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Farmers, Fences, and Fine Sediment:

Using Long-Term Water Quality Data and Aerial Imagery to Quantify the Impact of Best Management Practices in Marsh Creek, ID

By

Graham Alexander Meese

A thesis

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Committee Approval

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Abstract

Farmers, Fences, and Fine Sediment: Using Long-Term Water Quality Data and Aerial Imagery to Quantify the Impact of Best Management Practices in Marsh Creek, ID

Thesis Abstract—Idaho State University (2018)

Agricultural and grazing practices have detrimental and long-lasting effects on water quality. Large investments in conservation actions aim to mitigate these impacts and restore ecosystem services. Since 1982, an estimated $62.9M have been spent on conservation and restoration programs within Marsh Creek in southeast Idaho. Between 1969 and 2017 suspended sediment (SS) flux in Marsh Creek was reduced 75%. A time-series comparison between conservation investments and SS flux shows a strong negative correlation between dollars spent and flow-normalized flux with lags of 6 and 7 years. Despite considerable investment in conservation actions, from 2004 to 2012 Marsh Creek still exceeded high- and low-flow SS limits more than 50% of the time. Future conservation investments will yield the greatest benefit through continued reduction of near-channel sediment sources, water quality monitoring, reconnecting the channel with the floodplain, and community education.

Keywords:

Best management practices (BMPs), conservation, conservation reserve program (CRP), environmental quality incentives program (EQIP), section 319 grants, river restoration, conservation history, stream water quality, agricultural water quality, water quality trend analysis, aerial image analysis, suspended sediment, bank erosion, semi-arid, southeast Idaho, Idaho State University, Bannock County, Marsh Creek, Portneuf River

Chapter 1: Introduction

1.1. Anthropogenic impacts on water quality

Urban, industrial, and agricultural growth have degraded the quality and quantity of water in many rivers worldwide (Ridoutt and Pfister, 2010). The global human population has grown from 3 billion to 6.9 billion between 1960 and 2010 (Hooke, 2012) and it is estimated to grow to 11.2 billion people by 2100 (Bongaarts, 2016). With an increasing global population comes a demand for agricultural intensification and expansion while simultaneously understanding and mitigating its impacts on global freshwater ecosystem services.

The agricultural and livestock industries are the largest users of both surface water and groundwater, and their impacts on water quality are in most cases detrimental (Foley et al., 2005). Crop and pasture land occupy roughly 40% of the global land surface, and technological advancements have doubled yields over the past four decades (Foley et al., 2005). However, these efficiencies come at the cost of high fertilizer inputs, limited return flow, and reduced groundwater recharge (Ward and Pulido-Velazquez, 2008). Compared to undeveloped watersheds, agricultural watersheds have been shown to have increased fine sediment loads (Walling, 1999), increased nutrient concentrations (Allan et al., 1997), decreased in-stream biodiversity (Feld et al., 2011), and modified flow regimes (Poff et al., 2007; Belmont et al., 2011).

1.1.1. Pollutant sources and pathways

Pollutant sources can be classified into two general categories: point source and nonpoint source. The distinction between the two is dependent on scale, but practically speaking, point source pollution is a single identifiable source of discharge into a water body such as a pipe from a wastewater treatment facility. Over the last few decades, point source pollution has been greatly reduced, and now nonpoint source (NPS) pollution is the largest contributor to poor water quality (Keiser and Shapiro, 2017). NPS is any diffuse pollutant source that cannot be directly measured as it enters a waterbody, making it much more difficult to manage. Common NPS pollution in streams and rivers are runoff from agricultural fields, stream bank erosion, atmospheric deposition and contaminated groundwater.

Agriculture has been identified as the largest contributor to NPS pollution, causing increased sediment and nutrient loads, decreased biodiversity and modified flow regimes (Walling, 1999; Allan, 2016; Feld et al., 2011; Poff et al., 2007; Schilling et al., 2011). Suspended sediment is of particular concern as it acts as a transport vector for nutrients (especially phosphorus), heavy metals, and pesticides (Walling and Collins, 2016). The main sources of sediment in agricultural watersheds include erosion of disturbed upland soils, gully formation, stream-bank erosion, and sheet and rill erosion on irrigated cropland (Schilling et al., 2011). Once sediment enters the fluvial system, the transport times can vary from years to thousands of years dependent on the longitudinal, lateral, and vertical connectivity of the channel and the transport distance (Fryirs, 2013; Pizzuto et al., 2014).

1.1.2. Federal policies and programs to protect and improve water quality

In the United States, a wide variety of anthropogenic impacts on water quality were federally recognized in 1948 with the passage of the Federal Water Pollution Control Act. This body of periodically growing legislation, now more commonly known as the Clean Water Act (CWA) after the 1972 amendments, was enacted, “to restore and maintain the chemical, physical, and biological integrity of the Nation’s waters” (EPA, 1972). The 1972 amendments set regulations for point sources of pollution, commonly associated with industrial operations. Significant reductions in pollutant loads have been observed due to the passage of the CWA

(Oelsner et al., 2017), and the largest reductions were observed prior to 1972 due to the development of waste water treatment plants enforced by the 1948 Federal Water Pollution Control Act (Keiser and Shapiro, 2017).

In Section 208, the 1972 amendments also addressed nonpoint source pollution for the first time, directing states to identify areas of concern and create management plans for nonpoint source pollution. Although this program was unsuccessful at the national level because it failed to create enforceable regulations and funding was completely used up by 1980 (Szalay, 2010), it did initiate many of the first water quality monitoring projects in watersheds across the US (McSorley, 1977). The CWA was later amended in 1987, strengthening regulations on nonpoint source pollution in Section 319, creating the Nonpoint Source Management Program (EPA, 2002). Section 319 requires state regulatory agencies to monitor pollutant levels, set total maximum daily loads (TMDLs) for pollutants in streams, rivers, and lakes, and to create a list of all waterbodies deemed impaired, called a 303(d) list (EPA, 2002). Agricultural catchments are of particular concern as they are often deemed impaired due to high nutrient concentrations and suspended sediment loads (Allan et al., 1997). If a stream exceeds a TMDL threshold, it is required by law under the CWA that action be taken by the governing agency to improve the quality of the water, often through the implementation of Best Management Practices (BMPs) (EPA, 2002). A BMP is a standard treatment that is used for controlling a given pollutant. Conservation and restoration project utilize a suite of BMPs to achieve a target objective. For example, direct cattle access has been shown to destabilize the banks of channels which has been identified as the primary source of fine sediment (Guilinger, 2017; Peppler and Fitzpatrick, 2005). To reduce sediment, a stream bank restoration project is likely to include BMPs such as riparian exclosure fencing, grazing management plans, and bioengineering treatments like planting vertical and horizontal willow bundles (Scully et al., 2003). Since 1972, over $1 trillion dollars have been spent in the United States to improve water quality; yet over 50% of streams are still deemed impaired (Oelsner et al., 2017; Keiser and Shapiro, 2017).

1.1.3. Conservation and restoration efforts

As a consequence of the anthropogenic impacts on water quality, river conservation and restoration projects occur globally and are an integral part of natural resource management (Wohl et al., 2005). Billions of dollars are spent each year on the implementation of BMPs that target specific reaches within a river that exceed TMDLs, but the biological and geomorphic responses to implementation can often lag (Meals et al., 2010), and most rivers rarely return to their undisturbed state (Schilling et al., 2011). Classification systems are commonly used in aiding restoration efforts (i.e. Rosgen, 1994; Buffington and Montgomery, 2013). These assessments are frequently based on reference reaches and do not directly account for changes in channel-forming processes or the legacy of land use and its effect on the ecological and geomorphic processes (Palmer et al., 2005).

1.2. Documenting the impact of conservation and restoration

BMPs aimed at reducing suspended sediment can have numerous objectives from reducing in-stream loads to improving ecological conditions (Feld et al., 2011). The effect of individual BMPs are well known through small-scale experiments, but when the spatial scale increases to the watershed level, many different BMPs are employed throughout time and space, within varying land uses, making it much more difficult to quantify the impact of each project at either the reach or watershed scale (Cherry et al., 2008).

Many different methods are used to measure BMP success and depend on spatial and temporal extent. The effectiveness of different BMPs are generally tested at local in-field and edge-of-field monitoring studies, but these can be vary dramatically depending on the site specific conditions and application (Cherry et al., 2008). In-stream ecological surveys and site assessments are commonly used to monitor the health of a stream segment, but observations are often made only once a year at one location, offering only a snapshot in space and time (BLM, 2017). Watershed modeling is a valuable tool as the spatial grain and focus of the model can be built to suit the user's interest, but models can be misused and provide inaccurate estimations if they are not properly calibrated with watershed specific field data (Belmont and Foufoula-Georgiou, 2017). The use of repeat aerial image analysis helps quantify changes in channel planform geometry between two points in time. Though it provides both large spatial extent and fine grain resolution, when viewed alone it does not directly indicate the drivers of geomorphic change (Rowland et al., 2016). Measuring BMP effectiveness at the watershed scale is often done through paired watershed studies, but this requires assumptions about similarity between the two locations that may include gross oversimplifications of complex systems (Loftis et al., 2001). Trend detection in water quality records integrate the changes that have occurred within the watershed upstream from the sampling location, but it can be difficult to attribute changes in the watershed to specific changes in land use and conservation due to confounding variables that also influence water quality (Hirsch et al., 2010). If resources are available, multiple methods can be combined to gain a more complete understanding of the effect of conservation actions.

Despite the public visibility of and interest in river restoration, the scientific community's understanding of fluvial systems is incomplete, and the integrated effect of anthropogenic perturbations on lotic ecosystems and channel planform equilibrium is different between each watershed (Malakoff, 2004). Specifically, BMPs that target fine sediment in agricultural watersheds often fail to yield quantitative improvements (Schilling et al., 2011; Palmer et al., 2005; Tomer and Locke, 2011). This is in part because conservation and restoration projects are often aimed at watershed-level objectives, yet individual projects often occur at the reach scale and are fragmented in space and time, impacting only a small fraction of the watershed. Another large disparity in quantifying improvement is a lack of data. Out of all the projects in the National River Restoration Science Synthesis database, only 10% included an assessment or evaluation component (Palmer and Allan, 2006). Furthermore, BMPs are often implemented inefficiently due a lack of watershed-specific understanding of sources and pathways of contaminants, ecological function, and the legacy of past land use (Tomer and Locke, 2011). Successful large-scale assessment, implementation, and monitoring projects targeting fine sediment have been conducted in large rivers, such as the Colorado River (Cohn, 2001), however, few successful long-term monitoring projects have been completed in smaller watersheds (Bernhardt et al., 2005; Palmer and Allan, 2006; Brooks et al., 2010). Additionally, the lack of long-term monitoring projects is due in part to the persistence of legacy effects, data access, the quality and representativeness of pre- and post-restoration data, and the lag time between completed projects and measurable improvements in water quality. Furthermore, detecting change can be time-consuming, expensive, and require historic records (Drewes, 1988; Jones et al., 2012; Minella et al., 2008; Palmer and Allan, 2006).

1.2.1. Trend detection in water quality data

The U.S. Department of Agriculture (USDA) alone spends $3.5 billion dollars annually on conservation programs. Naturally, it is important to have methods to detect trends in water quality to assess the success of restoration and conservation actions where historical water quality records are available (Tomer and Locke, 2011). Long-term water quality data sets were not widely collected in the United States until the 1970's driven by the growing legislation within the CWA and the billions of dollars spent to reduce pollutant inputs (Smith et al., 1987). With growing investment in conservation came a need to detect and measure changes, but water quality data can be challenging to analyze using parametric or non-parametric tests because the data often fails to meet statistical assumptions such as being normally distributed, collected at equal time intervals, and conforming to predefined trend relationships (Hirsch et al., 1991). Initial studies used non-parametric tests, such as the Seasonal Kendall test (Smith et al., 1982), which works well in detecting a trend within the traditional hypothesis testing framework. Unfortunately, water quality managers often need to know more than just if an analyte is changing; they need to know when the change occurred, at what rate, and the magnitude of change to effectively guide future efforts. Hirsch et al., (2010) recognized this need and developed a flexible model employing Weighted Regressions based on Time, Discharge, and Season (WRTDS), allowing for the detection and description of long-term trends (Hirsch et al., 2010). The main advantages of the WRTDS method are numerous: it allows for unevenly spaced data with large gaps, it does not assume a static concentration-discharge relationship, it does not assume that the seasonal variations are static through time, and it makes no assumptions about the shape of the long-term trend (Hirsch et al., 2010). In addition, the WRDTS model generates a flow-normalized concentration and flux that helps to eliminate the influence of interannual changes in mean discharge. It accomplishes this by calculating each daily concentration using a discharge randomly selected from a probability distribution function of all of the discharge values that have occurred on that given day of the year. Uncertainties in this method are quantified using a block bootstrap method created for specified confidence intervals that account for short-term serial autocorrelation (Hirsch et al., 2015).

1.2.2. Repeat aerial imagery to quantify planform change

A unique way to quantify changes in fluvial systems is through the use of repeat aerial imagery. For some time, researchers have used repeat imagery to detect changes in channel planform over short intervals and large spatial scales (e.g. Nelson et al., 2013; Aalto et al., 2008; Rowland et al., 2016). In human-altered landscapes, these types of analyses provide insight into a river's geomorphic response to land use, channel modification, conservation actions, and water withdrawals. The recent increase in tools for geospatial analysis and increasing access to high-resolution satellite imagery have allowed for this type of analysis to expand to greater spatial extents and temporal resolutions (Rowland et al., 2016; Rhoads et al., 2016; Nelson et al., 2013; Belmont et al., 2011).

1.3. Study area

Marsh Creek is a low-gradient, underfit, meandering stream that flows south-to-north through a broad (1-2 km) agriculturally dominated valley in southeast Idaho. It drains approximately 1,100 km^2, with elevations ranging from 1380-2700 meters, and is the largest tributary to Portneuf River (figure 1.1). Marsh Valley formed during the Neogene due to basin and range tectonics, and has since undergone extensive filling and subsequent incision due to the passage of the Yellowstone hotspot (Thackray et al., 2011). During the late Pleistocene, Marsh Valley was the uppermost flood corridor of the Bonneville Flood, draining pluvial Lake Bonneville at Red Rocks Pass roughly 17 ka. Thackray et al. (2011) suggest that there was non-catastrophic drainage of Lake Bonneville prior to and post Bonneville Flood (ca. 17 ka), aiding in the formation of the broad southern end of Marsh Valley. The northern end of the valley is considerably narrower, confined by the incised Pleistocene alluvial fans and the 430 +/- 70 ka Portneuf basalt (Thackray et al., 2011; Rodgers et al., 2006).

1.3.1. Climate and hydrology

The climate in the Marsh Valley is semi-arid, with an average precipitation of 400 mm in the valley and up to 500-760 mm at the highest elevations. Monthly temperatures range from an average high of 32°C in July to an average low of -8°C in January (Hammes, 2010; ISCC, 2002). Precipitation falls predominantly as snow between November and March, with peak runoffs in Marsh Creek of 2-28 m^3/sec typically generated by rain on snow events in the early spring (Hammes, 2010). Summer base flows range from 0.3-1.4 m^3/sec and are sustained by numerous springs throughout the valley but reduced by agricultural surface and groundwater withdrawals during the growing season.

The headwaters of Marsh Creek are left- and right-hand fork Marsh Creek located in the southern end of the Portneuf Range. The main tributaries to Marsh Creek are Birch Creek, Hawkins Creek, Garden Creek, Goodenough Creek, Bell Marsh Creek, and Walker Creek. All of these tributaries flow east off the Bannock Range. The hydrology in these tributaries is flashy in the winter and spring. During summer base flows, many of these tributaries run dry due to diversions and become disconnected from the main stem of Marsh Creek (Hammes, 2010; Guilinger, 2017).

1.3.2. Human history of Marsh Creek

Prior to the arrival of settlers, the Portneuf River drainage had been used for centuries by the Shoshone and Bannock tribes. These tribes were nomadic and followed the seasonal movement of bison throughout the region (Eaton, 1935). In 1834, Fort Hall was built just north of present-day Pocatello as a fur trading post and was an important stopping point along the Oregon Trail. In 1868, the Fort Hall Indian Reservation was created and included the Marsh Creek watershed. At its initiation the reservation was 1.3 million acres, but in 1882 the reservation was reduced in size for the construction of the Union Pacific Railroad through Marsh Valley, then in 1889, the entire Marsh Creek watershed was removed from the reservation, shrinking it substantially (Sanger, 2018).

The first significant modification of the channel occurred when fur trappers removed beaver between 1830-1860. In 1878-1879 the channel was next affected by the construction of the Utah & Northern narrow gauge railroad within Marsh Valley, but it was soon abandoned in 1882 when frequent flooding required the railroad to rerouted down the Portneuf River valley (Link and Phoenix, 1996; DHBC, 2016). The construction of the railroad brought an influx of settlers to the area with the founding of Pocatello as a major railroad hub in 1882 (Link and Phoenix, 1996). In 1906, the Portneuf Marsh Valley Canal Company was formed, building the largest diversion in the Portneuf watershed (figure 1.1), taking water from the Portneuf River at Topaz landing, just downstream of Lava, south into Marsh Valley to irrigate the southeast benches above Downey, Arimo, and Virginia (PMVCC, 2018). This canal is roughly 43 km long and diverts up to 70% of summer flow in the Portneuf River for agricultural use and to power a small hydroelectric plant (ISCC, 2002).

Agriculture and livestock production have supported the communities in Marsh Valley since the 1860's (DHBC, 2016). Today roughly 80% of the Marsh Creek watershed is used for crop and rangeland (IDEQ, 2003), and the USDA Agricultural Census indicates that the areas in production have been relatively stable since the 1980's (USDA, 2017; IDEQ, 2003). As of 2013, there were 35 animal facilities that utilized Marsh Creek as the only access for stock water (IASCD, 2013).

1.3.3. Water quality and conservation actions in Marsh Creek

Marsh Creek is considered ‘impaired’ under the CWA because it exceeds water quality limits for nutrients and sediment and does not meet the standards for its designated beneficial uses for cold water aquatic life and secondary contact recreation (EPA, 2016). As of 1977, the majority of the sediment was coming off dryland farms on the West Bench of Marsh Valley and SS loads were an order of magnitude greater than what we measure today (McSorley, 1977). In 2017, the banks were identified as the primary source of sediment, indicating that there has been a shift in the dominant source of fine sediment through time (figure 1.2) (Guilinger, 2017).

Conservation and restoration actions in Marsh Creek have been fragmented in space because 66% of the land within the watershed, and 93% of the land adjacent to the mainstem channel, is privately owned. Importantly, the dominant sources of SS are derived from these private lands. Conservation programs on private land depend on landowners who are willing to participate and have the financial means of helping to pay for a portion of their projects. The main agencies responsible for the implementation of these cost-share programs are the Natural Resource Conservation Service (NRCS), the Idaho Department of Environmental Quality (IDEQ), and the Portneuf Soil and Water Conservation District (PSWCD).

Conservation actions to improve water quality have occurred within the Marsh Creek watershed since the 1930’s but funding for large scale conservation actions was not available until the 1980’s (Hammes, 2010; Koester, 1995). These projects aimed at improving the water quality in Marsh Creek by stabilizing the soils on the upper benches (Koester, 1995; PSWCD, 1994), which had been identified in 1965 as the dominant source of sediment in Marsh Creek (figure 1.2) (Merrell and Onstott, 1965). The main programs that support upland conservation actions are the State Agricultural Water Quality Program (SAWQP) led by the IDEQ and

PSWCD, and the Conservation Reserve Program (CRP) administered by the NRCS. The CRP has been active since 1986 and is the longest and largest conservation program that has occurred in the Marsh Creek watershed, removing subprime land from production, stabilizing the soils, and reducing erosion (USDA, 2018a). Together these two programs have implemented BMPs on roughly 50% of the upland benches in the Marsh Creek watershed at an estimated cost of $57,659,000 (Koester, 1995; EWG, 2017; PSWCD, 1994).

In the 1990s, as the upland sediment source stabilized, conservation actions shifted to the lowlands, aimed at reducing streambank erosion and sheet runoff from irrigated fields. The main programs implementing near-channel projects are the Environmental Quality Incentives Program (EQIP) managed by the NRCS and 319 grants managed by the IDEQ. Between these two programs, 44 km of fencing and 30 off-channel watering troughs were constructed, aimed at reducing the impact of livestock on streambank erosion.

Unfortunately, these considerable efforts have not been enough to decrease suspended sediment loads below target limits: Marsh Creek exceeds its high- and low-flow limits more than 50% of the time (IDEQ, 2010). On average, the suspended sediment concentrations in Marsh Creek are two times higher than the Portneuf River at their confluence, necessitating continued implementation of BMPs and highlighting the importance of further monitoring and research.

1.4. Problem statement

It is difficult to measure watershed-scale responses to conservation actions, yet this is critical information for increasing the effectiveness of future efforts. This thesis focuses on the compilation and analysis of historic water quality records, repeat aerial imagery, and conservation actions to assess the effectiveness of conservation actions on long-term trends in suspended sediment loads in Marsh Creek, an agriculturally impacted stream in southeast Idaho

(Figure 1.1). In the thesis that follows, I use all available historic data sources to assess whether changes in total suspended sediment loads over time are attributable to the progressive implementation of conservation actions.

