THE EFFECTS OF WATER QUANTITY ON FISH ASSEMBLAGE COMPOSITION IN THE UPPER CIMARRON RIVER

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Abstract: Many prairie streams suffer from altered flow regimes as a result of surface and groundwater extraction for irrigation. Changes in flow regimes can alter resident fish assemblage structure and abundance. To determine the effect of water withdrawal on the resident fish assemblage of the upper Cimarron River, I developed and pursued two objectives: 1) quantify the magnitude of stream flow loss in the upper Cimarron River and its effect on the fish assemblage and 2) determine concurrent fish assemblage differences among sites that differ in water quantity. To quantify stream flow loss, I identified a temporal change in stream flow using segmented regression and Indicators of Hydrologic Alteration (IHA) software to compare two periods surrounding the temporal change ("pre" and "post-impact") to determine the magnitude. To determine the effect of stream flow loss on the resident fish assemblage, upper Cimarron River fish collection records from Oklahoma State University, the University of Oklahoma, and the University of Kansas were separated by date into pre and post-impact communities and then compared. To compare concurrent fish assemblage differences among sites with different water quantities, I sampled the Ditch Valley area of the Cimarron River where a diversion of stream flow into an irrigation canal provides four distinct sample sites with different water flows (upstream, canal, diverted river, and downstream). Temperature, salinity, and discharge were measured for each site. Fish were sampled using a seine bi-monthly between May 2012 and December 2012, with an additional sampling in June 2013. A significant change in upper Cimarron River stream flow was detected in 1986, resulting in decreased flows and a change in fish assemblage structure. Post-impact assemblage favored tolerant species able to adapt to reduced water flows. Flow reductions appear to be correlated with of increased groundwater withdrawal for irrigation. Historical drought made comparisons of Ditch Valley fish communities difficult, but general trends were apparent. Species richness was positively correlated with water quantity and fish occurrence in the simplified habitat of the canal was most likely related to entrainment or food resource availability.

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CHAPTER I

INTRODUCTION TO THESIS

Despite their importance, freshwater ecosystems globally are facing increasing peril from anthropogenic activities (Malmqvist and Rundle 2002, Dudgeon *et al.* 2005, Vörösmarty *et al.* 2010). Over the last few centuries, humans have significantly altered streams and rivers by damming, channelizing, and diverting stream flow, as well as extracting water for “off-stream” usages, such as human consumption, agriculture, and industry (Pringle 2000, Vörösmarty *et al.* 2010). These actions have diminished ecosystem functions and reduced the quality of habitat for our wildlife resources (Pringle 2000, EPA 2013). Currently, only a small proportion of the world’s water systems remain unaffected by humans (Vörösmarty *et al.* 2010), and that number is expected to decline as demand for freshwater increases in response to expected population growth (Pringle 2000, Malmqvist and Rundle 2002).

Prairie streams in the southern plains have fared no better than those globally. In fact, prairie streams, such as the Cimarron River (Taylor and Miller 1990), may be even more imperiled because many of the former prairies that encompassed these freshwater systems have been fragmented and altered (Dodds *et al.* 2004). Prairie streams also suffer from surface and groundwater extraction for use in agriculture and industry, resulting in streams that are dry for much of the year (Young *et al.* 2005, Dodds *et al.* 2004, Steward *et al.* 2013). Combined, these activities have altered the flow regime, which dictates when, how much, and how often water is available to wildlife (Malmqvist and Rundle 2002, Dewson *et al.* 2007, Carlisle *et al.* 2010). Because flow regimes control many of the physical, chemical, and biological processes in aquatic ecosystems (Poff *et al.* 1997, Carlisle *et al.* 2010), aquatic species that reside in these systems for any part of their life cycle rely on its variation in flow.

Alterations to natural flow regimes can have severe consequences to the organisms adapted to them (Geist 2011). Extended periods of low flow, often exacerbated by human activities such as groundwater extraction and water diversion, are particularly detrimental. Matthaei et al. (2010) pointed out that flow reduction in the form of water extraction is an increasingly dominant stressor in western portions of the United States and Reash and Pigg (1990) stated that stream flow is probably the most important variable affecting biological communities and species richness.

The Cimarron River flows 1,117 km from its origin in northeastern New Mexico, to its confluence with the Arkansas River at Keystone Reservoir in Oklahoma. The majority of this flow is in Oklahoma, but there are brief forays into Colorado and Kansas. The Cimarron River is largely undammed and considered one of the longest free-flowing water systems in the United States. It is described as an intermittent 'losing' stream characterized by high rates of evaporation, infiltration and dissolved solids. It is a relatively unshaded, shallow river with a shifting sand substrate (Hargett et al. 1999, Pigg 1988, Reash and Pigg 1990). Historically, surface flows in the river were highly variable (especially in the semi-arid portion in the west) and driven by pulsed precipitation, with large flows during heavy rains and lack of flow during extended dry periods. Many of the fish species found in the Cimarron River have life history strategies that match these harsh conditions.

One such species adapted to these harsh conditions is the threatened Arkansas River shiner. This small cyprinid was once abundant in the western portion of the Arkansas River basin in New Mexico, Kansas, Oklahoma, and Texas (USFWS 2011). Current surveys of the Cimarron River, however, have found no evidence of the species in since 1992 (Daniel Fenner, U.S. Fish & Wildlife Service, personal comminication). Changes to the natural flow regime may help explain the dissapearance of this species from the Cimarron River (Cross *et al.* 1983).

Unfortunately, researchers have reported that water flows in the Cimarron River have been declining and several species of native fish are either extirpated or becoming more rare (Cross *et al*. 1983, Cross *et al.* 1985, Larson 1991). Surveys of the upper portion of the Cimarron River have occurred for many years, particularly since 1987 when the Arkansas River shiner became protected under the Endangered Species Act (ESA), but a current comparison between historical and current fish communities has not been conducted to document changes in the fish assemblage. I developed and pursued two main objectives that are addressed in the following chapters of this thesis: 1) quantify the magnitude of stream flow loss in the upper Cimarron River and its effect on the fish assemblage and 2) determine concurrent fish assemblage differences among sites differing in water quantity.

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CHAPTER II

THE EFFECTS OF REDUCEED STREAM FLOW ON THE FISH ASSEMBLAGE IN THE UPPER CIMARRON RIVER

INTRODUCTION

A natural flow regime is the dynamic historical sequence of high and low flows that consists of five critical elements: magnitude, timing, frequency, duration, and rate of change (Poff *et al.* 1997, Mathews and Richter 2007, Poff and Zimmerman 2010). While stream flow is ultimately derived from precipitation, whether rain or snow, only a small portion of it is delivered directly into a stream system (Poff *et al.* 1997). The rest enters the stream over time by some combination of surface, soil, and ground water flow (Poff *et al.* 1997). Climate, geology, topography, soils, and vegetation mediate the rate at which water enters the stream and the pathways by which it is delivered; thereby influencing the magnitude, timing, frequency, duration, and rate of change (Poff *et al.* 1997). Because the natural flow regime influences species distribution and abundance, human activities that alter any of these factors can result in substantial and cascading effects on the terrestrial and aquatic wildlife that rely on it (Poff *et al.* 1997, Pringle 2000, Carlisle *et al*. 2010). One increasingly prevalent stressor that has altered natural flow regimes, especially in arid portions of the United States, is water extraction (Matthaei *et al.* 2010).

