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Development and Application of Hydraulic and Hydrogeologic Models to Better Inform Management Decisions

Trinity L. Stout
Utah State University

DEVELOPMENT AND APPLICATION OF HYDRAULIC AND HYDROGEOLOGIC

MODELS TO BETTER INFORM MANAGEMENT DECISIONS

by

Trinity L. Stout

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Approved:

____
Bethany T. Neilson, Ph.D.
Major Professor

____
Jeffery S. Horsburgh, Ph.D.
Committee Member

____
Patrick Belmont, Ph.D.
Committee Member

____
Mark McLellan, Ph.D.
Vice President for Research and
Dean of the School of Graduate Studies

UTAH STATE UNIVERSITY
Logan, Utah

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ABSTRACT

Development and Application of Hydraulic and Hydrogeologic

Models to Better Inform Management Decisions

by

Trinity L. Stout, Master of Science

Utah State University, 2017

Major Professor: Dr. Bethany T. Neilson
Department: Civil and Environmental Engineering

Water is arguably one of the most important and limited resources in semi-arid regions. As populations continue to grow, so will the demand for water and the development of water resources. Oftentimes, management decisions can alter both the quality and quantity of water necessary for maintaining environmental quality and the sustainability of human water supplies. Because of the importance and overall impact of management decisions, a variety of approaches have been established to describe, measure, and predict changes in environmental systems. Along these lines, this research focused on the development and application of two specific models for assessing stream restoration and groundwater recharge.

The first study focused on understanding the impacts of beaver dams in mountain streams and their effectiveness as a restoration tool. One-dimensional hydraulic models for reaches both with and without beaver dams were developed to compare hydraulic responses (e.g., channel depth, width, velocity distributions). Model results indicated statistically significant shifts in the channel hydraulics within the beaver impacted reach. Observations of substrate size distributions for different geomorphic/habitat units within each reach also indicated increased variability and spatial heterogeneity due to beaver dams. Within the hydraulic model, three different approaches to adding beaver dams were applied and demonstrated that a relatively low number of dams would result in significant changes in channel hydraulics. Such predictions provide preliminary guidance regarding the number of dams per unit stream length required to begin meeting some restoration goals.

The second objective investigated previous research that developed a simplified conceptual model and empirical relationship to predict the proportion of precipitation that enters an aquifer by developing a recharge/precipitation (R/P) term and relating it to a variety of hydrogeological, topographic, and land cover parameters. We applied this relationship to two western, mountain watersheds to determine if the driving forces defined in these relationships remain relevant when applied under different conditions. The independent application of the method to each watershed stressed the importance of meeting simplifying assumptions, illustrated the need for more comprehensive geospatial datasets, and demonstrated that existing simplified empirical models may not be suitable for estimating groundwater in mountain watersheds.

(116 Pages)

PUBLIC ABSTRACT

Development and Application of Hydraulic and Hydrogeologic

Models to Better Inform Management Decisions

Trinity L. Stout

Water is one of the most important and limited resources in regions with little rainfall. As populations continue to grow, so does the need for water. Individuals in water management positions need to be well informed in order to avoid potential negative effects concerning the overall quality and amount of water available for both people and the environment. In order to provide better information for these individuals, computer models and mathematical relationships are commonly developed to estimate the outcome of different situations regarding surface water and groundwater. Along these lines, this study focused on two modeling studies that provide information to managers regarding either stream restoration techniques or the amount of groundwater available.

The first study investigated the effects that beaver dams have on streams. In order to do this, a computer model was developed to represent a section of stream with beaver dams and a section without. The model provided information regarding changes in the average depth, width, and velocity of the stream as a result of having beaver dams. We also measured changes in sediment size distributions between the two stream sections to confirm that beaver dams additionally impact sediment movement and channel shape. Results indicated that only a few dams are actually needed to achieve many of the desired changes in stream restoration.

The second study involved testing an equation that was used to predict how much precipitation would become groundwater in a Midwestern watershed. Variables in the equation included measurements of natural or developed land, movement of water through soil, the depth of the water table, and hillslope steepness. We tested the equation in two western watersheds to determine if variables used in the earlier study remain relevant when applied under different conditions. The independent application of the method to each western watershed stressed the importance of meeting simplifying assumptions and developing more complete datasets. We also found that the application of existing simplified empirical relationships may not be suitable in estimating groundwater recharge in mountain watersheds.

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CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER 1

GENERAL INTRODUCTION

One does not have to look far to begin to understand the importance of water in the semi-arid to arid regions of the Western United States. During 2013-2014, California experienced low precipitation and warm temperatures that affected the amount of water available for agriculture and impacted fisheries and other ecosystems (Swain et al., 2014). Proposed pipelines are intended to convey water from northern Nevada and Utah to Las Vegas, NV and from Lake Powell to Washington County, UT to offset the rapidly expanding population's demand for water (Archibold and Johnson, 2007). Changes in precipitation and human-induced flows in the Colorado River have impacted the live storage capacity of Lake Mead (Barnett and Pierce, 2008). The quantity and quality of the available water has significant impacts on anthropogenic usage (consumption, agriculture, or recreation) (Houck, 1999) and ecological structure (channel morphology, riparian and aquatic habitat) (Gorman and Karr, 1978). The demand for water will continue to increase as populations grow, and further uncertainty is introduced by an ever-changing climate. Although difficult, the ability to make informed water resource and environmental management decisions is critical, especially considering the inherent uncertainty and spatio-temporal variability associated with the physical, environmental, economic, social, and political aspects of water resource systems (Hipel and Ben-Haim, 1999).

Management decisions impacting both anthropogenic and ecological aspects of water usage have to be made frequently. The effects of these decisions can often have significant impacts (either positive or negative) on the environment. In an attempt to harness and use as much of our water recourses as possible, environmental processes and conditions have been altered, often leading to the degradation of streams and riparian habitat (Graf, 2006; Ligon et al., 1995; Schmidt and Wilcock, 2008). Channelization of streams to help with flood control have had the unintended effect of limiting the ability of the stream to access floodplains and rework sediment, leading to a decrease in habitat availability and complexity (Chapman and Knudsen, 1980; Lau et al., 2006). Changes in land use impact runoff and erosion, changing channel morphology and stream ecosystems (Allan, 2004). Historically, groundwater and surface water have been treated as two separate resources (Winter, 1998) in the water resource management decision making process, and, in some cases, excessive pumping of groundwater has contributed to a decrease and even a complete loss of surface water (Wahl and Wahl, 1988).

Because of the importance and impact of management decisions, a variety of approaches have been established to describe, measure, and predict changes in environmental systems due to human interaction and water demand (Fleckenstein et al., 2010; Hipel and Ben-Haim, 1999). Models have been created to predict the impacts and responses due to stream restoration (Bennett et al., 2008; Kasahara and Hill, 2008), determine the amount of annual groundwater recharge (Scanlon et al., 2002), or demonstrate changes in surface water quality (Tong and Chen, 2002). This thesis focused on two studies in which models were applied with the intention of better informing future/imminent management decisions regarding 1) stream restoration projects and 2) the estimation of groundwater recharge to aquifers.

Numerous studies document degradation to stream ecosystems and a variety of restoration and rehabilitation methods have been introduced (Shields Jr et al., 2003). Some of the more conventional stream restoration methods used can be costly and require heavy machinery. Because of this, new methods and techniques are being developed to achieve restoration goals while keeping cost and disturbance minimal, such as the introduction of beaver dams or the installation of beaver dam structures (Pollock et al., 2014). Oftentimes the impacts and effectiveness of different restoration techniques are not quantitatively known pre-implementation. Palmer and Bernhardt (2006) call for the development of methods to synthesize and evaluate the impacts of stream restoration projects. Majerova et al. (2015) mention the need for further quantitative field studies and modeling work to determine and quantify the effectiveness of specific restoration efforts. Focusing in on the specific method of using beaver dams as a restoration tool, the first study aimed to meet the call for more quantitative and predictive work regarding stream restoration efforts.

Much of the groundwater recharge work completed has focused on arid to semi-arid regions (De Vries and Simmers, 2002). Unfortunately, there is no direct way to measure groundwater recharge, so a variety of methods have been developed to estimate rates, and several must be applied to increase the reliability of the estimate (Scanlon et al., 2002). Each method requires a substantial amount of data that are difficult to obtain. As a result, new methods and relationships are being developed to estimate recharge rates using readily available data to help managers make informed decisions (Cherkauer and Ansari, 2005). However, many of the new methods are site specific and remain to be tested under different conditions. The second study focused on applying one of these relationships in a mountain watershed.

Overall, there is a clear need for further work in modeling the impacts of stream restoration and developing better estimates of groundwater recharge. This thesis is focused on investigating the development and application of hydraulic and hydrogeologic models with the intent to better inform resource managers regarding both of these issues. Chapter 2 will discuss the development of a hydraulic model to relate the impacts of beaver dams on channel hydraulics and substrate characteristics in the context of stream restoration, and determine an appropriate beaver dam density. Chapter 3 investigates the applicability of a previously established relationship between watershed characteristics and variability in recharge and precipitation when it is applied to two mountain watersheds. The thesis concludes with a general discussion of findings and engineering significance of the entire work.

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CHAPTER 2[1]

IMPACTS OF BEAVER DAMS ON CHANNEL HYDRAULICS AND SUBSTRATE CHARACTERISTICS IN A MOUNTAIN STREAM

Abstract

Beaver dams have significant impacts on the hydrology, temperature, biogeochemical processes, and geomorphology of streams and riparian areas. They have also been used as a viable tool in restoring impaired riverine systems. Due to the dynamic nature of beaver dams, these impacts vary and are difficult to quantify. To begin understanding the impacts of beaver dams in mountain streams, we developed 1D hydraulic models for a beaver impacted reach that includes eight dams and a non-impacted reach to compare hydraulic responses (e.g., channel depth, width, velocity distributions). We also compared observations of substrate size distributions for different geomorphic/habitat units within each reach. Results from the models indicated shifts in channel hydraulics through statistically significant increases in depths and widths as well as a decrease in flow velocities through the beaver impacted reach. These hydraulic adjustments, as a result of beaver dams, are consistent with observed changes in the increased variability and spatial heterogeneity in sediment size distributions. Through the application of three different modeling approaches we found that a relatively low number of beaver dams would result in significant changes in channel hydraulics. Such predictions are shown to provide preliminary information regarding the number of dams per unit stream length required to begin meeting various restoration goals.

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Introduction

Beaver dams have significant impacts on the hydrology, temperature, biogeochemical processes, and geomorphology of streams and riparian areas. Research focused on the influences on hydrology and hydraulics has shown that beaver dams decrease flood peaks and flow velocities while increasing surface water storage and base flow during summer months (Green and Westbrook, 2009; Nyssen et al., 2011; Westbrook et al., 2006). Decreased velocities through beaver ponds result in increased sediment deposition and improved stream bank stability (Pollock et al., 2007). According to Westbrook et al. (2006), beaver dams elevate the water table and attenuate the expected water table decline during summer months, allowing for increased interaction with riparian areas. They also found that as the new hydrologic regime created by beaver dams is maintained, the formation and overall persistence of wetlands is encouraged. Other benefits of beaver dams include floodplain development, channel meandering, and the creation of more complex channels by introducing spatial heterogeneity in hydraulic characteristics such as channel depth, width, cross sectional area, instream velocity, and channel roughness (Green and Westbrook, 2009). Because of the range of impacts beaver dams have on hydrologic and hydraulic characteristics, beavers are starting to be used as a viable tool in restoring and improving impaired streams and riparian habitats (Pollock et al., 2014; Wheaton et al., 2004; Wheaton et al., 2012).

Beavers are specifically beneficial to stream fish populations in their ability to create complementary habitat (Schlosser, 1995). In order to maintain healthy fish populations, access to flowing (lentic) and still (lotic) waters are needed for development of fish through various life stages (Rosenfeld et al., 2000; Schlosser, 1995). Snodgrass and Meffe (1999) stated that beaver dams increase habitat heterogeneity by introducing lentic patches in lotic corridors. One of the fish species of concern in the intermountain west is the Bonneville cutthroat trout (*Oncorhynchus* *clarkii utah*), a subspecies of trout native to parts of Utah, Nevada, Wyoming, and Idaho – a region referred to as the Bonneville Basin (McHugh and Budy, 2006). Harig and Fausch (2002) found that in order to facilitate the translocation of cutthroat trout, a stream needs to have sufficient deep-pool habitat. However, White and Rahel (2008) determined that without beaver impoundments, there is little formation of necessary pool habitat in headwater streams. They also showed that negative impacts of drought on Bonneville cutthroat trout populations were mitigated in reaches containing beaver ponds. Other streams in the same study drainage had more age-0 fish, yet saw a decrease in juvenile fish during drought periods while the beaver impacted reaches maintained higher populations of both juvenile and adult fish.

Many restoration projects on dammed and regulated rivers have focused on rehabilitating salmonid spawning habitat, and improving habitat in general (Pollock et al., 2007; Wheaton et al., 2004). In order to determine the effectiveness in meeting restoration goals of improved fish habitat through the development of beaver dam complexes (or series of beaver dams), specific metrics must be chosen and measured. Indicators of improved habitat availability and diversity are the increased variability in hydraulic characteristics (such as channel depth, width and velocity) and increased spatial variability in geomorphic characteristics such as grain size distributions of channel substrate (Roper et al., 2002).

Use of hydraulic characteristics (e.g., channel depth, width, and velocity) as indicators of available habitat is well documented in the literature. For example, the Physical Habitat Simulation (PHABSIM) Software simulates a relationship between streamflow and habitat, and changes in flow and hydraulics are equated with changing/available habitat (Milhous and Waddle, 2012). Similarly, Rabeni and Jacobsen (1993) used distinct morphological and hydraulic characteristics to classify different habitat units. Ghanem et al. (1996) predicted hydraulic characteristics to describe physical habitat conditions through a two dimensional hydraulic model.

Lamouroux et al. (1998) used statistical hydraulic models as a part of estimating habitat suitability. Although overly simplistic, diversity in hydraulic characteristics can be used as a first cut surrogate as a means to provide preliminary information regarding potential habitat influences due to beaver dam development.

A close relationship between hydraulic characteristics and geomorphic features of the stream exists and could be used as an indicator of aquatic species habitat availability and diversity. Brierley and Fryirs (2013) showed that geomorphic diversity in streams determines the diversity of the habitat, its availability and the viability. Wheaton et al. (2010) linked geomorphic changes to changes in the physical habitat at a scale that fish experience by comparing the differences in digital elevation models (DEMs) from before and after a high flow event. Further, as part of the protocol for the Columbia Habitat Monitoring Program (CHaMP), geomorphic information, specifically spatial substrate composition and distribution data, is obtained (Roegner et al., 2009) and illustrates the importance of understanding geomorphic diversity when determining overall habitat availability.

With the introduction of beaver dams to a system, a cyclical feedback between changing hydraulics and thus changing geomorphic properties is initiated. If dams fail, the system is pulsed with stored sediment and the cycle is reset (Butler and Malanson, 2005; Levine and Meyer, 2014). Beaver dams initially increase depth and decrease velocity, in turn altering sediment erosion, transport and deposition trends upstream of the dam while scour pools form at the downstream side. Altered sediment transport trends translate into roughness values that vary longitudinally, in turn affecting the channel hydraulics. As a result of beaver dam construction, trends in hydraulics and geomorphic properties are disrupted and create increased spatial heterogeneity for both metrics as a function of time (Green and Westbrook, 2009).

Few studies have quantitatively described the influence of beaver dams on channel hydraulics. However, Green and Westbrook (2009) studied the effects of beaver dam removal on the type and percent coverage of riparian vegetation, channel hydraulics (such as depth, width, velocity, and stream power), and sediment yield. Their study was performed through analysis of aerial imagery, field observations of bankfull indicators, velocity measurements, and Manning's equation. While the study covered longer temporal scales (36 year period), detailed information regarding how the hydraulics vary over space and different flow conditions was missing. There is a clear need for more detailed methods that assess the influence of beaver dams on channel hydraulics over different flow ranges as they relate to changes in stream ecosystems. Majerova et al. (2015) also highlighted the need for a better understanding of spatial and temporal variability of streamflow and temperature in beaver impacted reaches. They concluded that if beaver dam complexes are to be successfully used as a restoration tool, a better understanding of their influences on stream ecosystems needs to be reached through more quantitative field and modeling studies.

To begin understanding these influences, we use a 1D hydraulic modeling approach to consider the temporal and spatial shifts in hydraulic variability in reaches with and without beaver dams. Field data describing the spatial heterogeneity of grain size distributions following the introduction of beaver dams on a mountain stream were gathered. In our study, we illustrate and quantify the effectiveness of beaver dams in meeting specific restoration goals of increased habitat availability and diversity using surrogate hydraulic measures. Due to the complexity of connections between beaver dams and stream restoration goals, this simplified approach provides only initial understanding that will require further development and testing in various field settings. However, the model comparisons developed illustrate impacts that beaver dams can have on hydraulic characteristic distributions, establish how these distributions shift as beaver dam density increases, and provide information regarding the density of dams (number of beaver dams per km) that could significantly change reach scale hydraulic characteristics. As a result, the density and location of beaver dams presented here provide initial guidance based on a modeling methodology rather than a definite restoration approach that would be relevant for any stream system.

Methods

Study Site Description

Curtis Creek is a tributary to the Blacksmith Fork River near Hyrum, Utah. This snowmelt dominated, first order mountain stream drains 59.5 km^2 of the Bear River Range (N.M. Schmadel et al., 2014a). The creek flows through Hardware Ranch, a Wildlife Management Area managed by the Utah Division of Wildlife Resources (UDWR). In 2001, approximately 440 meters of the creek was relocated and the channel was re-built in order to avoid damage to structures in the area (Figure 2-1). In 2005, the lower portion of the study reach was fenced in which allowed for riparian vegetation recovery. In the summer of 2009, beaver dams began being built in this lower (fenced-in) portion of the study reach. In 2012, when this study started, nine beaver dams with heights ranging from 0.44 to 1.29 m were already established in the reach (Figure 2-1). Eight of the dams were located in the main channel (Figure 2-1, beaver dams with numbers) and one in the old channel at the downstream end of the reach. Between 2013 and 2014, additional smaller dams were built that changed the local water surface profile (Figure 2-1, beaver dams without numbers). Due to a small snowpack and generally low flow conditions from summer 2012 to 2014, no significant sedimentation in the channel and ponds was observed throughout the study reach. Localized channel aggradation, degradation, sediment movement, and bar formation were observed surrounding a partially failed dam in the upper section of the beaver

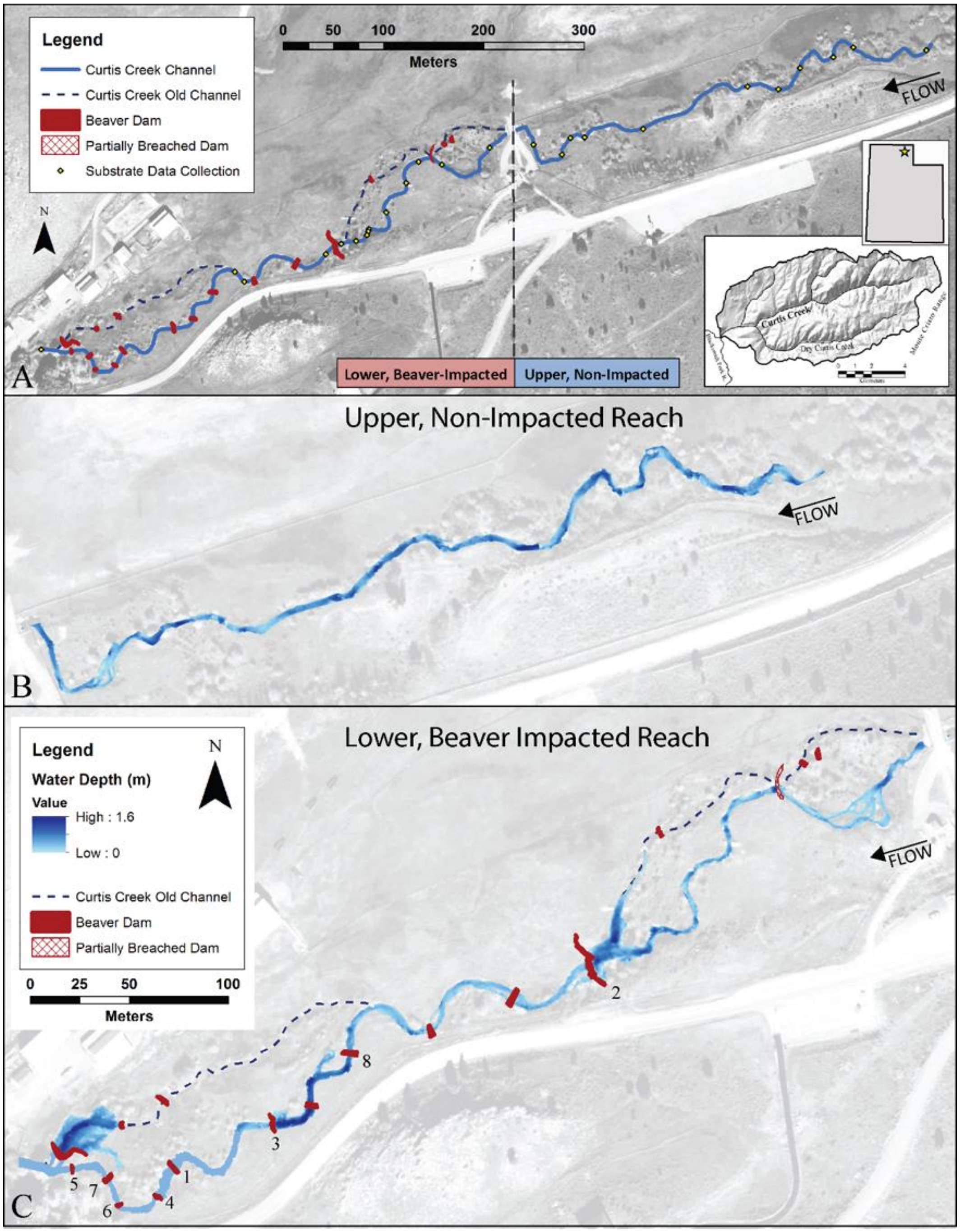

FIGURE 2-1 Site map of the study reach at Curtis Creek near Hardware Ranch, UT. The beaver impacted reach is indicated in red, while the non-impacted reach is in blue (Figure 1A). Flow is from right to left. The main channel is indicated with blue line (Figure 1A) and different shades of blue representing water depth ranging from 0 to 1.6 meters (Figure 1B, 1C). The location of the old channel is indicated with dashed blue line. Substrate data locations are shown as yellow circles along both reaches (Figure 1A). Beaver dams present in the lower reach are numbered in the same order as they were constructed by the beaver, same as they were placed in the model (Natural Sequence, Figure 1C).

impacted reach (Figure 2-1). However, channel changes affecting hydraulic characteristics throughout the rest of the study reach were minimal during this study and did not require alteration of channel geometry in the model. Similar to other work completed within this area (Majerova et al., 2015), the Curtis Creek study reach (Figure 2-1) was divided into two main study reaches: a lower, beaver impacted reach (750 meters long), and an upper, non-impacted reach (535 meters long).