Chapter 1 Figures

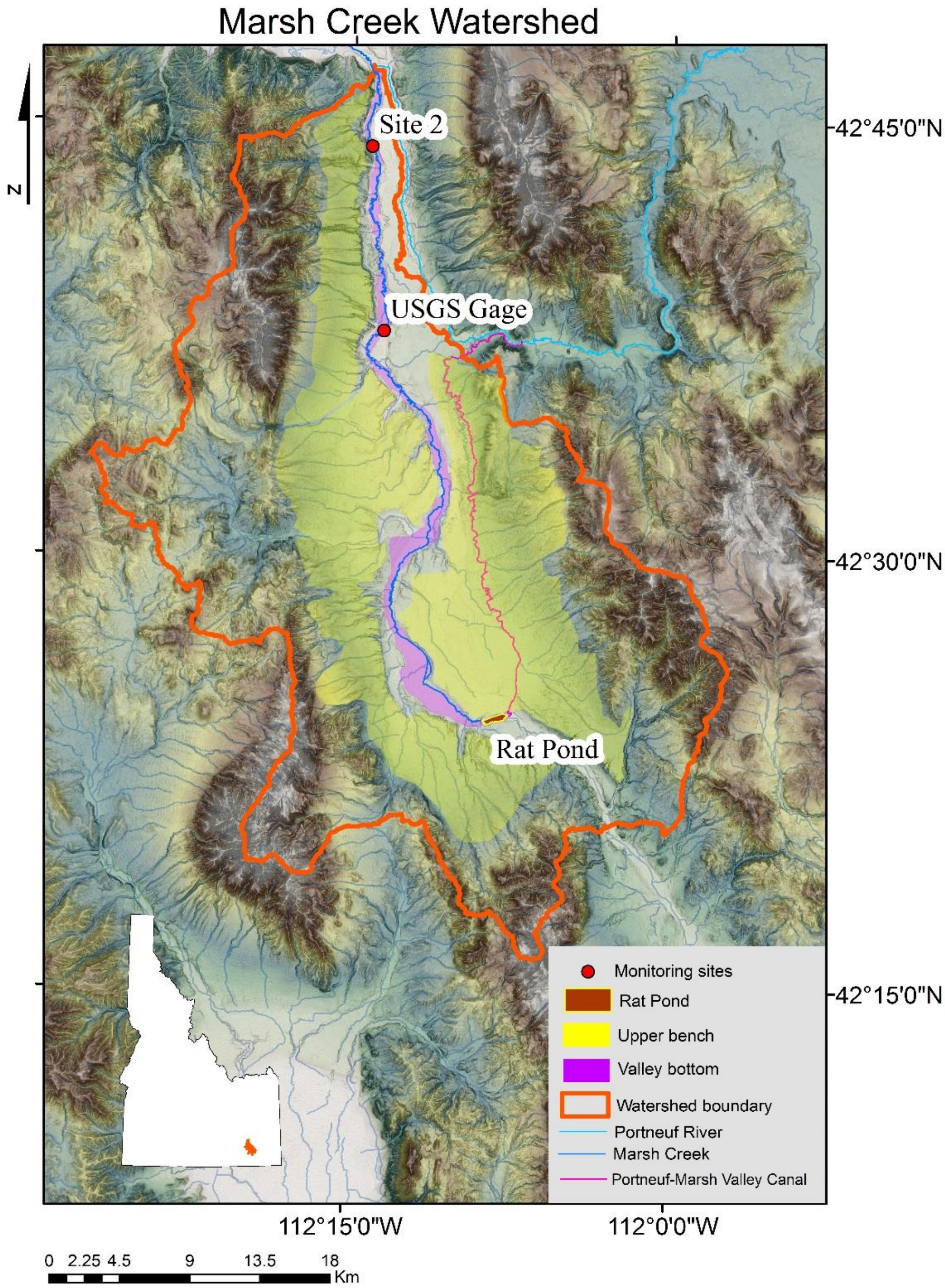

Figure 1.1: Map of Marsh Creek watershed in Southeast Idaho. This analysis focuses on the main stem of Marsh Creek from the confluence with the Portneuf River upstream to the Rat Pond. The orange line is the watershed boundary, and the yellow area shows the upland benches surrounding Marsh Creek that were dry-farmed. The purple area shows the lowland valley adjacent the Marsh Creek where irrigated agriculture occurs as well as livestock grazing and overwintering. The Marsh Valley Canal (magenta line) transfers water from the Portneuf River south into Marsh Valley to irrigate the eastern bench.

Figure 1.2: Left: Image of sheet and rill erosion on the upper benches in the northwest portion of Marsh Creek from Merrell and Onstott (1965). The upper benches were used for dryland agriculture and were identified in this initial study as the dominant source of suspended sediment in Marsh Creek. Right: Image of repeated bank failures, steep, unvegetated banks, and a cattle access point from a kayak survey of Marsh Creek in 2016 (Guilinger, 2017). These images represent the upland and lowland sources of sediment identified by prior research. The dominant source of sediment has shifted between 1965 and 2017 from upland erosion to lowland bank erosion as shown through a sediment fingerprinting analysis (Guilinger, 2017).

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Chapter 2: Farmers, fences, and fine sediment: Have conservation investments improved water quality in Marsh Creek, an agriculturally impacted stream in southeast Idaho?

Abstract

Agricultural and grazing practices have had detrimental and long-lasting effects on water quality in our nation's rivers and streams, including fine sediment pollution. Large investments in conservation actions aim to mitigate these impacts and restore ecosystem services. To assess the return on these investments, we compare a time series of conservation projects against historic records of suspended sediment flux in Marsh Creek, an agriculturally impacted watershed in southeast Idaho. Repeat aerial photograph analysis was also used to identify anthropogenic modifications to the channel and the natural geomorphic response, providing insight into sediment sources within the basin. The aerial photograph analysis and previous sediment fingerprinting efforts indicate that the current sediment in transport comes from the banks of the stream, exacerbated by the presence of livestock. Detection of long-term water quality trends in lower Marsh Creek using the weighted regressions on time, discharge, and season model indicate that flow-normalized flux of suspended sediment decreased by 23.5×10^6 kg/yr between 1969 and 2017, a 75% reduction. The correlation between conservation investments and suspended sediment flux was tested using a generalized least squares (GLS) regression and a cross-correlation analysis. The GLS regression revealed a strong negative correlation ($\beta = -0.74$) between dollars spent and flow-normalized flux ($p = 0.0012$), and the cross-correlation analysis of the 1st order differenced values revealed significant correlations of -0.41 and -0.39 at lags of 6 and 7 years, respectively. The combination of the long-term water quality trend analysis and the cross-correlation analysis provide valuable insight into the positive impact conservation efforts have made and the expected lagged response in water quality improvements, helping to guide future conservation and monitoring efforts.

2.1. Introduction

2.1.1. Agricultural impacts on water quality

The agricultural and livestock industries are among the top users of both surface water and groundwater, and their impacts on water quality are in most cases detrimental (Foley et al., 2005). Compared to undeveloped watersheds, agricultural watersheds have been shown to have increased fine sediment loads (Walling, 1999), increased nutrient concentrations (Allan et al., 1997), decreased in-stream biodiversity (Feld et al., 2011), and altered flow regimes (Poff et al., 2007; Belmont et al., 2011).

Since the passage of the Clean Water Act (CWA) in 1972 over 1 trillion dollars have been spent to improve water quality yet over 50% of streams are still deemed impaired (Oelsner et al., 2017; Keiser and Shapiro, 2017). Though significant reductions in pollutant loads have been observed since the CWA, the largest changes were due to limiting point sources such as waste water treatment plants enforced by the 1948 Federal Water Pollution Control Act (Keiser and Shapiro, 2017). Today, water quality managers are faced with the difficult task of controlling and mitigating nonpoint source pollution associated with agricultural areas. This often requires watershed-specific information about sediment sources and conveyance mechanisms to effectively and efficiently solve water quality issues (Walling, 1999; Belmont and Foufoula-georgiou, 2017).

Globally, ~40% of agricultural watersheds are impaired due to fine sediment pollution caused by irrigation, tilling, overgrazing, stream bank destabilization and lack of crop cover (Foley et al., 2005). In these systems, sediment enters the aquatic system via sheet and rill, gully, stream channel, and irrigation-induced erosion (ISCC, 2002). Once in the aquatic system, suspended sediment can cause compounding effects because it not only impacts aquatic life and aesthetics, but also aids in the transport of nutrients, heavy metals, pesticides, and other organic pollutants (Walling and Collins, 2016; Feld et al., 2011).

2.1.2. Conservation and restoration efforts

Conservation and restoration projects occur globally and are an integral part of natural resource management (Wohl et al., 2005). Billions of dollars are spent each year on the implementation of best management practices (BMPs) which aim to reduce agricultural impacts, but the biological and geomorphic responses to implementation can often lag (Meals et al., 2010), and these systems rarely return to their undisturbed state (Schilling et al., 2011). Specifically, BMPs that target sediment in agricultural watersheds often fail to yield quantitative improvements (Schilling et al., 2011; Bernhardt et al., 2005) due to prolonged response times, the short time scales over which the projects are evaluated, and because reach-scale projects are often too fragmented in space and time to impact watershed-level resource concerns (Meals et al., 2010). Another limitation in quantifying improvements is a lack of post-project monitoring data. Out of all the projects in the National River Restoration Science Synthesis database, only 10% include an assessment or evaluation component (Palmer and Allan, 2006). Few successful long-term monitoring projects have been completed in small-scale watersheds, yet low-order streams make up a large portion of the impacted landscape (Bernhardt et al., 2005; Palmer and Allan, 2006; Brooks et al., 2010).

2.1.3. Challenges in measuring success in restoration and conservation actions

The success of restoration and conservation projects can be measured in many different ways, driven by spatial scale and the target resource concerns (Wohl et al., 2005; Palmer et al., 2005). The main techniques to measure conservation success are through repeat ecological surveys (Palmer et al., 2005; Pander and Geist, 2013), repeat channel condition surveys (BLM, 2017), repeat aerial imagery analysis (Rowland et al., 2016), water quality trend analysis (Hirsch et al., 2010), and watershed modeling (e.g. Tomer and Locke, 2011). These different techniques vary in their spatial and temporal extent. Combining long-term trends in watershed-scale water quality records, and repeat aerial imagery analysis provides a unique integrated measurement of the changes within the basin (Hirsch et al., 2010).

Long-term water quality data sets were not widely collected in the United States until the 1970s, driven by the growing legislation within the CWA and the billions of dollars spent to reduce pollutant inputs (Smith et al., 1987). Water quality data can be challenging to analyze using parametric or non-parametric tests because the data often fails to meet statistical assumptions, such as being normally distributed, collected at equal time intervals, and conforming to pre-defined trend relationships (Hirsch et al., 1991). Hirsch et al. (2010) recognized this need and developed a flexible model using weighted regressions based on time, discharge, and season (WRTDS), allowing for the detection and description of long-term trends (Hirsch et al., 2010). In addition to generating daily concentrations and fluxes, the WRDTS model generates a flow-normalized concentration and flux that helps to eliminate the influence of interannual changes in discharge (Hirsch et al., 2015).

Beyond measurements of in-stream flux, changes in sediment sources and the effectiveness of restoration actions can be measured through repeat aerial imagery analysis. In human-altered landscapes, these types of analysis provide insight into a river's geomorphic response to land use, channel modification, conservation actions, and water withdrawals by revealing where and how changes have occurred (Rowland et al., 2016; Rhoads et al., 2016; Nelson et al., 2013; Belmont et al., 2011). Repeat aerial image analysis can be used to assess whether the channel has been a source of sediment through time, and can be used to reveal the human impact on the channel via direct modification.

2.1.4. Research objective

This chapter utilizes all available water quality records of suspended sediment (SS), compiled land use changes, and conservation actions to quantify long-term trends in SS loads and their response to the progressive implementation of conservation actions within Marsh Creek (figure 2.1). We use a timeline to illustrate when and where land use changes and conservation actions have occurred, and repeat aerial imagery to quantify the integrated planform geomorphic response to the conservation and restoration efforts within the watershed. This information is critical for gaining a better understanding of the effect these actions have had on reducing sediment loads at the watershed scale, and should prove useful in guiding future conservation investments.

2.2. Watershed description

Marsh Creek is a low-gradient, underfit, meandering stream draining approximately 1,100 km^2 and is the largest tributary to Portneuf River (figure 2.1). Marsh Valley formed during the Neogene due to basin-and-range extensional tectonics, and has since undergone extensive filling and subsequent incision due to the passage of the Yellowstone hotspot (Thackray et al., 2011). Sedimentary deposits of alternating lacustrine and alluvial sediments in Marsh Valley indicate that there have been poor drainage patterns over the past 640 ka. During the late Pleistocene, Marsh Valley was the uppermost corridor of the Bonneville Flood, draining pluvial Lake Bonneville at Red Rocks Pass roughly 17.5 ka (Thackray et al., 2011; Amidon and Clark, 2014). The broad southern end of Marsh Valley carved into the basin fill suggests non-catastrophic overflow of Lake Bonneville before and after the catastrophic flood. The northern end of the valley is considerably narrower, confined by the incised Pleistocene alluvial fans and the inverted topography of the 430 +/- 70 ka Portneuf Valley Basalt which separated the Portneuf River and Marsh Creek as they run subparallel for ~30 km before joining together (Thackray et al., 2011; Rodgers et al., 2006).

The climate in the Marsh Valley is semi-arid, with an average precipitation of 400 mm in the valley and up to 500-760 mm at the highest elevations (Hammes, 2010; ISCC, 2002). Monthly temperatures range from an average high of 32°C in July to an average low of -8°C in January. Precipitation falls predominantly as snow between November and March, with peak runoff in Marsh Creek of 2-28 m^3/sec typically generated by rain on snow events in the early spring (Hammes, 2010). Summer base flows range from 0.3-1.4 m^3/sec and are sustained by numerous springs throughout the valley, but are reduced by agricultural surface and groundwater withdrawals during the growing season.

The spatial extent of this analysis encompasses the lower 70 kilometers of Marsh Creek from its confluence with the Portneuf River upstream to a landslide-dammed lake referred to locally as the 'Rat Pond' (figure 2.1).

2.2.1. Agencies and programs responsible for conservation

Conservation and restoration actions in Marsh Creek are fragmented in space and time because the majority of land within the watershed is privately owned. Conservation programs on private land requires willing landowners who have the financial means to help to pay for a portion of their projects. The main agencies responsible for the implementation of these cost-share programs are the Natural Resource Conservation Service (NRCS), the Idaho Department of Environmental Quality (IDEQ), and the Portneuf Soil and Water Conservation District (PSWCD), though other agencies often coordinate to assure success (table 2.1). Upland conservation and restoration projects have been carried out on public land in tributaries to Marsh Creek, but the main sources of sediment today come from the bed and banks of mainstem Marsh Creek, 93% of which is privately owned (Guilinger, 2017; Hammes 2010).

The NRCS has administered the majority of the programs in Marsh Valley including the Conservation Reserve Program (CRP), the State Acres for Wildlife Enhancement (SAFE) program, the Environmental Quality Incentives Program (EQIP), the Conservation Stewardship Program (CSP), and the Agricultural Water Enhancement Program (AWEP), and provides Conservation Technical Assistance (CTA). These cost-share programs help private landowners improve water quality, minimize erosion, improve habitat, increase agricultural efficiency, and minimize agricultural impacts on the surrounding ecosystems.

The IDEQ administers the Nonpoint Source Management Program and Section 319 of the 1987 amendments to the CWA. Section 319 of the CWA created cost-share programs for private landowners help improve water quality in water bodies that are deemed impaired (IDEQ, 2010). The 319 projects in Marsh Valley have focused on erosion from agricultural fields, limiting livestock access to streams, and stream bank restoration and revegetation projects.

2.2.2. Water quality issues

Marsh Creek used to be a different stream than what we observe today. In 1912 local fishermen complained to the game warden of a sawmill degrading the water quality in Marsh Creek, killing thousands of trout fry, stating that Marsh Creek was considered to be one of the best streams in Idaho for fishing (DHBC, 2016).

Marsh Creek is currently considered impaired under the CWA because it exceeds TMDLs for nutrients and sediment and does not meet the standards for its designated beneficial uses for cold water aquatic life and secondary contact recreation (EPA, 2016). Marsh Creek was first evaluated and considered impaired by the IDEQ in 1994 and TMDLs were approved in 2010 for total nitrogen, total phosphorus, and fine sediment (IDEQ, 1999, 2003). TSS thresholds of 35 mg/l and 80 mg/l, for low flows and high flows, respectively, were set in 2001. From 2004 to 2012, these thresholds were exceeded more than 50% of the time (Harris, 2018). As of 1965, the water in Marsh Creek degraded the water quality in the Portneuf River due to high levels of total solids and fecal coliform bacteria, which were attributed to poor agricultural practices and grazing on the northwest benches above Marsh Creek (Merrell and Onstott, 1965). Water quality samples taken between 1969 to 1974 throughout the basin indicated that the tributaries delivered large amounts of sediment to Marsh Creek, estimating SS loads an order of magnitude higher than we see today (McSorley, 1977). In 2017, the banks were identified as the primary source of suspended sediment, indicating that there has been a shift in the dominant source of fine sediment through time (Guilinger, 2017).

2.2.3. Timeline of human impacts and conservation response

A comprehensive history of land use impacts and conservation actions is necessary to compare against the long-term trends in the water quality data (figure 2.2). Agriculture and livestock production have supported the communities in Marsh Valley since the late 1800's (DHBC, 2016). Historic accounts of grazing reported 10,000 cattle in Marsh Valley in 1875 and 200,000 sheep in 1905 (Hammes, 2010), and since 1974, there have been an average of 25,400 head of cattle a year in Bannock County (USDA, 2017). There are an estimated 122 animal facilities identified in the Marsh Creek watershed, 35 of which utilize Marsh Creek as their only access to water (IASCD, 2013).

Initial studies of erosion and water quality indicated that the Marsh Creek watershed has been heavily impacted by overgrazing and poor farming practices, particularly in the uplands

(McSorley, 1977; Merrell and Onstott, 1965). In the mid-1980s the PSWCD, the Soil Conservation Commission, and IDEQ carried out restoration efforts in the Lone Pine and Arkansas Basins under the State Agricultural Water Quality Program (SAWQP) (Koester, 1995; Drewes, 1988). These projects were both very large covering a total of 140 km^2 (9,281 acres) and attempted to stabilize the soils on slopes surrounding Marsh Creek to the southeast by implementing soil retention structures such as terraces and sediment retention basins, as well as conservation tillage practices (figure 2.3) (Koester, 1995). The total cost to implement BMPs in these two projects was $554,506 with a cost share match of $359,017, totaling $948,523 (Koester, 1995; PSWCD, 1994). IDEQ conducted the first local effort to determine the effectiveness of BMPs by collecting water quality samples from pre- and post- BMP installation in the Lone Pine basin (Drewes, 1988). The report concluded that suspended solids and total phosphorus concentrations decreased, but not significantly due to the lack of pre- and post-treatment data.

The Conservation Reserve Program (CRP) began in 1986 and is the largest and most expensive program in the basin (figure 2.3). The State Acres for Wildlife Enhancement (SAFE) program works in conjunction with the CRP to provide habitat for the Sharp-tailed Grouse, allowing the cap on total acres in CRP to be raised (NRCS, 2017). From 1982 to 2016, 90% of the total dollars spent on conservation within the Marsh Creek watershed were through the CRP, totaling an estimated $56,710,420. Since 1986, an average of 205.6 km^2 (50,805 acres, figure 2.4) have been enrolled in the CRP, removing roughly 50% of the land occupying the upper benches from production and establishing vegetative cover to reduce soil erosion and provide habitat for the Sharp-tailed Grouse.

Near-channel actions did not begin until the mid-1990's (figure 2.2). In 1995, the Arimo Ranch, PSWCD, and IDFG carried out the single largest riparian restoration project that has occurred along Marsh Creek, funded by a $38,000 Section 319 grant (Scully et al., 2003). This project established 6,705 meters (22,000 ft.) of exclosure fence, planted 500 willows and enrolled 6.07 km^2 (1,500 acres) to prescribed riparian grazing where fences were not installed (table 2.1) (IDEQ, 2003). This project also included a post-implementation fisheries survey finding that the majority of fish sampled were Utah suckers, redside shiners, Utah chub, and carp with 2% of the sampled fish being salmonids including 6 brown trout, 4 cutthroat, 2 mountain whitefish and 1 rainbow trout (Scully et al., 2003). Stream bank stability was reported as improved, but the reach was still degraded with warm turbid water and a high width-to-depth ratio (Scully et al., 2003).

The Environmental Quality Incentives Program (EQIP) began in 1996 (table 2.1). Some of the main types of conservation practices that have been implemented as part of the EQIP program are grazing management, irrigation systems, and livestock confinement systems. EQIP is the second longest running program within the basin and has provided an estimated $2,052,280 in funding (figure 2.5) (EWG, 2017).

The Conservation Stewardship Program (CSP) has been active in Bannock County since 2010 and has provided an estimated $1,145,011 in funding. The CSP provides cost-share assistance to landowners to maintain and enhance existing BMPs including nutrient and grazing management plans.

From 2008-2015, a three-phase project funded by 319 grants, administered by the IDEQ, focused on reducing sediment, nutrient, and bacteria inputs to the stream from livestock feeding operations (table 2.1). These projects focused at areas located near or on Marsh Creek, and installed 15,451 meters of fencing, relocated 7 corrals off the stream or tributaries, built 12 off-stream watering troughs, 4 corral berms and created grazing management plans. The total cost of these projects was $1,002,074. This 319 grant also supported an increase in water quality monitoring efforts throughout the basin to identify and target source areas and longitudinal trends (IASCD, 2013).