Water extraction is the removal of water for use “off-stream,” and includes activities such as irrigation, industrial use, inter-basin transfers, and human and animal consumption (Vorosmarty *et al.* 2010). The largest source of water extraction in arid climates is, by far, the removal of surface and groundwater for irrigation (Carlisle *et al*. 2010). In 2005, 37% of all freshwater and 67% of extracted groundwater was used for irrigation (Kenny *et al.* 2009). All told, more than 24 million hectares in the U.S. were irrigated in 2005 using almost 564,000 m^3 of freshwater (Kenny *et al.* 2009). The extraction of water from a stream or its corresponding aquifer can have several effects to the natural flow regime, the most obvious of which is a decrease in stream flow (Poff *et al.* 1997).

Declining stream flow has been a major determinate of species loss in some stream systems (Xenopoulos *et al.* 2005). Effects of reduced stream flow include changes in nutrient levels and decreases in physical habitat (depth, wetted width, and flow velocity), as well as increases in fluctuations and maximum levels of temperature and conductivity (Dewson et al. 2007). Fishes within Great Plains prairie streams have adapted to harsh, rapidly changing environmental conditions (Taylor *et al.* 1993, 1996), but modified flow regimes may exacerbate conditions to the point that a species’ tolerance may be exceeded. For example, Stevenson’s (1997) work with benthic algae indicated that reductions in current velocity has an effect on abiotic stressors, such as pH and salinity, which in turn directly affects the organism’s ability to utilize available resources to function. This inability to utilize resources may result in a species experiencing reductions in feeding, reproduction, or growth, making it more prone to disease and predation as a result of increased stress.

In addition to exceeding tolerances, physical changes in the environment may also alter species interactions. For example, the plains killifish (*Fundulus zebrinus*) and the Red River Pupfish (*Cyprinodon rubrofluviatilis*) commonly occur together in the upper Red River basin, but the latter outnumbers the former at high salinities, whereas the reverse is true at lower salinities (Echelle *et al.* 1972). In part, this pattern seems to reflect differential competitive abilities, with the pupfish outcompeting the killifish in species-poor assemblages at high salinities, but not in the more species-rich assemblages at lower salinities (Echelle *et al.* 1972). Such changes in fish assemblage structure occur because species able to tolerate extreme conditions remain while those that cannot disappear. Taylor et al. (1993) pointed out that in variable environments, physical factors play an important role in determining assemblage structure. Therefore, differences in fish population and assemblage dynamics should occur concomitantly with reduced flow and greater fluctuations in environmental conditions.

The majority of the upper Cimarron River flows through the panhandle of Oklahoma and southwestern Kansas where irrigation driven agriculture is the major economic force (Harrington *et al.* 2010, USDA 2014). In fact, farmland comprises 97% of all land in Beaver and Cimarron Counties, through which the Cimarron River flows (USDA 2014). Areas surrounding the Cimarron River are a mixture of cattle ranches, dry crops, and irrigated cropland. Cropland can be irrigated by diversion of surface water or by center point irrigation that withdraws groundwater from the alluvial aquifer or the underlying High Plains Aquifer (HPA).

The HPA is relatively shallow, rising to within 50-400 feet of the surface in some parts of western Oklahoma (Luckey *et al.* 2000) and is responsible for most of the base flow in the river during low flow periods (Kendy and Bredehoeft 2006). Water withdrawal for irrigation started slowly in the Oklahoma panhandle in the 1930's and continued at this pace for the next 30 years (Hart *et al.* 1976). Withdrawals for irrigation increased exponentially, however, beginning in 1964 (Hart *et al.* 1976). Whether removal is from surface-water diversion or groundwater extraction, there is an initial decrease in stream flow (Poff *et al.* 1997, Burt *et al.* 2002). My objectives were: 1) to evaluate water flow regimes in the upper Cimarron River in relation to precipitation and groundwater extraction, and 2) to examine the possibility of associated changes in the fish assemblage.

METHODS

Stream Flow Assessment

To examine stream flow regime in the upper Cimarron River, I obtained daily stream flow data from the U.S. Geological Survey (USGS) stream gauge farthest downstream in the watershed (HUC #07156900 north of Forgan, Oklahoma) (Figure 1). Average daily discharge values from 1966 to 2012 were downloaded, converted to cubic meters per second (m^3/s), and averaged by calendar year to obtain the annual mean daily discharge. I then calculated a segmented regression (piecewise) using SigmaPlot (v 13.0) software to identify points of change ("pre" and "post-impacts").

Once impact points were identified, I used Indicators of Hydrologic Alteration 7.1 (IHA) software to analyze and compare pre and post-impact periods as they relate to components of the natural flow regime (Richter *et al.* 1996, Mathews and Richter 2007). Because the data were not normally distributed, medians and percentages above and below the median (± 25%) were used (TNC 2009). Extreme low flows were calculated as < 10% of all daily flows for the selected period. I used significance counts (< 0.10) as indicators of statistical significance for IHA parameters (TNC 2009).

Watershed assessment of precipitation and snowfall

Average daily rain and snowfall dating back to 1966 were downloaded from thirteen gauges administered by the National Climate Data Center and located upstream of the USGS stream gauge at Forgan, Oklahoma. I transformed the data into centimeters and mean annual rainfall and snowfall were calculated. To fill data gaps, mean daily rainfall and snowfall amounts across all stations were used and daily values were then summed to obtain annual amounts. Annual values were then averaged pre and post-impact and compared with ANOVA to test for significance ($\alpha < 0.10$).

Groundwater Well Assessment

To calculate groundwater usage, I used the online database of wells available through the Oklahoma Water Resource Board, Kansas Geological Survey, Colorado Division of Water Resources, and New Mexico Office of the State Engineer. Wells were organized by completion date or by date of the permit if the completion date was missing. Wells were organized by year and the mean number of wells constructed each year in the Upper Cimarron River watershed was determined. The mean numbers of wells built were then averaged pre and post-impact and compared with ANOVA to test for significance (α < 0.10). Additionally, I then summed the cumulative number of wells by year for each year between 1966 and 2012.

Fish Assemblage Assessment

Using the collection records of Oklahoma State University, University of Oklahoma, and Kansas University, I compiled a list of all species present within the upper Cimarron River pre and post-impact using location or lat/long records. Because survey protocols could not be verified, or were not standardized for each collection, species abundances were discarded and converted to presence-absence. I compared the composition of fish assemblages pre and post-impact using the Jaccard Index of Similarity. The Jaccard index (S_j) ranges from 0 to 1, with 0 indicating no similarity and 1 indicating identical assemblages (Phillips and Johnston 2004).