Field Data Collection

Topographic data for the study reach were collected between 2012-2013 using a differential rtkGPS (Trimble® R8, Global Navigation Satellite System, Dayton, Ohio) in order to develop channel, flood plain, and terrace geometry for hydraulic modeling. Channel topography was surveyed at a fine resolution (1.0-4.5 points/m^2) in order to capture the variability in channel geometry. The resolution of points decreased further away from banks (less than or equal to 1 point/m^2), as these areas were more uniform and not critical for hydraulic modeling. The survey point density was 2.2 points/m^2 on average. Banks were surveyed as breaklines to allow for crisp construction of channel form. Beaver dams were surveyed at the dam crest and at the bottom of the downstream and upstream side of the structure to establish the width, length, and volume of the dam. In 2014, a survey of the thalweg was performed to verify any channel change that had occurred during the time of the study was minimal (< 10 cm difference between initial survey and 2014 thalweg).

Water surface elevations (WSEL) were surveyed longitudinally along the stream at three different flows. The point density varied from 1 point per 0.3 m of stream to 1 point per 20 m of stream depending on variability of the water surface. In 2012, WSEL was measured for the lower, beaver impacted reach during base flows (0.19 m^3/s). The WSEL for the upper, non-impacted reach was surveyed in 2013 while flows were also low (0.30 m^3/s). During mild spring runoff in

2014, a complete WSEL survey was performed for both the upper and lower reach (0.93 m^3/s). The average slope for the study reach, determined from this water surface profile, is 0.017 for the lower, beaver impacted reach and 0.023 for the upper, non-impacted reach. At the same time all the WSEL surveys took place, discharge was measured using a Marsh McBirney Inc® Flo-Mate™ (Model 2000, Frederick, Maryland). Previous studies discuss the importance of groundwater/surface water interactions through this area(Majerova et al., 2015; N.M. Schmadel et al., 2014a), however, for our study, we assumed no groundwater/surface water exchange and the flow measured at the upstream boundary of the study reach was assumed to remain constant throughout the reach.

The detailed channel topography survey, a subsequently constructed water depth map, and WSEL slopes were combined with field observations (Brierley and Fryirs, 2013) to identify pools and riffles for substrate data collection. Substrate data were collected longitudinally along both reaches to characterize substrate size distribution for riffles, pools, and bars (Figure 2-1). There were 12 pebble counts performed in the upper reach and 15 in the lower reach. Each pebble count was approximately a 100 count and was performed randomly along the transect following the procedure outlined in Harrelson et al. (1994). In beaver ponds or natural pools where the substrate size was less than 2 mm, a grab sample was collected and analyzed with sieves to better determine the sand and silt fractions (smallest sieve size was 0.065 mm). Five grab samples were collected in the lower, beaver impacted reach and two grab samples were collected in the upper, non-impacted reach. Substrate was analyzed for diameter percentiles D16, D50, and D84 (Bunte and Abt, 2001).

HEC-RAS Model Development and Calibration

A DEM for the entire study reach was created in ArcMap 10.1 using the topographic survey data. Cross-sections capturing the channel and flood plain were derived from the DEM every five meters using HEC-GeoRAS and were imported into HEC-RAS. Initially, two separate models were created to represent the lower, beaver impacted reach and the upper, non-impacted reach. Beaver dams in the model were created at specific cross-sections in the lower reach using a combination of a blocked obstructions and permanent ineffective flow areas designed to simulate "leaky" dams as described by Woo and Waddington (1990). A blocked obstruction was placed in the bottom of the channel to ensure backwater effect even at baseflow conditions. A series of close, horizontally spaced permanent ineffective flow areas provided narrow, vertical areas that were included in flow calculations, essentially allowing water to flow through the rest of the dam face. 660 meters of the lower, beaver impacted and 523 meters of the upper, non-impacted reach could be modeled due to sparse topographic and WSEL points at the boundaries of both reaches.

Once the models were developed, they were calibrated against observed WSEL from the low flow conditions (0.19 m^3/s for the lower reach and 0.30 m^3/s for the upper) by adjusting Manning's n roughness values for small sub-sections of the reach (10-50 m resolution). During the calibration, ranges of reasonable roughness values were justified through field observations and substrate data. Uniform Manning's n values of 0.035 were assumed for areas impacted by beaver dam backwater, since depths in these locations were determined by downstream impoundments and not channel roughness. The height of the blocked obstruction and the spacing of the permanent ineffective flow areas were adjusted to calibrate WSEL in the backwater (ponded) areas. The lower flow conditions were selected for the model calibration because the influence of beaver dams on fish tend to focus on low flow conditions (Kemp et al., 2012). The calibrated model was validated against observed WSEL from the higher flow conditions (0.93 m^3/s) for both the lower and upper reaches.

After completion of the lower, beaver impacted and upper, non-impacted models, a third model was created to represent the lower reach prior to beaver colonization (lower, pre-beaver) in order to compare the lower and upper reach conditions without the influence of beaver dams. Geometry for the lower, pre-beaver model was created by combining the lower, beaver impacted reach information in sections without beaver dam influences. The areas in and around beaver dams were re-constructed through a combination of interpolation, aerial imagery, and data (stream depth, width) collected prior to beaver colonization.

After the three models were completed, discharge was simulated in steady-state for both 0.19 m^3/s and 0.93 m^3/s conditions. Hydraulic characteristics were calculated every five meters at each cross section in the models. Comparisons between the three models focused on the differences in the spatial distributions of key hydraulic characteristics over the entire study reach such as channel depth, width, and velocity.

Model Application and Comparisons

With the HEC-RAS model simulations completed, three different comparisons (Table 2-1) were performed in order to determine if the construction of beaver dams caused significant changes in the distribution of channel hydraulics (depth, width, velocity) between modeled reaches and to quantify the changes through a series of statistical tests. The first comparison was between the lower, beaver impacted reach and the upper, non-impacted reach. The second comparison investigated how hydraulics in the lower, beaver impacted reach compared to the lower, pre-beaver reach. The third comparison compared differences within the lower, pre-beaver reach and the upper, non-impacted reach. Comparisons 1 and 2 were designed to determine the impact of beaver dams on channel hydraulics while the third comparison was to determine if there were any natural variations in hydraulics between the upper and lower reaches before beaver impacted the system. The three comparisons were performed at both lower (0.19 m^3/s) and higher (0.093 m^3/s) flows to determine how beaver dams affect hydraulic characteristics over a range of flows. Because the distributions of hydraulic characteristics are non-parametric, Wilcoxon Rank-Sum tests were performed to determine statistical significance using an alpha value of 0.05 (Berthoux and Brown).

TABLE 2-1 Model Comparisons depict which reaches are being compared. Comparisons were performed twice for different flows (0.19 and 0.93 m3/s).

Modeled Reach	Comparison 1	Comparison 2	Comparison 3
Upper, Non-Impacted (UNI)	X		X
Lower, Beaver Impacted (LBI)	X	X	
Lower, Pre-Beaver (LPB)		X	X

Beaver Dam Densities

Upon quantifying impacts of beaver dams, we determined the minimum number of beaver dams required to create a significant change in channel hydraulics using three different approaches. Each approach required nine simulations where the depth and velocity within each reach were compared as beaver dams were incrementally added to the reach. The first approach mimicked the natural sequence of dam construction, while the last two approaches followed the idea of a resource manager selecting optimal dam locations (Pollock et al., 2014).

The first approach used the Natural Sequence of Dams (NS) construction. Beaver dams were added to the modeled study reach in the order they were naturally constructed. After the addition of each dam, the model was run to show the sequential impact of natural beaver dam complex evolution on the distribution of depth and velocity. The second approach, Spatially

Equivalent Dams (SE) construction, involved segmenting the lower reach into eight equal sections. Dam locations were selected by effectively halving the reach and continuing to add dams at equal increments, alternating downstream to upstream. This method demonstrated the impact of beaver dam evolution if the structures were located in a non-strategic manner. The third approach, Maximum Effect of Dams (ME) construction, utilized a more sophisticated method for beaver dam location selection by determining channel slopes between cross sections. A moving average of slope of 10 cross sections (approximately 40-50 m river length) was calculated and sorted to determine where the lowest slope cross sections were located. Those locations with the lowest slopes were selected for beaver dam placement. In theory, by constructing a beaver dam at the downstream end of a low-slope section, the backwater effect of a beaver dam is maximized. As a general rule, dams were not constructed within 40-50 m upstream and downstream of a previous dam to ensure an uninhibited backwater effect. The sequence of dam addition for this approach was based solely upon the remaining sections with the lowest slope.

The method of dam construction for the three approaches used blocked obstructions on the bottom of the channel with heights extending to the adjacent bank elevations. The "leaky dam" method used during model calibration was possible by adjusting ineffective flow areas to get modeled WSEL to match observed water surface for particular flows. Because there was no observed WSEL dataset available for scenarios run to determine the effects of different beaver dam densities, a blocked obstruction was simpler to construct and less subjective. The blocked method was compared to the "leaky dam" method using a paired t-test to ensure similar results in the distribution of hydraulic characteristics were obtained.

In order to estimate the threshold where additional dams no longer change distributions of hydraulic parameters, multiple Wilcoxon Rank Sum tests were performed for the NS, SE, and ME model constructions. Each time a dam was added to the modeled reach, a new distribution of hydraulic characteristics was produced. By developing a series of models with an increasing number of dams, a sequential shift in hydraulics was created. By comparing the distribution of hydraulic characteristics (using the Wilcoxon Rank Sum test) from each model run with the distributions from model runs with a greater number of dams, eventually a point was reached at which the addition of further dams in the model did not cause a significant change in the distribution of depth and/or velocity.

Results

Substrate Data Results

Substrate samples were analyzed for the D16, D50, and D84 percentiles and compared longitudinally in order to determine spatial heterogeneity between the upper, non-impacted and lower, beaver impacted reaches (Table A-1). Size distributions for riffles, pools, and bars (Figure 2-2) illustrate a downstream fining trend in the observed sediment size distribution for the non-impacted reach. These trends did not continue in the beaver impacted reach and the substrate variability introduced to the impacted reach was apparent. The D50 range for the lower, beaver impacted reach is 0.3-32 mm (medium sand to coarse gravel) for pools, 10.1-31.8 mm (medium gravel to coarse gravel) for bars, and 16-48.8 mm (medium/coarse gravel to very coarse gravel) for riffles. The respective ranges for the upper, non-impacted reach for pools, bars, and riffles are 5.5-35.5 mm (fine gravel to very coarse gravel), 38-52.8 mm (very coarse gravel), and 40.5-56.7 mm (very coarse gravel), respectively. Beaver ponds in the lower, beaver impacted reach have finer substrate, however the substrate in natural pools are similar in distribution to pools in the upper, non-impacted reach. Riffles located between beaver ponds in the lower reach also have similar size distributions to those in the upper, non-impacted reach. Information regarding the complete range of sediment size distributions (including D16 and D84) is included in the Supplemental Information (Table A-1).

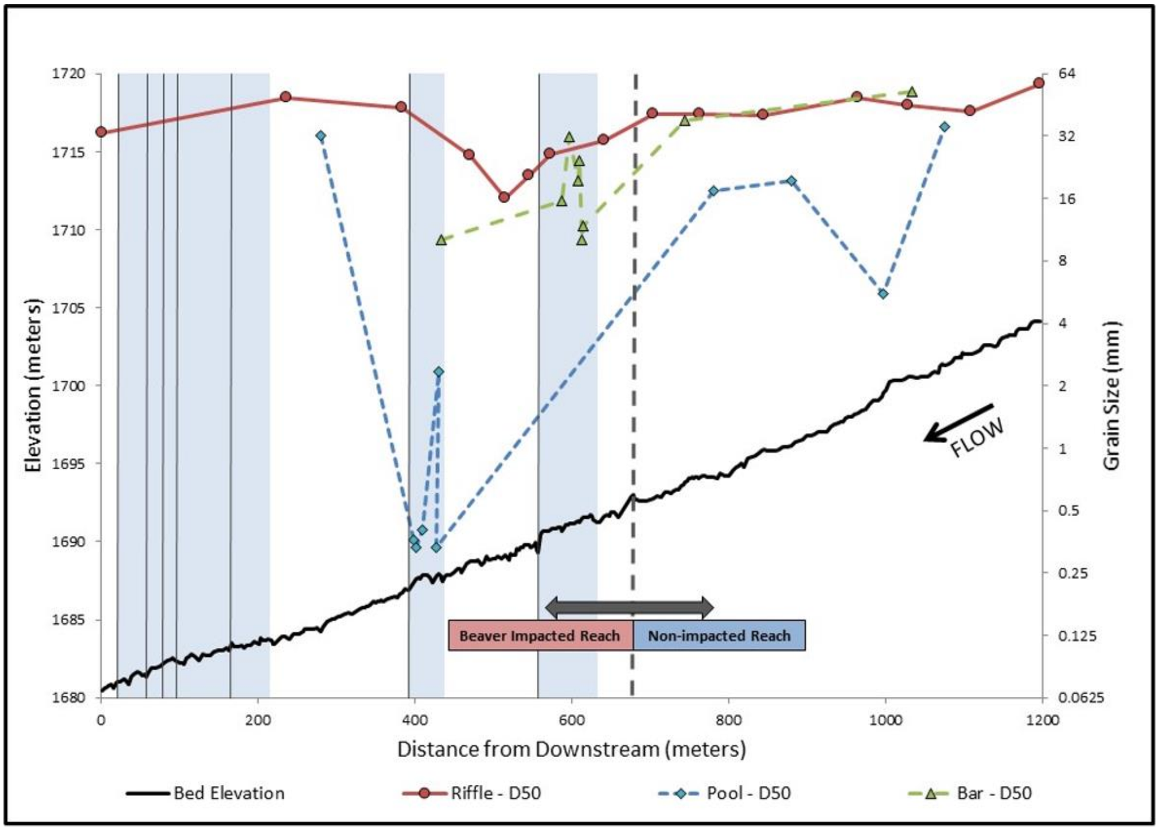

FIGURE 2-2 The median (D50) size distribution for riffles (red), pools (blue), and bars (green) is shown longitudinally. The vertical black lines and shaded blue regions mark the location of beaver dams and their respective backwater effects. The downstream fining trend in sediment size distribution observed in the upper, non-impacted (UNI) reach did not continue in the lower, beaver impacted (LBI) reach and the substrate variability introduced to the impacted reach can be seen.

<u>Model Results and Reach Comparisons</u>

After the models were calibrated and validated (Figure A-1), distributions of hydraulic characteristics (depth, width, velocity) were compared for the three model reaches and for each of the flows (0.19, 0.93 m^3/s). By simply observing the averages, standard deviations, and ranges for the modeled reaches and flows (Table 2-2), it is clear that beaver dams do impact the hydraulics of the system. However, statistical comparisons (t-test, $\alpha = 0.05$) proved valuable in determining if the impacts were, in fact, significant when comparing the distribution of hydraulic characteristics for the upper, non-impacted, lower, beaver impacted, and lower, pre-beaver models (Table 2-3). Comparison 1 (upper, non-impacted vs. lower, beaver impacted) and 2 (lower, pre-beaver vs. lower, beaver impacted) showed all three hydraulic characteristics were significantly different. Depths increased by 91 and 50%, widths increased 53 and 74%, and velocities decreased 33 and 31%, respectively. Comparison 3 (upper, non-impacted vs. lower, pre-beaver) also indicated widths (12% decrease) and depths (27% increase) were different, but still had similar velocities (only 3% difference). The relative impact of beaver dams on channel hydraulics at low and high flows are also similar. Therefore, the remainder of the paper focuses only on the lower (0.19 m^3/s) flow condition, however, corresponding figures for the 0.93 m^3/s simulation are included in the Supplemental Information (Figure A-2).

In order to show the spatial variability in depth, width, and velocity, data were plotted longitudinally for each of the three modeled reaches (Figure 2-3). The upper, non-impacted and lower, pre-beaver reaches show low variability and high spatial uniformity throughout the reach. However, the lower, beaver impacted reach introduces greater variability and spatial heterogeneity in hydraulics.

<u>Beaver Dam Densities</u>

A set of sequential cumulative distributions was generated for each of the three approaches in determining the beaver dam density where hydraulic changes no longer occurred. The cumulative distributions show the gradual shift in hydraulic characteristics due to beaver dam complex evolution (Figure 2-4). Wilcoxon Rank Sum tests determined the point at which statistical significance was reached for each approach (Figure 2-5). Using the NS approach, no

TABLE 2-2 Average values for depth (m), width (m), and velocity (m/s) for the upper, non-impacted (UNI), lower, beaver-impacted (LBI), and lower, pre-beaver (LPB) results for both 0.19 and 0.93 m3/s modeled flows.

Parameter	Model	Modeled Flow						Percent increase in average due to flow changes
		0.19 m^3/s			0.93 m^3/s			
		Average	St.Dev.	Range	Average	St.Dev.	Range	
Depth (m)	LBI	0.2	0.14	0.03-0.66	0.28	0.13	0.06-0.71	38.1
	LPB	0.14	0.07	0.05-0.41	0.27	0.09	0.13-0.59	92.9
	UNI	0.11	0.04	0.04-0.28	0.22	0.06	0.08-0.43	100
Width (m)	LBI	4.88	3.97	0.60-17.7	7.59	7.78	1.51-24.5	49.3
	LPB	2.7	0.81	0.60-6.08	3.77	1.06	1.92-10.7	39.6
	UNI	3.06	1.1	0.89-9.10	4.55	1.6	2.25-13.3	48.7
Velocity (m/s)	LBI	0.45	0.33	0.02-1.57	0.78	0.4	0.08-1.81	71.1
	LPB	0.65	0.26	0.11-1.48	1.06	0.32	0.35-1.67	63.1
	UNI	0.67	0.25	0.16-1.31	1.06	0.25	0.44-1.59	58.2

significant changes in depth and velocity occurred after the addition of three and five dams, respectively. Similar results were obtained using the SE approach. The ME approach required five beaver dams for depth and six for velocity before further changes became insignificant. With this information, we determined the beaver dam density for Curtis Creek (and perhaps other streams of similar flow regime, order, gradient, and substrate) by dividing the minimum number of dams required by the length of the modeled reach (660 m). Declaring statistical significance for a certain method depends on the hydraulic variable in question. It generally required less dams to reach a point of significance for depth, and required more dams for velocity. In order to apply this information, a user should first determine if it is more important to alter depth or velocity for restoration goals. Depending on the target variable, the optimal beaver dam density with respect to individual hydraulic parameters was 4.5 – 7.6 beaver dams per kilometer for depth and 7.6 – 9.1 beaver dams per kilometer for velocity (Table 2-4) for Curtis Creek.

TABLE 2-3 Comparisons of the upper, non-impacted (UNI), lower, beaver impacted (LBI), and the lower, pre-beaver (LPB) reaches at 0.19 m^3/s and 0.93 m^3/s indicate the differences due to beaver dams (Comparisons 1 and 2) and channel geometry (Comparison 3). For both flows, Comparisons 1 and 2 are statistically significant in all three hydraulic variables, while only depth and width values are significantly different in Comparison 3. In the comparisons, the first reach listed is considered the original distribution against which the second reach is compared. Therefore, a positive value indicates an increase in the average values of the second model reach and a negative value indicates a decrease.

