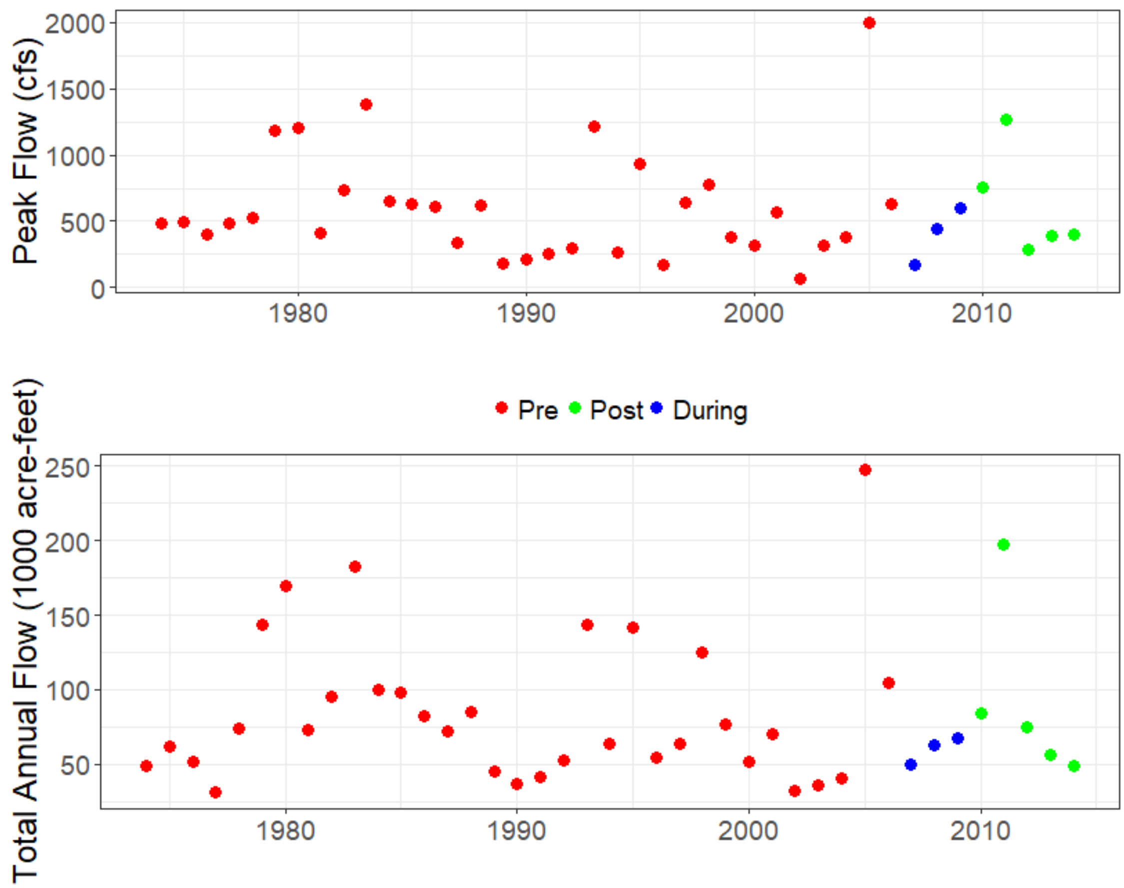

Figure 5. (a) Yearly peak Sevier River streamflow at Hatch USGS gage and (b) total annual streamflow at the same gage. Colors indicate restoration status.

addition, I assigned each polygon two attributes: sediment source and restoration status (including year restored). Cut bank sediment sources were categorized into three types: : alluvium, colluvium, and old valley fill. I supplemented aerial imagery with topography to guide my sediment source type decisions. I chose an alluvium source when it was the clear that the river was migrating in sediment it had likely deposited in the past, I chose colluvium when the stream ran up against a steep slope, and I chose old valley fill when the river was eroding remnants of Pleistocene valley fill. One can generally tell the difference between alluvium and Pleistocene remnants according to vegetation type and surface elevation. Old valley fill was often populated by sage brush and other shrubs and

was generally at least 1 meter higher in elevation than alluvium. Alluvium was found at lower elevations and vegetation was greener with short grasses. I assigned each polygon its restoration year (or unrestored) using maps provided by Nic Braithwaite of UDWR (Figure 2).

Next, I converted the two-dimensional polygons to three-dimensional volumes following Equation 1. Due to limited field access and the lack of high-resolution elevation/morphology data in the region, I uniformly applied bank height values to Hatch and Panguitch Valleys after field visits and surveying cut banks in both locations. These values were two meters for Hatch Valley and one meter for Panguitch Valley.

Equation 1. Calculations for cut bank movement mud movement.

$$\frac{Lateral\ Movment\ (m^2) * Bank\ Height(m)}{Reach\ Length\ (km) * Time\ Gap(yr)} * Mud\ Modifier\ (\%)$$

$$= \frac{m^3\ Cutbank\ Mud}{km\ yr}$$

I calculated reach length by digitizing the river centerline using 2004 and 2014 aerial photos to account for potentially large geomorphic changes caused by the 2005 flood event (Table A1). In general, the river length differences between years were minimal except that the unrestored reach was about ~three km shorter in 2014 compared to 2004—a ~12% decrease in length. Conversely, the Panguitch reach length increased by 9% during the same time period.

I included a “mud modifier” in Equation 1 to estimate turbidity-relevant sediment volumes. This restoration sought to reduce turbidity, and, as “mud” (particle size < 63 µm) is the major contributor to high turbidity measurements (Merten et al., 2014), this

modifier better describes the effects of restoration. The mud modifier was different for each sediment source type and described the fraction of mud in each type. To estimate this modifier, I performed a grain size analysis.

My grain size analysis consisted of 13 (locations listed in Table A2) cut bank sediment samples. At each site, I collected roughly 250 grams of cut bank sediment at the vertical midpoint to avoid aeolian contamination. I used aerial photos, topographic data, and in-field assessments to define sediment source type for each sample. Samples were air dried, and I performed a dry-sieve at 1-mm following the methods described by Folk (1974). I then completed a laser diffraction particle size analysis on the fractions smaller than 1 mm using a Malvern Mastersizer 2000. The Malvern Mastersizer 2000, and laser diffraction particle analysis in general, is a fast, reliable, reproducible, and precise method for developing particle size distributions of fine sediment samples (Sperazza et al., 2007). Ryzak and Bieganowski (2011) and others have developed optimal methodology which has informed the methods defined by Luminescence Lab at Utah State University where I completed this analysis (Dr. Tammy Rittenour, personal communication, 2018). I used the resulting grain size distributions (Figure A13) to calculate mud fractions (% less than 63 μm) for each sample. I then found an average mud fraction for each sediment source type. I used these values as the “mud modifier”, and they were 62.5% and 51.8% for alluvium and old valley fill respectively. I did not identify any colluvium sources of eroding material.

I delineated the longitudinal extent of each cut bank for each photo year as shown in Figure 6 (further examples for each year are shown in Figures A1, A2, A3, A4). Cut bank lines were converted to polygons to represent lateral cut bank movement between

photo years (Figure 7). Note the unnatural, straight line cutoff of the 1974 to 2004 polygon toward the bottom right of Figure 7. This is an example of the uncertainty and ambiguity when delineating these features which I address in a later section of this research. I delineated 928 polygons.

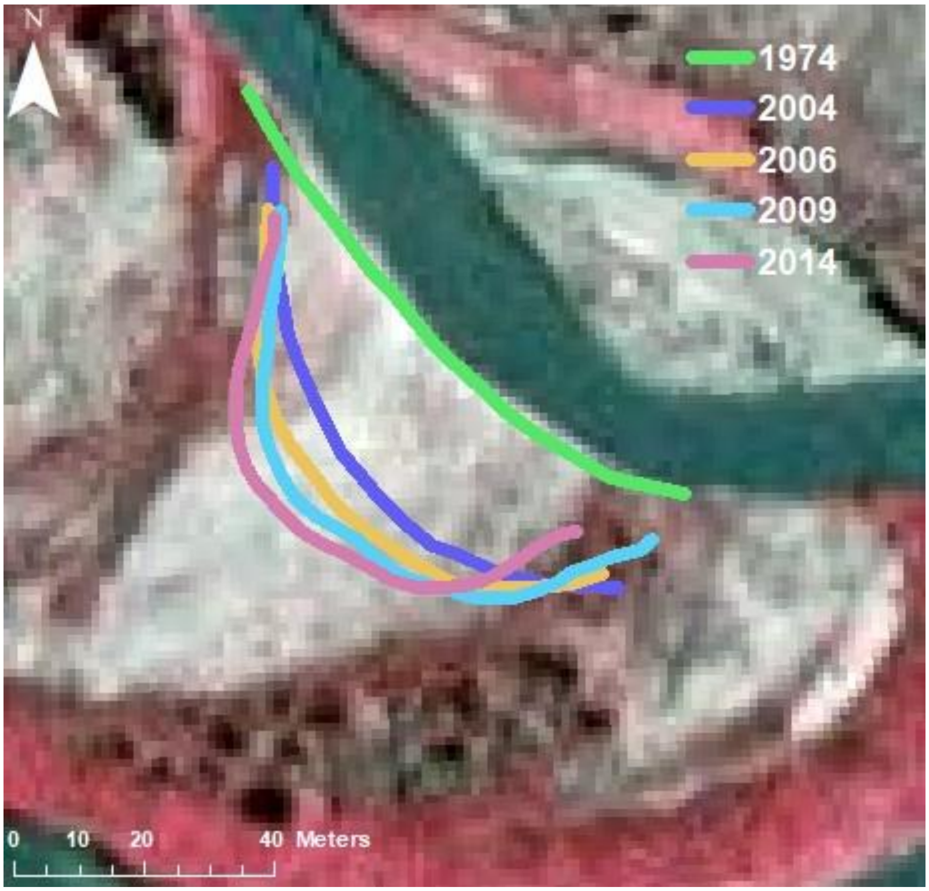

Figure 6. Cut bank delineation example with 1974 photo.

Cut Bank Delineation Uncertainty

Delineating features in historical aerial photos can be ambiguous and imprecise. Reasons for this include resolution, plane angle, shadows, stream stage, and artificial bank alteration (e.g. the bank sloping performed on the Sevier). Here, the target elements were cut banks whose precise boundaries were uncertain in the absence of ground

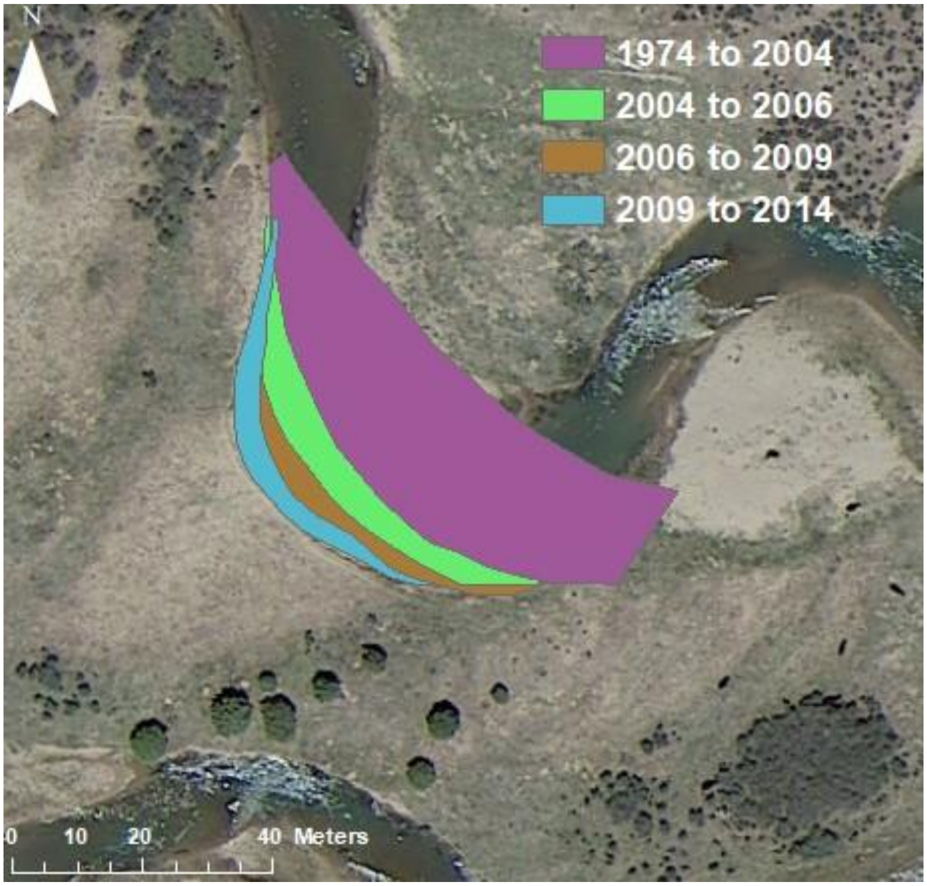

Figure 7. Example polygon results from aerial photo cut bank delineation for four time periods. Photo is from 2014.

observations or high-resolution topographic data. To delineate these features, I used knowledge of general river dynamics (i.e. where cut banks are likely to occur) and site-specific field observations. In particular, identifying precise horizontal bank location is achallenge. It is difficult to laterally delineate banks that are tall and sloping with sparse vegetation, thus estimates are uncertain. For example, cut banks often appeared as white to light brown strips whose bank line one could reasonably digitize on either edge—or somewhere in the middle. In my photos, this horizontal discrepancy ranged from one to four meters.

To quantify this uncertainty and corroborate results, I randomly resampled 25 (of 226 total) cut bank locations focusing on feasible polygon delineation extent. I digitized the maximum and minimum reasonable extent of each polygon and time period of my initial photo analysis, considering the aforementioned factors that cause ambiguity.

HEC-RAS Model Development

I used Hydrologic Engineering Center's River Analysis System (HEC-RAS) to simulate a one-dimensional, steady flow for ~140 kilometers of the Upper Sevier River from Asay Creek to the end of Circleville Canyon (Figure 1). My model consisted of three "hydraulic reaches": Asay Creek, Mammoth Creek, and the Sevier River from the confluence to Circleville Canyon. A HEC-RAS model is a prerequisite for the use of SIAM. HEC-RAS models one- and two-dimensional river hydraulics, sediment dynamics, and water quality using both steady and unsteady flows and was developed by the U.S. Army Corps of Engineers (USACE, 2010). HEC-RAS has been applied to a wide variety of topics from flood prediction (Timbadiya et al., 2011) to river restoration (Buchanan et al., 2012). Table 3 lists the data necessary to develop my HEC-RAS model and their sources.

Table 3. HEC-RAS model data and source

Data	Source
Channel Geometry	Synthetic, trapezoidal geometry based on channel width from aerial photography, field survey bank height estimates Elevation from 10 m DEM/TIN
Streamflow	USGS gages: 1) Sevier near Hatch (1017450) 2) Mammoth Creek (10173450)
Manning's n	Chow (1959)

Due to the lack of high-resolution channel elevation data, I developed a representative, synthetic, trapezoidal channel using elevations from a TIN (made from 10m DEM) as bed elevation, a digitization of bankfull channel width from a high flow aerial photograph (2011, peak flow ~1,300 cfs), and estimated bank heights from my aerial photo analysis (one meter for Panguitch and two meters for Hatch) (Figure 8). I developed these cross sections at the upstream and downstream extent of each restorable reach (roughly two river-kilometers long) in Panguitch Valley, and I maintained the same two km maximum gap between cross sections in Hatch Valley. I used the built-in HEC-RAS interpolation function to interpolate cross sections every 250 meters between my user created cross sections for both valleys. While the synthetic geometry does not capture the fine details of bed topography, I sought to examine broad sediment transport patterns which added local complexity would only obscure. Likewise, interpolating two kilometers between cross sections does not capture stream meanders. To address this, I made sure each reach represented real-world reach length.

I used a Manning's n of 0.045 for the Sevier River as it is a winding stream with minimal bed vegetation (Chow, 1959). I chose 0.04 for Mammoth Creek as it is less sinuous and has less vegetation than the Sevier River (Chow, 1959). I selected 0.07 for Sevier floodplain roughness due to its relative flatness and lack of large trees or shrubs, and I selected 0.09 for Mammoth Creek because there are more trees in its floodplain (Arcement Jr. and Schneider, 1989).

Sediment Budget Modeling with SIAM

Sediment Impact Analysis Methods (SIAM) is a sediment budgeting tool packaged with HEC-RAS. SIAM compares sediment transport capacity and supply and

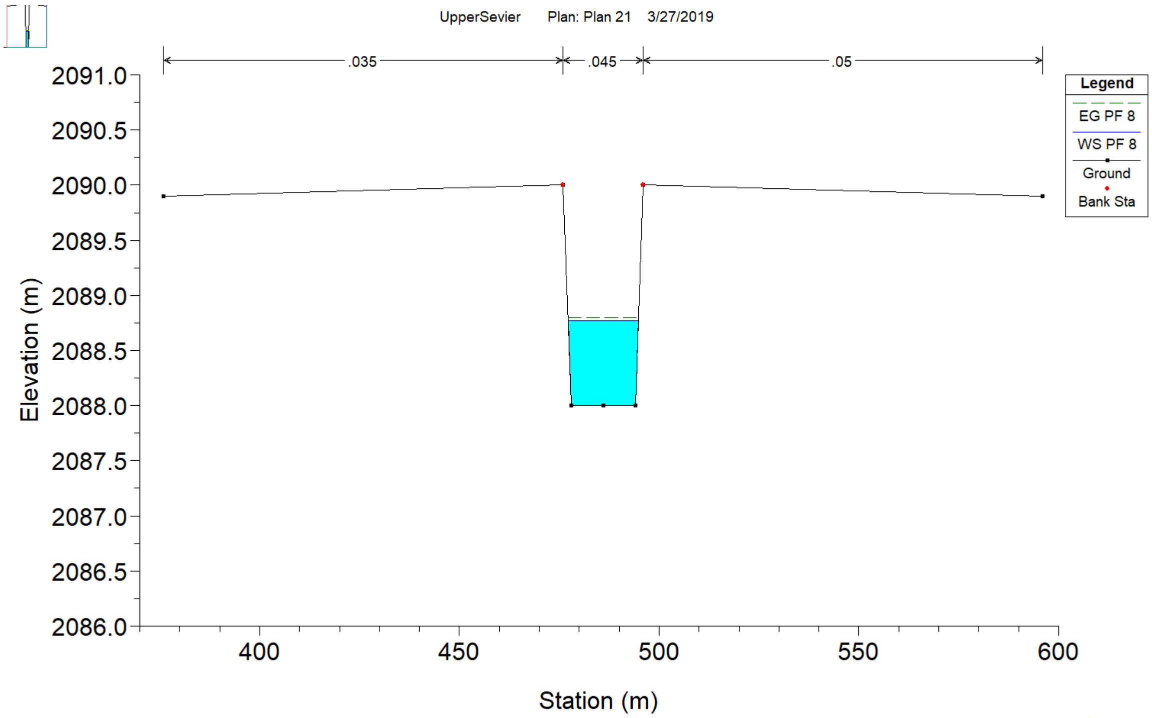

Figure 8. HEC-RAS example cross section with flow profile at 10 cms.

identifies whether stream reaches are in sediment surplus, deficit, or equilibrium. SIAM is a steady state model that operates on a yearly time step. SIAM inputs include defining sediment reaches, flow duration curves, bed material content, local supplies in each sediment reach, and a sediment transport equation. The results identify theoretical sediment imbalances.