RESULTS

Stream Flow Assessment

Segmented regression displayed a downward trend in mean daily discharge over time, with a significant (P <0.01) break in the rate of change beginning in 1986 (Figure

2). Mean daily discharge and year were strongly correlated (r^2= 0.79). Subsequent analyses thus considered 1966-1985 as pre-impact and 1986-2012 as post-impact.

Significant changes in stream flow regime were evident from pre to post-impact in 51 of 67 parameters compared by IHA, including 10 of 11 measuring magnitude and duration, 5 of 6 measuring timing, and 3 of 3 measuring frequencies and rate of change (Table 1). Most dramatically, 1-day maximum flow (Figure 3) decreased ~80% post-impact, while extreme low flows nearly doubled in frequency and duration (Figures 4 and 5). Examples of other changes include an approximately 85% decrease in high-flow frequency (Figure 6). High flow peak and duration also became more dispersed around the median. Additionally, stream flow took longer to rise and was quicker to fall post-impact. Finally, small floods that typically occurred in July pre-impact arrived earlier in May post-impact.

Precipitation and Snowmelt Assessment

Mean annual precipitation changed little in the Upper Cimarron River between pre and post-impact periods. The period prior to impact experienced 42.78 and 80.72 cm of mean annual rainfall and snowmelt respectably; while post-impact averaged 46.28 and 83.63 cm (Figures 7 and 8). The ANOVA indicated no statistical difference between pre and post-impact rainfall (P = 0.14) or snowfall (P = 0.70).

Groundwater Well Assessment

The number of groundwater wells built per year increased by an average of 34 wells between pre to post-impact periods (Figure 9), although this was not statistically different between the two periods (P =0.26). Initially, the cumulative number of wells increased little between 1966 and 1974 constructing 569 wells in a nine year period ($\bar{x}$ = 63) (Figure 10). In contrast, between 1975 and 1977 a total of 1,386 wells were built over a three year period ($\bar{x}$ = 462). Following 1977, the cumulative number of wells constructed continued to increase, but at a much slower rate ($\bar{x}$ = 224).

Fish Assemblage Assessment

Pre-impact collections of the upper Cimarron River reported 21species representing 7 families. Most of these were native small-bodied cyprinids (n = 8), but included three nonindigenous cyprinids (goldfish, common carp, and Red River shiner), one catostomid (white sucker), and one centrarchid (largemouth bass) (USGS 2012). In contrast, post-impact surveys reported 27 species representing 9 families. Similar to pre-impact surveys, the majority of post-impact species were small-bodied cyprinids (n=9). The number of nonindigenous species in post-impact surveys increased to eight, including four cyprinids (goldfish, common carp, Red River shiner, and river shiner), one cyprinodontid (Red River pupfish), one catostomid (bigmouth buffalo), one centrarchid (largemouth bass), and one percid (common logperch) (USGS 2012). While a few of the species identified as nonindigenous in this analysis are native to the Cimarron River, they are classified as a nonindigenous aquatic species in the upper portion of the Cimarron River according to the United States Geological Survey.

Three species reported as present pre-impact were not detected in post-impact collections. These were the peppered chub, channel catfish, and white sucker. Post impact collections documented nine species not found in earlier collections, of which four were nonindigenous species (Red River pupfish, bigmouth buffalo, common logperch and river shiner) (USGS 2012). Jaccard's similarity index indicated a moderate amount of similarity between pre and post-impact communities ($S_j = 0.60$).

DISCUSSION

The quantity and timing of stream flow are essential to the biological integrity of a stream system (Poff *et al.* 1997). While it is obvious that the quantity of water has declined in the Upper Cimarron River, the cause or causes are not as apparent. I focused on irrigation as the likely primary driver because agricultural activities rely on large amounts of water to economically sustain these activities (Pringle 2000). Diversion of surface flow is usually the first impact to stream flow, but as surface flow declines to the extent that landowners can no longer draw sufficient quantities of water directly from surface waters (Eheart and Tornil 1999), their reliance shifts to groundwater wells. Additionally, landowners farthest from the river do not typically have access to surface water resources and must rely on groundwater for their irrigation needs (Eheart and Tornil 1999). Unfortunately, the groundwater resources upon which they rely also sustains the base flow of streams (Kendy and Bredehoeft 2006), which declines with high irrigation demand during the growing season (June through August) when precipitation is normally minimal (Eheart and Tornil 1999).

Surface flow and their corresponding groundwater aquifers interact in a variety of ways (Sophocleous 2002). When water is removed from the underlying aquifer through a well, a cone of depression is established that intercepts water from surrounding water resources (Burt *et al* 2002). The amount of water withdrawn from the aquifer must be offset by reduced groundwater storage, increased recharge, reduced evaporation and evapotranspiration, or reduced stream flow (Theis 1941). This cone of depression can divert water away from surface flow (Theis 1941, Sophocleous 2002), by intercepting precipitation and irrigation returns, further affecting the timing and amount of stream flows (Burt *et al.* 2002).

The High Plains Aquifer (HPA) serves as the source of all groundwater in the upper Cimarron River (USGS 2013a). Over 165,000 wells pump groundwater from the HPA and its principal source of recharge is precipitation (USGS 2013b) which has not changed appreciably in the last 50 years. The portion of the Cimarron River overlying the HPA is an arid region with high evapotranspiration and low rates of recharge (Sophocleous 2005). Stream flows have declined because water in the area is withdrawn at a rate of 12-40% greater than the rate of recharge (Sophocleous 2005).

Apart from lower flow, the upper Cimarron River system has become more stable. Cross *et al.* (1985) reported that the Cimarron River appears more stable than one would expect of a plains stream of this size (22,108 km^2 upstream of gauge), and the loss of stream flow and variability likely affects the resident fish assemblage. Because stream systems experience a time lag between withdrawal of groundwater and decreased stream flows (Sophocleous 2005, Gido *et al.* 2010), the cumulative effect of large numbers of wells built in previous years may not manifest itself until a later point in time (Burt *et al.*

2002). This time lag may explain why year-to-year variability in mean daily discharge was high until 1979 but which subsequently declined to fairly constant, relatively low discharge levels (Figure 2). When we compare the stream flow declines observed in the piecewise regression with the cumulative number of wells per year, a correlation between the two becomes evident. Well construction grew exponentially between 1975 and 1978 increasing the number of well in the upper Cimarron River watershed from 569 wells to 2,190, becoming the most likely cause of declining stream flow and variability that is occurred in 1979.

Another interesting observation is that when groundwater is removed for irrigation, an increase in stream flow downstream typically occurs as water that is not taken up by plants or evaporated ends up as surface flow to the stream (Eheart and Tornil 1999). Interestingly, if this occurs within the upper Cimarron watershed, it is either in such small quantities that no appreciable amount of increase is observed or infiltration in this area is so high that surface flow to the stream does not occur. Most likely it is a combination of both as center point irrigation has become more efficient (Luckey 2000) and it may be that there is little water that is not taken up by plants. Any remaining water probably infiltrates to aquifer or is evaporated in the arid climate of this area.