	Variable	Percent Difference in Average Values (0.19 m^3/s)	Percent Difference in Average Values (0.93 m^3/s)
Comparison 1 UNI vs. LBI	Depth	90.9	31.8
	Width	53.3	53.8
	Velocity	-32.8	-27.4
Comparison 2 LPB vs. LBI	Depth	50.0	7.4
	Width	73.7	85.7
	Velocity	-30.8	-27.4
Comparison 3 UNI vs. LPB	Depth	27.3	22.7
	Width	-11.8	-17.1
	Velocity	-3.0	0.0

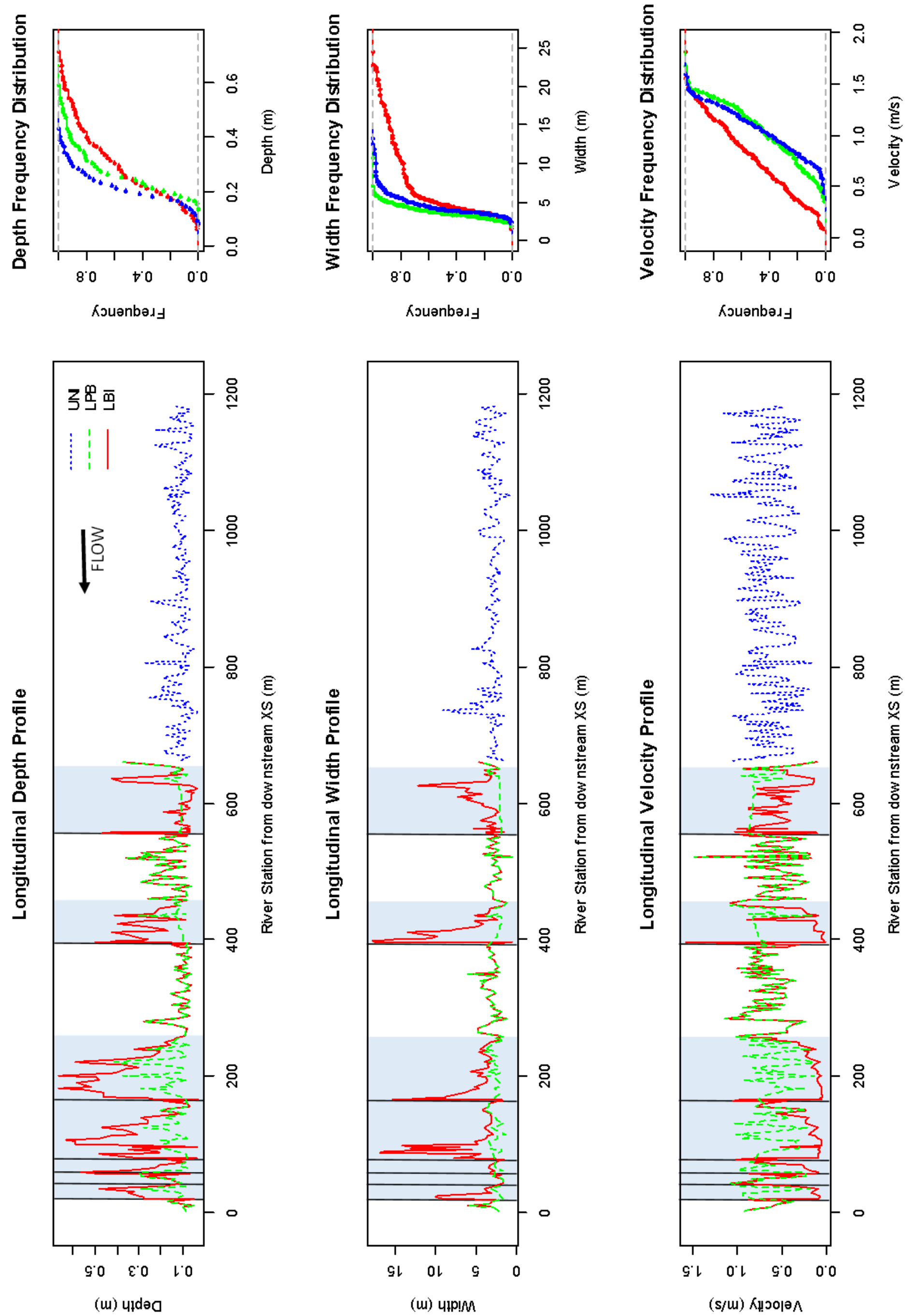

FIGURE 2-3 Depth, width, velocity values are plotted longitudinally along with the respective cumulative distribution curves for the 0.19 m^3/s simulation. The upper, non-impacted (UNI) data are shown in blue, lower, beaver impacted (LBI) data are shown in red, and lower, pre-beaver (LPB) data are represented by green color. Beaver dam locations are indicated by solid black vertical lines and the backwater effect is shown in shaded light blue. Flow is from right to left, with river station 0 as the most downstream location in the model. Through much of the lower reach, LBI and LPB values are the same. Difference can be seen where there is a change due to beaver dams.

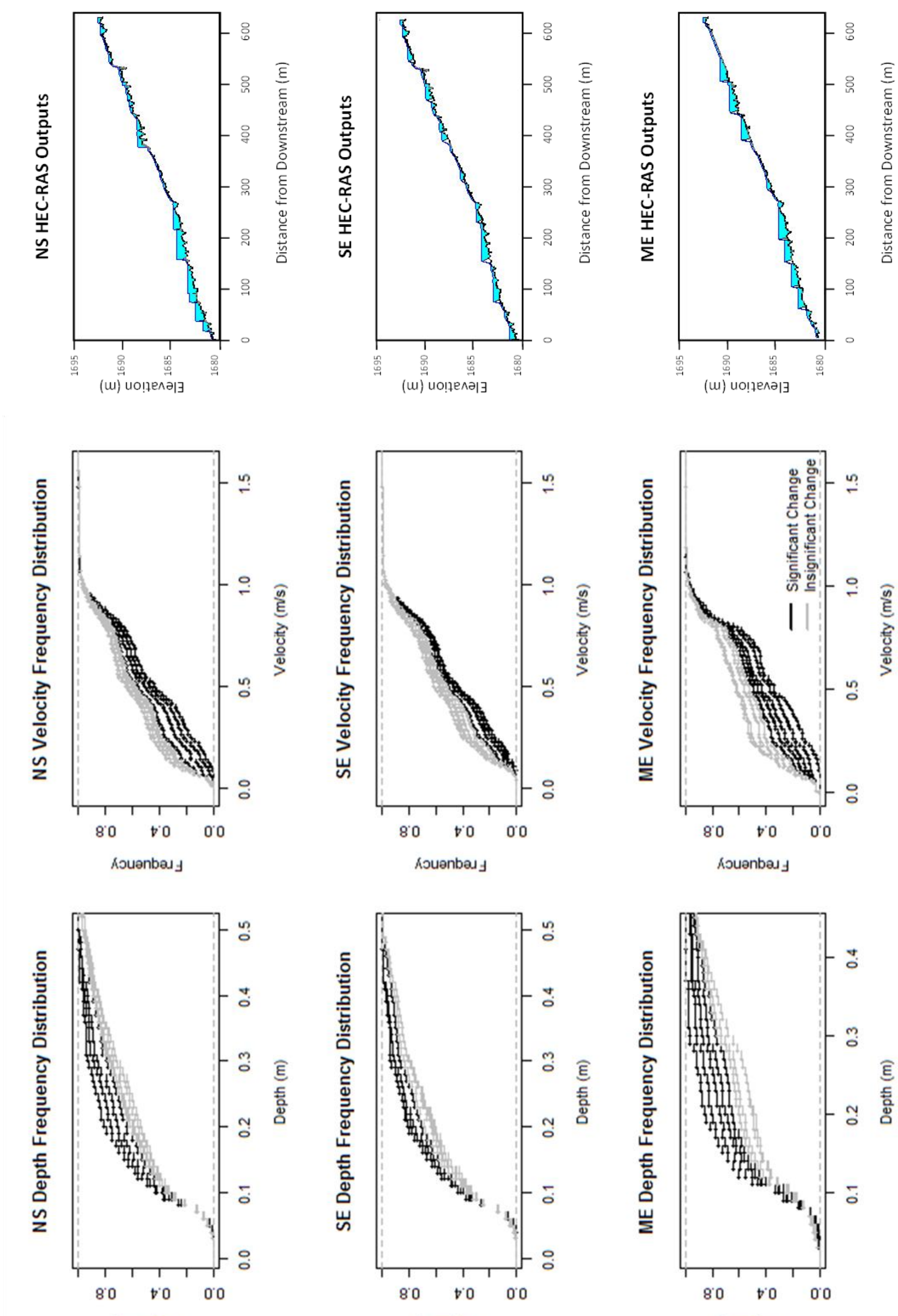

FIGURE 2-4 Sequential cumulative distributions from the addition of more beaver dams are shown for both depth and velocity values. Black indicates distributions from added dams that still contribute to a significant change in the respective hydraulic characteristic. Grey distributions represent dams that no longer result in a significant change. Longitudinal plots are included showing the backwater effects of the dams constructed in each method.

Natural Sequence (NS) Approach

Depth	Initial number of dams in channel	Subsequent number of dams in channel: 0	1	2	3	4	5	6	7	8
	0	-	0.447	0.064	0.002	5E-04	2E-04	7E-05	6E-06	1E-06
	1	-	-	0.291	0.017	0.007	0.003	0.001	2E-04	5E-05
	2	-	-	-	0.161	0.084	0.044	0.026	0.005	0.002
	3	-	-	-	-	0.75	0.558	0.422	0.161	0.079
	4	-	-	-	-	-	0.792	0.631	0.276	0.146
	5	-	-	-	-	-	-	0.827	0.403	0.228
	6	-	-	-	-	-	-	-	0.534	0.322
	7	-	-	-	-	-	-	-	-	0.712
	8	-	-	-	-	-	-	-	-	-

Velocity	Initial number of dams in channel	Subsequent number of dams in channel: 0	1	2	3	4	5	6	7	8
	0	-	0.257	0.014	5E-05	4E-06	4E-07	7E-08	4E-10	1E-11
	1	-	-	0.179	0.003	4E-04	6E-05	2E-05	2E-07	1E-08
	2	-	-	-	0.11	0.032	0.009	0.004	2E-04	2E-05
	3	-	-	-	-	0.586	0.321	0.218	0.038	0.008
	4	-	-	-	-	-	0.66	0.505	0.133	0.038
	5	-	-	-	-	-	-	0.823	0.285	0.1
	6	-	-	-	-	-	-	-	0.394	0.15
	7	-	-	-	-	-	-	-	-	0.553
	8	-	-	-	-	-	-	-	-	-

Spatially Equivalent (SE) Approach

Depth	Initial number of dams in channel	Subsequent number of dams in channel: 0	1	2	3	4	5	6	7	8
	0	-	0.755	0.334	0.082	0.051	0.012	0.005	0.003	0.001
	1	-	-	0.509	0.144	0.092	0.022	0.009	0.006	0.002
	2	-	-	-	0.407	0.285	0.09	0.045	0.033	0.015
	3	-	-	-	-	0.806	0.392	0.254	0.205	0.125
	4	-	-	-	-	-	0.553	0.387	0.325	0.214
	5	-	-	-	-	-	-	0.805	0.724	0.56
	6	-	-	-	-	-	-	-	0.912	0.738
	7	-	-	-	-	-	-	-	-	0.83
	8	-	-	-	-	-	-	-	-	-

Velocity	Initial number of dams in channel	Subsequent number of dams in channel: 0	1	2	3	4	5	6	7	8
	0	-	0.711	0.268	0.272	0.008	4E-04	3E-05	5E-06	3E-07
	1	-	-	0.46	0.065	0.023	0.002	2E-04	3E-05	2E-06
	2	-	-	-	0.268	0.125	0.015	0.002	5E-04	7E-05
	3	-	-	-	-	0.676	0.19	0.058	0.023	0.005
	4	-	-	-	-	-	0.371	0.14	0.065	0.018
	5	-	-	-	-	-	-	0.572	0.358	0.152
	6	-	-	-	-	-	-	-	0.726	0.384
	7	-	-	-	-	-	-	-	-	0.598
	8	-	-	-	-	-	-	-	-	-

Maximum Effect (ME) Approach

Depth	Initial number of dams in channel	Subsequent number of dams in channel: 0	1	2	3	4	5	6	7	8
	0	-	0.257	0.044	0.008	3E-04	2E-05	1E-06	2E-09	4E-10
	1	-	-	0.379	0.124	0.013	0.002	2E-04	1E-06	4E-07
	2	-	-	-	0.488	0.095	0.019	0.004	6E-05	3E-05
	3	-	-	-	-	0.342	0.113	0.036	0.001	8E-04
	4	-	-	-	-	-	0.552	0.27	0.03	0.02
	5	-	-	-	-	-	-	0.609	0.112	0.082
	6	-	-	-	-	-	-	-	0.285	0.227
	7	-	-	-	-	-	-	-	-	0.906
	8	-	-	-	-	-	-	-	-	-

Velocity	Initial number of dams in channel	Subsequent number of dams in channel: 0	1	2	3	4	5	6	7	8
	0	-	0.201	0.011	0.001	2E-05	7E-07	3E-09	3E-13	5E-14
	1	-	-	0.222	0.054	0.005	4E-04	1E-05	7E-09	2E-09
	2	-	-	-	0.465	0.094	0.018	0.001	4E-06	2E-06
	3	-	-	-	-	0.344	0.108	0.013	1E-04	6E-05
	4	-	-	-	-	-	0.527	0.132	0.005	0.003
	5	-	-	-	-	-	-	0.373	0.029	0.018
	6	-	-	-	-	-	-	-	0.197	0.146
	7	-	-	-	-	-	-	-	-	0.872
	8	-	-	-	-	-	-	-	-	-

<0.05	p-value indicates significant difference in comparison
>0.05	No significant difference
-	No comparison made

FIGURE 2-5 Comparisons of reach depth and velocity distributions to determine the number of beaver dams where significant changes occur using the Natural Sequence (NS), Spatially Equivalent (SE), and Maximum Effect (ME) approaches. The rows show how many dams were included initially in the study reach for each method. The column numbers represent the addition of subsequent dams within the study reach to provide a comparison of all dam combinations. The p-values are where significant changes in distributions occur (shown in grey). If you begin at row zero (initial simulation contains zero dams), moving across the columns show that three additional dams are needed to introduce significant change in the distribution of depth values. Row two (initial simulation contains two dams) requires the construction of five dams for the distribution of depth values to significantly change from the depth distribution with two dams present. Once you have three dams, the addition of further dams no longer introduces a significant change. From here we conclude that the number of dams required to significantly alter depth values using the NS method is 3.

TABLE 2-4 For each of the three approaches, the number of beaver dams required in the model before changes in the distributions of depth and velocity values due to the addition of more dams are no longer statistically significant was determined. The determined values are given along with the resulting estimated beaver dam density.

Approach	Number of Dams Needed for Change in Depth	Resulting Beaver Dam Density (Dams/km)	Number of Dams Needed for Change in Velocity	Resulting Beaver Dam Density (Dams/km)
Natural Sequence	3	4.5	5	7.6
Spatially Equivalent	3	4.5	5	7.6
Maximum Effect	5	7.6	6	9.1
Average	3.7	5.5	5.3	8.1

Discussion

Numerous studies have investigated the impacts of beaver dams on local hydrology, geomorphology, and channel hydraulics (Meentemeyer and Butler, 1999; Nyssen et al., 2011; Pollock et al., 2007; Westbrook et al., 2006) and have led to beaver being used as a restoration tool (Pollock et al., 2007; Pollock et al., 2014). Hydraulic variability has been used to represent aquatic species habitat (Ghanem et al., 1996; Lamouroux et al., 1998; Milhous and Waddle, 2012; Rabeni and Jacobsen, 1993) and some studies have quantified shifts in hydraulic characteristics due to beaver dams (Green and Westbrook, 2009; Smith and Mather, 2013). However, there is a gap in the literature that establishes the shifts in hydraulic characteristics as a quantifiable metric for determining the success of restoration projects. When beaver dams are used as the mechanism of influencing change, the changes in hydraulics and substrate composition can be used as clear indicators of improved fish habitat availability and variability, and thus as an indicator of success for many restoration efforts.

Substrate Data

Similar to other studies (Green and Westbrook, 2009; Smith and Mather, 2013) there is evidence of disrupted sediment trends throughout our study reach as a result of beaver dam activity. We observed a slight fining trend in the upper, non-impacted reach for the D50 of the riffles, bars, and pools (Figure 2-2). However, the fining trend is immediately disrupted upon entering the lower, beaver impacted reach and no distinct trend is manifested throughout the remainder of the reach. Instead, spatial heterogeneity and patchiness (Sullivan et al., 1987) is evident as beaver dams influence the distribution of surface substrate composition. Beaver dam construction, maintenance, and failure throughout the reach have significantly impacted longitudinal sediment storage and transport patterns (Butler and Malanson, 2005). Decreased velocities through beaver ponds encouraged the deposition of fines in the channel (Levine and Meyer, 2014) as observed throughout areas of beaver dam backwaters (river distance 400 in Figure 2-2). Scour pools and riffles of coarser substrate form below beaver dams as the sediment-starved water moves downstream (Gurnell, 1998; Smith and Mather, 2013) leading to the formation of a step pool sequence (Naiman et al., 1988) (see river distance 0 and 400 m in Figure 2-2). Evidence of redistributed sediment following beaver dam failure is seen both downstream and upstream of the dam located near river distance 600 m. Upstream of this dam, the sediment stored in the former beaver pond formed a cluster of bars which transformed the single thread channel to multiple threads. Some of the stored sediment was evacuated downstream from the dam (Butler and Malanson, 2005; Levine and Meyer, 2014), and reworked existing riffles causing their slight fining. Qualitative observations in subsequent years showed that some of the finer sediment was transported farther downstream in pulses creating a new riffle surface and small in-channel bars. Newly reworked and redistributed gravel helps meet necessary requirements for the formation of redds and successful spawning (Bjornn and Reiser, 1991; Harig and Fausch, 2002). Areas in the lower reach that were not impacted by beaver dams show similar sediment size distributions as the riffles, pools and bars in the upper, non-impacted reach (river distance 250-300 m). Using substrate data as a characteristic of geomorphic processes, there is a clear increase in temporal variability and spatial heterogeneity as a result of beaver dams in the lower, beaver impacted reach. Heterogeneity in geomorphic features can then be linked to habitat diversity and availability (Brierley and Fryirs, 2013) and species richness (Guégan et al., 1998).

Reach Comparison Results

Understanding the impact on hydraulics as a function of beaver dam construction, maintenance and/or failure is essential for determining the magnitude of change that can be expected throughout other aspects of the system. Overall, the impacts of beaver dams on general physical conditions of streams are well documented (Gurnell, 1998; Pollock et al., 2003; Rosell et al., 2005). However, few studies quantify the shifts in hydraulics. Even though the change in depth and width may be noted, their primary focus is not quantitative nor do they provide comparisons to conditions prior to colonization (Naiman et al., 1988; Nyssen et al., 2011; Westbrook et al., 2006; Woo and Waddington, 1990).

Similar to other studies (Green and Westbrook, 2009; Smith and Mather, 2013), we observed an increase in average depth and width, and an average decrease in velocity (Table 2-3). Furthermore, we found statistically significant shifts reach depth, width, and velocity even with a low number of beaver dams (Figure 2-5). Green and Westbrook (2009) observed similar trends, noting a decrease in widths and an increase in velocities following the removal of beaver dam. While the decrease in width was not quantified, the average increase in channel velocity was determined to be about 81% at bankfull flows.

A majority of results in the literature associated with changes in hydraulic characteristics come from measurements upstream and downstream of dams which does not allow for direct comparison with our study. Meentemeyer and Butler (1999) saw ranges of 19-100% decrease in velocity upstream of dams compared to downstream velocities during low flow periods. Smith and Mather (2013) highlighted differences in stream depth, width, and velocity between sites located upstream and downstream of beaver dams, however results were reported as a total change in depth and cannot be compared as a percent difference. Regardless, similar trends were observed and indicate that beaver dams increase depth and width, resulting in an overall decrease in velocity.

As we expected from results in previous studies, the introduction of beaver dams in the lower, beaver impacted reach had significant impacts on the distributions of depth, width, and velocity values when compared to the upper, non-impacted and lower, pre-beaver reaches (Table 2-3). While the shifts in the average values and overall variability in Comparison 1 (upper, non-impacted vs. lower, beaver impacted) and Comparison 2 (lower, pre-beaver vs. lower, beaver impacted) were both significantly different, some of the variability is due to the natural differences in the upper and lower reach as seen in Comparison 3 (upper, non-impacted vs. lower, pre-beaver). The lower, pre-beaver reach was 23-27% deeper on average and 12-17% narrower than the upper, non-impacted reach, but velocities remained similar at only 0-3% difference. This could be due in part to the channel being relocated and rebuilt in the lower reach. The banks were stabilized and the stream could not adjust widths, causing many sections to degrade, become rectangular, and disconnect from the floodplain. Although Comparison 3 indicates a significant difference in channel geometry between the lower, pre-beaver and upper, non-impacted reaches, the introduction of beaver dams still caused a large shift in hydraulic characteristics (Comparison 2). Overall, it is evident that beaver dams introduce variability in depth and velocity values between reaches and that the variability cannot be solely attributed to natural differences in channel geometry and slope. These three comparisons also indicate that other impacts related to channel hydraulics, such as improvements in fish habitat heterogeneity (Kemp et al., 2012), riparian corridor structure and productivity (Westbrook et al., 2006), alterations in hydrology (Nyssen et al., 2011; Westbrook et al., 2006; Woo and Waddington, 1990), increases in sediment storage (Butler and Malanson, 1995; 2005; Green and Westbrook, 2009), and influences in downstream aquatic ecology (Fuller and Peckarsky, 2011), may also differ.