There are several transport equations available in SIAM. Here, I used the default: Ackers-White, a total load transport function whose upper applicability boundary is 7 mm diameter particles (Brunner, 2016). As the sediment sizes in this analysis were all finer than 7 mm, I chose to use the default method rather than further complicate the analysis by investigating other methods.

Sediment Reaches

Sediment reaches are groups of cross sections with consistent hydraulic and sediment properties (USACE, 2010). For the Sevier River near Hatch, I defined sediment reaches to be the same as reaches in my aerial photo analysis. I defined Mammoth and Asay Creek as their own sediment reaches because they were not an important part of my SIAM analysis. In Panguitch Valley, I defined sediment reaches as roughly ~2 rkm stretches where restoration might be applied. That is, reaches with higher sinuosity and actively eroding cut banks. I qualitatively evaluated this by examining cut bank movement across my aerial photographs. I separately isolated and defined the straight reaches where this type of bank restoration would not be appropriate. I defined a total of 57 sediment reaches.

Flow Duration Curves

I acquired USGS data for both the Sevier River near Hatch and Mammoth Creek, but Asay Creek was ungauged. I estimated Asay Creek's flow with a water budget (i.e. Sevier River flow minus Mammoth Creek flow at each time step). Then, I developed a flow duration curve for each river using R software package "hydroTSM". An example of 2014's flow duration curve for the USGS gage near Hatch is shown in Figure A7. Once I created a duration curve for each year from 2009 to 2014 at each site, I discretized the curve by extracting values at 15 exceedance intervals because SIAM requires flow duration a discrete input. I then took an average for the time period at each site and each of the 15 exceedance intervals (Figure 9). A key assumption is that I assumed no streamflow loss from the Sevier near Hatch gage through Circleville Canyon. In reality,

there are several diversions and return flows in Panguitch Valley as agricultural is the dominant land use.

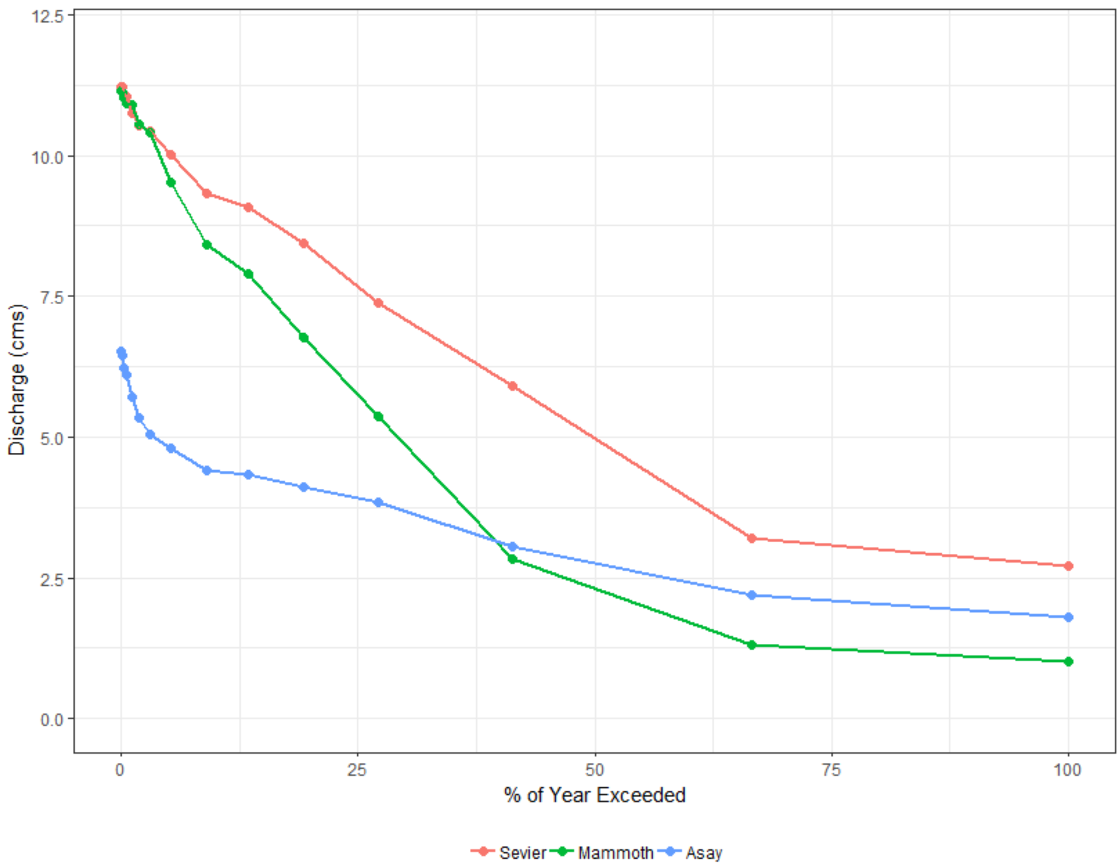

Figure 9. Flow duration curves for the Mammoth Creek, Asay Creek, and the Sevier River downstream of the confluence.

Bed Material

Bed material is a SIAM input required to calculate transport capacity of the stream. A single bed grain-size distribution was applied to both Hatch and Panguitch Valley based on qualitative field observations (Table 4). The values in the cut bank column of Table 4 are overall averages for particle size classes using my grain size sample results. I assigned a coarse bed material grain-size distribution to steeper reaches to prevent the model from eroding those reaches.

Table 4. SIAM model bed and cut bank material content.

Class	Particle Diameter (mm)	Panguitch (% Finer)	Hatch	Coarse	Cut Bank
Clay	0.004	0	0	0	0
VFM	0.008	0	0	0	17.3
FM	0.016	4	0	0	9.8
MM	0.032	10	4	0	13.4
CM	0.0625	10	5	0	15.6
CFS	0.125	15	10	0	17.7
FS	0.25	20	20	0	16.7
MS	0.5	15	20	0	8.1
CS	1	10	15	0	1.3
VCS	2	10	15	4	0
VFG	4	4	6	4	0
FG	8	2	3	4	0
MG	16	0	2	20	0
CG	32	0	0	40	0
VCG	64	0	0	26	0
SC	128	0	0	2	0
LC	256	0	0	0	0
SB	512	0	0	0	0
MB	1024	0	0	0	0
LB	2048	0	0	0	0

Extrapolating Cut Bank Erosion for Panguitch Valley

To improve my SIAM model's representation of the real world, I included cut bank sediment inputs for each SIAM reach. Measured results from my aerial photo analysis were used where available in Hatch Valley and a portion of Panguitch Valley. There were several reaches in Panguitch Valley where cut bank erosion was unknown, so I developed a relationship between erodible bank length and cut bank movement to estimate local sediment inputs in those reaches. I chose erodible bank length as a simple and measurable reach characteristic.

I used georeferenced aerial photographs to delineate erodible bank length following a method similar to my initial polygon analysis. I examined the stream across

my historical photos, found outside bends/cut banks, identified regions of noticeable cut bank erosion, and digitized a polyline along the length of bank that is actively eroding. I then associated each line with its reach and summed the length. I then developed linear regressions (forced through zero as, if there is no erodible bank, there cannot be erosion) to quantify the relationship between erodible bank length and eroded cut bank volume.

I input these cut bank local reach supplies into SIAM by creating a universal "cut bank" source which I set as the average value across all reach cut bank inputs. I then set an individual multiplier in each reach that modified the universal input to match actual input to that reach. For example, the average cut bank input for a reach was 1,122 tonnes per year. The observed cut bank supply in the reach restored in 2008 was 1,815 tonnes per year. Thus, I set the multiplier in the 2008 reach to 1.617. In addition, I separated the initial 1,112 tonnes per year into grain size classes following the content results for cut banks in Table 4.

Calculating Vertical Bed Movement

To compare the sediment impacts between reaches, I calculated reach-normalized, vertical bed movement from SIAM's local balance output following Equation 2. I obtained reach length and width by digitizing them using aerial photographs.

Equation 2. Calculating vertical bed movement where Δz is vertical stream bed movement.

$$\Delta z\ (m) = \frac{Local\ Balance\ (m^3)}{Reach\ Length\ (m) * Average\ Reach\ Width\ (m)}$$

RESULTS

Water Quality Monitoring Near Hatch, UT

Stream Temperature

The sites on the Upper Sevier River near Hatch displayed a typical temperature pattern reaching a peak temperature of 23.9 °C at the downstream site on June 1, 2018 and minimum temperatures of 0 °C several times during winter and early spring (Figure 10). Stream temperature followed air temperature patterns (Figure A8).

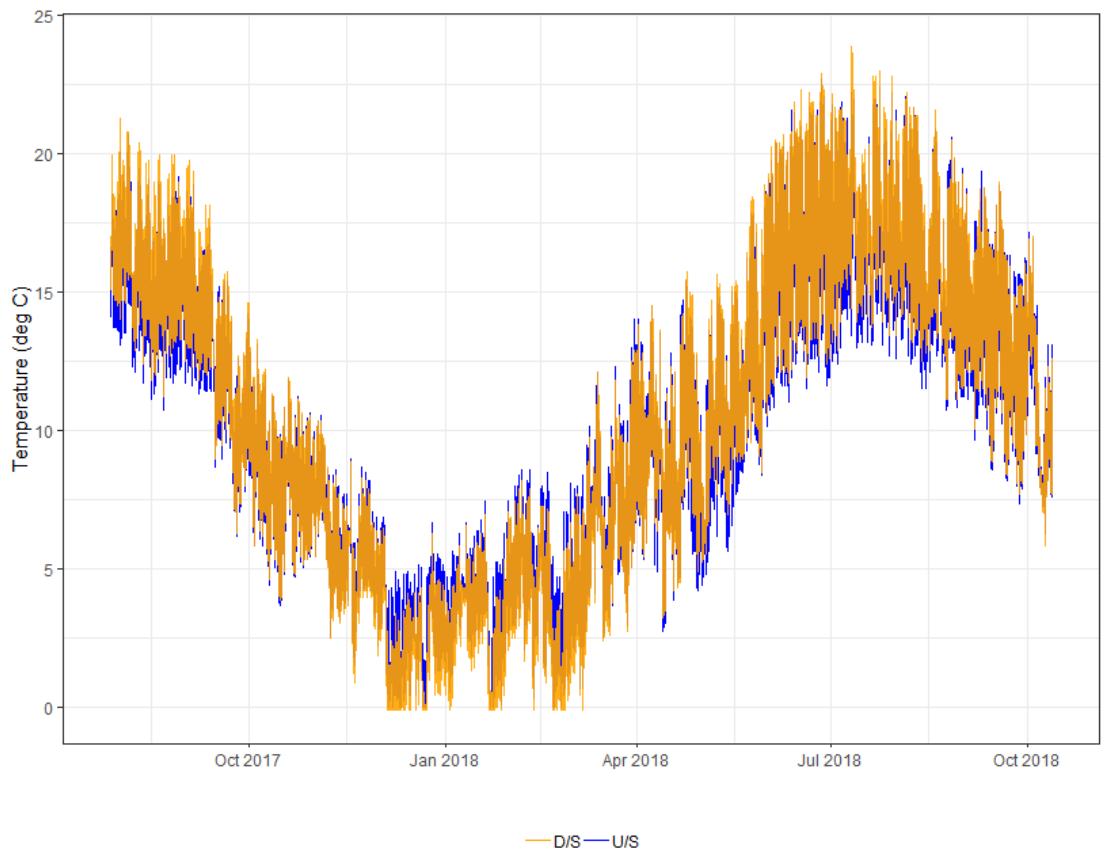

Figure 10. Hourly Sevier River water temperature at upstream (U/S) and downstream (D/S) monitoring sites from July 28, 2017 to October 14, 2018.

Both sites had similar seasonal temperature fluctuations and were generally within 1°C of each other with one notable exception (Figure 11c). During winter 2018, upstream

high temperatures were more than 2 °C warmer than the downstream site. Winter 2018 stream temperatures are shown alongside air temperatures in Figure A9. The diel variation in air temperatures was much larger (up to 5 °C warmer and cooler) at the upstream site than the downstream site. It is clear from these figures that summer was the only seasons that was in danger of violating the state's 20 °C threshold, thus I completed the threshold analysis on those periods.

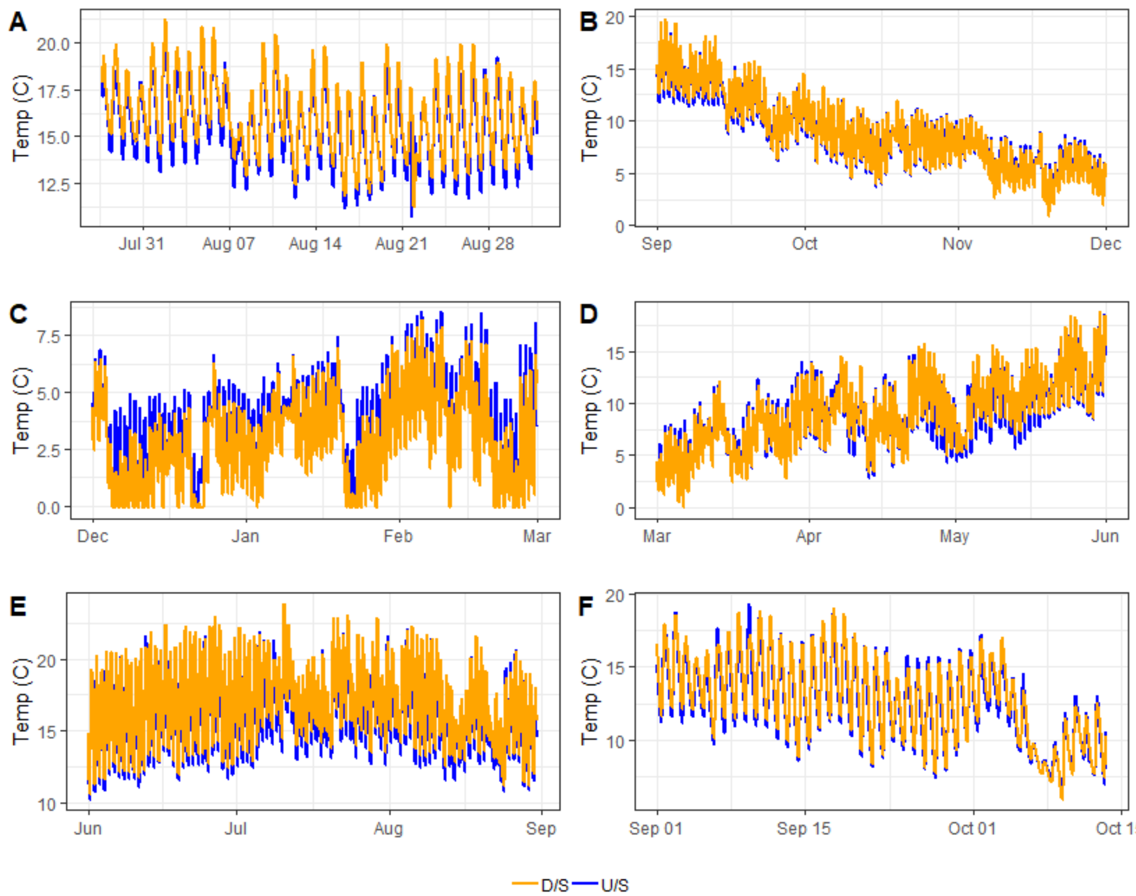

Figure 11. Hourly Sevier River water temperature at upstream (U/S) and downstream (D/S) monitoring sites for (a) summer 2017, (b) fall 2017, (c) winter 2018, (d) spring 2018, (e) summer 2018, and (f) fall 2018.

Neither the daily mean nor minimum stream temperatures violated the Utah 3A threshold during any season (Figure 12). However, both the upstream and downstream

site maximum daily temperatures violated the threshold for 60 and 63 days of summer 2018, respectively. The warmest temperatures measured during this study were 22.7 °C (upstream) and 23.9 °C (downstream). So, maximum daily temperatures violated Utah's threshold, but neither site exceeded the lethal biological threshold for rainbow trout of 24 °C. However, the downstream site came within 0.1 °C of the 24 °C threshold suggesting water temperatures may stress trout river managers should continue to monitor stream temperatures.

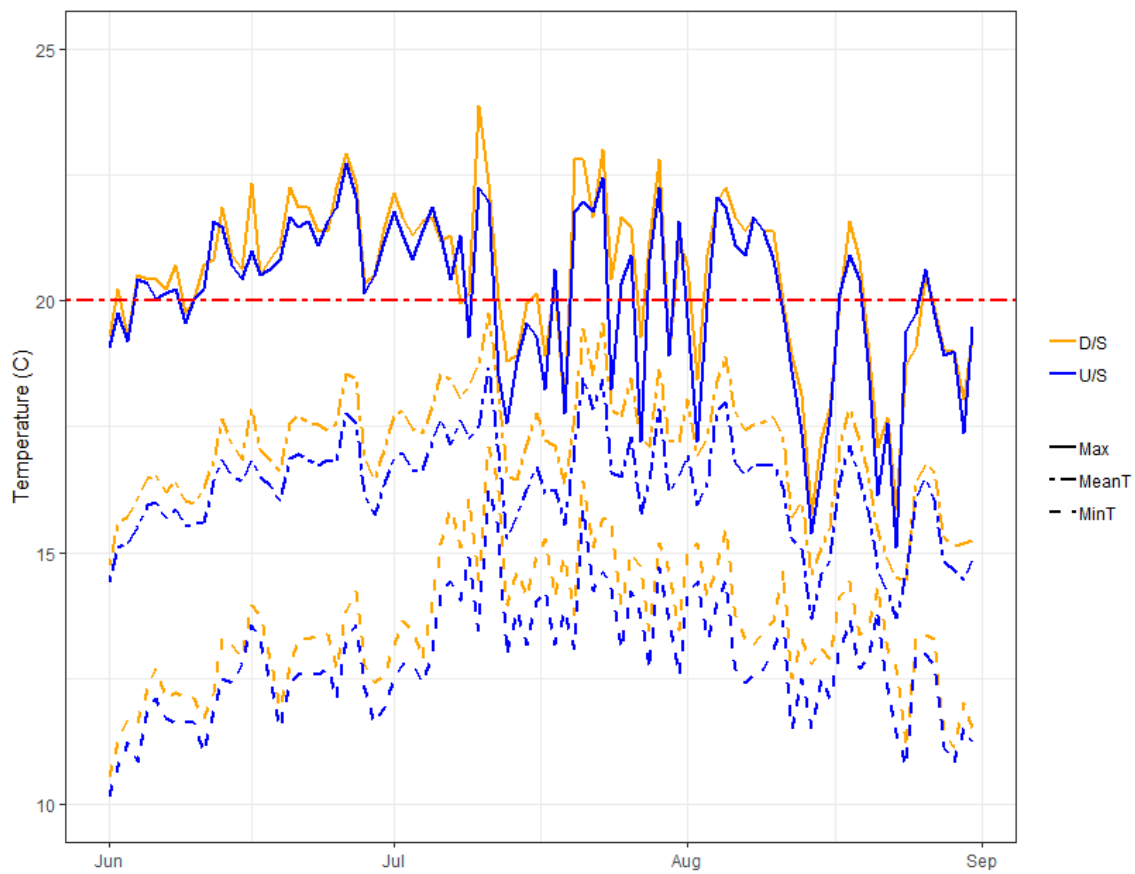

Figure 12. Daily maximum, mean, and minimum stream temperatures at the upstream and downstream sites for summer 2018. The red line indicates Utah's 3A standard for water temperature.