2.3. Methods

2.3.1. Obtaining historical information and conservation actions

Anonymized summaries of conservation actions were obtained from the NRCS and IDEQ offices as well as from the USDA Agricultural Census, the Environmental Working Group (EWG), and manually digitized from satellite imagery (EWG, 2017; PSWCD, 1995; USDA, 3013, 2017).

NRCS farm bill programs were obtained as aggregate data at the Bannock County and Portneuf watershed scale. Conservation actions in the Marsh Creek watershed were estimated as 80% of the total amount for Bannock County based on a personal communication from Nate Matlack, District Conservationist with NRCS. The EQIP and CSP data used in this thesis were obtained from the EWG and contained annual data on the number of contracts and annual dollars spent (EWG, 2017). Reported dollars spent do not include the cost-share dollars spent by private landowners unless otherwise stated. Dollars spent were adjusted for inflation.

2.3.2. Water quality record

The water quality record was analyzed using the weighted regressions on time, discharge and season (WRTDS) trends model (Hirsch et al., 2010). The water quality record was collected as grab samples and turbidity measurements at two different sites along Mash Creek. The water quality record spans from 1969-2018, containing significant gaps between 1974 to 1979 and

1981 to 1991 (table 2.2). Turbidity was used as a proxy for TSS by developing a rating curve (Minella et al., 2008). The data used to develop this rating curve was collected at site 2 (figure 2.1) by the IDEQ between 2003 and 2012 and included 33 paired measurements of turbidity and TSS. The following linear regression was fit to the data and one outlier was removed based the Cooks Distance metric. The remaining points exhibited a strong correlation with an $R^2 = 0.94$ (Cusack, 2016).

$$tss\ (mg/l) = 2.7167 - 2.5935\ x\ turbidity\ (NTU) \qquad \text{Equation 2.1}$$

Because data were collected at both site 2 (a.k.a. 'Tripplett's' in Cusack, 2016 or IDEQ documents) and the USGS gaging station (site 13075000; a.k.a. 'site 6' in Guilinger, 2017) it was important to create a single representative measurement location within the basin. TSS data at the USGS gage were estimated based on data collected at site 2, 20.7 km downstream (figure 2.1). Contemporaneous TSS data collected by Guilinger in 2016 from site 2 and the USGS gage are well-correlated, but the correlation depended on antecedent conditions and discharge (Guilinger, 2017). First, the TSS data were averaged to a daily time step to account for the lag between the two sites. The data were then log-transformed and split at the 50% exceedance probability flow (1.7 m^3/sec or 60 ft^3/sec), based on different concentration-discharge relationships observed at high- and low-flows. A linear regression was then fit separately to the low-flow and high-flow datasets.

$$Q \le 1.7 \frac{m^3}{sec}: log\left(tss_{USGS\ gage}\right) = -2.748 + 2.252 *\ log(tss_2) \qquad \text{Equation 2.2}$$

$$Q > 1.7 \frac{m^3}{sec}: log\left(\ tss_{USGS\ gage}\right) =\ -0.864 + 1.367 *\ log(tss_2) \qquad \text{Equation 2.3}$$

In these equations, $tss_{USGS\ gage}$ is the TSS [mg/l] at the USGS gaging station, tss_2 is the TSS at site 2 [mg/l] and the log function has a base of 10. The models had R^2 values of 0.61 and 0.65 respectively. A discharge correlation between the two sites was much stronger than the TSS relationship with an R^2 of 0.92 (Harris, 2018). Although 87% of the TSS records were collected at site 2, the TSS record was shifted to the USGS gage, because the earliest records available are from the USGS gage.

2.3.3. Detecting long-term trends in water quality data

Measured and estimated water quality data for the USGS gage were analyzed using the Exploration and Graphics for RivEr Trends (EGRET) and EGRET confidence interval (ci) packages available in the R statistics software package to detect long-term trends in annual suspended sediment loads, as well as to provide graphical tools to display the data. This analysis utilized methods developed by Hirsch et al. (2010; 2015) that employ a weighted regressions model based on time, discharge, and season (WRTDS) to estimate a daily concentration for the period of record. The core of the WRTDS method is rooted in the equation for calculating the concentration for each day in the period of record:

$$\ln(C_{ij}) = \beta_0 + \beta_1 \ln(Q_{ij}) + \beta_2 T_{ij} + \beta_3 \sin(2\pi T_{ij}) + \beta_4 \cos(2\pi T_{ij}) + \epsilon_{ij} \quad \text{Equation 2.4}$$

where $\ln(C_{ij})$ is the natural log of concentration [mg/l] for a given day (i) of year (j), $\ln(Q_{ij})$ is the natural log of mean daily discharge [m^3/sec], T_{ij} is time [years] represents the long-term trend in the water quality record, $\sin(2\pi T_{ij})$ and $\cos(2\pi T_{ij})$ represent annual seasonal changes in the water quality record, β_{0-4} are fitted coefficients, and ϵ_{ij} is the residual error (Hirsch et al., 2010). This equation estimates a daily concentration for an unknown day using a locally-weighted three-dimensional regression where known measurements are assigned a weight based their distance in time, season, and discharge (Hirsch et al., 2010).

This method was used because it is one of the most flexible statistical models available that can calculate both concentration and flux. There are numerous advantages to the WRTDS method: it allows for unevenly spaced data with large gaps, it does not assume a static concentration-discharge relationship, it does not assume that the seasonal variations are static through time, and it makes no assumptions about the shape of the long-term trend (Hirsch et al., 2010). In addition, it generates a flow-normalized concentration (FNC) and flow-normalized flux (FNF) that eliminates interannual variations driven by the discharge record. The FNC and FNF calculations assume stationarity in the discharge record and are created by calculating each daily concentration from the probability distribution function of discharge built for each day of the year from all the flows that have occurred on that given day. Uncertainties in this method are quantified using the a block-bootstrap method that was developed to estimate the probability of type-one errors using Monte Carlo simulations to represent the variability within the dataset (Hirsch et al., 2015). Data is selected in blocks within the bootstrap method to account for short-term autocorrelation in hydrologic data such as storm events. A 90% confidence interval was used to increase the likelihood of detecting trends given that water quality data is often highly variable (Hirsch et al., 2015). Uncertainties in the flow-normalized trends are reported using the likelihood approach because it provides more intuitive utility for decision makers. Approximate p-values are also generated.

Two different time periods (1969-2017 and 1991-2017) were run using the WRTDS method to test the assumption of stationarity in the discharge record made in the FNF calculation. The results from these two different runs are considered separately due to the uncertainty in the quality and methods of the historic data sets. A complete list of specific functions and methods employed by the WRTDS method are not discussed here, the WRTDS method is detailed in Hirsch et al. (2010) and the block bootstrap method for estimating uncertainty is detailed in Hirsch et al. (2015).

2.3.4. Analyzing trends between water quality and BMPs

Individual conservation actions can be accounted for using many different metrics, such as dollars, area, length, and number. To include all types of conservation actions that reduced erosion and sediment loading, conservation investments were represented by the dollars spent on a project, excluding the cost-share match because this value was unknown for many of the projects. Annual dollars spent were adjusted for inflation then aggregated to cumulative annual dollars spent to best represent the impact many different projects throughout space and time might have on improving water quality.

Water quality was represented by the flow-normalized suspended sediment flux because the suspended sediment record is the longest and most complete water quality record available and reducing erosion and suspended sediment loads have been the primary target of conservation efforts. The long-term trends in suspended sediment flux were estimated using the flow-normalization method in the EGRET stats package to remove the interannual variations in hydrology.

The relationship between cumulative conservation investments and annual flow-normalized suspended sediment flux was assessed using a generalized least squares (GLS) model and a cross-correlation analysis. Autocorrelation was removed from the GLS model using an autoregressive moving average (ARMA). The ARMA order was selected using the AIC and BIC criterion. This analysis was performed using the Nonlinear Mixed-Effects Models (nlme) package in R statistics. The GLS model does not account for any lag time between the two datasets, thus necessitating the cross-correlation analysis.

The cross-correlation analysis was performed using the ccf function in R statistics to investigate the possible lag response times between conservation actions and measurable improvements in water quality. 1^{st}-3^{rd} order differencing was used on both the FNF and cumulative dollars spent to remove the autocorrelation and trend each dataset. The time period compared in both analyses was from 1982 to 2016, set by the first known conservation project (table 2.3).

2.3.5. Channel planform change detection

Historic aerial imagery provides a valuable reference point for measuring channel planform change. Shifts in channel planform measure the geomorphic response to land use changes, agricultural and livestock impacts, and conservation actions. This study compares aerial imagery from 1941 and 2013 for Marsh Creek from the confluence with the Portneuf River upstream 70 km to the 'Rat Pond,' documenting 72 years of cumulative change.

42 single frame black and white aerial photographs from 1941 were used in this study. The images have a scale of 1:31,680 and were scanned in color at 2400 dpi. This study took advantage of the structure-for-motion (SfM) software Agisoft Photoscan to georectify, georeference, and stitch the historic aerial images into a single orthomosaic raster (Agisoft LLC, 2016). Detailed information on the workflow used in Agisoft is available in Appendix A. The georeferenced raster was then imported into ArcMap 10.4 and georeferenced again to improve the spatial accuracy using 0.5 m resolution National Agriculture Imagery Program (NAIP) satellite imagery from 2013 and a 1-meter resolution DEM collected in 2015 (USDA, 3013; FEMA, 2016). Over 150 ground control points (GCPs) near the channel were placed between the

1941 raster and the NAIP and DEM datasets and fit using the spline transformation. The modern imagery used for comparison was NAIP orthoimagery taken in 2013 at 0.5 meter resolution. (USDA, 2013). Both the 1941 and the 2013 images were projected in the spatial reference frame NAD 1983 UTM Zone 12N and linear units were measured in meters.

Binary channel masks were extracted from each set of images by manually digitizing the wetted edge of mainstem Marsh Creek in ArcMap 10.4. Ideally, the bankfull width would be compared instead of the wetted width, but the resolution of the 1941 images was not fine enough to be able to distinguish a bankfull width, only the transition from water to land. Because of this compromise, measuring the channel's geomorphic response was only considered where the bankfull width is approximately the same as the wetted width. This validation was performed by visually checking the entire length of the channel to see if the bankfull width shown in the LiDAR DEM was equivalent to the digitized channel mask.

These georeferenced binary channel masks were then rasterized and imported into a set of analysis algorithms collectively called Spatially Continuous Riverbank Erosion and Accretion Measurements or SCREAM (Rowland et al., 2016) that measure changes along each channel bank pixel by pixel. This new method offers a comprehensive set of channel metric outputs and reduces errors that arise in measurements made using only the channel centerline. The channel was divided into segments that were defined as 200 times the mean channel width to better detect spatial patterns in sinuosity, mean width, erosion rate, and accretion rate.

The two main sources of error in this repeat aerial image analysis were image registration and feature identification (Rowland et al., 2016). The minimum image registration error in this analysis is set by the 0.5-meter pixel size for the 2013 NAIP imagery. The 1941 image registration error is more difficult to constrain due to the high local accuracy of the spline fit. The error associated with feature identification is set by the 1-meter uncertainty in delineating the wetted edge of each channel in both the 1941 and 2013 imagery. The uncertainty in wetted edge delineation was measured by taking the average of repeat measurements of the width of the transition from the water (dark pixels) to the land (light pixels) in the 1941 imagery in ArcMap 10.4. Using the simple rule of sums and differences, the uncertainty in the change in width was two meters.

2.4. Results

2.4.1. Water quality trends using WRTDS Model

2.4.1.1. Trends in suspended sediment flux

The annual suspended sediment flux varied over two orders of magnitude throughout the period of analysis, depending on the mean annual discharge (figure 2.6). Overall, the annual suspended sediment flux declined from 1969 and 2017. Between 1969 and 1973, when the first water quality samples were available, the mean suspended sediment flux was 44.1 x 10^6 kg/year. The mean SS flux in 2015-2017 was 5.4 x 10^6 kg/year, showing a large decrease over the period of record. Although there is an observed decline in the annual SS flux, the fluxes are periodic, varying with the annual meteoric trends of wet and dry years, making it difficult to discern when and at what rate water quality improvements occurred.

2.4.1.2. Trends in flow-normalized suspended sediment flux

The flow-normalized flux (FNF) of suspended sediment is an important component of this analysis as it is the only way in which we can remove the hydrologic influence on the annual SS flux. The FNF was determined for two different time periods (1969-2017 and 1991-2017) to test the assumption of stationarity. The 1991-2017 FNF trend is consistently less than and sub-parallel to the 1969-2017 FNF trend (figure 2.7C), but it is within the bounds of uncertainty, indicating that any changes in hydrology are less than the uncertainty in the trend, validating the use of the FNF method. The consistently lower values in the FNF trend for the 1991-2017 time period indicates that the trend may even be underestimated, as there were higher discharges earlier in the record that may be underestimated, and lower discharges later in the record that may be overestimated (figure 2.6).

The annual FNF has decreased 23.5 x 10^6 kg/year between 1969 and 2017, a 75% reduction (figure 2.7A). Using a 90% confidence interval, the block bootstrap method estimated that the likelihood that the FNF is trending down is 89%, with an approximate 2-sided p-value of 0.23. On average the annual FNF has decreased 4.9 x 10^5 kg/year^2, but the trend is non-linear. The most rapid decreases observed were 2.06 x 10^6 kg/year^2 and 2.11 10^6 kg/year^2, which occurred between 1997 and 2004, and 2010 and 2012, respectively. Short periods of increasing FNF occurred from 1996 to 1997 and from 2004 to 2010.

The FNF for the 1991-2017 period showed a similar sub-parallel trend to the longer-term record, but with lower FNF values (figure 2.7C). The annual FNF decreased 14.5 x 10^6 kg/year between 1991 and 2017, at an average rate of 5.6 x 10^5 kg/year^2, also showing a 75% reduction. Using a 90% confidence interval, the block bootstrap method estimated that the likelihood that the FNF was trending down was 89%, with an approximate 2-sided p-value of 0.24. The 1991-2017 did not have as pronounced periods of increased FNF but mirrored the 1969-2017 record during the time periods mentioned in the full record.

2.4.2. Correlation and lag between cumulative investment in BMPs and water quality

The time series comparison between cumulative conservation investments and FNF using the generalized least squares (GLS) regression model shows a -0.74 beta correlation, with a p = 0.0012 (figure 2.8). The best fit auto regressive moving average (ARMA) order was (1, 1), chosen using the lowest AIC and BIC criterion, which adjusted the model fit and significance test to account for autocorrelation in the data.

The cross-correlation analysis was performed on 1^{st}-3^{rd} order differenced data, measuring the change in progressive dollars spent and change in FNF between each year at 0 +/- 20 lags (figure 2.9). This analysis utilized a 34-year long record and revealed significant correlations of -0.41 and -0.39 at lags of 6 and 7 years, respectively, indicating that a measurable change in water quality occurs 6-7 years after the dollars are spent on conservation actions. 2^{nd} and 3^{rd} order differencing measured the change in progressive dollars spent and change in FNF between 3- and 4-year periods, respectively, throughout the period of record. These analyses also revealed significant correlations at 6- and 7-year lags. Due to relatively short record, significant correlations at large lags, especially in the higher order differenced data, may only be based on a few pairs of data points, so these correlations, although significant at 15 years, were not considered robust in this analysis.

2.4.3. Channel planform change using SCREAM

The geomorphic response to the integrated land-use changes and conservation actions between 1941 and 2013 measured using repeat aerial image analysis have revealed both natural and anthropogenic changes in the channel as well as human modifications that occurred prior to 1941. The comparison of wetted widths did reveal some segments where this analysis was invalid due to large changes in width driven by irrigation withdrawals (figure 2.10, Figure B.1), but in many sections throughout the basin, the wetted width and bankfull width were found to be equivalent.

The changes in the length of channel from 1941 to 2013 were revealed through the changes in sinuosity (figure 2.11). Overall, the length of the channel was reduced 1.2 km, but both shortening and lengthening were observed throughout the basin. The channel moved naturally in very few places via meander migration, avulsions, or cutoffs. Anthropogenic straightening only occurred in a few locations in the lower 30 km where the channel length was reduced 2.2 km. A 10-km-long, previously straightened section in the upper basin was identified in the 1941 imagery, providing a minimum age for this highly modified section (figure 2.11). Just downstream from this straightened section, the channel lengthened by 1.2 km, a 2.5% increase.

Five sections within the analysis were deemed invalid due to large differences between the wetted width and the bankfull width identified in the 2013 imagery (figure 2.10). Widening was observed throughout the basin but was often within the 2 meters of uncertainty. The greatest widening occurred in a segment 15 km upstream from the confluence, widening 3.5 meters, a 45% increase. The lower 30 km of Marsh Creek widened an average of 2.2 meters and this change was qualitatively validated using the imagery from both time periods.

2.5. Discussion

2.5.1. The impact of land use and conservation on suspended sediment flux

During the 20th century the main source of sediment to Marsh Creek has shifted from the upland benches of Marsh Valley (McSorley, 1977; Merrell and Onstott, 1965) to near-channel sources (Guilinger, 2017; Scully et al., 2003), but the timing and magnitude of this shift has been unknown. Similar shifts in sediment source have been observed in the agricultural catchments throughout the Midwest (Schilling et al., 2011; Belmont et al., 2011).

The observed shift in dominant sediment source, the decreasing FNF, the strong negative correlation between the FNF and dollars spent on conservation, and the cross-correlation analysis provide supporting evidence that upland BMPs were effective at reducing fine sediment inputs from erosion on the upper benches (figure 2.3). The greatest increase in conservation actions occurred between 1988 to 1993 with the CRP and SAWQP implementing BMPs on 28% of the watershed. The largest observed decrease in the flow-normalized flux (FNF) occurred between 1992 and 2004 and was likely in response to the reduced sediment inputs from the upper benches. This delayed response in FNF to conservation actions is captured in the cross-correlation analysis where significant correlations were found at lags of 6 and 7 years (figure 2.9), affirming the expectation that measurable improvements in water quality occur years after investments in conservation have been made (Meals et al., 2010).

Near-channel erosion was not identified as a dominant source in the early reports, but livestock have been present in Marsh Creek since the early 1900s impacting the stability of the banks via grazing and the use of Marsh Creek for water. The stabilization in the FNF in 2004 may represent reduction of upland sources but the persistence of lowland bank sources (figure 2.7). This inference is supported by the work done by Guilinger (2017) indicating that the sediment in transport today is coming from the banks and supported by the observed increases in channel width between 1941 and 2013. A simple back-of-the-envelope calculation using the total sediment flux exported between 1991-2017, assuming a bulk soil density of 1300 kg/m^3, reveals that if all this sediment were sourced from a 75-kilometer long, 1-meter tall bank, the channel would have had to widen 2.1 meters. Although this is just an estimation, it indicates that it is plausible and consistent with our imagery analysis that over the past 25 years the banks were the primary source of sediment.

A second large decline in the FNF occurred between 2010 and 2012. This change in the FNF could be a response to the progressive near-channel actions and the initiation of the Section 319, phase 1 grant in 2008 but no quantitative analysis was made due to the short duration of the change. Near-channel actions began in 1995 with the riparian restoration project at Arimo Ranch, followed by the beginning of EQIP in 1996 (table 2.2). Although the initiation of these projects may have contributed to the large decrease in FNF between 1992 and 2004, it was likely not the dominant driver of this change due to the expected lag time of 6 to 7 years from the cross-correlation analysis (figure 2.3). The 30 off-channel watering troughs installed since 1996 have also played a role in reducing streambank erosion, but these do not remove livestock from grazing the riparian vegetation, making them less effective at stabilizing the banks than riparian exclusion fencing.

The FNF decreased significantly through time, but the rate of change has declined since 2004, and the SS concentrations are still exceeding the TMDL limit over 50% of the time (figure 2.7). The upland source of SS identified in initial reports (McSorley, 1977; Merrell and Onstott, 1965) was treated with over 50% of this source area enrolled in conservation programs, but visual inspections of the banks from a kayak survey and from field visits indicate that there are still ubiquitous bank failures occurring today (Guilinger, 2017). Bank instabilities occur due to improper grazing practices, the use of the stream for stock water, and increased hydraulic conveyance due to channelization (Guilinger, 2017; Peppler and Fitzpatrick, 2005). Conservation programs targeting bank instabilities began in the mid-1990s but have only treated an estimated 15% of the banks throughout the mainstem. Future declines in the FNF due to streambank stabilization projects are likely to occur more gradually through time, compared to the decline between 1997-2004 associated with the reduction in upland sediment input, due to the ubiquitous bank failures, available funding, and willing landowners to implement projects.

Although a strong correlation exists between cumulative dollars spent and FNF and significant 1st order differenced cross-correlations at 6- and 7-year lags were observed, BMPs are not exclusively responsible for changes in the FNF. Other factors may have influenced this outcome including technique modernization, changes in irrigation practices or regional environmental drivers. It is also important to consider that the aggregate effect of many individual practices intended to resolve watershed level resource concerns can fail because projects implemented at the reach scale are too fragmented in space and time, impacting only a small fraction of the watershed (Feld et al., 2011; Bernhardt et al., 2005).