Beaver Dam Densities

In order to effectively restore impaired stream systems, actions must be implemented that directly address the cause of degradation without exceeding the physical or biological potential of the site (Beechie et al., 2010). Oftentimes, conventional stream restoration techniques can be costly, involving heavy machinery and intense channel modification and re-vegetation (Wheaton et al., 2012). Recently, river managers are turning away from the conventional approach of using a hard engineering solution and are instead searching for ecologically based methods for improving impaired and degraded streams. Many channel restoration projects aim to reconnect incised, channelized streams with the floodplain, reduce erosion, and increase stream bank stabilization, improve the riparian corridor, and increase overall channel and habitat complexity (Pollock et al., 2007; Pollock et al., 2014; Roper et al., 2002; Wheaton et al., 2004). Previous research has shown the ability of beaver dams to have a significant impact on each of these areas. As a result, many restoration projects are beginning to use beaver and beaver dam structures as an ecological and cost effective approach in restoration and management (Andersen and Shafroth, 2010; Burchsted et al., 2010; Curran and Cannatelli, 2014; DeBano and Heede, 1987; DeVries et al., 2012; Pollock et al., 2014).

This simple modeling approach provides a first cut for determining the beaver dam densities required to begin meeting restoration goals related to depth and velocity distributions. Both the NS and SE approaches required fewer beaver dams to reach statistical significance than the ME approach for both depth and velocity (Figures 2-4 and 2-5). The NS and SE approaches required 3 dams for depth and 5 dams for velocity while the ME approach required 5 for depth and 6 for velocity. We originally hypothesized the ME approach would require fewer dams to reach a point of insignificant change and it seemed counterintuitive that more dams were needed with a maximized backwater effect. However, in the NS and SE approach, there was no further significant change due to the ineffective location of the dams. Some dams ended up with little backwater due to either steeper slope or close proximity of another dam. These dams had a minimal, localized impact on the hydraulics of the stream, and were not able to significantly alter the overall distribution of values. However, using the ME approach, locations were selected where the backwater effect was continually maximized and not inhibited by slope or other dams, allowing for later dams to still have a significant impact on the system.

By understanding the beaver dam density where hydraulic variability is maximized, restoration goals of habitat availability can be assessed while minimizing required resources. Whether beaver are introduced to the system and left to establish complexes alone or if structures are installed to encourage dam building (Pollock et al., 2007; Pollock et al., 2014), the essential complementary habitat (Schlosser, 1995) will be established to improve necessary deep pool habitat for maintaining fish populations (Harig and Fausch, 2002; Snodgrass and Meffe, 1999; White and Rahel, 2008). This approach could also be applied to a variety of other situations with different criteria and objectives. If the goal is to have deep pool habitat, beaver dams or similar structures could be built where natural pools already exist or where channel slopes are greater. The ME approach could be applied in locations where channel incision is a concern, allowing for an extended backwater effect while not requiring a dam with a height that would likely fail in high flows (Wheaton et al., 2012). For each individual case, it is important to consider that the material used in dam development and the total pool volume created will greatly impact habitat creation.

Beaver dams introduce spatial heterogeneity in sediment size distributions and increases variability in channel hydraulics. These metrics can be used as a surrogate for habitat variability and availability that is necessary for fish and other aquatic species for growth, survival and reproduction (Harig and Fausch, 2002; Snodgrass and Meffe, 1999; White and Rahel, 2008). Overall, habitat suitability is dependent on chemical, thermal, and physical characteristics (Dodds and Whiles, 2010), each of which are directly influenced by channel hydraulics (N. M. Schmadel et al., 2015; N.M. Schmadel et al., 2014b). From a management standpoint, specific restoration goals related to improving fish habitat can be achieved through the placement of beaver dams on a reach. Whether beaver naturally construct the dams themselves, or if beaver dam structures are installed at selected locations, a beaver dam density can be determined that will maximize restoration efforts while minimizing the resources used. However, this simplistic relationship between channel hydraulics and habitat availability should be carefully examined and predictions should be tested before any restoration using beaver dams takes place. This modeling approach is simply intended as a starting point for restoration projects. Post-restoration monitoring is recommended to continue to evaluate the progress and adjust individualized approaches.

Conclusion

This study quantifies the impacts of beaver dams on channel hydraulics and substrate characteristics within a mountain stream through the development and application of a 1D hydraulic model and analysis of longitudinal substrate data. Through the comparison of a lower, beaver impacted reach and upper, non-impacted reach, it is evident that beaver dams disrupt general trends in sediment distributions. Through beaver dam construction, maintenance and failure, sediment is stored and reworked, introducing greater spatial heterogeneity and variability in substrate. Results from a 1-D hydraulic model indicate that the introduction of beaver dams cause significant change in the distributions of the depths, widths, and velocities as shown by comparing the upper, non-impacted, lower, beaver impacted, and lower, pre-beaver reaches. Even though there were some differences in hydraulics due to natural variability in channel geometry alone, there was still significant change introduced by beaver dams. Not only was there an increase in variability in hydraulic values, but spatial heterogeneity in depth, width, and velocity was also observed throughout the reach. We also show that a relatively low number of dams are required to cause a significant change in these hydraulic parameters. Three approaches to estimating the influence of different beaver dam densities (Natural Sequence, Spatially Equivalent, and Maximum Effect) showed there are differences regarding the number of dams necessary to significantly alter depth and velocity distributions. Overall, this study has demonstrated the potential influence of beaver dams in meeting restoration goals of increased habitat availability through increased hydraulic variability and provides an estimate on the best use of resources in meeting these goals for mountain streams similar to Curtis Creek.

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CHAPTER 3

DETERMINING THE APPLICABILITY AND EFFECTIVENESS OF AN EMPIRICAL RELATIONSHIP IN ESTIMATING GROUNDWATER RECHARGE IN TWO WESTERN, MOUNTAIN WATERSHEDS BASED ON PRECIPITATION AND DESCRIPTIVE SURFACE CHARACTERISTICS

Abstract

A majority of research regarding groundwater recharge in the western United States has focused on basin-fill aquifers and has given little attention to recharge rates within the adjacent mountain blocks. Developing estimates of recharge can be difficult, data intensive, and inherently error-prone. Researchers in Wisconsin successfully developed a simplified conceptual model and empirical relationship to predict the proportion of precipitation that enters an aquifer by developing an R/P term and relating it to a variety of hydrogeological, topographic, and land cover parameters. The researchers acknowledge that the relationship had not been tested outside of Wisconsin, yet were confident that the ratios developed would still govern groundwater recharge rates if applied to other systems. We applied their relationship to two western, mountain watersheds to determine if the driving forces defined in these relationships remain relevant. The independent application of the method to each watershed stressed the importance of meeting simplifying assumptions, highlighted the need for more comprehensive geospatial datasets, and demonstrated that existing simplified empirical models may not be suitable for estimating groundwater recharge in mountain watersheds.

Introduction

Groundwater (GW) – surface water (SW) interactions vary temporally and spatially, and significantly influence the quantity and quality of both resources (Winter, 1999). Traditionally, GW and SW have been viewed as two separate resources in management decisions (Winter, 1998) even though earlier research has shown their interconnectedness (Rorabaugh, 1964; Theis, 1941; Wahl and Wahl, 1988). Interactions between GW and SW bodies can be difficult to quantify, and new methods are constantly being developed to improve estimates and understanding of processes (Fleckenstein et al., 2010). Withdrawal of water from streams can deplete GW (Winter, 1998) while over pumping can decrease the total stream flow (Wahl and Wahl, 1988). Understanding the GW-SW interactions in basin-fill aquifers has been the focus of the majority of research, with little attention being given to mountain block hydrology and aquifers (Ajami et al., 2011; Wilson and Guan, 2004). While aquifer demand is continually increasing, managers need a better understanding of the sources, processes and characteristics that govern GW-SW interactions for both basin-fill and mountain aquifers.

Recharge to aquifers is the result of the percolation of either precipitation or streamflow. Sources of recharge to basin-fill aquifers are generally lumped into two categories of Mountain Front Recharge (MFR) or Mountain Block Recharge (MBR) based on geographic location and flow paths into the basin-fill aquifer (Wilson and Guan, 2004). MFR enters the basin-fill aquifer from the surface as infiltration through the vadose zone to the saturated zone, either in the form of precipitation or as streamflow moving across the mountain front zone. MBR is classified as subsurface inflow from saturated zones originating in the adjacent mountain blocks (Figure 3-1).

Recharge to mountain aquifers is a part of the complex mountain block hydrology. All mountain aquifers cannot be classified together due to variable geology, and sources cannot be categorized simply as either MFR or MBR. Instead, identifying the different sources of

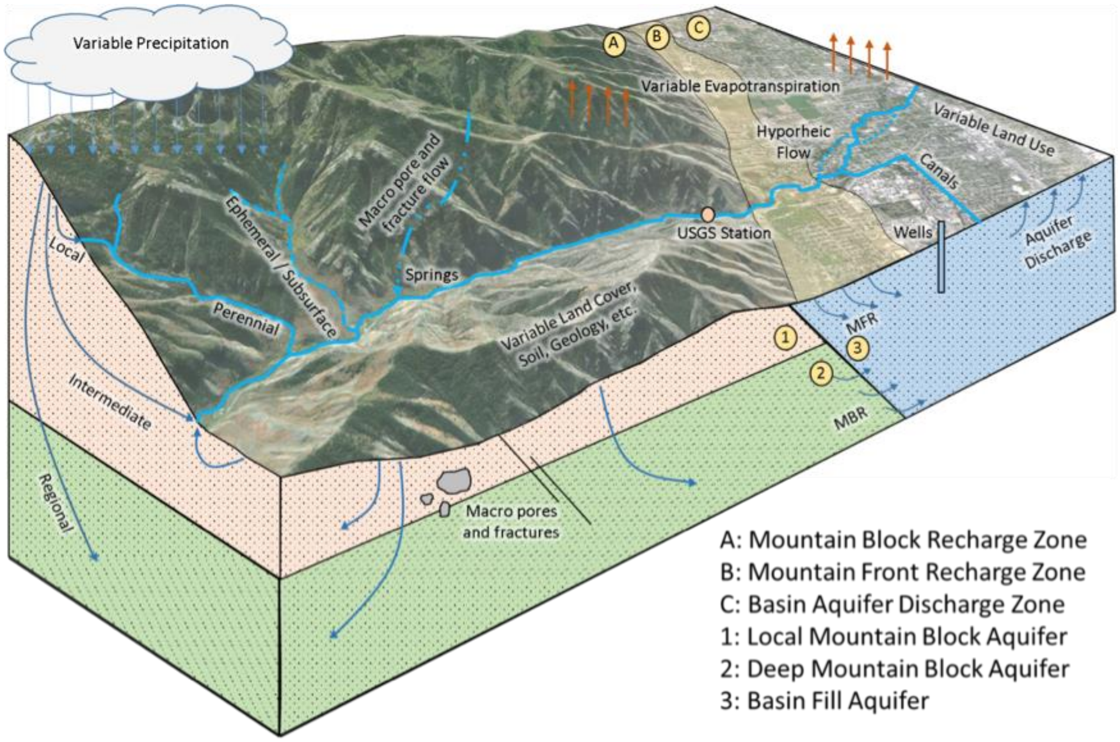

FIGURE 3-1 Three types of aquifers are considered throughout this study: local mountain block, deep mountain block, and basin-fill. A variety of processes, watershed characteristics, and fluxes determine recharge to each. The USGS gaging stations discussed in this study are located near the mouth of each canyon and measure flow and baseflow from the Local Mountain Block aquifer only.

groundwater to mountain aquifers requires a better understanding of the complex processes involved in mountain block hydrology. While basin-fill aquifers are recharged by MFR or MBR, sources of recharge to mountain aquifers become differentiated by the overall length of the flowpaths (Wilson and Guan, 2004). Some precipitation enters the aquifers through percolation and infiltration and shortly (spatially and temporally) leaves as baseflow or interflow. Other flow enters the aquifer higher in the mountain block or in other drainages and travels greater distances before contributing to baseflow. Differences in flowpaths (local, intermediate, regional) are dependent upon topographic relief (Toth, 1963) as well as geology (Spangler, 2001). While difficult to quantify, understanding the length of flowpaths is essential in understanding gains and losses along river systems to aquifers. Percolation through bedrock is dependent upon the underlying geology (which is variable over space) and the type and intensity of precipitation (which are variable over space and time). While many studies focus on identifying sources of groundwater, this study focuses only on the total volume of water that enters and exits aquifers. Therefore we use the term "recharge" in the general sense to identify all sources, regardless of the origin.

In order to clarify the terminology used and processes referenced, a simple conceptual model was developed (Figure 3-1). The mountain block aquifer was broken into two conceptual portions: one that contributes recharge to basin-fill aquifers as MBR (deep mountain block) and the portion of the aquifer that contributes to baseflow in the mountain stream before entering the valley (local mountain block), ultimately recharging basin-fill aquifers as MFR. Different watershed characteristics, recharge sources, flowpaths, and fluxes are identified.

Several factors determine the amount of precipitation that enters aquifers. Developing an understanding of these factors is crucial to estimating recharge (Anderholm, 1998; Wilson and Guan, 2004) in either basin fill or mountain aquifers. Watershed and meteorological characteristics such as precipitation (type and magnitude), interception, evapotranspiration (ET), bedrock percolation, geology (including formation type and faults), watershed and river gradient, soil characteristics, and general differences in flow paths through the mountain block each play a role in the overall mountain block hydrology and groundwater recharge. Many of these characteristics are included in a variety of hydrologic studies and models, however some are often difficult to quantify. For example, evapotranspiration (ET) is difficult to quantify, but has a significant impact on shallow subsurface flows and changes with land use, vegetation cover, and elevation. Because of the difficulty in estimating ET, some groundwater recharge studies only considered months for which evapotranspiration was minimal or negligible (Chen and Lee, 2003).

Because recharge cannot be measured exactly, a variety of methods have been developed to form estimates. These methods are categorized as hydrometeorologic, potentiometric, or surface water flow depending on the data used for each (Sophocleous, 1991). Common methods within each of these three categories include performing a water balance (Finch, 1998; Rushton and Ward, 1979), measuring water table fluctuations (Ketchum et al., 2000), and stream hydrograph separation (Mau and Winter, 1997). Because all estimation techniques are error-prone, it is suggested that two or more techniques be used to determine recharge rates (Halford and Mayer, 2000; Kao et al., 2012; Mau and Winter, 1997). However, many mountainous watersheds are data limited, making the usage of multiple techniques difficult. Often, the only continuous, reliable data available are discharge measurements from stream gaging stations, making stream hydrograph separation the only available technique for estimating groundwater recharge to mountain aquifers.

Stream hydrograph separation stems from the generally accepted idea that baseflow measurements can be used to represent GW recharge rates (Halford and Mayer, 2000). This assumption originates from a simplified groundwater balance of a small watershed, later described by Cherkauer and Ansari (2005) as:

$$\mathrm{I} + \mathrm{GW_{in}} = \mathrm{Q_{bf}} + \mathrm{GW_{out}} + \mathrm{ET} + \mathrm{NP} + \frac{\Delta \mathrm{S}}{\mathrm{t}} \qquad \text{Equation 1}$$

Where I is infiltration into the system, GW_{in} is groundwater influx to the watershed through aquifers, Q_{bf} is groundwater discharge to stream baseflow, GW_{out} is groundwater efflux from the watershed through aquifers, ET is evapotranspiration losses from the watershed, NP is net pumpage of groundwater into or out of the watershed, and $\Delta S/t$ is the rate of change of groundwater storage with respect to time (Cherkauer and Ansari, 2005). If watersheds can be selected where $GW_{in} = GW_{out} = NP = \Delta S/t = 0$, and if recharge is defined as net groundwater recharge (I – ET), then the previous equation simplifies to:

$$\text{Recharge} = \text{Net Recharge} = Q_{bf} = \text{Stream Baseflow} \qquad \text{Equation 2}$$

This simplified model assumes that groundwater and surface water divides coincide, that there is no human transport of water into or out of the watershed, and that groundwater storages do not significantly change year to year. The model also suggests interflow to be a negligible component of streamflow and assumes that the hydrograph can be separated into direct surface runoff and groundwater discharge (Kulandaiswamy and Seetharaman, 1969). Therefore, watersheds with significant surface water storage (lakes, reservoirs, wetlands) must be avoided to meet this assumption. Lastly, the stream hydrograph separation method for determining groundwater recharge requires small-scale temporal resolution flow measurements, most often provided by the U.S. Geological Survey (USGS) or constructed through field observations at independent sites.

Approaches for baseflow estimation using stream hydrograph separation can be grouped into two categories: graphical hydrograph separation and using tracer mass balances. The graphical approaches rely solely on stream discharge data, while the mass balance methods require chemical concentrations in stream discharge and end-member constituents, such as runoff and baseflow (Miller et al., 2015). Different procedures of graphical hydrograph separation and mass balance approaches are outlined well in the literature (Chen and Lee, 2003; Cherkauer and Ansari, 2005; Kao et al., 2012; Mau and Winter, 1997; Miller et al., 2015; Yeh et al., 2007). Chen and Lee (2003) suggest that graphical methods may be more suitable for estimating groundwater recharge in mountainous regions than in the plains regions or regions with less topographic relief due to the differences in the hydraulic properties of the soil and available data. Each of the methods described for estimating recharge requires a significant amount of data and can only be performed in watersheds where monitoring infrastructure exists.

Even with the complexity of estimating groundwater recharge, it is crucial that water resource managers have adequate information to make informed decisions in order to meet the needs of an ever growing population. The availability of recharge cannot be adequately factored into the planning process until it has been quantified. Because of the scarcity or difficulty in obtaining necessary data for estimating GW recharge, Cherkauer and Ansari (2005) suggested the need to develop a method that is useful to GW resource managers and meets four conditions. The method should (1) focus on the influx, or sources of recharge, to saturated systems, (2) be able to accurately define those sources down to the scale of several square kilometers, (3) rely exclusively on readily available data, and (4) be readily applicable across the regional scale.

In order to meet the suggested conditions, Cherkauer and Ansari developed a method to be used as a first approach to understanding recharge rates based on available surface information. They developed a simplified conceptual model (Figure 3-2), where the watershed is represented as a rectangle with the horizontal length being the length of the main channel (L_c), the width being the (drainage area / L_c), and the average length of the surface flow path (L_f) to the channel being half the width. Watersheds were assumed to be internally homogenous with each physical property (elevation, slope, effective soil conductivity, etc.) represented as the mean value of the entire drainage area. The model assumes that as precipitation falls within the watershed, water flows toward the main channel. Along the flow path, some of the water infiltrates and enters the river as baseflow, while the remainder of the water enters as surface runoff. The entire

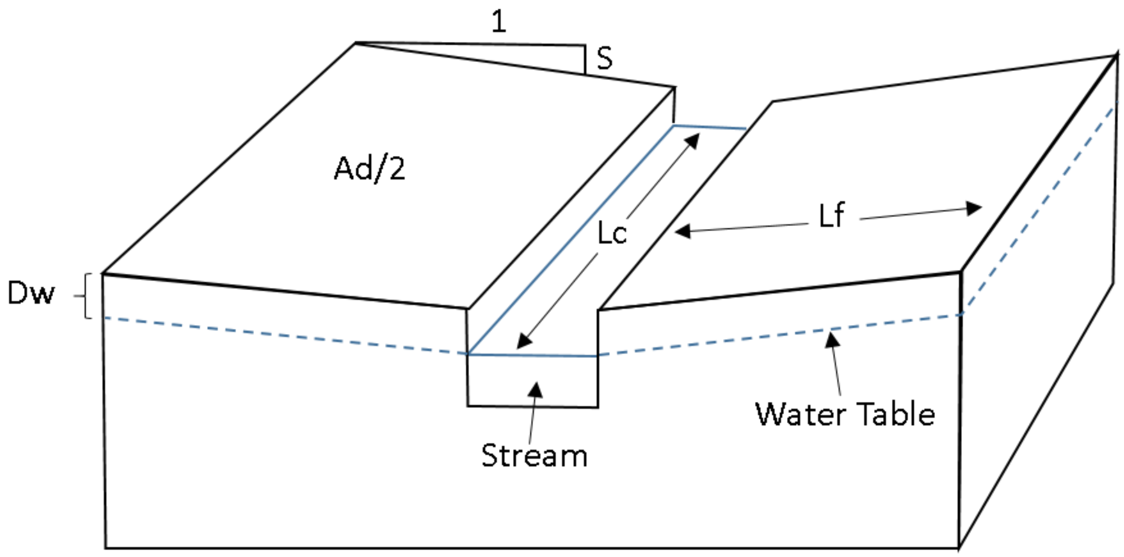

FIGURE 3-2 Conceptualized geometric dimension of study watershed. Half of the watershed of drainage area A_d is shown; other half is mirror image. L_c is length of main channel. L_f, length of overland flow, is $A_d/(2\ L_c)$, S is average surface slope toward stream, and D_w is the average depth to the water table. Figure adapted from Cherkauer and Ansari (2005).

drainage area, therefore, acts as a recharge area, and the stream is considered the only point of groundwater discharge. The previously mentioned physical properties of the watershed determine the partitioning of precipitation to either groundwater or runoff (Cherkauer and Ansari, 2005). The developed model is highly simplified and does not account for spatial heterogeneity in physical properties. It also does not account for antecedent soil moisture, assumes annual total precipitation depth is uniform across the watershed and that it occurs with uniform intensity, and that there are no other groundwater discharge points besides the stream.