Changes in stream flow can alter a fish assemblage by replacing some species with species more suited to the new conditions (Poff *et al.* 1997, Carlisle *et al*. 2010, Gido *et al.* 2010, Poff *et al.* 2010). For example, reduced minimum flows could favor species that guard nests over species that do not protect offspring after spawning (Carlisle *et al*. 2010). In extreme low flows, oxygen depletion in water surrounding eggs can result in delays in development and suffocation. Nest guarders circulate water around their eggs as they chase away predators, thus ensuring sufficient oxygen levels for developing young (Carlisle *et al*. 2010). In contrast, eggs that are left without parental care receive no additional water circulation and can deplete the DO from the surrounding water. This is but one possible reason that the white sucker, a simple nester that spawns over coarse substrate (Miller and Robison 2004, Tomelleri and Eberle 2011) has not been found in the upper Cimarron River since 1963. In contrast, yellow bullhead, central stoneroller, and bluegill are all nest guarders (Miller and Robison 2004) that are newly established in the upper Cimarron River. Another potential cause in the decline of the white sucker is the failure of their larvae to reach suitable nursery habitat. Larval suckers drift upon emergence (McPhee 2007) where they are transported downstream to backwater nursery habitats. As flows decrease, the abundance of suitable nursery habitat becomes rarer. Furthermore, a reproductive strategy of some native prairie fishes is broadcast spawning of semi-buoyant eggs that develop as they drift downstream (Durham and Wilde 2006). Reduced stream flows cause egg drift to be truncated, leaving them to develop in sub-optimal habitats or settle out and become smothered by sediment (Durham and Wilde 2006). Many of these species, such as the peppered chub (*M. tetranema*), plains minnow (*H. placitus*), Arkansas River shiner (*N. girardi*), and flathead chub (*P. gracilis*) have either been extirpated or reduced in abundance since 1986. The Arkansas River shiner is federally threatened and probably extirpated from the Cimarron River with none collected since 1992 (Daniel Fenner, U.S. Fish & Wildlife Service, personal comminication), and the flathead chub is nearly extirpated.

While I classified stream flows as either pre (1966-1985) or post-impact (1986-2012), human impacts to the upper Cimarron River likely go back much farther than

1966 and the term pre-impact can be viewed as arbitrary. The construction of wells and the removal of large amounts of groundwater from the HPA, however, seem to adequately explain the reductions in upper Cimarron River stream flow during the periods analyzed, and likely is a major factor causing the observed changes in fish assemblage. Historically, most of this water was used for irrigation (Luckey *et al.* 2000), but recently landowners in southwestern Kansas and the Oklahoma panhandle have begun selling water to energy companies for oil and gas extraction. As the demand for water increases, additional impacts to the resident ichthyofauna are likely.

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Tables and Figures

Table 1. Indicators of Hydrologic Alteration (IHA) scorecard generated from stream flow data collected by USGS gage #07156900 on the Cimarron River north of Forgan, OK, investigating changes in stream flow pre and post January 1, 1986. Significance counts can be interpreted similarly to p-values where < 0.10 represents a significant change between pre and post-impact values (bold).

	Medians		Coefficient of Dispersion (CD)		Deviation Factor		Significance Count	
	Stream flow (m^3)							
IHA Group	Pre-impact	Post-impact	Pre-impact	Post-impact	Medians	C.D.	Medians	C.D.
Group 1: Monthly								
January	**1.76**	**1.10**	0.46	0.33	**0.37**	0.27	**0.00**	0.61
February	**1.80**	**1.16**	0.39	0.38	**0.35**	0.03	**0.00**	0.97
March	**1.87**	**1.10**	0.29	0.36	**0.41**	0.25	**0.00**	0.43
April	**1.86**	**1.12**	0.34	0.28	**0.40**	0.19	**0.00**	0.55
May	**1.71**	**0.93**	0.28	0.39	**0.45**	0.42	**0.00**	0.15
June	**1.40**	**0.79**	0.54	0.43	**0.43**	0.21	**0.00**	0.52
July	**1.12**	**0.65**	0.42	0.43	**0.42**	0.04	**0.00**	0.93
August	**1.15**	**0.62**	0.62	0.41	**0.46**	0.34	**0.00**	0.60
September	**1.06**	**0.74**	0.64	0.33	**0.31**	0.49	**0.00**	0.41
October	**1.40**	**0.82**	0.65	0.21	**0.41**	0.68	**0.00**	0.29
November	**1.66**	**0.91**	0.42	0.28	**0.45**	0.33	**0.00**	0.56
December	**1.74**	**0.99**	0.37	0.31	**0.43**	0.15	**0.00**	0.73
Group 2: Magnitude and duration of annual extremes								
1-day minimum	**0.65**	**0.45**	**0.36**	**0.19**	**0.30**	**0.48**	**0.03**	**0.07**
3-day minimum	**0.67**	**0.46**	**0.44**	**0.24**	**0.31**	**0.45**	**0.02**	**0.09**
7-day minimum	**0.73**	**0.49**	**0.46**	**0.21**	**0.33**	**0.55**	**0.02**	**0.03**
30-day minimum	**0.96**	**0.56**	0.57	0.25	**0.41**	0.56	**0.00**	0.11
90-day minimum	**1.24**	**0.69**	0.62	0.31	**0.44**	0.50	**0.00**	0.48
1-day maximum	17.88	3.31	2.01	1.70	0.81	0.15	0.22	0.80
3-day maximum	**9.85**	**2.28**	2.36	1.23	**0.77**	0.48	**0.07**	0.47
7-day maximum	**5.72**	**1.80**	2.15	0.77	**0.69**	0.64	**0.02**	0.28
30-day maximum	**3.31**	**1.42**	1.27	0.28	**0.57**	0.78	**0.01**	0.11
90-day maximum	**2.48**	**1.24**	0.90	0.25	**0.50**	0.72	**0.00**	0.13
Number of zero days	0.00	0.00	0.00	0.00				
Base flow index	**0.40**	**0.55**	0.25	0.30	**0.36**	0.22	**0.00**	0.26