Seven-day rolling average maximum stream temperatures (7DADM) exceeded the 20 °C threshold 65 % of the summer at the upstream site and 74.4% of the summer at the downstream site (Figure 13). In addition, both sites had two extended, consecutive-day periods above the 7DADM threshold in June and half of July. The downstream site reached a maximum 7DADM of 21.9 °C twice—once in late June and again in late July. The upstream site reached a maximum 7DADM of 21.7 °C in late June. Neither site reached the biological maximum of 22 °C.

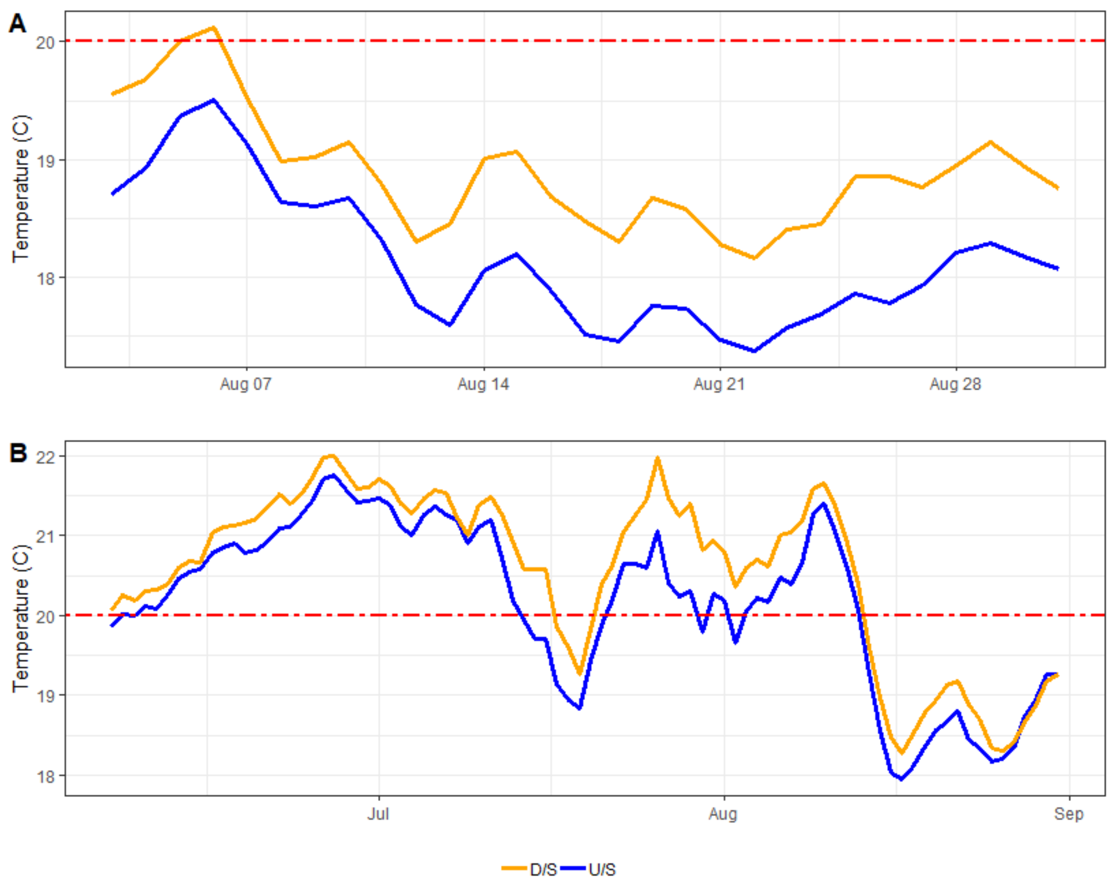

Figure 13. 7-day rolling average maximum stream temperatures for the upstream and downstream sites during (a) summer 2017 and (b) summer 2018. The red line indicates Utah's 3A standard.

Seven-day rolling average daily average stream temperature (7DADA) at the downstream site was, on average, 1 °C warmer than the upstream site, as the river heated longitudinally through summer (Figure 14). Neither site violated the 19 °C threshold for rainbow trout during summer 2018. The upstream site reached a maximum average weekly average temperature of 17.6 °C and the downstream site 18.6 °C—both in early July.

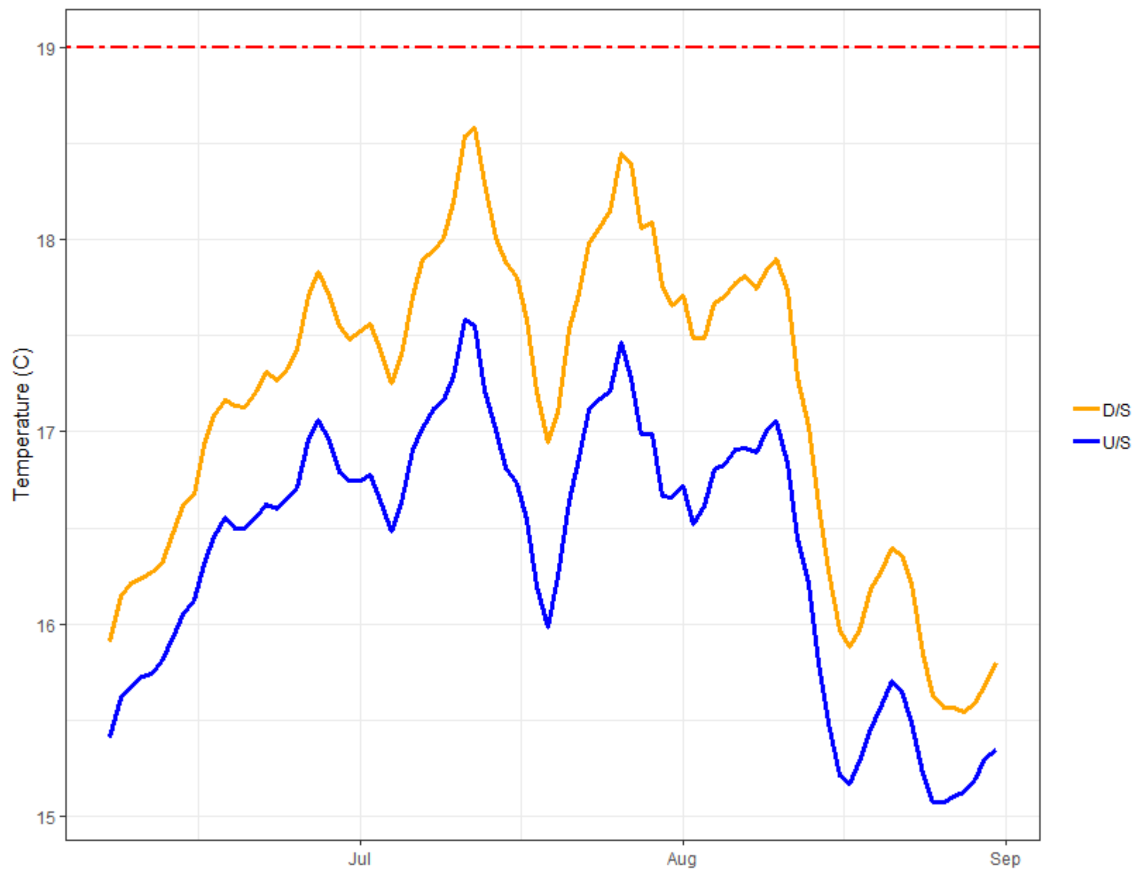

Figure 14. 7DADA results for upstream and downstream sites during summer 2018. The red line indicates the rainbow trout biological upper threshold.

<u>Turbidity</u>

Turbidity generally followed fluctuations in streamflow with larger discharge events driving turbidity peaks with a falling limb as streamflow subsided (Figure 15, and Figure 16a,d,e for detail). Both figures above have their left y-axes cropped and the full extent of turbidity measurements is shown in Figure A10. The downstream site was almost always more turbid, and more variable, than the upstream site (Figure 15).

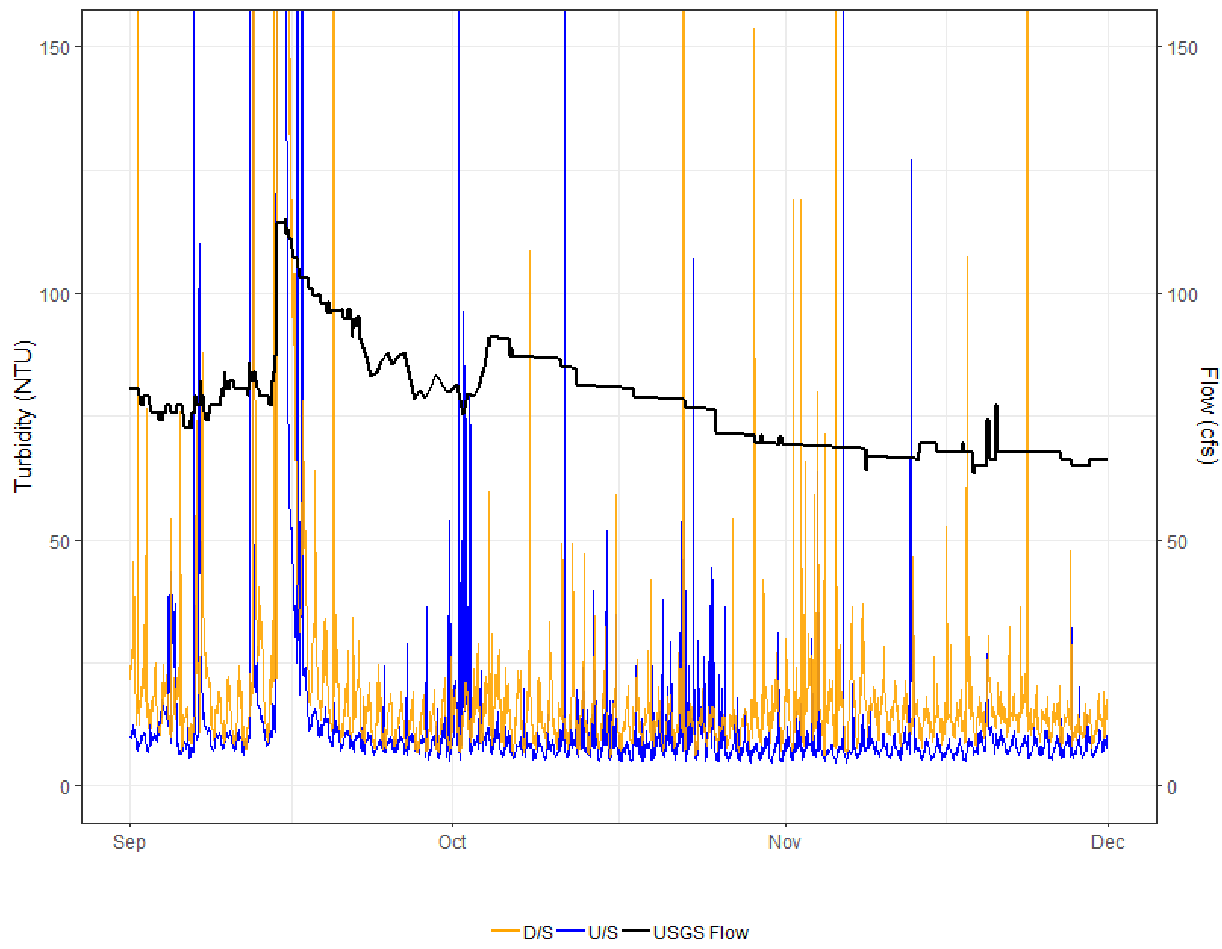

Figure 15. Hourly Sevier River turbidity at upstream (U/S) and downstream (D/S) monitoring sites with streamflow at USGS gage near Hatch.

Turbidity at the downstream site was substantially flashier than the upstream site during both summer periods (Figure 16a,e). Counterintuitively, turbidity steadily rose at both sites from ~10 NTU to ~25 NTU during winter 2018 (Figure 16c) as streamflow dropped steadily from 65 cfs to 50 cfs by end of season. In addition, two periods of seemingly unprompted elevated turbidity occurred at the upstream site in mid-March 2018 and early October 2018 (Figure 16d,f). The mid-March event was a much larger magnitude than the October event.

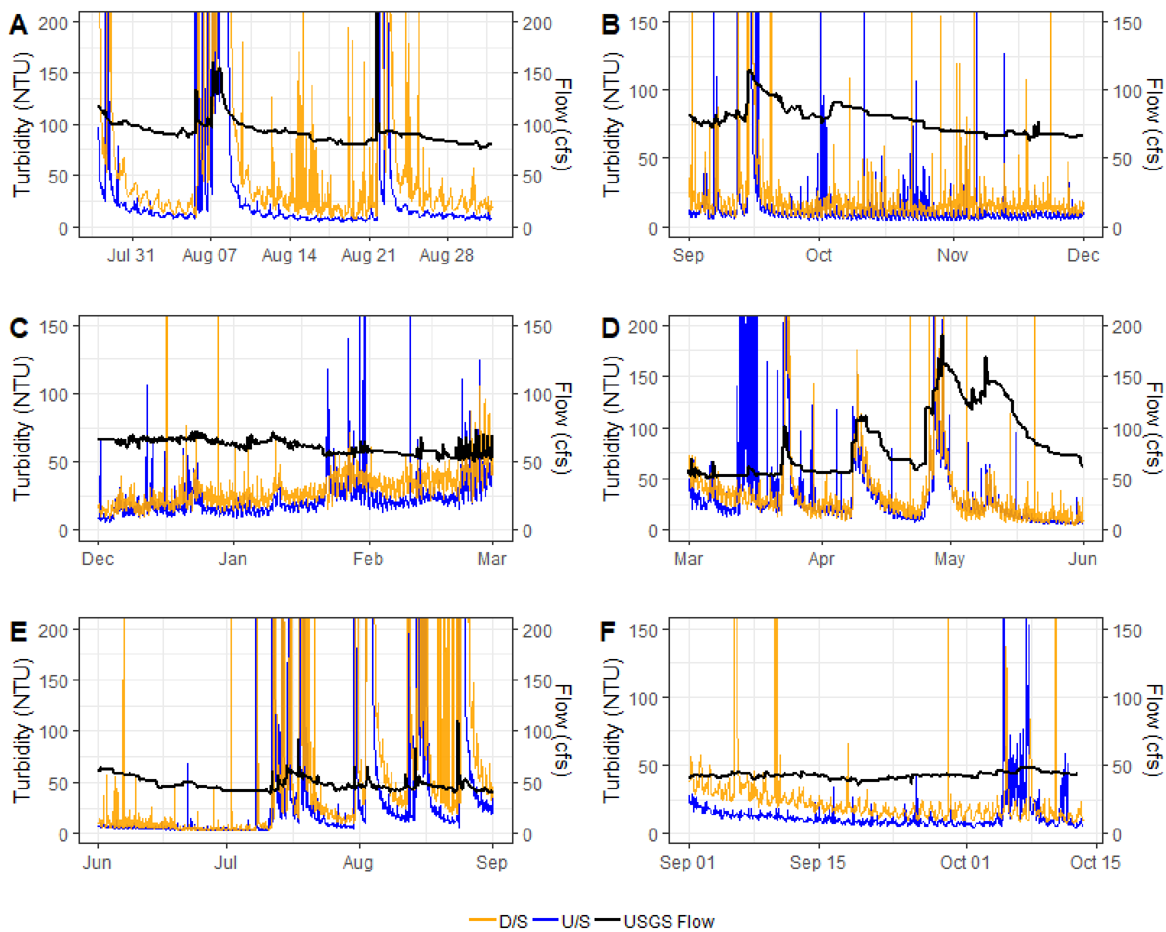

Figure 16. Hourly Sevier River turbidity at upstream (U/S) and downstream (D/S) monitoring sites with streamflow at USGS gage near Hatch for (a) summer 2017, (b) fall 2017, (c) winter 2018, (d) spring 2018, (e) summer 2018, and (f) fall 2018. Note that the y-axis changes between panels.

In all seasons except Spring 2018, the downstream site was indeed more turbid than the upstream site (shown as positive values in Figure 17). If the error bars include zero, there was no statistical difference between the two datasets—which only occurred during spring 2018. Over the full period, the downstream site was, on average, 13.2 NTUs more turbid than the upstream site (recall the turbidity thresholds of 20, 50, and 160 NTUs). Summer was the season with the largest discrepancy between upstream and downstream sites at more than 30 NTUs difference during both 2017 and 2018. Confidence intervals were also larger during summer. In addition, the downstream site was more turbid in fall and winter by roughly 7 NTUs.

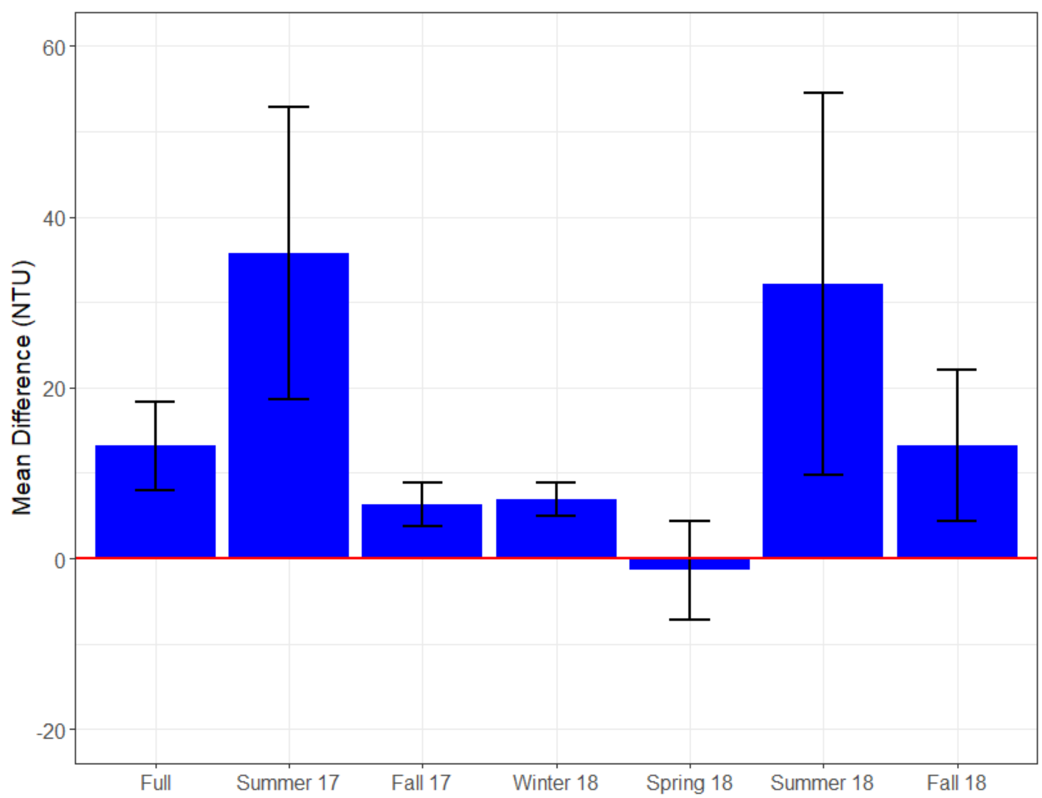

Figure 17. Mean of differences between daily mean turbidity at downstream and upstream sites for full observational period and seasons. Error bars show 95% confidence intervals.