In order to relate the surface characteristics to recharge rates, they first developed an estimate of recharge. Using the recession curve analysis presented by Linsley et al. (1982), Cherkauer and Ansari (2005) determined baseflow for multiple study watersheds at selected monitoring stations. Total annual precipitation was collected and related to annual recharge volumes through a dimensionless ratio of recharge per unit precipitation (R/P). Using total annual volumes of both precipitation and recharge, coupled with the model assumption that the entire watershed acts as a groundwater recharge area, the spatial and temporal variability in precipitation is eliminated. Physical characteristics used to describe the watershed included topography, geology, and land cover were quantified. A regression analysis was then performed between the R/P ratio and the physical characteristics of the watershed. To determine the viability of the regression equation, three levels of testing were used. The relationship was (1) tested against a second data set, (2) used to calculate recharge rates at sites outside the study area, and was (3) tested by comparing it to recharge rates developed through other methods at the same locations. The method remains to be tested outside the humid region of southeastern Wisconsin; however, Cherkauer and Ansari believe the dimensionless ratios of flux, travel distance, and area will control the relationship outside of their study area, and that only the regression coefficients will change.

While the method applied by Cherkauer and Ansari (2005) predicted recharge within 20% for the systems in which they were working, the same spatial characteristics may not be able to accurately portray the complexity of hydrological processes in mountainous watersheds in the western United States. As mentioned earlier, the processes that determine the amount of precipitation that makes it to aquifers include the amount and type of precipitation, evapotranspiration, soil, geology, vegetation, etc. In mountain regions, topographic complexity introduces high variability in precipitation, temperature, and vegetation (Bales et al., 2006) making area averages difficult to determine and oftentimes meaningless (Gee and Hillel, 1988). Shallow soils and steep slopes underlain by bedrock of differing permeability decrease the available groundwater storage in mountain regions (Wilson and Guan, 2004). Several studies have tried to quantify travel times and sources of groundwater recharge through complex mountain watersheds (Spangler, 2001) and determine the volume and rate of GW flow through mountain block regions into basin-fill aquifers (Manning and Solomon, 2003), ultimately showing that groundwater and surface water divides do not coincide, and that the stream is not the only point of discharge from the aquifer.

Overall, many of the simplifying assumptions made by Cherkauer and Ansari (2005) while developing their empirical relationship are not justifiable when dealing with western, mountain watersheds. However, our objective was to apply Cherkauer and Ansari's methods to two mountain watersheds to determine the applicability of applying an existing simplified empirical relationship in more complex watersheds, and determine if the driving characteristics identified still remain the driving parameters in estimating groundwater recharge. The success of these objectives will be determined by the ability of the relationship to accurately predict changes in R/P values, thus demonstrating that the relationship has captured and represented the different spatial and temporal processes and parameters that govern groundwater recharge.

Methods

Site Descriptions

In order to determine the applicability and effectiveness of using topography, hydrogeology, land cover, precipitation, and other watershed characteristics to estimate GW recharge in western, mountainous watersheds, sites were selected and data collected within two local watersheds: the Logan River and Red Butte Creek. Both watersheds are part of an interdisciplinary research and training program called iUTAH (innovative Urban Transitions and Aridregion Hydro-sustainability). This project has developed a network of climate and aquatic monitoring stations called GAMUT (Gradient Along Mountain to Urban Transitions) stations. Each of these watersheds share many common characteristics, yet are different enough to make a comparison to determine the suitability of this approach to two different types of mountainous western watersheds.

Logan River

The Logan River (LR) (Figure 3-3a) is centrally located in the Bear River mountain range east of Logan, Utah. With headwaters near the Utah-Idaho border, this third order river flows southwest through a watershed characterized by limestone geology and karst topography (Spangler, 2001). The watershed is mostly natural land cover with little development other than the paved and dirt roads and the occasional cabin or summer home. The system receives the majority of the precipitation in the form of snow. The hydrograph is snowmelt dominated and receives groundwater discharge throughout the summer from springs and diffuse sources. The river is gaged by the USGS (Site 10109000) above State Dam near Logan, Utah. Above this point, the river drains 214 square miles. Above the USGS gaging site, the river is dammed in two locations. Each of the dams on the river are used for hydroelectric generation that are simply flow through dams with minimal storage. One main diversion (USGS Site 10108400) diverts water from the river before the main gaging station (10109000). Additionally, Logan City pulls a significant amount of drinking water from Dewitt Springs. Daily flow data for the springs are recorded via Logan's supervisory control and data acquisition (SCADA) system.

Red Butte Creek

Red Butte Creek (RBC) (Figure 3-3b) drains a small watershed located at the northeast end of the Salt Lake valley. This small, second order stream flows southwest through quartzite, limestone and sandstone before entering the valley by the University of Utah (Ehleringer et al., 1992; Mast and Clow, 2000). The watershed is characterized by mostly natural land cover, with the only development being a single dirt road to the top of the watershed. The creek quickly

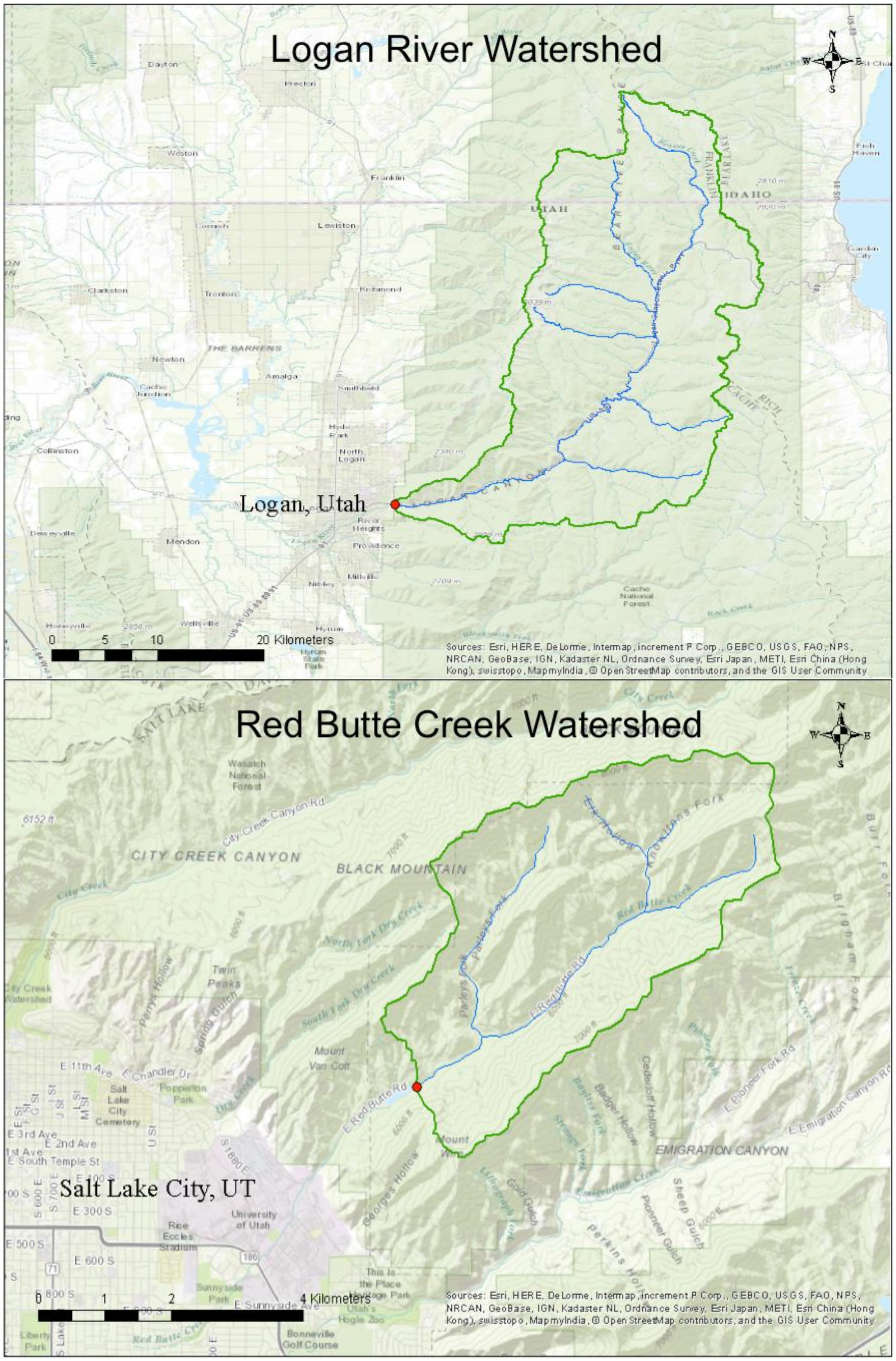

FIGURE 3-3 Logan River (a) and Red Butte Creek (b) Watersheds near Logan and Salt Lake City, Utah, respectively. The two sites differ significantly in various watershed characteristics, most notably the size. Both watersheds are gaged by the USGS (red point, both maps).

transitions from a relatively pristine watershed to a highly urbanized area as it flows from the headwaters and drains into the Jordan River. Climate in the region is characterized by hot dry summers, followed by long cold winters (Ehleringer et al., 1992). Precipitation in the watershed occurs mainly as snow, and the hydrograph is generally snowmelt dominated (Mast and Clow, 2000). Much of the upper watershed has limited public access, as the area is managed as a Research Natural Area. Within the research area is a single impoundment, Red Butte Reservoir. Originally constructed by the U.S. Army, the reservoir is managed by the Central Utah Water Conservancy district as habitat for a refuge population of the endangered June sucker (*Chasmistes liorus*). The reservoir generally maintains a constant level, only decreasing storage to be able to capture and mitigate spring runoff events (Mast and Clow, 2000). A single USGS gage measures flow above the reservoir (Site 10172200). The stream at the USGS station drains 7.25 square miles.

Data Acquisition

Acquired data were categorized as either hydrologic or spatial data. Hydrologic data consisted of discharge and precipitation and were used for the development of R/P ratios for both the Logan River and Red Butte Creek. Discharge was downloaded from the USGS for stations 10109000 (Logan River) and 10172200 (Red Butte Creek). Because of the diversions upstream of the USGS station on the Logan River, discharge data from the USGS station 10108400 (Highline Canal diversion) and daily usage values from Dewitt Springs (drinking water source) reported by the Utah Division of Water Rights were also downloaded. No further data were needed from the RBC watershed as there are no diversions above the USGS gaging station. Annual precipitation data were downloaded from the PRISM Climate Group for the contiguous United States in the form of a raster with 4km cells. A total of 35 years (1981-2014) worth of discharge and precipitation data were compiled.

Spatial data were collected as GIS coverages and were categorized as either topographic, hydrogeologic, or land cover (Table 3-1), using the same distinction as Cherkauer and Ansari (2005). Topographic data included watershed area, watershed slope, and length of channel. The area and slope datasets were developed from the National Elevation Dataset (NED). Watershed delineations were automated using a Python script and the USGS stations as pour points. Results from the delineation included watershed area and average slope. Channel length was determined from the National Hydrography Dataset (NHD) and indicates the cumulative length of channels in the drainage network.

Hydrogeologic data included coverages of hydraulic conductivity and the depth to the water table. For RBC, the data were available from the NRCS Soil Survey Geographic Database (SSURGO) as GIS coverages (Soils Hydrologic Group and Water Table Depth). Unfortunately, these coverages had missing data for much of the Logan River watershed. Instead, soils data were downloaded from the NRCS State Soil Geographic Database (STATSGO) from which hydraulic conductivity for the LR watershed was extracted. The hydraulic conductivity data are comparable between the SSURGO and STATSGO data, with the greatest difference being in the resolution of the datasets. The Water Table Depth coverage was also missing data over the Logan River watershed and sparse over RBC. In order to develop this coverage for the Logan River, spring and well data were acquired from the Utah Division of Water Rights. A depth to groundwater was obtained from each well log, while springs were assigned a depth of zero. Stream coverages were converted to points in GIS and also assigned a depth of zero. A point shapefile was generated using the latitude and longitude of each well, spring and stream point. Elevation data were assigned to the points from the NED coverage. The depth of each point was subtracted from the elevation to form a new field of GW elevation. Using the kriging interpolation method in GIS, a raster of GW elevation was generated. The GW elevation layer was subtracted from the NED

TABLE 3-1 Different data sources and layers used throughout the project. Many of the ArcGIS coverages were incomplete over the Logan River watershed, leading to the need of outside data sources.

Source	Dataset	Layer	Data
US Geological Survey	USGS National Elevation Dataset	NED30m	Area, Slope
US Geological Survey	USGS National Hydrography Dataset	USA_NHD_HighRes	Length of Channel
Natural Resources Conservation Service	NRCS Soil Survey Geographic Database	USA_Soils_Water_Table_Depth	Depth to water table
US Geological Survey	USGS National Land Cover Database 2006	USA_NLCD_2006	Land cover (percentages)
Natural Resources Conservation Service	NRCS Gridded Soil Survey Geographic Database	USA_Soils_Hydrologic_Group	Hydraulic conductivity
Natural Resources Conservation Service	State Soil Geographic Database	STATSGO	Hydraulic Conductivity
Utah Division of Water Rights	Utah Division of Water Rights	Spring and Well Data	Depth to water table
PRISM Climate Group	PRISM Spatial Climate Dataset	Annual Precipitation Data	Total Precipitation
US Geological Survey	Daily Flow Data	NA	Total flow and baseflow

DEM to form a new Water Table Depth coverage. This method of developing a Depth to GW coverage was not possible in RBC due to the lack of well data. The data from SSURGO dataset (Water Table Depth coverage) were used instead as a best estimate, even though the coverage was not complete.

Land cover data included estimates of the percent of the watershed that was Natural, Developed, or Agriculture. This information came from the National Land Cover Dataset (NLCD). For this study, a 2006 coverage was used to best represent the land cover over the 35 years of hydrologic data. It was assumed that the 2006 NLCD coverage would remain consistent and be representative of the land cover over the 35-year span of hydrologic data since both watersheds have experienced little change in the proportion of natural, developed, and agricultural land cover based on a comparison of 2006 and 2011 data. It was assumed the proportions remained similar since 1981.

Data Analysis

Due to the general lack of data in both watersheds besides USGS gaging stations, graphical stream hydrograph separation was selected as the most appropriate method for estimating recharge for this study (Scanlon et al., 2002). A variety of software packages exist for graphical baseflow separation. The Web-based Hydrograph Analysis Tool (WHAT) (Lim et al., 2005) was selected due to availability and ease of use. Data could either be retrieved from the USGS site or could be uploaded. The WHAT has the ability to separate baseflow using either the one parameter digital filter (Lyne and Hollick, 1979) or the recursive digital filter (Eckhardt, 2005). For this study, the recursive digital filter was selected with a filter parameter of 0.98 and a Baseflow Index (BFI) maximum of 0.80 to represent perennial streams with porous aquifers. Data returned from the WHAT came in the form of daily estimates of total flow, direct runoff, and baseflow in ft^3/s. Daily flow rates were converted to volumes and summed over each year. Flow volumes were divided by watershed area so that the annual flow, annual direct runoff, and annual baseflow were reported as depths in meters. Annual precipitation totals were determined by averaging the raster cells from the PRISM dataset, using the watershed as a boundary. With annual baseflow (as an estimate of recharge) and precipitation values determined and having consistent units, annual R/P ratios were calculated.

Watershed average values for each of the coverages were also determined. Each coverage was clipped to the respective watershed boundary and subsequently summarized. Complete GIS analyses included watershed delineation, determination of slope from National Elevation Dataset coverages, calculation of total channel length from the National Hydrography Dataset, determination of the percent of natural, developed, and agricultural coverage, and calculation of the average depth to groundwater and soil hydraulic conductivity. In order to facilitate repetitive calculations, analyses were performed using Python scripting.

Relationship Between Recharge, Precipitation, and Spatial Characteristics

Cherkauer and Ansari (2005) successfully developed an empirical relationship between recharge, precipitation, and watershed spatial characteristics. The equation developed was in the form:

$$\frac{R}{P} = X_1\left(\frac{K_v}{S*(D)^{0.3}}\right) - X_2\left(\frac{D_w}{L_f}\right) + X_3(N) + X_4 \qquad \text{Equation 3}$$

where R/P is the ratio of recharge per unit precipitation, K_v is effective vertical soil conductivity (m/d), S is average watershed slope (m/m), D is the percent of developed land cover in the watershed, D_w is the average depth to the water table (m), L_f is the length of flow to the channel (km), which is area/(2*channel length), N is the percent of natural land cover in the watershed. X_1, X_2, X_3, and X_4 are adjustable coefficients to help fit the empirical relationship to the data.

The first term of the equation represents the ratio of vertical flux of water through the soil to the horizontal flux across the ground surface. The greater the slope or portion of developed area, the smaller the first term becomes, indicating less precipitation makes it to the water table as recharge. Similarly, if the hydraulic conductivity were small, infiltration would be less and more precipitation would become runoff, decreasing the amount of recharge. The second term also represents a ratio of vertical distance over horizontal distance traveled before entering either the water table as recharge or leaving the watershed as direct runoff. The third term is intuitive as recharge varies directly with the amount of natural cover, just as it varies indirectly with the proportion of developed land cover. The terms are presented in the equation in order of descending importance (Cherkauer and Ansari, 2005).

For this study, the equation needed to be slightly altered due to the nature of the watersheds. Both of the watersheds are characterized, for the most part, by natural cover. In order to preserve the meaning of the first term, a value of one was added to the percent development term, D. If the percent development becomes less than one or approaches zero, the value of the first term begins to increase drastically. Where the first term has the greatest impact (Cherkauer and Ansari, 2005), avoiding inflated values is critical. The impact of adding one to the D term was tested in order to ensure there was no significant change in values of percent developed cover above 1% (Figure B-1).

With annual R/P values and spatial characteristics determined, a regression analysis was performed independently for both LR and RBC watersheds to determine the coefficients for the equation. The regression was performed by minimizing the difference between observed and calculated R/P values by adjusting coefficients. Different starting estimates for the coefficients were used during the analysis (zero, positive and negative one, a positive and negative fraction, and a positive and negative large number). Data from 1981-2004 were used for the regression analysis, while data from 2005-2014 was withheld as a validation dataset, after determining precipitation values from the two time periods were comparable. Multiple solutions were reached depending on the starting value used for the coefficients. Any set of coefficients that had a value of zero or were more than 4 orders of magnitude greater or less than zero were discarded. The remaining solutions were tested using a bootstrapping technique that would run 10,000 simulations and check the distribution of simulated R/P values against the observed distributions. In the bootstrapping technique, a normal distribution (instead of a value) is assigned to each of the parameters of the equation when calculating R/P values. Because the surface characteristics used in this relationship had no measurable annual variability, standard deviations were estimated in two ways to try to reproduce the uncertainty similar to the original study. First, the standard deviation of each parameter was estimated to be 10% of the mean, and 5% the second time. Bootstrapping was performed twice for each watershed using the two different estimates of standard deviation. The solution with a distribution of R/P values most similar (smallest percent difference) to the observed R/P mean and standard deviation for both bootstrapping attempts was selected as the best solution for that watershed. The final equation for each watershed was then applied to the validation dataset (2005-2014) to determine if the relationship would produce similar results.

Application of Empirical Relationships to Different Watersheds

Once relationships were established independently for both the LR and RBC, they were tested against the dataset from the other watershed. This allowed a test of the transferability of relationships between watersheds that have some similar, but also unique, characteristics.

Results

Data Acquisition

The majority of the analysis for this work involved accessing, processing, and analyzing existing GIS coverages. However, in order to estimate an average depth to groundwater in the Logan River watershed, a GIS coverage had to be generated. Because the coverage was created from all available (although limited) data, there were no independent datasets against which the raster values could be compared. A figure of this developed coverage, along with other figures of the different coverages used are included in Figure B-2 along with a brief description.

Data Analysis

Baseflow separation performed using the WHAT returned daily estimates of total flow that were partitioned into direct runoff and baseflow for the calendar years 1981-2014 for both the LR and RBC (Figure 3-4). These were converted to annual depths of average total flow, direct runoff, and baseflow (Table B-1). The averages and standard deviations of the annual flow depths, average annual precipitation, and average annual R/P values for both watersheds are also compared (Table 3-2). The Logan River had an average R/P value of 0.37 with a range of 0.25 – 0.59. The average R/P value for Red Butte Creek was 0.19 with a range of 0.06 – 0.42. While average precipitation depths were similar between watersheds (Figure B-3), the area-normalized total flow depth and baseflow depth of the Logan River was more than double Red Butte Creek. R/P values in the Logan River were just under double the Red Butte Creek averages. Annual precipitation values determined from the PRISM model, baseflow, and observed R/P values appear to trend together over time, with years of higher precipitation also having higher baseflow (Figure 3-5). Because the R/P values also follow the same trend, it indicates that the relationship between baseflow and precipitation is not well correlated (Figure B-4). If it were, there would be no variability in R/P as an increase in baseflow would be normalized by an equivalent increase in precipitation.

GIS analyses of the watershed characteristics produced significantly different results for estimates of the hydrogeology, land cover, and topography for the Logan River and Red Butte Creek (Table 3-3). The Logan river watershed has a much larger drainage area, total length of channels in the drainage network, higher hydraulic conductivity, and depth to water table than Red Butte Creek. However, the average slope of the Red Butte Creek watershed is higher than the Logan River. Proportions of land cover are very similar between watersheds.