Table 1. Continued

IHA Group	Medians: Stream flow (m^3) Pre-impact	Medians: Stream flow (m^3) Post-impact	Coefficient of Dispersion (CD) Pre-impact	Coefficient of Dispersion (CD) Post-impact	Deviation Factor Medians	Deviation Factor C.D.	Significance Count Medians	Significance Count C.D.
Group 3: Timing of annual extremes								
Date of minimum	211.50	209.00	0.11	0.13	0.01	0.21	0.90	0.71
Date of maximum	**207.00**	**147.00**	0.30	0.28	**0.33**	0.09	**0.06**	0.72
Group 4: Frequency and duration of pulses								
Low pulse count	11.00	11.00	**0.64**	**1.00**	0.00	**0.57**	1.00	**0.06**
Low pulse duration	**3.00**	**7.00**	**0.83**	**3.50**	**1.33**	**3.20**	**0.00**	**0.02**
High pulse count	19.50	1.00	**0.83**	**2.00**	0.95	**1.40**	0.11	**0.08**
High pulse duration	**2.25**	**1.50**	0.44	0.67	**0.33**	0.50	**0.00**	0.24
Low Pulse Threshold	1.19							
High Pulse Threshold	2.01							
Group 5: Rate and frequency of change in conditions								
Rise rate	**0.13**	**0.03**	0.86	1.00	**0.78**	0.16	**0.00**	0.73
Fall rate	**-0.13**	**-0.03**	-0.83	-1.00	**0.78**	0.20	**0.03**	0.76
Number of reversals	**142.00**	**116.00**	**0.12**	**0.20**	**0.18**	**0.61**	**0.00**	**0.07**
EFC Parameters								
Extreme low peak	0.79	0.79	0.11	0.14	0.00	0.33	0.49	0.42
Extreme low duration	**3.00**	**5.00**	**0.67**	**2.20**	**0.67**	**2.30**	**0.05**	**0.03**
Extreme low timing	218.00	203.00	**0.12**	**0.20**	0.08	**0.69**	0.33	**0.09**
Extreme low freq.	**5.50**	**9.00**	1.55	0.67	**0.64**	0.57	**0.00**	0.19
High flow peak	2.49	2.28	**0.15**	**0.23**	0.09	**0.53**	0.15	**0.07**
High flow duration	3.50	2.00	**0.54**	**0.94**	0.43	**0.75**	0.15	**0.04**
High flow timing	144.00	170.50	0.28	0.23	0.14	0.18	0.19	0.67
High flow frequency	**16.00**	**2.00**	0.86	2.00	**0.88**	1.33	**0.06**	0.13
High flow rise rate	**0.47**	**0.87**	0.55	1.00	**0.85**	0.81	**0.00**	0.10
High flow fall rate	**-0.31**	**-0.55**	-0.65	-0.78	**0.79**	0.21	**0.00**	0.56
Small Flood peak	33.98	22.71	0.63	0.16	0.33	0.74	0.13	0.33
Small Flood duration	21.00	6.00	1.17	0.83	0.71	0.29	0.31	0.65
Small Flood timing	**212.00**	**128.00**	0.26	0.07	**0.46**	0.73	**0.05**	0.12
Small Flood freq.	0.00	0.00	0.00	0.00				
Small Flood riserate	14.12	19.68	1.90	0.54	0.39	0.71	0.45	0.36
Small Flood fallrate	-3.10	-3.51	-1.90	-0.53	0.13	0.72	0.81	0.18

Table 1. Continued

	Medians		Coefficient of Dispersion (CD)		Deviation Factor		Significance Count	
	Stream flow (m^3)							
IHA Group	Pre-impact	Post-impact	Pre-impact	Post-impact	Medians	C.D.	Medians	C.D.
Large flood timing	140.00		0.10					
Large flood freq.	0.00	0.00	0.00	0.00				
Large flood riserate	34.68		0.40					
Large flood fallrate	-16.59		-0.79					

Table 2. Species presence in the upper Cimarron River pre and post-impact water flow years. Data was obtained from collections from Oklahoma State University, Oklahoma University, and Kansas University. Asterisk denotes non-native species to the upper Cimarron River.

Common Name	Scientific Name	1966-1985	1986-2011
	Catostomidae		
White sucker*	*Catostomus commersoni*	X	
Bigmouth buffalo*	*Ictiobus cyprinellus*		X
	Centrarchidae		
Green sunfish	*Lepomis cyanellus*	X	X
Orange spotted sunfish	*Lepomis humilis*	X	X
Bluegill	*Lepomis macrochirus*		X
Longear sunfish	*Lepomis megalotis*	X	X
Largemouth bass*	*Micropterus salmoides*	X	X
	Clupeidae		
Gizzard shad	*Dorosoma cepedianum*		X
	Cyprinidae		
Central stoneroller	*Campostoma anomalum*		X
Goldfish*	*Carassius auratus*	X	X
Red shiner	*Cyprinella lutrensis*	X	X
Common carp*	*Cyprinus carpio*	X	X
Plains minnow	*Hybognathus placitus*	X	X
Peppered chub	*Macrhybopsis tetranema*	X	
Emerald shiner	*Notropis atherinoides*		X
Red River shiner*	*Notropis bairdi*	X	X
River shiner*	*Notropis blennius*		X
Arkansas River shiner	*Notropis girardi*	X	X
Sand shiner	*Notropis stramineus*	X	X
Suckermouth minnow	*Phenacobius mirabilis*	X	X
Fathead minnow	*Pimephales promelas*	X	X
Flathead chub	*Platygobio gracilis*	X	X
	Cyprinodontidae		
Red River pupfish*	*Cyprinodon rubrofluviatilis*		X
	Fundulidae		
Northern plains killifish	*Fundulus kansae*	X	X

Table 2. Continued

Common Name	Scientific Name	1966-1985	1986-2011
	Ictaluridae		
Black bullhead	*Ameiurus melas*	X	X
Yellow bullhead	*Ameiurus natalis*		X
Channel catfish	*Ictalurus punctatus*	X	
	Percidae		
Arkansas darter	*Etheostoma cragini*	X	X
Common logperch*	*Percina caprodes*		X
	Poeciliidae		
Western mosquitofish	*Gambusia affinis*	X	X