The downstream sites violated the 20 NTU turbidity standard more often than the upstream site (Figure 18a). Over the entire period of observation, the downstream and upstream sites violated the 20 NTU threshold 58.8% and 37.6% of the time respectively. The sites violated the threshold most often during winter 2018 (downstream, 86.7%) and spring 2018 (upstream, 64.1%). Fall 2017 was the season with the least violations at both sites. These results suggest that both sites are violating the 20 NTU threshold although the downstream site was substantially worse.

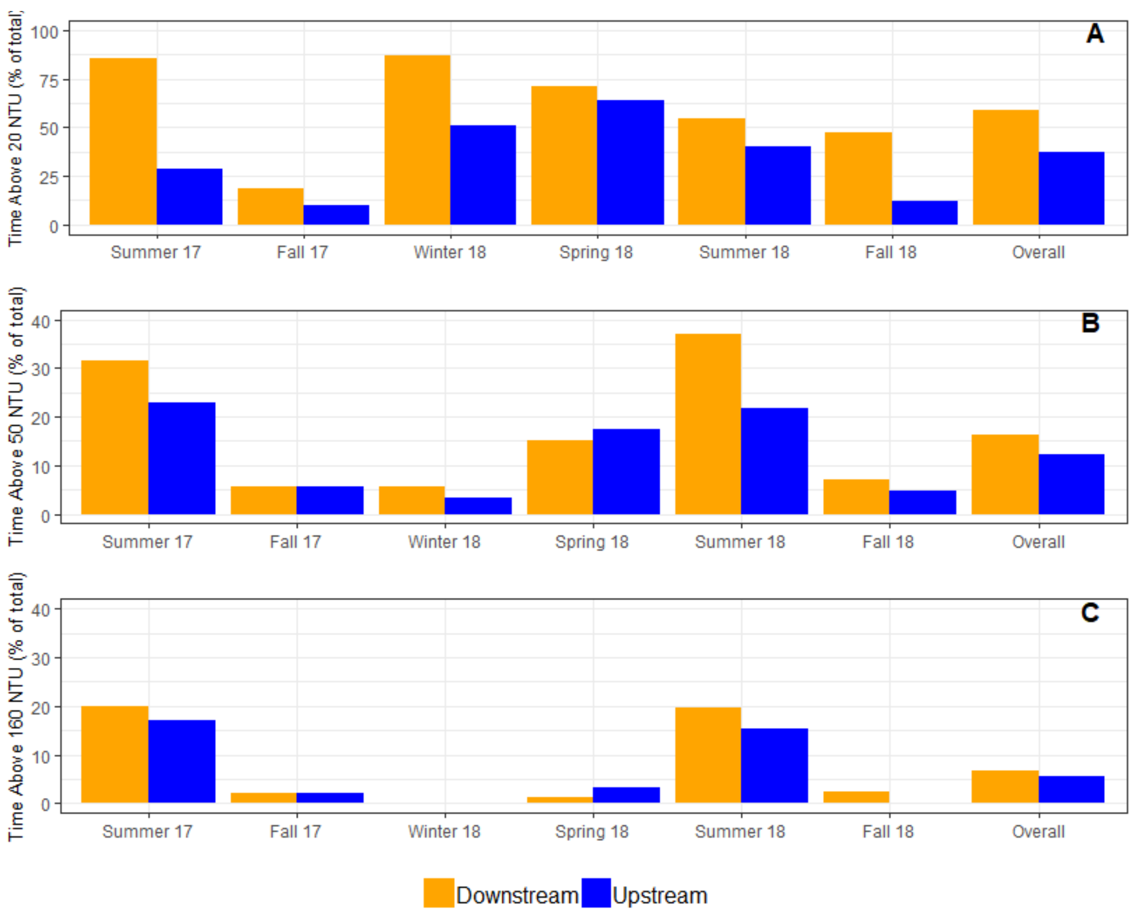

Figure 18. Daily mean turbidity threshold violations as fraction of total days for upstream and downstream sites by season. Turbidity thresholds are (a) 20 NTU, (b) 50 NTU, and (c) 160 NTU. Note that the y-axis of (b) and (c) differs from (a).

Both sites violated the 50 NTU threshold at least one day in each season (Figure 18b). This threshold was an absolute, instantaneous threshold recommended by Idaho DEQ so a single violation would constitute an impairment. The downstream and upstream site violated the 50 NTU threshold 16.3% and 12.2% of the time respectively. Most of the 50 NTU violations occurred in summer 2018 and summer 2017 at 31.4% and 22.8% for downstream and upstream respectively. Winter 2018 violated the 20 NTU threshold at least 50% of the time, but violations of the 50 NTU threshold during the same season were under 6% for both sites. This suggests that the baseline turbidity was higher in winter than other seasons, but that winter had fewer turbidity spikes (Figure 16). Upstream violations exceed downstream violations during just one time period: spring 2018.

Violations of the 160 NTU occurred 19.6% of the time for the downstream site and 15.2% at the upstream site during summer 2018 (Figure 16c). Violations of the 160 NTU turbidity threshold were rare in fall, winter, and spring. The overall frequency of violations was 6.6% and 5.7% at downstream and upstream respectively.

Sediment Flux Analysis

<u>Cut Bank Delineation Uncertainty</u>

The overall uncertainty associated with digitizing stream cut banks was about ±20% (Table 5). The values in the “Original”, “Maximum”, and “Minimum” columns are the total volumes of eroded mud over the corresponding time period for all resamples sites. The maximum relative difference values were consistent across photo time periods with an average of 24.1%. The minimum relative differences values were less consistent but resulted in a similar value of -22.7%. Individual polygon discrepancies ranged from

65% smaller to 100% larger. Discrepancies between cut bank polygon digitization are visualized in Figure 19. Here, the upstream minimum polygon was ~14% smaller than the original, while the upstream maximum polygon was 1% larger than my initial delineation. Full results for this example are shown in Table A3. These digitizing uncertainties are the smaller than the overall uncertainty, but they nonetheless illustrate the results of this analysis.

Table 5. Cut bank digitizing uncertainty with minimum and maximum volumes and percentage change compared to my initial analysis. This includes five photos and three time periods.

	Total Mud Volume (m3)				
Photo Year	Minimum	Original	Maximum	Max Relative Diff (%)	Min Relative Diff (%)
1974 - 2004	22,284	25,985	32,552	25.3	-14.2
2004 - 2006	6,522	9,811	12,154	23.9	-33.5
2009 - 2014	4,297	5,399	6,656	23.3	-20.4
			Average	24.1	-22.7

Effect of Streambank Restoration on Cut Bank Erosion

There was less streambank erosion in the post-restoration time period than in the pre-restoration time period across all reaches except Panguitch (Table 6). I included the results for the 2006 – 2009 period in Table 6, although this was the period when restoration was completed so these results are not discussed further. The largest erosion rates occurred during 2004 – 2006, the period with a major streamflow event. Of course, streamflow also changed during the post-restoration period, thus the “never restored”

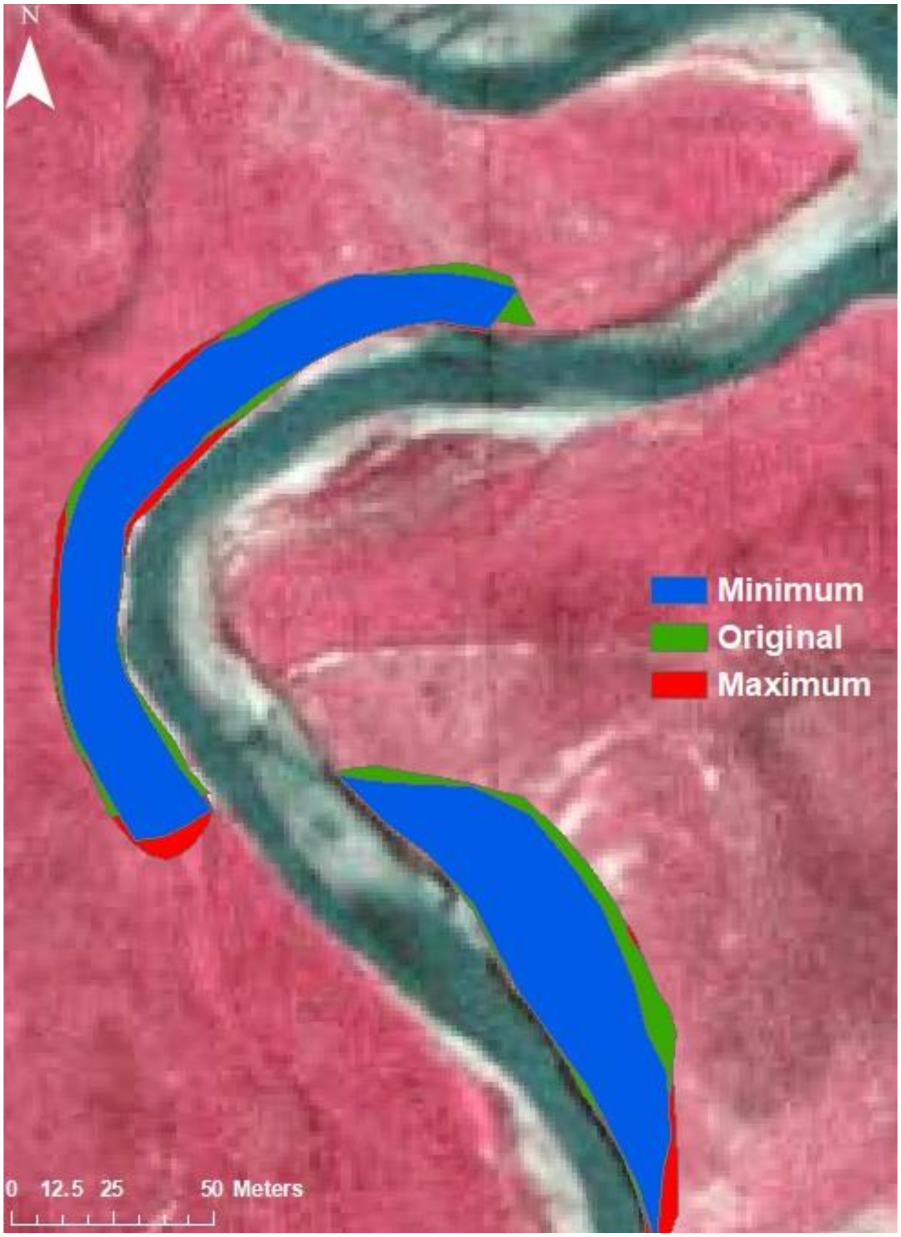

Figure 19. Cut bank resample examples with initial analysis and maximum and minimum extents. Flow direction is North.

reach acted as a control. Ratios of erosion in the post-restoration period versus erosion in the pre-restoration period are given in Table 7. Erosion rates dropped just 10% (far right column) in the control reach while falling, on average, 55% in the restored reaches. Thissuggests that erosion in the restored reaches decreased beyond the background rate.

Table 6. Volume of cut bank eroded by restoration status and time period. Values are in cubic meters of mud per year per river kilometer.

	Pre-Restoration	Pre-Restoration	During Restoration	Post-Restoration
Restoration Status	1974 to 2004	2004 to 2006	2006 to 2009	2009 to 2014
Never Restored	174	1,033	175	157
Restored 2007	398	2,092	210	138
Restored 2008	276	871	182	144
Restored 2009	230	620	162	112
Avg Restored	302	1,194	185	131
Panguitch	251	1,324	223	483

Table 7. Post- to pre-restoration erosion rate ratios. Values <1 mean erosion decreased in the post-restoration period and values >1 mean erosion increased in the post-restoration period. This table shows results with and without the flood event of 2005 in the pre-restoration time period.

Restoration Status	2009-2014/1974-2006 Ratio (with 2005 flood)	Restoration Status	2009-2014/1974-2004 (without 2005 flood)
Never Restored	0.69	Never Restored	0.90
Restored 2007	0.27	Restored 2007	0.35
Restored 2008	0.46	Restored 2008	0.52
Restored 2009	0.44	Restored 2009	0.49
Avg Restored	0.39	Avg Restored	0.45
Panguitch	1.52	Panguitch	1.93

The 2005 flood event produced 5 to 10 times more erosion than any other measured period. The values second column from the left (Table 7) include the 2005 flood in the pre-restoration time period and results adjusted accordingly—all ratios decreased. Interestingly, erosion rates increased by 93% in Panguitch Valley from the 1974 – 2004 period to the 2009 – 2014 period.

Following restoration, cut bank erosion was reduced in all reaches (except Panguitch), and these declines are particularly evident when visualized with uncertainty in Figure 20. The red line represents the change in erosion in the control reach from pre- to post-restoration while the black line represents the average change in erosion from pre- to post-restoration for restored reaches. Figure A11 and Figure A12 include and demonstrate the effects of the 2005 streamflow event. When including the flood event, in decline in erosion rates appears even more precipitous demonstrating the relatively large increase in erosion during the 2004 – 2006 time period.

<u>Extrapolating Cut Bank Erosion for Panguitch Valley</u>

Linear regression results suggest that the restored reaches eroded substantially less bank volume per erodible bank length than the unrestored reaches in the post-restoration era (Figure 21). I used an in-between value of 0.85 m3/year cut bank erosion per meter of erodible bank to calculate the erosion volumes for the unknown, unrestored reaches in Panguitch Valley. I chose this by finding a compromise between the regressions in Figure 21b,c and ignoring the regression on restored reaches in the post-restoration time period. I chose a value closer to the pre-restoration era regression because of its increased sample size and superior quality of fit.

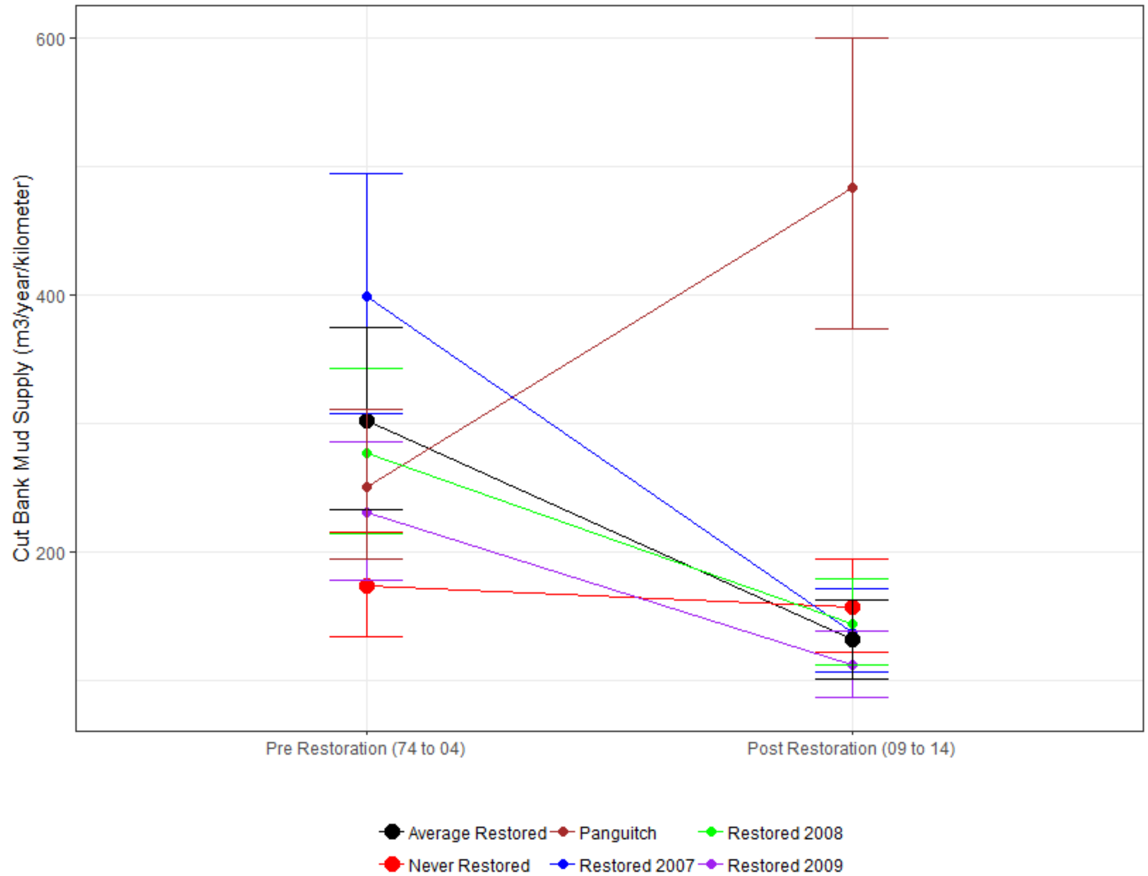

Figure 20. Aerial photo analysis results with uncertainty analysis results expressed as error bars.

The ratio of eroded bank volume to erodible bank length decreased in restored reaches by an average 0.16 from pre-restoration to post-restoration while the never-restored reaches in Hatch Valley ratio fell by just 0.02 (Table 8). Indeed, while length of erodible bank did not change, cut bank erosion decreased by roughly 400 m3/yr in the reach that was restored in 2008. Similarly, both erodible bank length and cut bank erosion decreased in the reach restored in 2009.