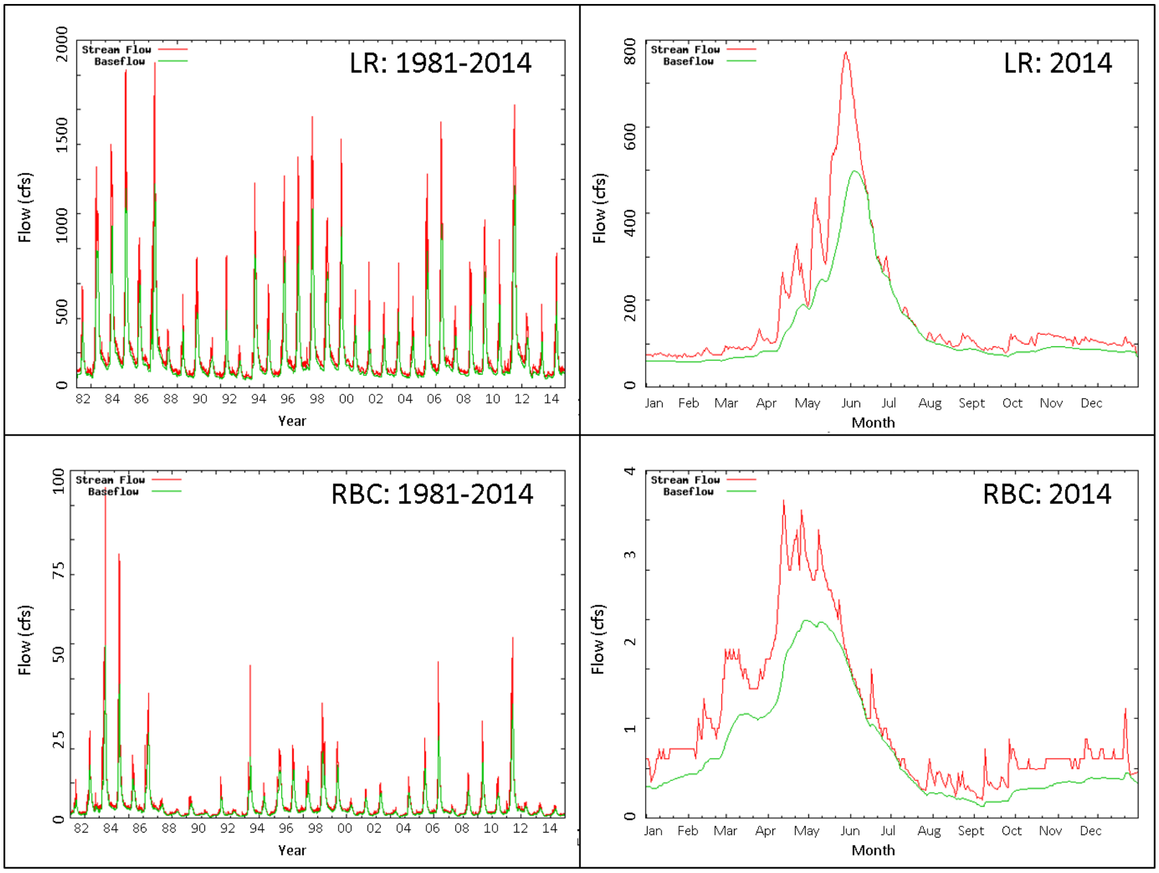

FIGURE 3-4 Results from the Web-based Hydrograph Analysis Tool (WHAT). Discharge time series for both the Logan River and Red Butte Creek are shown from 1981-2014. Red lines indicate the total flow and green represent calculated baseflow using recursive digital filters. Total flow and baseflow from the 2014 calendar year is also shown to demonstrate the baseflow separation.

TABLE 3-2 Discharge and precipitation values from 1981 – 2014 are compared to demonstrate the differences between watersheds.

Average Observed Values 1981-2014	Logan River		Red Butte Creek	
	Average	St. Dev	Average	St.Dev
Total Flow (m)	0.42	0.16	0.19	0.14
Total Direct Runoff (m)	0.09	0.04	0.04	0.03
Total Base Flow (m)	0.33	0.13	0.15	0.10
Total Precipitation (m)	0.89	0.19	0.76	0.16
R/P	0.37	0.10	0.19	0.10

FIGURE 3-5 Precipitation, baseflow and R/P time series for both the Logan River and Red Butte Creek watersheds. Precipitation totals were determined from PRISM estimates. Baseflow values were calculated using the WHAT. Both are reported as depths (m). The R/P values are simply the ratio of baseflow over precipitation.

TABLE 3-3 Spatial characteristics describing the Logan River and Red Butte Creek watersheds are given below. These were used to develop an empirical relationship to estimate the proportion of precipitation that eventually becomes groundwater.

	Variable	Logan River	Red Butte Creek
Hydrogeology	Hydraulic Conductivity(m/d)	0.91	0.10
	Depth to Water Table (m)	202.36	0.61
Land Cover	Natural (%)	98.83	100
	Developed (%)	1.06	0
	Agriculture (%)	0.12	0
Topography	Drainage Area (km2)	558.52	18.77
	Length of Channel (km)	128.75	8.81
	Slope (m/m)	0.34	0.51

Relationship Between Recharge, Precipitation and Spatial Characteristics

Results from the regression analysis that were tested using bootstrapping are included in Figure B-5. The solutions that proved to be best for each watershed ended up having very similar coefficients (Table 3-4) with the largest differences being in the second term that describes the depth the water table and the length of flow and the third term describing the portion of natural land cover.

Using the determined coefficients, the final empirical relationship for the Logan River is:

$$\frac{R}{P} = 0.01\left(\frac{K_v}{S*(1+D)^{0.3}}\right) - 0.0086\left(\frac{D_w}{L_f}\right) + 0.0115(N) + 0.01 \qquad \text{Equation 4}$$

The final empirical relationship for Red Butte Creek is:

$$\frac{R}{P} = 0.01\left(\frac{K_v}{S*(1+D)^{0.3}}\right) - 0.0101\left(\frac{D_w}{L_f}\right) + 0.0018(N) + 0.0099 \qquad \text{Equation 5}$$

With both relationships developed, the percent difference between observed and predicted values was determined for both watersheds. The average percent difference between the observed and predicted R/P values for the Logan River was -7% with a range between +40% and -50%. The average percent difference using absolute value of the differences was 22%. Red Butte Creek had an average percent difference of -31% and a range between +50% and -200%. The average percent difference using the absolute value of the differences was 55%. In general, the LR relationship predicted R/P with better precision than the RBC relationship (Figure 3-6). The LR relationship also performed better during the validation dataset than the development set, while the RBC relationship had consistent error for both datasets (Table 3-5). Negative values represent a predicted value larger than the observed value. Both relationships tend to overestimate R/P values.

When comparing predicted recharge values with observed baseflow, the Logan River predicted recharge within 21% on average. Predicted and observed data were correlated with an $R^2 = 0.50$ (Figure B-6). Red Butte Creek predicted recharge within 54% on average and had a correlation of $R^2 = 0.63$ (Figure B-6).

TABLE 3-4 Coefficients for the empirical relationships are given below. Coefficients are fairly similar between watersheds with the exception of 'X_2' and 'X_3'.

Coefficient:	LR	RBC
X_1	0.01004	0.00998
X_2	0.00862	0.01005
X_3	0.01146	0.00183
X_4	0.01001	0.00992

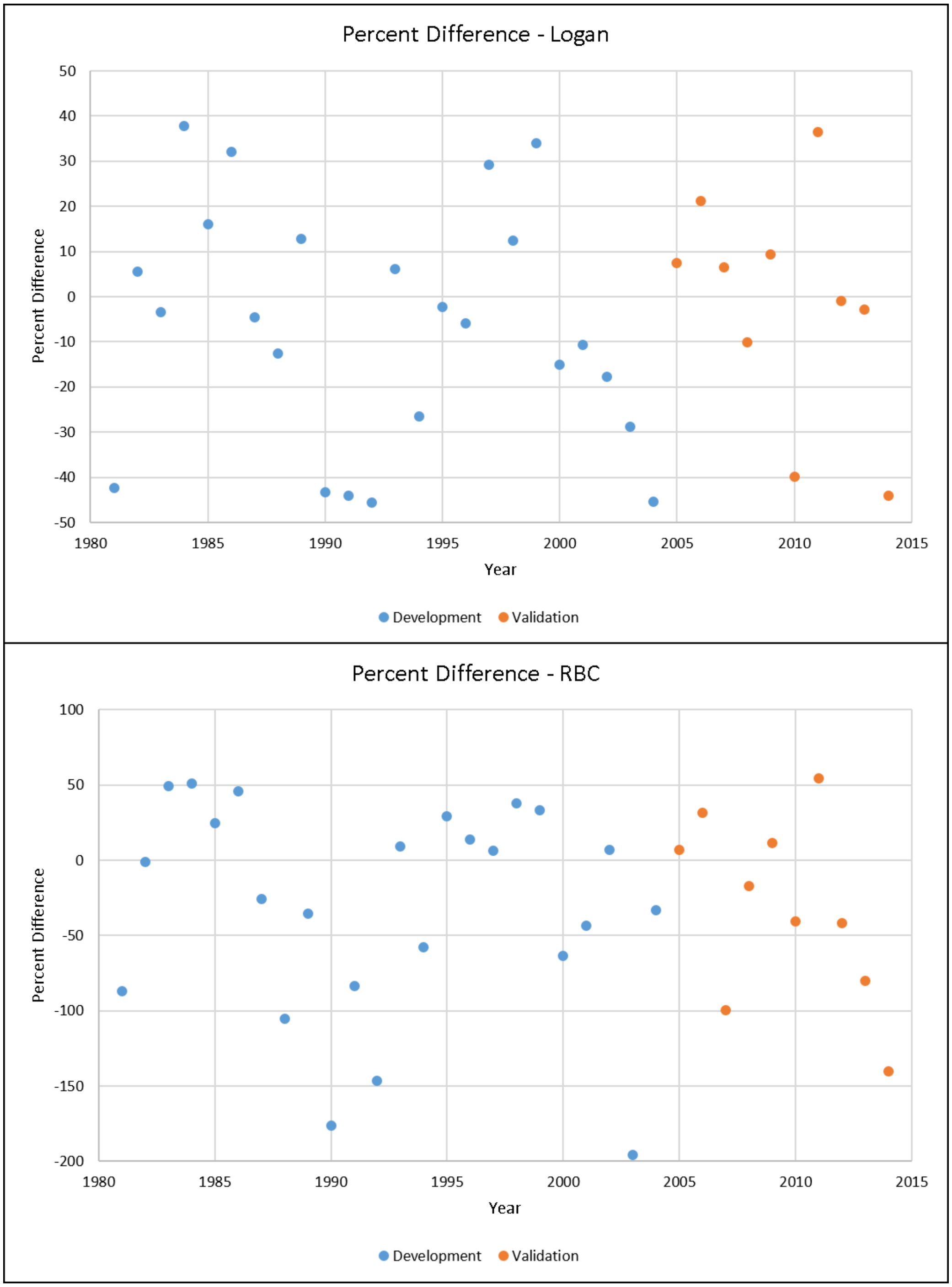

FIGURE 3-6 Percent differences between observed and predicted R/P values for the Logan River and Red Butte Creek watersheds. A perfect prediction would have a percent difference of zero, indicated by the black horizontal line. Results from the development and validation datasets are shown in blue and orange, respectively. The range of error for the Logan River is from about positive 50% to negative 50%. The range of error for Red Butte Creek is from about positive 50% to negative 200%. Negative values indicate an overestimate of R/P.

TABLE 3-5 The mean and standard deviation of the percent differences in observed and predicted R/P values are shown for the development and validation datasets for both the Logan River and Red Butte Creek. The Logan River relationship held more true to the observed data than the Red Butte Relationship.

Data Set	% Difference	Logan	RBC
Development Set	Mean	-6.8	-31.1
	St.Dev	26.5	71.4
Validation Set	Mean	-1.7	-31.5
	St.Dev	25.0	61.4

Application of Empirical Relationships to Different Watersheds

Even though both relationships were very similar and predicted recharge within their respective watersheds with a fair degree of success, applying the relationship of one watershed to the other did not produce good results. When the LR relationship was applied in the RBC watershed, R/P predicted values were 700% higher on average than observed values with a standard deviation of 413%. The RBC relationship under predicted R/P values by 300% on average with a standard deviation of 51%.

Discussion

Due to data limitations in western watersheds, developing estimates of groundwater recharge remains difficult and inherently inaccurate. Previous work (Cherkauer and Ansari, 2005) successfully developed a relationship that estimated groundwater recharge rates based on surface characteristics. We applied their simple relationship to the Logan River and Red Butte Creek watersheds to estimate the portion of precipitation that enters mountain aquifers. Results from the developed relationships varied between each other and the original study.

Data Acquisition and Analysis

While there are obvious differences between R/P results from the Logan River and Red Butte Creek (Table 3-2), it is worth first comparing results with those obtained during the original study. Cherkauer and Ansari (2005) used information from 12 watersheds over the span of two years and found an average R/P value of 0.123 with a range of 0.050 – 0.251. The average percent difference between observed and predicted data (using the absolute value of the differences) was 15% and with an overall range from +31% to -26.5%. Overall, the Logan River has much higher average R/P values than the original study, but has the most comparable results as far as the magnitude and range of the percent differences is concerned. Red Butte Creek has a similar range of R/P values, but vastly different results when comparing the overall percent differences. Cherkauer and Ansari explain there is a 37% expected error using this method from the propagation of error due to uncertainty in the datasets. Even if we assume that potential systematic errors associated with applying their method to entirely different physiographic and climatic regions are minimal, the ranges of both the Logan River and Red Butte Creek extend beyond the explainable error of 37% (from dataset variability), with Red Butte Creek being significantly worse.

There are also large differences in watershed characteristics (that were included in the empirical relationship) between the watersheds in the original study and the Logan River and Red Butte Creek watersheds. The original study investigated relatively small watersheds (2.9 – 48.7 km^2) with low slopes (0.0095 – 0.082) and a low percentage of natural land cover (8.0% – 26.9%). Hydraulic conductivities ranged from 0.33 – 2.7 m/d and the depth to the water table was relatively shallow (9.1 – 32.6 m). The Logan River had comparable hydraulic conductivity

(0.91 m/d) but had much greater area (558.5 km^2), steeper slopes (0.34), higher percentage of natural cover (98.8%), and greater depths to groundwater (202 m) (Table 3-3). Red Butte Creek was only similar in watershed area (18.8 km^2), but had steeper slopes (0.51), a higher percentage of natural cover (100%), lower hydraulic conductivity (0.10), and lower estimated depth to groundwater (0.61 m) (Table 3-4).

Aside from difference in results between this study and the original study, the Logan River and Red Butte Creek watersheds also produced very different estimates when compared to each other. While they are similar in proximity, and precipitation (Figure B-3), there were other characteristics that made them difficult to compare. Both watersheds feature a mixture of limestone, dolomite, quartzite and sandstone. However, the Logan River watershed has prominent karst features including sinks and macropores that influence groundwater and surface water hydrology. Karst hydrology poses an increased complexity in trying to estimate recharge to aquifers. Because of the complexity, Ahiablame et al. (2013) simply removed karst watersheds from their analysis in developing estimation of annual baseflow at ungaged sites. However, our study was not focused on understanding karst hydrology, but on determining the applicability of the empirical relationship to two different watersheds and the different hydrological characteristics within.

Another striking difference between the watersheds is the overall size. The Logan River watershed is several hundred square kilometers larger, and even has sub-watersheds larger than Red Butte Creek. The complexity of larger systems may mute some of the more pronounced processes that govern groundwater and surface water hydrology in smaller catchments. For example, the Red Butte Creek watershed drains practically due west with a single steep north-facing slope. During spring runoff, there is not a significant amount of snow that remains in the watershed once snowmelt begins due to the lack of storage on north-facing slopes. In contrast, the

Logan River watershed has multiple subwatersheds with numerous north-facing slopes that store snow until late in the season and attenuate spring runoff. In developing an understanding of the processes that truly govern the partitioning of groundwater and surface water in mountain watersheds, it may be worth investigating multiple clustered drainages.

While the distinct differences in spatial characteristics undoubtedly contribute to the variability seen in the distribution of observed and predicted R/P values and the average percent differences, it is more likely that a significant amount of the variability comes from many of the simplifying assumptions made in the original study that could not be justified in the Logan River and Red Butte Creek watersheds. These assumptions include statements concerning using baseflow separation as an estimate of recharge, the assumptions contained within the original conceptual model, including measurements of precipitation and recharge areas, and the baseflow separation method itself.

When equating baseflow with recharge (Equations 1 and 2), it is assumed that groundwater and surface water divides coincide, that there is no human transport of water into or out of the watershed, and that groundwater storages do not significantly change year to year. The model also suggests interflow to be a negligible component of streamflow and assumes that the hydrograph can be separated into direct surface runoff and groundwater discharge (Kulandaiswamy and Seetharaman, 1969). A few of these assumptions may be justifiable in the Logan River and Red Butte Creek watershed. However, studies in the Logan River have long confirmed that groundwater enters the Logan River basin from outside delineated watershed boundaries (Spangler, 2001).

The simplified conceptual model developed in the original study assumed precipitation occurred uniformly across the watershed, the entire drainage area was conceived as a recharge area, and that groundwater discharge occurs only at the stream. Each of these three assumptions fall apart quickly in western, mountainous watersheds due to the variability introduced by topography, geology, and GW flowpaths. For example, precipitation is more difficult to measure in mountain watersheds (Wilson and Guan, 2004) and can vary dramatically over space due to quick changes in topography (Bales et al., 2006). Both the Logan River and Red Butte Creek watersheds have recently been instrumented with several new climate stations, but did not have a sufficient number of years of data to be used in this study. The NRCS SNOTEL sites provide adequate temporal data; however, they are located in locations that only represent precipitation estimates for higher altitudes. The only dataset that provided the necessary temporal and spatial data was the PRISM dataset. Even using the best estimate of precipitation available, values were still averaged over the entire watershed. Further, the spatial variability in precipitation was lost, which would not have mattered if the entire watershed area could be considered a recharge area.

Unfortunately, recharge in semi-arid regions of the west occurs differently than in humid areas like the original study. Gee and Hillel (1988) explain that there are two kinds of recharge: continuous, spatially distributed (diffuse) recharge across the entire vadose zone, and transient (occasional), concentrated recharge through distinct pathways. Either type can become dominant based on precipitation distribution, soil type, plant cover, etc. Continuous recharge tends to decrease in the semi-arid regions of the west as potential evapotranspiration begins to exceed precipitation and infiltration rates. Mountain regions in the west are likely a combination of both diffuse and concentrated regions. Much of the precipitation comes in the form of snow (Mast and Clow, 2000; Spangler, 2001), and during snowmelt, infiltration exceeds potential evapotranspiration. Both the Logan River and Red Butte Creek geology have documented faults (Mast and Clow, 2000; Spangler, 2001), and the Logan River has significant karst features and macropores (Spangler, 2001) that act as pathways for concentrated recharge. It is clear that the entire watersheds cannot be treated as recharge zones, which nullifies the assumption that an average precipitation value can be used.

The third assumption made in the original conceptual model is that all groundwater discharge occurs at the stream. However, it is known that much of the groundwater that enters the mountain block continues downward, eventually entering basin-fill aquifers as MBR (Wilson and Guan, 2004). Using stable isotope data, research on the basin fill aquifer near Red Butte Creek distinguished between sources of MFR and MBR (Manning and Solomon, 2003). Similar conditions have been reported across the several mountain watersheds and adjacent basin-fill aquifers (Anderson et al., 1994; Feth et al., 1966; Wilson and Guan, 2004).

Another source of variability and uncertainty encountered in this work was the method of baseflow separation used to estimate recharge rates. Other methods used for estimating groundwater recharge could not be used for this study due to a lack of data (water balance, geochemical tracers) or inappropriate scale (well hydrograph analysis). While there is some concern that there are problems associated with recharge rates from stream discharge records (Halford and Mayer, 2000), it has been reported to be an appropriate method for mountain systems (Chen and Lee, 2003), and work is continuously being done to improve results (Ajami et al., 2011; Miller et al., 2015). One of the concerns with baseflow separation is that variability in bank storage discharge to the river as a result of short term surface flow fluctuations could lead to an over-estimation of recharge (Scanlon et al., 2002). To mitigate the effects of short-term variability, baseflow separation was performed over an entire calendar year. In estimating recharge, it is recommended to use multiple methods to decrease uncertainty. Unfortunately, due to the lack of data, only baseflow separation using recursive digital filters could be applied to both the Logan River and Red Butte Creek watersheds.

With so many simplifying assumptions not being met, it is clear that the selected method of estimating groundwater was not representative of the watershed characteristics and that another method should have been chosen. Unfortunately, a general lack of reliable or consistent geospatial data in the mountain regions limited the ability to use other empirical relationships and introduced greater variability to the method we used. While many geospatial datasets have been made available, there were gaps in many of the coverages needed for this study, including soil and depth to water table coverages. Most notable were differences in the Depth to Water Table coverages. Red Butte Creek had sparse coverage near the stream from SSURGO data and estimated water table depth at 0.61 meters. The manually developed layer used for the Logan River estimated an average water table depth of 202 meters (Table 3-3). This drastic difference in depth to groundwater is clearly an error, resulting from having to estimate values from incomplete data. While the depth to groundwater may be less than a meter near the stream in the Red Butte drainage, it is not representative of the watershed average. The original study documents access to more complete, readily available datasets, and did not have to reconcile such differences.