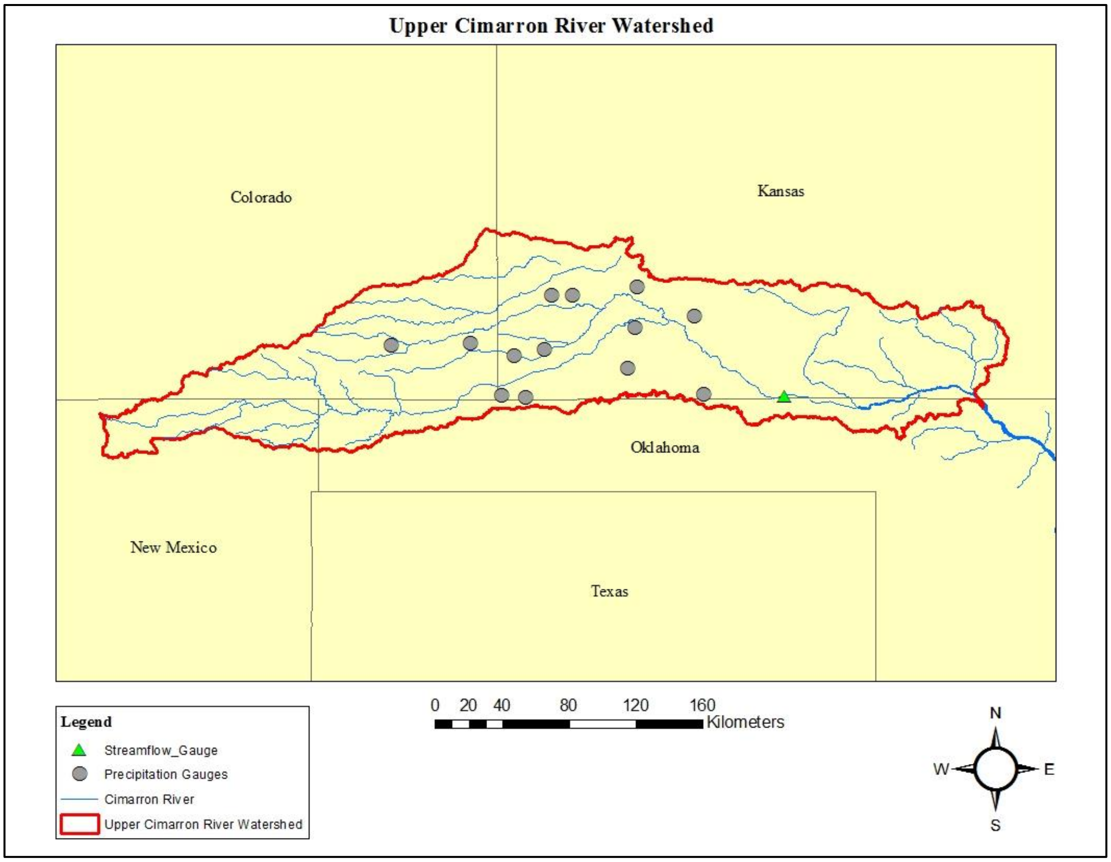

Figure 1. Map depicting the location and extent of the Upper Cimarron River watershed spanning portions of Oklahoma, Kansas, Colorado, and New Mexico. Precipitation and stream flow data from 1966 to 2012 were obtained from the depicted gauges.

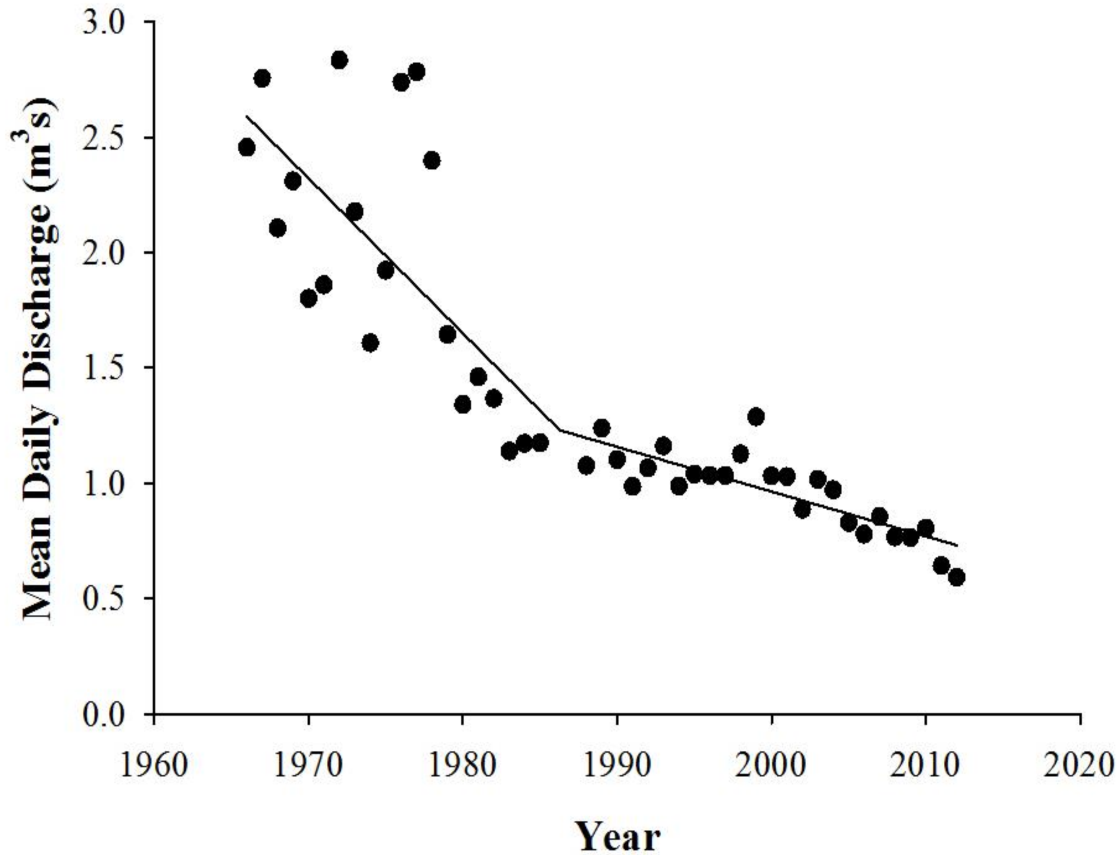

Figure 2. Segmented (piecewise) regression representing stream flow in the Cimarron River north of Forgan, Oklahoma, from 1966 to 2012. The scatterplot represents the annual mean daily discharge data used to develop the segmented regression. The mean daily discharge for the years 1986 and 1987 were omitted due to incomplete data. Stream flow data was obtained from USGS stream gauge #07156900.

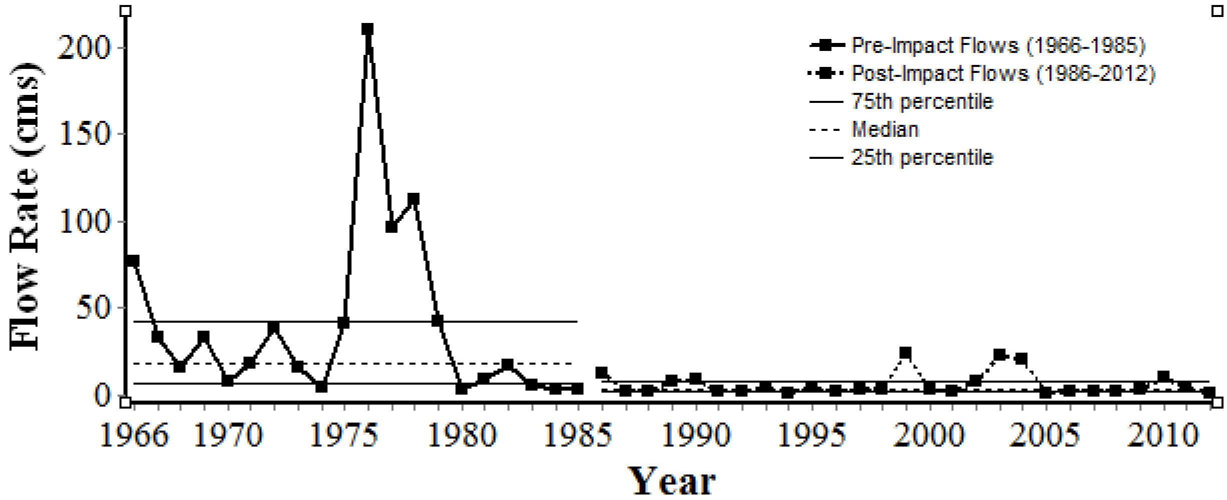

Figure 3. 1-day maximum stream flow for pre and post-impacted Cimarron River in the Upper Cimarron River watershed. Stream flow data were collected from USGS gauge 07156900 on the Cimarron River north of Forgan, Oklahoma. Data are based on calendar year and excludes data between October 1986 and October 1987 due to gauge malfunction.