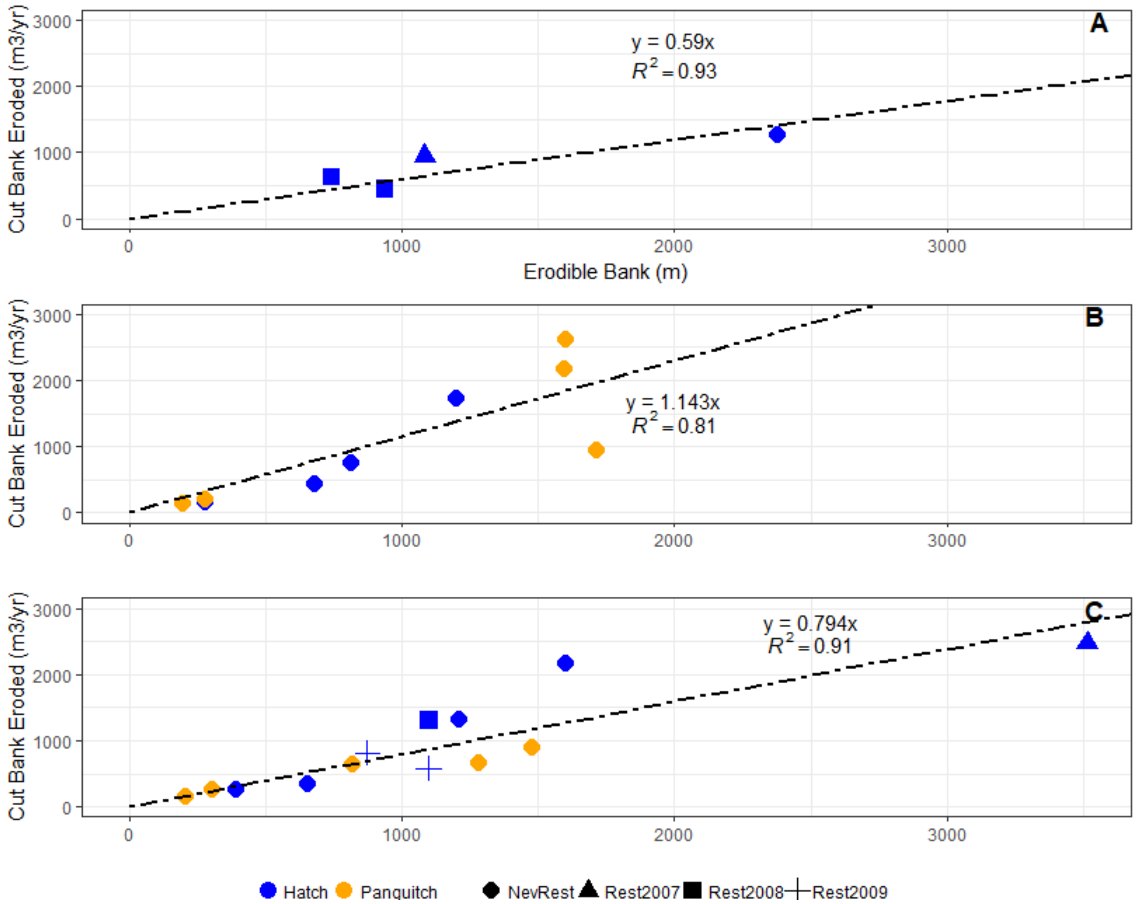

Figure 21. Linear regressions between erodible cut bank length and volume in (a) restored reaches in the post-restoration time period, (b) unrestored reaches in the post-restoration time period, and (c) all reaches in the pre-restoration time period including R2 and fit slope. Each point represents a reach and point shape shows reach restoration status.

<u>Sediment Budget Modeling with SIAM</u>

The difference between a reach's transport capacity and the sediment inputs from upstream and in-reach sources is called local balance. It varied greatly in this model application, from +200,000 m3 in a Hatch Valley reach to -600,000 m3 in Circleville Canyon. It is difficult to compare reaches using direct local balance values as they have different physical characteristics (e.g. length, width) and thus, the impact the local balance has on the reach varies. Instead, I calculated vertical bed movement (Figure 22b).

Table 8. Eroded cut bank volumes (m3) per erodible bank length (m) by reach for pre- and post-restoration periods. Differences indicates the change in this ratio between periods. Positive values mean the ratio increased post-restoration and vice versa. Averages for restored reaches, never restored, and never restored in Hatch are included.

		Pre-	Post-	
Valley	Status	Vol to Bank	Vol to Bank	Difference
Hatch	Never	0.53	0.93	0.40
Hatch	Never	1.09	0.65	-0.44
Hatch	2007	0.70	0.53	-0.18
Hatch	2008	1.19	0.87	-0.32
Hatch	Never	0.67	0.55	-0.12
Hatch	2009	0.53	0.47	-0.06
Hatch	2009	0.94	0.84	-0.10
Hatch	Never	1.35	1.43	0.08
Panguitch	Never	0.78	1.36	0.58
Panguitch	Never	0.71	0.69	-0.03
Panguitch	Never	0.61	1.63	1.03
Panguitch	Never	0.85	0.73	-0.12
Panguitch	Never	0.52	0.55	0.03
			Avg Restored	-0.16
			Avg Never	0.16
			Avg Nev Hatch	-0.02

Vertical bed movement describes what would happen to the stream's bed elevation given the calculated evacuated or stored sediment (local balance) assuming unchanging channel width and reach length. One general trend is that reaches with surplus sediment tended to immediately follow reaches with sediment deficits.

The reaches with labels 278, 238, and 235 made the up the stretch of the Sevier near Hatch that I monitored in my field data analysis (Figure 22b). My monitoring sites bracketed three reaches (from upstream to downstream): unrestored (2 rkm), restored (1.8 rkm), and restored (5.1 rkm). Vertical bed movement for these reaches was -3.8 m, +0.5 m, and +2 m, respectively. These SIAM values suggest these reaches were: severely degrading, minor aggradation, and severely aggrading respectively.

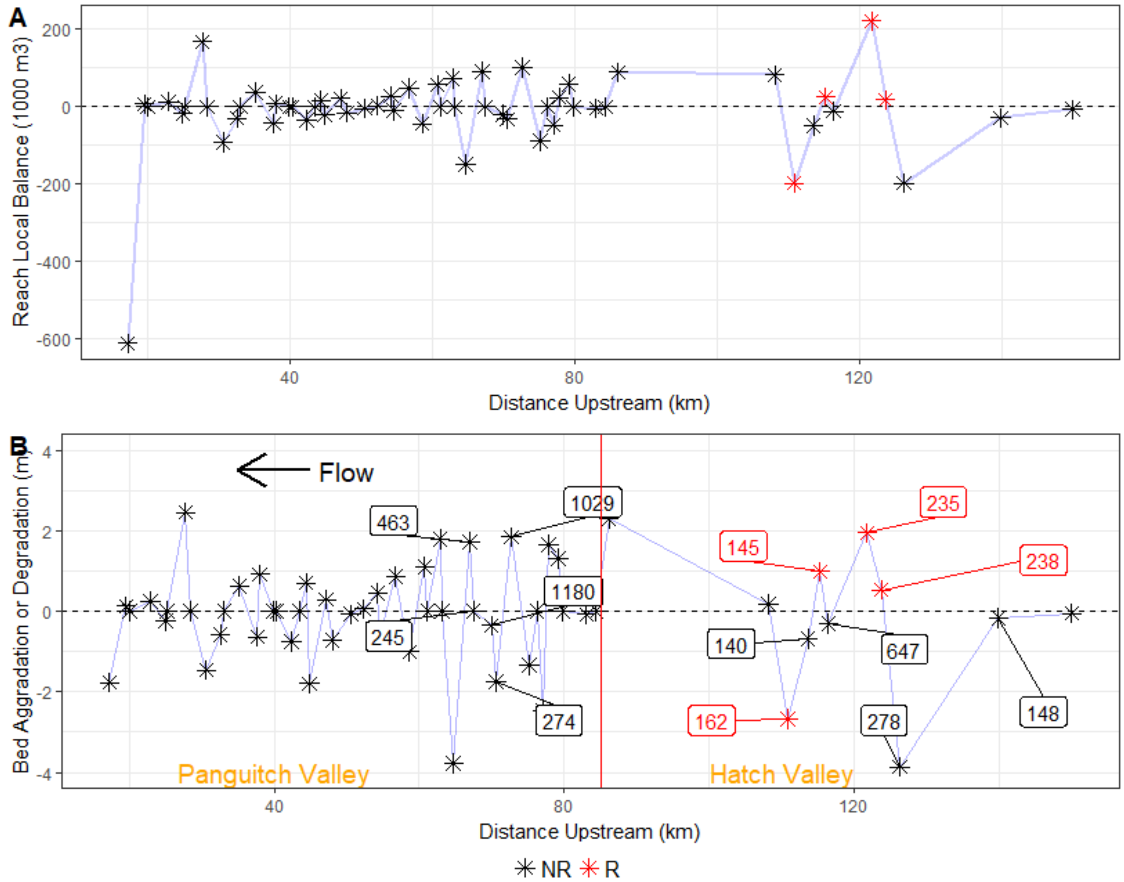

Figure 22. (a) Local balance per year and (b) bed aggradation/degradation per year against distance upstream of Circleville Canyon. The red line distinguishes valleys. Red points are reaches that were restored (as of 2014) and black points were never restored. Value labels are measured cut bank movement (m3/km/year total sediment) in each reach in the post-restoration era from aerial photo analysis

I also examined the relationship between vertical bed movement and cut bank erosion rates (from my historical photo analysis), but I did not find a clear correlation between them (Figure 23). It appeared that there may have been a weak relationship between them. That is, cut bank erosion was slower (<300 m3/km/year) during bed degradation and faster during aggradation (>300 m3/km/year). Unfortunately, R2 was low at just 0.11. Including the estimated (from the erodible bank extrapolation) cut bank movement makes the conclusion even less clear (R2 = 0.02). Reach transport capacity

and reach gradient followed each other closely suggesting gradient is likely a major driver for this model (Figure 24). I also examined the relationship between cut bank movement, and both reach transport capacity and reach gradient but did not find a significant correlations (Figure A14, Figure A15).

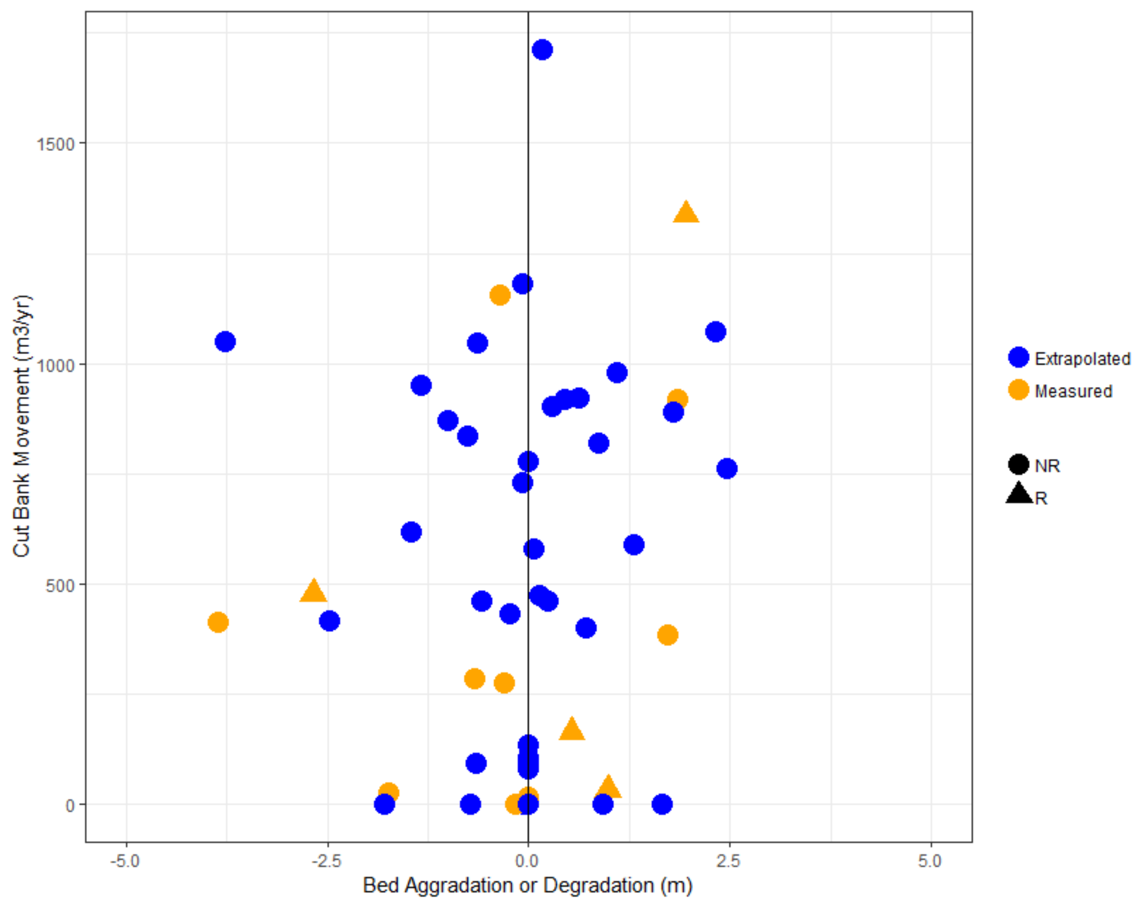

Figure 23. Vertical streambed movement compared to cut bank movement for both measured and extrapolated cut banks. Point shape indicates restored (R) or never restored (NR). Each point is a reach in SIAM.

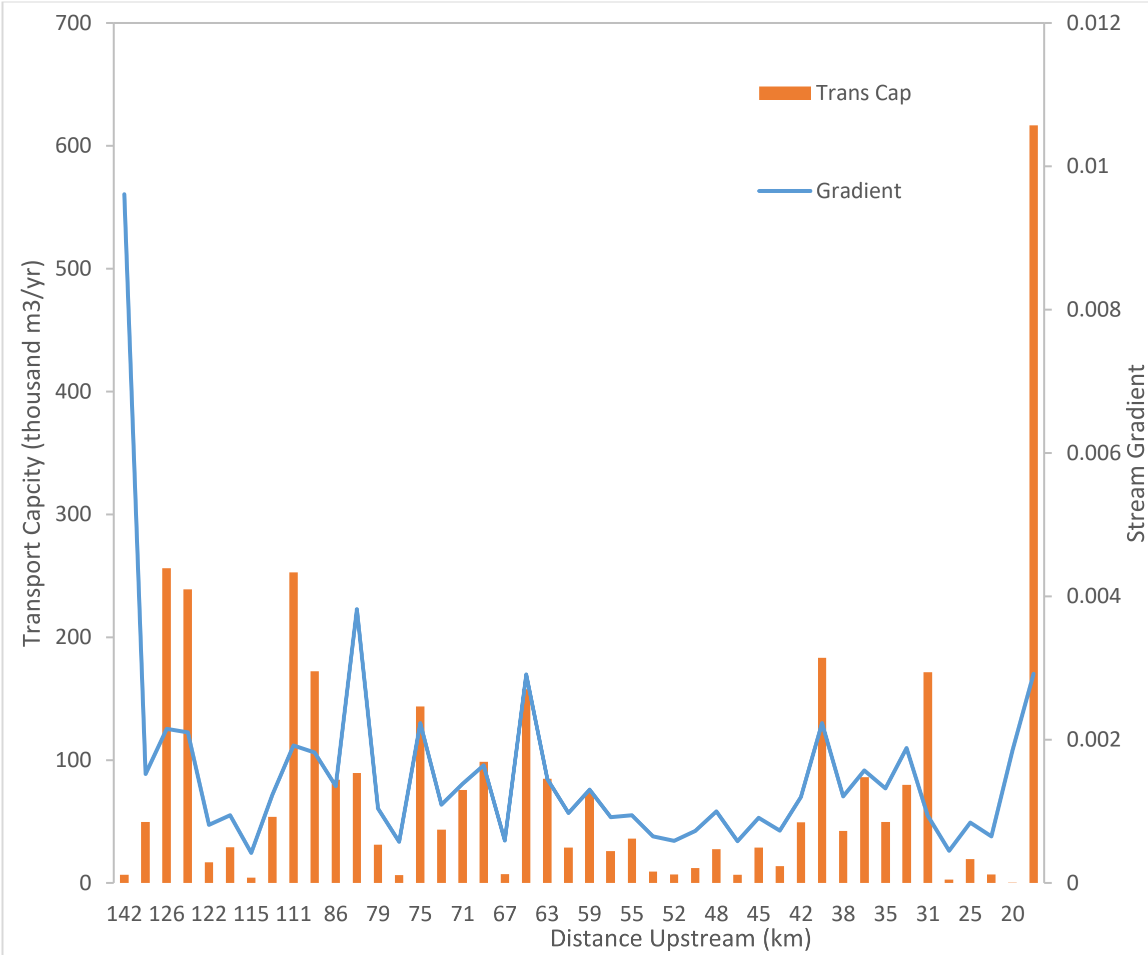

Figure 24. Sevier River reach transport capacity and slope from SIAM. River flows from left to right in figure.

DISCUSSION

Major Findings and Discussion

Water Quality of the Upper Sevier River Near Hatch, UT

Stream temperatures warmed longitudinally through the Sevier River study reach, although the downstream site was less than 1°C warmer than the upstream site. Both maximum daily temperatures and 7DADM violated Utah’s class 3A stream temperature standards during summer 2018, and both sites came within 0.1-0.2°C violating biological standards but ultimately did not exceed them. Likewise, 7DADA came within 1-2°C of violating the biological threshold for rainbow trout but did not. This finding suggests that restoration practitioners should reduce daily maximum stream temperatures through restoration (WRI, 2016). Given that another goal of this restoration was improving biological physical habitat (e.g. creating pools (WRI, 2016)), it is likely that cooler temperatures persist in deeper pools thus stream temperatures at both sites can support brown and rainbow trout. During periods of high temperatures, salmonids utilize thermal refugia—areas where stream temperatures remain cool (Elliott, 2000; Torgersen et al., 1999; Dzara et al., 2019). It is also important to note that summer 2018 was a low flow season. Low flows contribute to warm stream temperatures due to reduced assimilative heat capacity of the river, so this section of the Sevier River may have cooler stream temperatures in higher flow years.

If Sevier River stream temperatures violated the 20 °C state threshold and neared biological standards near Hatch, it may be difficult to establish a cold-water fishery ~40 km downstream at Circleville Canyon. The river must traverse Panguitch Valley where flow is diverted, used for irrigation, and a portion of it is returned. Water diversions have

the potential to warm stream temperatures a great deal (Elmore et al., 2016) . Indeed, stream temperatures increases related to irrigation have been measured in Panguitch Valley (Kevin Heaton, personal communication, 2018). Of course, if restoration efforts successfully establish woody riparian vegetation the rate of longitudinal heating could be slowed (Roth et al., 2010; LeBlanc and Brown, 2000). Lastly, the Upper Sevier River was not listed as impaired for temperature during the 2004 TMDL, but given these findings, restoration practitioners should monitor it closely.

Turbidity monitoring suggested that overall, the downstream site was 13 NTUs more turbid than the upstream site, but, while the downstream site violated the turbidity standards more often, both sites spent more than roughly 40% of the observed time period above the 20 NTU thresholds. Likewise, both sites violated the 50 and 160 NTU thresholds frequently during spring and summer—critical seasons when aquatic biota may already be stressed by warm stream temperatures and low flows. Spring 2018 was the only time period during which turbidity was lower at the downstream site than the upstream site. This was likely a result of the anomalous, elevated turbidity measurements at the upstream site during mid-March. These and other seemingly unprompted turbidity spikes may have been caused by nearby instream activity. Cattle were allowed full stream access at the upstream site where I witnessed long term grazing but were confined to a single, nearby cattle lane at the downstream site. In addition, fishing is common at both sites, but the downstream site was near an RV park where fishing is actively encouraged. Both activities disrupt the stream bed and increase turbidity.

In general, these turbidity findings suggest that the completed restoration did not improve stream turbidity, but that may not be the case. Since I did not measure turbidity

at these sites before and after restoration, it is impossible to know whether these results show a degradation or improvement. This restoration could have reduced the turbidity increases associated with bank degradation, but there may be a natural, background turbidity increase occurring downstream. In addition, there are turbidity drivers that are likely enhanced by agricultural degradation that this restoration did not address. For example, surface runoff across upland and riparian agricultural land has been shown to increase stream turbidity (S. Li et al., 2008). Indeed, USFS (2004) estimated that the stream bank erosion constitutes 41% of yearly suspended solid load with 59% from upland erosion. Even if restoration improved turbidity, the upstream site frequently violated turbidity standards. Perhaps restoration practitioners should examine the upstream (and upland) sources of fine sediments in conjunction with downstream restoration.