Relationship Between Recharge, Precipitation and Spatial Characteristics

This study tried to apply the relationship developed in the original study individually and independently to two separate mountain watersheds. Even though coefficients were selected that made the relationships reproduce distributions of predicted R/P similar to the observed R/P distributions, it became clear that in order for the method from the original study to be more successful, it should be applied to a cluster of watersheds. In doing so, some of the noise and nuances of smaller watersheds may be muted and a more representative empirical relationship can likely be developed. While some variability in the empirical relationship can be explained by variability in the datasets and differences in spatial characteristics, a large amount of uncertainty is introduced by not meeting the required simplifying assumptions and the general lack of quality geospatial data. However, when comparing the results from the two different watersheds, it became evident that the established relationship developed by Cherkauer and Ansari (2005) simply did not adequately describe the variability of key hydrologic processes within mountain watersheds.

As mentioned previously, the original study used two years of data and multiple watersheds to develop their relationships. This study only accounted for variability introduced by 35 years of precipitation and recharge estimates, however, they were simply reduced to a single average R/P value for each watershed. With a single dataset and a single average value, an infinite number of solutions could theoretically be developed to connect those two points. This was observed during the regression analysis when multiple solutions were determined. Also, by applying the approach to one watershed at a time, it negates the importance of the spatial characteristics being analyzed. Equation 3 simply reduces to R/P = constant, when there is no variability in spatial characteristics that is accounted for. Solving for recharge we achieve a relationship in the form of R = P*constant (in the form of the slope-intercept where y = mx). This would suggest that annual recharge volumes are directly related to precipitation. While some studies have previously estimated recharge by simply multiplying precipitation value by a 'rule-of-thumb' fraction, it has been shown that this approach can be deceptive and misleading since it completely disregards other processes (Gee and Hillel, 1988). While data from these study areas show a correlation between precipitation and recharge (Figure B-4), the correlation remains relatively weak (R^2 values of 0.58 and 0.66 for the Logan River and Red Butte Creek, respectively), and the data show that there is more to determining recharge rates. By using spatial characteristics for several clustered watersheds to determine the relationship between R/P, it is possible for parameters of the greatest importance could begin to be identified. Using multiple watersheds would also eliminate the chance of multiple solutions being achieved and remove some of the false successes achieved when the relationship is applied to individual watersheds.

Aside from just using clustered watersheds to improve the empirical relationship, other parameters should be considered. Many other studies that have developed empirical relationships have included parameters such as evapotranspiration (Ahiablame et al., 2013), growing degree days (Lorenz and Delin, 2007), annual freeze free days, relative humidity, and wetness index (Zhu and Day, 2009) along with the parameters used by Cherkauer and Ansari (2005). Each of these studies were also conducted using multiple watersheds. Lorenz and Delin (2007) suggest using decadal averages of precipitation, average recharge, and growing degree days to develop empirical relationships for R/P. Interestingly, each of the studies were also conducted in the humid and temperate watersheds in Wisconsin (Cherkauer and Ansari, 2005), Pennsylvania (Zhu and Day, 2009), Minnesota (Lorenz and Delin, 2007), and Indiana (Ahiablame et al., 2013). Unfortunately, due to the lack of geospatial data in mountain areas, the application of any of these other methods mentioned was limited.

As it stands, a successful empirical relationship has yet to be developed to estimate groundwater recharge rates based on surface characteristics and other meteorological data for western, mountainous watersheds where geospatial data are limited. Such an endeavor will be difficult for numerous reasons. First of all, basic assumptions in using baseflow as an estimate for recharge are not justified in mountain watersheds with carbonate geology and karst features or where adjacent basin-fill aquifers receive significant MBR. Other characteristics that influence the amount of recharge to mountain aquifers include better estimates of precipitation, temperature and evapotranspiration, each of which are highly variable with topography (Bales et al., 2006). A representation of the geology, in the form of faults, karst features, and other spatially variable hydraulic properties of the bedrock to better estimate concentrated recharge zones are also needed

(Guan, 2005). However, even if the parameters discussed were successfully represented, it is likely that an empirical relationship relating baseflow as recharge with spatial characteristics may still remain meaningless if the basic simplifying assumptions are not met.

Conclusion

Empirical relationships for estimating R/P ratios were developed for two mountain watersheds based on methods outlined in previous work done in Midwestern watersheds. These two western, mountain watersheds do not meet the most basic of simplifying assumptions used in the previous study. However, the relationship was still applied to determine the applicability of the method and to determine if the parameters identified still remain the driving force in partitioning precipitation into groundwater recharge and surface runoff. Applying the method led to average observed R/P values of 0.37 for the Logan River watershed and 0.19 for the Red Butte Creek. Because the method was applied independently to two individual watersheds, the regression analyses returned multiple solutions. Once bootstrapping was applied, a representative empirical relationship was determined for each watershed. However, it is difficult to relate dynamic temporal data with static spatial information. Because the relationships were applied to single watersheds, while there was variability in R/P values, there was no variability in spatial characteristics and the relationship became linearized, losing any representation of actual processes. Any successes in predicting R/P values using the developed relationships were simply an artifact of the correlation between recharge and precipitation. Understanding that there is more than just precipitation that determines recharge volumes, there is a clear need to consider other parameters, including temperature, evapotranspiration, and geologic features. However, the lack of sufficient geospatial data in mountain regions limits the type of approach that can be applied to estimate groundwater recharge. Even if sufficient data were collected and adequate spatial variability represented, new relationships would need to be developed to better represent driving processes in western, mountain watersheds that incorporate the appropriate assumptions.

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CHAPTER 4

GENERAL CONCLUSION

The development and application of models is essential for providing information to managers as part of the decision-making processes. This led to the application of a hydraulic model to understand stream restoration implications and an empirically derived mathematical model to predict annual groundwater recharge volumes.

The first modeling effort focused on quantifying the impacts of beaver dams on channel hydraulics and substrate characteristics through the development and application of a 1D hydraulic model and analysis of longitudinal substrate data. Results from the model indicate that the introduction of beaver dams causes significant change in the distributions of the depths, widths, and velocities while increasing spatial heterogeneity. Results from the substrate data indicate that through beaver dam construction, maintenance and failure, sediment is stored and reworked, introducing greater variability in substrate and disrupting general trends. A relatively low number of dams were found to cause significant changes in these hydraulic parameters. Overall, this study demonstrates the potential influence of beaver dams in meeting restoration goals of increased habitat availability through increased hydraulic variability and provides an estimate on the best use of resources in meeting these goals for mountain streams.

The second modeling approach applied empirical relationships for estimating R/P ratios to two mountain watersheds based on methods outlined in previous work done in Midwestern watersheds. These two western, mountain watersheds do not meet the simplifying assumptions used in the previous study. However, the relationship was still applied to determine the potential applicability of the method and if the parameters identified still remain the driving force in partitioning precipitation into groundwater recharge and surface runoff. The method successfully developed estimates of R/P ratios and even established coefficients for the empirical relationships, however, the results provided little insight into system behavior. Because the approach was applied independently to each watershed, there was no variability in the spatial dataset, which led to a linearized relationship between precipitation and recharge. A general lack of geospatial data in mountain regions limits the type of approach that can be used in estimating groundwater recharge. The difficulty in relating dynamic temporal data with static spatial data, insufficient data types, and the inability to meet simplifying assumptions limit the applicability of the existing empirical relationships to mountain watersheds. To make the approach of using empirical relationships to estimate recharge applicable, better geospatial data must be collected, data from multiple watersheds must be used, other parameters must be considered, and different simplifying assumptions must be developed.

Results from both of these studies can support decisions regarding stream restoration projects or in interpreting data regarding available groundwater and groundwater recharge rates. First and foremost, if there are sufficient, quality data to drive modeling efforts, meaningful guidance can be provided to better inform management decisions. However, the second modeling study highlights that models can provide reasonable answers without any connection to processes. While some researchers acknowledge and encourage the idea of equifinality to be embraced in modeling (Beven, 2006), managers should first develop an understanding of the assumptions, limitations, and uncertainty associated with each model. The mere successful generation of numbers through descriptive or predictive modeling does not indicate meaningful or applicable results without proper corroboration (Reckhow and Chapra, 1983). With the inherent uncertainty that exists in natural systems, combined with the variability of social, political, and economic factors, it is essential that managers be given accurate and applicable information regarding the processes and potential outcomes. The needs of an ever growing population must be met while still protecting the quality and quantity of water and the environment. Through the development of representative hydraulic, hydrologic, and hydrogeologic models, simulations can provide a means to better inform managers of the implications of decisions.

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CHAPTER 5

ENGINEERING SIGNIFICANCE

While both chapters demonstrate the development and application of models with the intent to better inform water managers, both have distinct engineering significance regarding the information they provide.

The work on determining the impact of beaver dams on channel hydraulics has clear stream restoration implications and a paper describing the results has been published by the journal Ecohydrology (DOI 10.1002/eco.1767). Recent studies have shown that beaver dams have proven to be a cost effective method for increasing variability in channel hydraulics and spatial heterogeneity in substrate distributions (Wheaton et al., 2012). Our model results indicate statistically significant differences in the distribution of hydraulic characteristics, and, therefore, potential difference in habitat, between a reach with or without beaver dams. Understanding that beaver have a positive impact on stream systems is a good start, yet resource managers need to be able to maximize the restoration efforts while minimizing cost and labor. By modeling three different approaches to beaver dam complex evolution, beaver dam densities can be established that would best meet restoration goals. This approach would be applicable to mountain streams and, once the validity of predictions have been tested, would allow resource managers to adequately improve the systems while using minimal resources.

This work contributes to advancing our understanding of the complexity of groundwater recharge in western regions where water resources are in high demand and highlights the need for the development and application of better approaches to quantify groundwater recharge rates in mountain regions. While work regarding the application of an empirical relationship to estimate groundwater recharge from precipitation and spatial characteristics could benefit groundwater resource managers in Midwestern regions, we show that the existing method is not suitable for estimating groundwater recharge in western, mountain regions. This project emphasizes the high level of variability, even between watersheds of regional proximity and highlights the need for more complete geospatial datasets in western, mountain regions. In order to develop an approach with appropriate parameters and assumptions, there is a need for comprehensive analyses of clustered watersheds to develop a more concrete understanding of the processes that most significantly govern groundwater recharge rates in mountain block regions.

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APPENDICES

Appendix A: Supplemental Tables and Figures for Chapter 2

TABLE A-1 Range of sediment size distributions for pools, bars and riffles in the lower and upper reaches.

		D16 RANGE		D50 RANGE		D84 RANGE	
		MIN	MAX	MIN	MAX	MIN	MAX
LOWER	POOL	0.088	5.31	0.334	32	1.172	128
	BAR	2.57	14.36	10.07	31.75	15.57	55.45
	RIFFLE	7.26	19.49	16	48.81	27.36	104.05
UPPER	POOL	0.3	10.55	5.53	35.45	13.59	98.93
	BAR	15.06	18.96	37.95	52.77	62.47	93.51
	RIFFLE	6.6	17.14	40.45	56.7	73.35	110.79

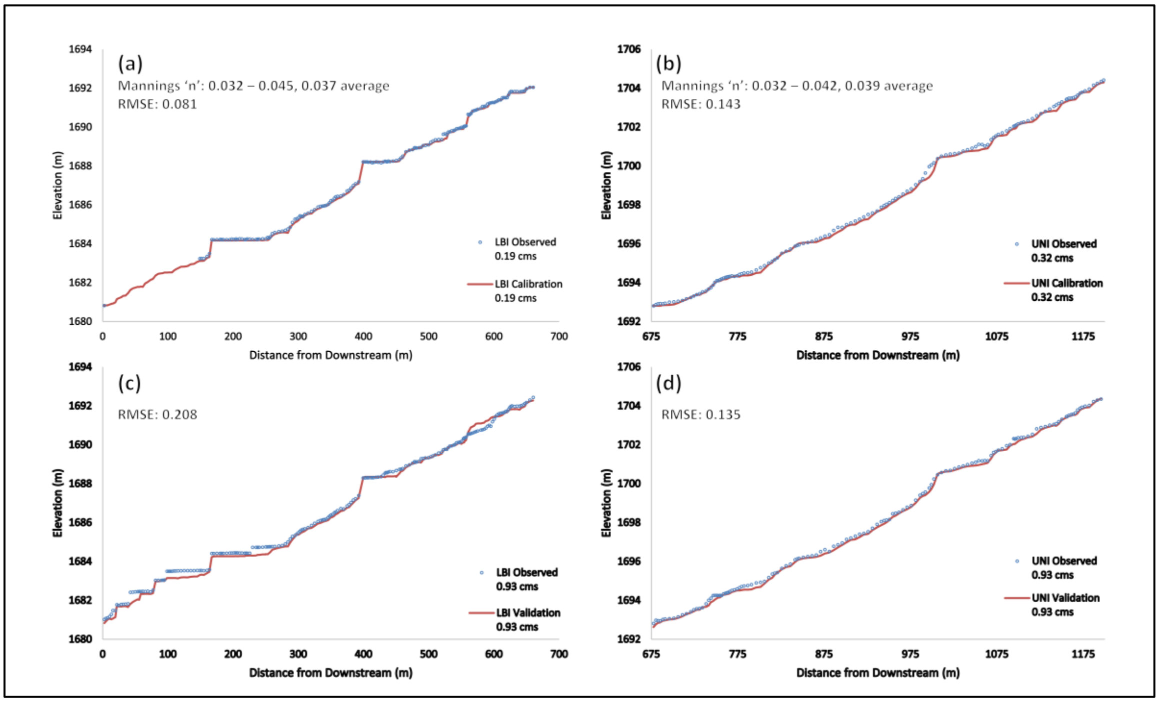

FIGURE A-1 Calibration and validation results for the study reach. The calibration data set for the lower, beaver impacted (LBI) reach (a) is compared with the non-impacted (UNI) reach (b) with the range of Manning's 'n' values used during model calibration and the root mean square error as a measure of model fit. The validation data set for the lower, beaver impacted (LBI) reach (c) and the upper, non-impacted (UNI) reach are also shown (d). Blue represents surveyed WSEL data and red is values produced by the model. The RMSE for the lower, beaver impacted reach is much higher due to increased WSEL in certain areas due to beaver activity between 2012 and 2014. Sections where beaver influence had not changed water surface elevations matched observed data well. Ranges and averages of roughness values used in the calibration for each reach are given.

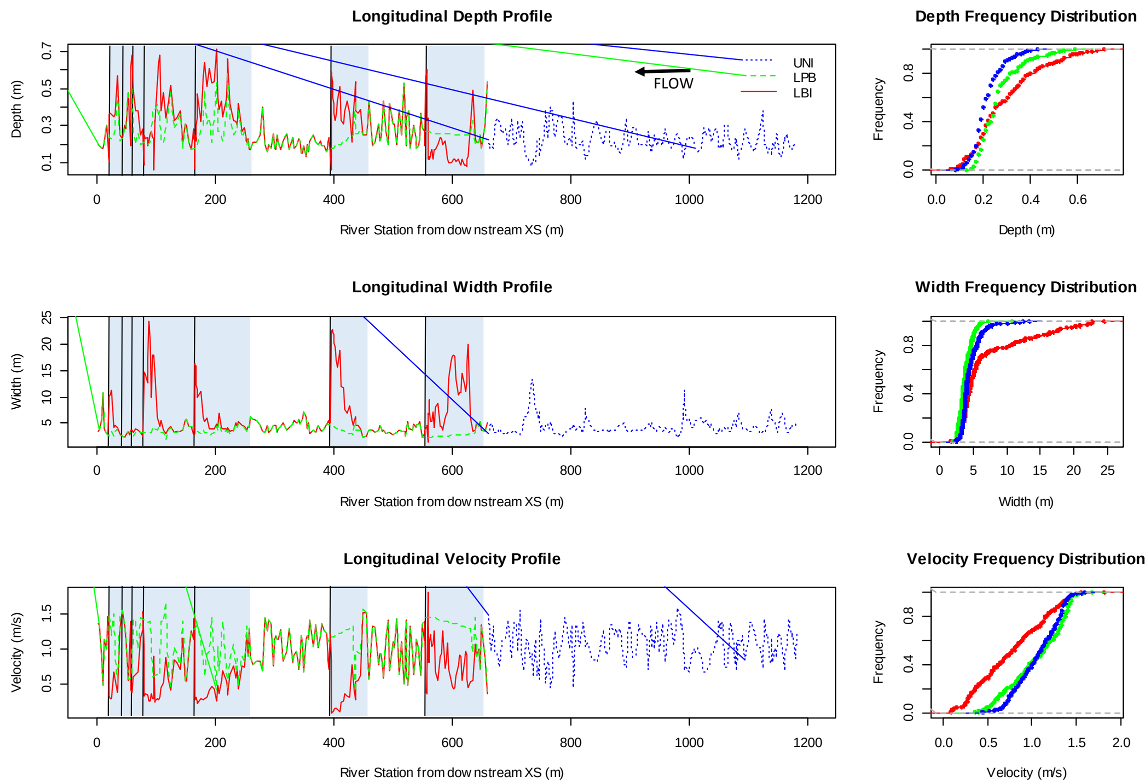

FIGURE A-2 Depth, width, velocity values are plotted longitudinally along with the respective cumulative distribution curves for the 0.93 m^3/s simulation. The upper, non-impacted (UNI) data are shown in blue, lower, beaver impacted (LBI) data are shown in red, and lower, pre-beaver data (LPB) are shown in green color. Beaver dam locations are indicated by solid black vertical lines and the backwater effect is shown in shaded light blue. Flow is from right to left, with river station 0 as the most downstream location in the model. Through much of the lower reach, LBI and LPB values are the same. Difference can be seen where there is a change due to beaver dams.

Appendix B: Supplemental Tables and Figures for Chapter 3

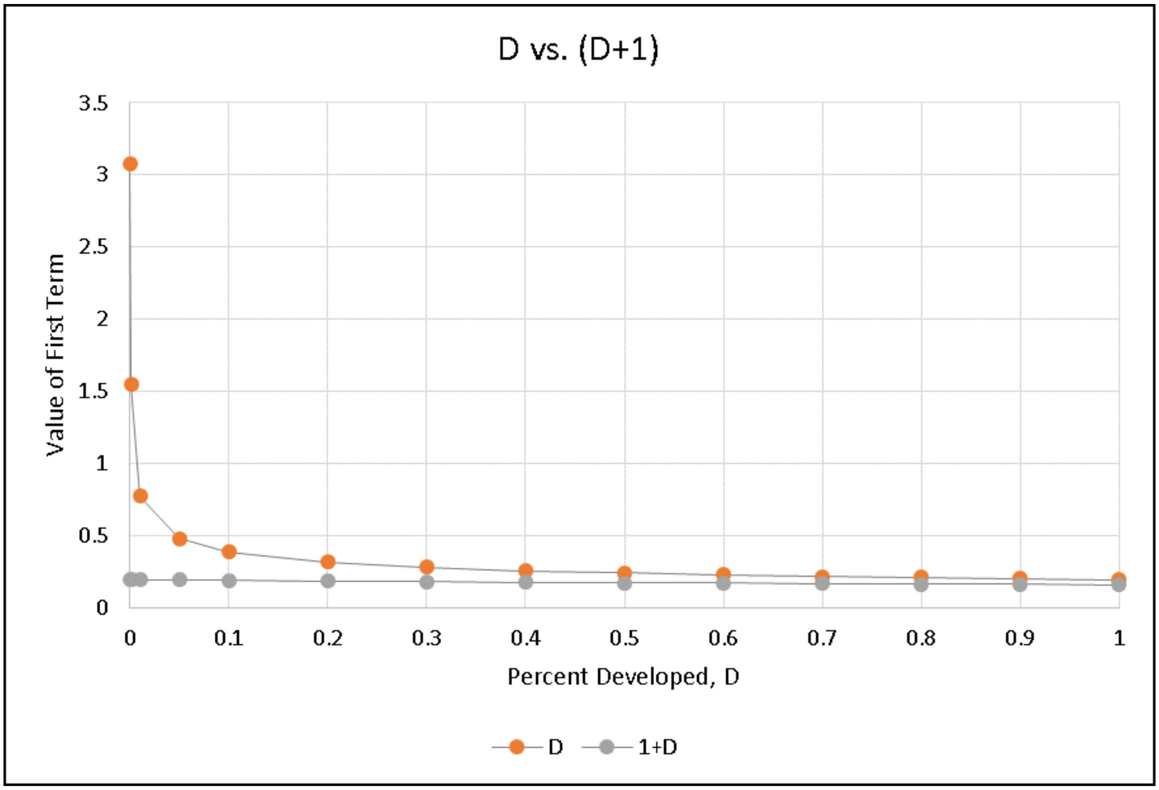

FIGURE B-1 Justification for adding one to the D term in the empirical relationship. As the percent of developed land drops below 1%, the value of the first term in the relationship begins to increase asymptotically.