2.5.2. The impact of land use and conservation on hydrology and channel planform geometry

There has been a declining trend in Marsh Creek's mean annual discharge since 1985 (figure 2.6). The trend in mean annual discharge is closely related to the climatic patterns of wet and dry years, and since 1985, both discharge at the USGS gage and maximum snow water equivalent (SWE) at the Wildhorse Divide SNOTEL site have been declining (figure 2.12). This decline in available water is correlated to larger climatic shifts, but the declining trend in mean annual discharge is occurring at a greater rate, shown by the trend in the normalized-differenced values through time (figure 2.12), indicating that the change in discharge may also be affected by other components of the watershed mass-balance. If discharge is decreasing at a greater rate than the decrease in precipitation, there could be a change in the rate of evapotranspiration (ET), a change in the groundwater system, or a change in storage.

An increase in ET could be due to well-intentioned conservation practices. For example, part of the EQIP provides funding for water use efficiency practices, switching from flood irrigation to sprinkler systems, which reduces the amount of water used and reduces sediment and nutrient inputs to the stream. The water needed to irrigate a given plot of land is less using sprinkler irrigation, but the proportion of consumptive use (water that is used to create biomass, or lost to ET), to non-consumptive use (loss to ground water) is much greater with sprinkler irrigation, reducing groundwater recharge, and increasing ET (Van Kirk, 2014). The installation and use of ground water wells has increased due to these changes in irrigation and urbanization throughout Marsh Valley (USDA, 2018b). Marsh Creek is fed by numerous springs throughout the basin, and the increase in ground water pumping and reduced ground water recharge have likely caused reduced summer base flows, increased instream deposition, and increased instream temperatures. The persistent high SS concentrations during low flows are also associated with strong diel signals. SS concentrations peak during the night and are thought to be caused by in-stream activity of nocturnal biota such as carp and crayfish (Cusack, 2016).

Another possible and less constrained or understood mechanism for a change in inputs in the watershed mass balance in Marsh Creek is changing flows in the Portneuf River. There is likely a subsurface connection between Marsh Creek and the Portneuf River where they run parallel for the northernmost 30 km until their confluence and due to the Portneuf Marsh Valley Canal, but this has not been studied in depth. Between McCammon and the confluence in Inkom, the Portneuf River sits on average 27 meters higher than Marsh Creek, while only being on average 2.1 km away, separated by the Portneuf basalt flow that overlies Quaternary gravels, and older alluvial, lacustrine, and volcanoclastic sediments of the Salt Lake Group. (Thackray et al., 2011) This topographic evidence for a hydraulic head gradient and highly permeable substrate between the two streams are strong evidence for a subsurface connection. This is also one possible explanation why the flows in Marsh Creek increase dramatically in this lower section.

The hydrology in Marsh Creek has also been impacted from straightening the channel to create more arable land, especially in areas that were historically wet meadows and shallow marshes (figure 2.11). Straightening the channel increases its gradient, increasing transport capacity and erosion, ultimately reducing the frequency of overbank flows. Disconnecting the channel from its floodplain also reduces the lag between the peak in precipitation and the peak in discharge, increasing peak flows due to increased water conveyance (Belmont and Foufoula-Georgiou, 2017). The straightening prior to 1941 in the upper basin could have caused a cascading effect whereby the channel directly below has lengthened and the channel in the lower basin responded by incising and widening to accommodate for the increased magnitude of channel-forming flows.

2.5.3. Changes in flow-normalized flux within the WRTDS model

The annual FNF is a critical component of this analysis because it removes the interannual periodicity in the SS flux driven by wet and dry years, enabling changes in land use and conservation efforts to be directly compared against long-term trends in water quality. One caveat of the flow-normalization is that the calculation of the flow-normalized load assumes stationarity in the discharge record, meaning that it should not be applied where management actions have “substantially” modified streamflow (Hirsch et al., 2010).

However, humans’ impact on the hydrologic cycle is unequivocal and that any statistical assumptions regarding stationarity in water quality and quantity are likely false (Milly et al., 2008). Thus, it is unlikely that the assumption of stationarity is true in any discharge record of sufficient length for use in the WRTDS method (at least 20 years). Practically speaking, the flow-normalized function should not be used if large changes in hydrology have occurred, such as the construction of a large dam, construction or removal of large diversions, or large changes in the consumptive use of water (Hirsch et al., 2010).

There have been no substantial instantaneous changes in land use or water withdrawals in Marsh Creek between 1969 and 2017, but conservation actions, small channel modifications, and climatic shifts have all played a role in the declining trend in flow over time. The difference in the FNF between the two time periods, 1969 to 2017, and 1991 to 2017 indicate that the discharge record is not stationary through time, but importantly this difference is within the 90% confidence intervals (figure 2.7). Furthermore, the drop in the FNF line for the shorter time-period indicates that the trend may even be underestimated, as there were higher discharges earlier in the record that would underestimated and lower discharges later in the record would be overestimated (figure 2.6).

2.5.3.1. Uncertainty in water quality data

The earliest water quality study found for Marsh Creek was published in 1977 and contained grab samples taken between 1969 and 1974 (table 2.2). This report stated that there were 7 sampling sites, but in the figures and data tables the location of the samples was “near McCammon,” and two different sampling sites were shown on the associated maps near McCammon. One was likely at the USGS gauging station, and the other at a road crossing a few km downstream. Although the exact location is unknown, the samples should be representative of the same area used in the long-term trend analysis. Another source of uncertainty in the 1969 to 1974 data is that turbidity was measured in JTU’s as a proxy for TSS. JTU’s and NTU’s are generally equivalent but JTU’s are a historic measurement made visually using a Jackson Candle Turbidimeter, and NTU’s are measured using a digital sensor that measures the backscatter of light at a 90-degree angle. The rating curve used to convert turbidity to TSS was developed using unspecified data prior to this study. Still, the equation used was very similar to Equation 2.1 developed by Cusack (2016).

$$tss = 34.32 + 3.31(JTU) \qquad \text{Equation 2.5}$$

The WRTDS model was built to process water quality records at the daily time step. Aggregating data to the daily timestamp when turbidity measurements are made every 15 min could misrepresent the true behavior if the stream was very flashy, but turbidity data collected by Stalder in 2017 at site 2 show that the maximum standard error was 5 NTU for a mean daily turbidity of 77 NTU (Stalder, 2018). The mean standard error in the mean daily turbidity between May and October of 2017 was 0.34 NTU, showing that aggregating 15-minute data to a mean daily timestamp does not decrease the representativeness of the data (Stalder, 2018).

The transformation of the long-term TSS record from site 2 to the USGS gage shifted 2618 or 87.7% of the daily TSS measurements. The TSS record was shifted to the USGS gage because shifting the discharge record to site 2 would still require shifting TSS records from the USGS gage to site 2, and these records comprise the earliest and most poorly documented records available.

Although there are varying degrees of uncertainty in each data set and in the transformations made, it was important to include all available data to generate a sufficiently long data set to detect long-term trends. The 90% confidence intervals (figure 2.7) generated by the block bootstrap method do not incorporate the uncertainties in each measurement, but they do span a large range, and despite these large uncertainties, it remains highly likely that the flow-normalized flux is trending down.

2.5.4. Channel planform changes measured using SCREAM

The 1941 orthomosaic image is a unique dataset for measuring planform change and a valuable outreach tool to show the local community what the land looked like ~70 years ago. Aerial images have played a pivotal role in the success of conservation and restoration actions by showing landowners the previous state of the creek or river on their property prior to their acquisition and management. This is exemplified with in two local restoration projects on Rapid Creek, and Pebble Creek, two tributaries to the Portneuf, where straightened sections of channel were put back into old meanders because the landowners were shown aerial photos of the channel prior to straightening (IASCD, 2013).

A comparison of the two generations of aerial photos did not reveal large meander migrations, avulsions, or cutoffs as one might have expected in a meandering stream over 70 years. This may be due to the low gradient, fine-grain nature of Marsh Creek. Meander migration normally occurs through bedload deposition on the point bar of a bend, forcing the main flow of the river towards the outside of the bend causing erosion (Leopold and Langbein, 1966). In Marsh Creek it is difficult for the fine sediment deposited on point bars to become established due to remobilization during storm events and melt periods during the spring. Prior to any human modification, Marsh Creek likely overtopped its banks more frequently, building its floodplain via deposition and meander migration enabled by more coarse bed sediment. The straightened section in the upper basin observed in the 1941 area is surrounded by polygonal features, indicating that this area was previously a shallow marsh not freeze-thaw features (see appendix B for examples). Channelization increased arable land and reduced flooding in this region, but this also reduced overbank deposition, increasing the longitudinal conveyance water and suspended sediment loads, and could have been one of the driving mechanisms causing bank instability downstream where the channel then adjusted to these changes.

Though the majority of changes in width were within the uncertainty in the measurement, after field inspection, the banks of the lower 30 km of Marsh Creek appear to have nontrivial widening (figure 2.10). This observed widening does not directly measure changes in the bankfull widths, but it appears that in both image sets that the water in the channel is fully covering the bed, and we know from field visits and previous kayak surveys that the banks in this area are steep, unvegetated, and failing in many locations (Guilinger, 2017). Furthermore, the lower 30 km was the only area to also show significant straightening between 1941 and 2013, indicating that the changes in channel width may be in response to the local straightening.

2.5.4.1. Sources of uncertainty in repeat aerial image analysis

The greatest source of uncertainty associated with generating the channel masks used in the SCREAM analysis are from the digitization of the 1941 channel banks and in the precision of georeferencing the 1941 imagery. The root mean square error (RMSE) from the 95 ground control points (GCPs) used in Agisoft was 11.4 meters. The Agisoft output was georeferenced in ArcMap using a spline fit and over 140 GCPs were placed near the channel, producing a RMSE of 0.01 m. This very small RMSE is because the spline transformation maximizes local accuracy for each GCP, a true 'rubber sheet' method, but it is difficult to rely on this very small RMSE knowing it is only accurate very close to where each GCP was placed. Most of the calculated changes in width, erosion rate, and accretion rate, fall within the bounds of uncertainty, but visual inspection confirms widening in lower reaches (figures 2.10 and 2.11). The widening and straightening in the lower 30 km provide additional supporting evidence to the conclusions made based on sediment fingerprinting and a longitudinal sonde array study, showing that the source of sediment in Marsh Creek is derived from the banks and that the longitudinal flux of SS increases in the lower 30 km (Guilinger, 2017).

The second source of uncertainty was in delineating the wetted edge of the channel in the 1941 imagery due to the large representative scale of the imagery (1:31,680). The wetted edge was chosen over the bank full width because it was very difficult to discern where the bankfull transition would be in the 1941 black and white imagery with no associated digital elevation model (DEM). In many sections throughout the basin, the channel does not contain an inset floodplain and is approximately rectangular, allowing for the wetted edge and bankfull width to be roughly equivalent. Previous field surveys have shown that Marsh Creek has high width-to-depth ratios and steep banks (Guilinger, 2017; Scully et al., 2003). If the flow is high enough to cover the bed of the river and the banks are vertical, then the wetted width and the bankfull width will be very similar. The bankfull width was found to be equivalent to the wetted edge throughout much of the basin, but one section near the confluence and four sections in the upper basin were identified where significant amounts of water are diverted, creating a rapid change in wetted width that is not representative of the bankfull width.

Using the wetted edge is also possibly problematic in that the flows should be the same between each time step. The 1941 aerial images were taken on three dates: 9/17/1941, 10/3/1941, and 11/8/1941. There was not a USGS gauging station on Marsh Creek until 1954, but the Portneuf at Pocatello gauge located downstream of Marsh Creek and provides a reference point to the historic flow data. Flows were 105 ft^3/sec., 111 ft^3/sec., and 168 ft^3/sec. respectively, on those dates, suggesting some variation in discharge. The 2013 NAIP imagery was taken on 11/4/2013 and the Portneuf at Pocatello gauge was at 143 ft^3/sec. Although there are differences in the flow, even within the 1941 images, photographs were taken in the same season, and the flows between the two time periods are not associated with any large storm events or summer base flow conditions. Variation in flows could cause a change in the wetted width, but the change detected would likely be within the 2-meter uncertainty associated with bank delineation.

2.5.5. Lag times between conservation efforts and water quality improvements

The lag time between conservation actions and improved water quality downstream is dependent on the sources of sediment, the BMPs used, and the rate at which sediment is flushed out of the system. Lag time is an important metric to discuss when assessing conservation efforts because measurable improvements may not occur for many years.

Mean velocities for suspended sediment have been estimated in the mid-Atlantic Region to be from 0.01 to 0.1 km/year, indicating that a single grain of silt or clay originating in the headwaters of Marsh Creek could take between 700 to 7000 years to leave the system (Pizzuto et al., 2014). This transport time is so long because a single grain is only in transport a fraction of the time and is stored in the channel or on the floodplain for much of its time within the basin. If the channel is largely disconnected from its floodplain and the transport of SS only occurs within the channel, which is the case in much of Marsh Creek, these transport rates could be greatly accelerated, and it could only take 2 months for sediment to travel from the headwaters to the confluence. This theoretical modeling perspective is important to acknowledge as changes in sediment delivery within the watershed will not result in direct changes in the water quality at the outlet. Studies that directly measured sediment load data to quantify the lag time associated with sediment delivery and transport found a large range of lag times ranging from 8 years to greater than 50 years (Brooks et al., 2010; Meals et al., 2010).

The cross-correlation analysis of the 1st order differenced values provided a Marsh Creek specific response time to conservation actions of 6 to 7 years. This is important to acknowledge because conservation actions will not generate an immediate response in the water quality measurements, and future monitoring efforts should be made for at least 10 years post-conservation to capture any changes in the water quality record.

2.6. Conclusion

Marsh Creek has changed significantly over the historic period (DHBC, 2016). Early agricultural and grazing practices caused significant erosion of the upper benches and banks, degrading water quality (Merrell and Onstott, 1965; Guilinger, 2017). Historical analysis indicates that the main source of sediment has shifted from the upland benches of Marsh Valley

(McSorley, 1977; Merrell and Onstott, 1965) to near-channel sources (Guilinger, 2017). Conservation actions have followed the source of sediment, beginning with the SAWQP and CRP on the upland benches, then shifting to near-channel actions under EQIP and Section 319 grants. Detection of long-term trends using the WRTDS model indicate that FNF has decreased an estimated 23.5 x 10^6 kg/year between 1969 and 2017, a 75% reduction (figure 2.7). Using a 90% confidence interval, the block bootstrap method estimated that the likelihood that the FNF is trending down is 89%. The largest observed decrease in the FNF occurred between 1992 and 2004 and was likely in response to the reduced sediment inputs from the upper benches (figure 2.3), which is supported by the 6- and 7-year lag times identified in the cross-correlation analysis (figure 2.9). This response is also supported by the strong negative correlation between the cumulative dollars spent on conservation and the long-term trend in FNF. The stabilization in the FNF between 2004 and 2010 may represent a shift in the dominant sediment from the uplands to the banks (Figure 2.7). Bank instabilities have been observed throughout the basin and near-channel conservation efforts have yet to control this source because of difficulty coordinating the mosaic of private landowners to adopt improved practices. Furthermore, near-channel conservation actions require willing landowners and funding through cost-share programs that currently do not have adequate funds to address the near-channel issues throughout the basin.

The geomorphic response to the integrated land use changes and conservation actions between 1941 and 2013 measured using repeat aerial image analysis show that much of the anthropogenic impacts to Marsh Creek were present in 1941 with three significant findings. First, the channel moved naturally in very few places due to meander migration, avulsions, or cutoffs within the measurement error of 2 meters (figure 2.11). Second, the channel in the upper basin had already been straightened prior to 1941 (figure 2.11). Third, significant widening has occurred, especially in the lower 30 km, providing another line of evidence that the source of suspended sediment in Marsh Creek is coming from the banks (figure 2.10).

An estimated $62.9M, or $84.5M adjusted for inflation, was spent on conservation programs within Marsh Creek since 1982, with 90% of this funding going to CRP. Over 85% of the upland benches have been a part of either the SAWQP or the CRP and 44 km of fencing and 30 off-channel watering troughs have aimed to reduce the impact of livestock on the streambank erosion. Unfortunately, these considerable efforts have not been enough to improve the suspended sediment loads below the target for the TMDL limits. Between 2004 and 2012 Marsh Creek exceeded its high and low flow limits more than 50 % of the time (IDEQ, 2018). On average, the suspended sediment concentrations in Marsh Creek are two times higher than the Portneuf River at their confluence, necessitating continued implementation of BMPs and strengthening the importance of monitoring and research.

The combination of the WRTDS method and the cross-correlation analysis is a unique and powerful method for quantifying the lag time between conservation actions and changes in long-term water quality trends. Repeat aerial image analysis also provided a unique insight into areas within the watershed that have undergone anthropogenic modifications and how the channel has responded. These methods can be applied to other watersheds with suitable imagery, conservation, and water quality datasets to gain a watershed-level response to anthropogenic impacts and conservation, and more efficiently guide future efforts.

2.7. Future efforts

The most pressing future work within the Marsh Creek watershed is continued bank stabilization, water quality monitoring, and education. Currently the DEQ is the only agency monitoring water quality in Marsh Creek via monthly grab samples at site 2, and continued monitoring is vital for reevaluating the success of continued conservation efforts. The water quality sonde should be redeployed at site 2 due to the poor water quality in Marsh Creek and to continue assessment of past and present conservation efforts.

Research conducted over the past six years by geologists, ecologists, ecosystem service scientists, and political scientists as part of the Managing Idaho's Landscapes for Ecosystem Services (MILES) program has provided new insights into the interactions of humans and the landscape, along with the changes that have occurred, and insights into the dominant physical processes at work at the landscape scale. Engaging the community within Marsh Creek and the greater Portneuf watershed in a discussion of how to improve water quality is an important next step to improving the ecosystem services that our rivers and streams provide. In April 2018, a special projects grant for $250,000 was awarded to the NRCS. This 2-year grant targets near-channel conservation actions in the lower basin of Marsh Creek, utilizing recent findings made at Idaho State University to justify the project. A considerable amount of work is still needed to stabilize the ubiquitous bank failures, but this special awards grant and continued outreach into the community will aid in the reductions of suspended sediment loads in Marsh Creek. Although the tributaries do not contribute the bulk of the sediment load to Marsh Creek, recent monitoring efforts by the IDEQ have identified Gordon Creek, Hawkins Creek, and Birch Creek, all tributaries to Marsh Creek, as having the worst water quality out of all the stream segments in the Portneuf River watershed.

Due to the complexities in using water quality to determine conservation effectiveness, developing a reduced complexity model that estimates annual costs and sediment load reductions associated with different combinations of conservation practices would be a very useful decision-making support tool, and was recently done in the Minnesota River basin with great success

(Belmont and Foufoula-Georgiou, 2017). The largest challenge to successful watershed modeling is the quality of the data used to calibrate the model. Marsh Creek is a unique watershed that contains long-term water quality and quantity records, climate data, complete topographic, soil, and land use layers, making it a good candidate for future modeling efforts (Osmond et al., 2012).

The data collected in this study could be further used to investigate different methods of quantifying the lag times and trends in pollutant loads. There is a general lack of follow up studies in river conservation and restoration science, and although the estimation of lag times and pollutant loads is very important from a management and monitoring standpoint, very few studies have been performed (Schilling et al., 2011; Meals et al., 2010).

This study also identified that the flows in Marsh Creek have been decreasing since the 1980s. Future efforts should investigate the potential causes of this decline, as well as the role of groundwater in moderating the water quality in Marsh Creek, and any possible changes to the groundwater system that have occurred throughout time. Water is our most precious resource, and it is our responsibility to preserve its quality and quantity for future generations.

Chapter 2 Tables

Program	Agency(s)	Active dates	Estimated cost	Main BMPs utilized
SAWQP-Lone Pine	PSWCD IDEQ ISCC NRCS	1985-1995	$327,275 match: $297,846	conservation tillage, residue management, pasture and hay planting, terraces, water and sediment control basins
SAWQP- Arkansas Basin	PSWCD IDEQ ISCC	1982-1992	$226,232 Match: $97,171	conservation tillage, residue management, pasture and hay planting, terraces, water-and sediment-control basins
CRP + SHARP	NRCS	1986 - present	$56,710,000	access control, conservation cover, upland wildlife habitat management
EQIP	NRCS	1996 - present	$2,052,280	off-channel watering facilities, irrigation systems, fencing
CSP	NRCS	2011-2014	$1,141,011	continue or enhance existing conservation practices
Arimo Ranch	PSWCD, IDFG, IDEQ	1995	$38,000	fencing, bioengineering, prescribed grazing
319 phases I,II,III	IDEQ, PSWCD, Conservation Basics LLC	2008-2015	$1,002,074 Match:$752,572	off-channel watering facilities, corral relocations, berms, fencing

Table 2.1: List of all known programs and projects active since 1980 in Marsh Creek and the main BMPs used. The Section 319 program still receives funding but currently does not have any active projects. The costs listed in this table are not adjusted for inflation.

Date range	Agency	Location	Type
1969-1974	IDOHW-DOE[a]	McCammon	grab samples
1979-1981	USGS	USGS Gage	grab samples
1991-2001	USGS	USGS Gage	grab samples
2003-2012	IDEQ	Site 2	continuous
2003-Present	IDEQ	Site 2	grab samples
2016-2017	ISU-Guilinger[b]	USGS Gage	continuous
2017-2018	ISU-Stalder[c]	Site 2	continuous

Table 2.2: Date ranges, agencies, locations (see figure 2.1), and type of sample taken for the suspended sediment record utilized in the WRTDS analysis (figure 2.7). Records taken at "site 2" were shifted to be representative of TSS at the USGS gage (see section 2.3.2 for details). Grab sample frequency varied from bi-weekly to bi-monthly. Continuous monitoring recorded turbidity values every 10- to 15-minutes. 1969-1974 records taken at McCammon do not have an exact sampling location, but the only sampling locations near McCammon would be the USGS gage or a road crossing ~ 2 km downstream from the gage.