Given the limitations of this study design, it is difficult to conclude whether restoration altered stream temperatures and turbidity. While establishing woody riparian vegetation can indeed cool stream temperatures (Roth et al., 2010), restoration practitioners were unable to do so thus any temperatures changes would have relied on subsequent changes in channel morphology. In addition, Meals (2001) found that this sort of restoration (e.g. cattle exclusion, streambank bioengineering) is capable of reducing fine sediment loads by 18-25% when completed on the watershed scale so it is possible that these restoration methods may need to be applied beyond the scale seen in this study to produce significant changes in water quality.

Effectiveness of Bank Stabilization on Cut Bank Erosion

Aerial photo analysis results suggest that the streambank restoration on the Upper Sevier River in Hatch Valley decreased cut bank erosion rates by 45% compared to control reaches. While applications of bioengineering techniques like those applied on the Upper Sevier are common and often recommended over hard stream engineering (Li and Eddleman, 2002), their effectiveness on erosion reduction has rarely been quantified. Simon et al. (2011) used a bank erosion model called BSTEM to simulate the effects of bank toe protection. They estimated that adding toe protection to bare eroding banks on the Big Sioux River in South Dakota could reduce annual stream bank sediment loads by 51%. In addition, Dave and Mittelstet, (2017) found that rock vanes (one of the techniques applied on the Upper Sevier River) applied on the banks of Nebraska's Cedar River reduced streambank erosion rates by 47.2%. While neither referenced stream is semi-arid nor located in the great basin, the results are comparable to those of this study. In addition, my aerial photo analysis uncertainty findings of roughly ±20% were comparable with similar studies. Day et al. (2013) calculated an uncertainty of 11-20% when using aerial photographs to assess bluff erosion. Uncertainty analyses improve result integrity and are rarely performed but are an asset for decision makers.

Although restoration greatly reduced cut bank erosion, it did not stop it completely. This is important because a common technique to improve degraded stream banks is to armor them with rip rap which completely prevents further bank erosion. Of course, riparian and aquatic ecological health depends upon stream dynamism (Richards et al., 2002). Reversing the results of degradation by encouraging stream bank vegetation

regrowth is an ecologically preferable method. These results suggest restoration greatly reduced cut bank erosion, but the banks are still actively eroding as a natural system does.

Interestingly, I found that cut bank erosion increased in Panguitch Valley. As there is no flow gaging of the Sevier River in Panguitch Valley, it is difficult to know whether the flows that occurred near Hatch propagated through to Panguitch. In addition, there are a few intermittent tributaries flowing from the Paunsaugunt Plateau to the Sevier River's east. The most notable of these tributaries occurs ~20 kilometers downstream of my aerial photo study area and follows Highway 12 out of the Red Canyon area. This tributary's flow is not measured and affects Panguitch Valley independently of Hatch Valley. The tributary could have contributed a large flow during my post-restoration period that did not affect my Hatch study area. In addition, surface water diversions in Panguitch Valley, which are more extensive than those in Hatch Valley, affect streamflow and water management may have changed over the time periods analyzed here.

Future Streambank Restoration

This research is intended to aid decision making for future streambank restoration of this type. Results showed that the streambank restoration performed on the Upper Sevier River near Hatch decreased cut bank erosion rates. Water quality results showed that stream temperatures and turbidity generally degraded downstream and the effects of restoration could not be disentangled from longitudinal changes. I recommend that future streambank restoration be monitored upstream and downstream, and before and after restoration is implemented. Restoration practitioners should also consider lag times of the effects restoration practices—particularly with respect to water quality (Meals et al.,

2010). Additionally, other sources of fine sediment should be quantified and, where possible, actively monitored. It may also be the case that the reach scale at which this streambank restoration occurred was simply not extensive enough to improve stream turbidity. If that is the case, predictive modeling is the ideal tool to upscale and examine the effectiveness of streambank restoration in different locations.

Unfortunately, my application of SIAM was insufficient but more comprehensive sediment modeling would be useful for restoration work of this type. I was unable to find any significant relationships between several SIAM variables, but the characteristics of the reaches I monitored may help explain the discrepancy between my turbidity and aerial photo analysis findings. The first reach below my upstream monitoring site was severely degrading and the next two reaches were aggrading. In reality, the initial reach is straight with high water velocity, so it is unlikely that any fines deposit. It is possible that the reach immediately below my upstream site was not storing its "fair share" of sediment. That is, there was considerably more fine sediment in the reach just upstream of my downstream site because the 2 rkm reach upstream was passing fine sediment through instead of storing it. This would result in more fine bed sediments at the downstream monitoring site and higher turbidity than might otherwise be expected. Nevertheless, while a more comprehensive sediment model would require substantially more data than my SIAM model, one can easily the see the value in being able to predict the effects of restoration to inform future decisions. Indeed, Enlow et al. (2018) recently developed a framework to evaluate streambank stabilization techniques that utilizes a suite of process-based models. Their framework includes setting quantifiable objectives, running simulation models, and making estimates of cost-effectiveness.

In addition, there may be a secondary use for the results of my cut bank erosion extrapolation analysis to inform future restoration. I performed a partial historical aerial photo analysis for Panguitch Valley but also made estimates of erosion throughout the remaining unknown reaches. As restoration moves downstream, these results could be used by practitioners to prioritize problematic, rapidly eroding reaches.

Finally, the effects of climate change on stream water quality in the intermountain west are not well studied, but should be considered when determining the location and extent of restoration projects. Ficklin et al. (2018) showed that natural and managed watersheds display similar responses to recent climate change, and van Vliet et al. (2013) projected a nationwide 1 to 2 °C average increase in maximum and mean global stream temperatures by the end of the century. In addition, Whitehead et al. (2009) suggested that low summer flows and higher stream residence times will result in lower turbidity levels, but it has also been predicted that precipitation will shift from snowmelt to rainfall (Adam et al., 2009), leading to increased erosion and transport of fine sediments via surface runoff.

Limitations

The Upper Sevier River was listed as impaired for suspended sediments for which I used turbidity as a surrogate because it is easier to measure continuously. Turbidity is a measure of the effects of suspended sediment but there is no universal, direct relationship between them (Bash et al., 2001). Developing a site-specific relationship between turbidity and suspended solids would improve the credibility of this analysis. In addition, I was unable to relate turbidity and streamflow at each site because of the poor quality of

my stage-discharge rating curve. Directly comparing turbidity requires the assumption that streamflow is the same at both sites which may not have been the case.

The SIAM model also had several assumptions and limitations. First, SIAM performs steady state, one-dimensional computations, operates on a yearly time step, applies reach-averaged hydraulic and physical properties, assumes unlimited bed supply, and does not update channel geometry. Second, vertical bed movement of ±3 meters in a year is very unlikely. Chase and McCarthy (2012) observed <0.1 meter per year. Of course, holding stream width constant restricts lateral movement so any geometry changes were forced to occur vertically. It is also likely that the reaches where major bed degradation occurred had coarser bed material than the inputs I used. I applied uniform bed material estimates as I lacked detailed data. In addition, the SIAM model did not include all local supplies in each reach. A more comprehensive version would include upland erosion sources in addition to instream, eroding banks and would more accurately predict reach sediment balances.

Due to a lack of SIAM model input data, I made several simplifying assumptions during model construction. These included a surface level analysis of Manning's n selection, partially synthetic channel geometry, and the reduction of a five-year flow regime to a single, representative year. This modeling analysis was meant to examine broad patterns which local complexities may have obscured. Another large assumption I made is that streamflow propagates from the Hatch area through Panguitch Valley to Circleville canyon. Given the agricultural activity in Panguitch Valley, water withdrawals and returns are common.

CONCLUSIONS

Shifts in land cover from native vegetation to agriculture have led to riparian degradation, elevated stream temperatures, and turbidity impairments. In turn, these water quality degradations negatively impact aquatic biota such as fish. As a result, stream restoration is sometimes used to address water quality issues. This is precisely the case for Utah's Upper Sevier River. A 3A, cold-water fishery, the Upper Sevier was listed as impaired for fine sediments in 2004. Utah funded restoration projects near Hatch, UT to improve water quality and address the designation. However, the state has not adequately monitored the effects of the completed restoration. My research evaluated a restored reach to determine its effectiveness.

Using continuous monitoring data, I quantitatively evaluated stream temperature and turbidity upstream and downstream of completed restoration. Both sites 7-day average daily maximum temperatures violated the state of Utah's class 3A stream maximum temperature standard during summer 2018. On the contrary, temperatures did not violate the rainbow trout specific thresholds for maximum or mean daily temperatures. The average downstream temperature was roughly 1 °C warmer, but, as rivers naturally warm up downstream, stream temperature increase was expected. Managers should continue to monitor stream temperatures throughout the year to support a naturally reproducing sport population as the Upper Sevier River is currently stocked with sport fish. The upstream site was on average roughly 13 NTU more turbid than the downstream site, but both sites frequently violated state and biological turbidity standards. As the upstream site was quite turbid, I recommend examining and perhaps targeting upstream fine sediment sources in addition to future downstream restoration.

Monitoring showed that temperature and turbidity conditions at the downstream site were worse than the upstream site, but without a before-after analysis, they were inconclusive with regards to the effectiveness of restoration.

Historical aerial photo analysis results demonstrated that the streambank restoration techniques employed on the Sevier River near Hatch dramatically decreased cut bank erosion. These findings were in line with those of similar studies. I also developed a simple sediment budget model for the Sevier River from Asay Creek to Circleville Canyon and found that a more comprehensive sediment model is necessary to address bank erosion predictions. I recommended pursuing other predictive modeling to inform future restoration decision making.

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APPENDIX

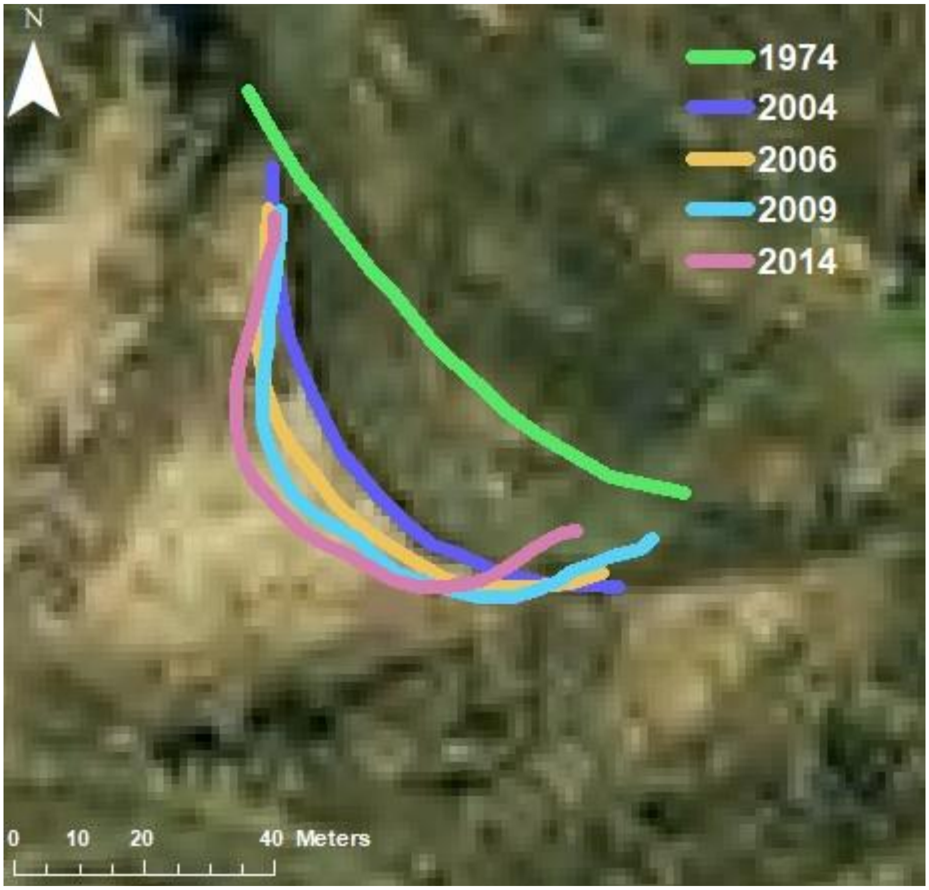

Figure A1. Cut bank delineation example with 2004 photo.

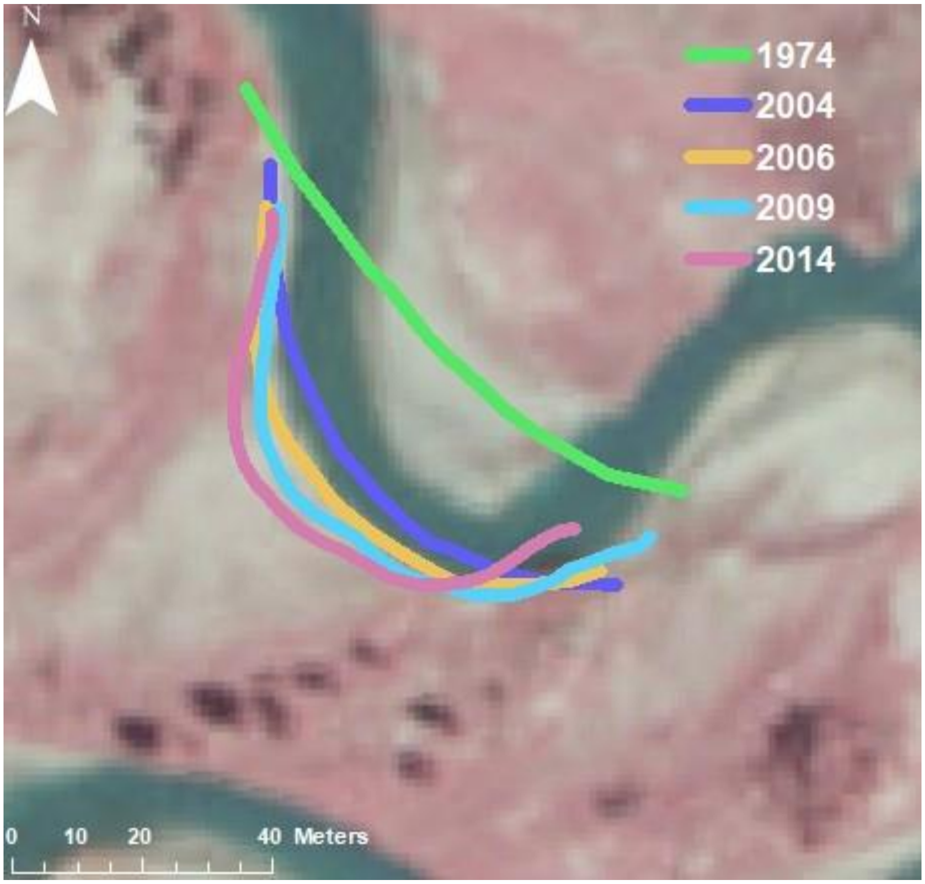

Figure A2. Cut bank delineation example with 2006 photo.

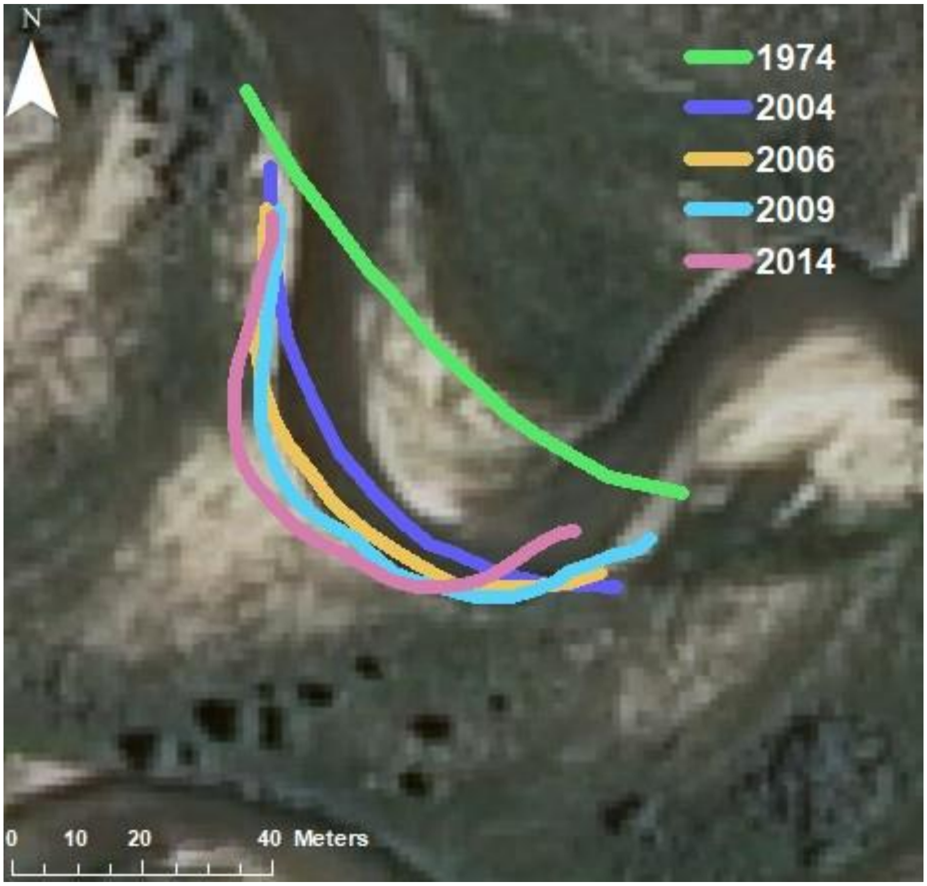

Figure A3. Cut bank delineation example with 2009 photo.

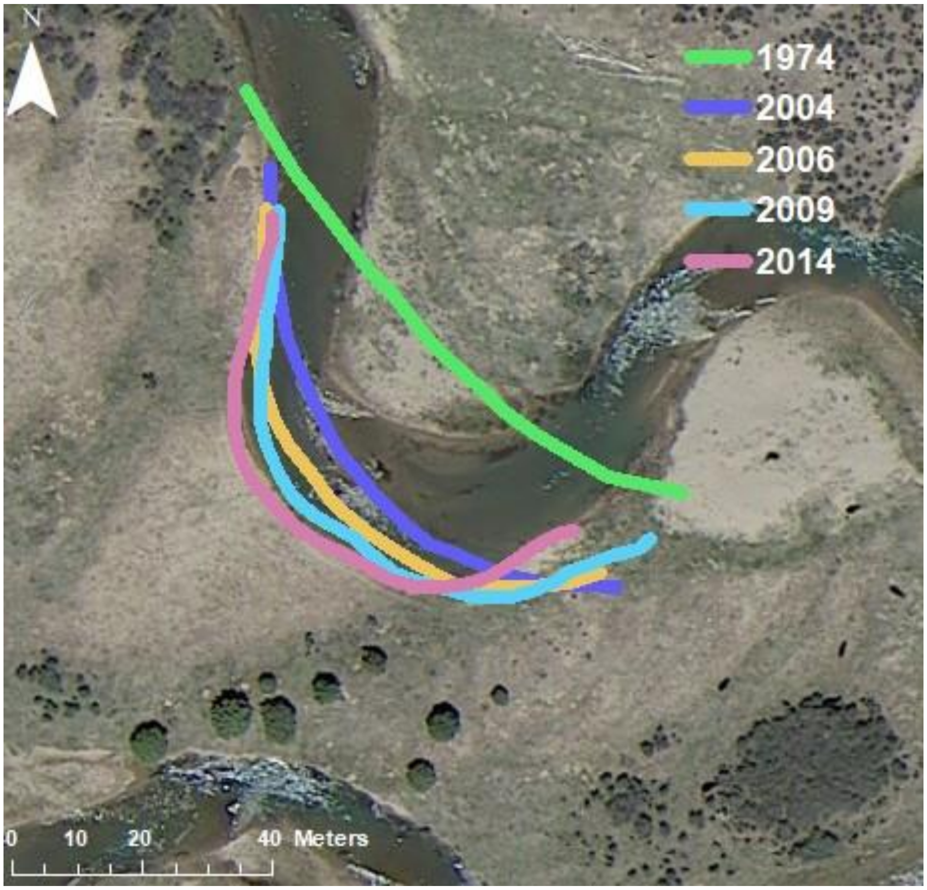

Figure A4. Cut bank delineation example with 2014 photo.