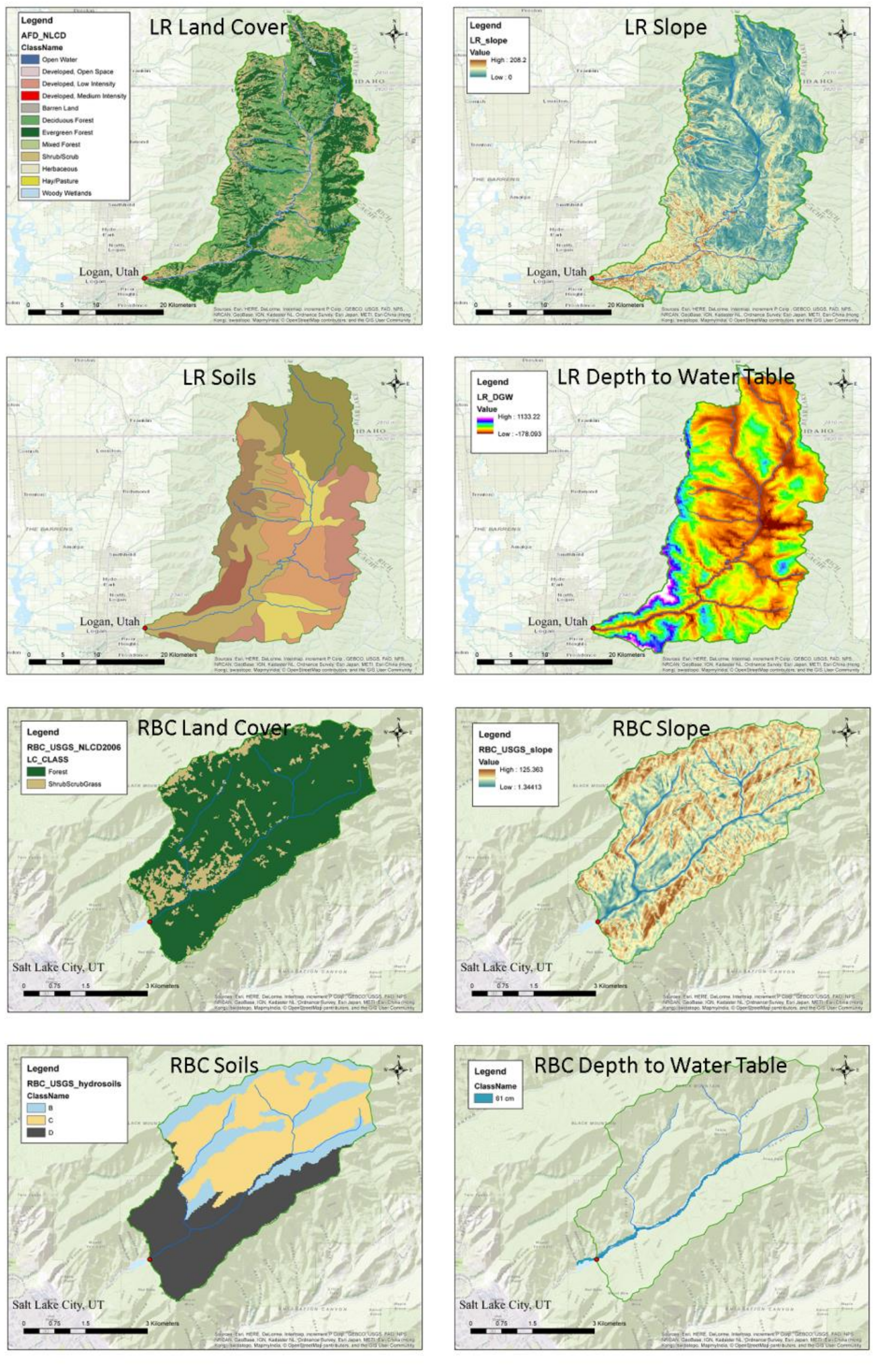

FIGURE B-2 ArcGIS Coverages used in the analysis. Land cover data was retrieved from the 2006 National Land Cover Dataset. Slopes were calculated from the National Elevation Dataset. Soil data were retrieved either as a shapefile (LR) from STATSGO data or a raster (RBC) from SSURGO data. Depth to water table was estimated using a raster developed from well and stream data (LR) or from SSURGO data (RBC).

TABLE B-1A Annual Flow, Precipitation, and R/P values for the Logan River

Year	Total Flow Depth (m)	Direct Runoff Depth (m)	Baseflow Depth (m)	Annual Total Precip (m)	R/P
1981	0.29	0.06	0.23	0.92	0.26
1982	0.61	0.13	0.48	1.24	0.39
1983	0.63	0.13	0.50	1.41	0.35
1984	0.73	0.15	0.58	0.99	0.59
1985	0.48	0.10	0.38	0.88	0.43
1986	0.78	0.18	0.61	1.13	0.54
1987	0.29	0.06	0.23	0.67	0.35
1988	0.27	0.06	0.22	0.68	0.32
1989	0.39	0.08	0.31	0.73	0.42
1990	0.24	0.05	0.19	0.75	0.25
1991	0.28	0.06	0.22	0.87	0.25
1992	0.20	0.04	0.16	0.65	0.25
1993	0.47	0.10	0.37	0.96	0.39
1994	0.28	0.06	0.22	0.77	0.29
1995	0.49	0.10	0.38	1.08	0.36
1996	0.51	0.11	0.40	1.15	0.34
1997	0.64	0.13	0.50	0.97	0.52
1998	0.55	0.11	0.44	1.05	0.42
1999	0.59	0.12	0.47	0.84	0.55
2000	0.32	0.06	0.26	0.82	0.32
2001	0.27	0.05	0.21	0.65	0.33
2002	0.27	0.05	0.22	0.70	0.31
2003	0.29	0.06	0.23	0.80	0.28
2004	0.28	0.06	0.22	0.88	0.25
2005	0.50	0.10	0.39	0.99	0.39
2006	0.56	0.12	0.44	0.95	0.46
2007	0.31	0.06	0.25	0.63	0.39
2008	0.36	0.07	0.28	0.86	0.33
2009	0.45	0.09	0.35	0.88	0.40
2010	0.34	0.07	0.27	1.03	0.26
2011	0.74	0.16	0.58	1.01	0.57
2012	0.33	0.06	0.27	0.74	0.36
2013	0.25	0.05	0.20	0.56	0.36
2014	0.34	0.07	0.27	1.07	0.25

TABLE B-1B Annual Flow, Precipitation, and R/P values for Red Butte Creek

Year	Total Flow (m)	Total Direct Runoff (m)	Total Base Flow (m)	Total precip (m)	R/P
1981	0.11	0.03	0.09	0.85	0.10
1982	0.26	0.06	0.20	1.07	0.19
1983	0.60	0.15	0.45	1.20	0.37
1984	0.46	0.11	0.34	0.89	0.38
1985	0.23	0.05	0.18	0.73	0.25
1986	0.42	0.09	0.33	0.95	0.35
1987	0.10	0.02	0.08	0.56	0.15
1988	0.06	0.01	0.05	0.55	0.09
1989	0.09	0.02	0.07	0.53	0.14
1990	0.05	0.01	0.04	0.59	0.07
1991	0.10	0.02	0.08	0.78	0.10
1992	0.06	0.01	0.05	0.59	0.08
1993	0.24	0.05	0.18	0.88	0.21
1994	0.11	0.02	0.09	0.75	0.12
1995	0.30	0.06	0.24	0.88	0.27
1996	0.25	0.05	0.19	0.88	0.22
1997	0.20	0.04	0.16	0.80	0.20
1998	0.35	0.08	0.28	0.91	0.30
1999	0.25	0.05	0.20	0.71	0.28
2000	0.10	0.02	0.08	0.71	0.12
2001	0.11	0.02	0.09	0.65	0.13
2002	0.13	0.03	0.10	0.51	0.20
2003	0.06	0.01	0.04	0.69	0.06
2004	0.13	0.03	0.10	0.71	0.14
2005	0.22	0.05	0.17	0.85	0.20
2006	0.31	0.07	0.24	0.86	0.28
2007	0.07	0.01	0.06	0.62	0.09
2008	0.14	0.03	0.11	0.66	0.16
2009	0.22	0.05	0.17	0.80	0.21
2010	0.15	0.03	0.12	0.89	0.13
2011	0.47	0.11	0.36	0.86	0.42
2012	0.10	0.02	0.08	0.58	0.13
2013	0.08	0.02	0.06	0.59	0.11
2014	0.07	0.01	0.06	0.72	0.08

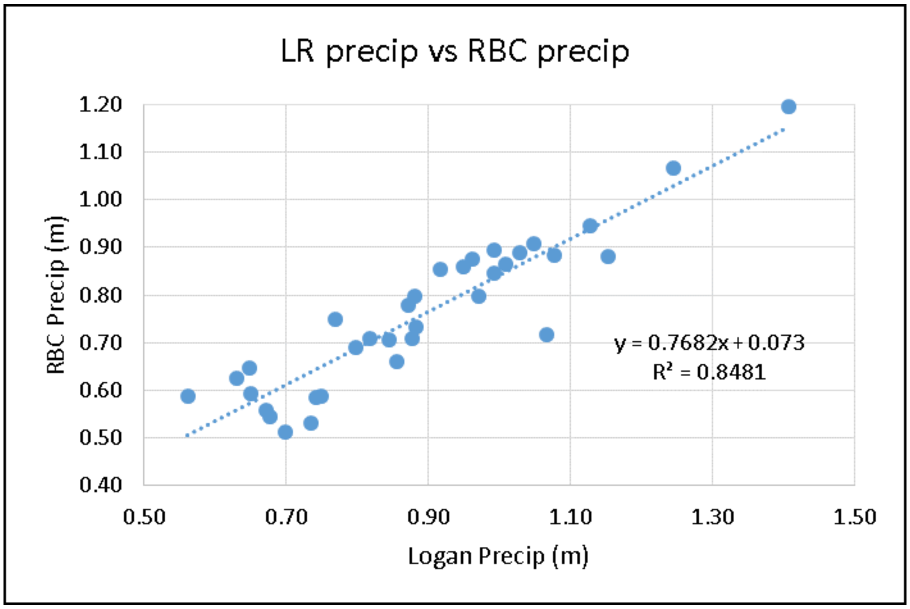

FIGURE B-3 Comparison of precipitation depths between the two study watersheds. While the LR watershed does receive slightly more precipitation, the precipitation amounts between the two were strongly correlated.

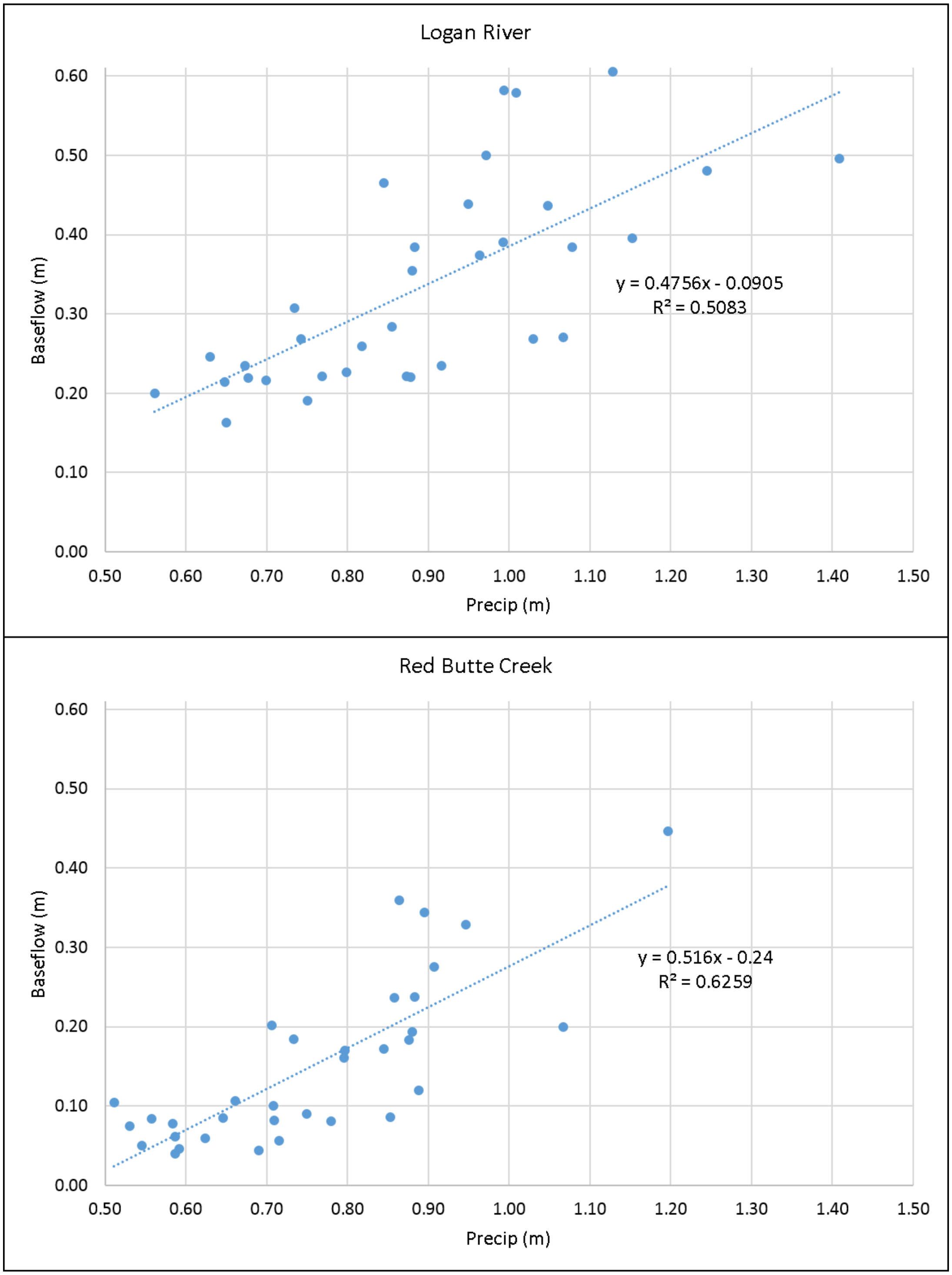

FIGURE B-4 Baseflow vs Precipitation for the Logan River and Red Butte Creek. While there is a correlation between precipitation and baseflow values, the obvious noise in the data indicate that other factors besides just precipitation dictate the amount of recharge that enters mountain aquifers.

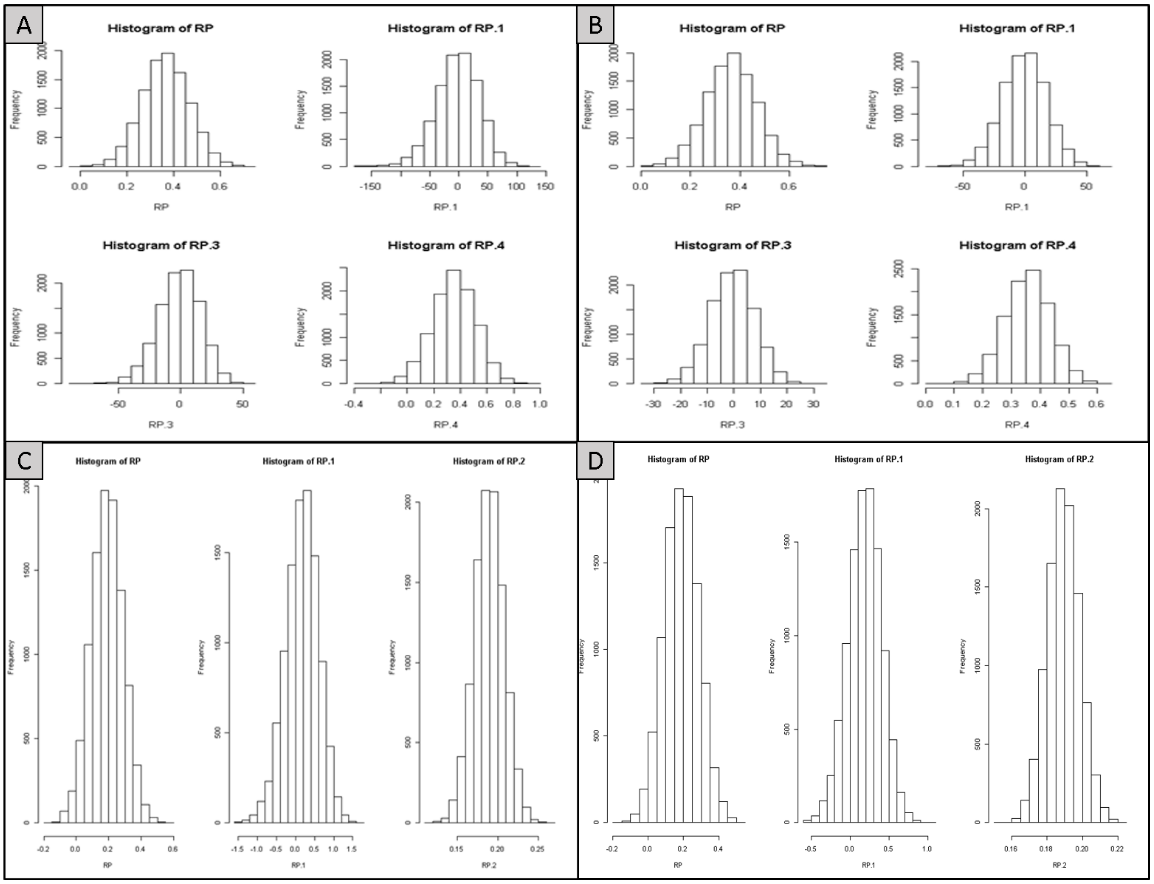

FIGURE B-5 Results from the bootstrapping for the Logan River relationship (A and B) and the Red Butte Creek relationship (C and D). Because there was no variability in the spatial datasets, standard deviations were estimated by multiplying the average value for each parameter by 0.1 (A and C) and 0.05 (B and D).

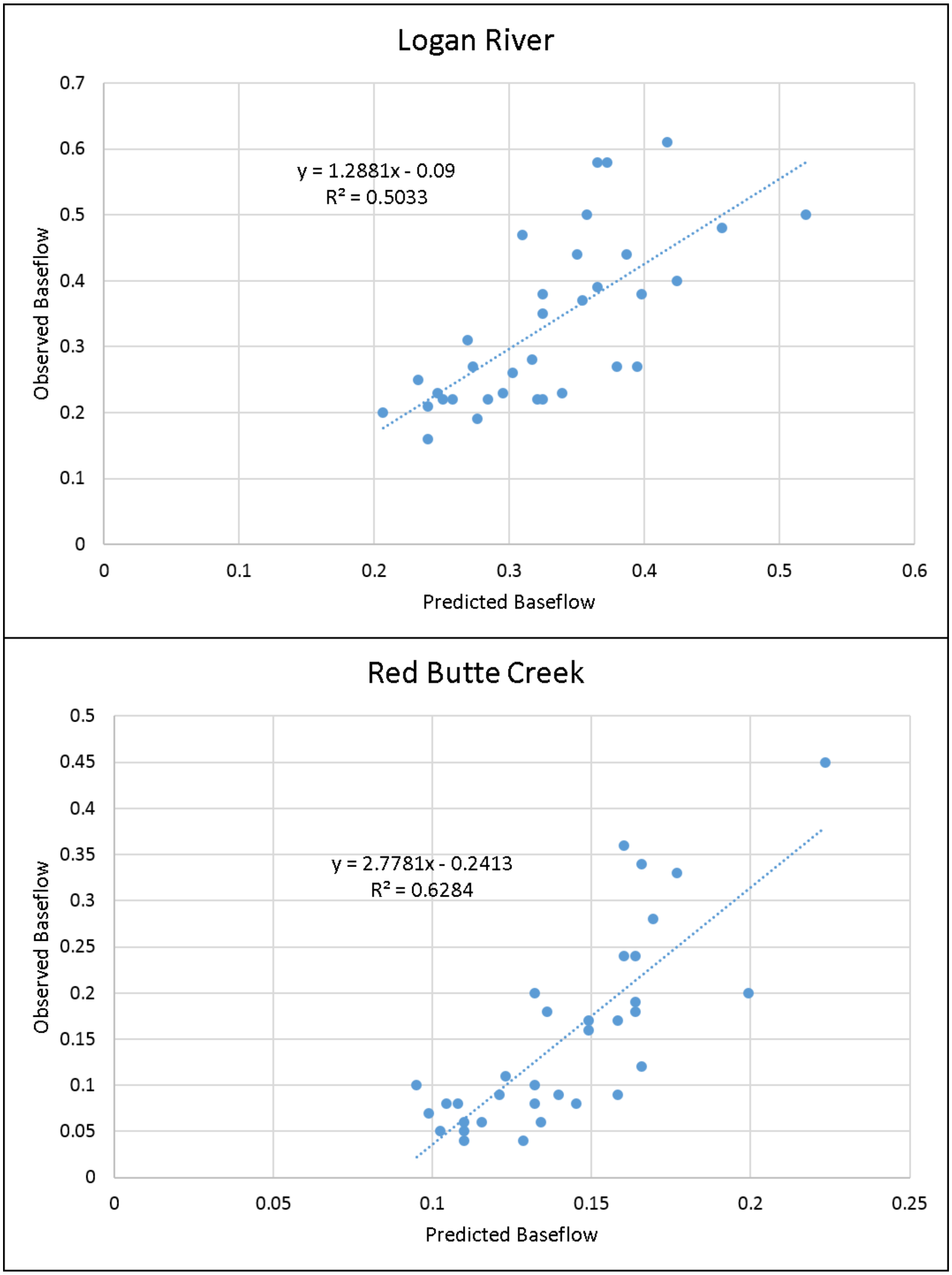

FIGURE B-6 Predicted and observed data were compared to determine the accuracy of the relationship. The Logan River predicted and observed data were correlated with an $R^2 = 0.50$. Red Butte Creek had an $R^2 = 0.63$.

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Trinity Stout
1600 E Canyon Rd.
Logan, UT 84321

Milada Majerova,

I am in the process of preparing my thesis in the Civil and Environmental Engineering department at Utah State University.

The article "Impacts of beaver dams on channel hydraulics and substrate characteristics in a mountain stream", of which I am first author, and which will appear in journal Ecohydrology (DOI: 10.1002/eco.1767), reports an essential part of my thesis research. I would like permission from you as a co-author to reprint the article as a chapter in my thesis, which may require some slight revision.

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(208) 709-4400
trinity.stout@aggiemail.usu.edu

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Signed: ______________________________

Date: 7/27/2016