[a]: (McSorley, 1977)

[b]: (Guilinger, 2017)

[c]: (Stalder, 2018)

Year	FNF	cumulative dollars spent	change in dollars spent	change in FNF
1982	22.08	$29,196	--	--
1983	22.76	$35,152	$5,956	0.68
1984	21.09	$72,993	$37,841	-1.67
1985	21.99	$432,484	$359,491	0.9
1986	19.05	$832,829	$400,345	-2.94
1987	19.24	$3,544,564	$2,711,735	0.19
1988	17.73	$6,736,942	$3,192,378	-1.51
1989	16.8	$9,762,409	$3,025,467	-0.93
1990	19.05	$12,635,362	$2,872,953	2.25
1991	22.17	$15,394,103	$2,758,740	3.12
1992	26.2	$18,069,919	$2,675,817	4.03
1993	24.88	$20,650,681	$2,580,761	-1.32
1994	23.13	$23,153,725	$2,503,044	-1.75
1995	21.9	$25,648,260	$2,494,535	-1.23
1996	21.9	$28,012,078	$2,363,818	0
1997	24.74	$30,229,638	$2,217,560	2.84
1998	21.4	$32,663,494	$2,433,856	-3.34
1999	18.15	$35,083,443	$2,419,949	-3.25
2000	16.87	$38,508,973	$3,425,530	-1.28
2001	15.75	$41,889,594	$3,380,621	-1.12
2002	13.93	$45,262,606	$3,373,012	-1.82
2003	11.6	$48,624,584	$3,361,978	-2.33
2004	10.35	$51,864,199	$3,239,615	-1.25
2005	10.48	$55,274,370	$3,410,171	0.13
2006	11.05	$58,710,996	$3,436,626	0.57
2007	11.59	$62,057,662	$3,346,666	0.54
2008	12.06	$65,041,699	$2,984,037	0.47
2009	12.56	$68,037,281	$2,995,582	0.5
2010	13.18	$71,123,150	$3,085,869	0.62
2011	11.08	$73,298,043	$2,174,894	-2.1
2012	8.97	$75,810,993	$2,512,949	-2.11
2013	8.34	$78,132,139	$2,321,147	-0.63
2014	8.08	$80,632,879	$2,500,740	-0.26
2015	7.95	$82,692,024	$2,059,145	-0.13
2016	7.87	$84,462,268	$1,770,244	-0.08

Table 2.3: Overlapping record (34 years) of cumulative dollars spent, adjusted for inflation, and the associated annual flow-normalized flux (FNF) of suspended sediment in 10^6 kg/year. Change in dollars spent, adjusted for inflation, and change in FNF were used in the 1st order differenced cross-correlation analysis (figure 2.9).

Chapter 2 Figures

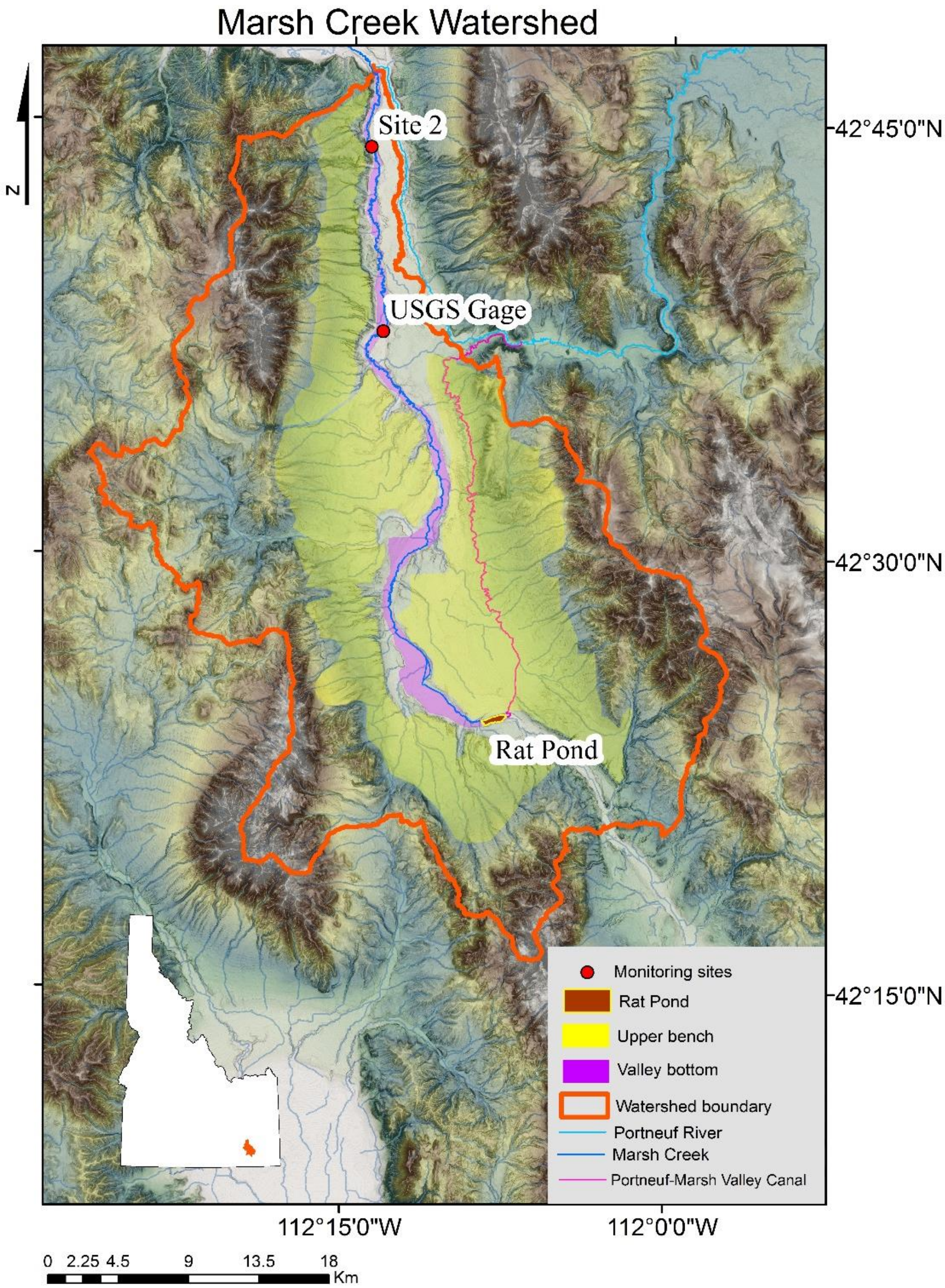


Figure 2.1: Map of the Marsh Creek watershed in southeastern Idaho. The focus of this analysis covers the main stem of Marsh Creek from the Rat Pond north to the confluence with the Portneuf River. The orange line is the watershed boundary, and the yellow shaded area shows the upland benches surrounding Marsh Creek. The purple shaded area shows the lowland valley adjacent to Marsh Creek where irrigated crops and livestock grazing occurs. The Marsh Valley Canal is shown as a purple line and transfers water from the Portneuf River south into Marsh Valley to irrigate the southeast benches.

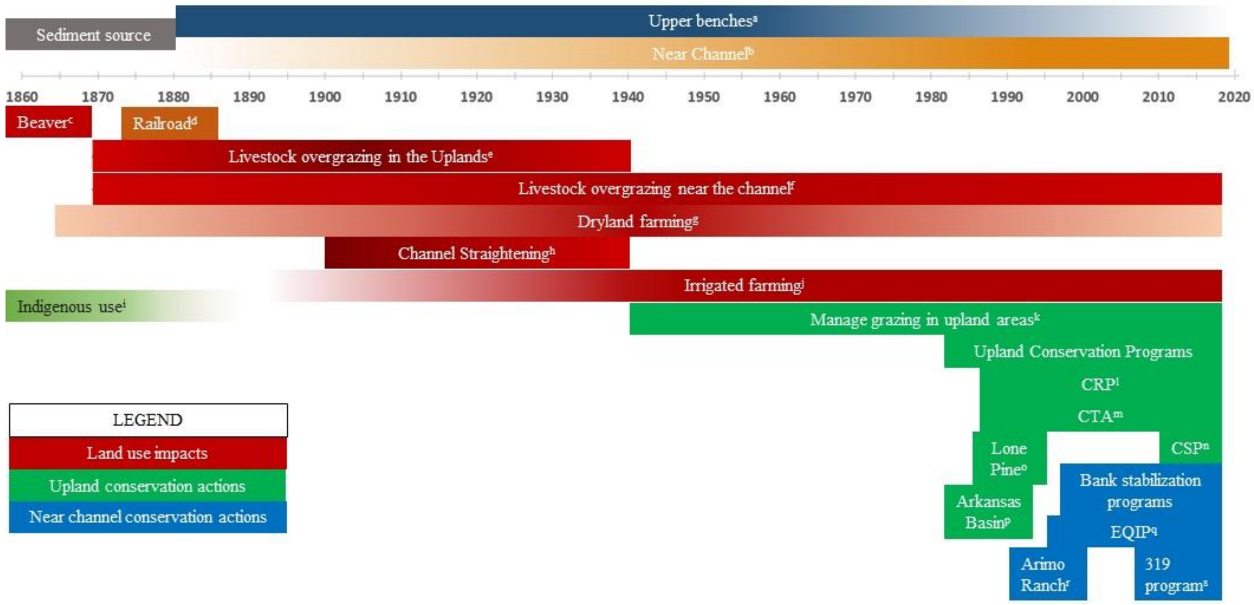


Figure 2.2: Timeline showing approximate start and end dates of human impacts in red, conservation efforts in the uplands in green, and near-channel actions in blue. The intensity of the color is intended to represent the approximate intensity of each land use impact on sediment loads in Marsh Creek. The length of the bars represents the approximate timeframe over which different impacts and conservation actions occurred.

[a]: Upper benches were the primary source of sediment in 1965 (Merrell and Onstott, 1965).
[b]: The banks were identified as the primary source of sediment in 2017 (Guilinger, 2017).
[c]: The 1st settlers were fur trappers, removing beaver from the region (DHBC, 2016).
[d]: The U & N railroad was built in 1879, but removed in 1882 (Link and Phoenix, 1996).
[e]: Overgrazing on slopes destabilized soils, causing severe erosion (Merrell and Onstott, 1965).
[f]: The lowland valley is used for livestock grazing and overwintering (ISCC, 2002).
[g]: Dryland farming on the upper benches was a major source of sediment (McSorley, 1977).
[h]: Aerial photograph analysis. Exact timing unknown. Minimum age set by 1941 imagery.
[i]: Shoshone and Bannock tribes inhabited Marsh Valley prior to settlers arrival (Sanger, 2018).
[j]: Irrigated farming occurs in the lowland valley (ISCC, 2002; DHBC, 2016).
[k]: Grazing management on the National Forest began as early as the 1930s (Hammes, 2010).
[l]: The Conservation Reserve Program is administered by the NRCS (EWG, 2017).
[m]: Conservation Technical Assistance provides free assistance to landowners (NRCS, 2018c).
[n]: The Conservation Stewardship Program enhances conservation practices (NRCS, 2018b).
[o]: Lone Pine basin, part of the State Agricultural Water Quality Program (PSWCD, 1995).
[p]: Arkansas basin, part of the State Agricultural Water Quality Program (PSWCD, 1994).
[q]: The Environmental Quality Incentives Program is administered by the NRCS (USDA, 2018b).
[r]: The Arimo Ranch project was funded by Section 319 Grants (IASCD, 2013).
[s]: Section 319 Grants are administered by the IDEQ under the CWA (IASCD, 2013).

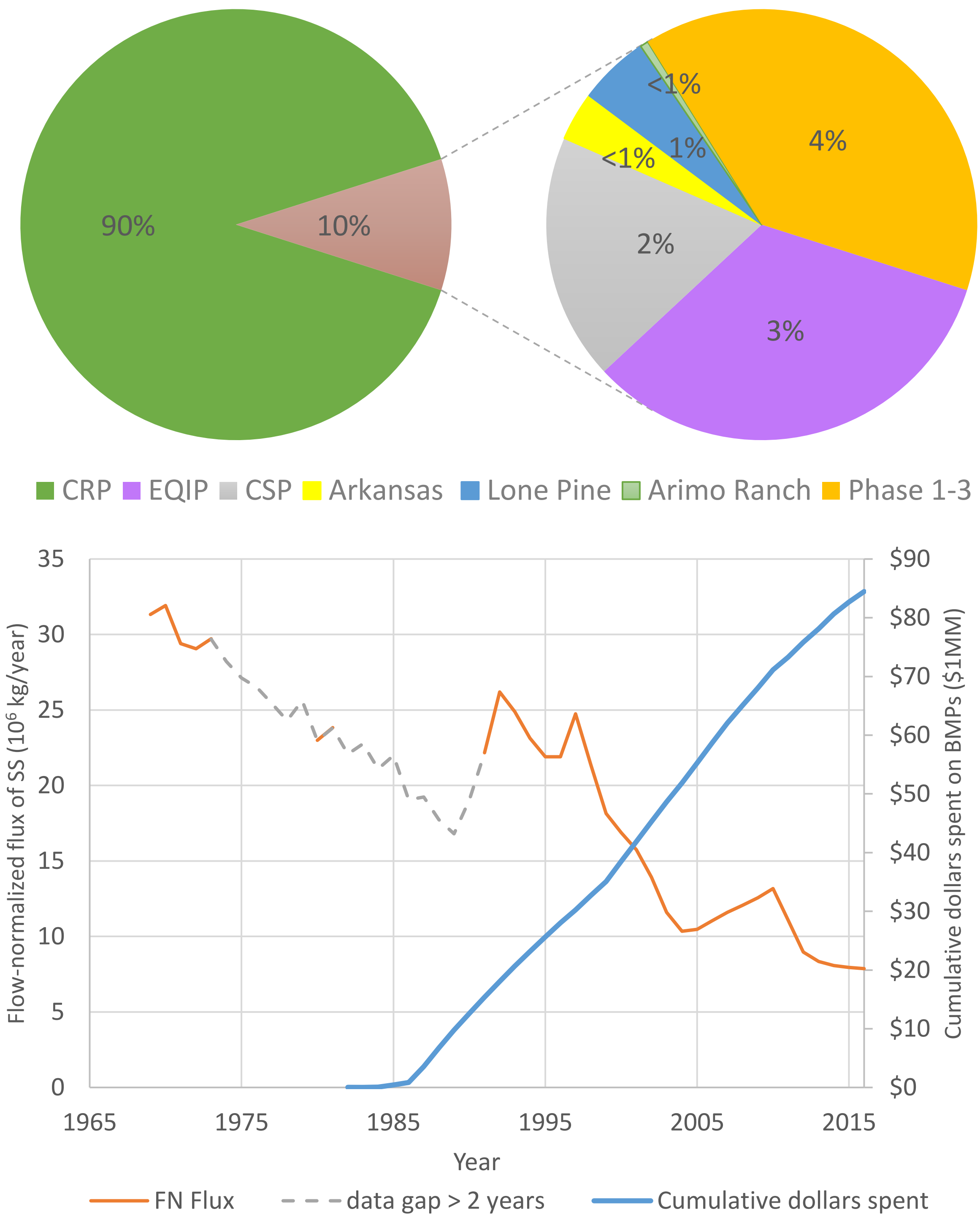


Figure 2.3: Top: Pie chart showing the percent of total dollars spend in each program or project, not adjusted for inflation. Bottom: Flow-normalized flux of annual suspended sediment from 1969 to 2017 plotted on the left y-axis and the cumulative dollars spent on BMPs, adjusted for inflation on the right y-axis. These two data sets are compared using the GLS regression (figure 2.8) and the cross-correlation analysis (figure 2.9). Data gaps greater than 2 years are estimated using the WRTDS method.

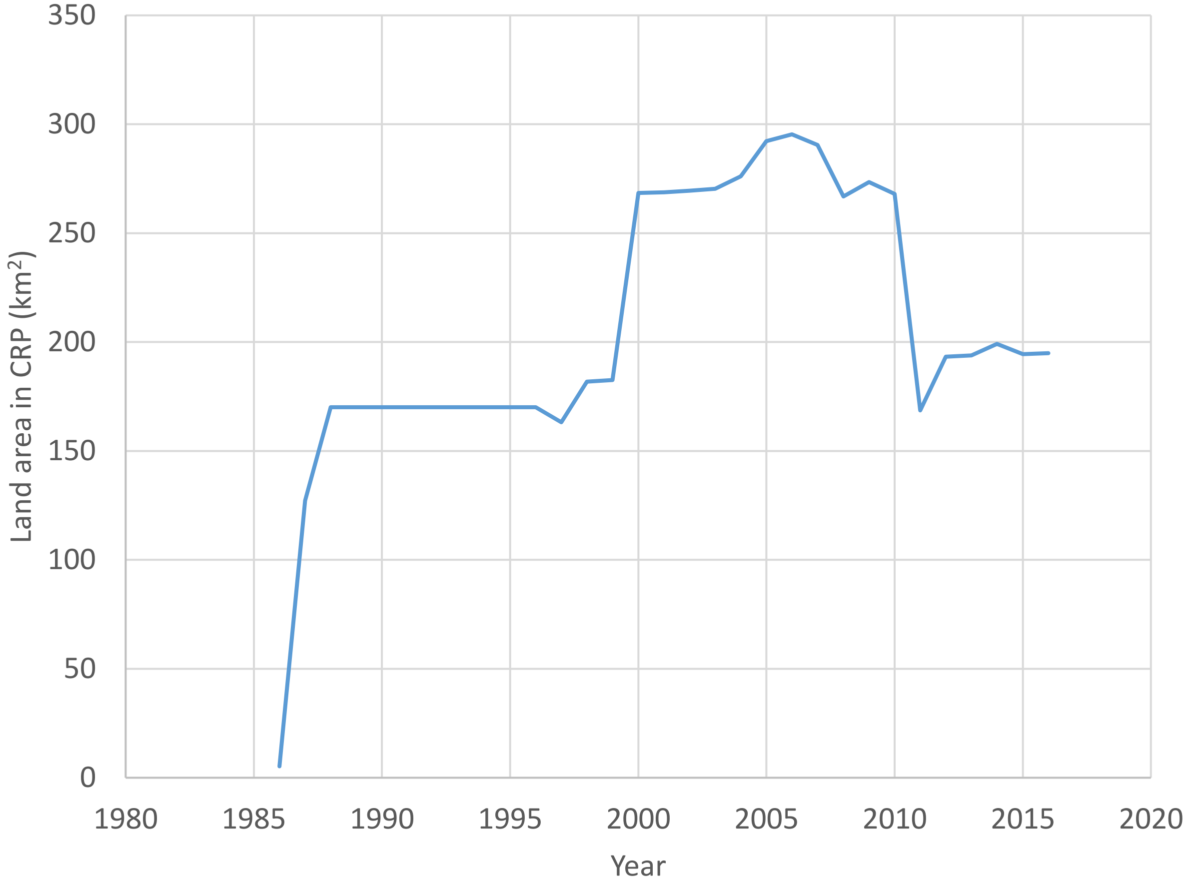


Figure 2.4: Land area enrolled in the Conservation Reserve Program (CRP) since its inception in 1986. Declines in enrolled area since 2010 are due to changes in funding within the US Farm Bill. Interviews with landowners and managers indicate that even after a 10-year CRP contract expires the land may not return to production due to the marginal yields produced (Taylor, 2018).