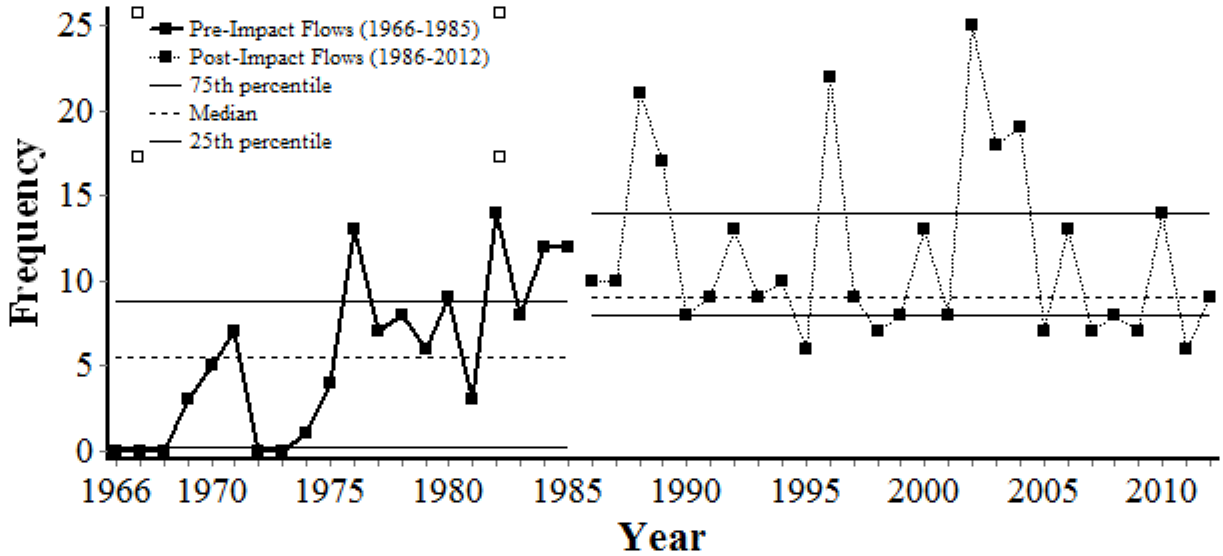

Figure 4. Frequency of extreme low flows for pre and post-impacted Cimarron River in the Upper Cimarron River watershed. Stream flow data were collected from USGS gauge 07156900 on the Cimarron River north of Forgan, Oklahoma. Data are based on calendar year and excludes data between October 1986 and October 1987 due to gauge malfunction.

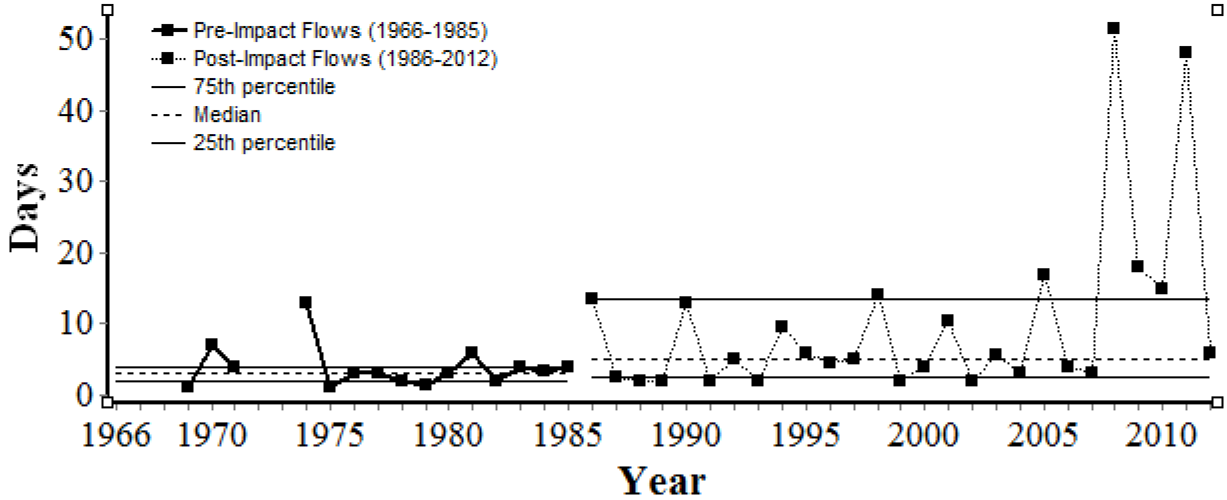

Figure 5. Duration of extreme low flows for pre and post-impacted Cimarron River in the Upper Cimarron River watershed. Stream flow data were collected from USGS gauge 07156900 on the Cimarron River north of Forgan, Oklahoma. Data are based on calendar year and excludes data between October 1986 and October 1987 due to gauge malfunction.

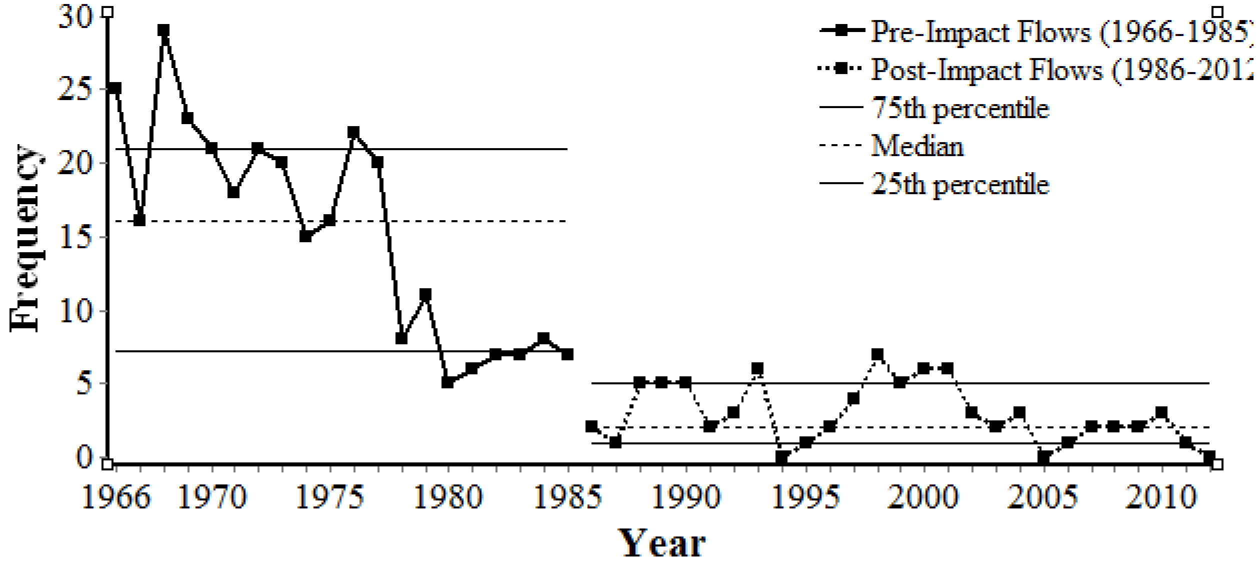

Figure 6. Frequency of high flows for pre and post-impacted Cimarron River in the Upper Cimarron River watershed. Stream flow data were collected from USGS gauge 07156900 on the Cimarron River north of Forgan, Oklahoma. Data are based on calendar year and excludes data between October 1986 and October 1987 due to gauge malfunction.