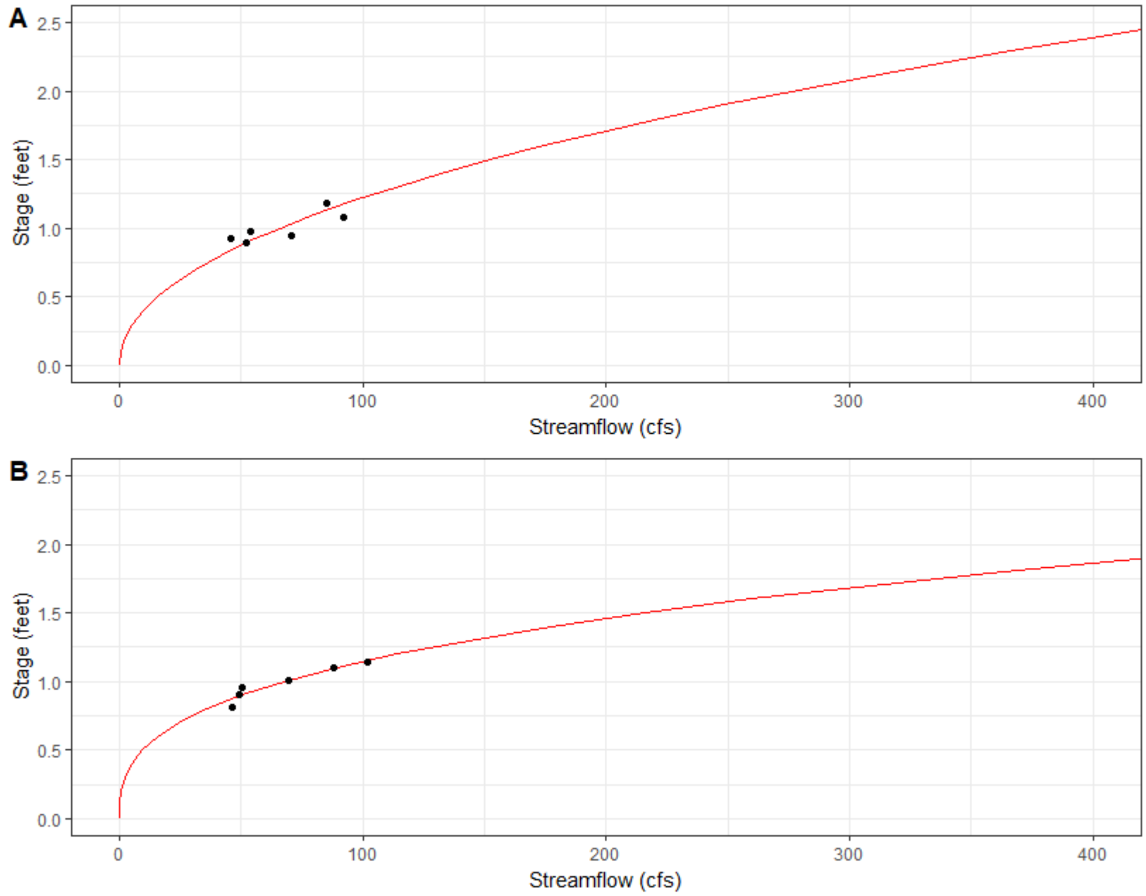

Figure A5. Sevier River rating curve with data points and nonlinear fitted regression for (a) upstream site (RMSE = 12.3 cfs) and (b) downstream site (RMSE = 6.6 cfs).

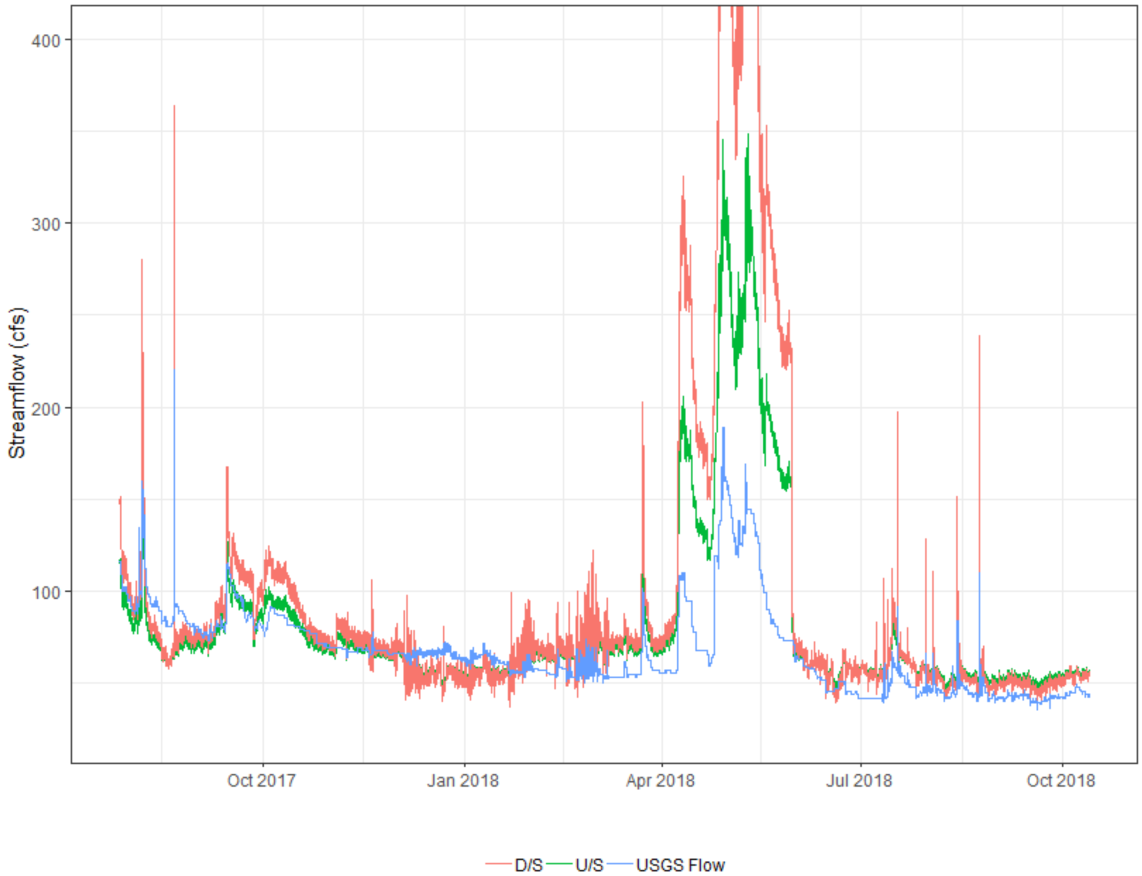

Figure A6. Sevier River streamflow at upstream site, downstream site, and USGS gage near Hatch, UT.

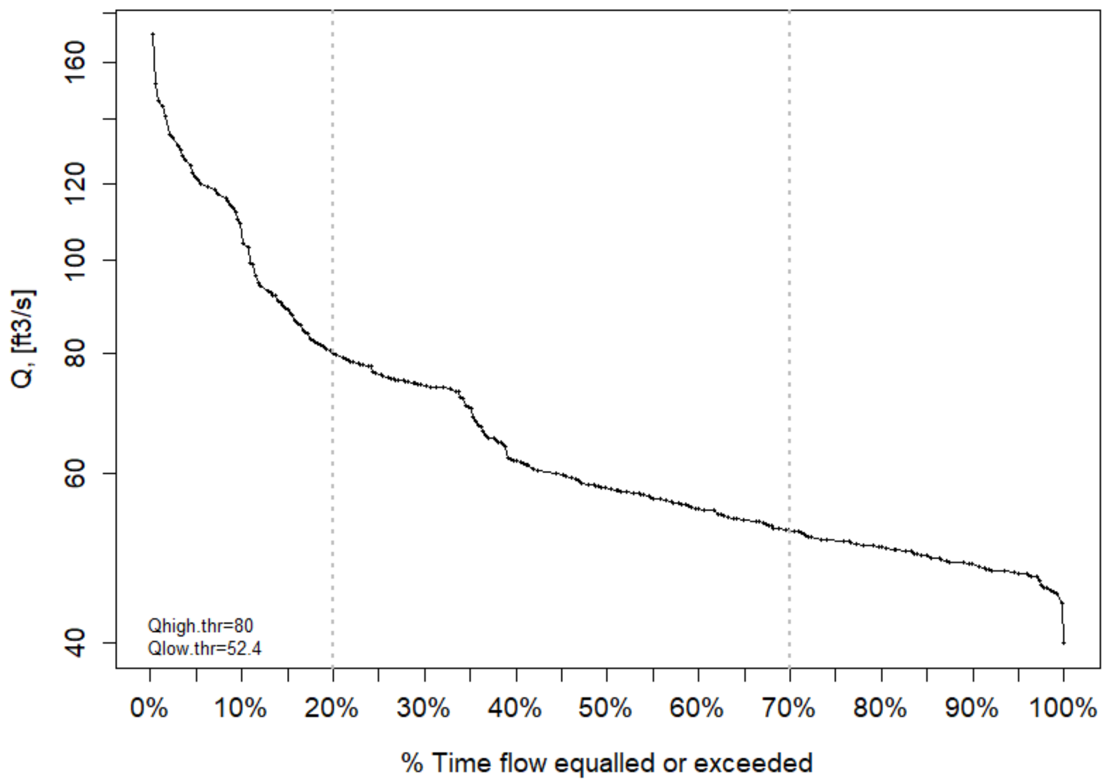

Figure A7. Flow duration curve for 2014 Sevier River USGS gage near Hatch.

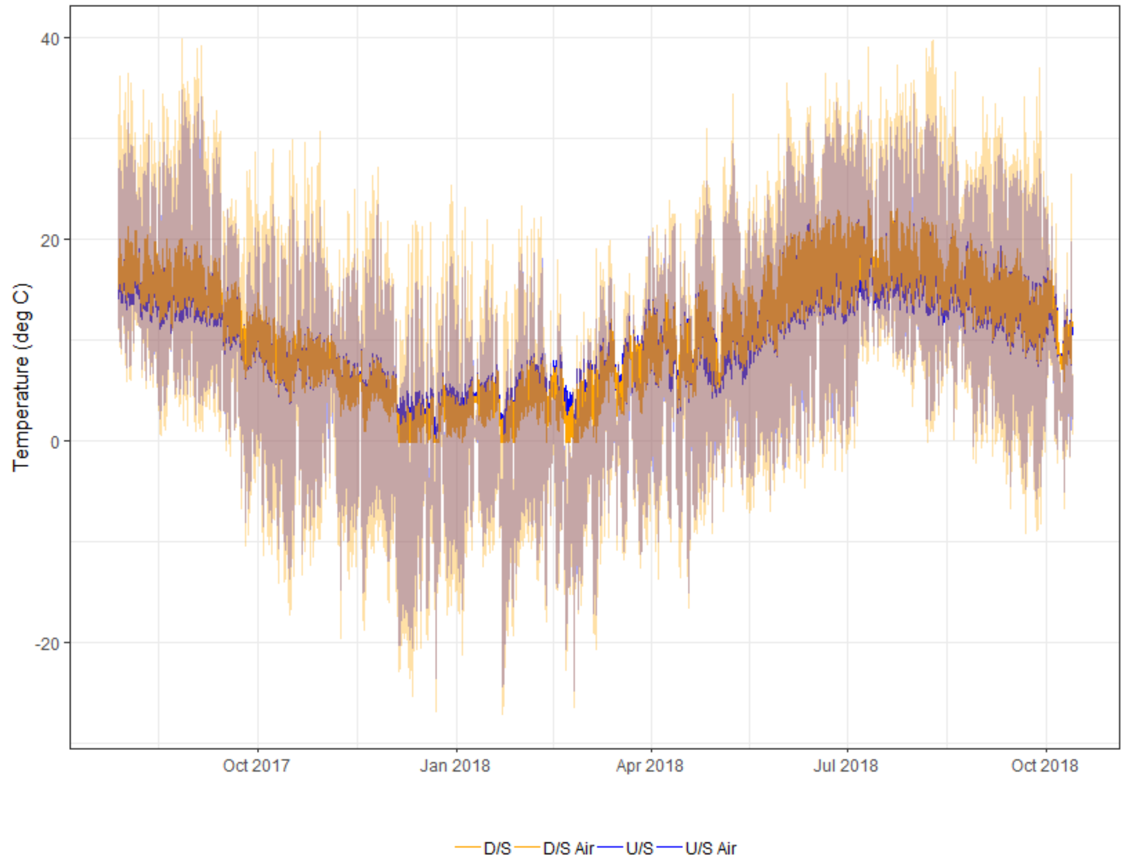

Figure A8. Hourly Sevier River water and air temperatures at upstream (U/S) and downstream (D/S) monitoring sites for entire period of observation.

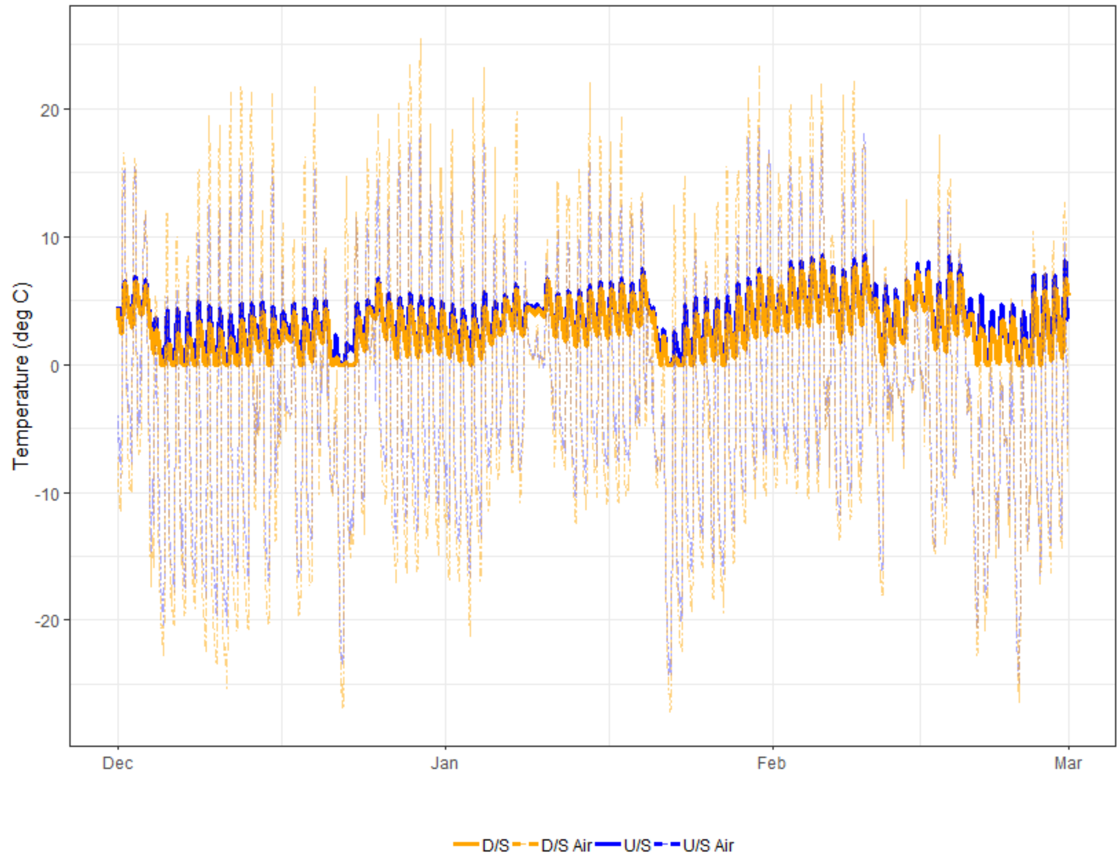

Figure A9. Hourly Sevier River water and air temperatures at upstream and downstream monitoring sites for winter 2018.

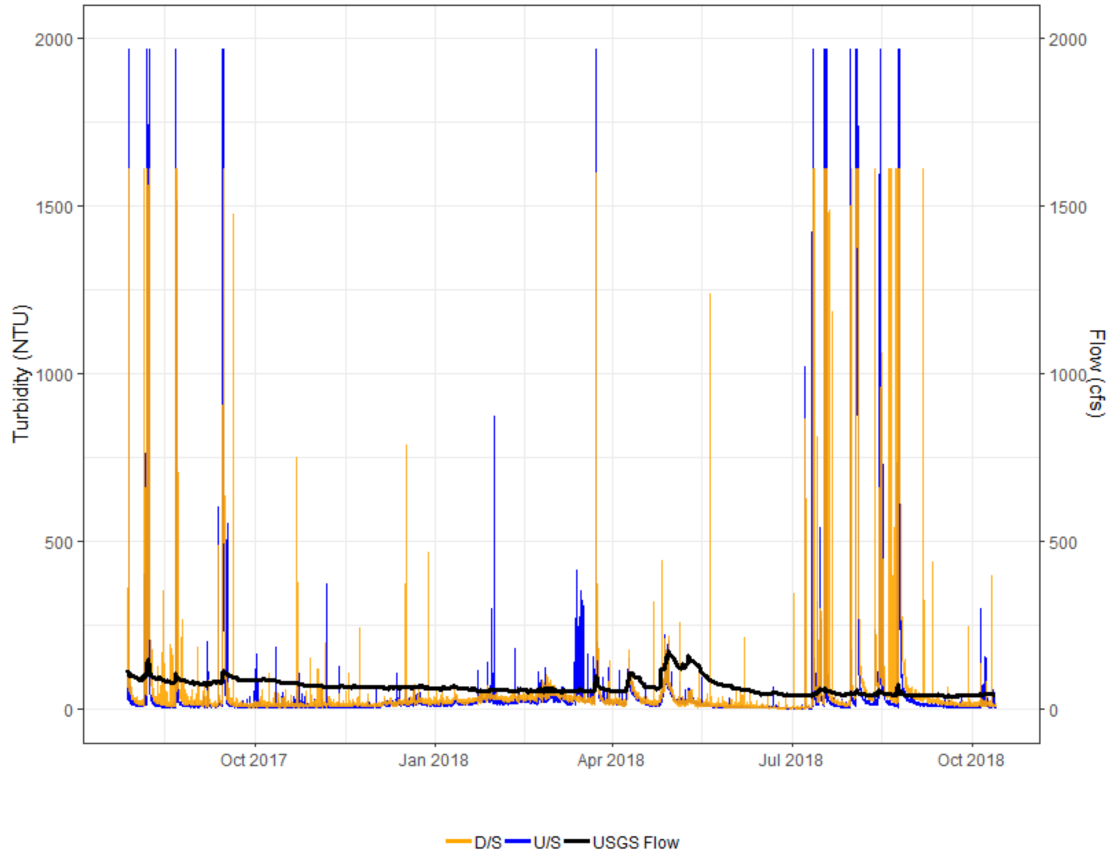

Figure A10. Turbidity at upstream and downstream sites with USGS flow showing full NTU range of turbidity measurements.