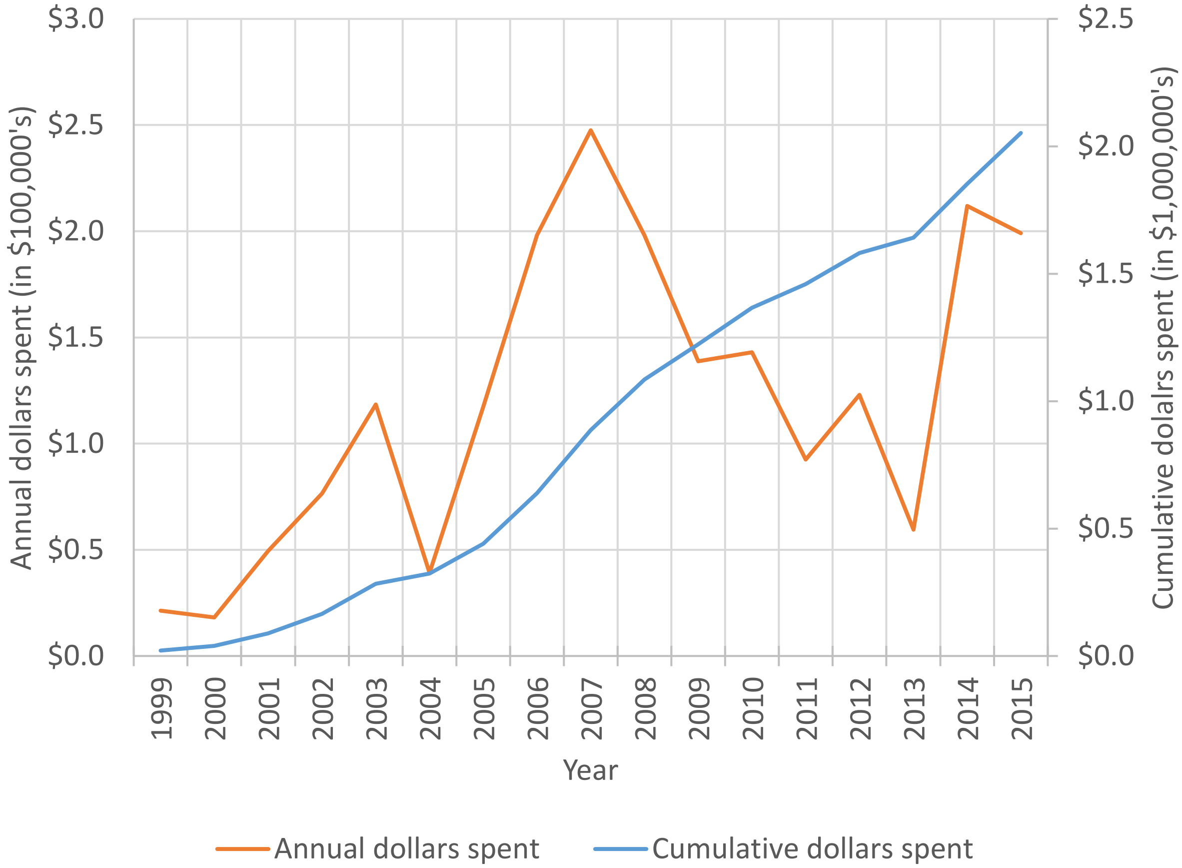


Figure 2.5: Annual and cumulative money spent, adjusted for inflation, in the Environmental Quality Incentives program (EQIP) in the Marsh Creek watershed. Data was obtained from the Environmental Working Group at the Bannock County level from 1999 to 2015. The amount of dollars spent in the Marsh Creek watershed is calculated as 80% of the total dollars spent in Bannock County based on a personal communication from Nate Matlack, District Conservationist with NRCS.

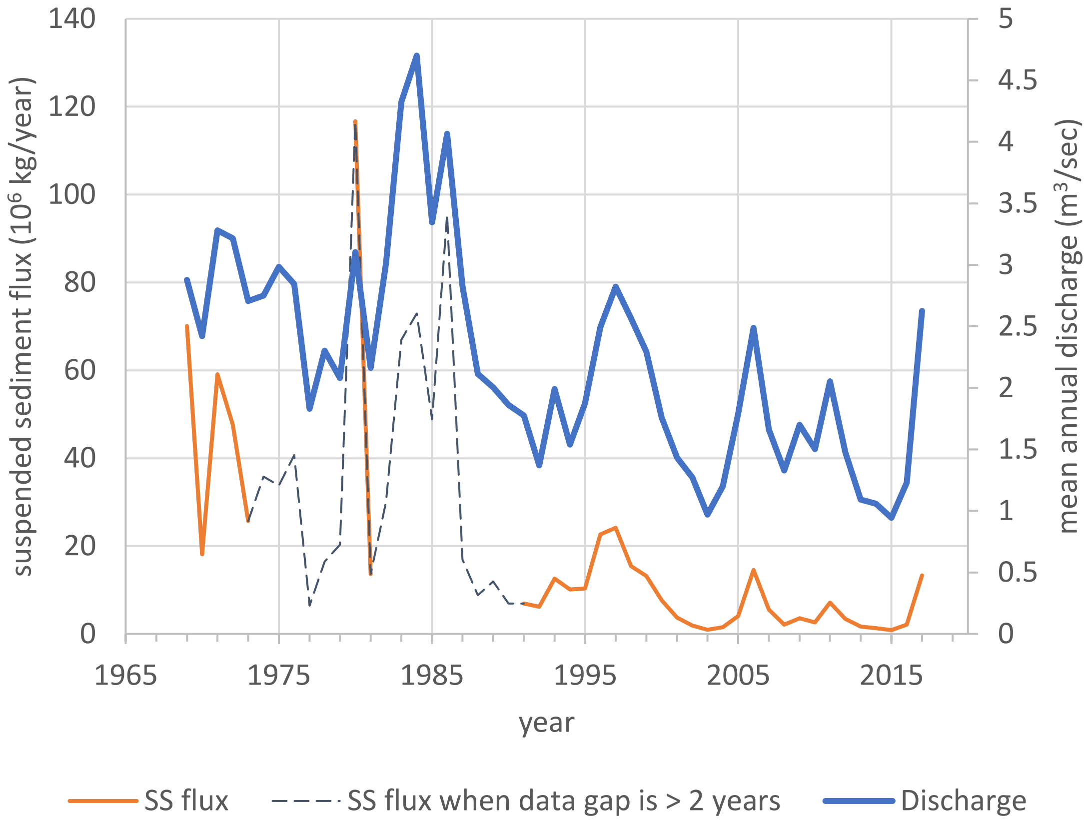


Figure 2.6: The modeled suspended sediment (SS) flux from 1969-2017 shown in orange line, with grey dashed line representing time periods where data gaps are greater than 2 years. Mean annual discharge in m^3/sec from the USGS gaging station near McCammon, ID shown as a blue line. The strong dependence of SS flux on discharge necessitates the flow-normalization method to determine the relationship between long-term trends in water quality data and conservation actions.

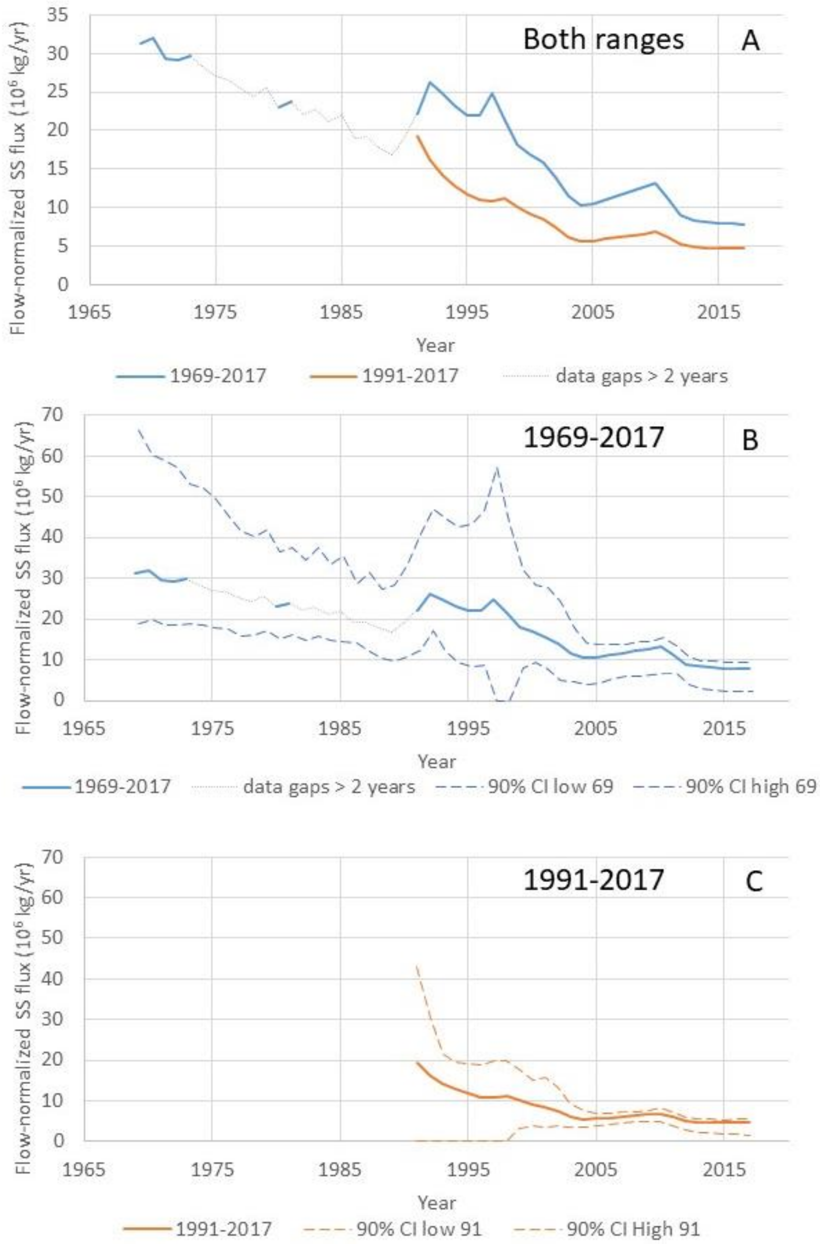


Figure 2.7: A: Annual flow-normalized flux (FNF) of suspended sediment (SS) in Marsh Creek for the full period of record, 1969-2017, with 90% confidence intervals generated by the block bootstrap method showing an estimated 23.5 x 10^6 kg/yr reduction. B: FNF of SS from 1991-2017, with 90% confidence intervals showing an estimated 14.5 x 10^6 kg/yr reduction. C: FNF of SS for the two periods of analysis. The two different time periods utilize the same TSS record but, due to the different periods of analysis, include different discharge records. This comparison was preformed to test the assumption of stationarity in the hydrologic record. Although the 1991-2017 FNF trend is consistently less than and sub-parallel to the 1969-2017 FNF trend, it is within the bounds of uncertainty, indicating that there have not been any substantial changes to the hydrology, validating the use of the FNF method. The drop in the FNF line for the shorter time period indicates that the trend may even be underestimated, as there were higher discharges earlier in the record that may underestimate the FNF and lower discharges later in the record that may be overestimate the FNF (figure 2.6).

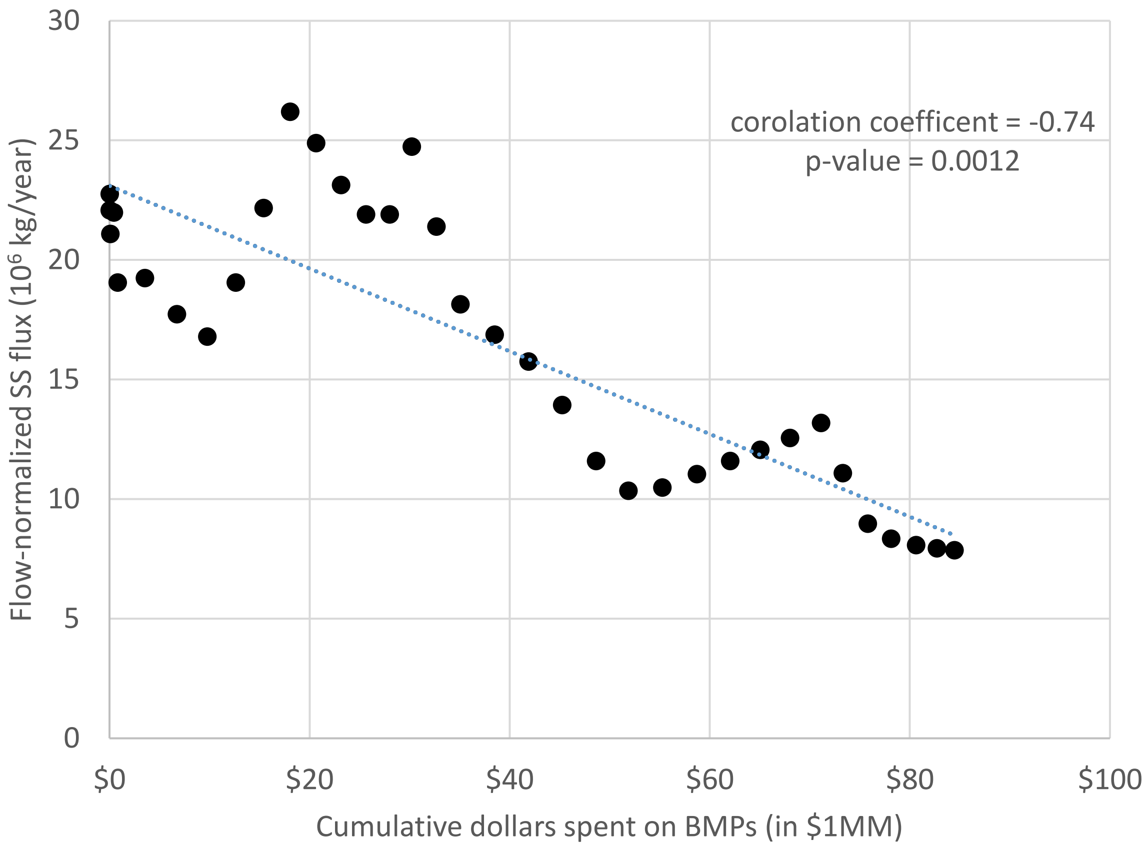


Figure 2.8: Generalized least squares regression between cumulative dollars spent on conservation actions, adjusted for inflation, on the x-axis and flow-normalized flux (FNF) on the y-axis. This regression accounted for autocorrelation using an ARMA model of order (1,1). A significant beta correlation of -0.74 was found with a p-value of 0.0012, suggesting that the progressive implementation of conservation actions has improved water quality over time. This model does not account for any lag time between conservation actions and the FNF, and because each dataset has a linear trend, it is likely that they would be well correlated. This relationship was tested further using a cross-correlation analysis (figure 2.9) to assess the influence of lag times and the trend in each dataset.

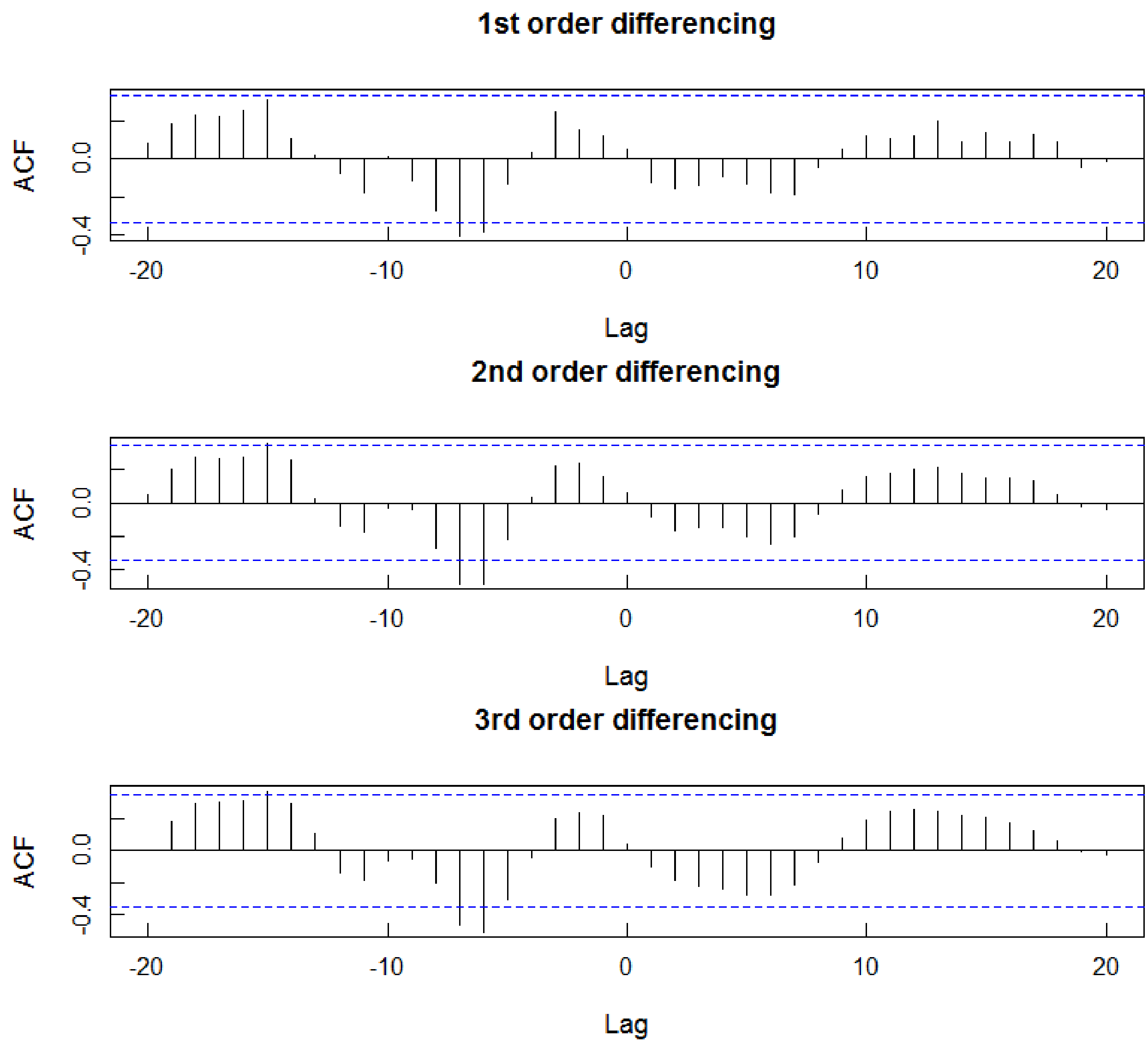


Figure 2.9: Cross-correlation analysis between the progressive dollars spent (independent variable) and flow-normalized flux (FNF) of suspended sediment (SS) (dependent variable). The number of lags in years is on the x-axis, and the auto-correlation function is on the y-axis, similar to the correlation between the two variables. The horizontal dashed lines in each plot are the 95% confidence intervals. 1^{st} order differencing utilized a 34 year long record, measuring the change in progressive dollars spent and change in FNF between each year at 0 +/- 20 lags. Lags of -6 and -7 exceed the confidence intervals, indicating that there is a significant relationship (-0.41, and -0.39, respectively) between the dollars spent on conservation and the FNF. 2^{nd} and 3^{rd} order differencing measured the change in progressive dollars spent and change in FNF between 3- and 4-year periods, respectively, throughout the period of record. Due to relatively short record (34 years), significant correlations at large lags, especially in the higher order differenced data, may only be based off a few pairs of data points, so these correlations, although significant at -15 lags, were not considered robust in this analysis.

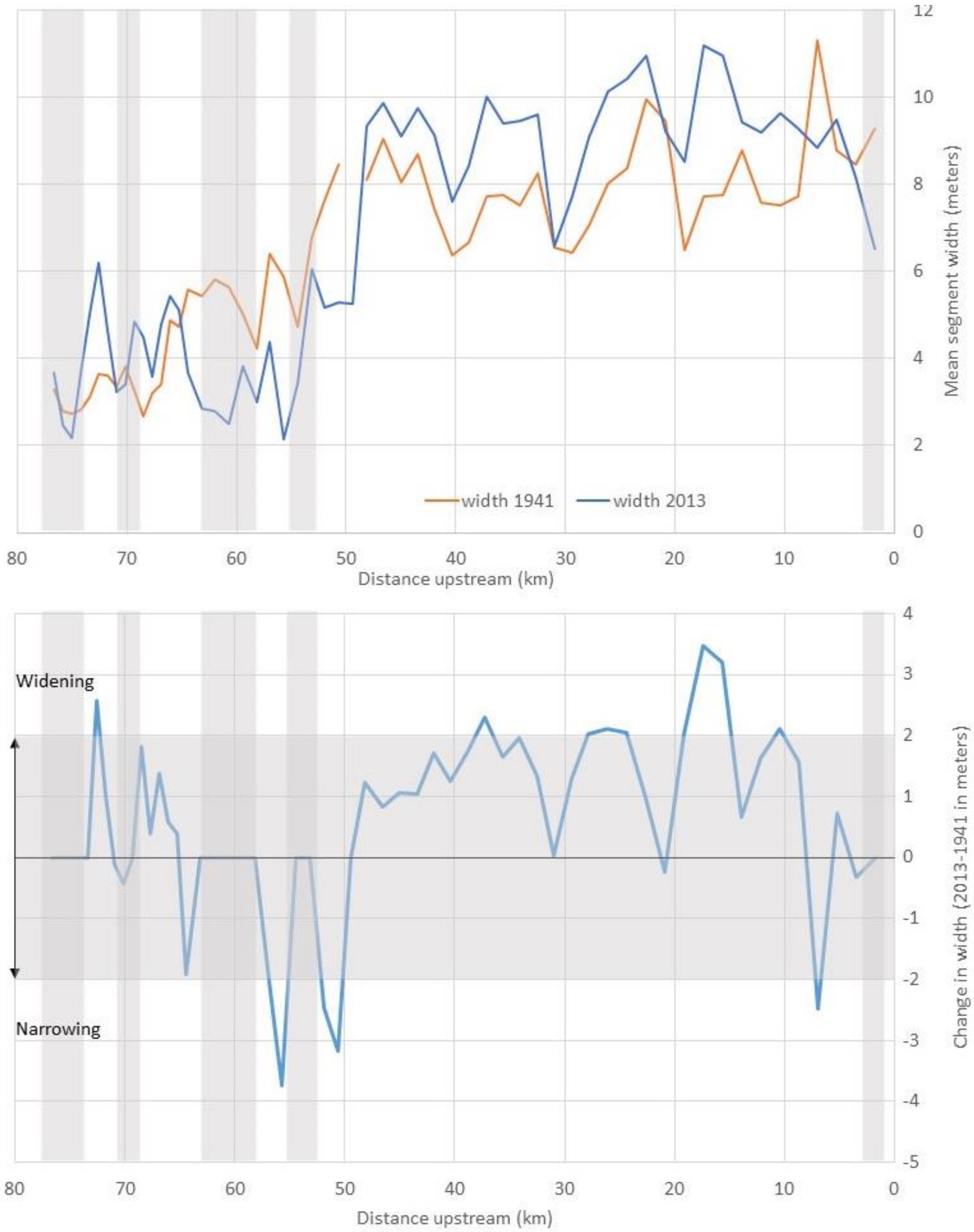
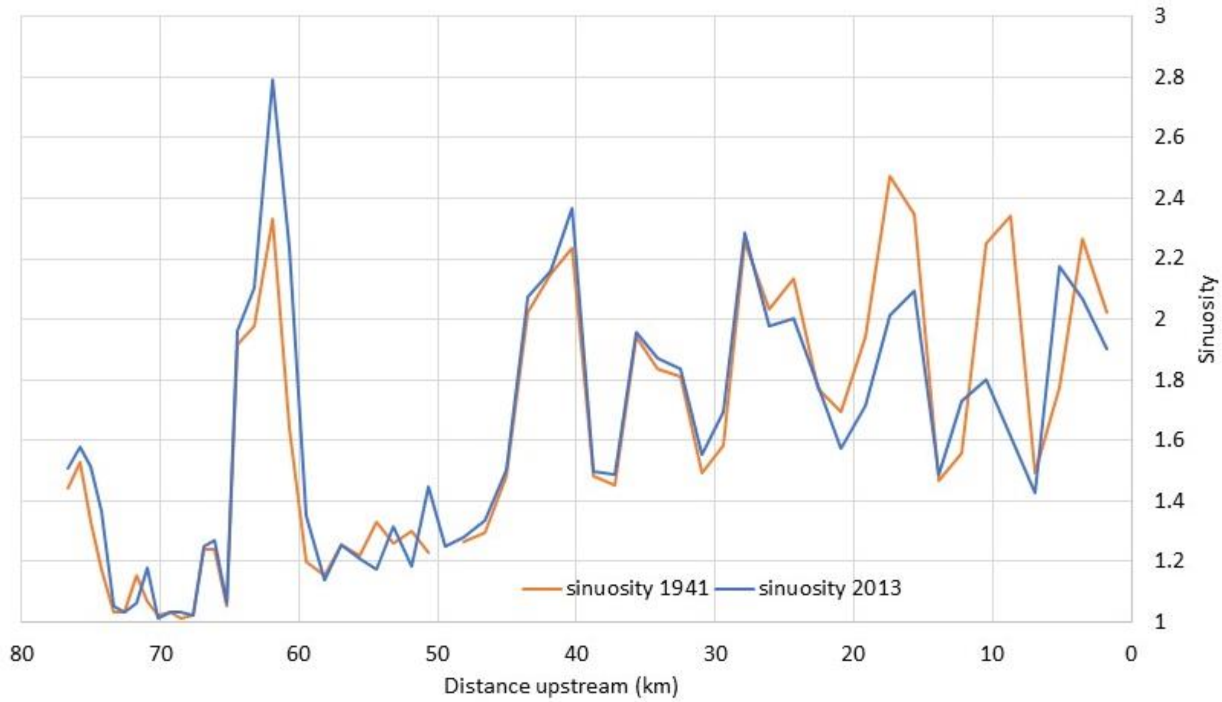


Figure 2.10: Top: Mean segment channel width for 1941 (orange) and 2013 (blue) on the y-axis and distance upstream from the confluence on the x-axis. Vertical grey bars indicate areas that were identified as invalid due to discrepancies between the wetted width and bankfull width identified in the 2013 imagery. These discrepancies were often associated with large irrigation withdrawals. The gap in the 1941 width at 50 km upstream is due to channel lengthening where the 2013 channel had an extra segment in the analysis. Bottom: Change in width showing widening as positive values and narrowing as negative values. Uncertainty in the change in width is 2 meters, shown by the horizontal grey box, making most of the change detected within the measurement uncertainty.



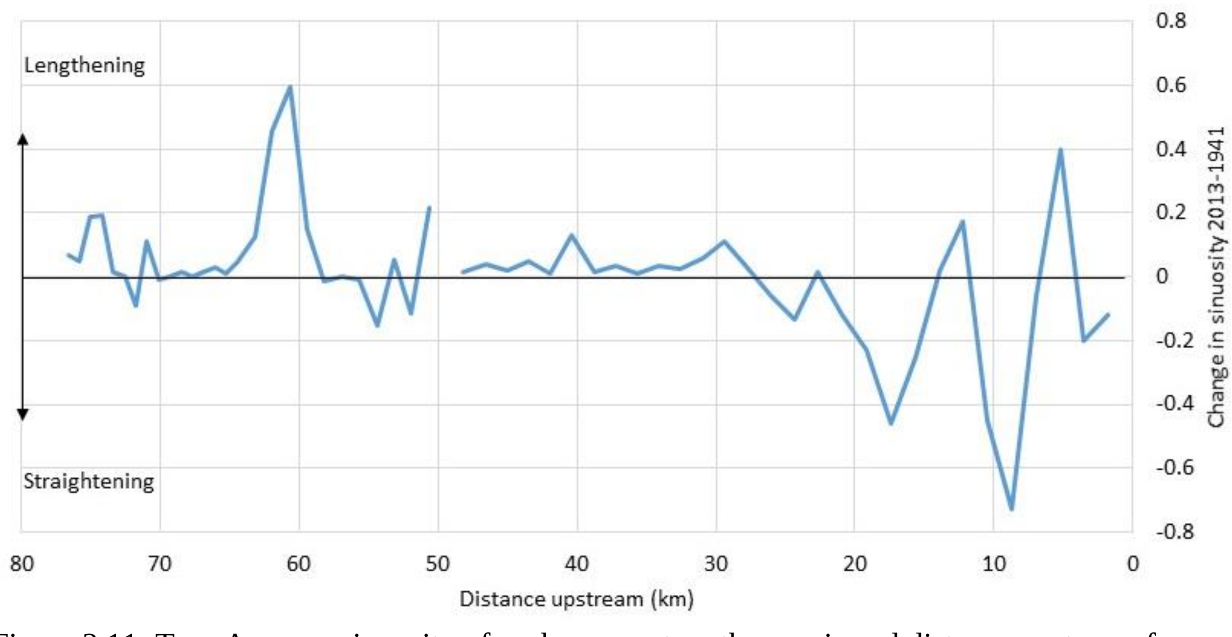


Figure 2.11: Top: Average sinuosity of each segment on the y-axis and distance upstream from the confluence on the x-axis. The orange line represents the 1941 channel, and the blue line represents the 2013 channel. Bottom: Change in sinuosity on the y-axis, indicating changes in channel length, both naturally occurring and anthropogenically induced. Negative changes in sinuosity indicate straightening, positive changes indicate lengthening. The lengthening at ~62 km upstream occurred just downstream from the ~10 km straightened section just upstream. Straightening occurred in the lower 30 km and, after visual inspection, appear to be associated with anthropogenic modifications as shown by prolonged straight sections.

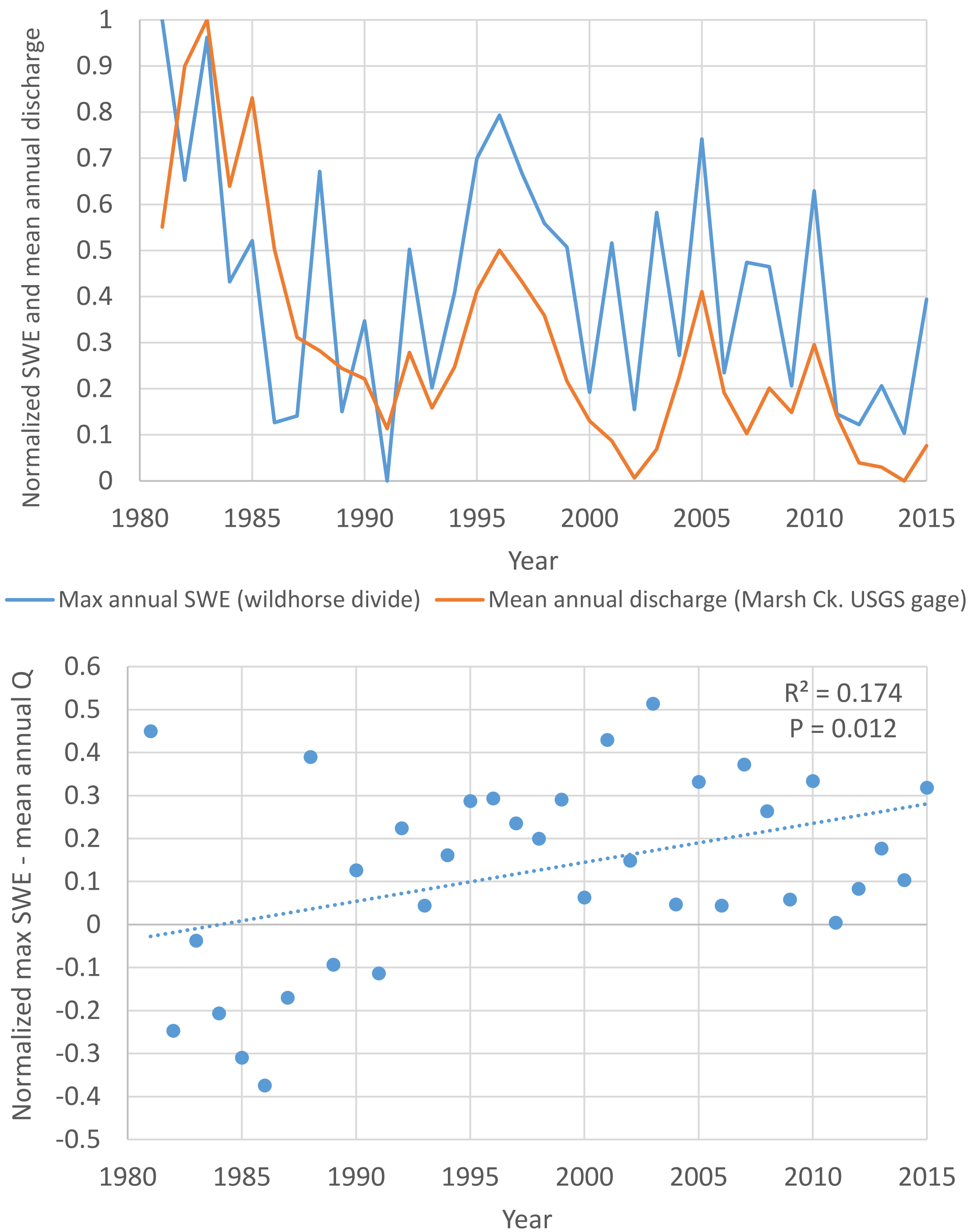


Figure 2.12: Top: Normalized annual max snow water equivalent (SWE) at the Wildhorse Divide SNOWTEL gage (site # 867) in blue and the normalized mean annual discharge at the USGS gage in Marsh Creek (site # 13075000) in orange for the full period of record at the SNOTEL station. Bottom: The difference between the normalized peak annual SWE and mean annual discharge for each year, showing a significant increasing trend (p=0.012).

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Chapter 3: Summary and recommendations for improving water quality

3.1. Introduction and disclaimer

The following recommendations are a guide for local stakeholders interested in reducing suspended sediment loads in Marsh Creek and the lower Portneuf River. This summary and recommendation builds on previous work by comparing the progressive implementation of conservation actions against historic records of suspended sediment flux in Marsh Creek. Changes in channel width and length were measured using aerial imagery collected in 1941 and 2013. Recommendations are made based on previous studies, stakeholder meetings, land owner surveys, aerial imagery, and an observed correlation and cross-correlation between conservation efforts and suspended sediment flux over the past 50 years. This report should not be used to guide any site-specific efforts. New projects will require site-specific analysis by the respective professionals. The authors of this summary and recommendation take no responsibility for actions taken in response to this document.

3.2. Study design and methods

We estimate long-term trends in suspended sediment flux using a weighted regressions on time, discharge, and season (WRTDS) model to predict daily concentrations over the period of analysis (Hirsch et al., 2010). Because water quality data were collected at two different sites, we scaled values from a downstream site (Site 2) to represent the suspended sediment concentrations measured at the long-term USGS gaging station (USGS Site 13075000) near McCammon (figure 3.1). In addition to generating daily concentrations and fluxes, the WRDTS model generates a flow-normalized flux (FNF) that eliminates the influence of interannual changes in discharge, allowing for the comparison of conservation and land use changes to the long term water quality record (Hirsch et al., 2015). The WRTDS model is part of the Exploration and Graphics for RivEr trends (EGRET), developed in R statistics.

Information regarding the history of conservation actions in the Marsh Creek were obtained from the Idaho Department of Environmental Quality (IDEQ), the Natural Resources Conservation Service (NRCS), the USDA Agricultural Census, and from the Environmental Working Group. Progressive conservation actions, represented by dollars spent, and the FNF were compared using a generalized least squares regression model and a 1st order differenced cross-correlation analysis (figure 3.2).

Channel planform change was measured between 1941 and 2013. The 1941 images was orthorectified, georeferenced, and stitched using Agisoft Photoscan (Agisoft LLC, 2016) and the wetted edge of the channel was digitized for each time period from the confluence of Marsh Creek and the Portneuf upstream to the Rat Pond. These channel masks were then imported into the Spatially Continuous Riverbank Erosion and Accretion Measurements tool (Rowland et al. 2016) to detect changes in channel location pixel by pixel. Changes in length and width were averaged over segments defined as 200 channel widths to reduce noise and detect longitudinal trends.

3.3. Summary of pertinent results and conclusions

- A review of previous research in Marsh Creek indicates that the main source of sediment has shifted from the upland benches of Marsh Valley (McSorley, 1977; Merrell and Onstott, 1965) to near-channel sources (Guilinger, 2017).
- Since 1982, an estimated $62.9M, or $82.5M adjusted for inflation, have been spent on conservation and restoration programs within Marsh Creek, with 90% of the funding spent on the uplands through the Conservation Reserve Program.
- The flow-normalized annual suspended sediment flux has decreased by 23.5×10^6 kg/year between 1969 and 2017, a 75% reduction, with the most rapid decreases occurring between 1997 and 2004, and between 2010 and 2012.
- The shift in sediment source, the strong negative correlation between the flow-normalized flux and dollars spent on conservation, and the significant cross-correlations at 6- and 7-year lags are evidence that the investments in the upland areas were effective at reducing fine sediment inputs.
- A comparison of aerial imagery from 1941 to 2013 revealed that the lower 30 km of Marsh Creek have undergone the most change. In this section, the channel length was reduced 2.2 km, and segments widened as much as 3.5 meters. Changes in other segments were below the limit of detection due to the slow rate of change in low gradient, fine grain stream systems.

Flow-normalized suspended sediment flux dropped by 75% over the past 45 years. Dollars spent on conservation projects have had measurable improvements in water quality seen 6 to 7 years later.

3.4. Recommendations for future conservation strategies

The task of restoring water quality in Marsh Creek necessitates conservation planning to be done at the watershed scale, aimed at restoring geomorphic and ecological processes, not just focused at point-specific issues (Wohl et al., 2005). In Marsh Creek, we know that the upland benches used to be the dominant source of sediment, but due to upland conservation practices, this source of fine sediment has been diminished. Guilinger (2017) showed that the Rat Pond is not a significant source of sediment and that **the banks throughout the basin are the primary source**, identifying the lower 30 km of Marsh Creek as a disproportionately large source.

The aerial photograph analysis in this report indicated that the channel in the lower 30 km has been anthropogenically straightened, increasing its erosive capacity. Lower Marsh Creek has also shown the greatest amount of widening, attributed to changes in land use and hydrology, and exacerbated by livestock overgrazing on riparian vegetation and direct access to the creek for water. Based on the current understanding of sediment sources and changes in the channel that have occurred, **future efforts should focus on conservation actions that stabilize the banks and reconnect the channel with its floodplain, especially in the lower basin**.

Conservation actions have done a good job at addressing the shift in dominant sediment sources with the continued support of the CRP and additional efforts towards bank stabilization efforts in the past 20 years. The NRCS was recently awarded a $250,000 special project grant to improve the quality of water in Marsh Creek, targeting the lower 30 km based off recent research showing that the banks in this section are a large source of sediment. These types of projects based on watershed-specific research are crucial for effective and efficient conservation implementation.

Farmers are the stewards of their land. This statement is the foundation on which conservation on private land must stem from, and the numerous personnel implementing the cost-share programs through the NRCS and Section 319 grants have done a great job at working with the land owners in this area, shown by land owner trust in these agencies (Taylor, 2018). This notion of cooperative stewardship between private landowners and state and federal agencies depends on trust and building and maintaining trust is vital for continued success. We should share the success stories of other local conservation projects and to exemplify how future participants could benefit. The 1941 aerial imagery generated from this report is a useful tool in demonstrating the changes in channel planform that have occurred through time. When communicating with landowners, it is important to be relatable, conversational, honest and upfront.

"*Nobody cares how much you know, until they know how much you care.*"
– Theodore Roosevelt

A survey of landowners in Marsh Creek conducted in 2017 showed that the majority of respondents with stream access were concerned about contamination, sediment, the volume of water, habitat, and groundwater recharge (Taylor, 2018). Additionally, out of those that were from lower Marsh Creek, 75% indicated that they are interested in participating in conservation programs, including bank stabilization and constructed wetlands on their property.

The general support for conservation actions should be strengthened with public outreach and education. There have been large advancements over the past six years in our understanding of sediment sources, improvements in water quality due to conservation, and ecological dynamics in Marsh Creek. Landowners need to know that investments in conservation are paying off (Guilinger, 2017; Stalder, 2018). This new information needs to be shared with the communities in Marsh Creek. One possible avenue to share this information could be to have a booth at the Downey Country Fair. This fair is held the first week of August and is well-attended by the whole county. Other possible outlets for disseminating this information would be to create a pamphlet that could be mailed and / or emailed out, to write a section for a regional newsletter like the Idaho Soil and Water Conservation Commission newsletter, and to contact local organizations like the Rotary Club and the Farm Bureau to see if they would be interested in hosting a presentation.

Another important outreach tool would be to have a pilot project that other landowners who were curious about participating in conservation programs could visit and talk with a participating landowner about their experience. This project should showcase the many different BMPs utilized in bank stabilization including off-site watering troughs, exclusion fencing, grazing management, and bioengineering techniques. This would require a willing landowner to host an "open house" after their project was completed and could be a critical component to demonstrating what participating in a conservation program really means.

3.5. Sediment reduction methods

Two different strategies can be taken to improve the water quality in Marsh Creek: reducing inputs from the source, and sediment retention strategies (Guilinger, 2017). Bank stabilization projects reduce the source of sediment and target the underlying issues causing of high suspended sediment loads but are challenging to successfully implement at the watershed scale. Bank stabilization projects may also fail to yield reductions in suspended sediment loads due to the presence of ubiquitous bank failures throughout the basin, and highly fragmented land due to private ownership. They can also be financially impractical if a farmer relies on meadow hay harvested from near the channel. Nonetheless, current efforts are on the right track with the special project grant targeting bank stabilization in the lower basin. Stabilizing all of the banks in Marsh Creek may be an impractical goal, necessitating other methods to reduce sediment loads.

Marsh Creek, inherent in its name, was once a very wet valley with frequent overbank flows and deposition evident in the topography of the southern basin (figure 3.3). Straightening and levying the channel reduced these overbank flows and made the land more productive for grazing and agricultural production, but by reducing the rivers connection with the floodplain, the watershed scale geomorphic processes were fundamentally altered. Overbank flow causes a decrease in the velocity of the water and deposition of the sediment in transport. Reconnecting the channel with its floodplain would store sediment within the basin and reduce the annual sediment load (Zedler, 2003). The lower 30 km have been shown to be a dominant source of sediment, thus any projects attempting to reconnect the river with its floodplain should focus on the lower 30 km of the creek to reduce the sediment load from Marsh Creek to the Portneuf River (Guilinger, 2017). Luckily, numerous locations within the lower section of Marsh Creek have floodplains that are lower than the existing channel bank height, making it feasible to implement some form of wetland restoration or construction with landowner support.

To make a wetland project successful, the land is normally removed from production and grazing, making it best suited for a conservation easement. The NRCS has the Agricultural Conservation Easement Program (ACEP) aimed at protecting and restoring wetlands, as well as the Wetland Reserve Enhancement Partnership (WREP) to provide technical assistance and financial leverage with other conservation partners. Permanent and 30-year easements are available through the ACEP, and in these programs, NRCS would pay for 100% of the easement, and between 75-100% of the restoration costs (NRCS, 2018a). Other possible sources of financial assistance in purchasing a conservation easement could be the City of Pocatello, and private non-profits like Trout Unlimited, Ducks Unlimited, and the Sagebrush Steppe Land Trust.

3.6. Conclusions

The conservation efforts made over the past 35 years are strongly correlated with a 75% reduction in the flow-normalized suspended sediment load, and previous research has shown that the source of sediment has shifted over time from the upland benches of Marsh Valley to the banks of Marsh Creek (Guilinger, 2017; Merrell and Onstott, 1965). Bank instabilities have been observed throughout the basin and although over 44 km of fence line and 30 offsite watering troughs have been installed throughout the basin, the banks are still the dominant source of sediment. Aerial photograph analysis indicates that the channel in the lower 30 km has been anthropogenically straightened, the greatest amount of widening is observed in that lower section. These observations strengthen previous research stating that the lower 30 km contributes a disproportionately high sediment load (Guilinger, 2017).