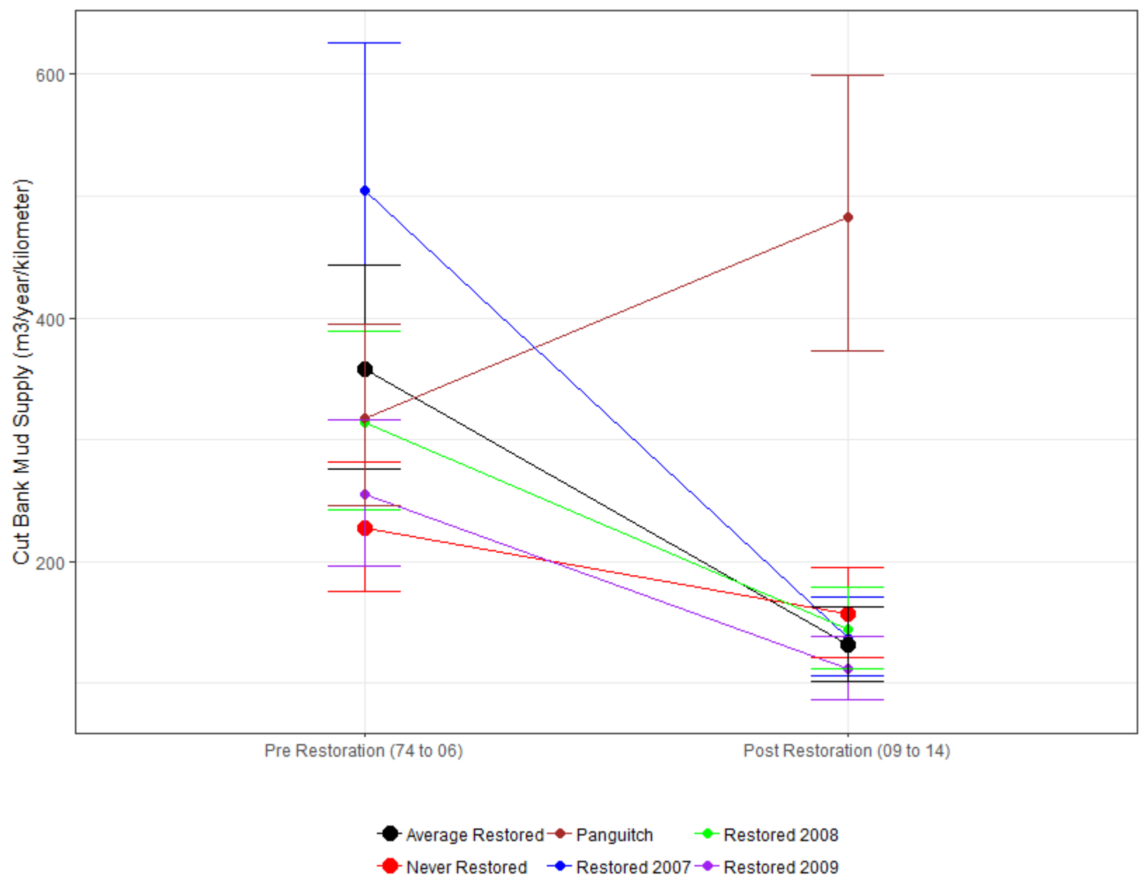

Figure A11. Aerial photo analysis results with uncertainty where pre-restoration time period includes 2005 flood.

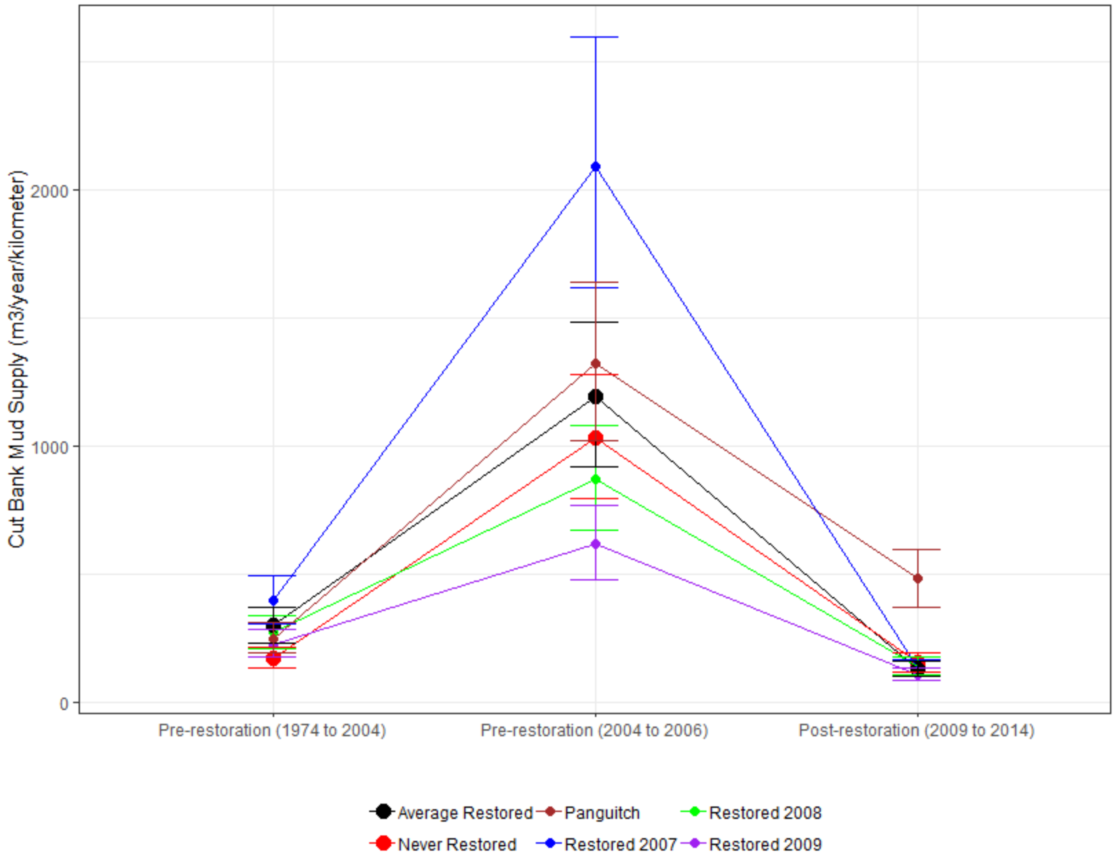

Figure A12. Aerial photo analysis results including uncertainty with 2004 to 2006 as individual period.

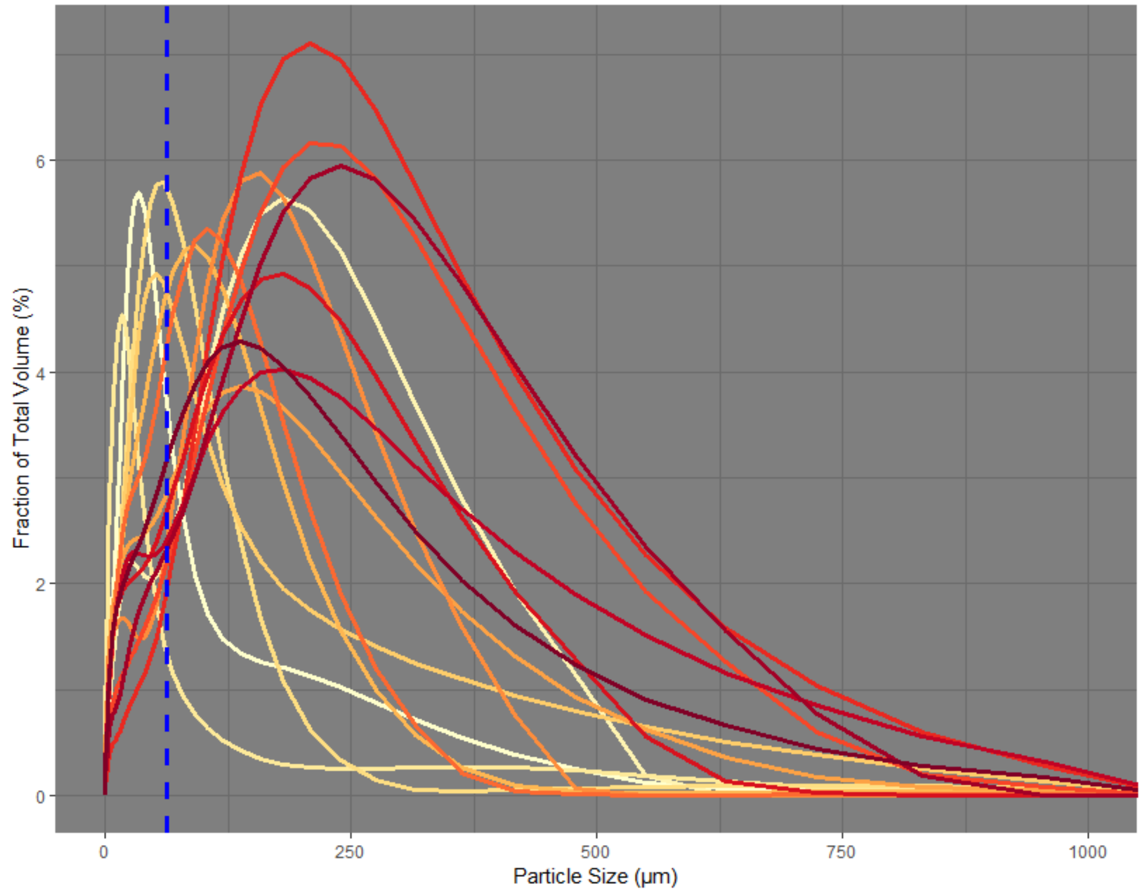

Figure A13. Grain size distribution for each sediment sample. The dashed line represents 63 um, grain sizes smaller than which are considered mud. Color gradient shows upstream (light yellow) to downstream (dark red).

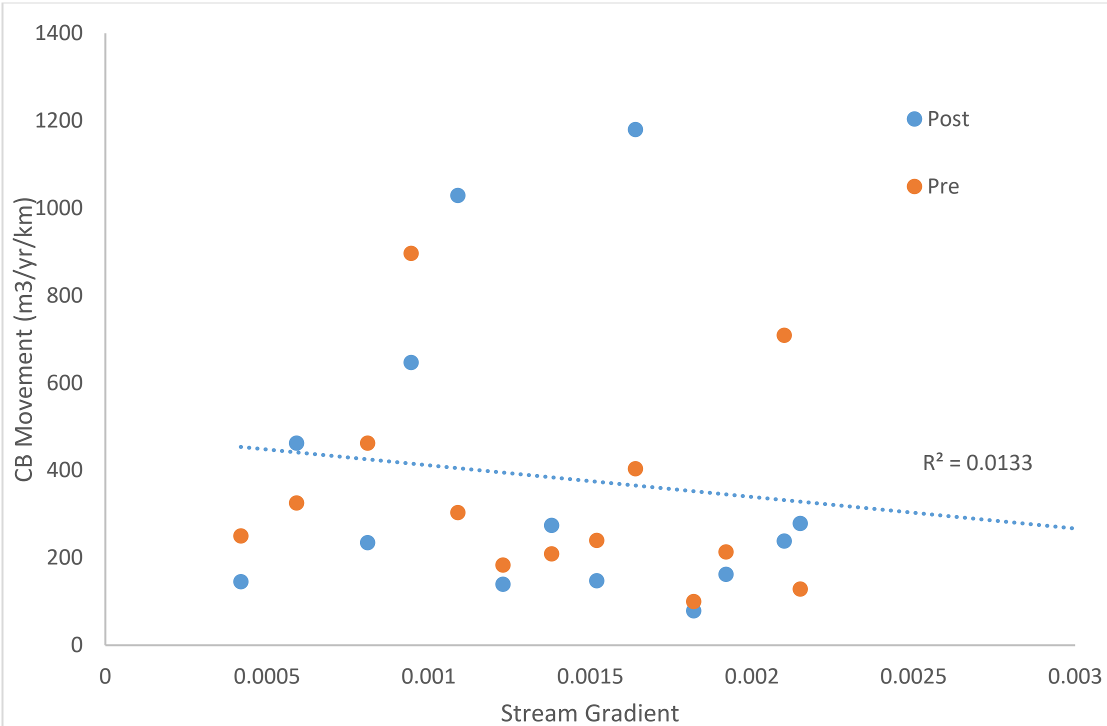

Figure A14. Cut bank movement from aerial photo analysis vs stream gradient of SIAM reach in pre- and post-restoration time periods. Trend line is for post-restoration data.

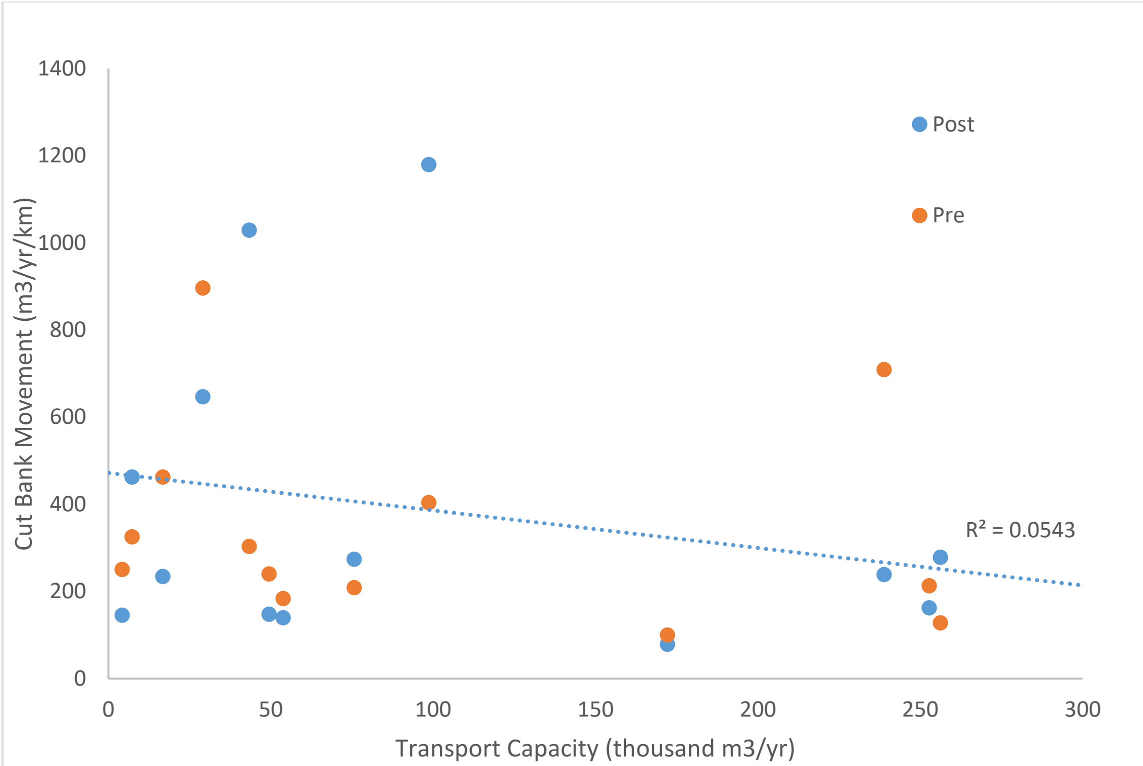

Figure A15. Cut bank movement from aerial photo analysis and SIAM calculated reach transport capacity for pre- and post-restoration time periods. Trendline is for post-restoration data.

Table A1. Reach lengths and changes for aerial photo analysis.

Status	2004 Reach Length (km)	2014 Reach Length (km)	Change (%)
Never Restored	23.63	20.84	-11.78
Restored 2007	1.95	1.93	-1.03
Restored 2008	5.15	5.29	2.8
Restored 2009	3.74	3.82	2.25
Panguitch	7.81	8.49	8.71

Table A2. Grain size analysis sample name, location, source type, and mud content.

Site Name	Valley	Site Easting	Site Northing	Sediment Source Type	%<63 um
P1	Hatch	376618.42	4170583.65	?	21.67
P2	Hatch	370301.66	4161454.15	Alluvium	41.50
P3	Hatch	376618.42	4170583.65	?	17.88
P4	Hatch	370435.37	4161338.42	Alluvium	79.46
s84	Hatch	372471.41	4165458.72	Alluvium	92.13
s85	Hatch	372510	4165360.41	Alluvium	70.39
Bridge	Hatch	373810.11	4168095.37	OVF	59.81
s89	Hatch	374233.07	4169111.63	OVF	57.38
s86	Hatch	374475.6	4169199.35	OVF	51.51
s87	Hatch	374467.36	4169236.04	OVF	44.12
s88	Hatch	374316.14	4169357.18	OVF	55.04
s91	Hatch	376609.76	4170965.9	Alluvium	44.29
s90	Hatch	376561.17	4171060.13	OVF	42.76
s93	Panguitch	374310.81	4192179.14	?	29.33
s95	Panguitch	374275.39	4192219.78	Alluvium	47.04

Table A3. Example cut bank resample results showing initial analysis (original) and resample results with relative differences.

	Min	Original (total m3 mud)	Max
D/S	1975	2037.9	2286
Relative diff (%)	-3.1	-	12.2
U/S	1625	1881	1894
Relative diff (%)	-13.6	-	0.7

Table A4. Sevier River flow measurements with site name, date/time, flow, and a comparison with the nearby USGS gage.

Site	Date	Time (MDT)	Flow (cfs)	Flow % Difference from USGS Gage
Downstream	9/16/2017	16:00	101.69	-3.2
Upstream	9/17/2017	9:00	85.07	-17.4
Upstream	11/3/2017	17:00	92.18	33.8
Downstream	11/3/2017	18:00	87.87	27.5
Upstream	5/30/2018	15:00	69.59	-4.0
Downstream	5/30/2018	16:00	70.59	-2.6
Upstream	6/17/2018	19:00	52.34	15.0
Downstream	6/18/2018	16:30	50.57	8.5
Upstream	10/13/2018	14:00	45.95	4.0
Downstream	10/13/2018	15:30	49.09	11.1
Downstream	11/10/2018	15:30	46.44	-5.2
Upstream	11/10/2018	17:00	53.54	6.4

Table A5. Mud content results for sediment source types. OVF = old valley fill.

	Mean % Mud	Standard Deviation
Alluvium	62.5	21.2
OVF	51.8	7.0

Equation A1. Stage-discharge relationship where Q is discharge in cfs, Cr is a constant, s is stream stage in feet, and β is a constant.

$$Q = C_r s^{\beta}$$