Future conservation efforts should utilize both bank stabilization and sediment retention strategies to improve water quality in Marsh Creek and the lower Portneuf River. Public outreach and education should also be increased, and a pilot project could be used to exemplify the different types of BMPs used for streambank stabilization, or a conservation easement for a wetland restoration project.

Chapter 3 Figures

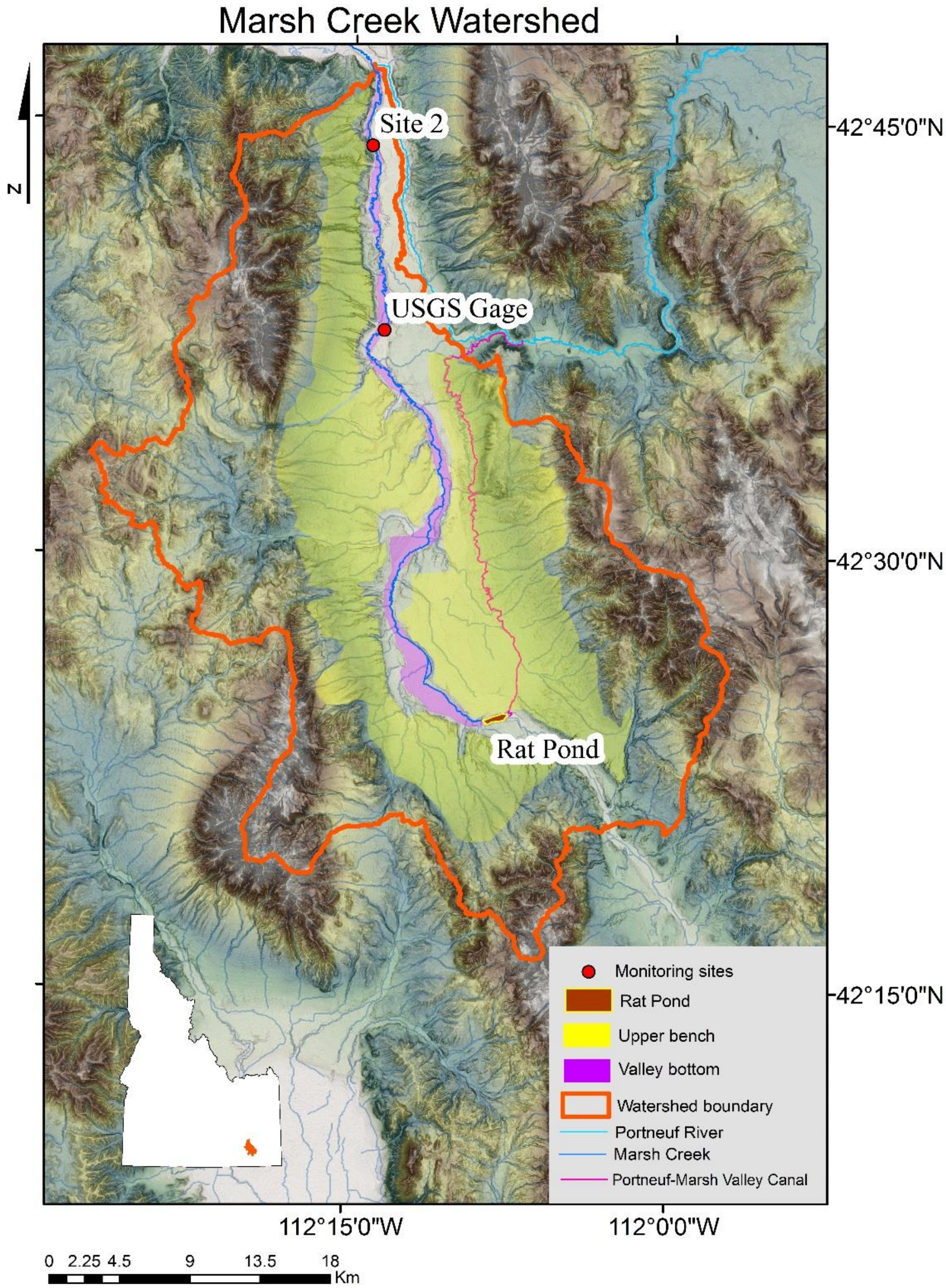

Figure 3.1: Map of the Marsh Creek watershed in southeastern Idaho. This analysis covers the main stem of Marsh Creek from the Rat Pond north to the confluence with the Portneuf River. The orange line is the watershed boundary, and the yellow shaded area shows the upland benches surrounding Marsh Creek that were the historic source of sediment. The purple shaded area shows the lowland valley adjacent the Marsh Creek where recent efforts have aimed to stabilize the banks. The Marsh Valley Canal is shown as a purple line and transfers water from the Portneuf River south into Marsh Valley to irrigate the south east bench.

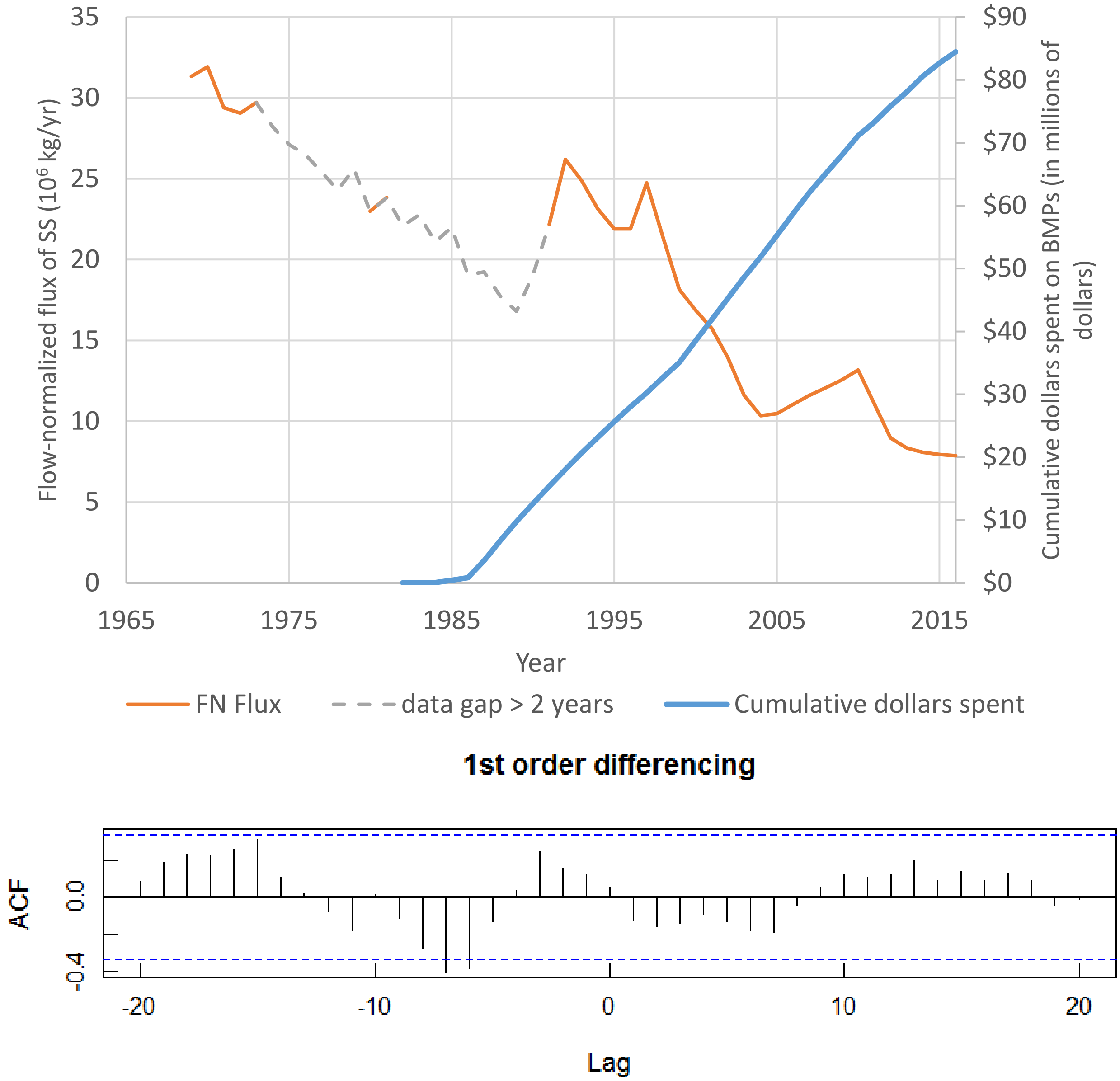

Figure 3.2: Top: Flow-normalized flux of suspended sediment (SS) from 1969 to 2017 plotted on the left y-axis and the cumulative dollars spent on BMPs, adjusted for inflation, on the right y-axis. A significant negative correlation between the cumulative dollars spent and flow-normalized SS flux was found using a generalized least squares regression. This model does not account for any lag time between conservation actions and the FNF, and because each dataset has a linear trend, it is likely that they would be well correlated. Bottom: The relationship between FNF and cumulative dollars spent was tested further using a cross-correlation analysis to assess the influence of lag times and remove the trend in each dataset. The horizontal dashed lines in each plot are the 95% confidence intervals. 1st order differencing utilized a 34-year long record, measuring the change in progressive dollars spent and change in FNF between each year at 0 +/- 20 lags. Lags of -6 and -7 exceed the confidence intervals, indicating that there is a significant relationship (-0.41, and -0.39, respectively) between the dollars spent on conservation and the FNF.

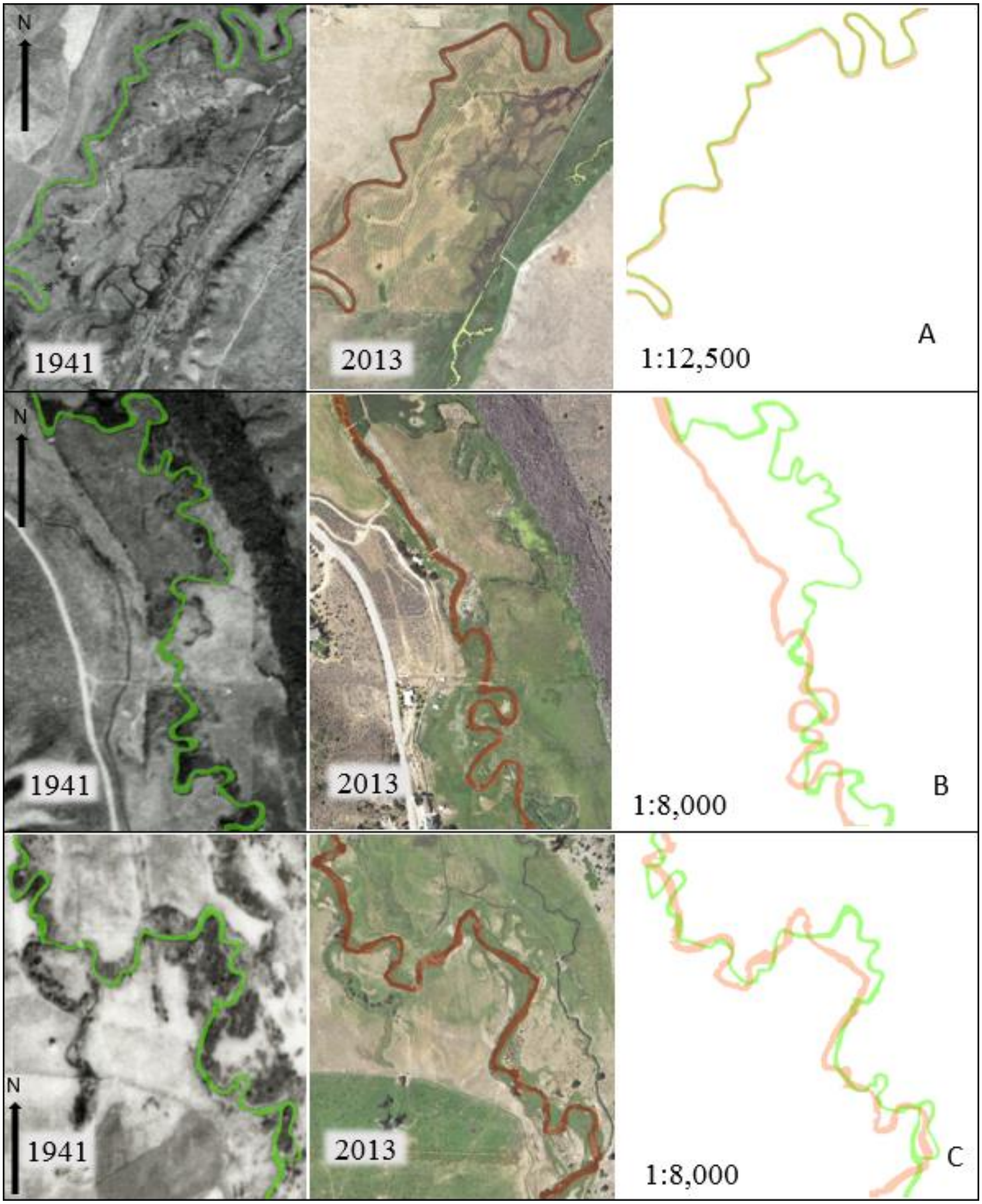

Figure 3.3: Aerial image comparison from 1941(green channel) to 2013 (orange channel) at three locations: A ~30 km, B~10 km, and C~18 km upstream from the confluence. Series A shows a wet meadow in 1941 that is now in agricultural production. Series B shows significant anthropogenic channel straightening, with widening occurring between the two time steps. Series C shows anthropogenic straightening, and a reduction in riparian vegetation, as well as channel widening.

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Appendix A. Agisoft workflow

Agisoft Photoscan was utilized to stitch 42 single frame black and white aerial photos from 1941. This dataset is a unique product and was used to detect channel planform change. The following workflow is included as an appendix because many different methods were attempted to create a georeferenced orthorectified raster, and this workflow worked the best. The most important aspect of this workflow was to preserve the details in the images with as little distortion as possible.

Step 1. Scan images using the best scanner possible at the highest resolution possible in color, even if the images are in black and white. The images used in this project were scanned at 2,200 dpi using an Epson Expression 11000XL scanner. Not all scanners are created equal.

Step 2. Import Photos- Import using the add photos button in the workspace pane.

Step 3. Settings- On the reference pane locate the settings icon (hammer and wrench) and set the coordinate system to “local” coordinate system.

Step 4. Calibrate Cameras- From the tools tab at the top, select camera calibration, then enter 209.55 mm into the focal length for each photo. *The camera calibration is specific to the camera used to take the photo. Make sure to look up the proper camera calibration for your particular dataset. If you can’t find the camera settings that it OK, just leave the camera calibration blank.

Step 5. Settings- From the tools tab select preferences. From the general tab check the box for write log to file. From the advanced tab check the boxes for enable VBO support and Keep Map depths. Click Apply.

Step 6. Align Photos- From the workflow tab select align photos and change settings to High, and generic. Click OK.

Step 8. Using ArcMap (or any GIS software) create a point shape file and mark locations on the present imagery that are also identifiable in the historic imagery. Use get xy data to assign a spatial reference to each point and use extract value to points to assign elevation values to each point. Export the attribute table as a .txt file

Step 7. To add ground control points (GCP) click add GCP and import xyz data. Switch the coordinate system to ESPG: 26912 (NAD 1983 UTM zone 12N) *or whatever coordinate system you are working in. Locate the GCP on each photo. After entering the same point on 2 photos the program will start find that GCP on other photos automatically. Remove any points with a large error. Use the update button to update accuracy. IMPORTANT- DO NOT USE THE MAGIC WAND. Also, masking can be done in each photo to remove flight number and any photo marks by using the select box, then add selection (control + shift + A).

Step 8. Build Mesh- From the workflow tab select build mesh and change settings to sparse cloud, high.

Step 9. Remove inaccurate points- From the edit tab at the top select gradual selection. From this window select the various options from the drop-down arrow and select points to remove. Adjust slider bar to adjust the number of points included. You may need to click on your model and then press delete for the points to be removed. The lasso tool can also be used to select and delete points that are obvious misfits

*IMPORTANT- Make a copy of your chunk before removing any points from your model by right clicking on your chunk and select make a copy. Then work on the COPY. You can also use save as to create a whole new project, that way if your model gets weird, you can always revert to the full model.

Step 10. Import DEM. From the tools tab select import DEM. Make sure the imported DEM covers the full extent of the model. *This allows the images to be projected onto a smooth surface and was a crucial step in a good final product.

Step 11. Build Orthomosaic- From workflow tab select build orthomosaic. Change settings to build off the DEM. Enter the correct coordinate system.

Step 12. Export Ortho- Right click on your ortho image icon in the workspace pane and select export. Export as .TIFF.

Appendix B. Supplemental figures

Figure B.1: Left: Aerial image from 2013 showing the wetted width of Marsh Creek in pink, flowing from south to north. Right: The same spatial extent as the picture shown on the left with a LiDAR derived slope map overlaid on the 2013 aerial imagery with the wetted edge of Marsh Creek shown in pink. The wetted width of the channel changes abruptly due to an agricultural irrigation diversion, causing the wetted width to not be equivalent with the bankfull width. Five sections of channel were identified by visually inspecting the channel, as shown in the right image, throughout the basin where similar diversions caused the wetted width and bankfull width to not be equivalent invalidating these sections from the change in channel width analysis.

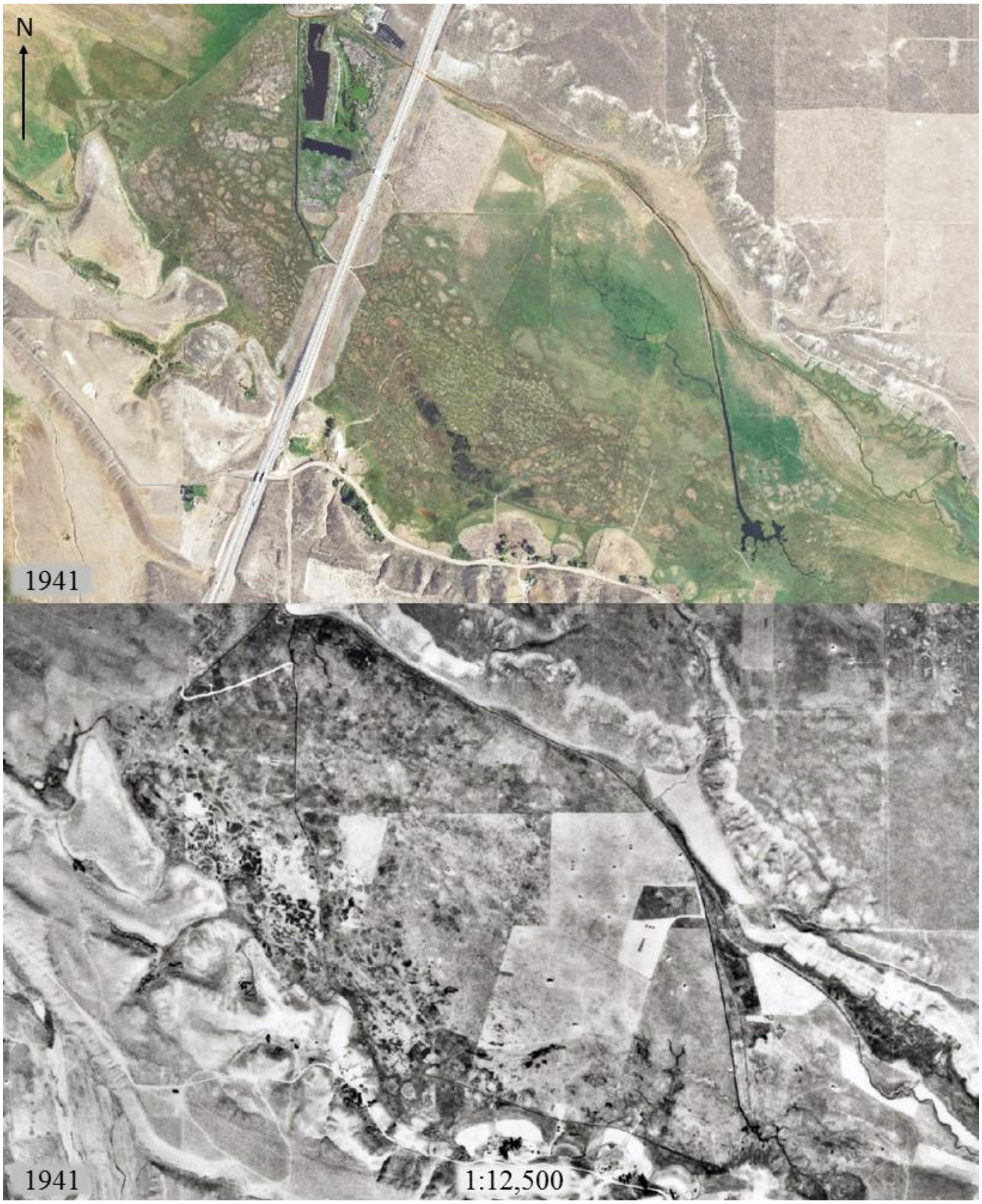

Figure B.2: Aerial photograph comparison of the same area between 1941 and 2013 showing the large region of polygonal topography of what once was a shallow marsh in the upper basin, ~65 km upstream from the confluence. In both images, the channel is straightened and the area is in agricultural production. The 2013 image shows a peat mine in the upper left-hand corner. The presence of the peat mine strengthens the inference that this section of the valley had been a wetland prior to channelization.