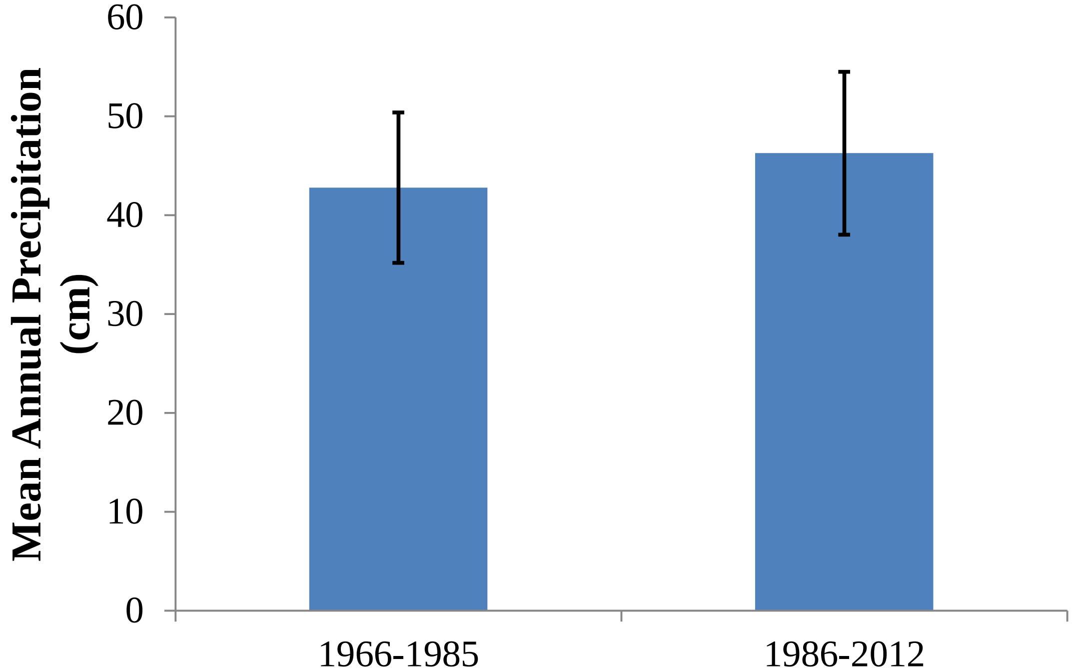

Figure 7. Mean annual precipitation in the upper Cimarron River watershed for years correlated with pre and post-impact water flow from Figure 2. Pre-impact includes years 1966-1985 and post-impact includes years 1986-2012. Precipitation data were collected from the National Oceanic and Atmospheric Administration's National Climatic Data Center's gauges. Gauges include: USC00140802 (Big Bow, KS), USC00142432 (Elkhart, KS), USC00057992 (Stonington, CO), USW00003028 (Springfield Comanche National Grassland, CO), USC00051268 (Campo 7, CO), USC00147922 (Sublette 7, KS), USC00146813 (Richfield 10, KS), USC00148287 (Ulysses 3, KS), USC00144695 (Liberal, KS), USC00143855 (Hugoton, KS), USC00058793 (Walsh 1, CO), USC00144114 (Johnson, KS), and USC00146808 (Richfield 1, KS). Data are based on calendar year. Error bars are ± 1 standard deviation.

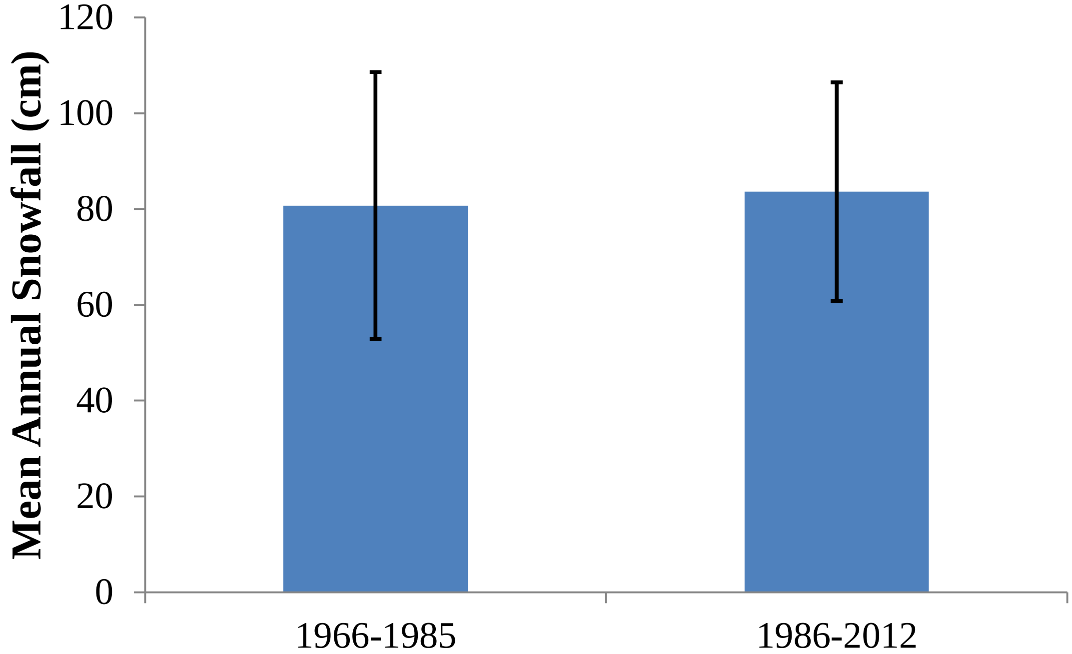

Figure 8. Mean annual snowfall in the upper Cimarron River watershed for years correlated with pre and post-impact water flow from Figure 2. Pre-impact includes years 1966-1985 and post-impact includes years 1986-2012. Precipitation data were collected from the National Oceanic and Atmospheric Administration's National Climatic Data Center's gauges. Gauges include: USC00140802 (Big Bow, KS), USC00142432 (Elkhart, KS), USC00057992 (Stonington, CO), USW00003028 (Springfield Comanche National Grassland, CO), USC00051268 (Campo 7, CO), USC00147922 (Sublette 7, KS), USC00146813 (Richfield 10, KS), USC00148287 (Ulysses 3, KS), USC00144695 (Liberal, KS), USC00143855 (Hugoton, KS), USC00058793 (Walsh 1, CO), USC00144114 (Johnson, KS), and USC00146808 (Richfield 1, KS). Data is based on calendar year. Error bars are ± 1 standard deviation.

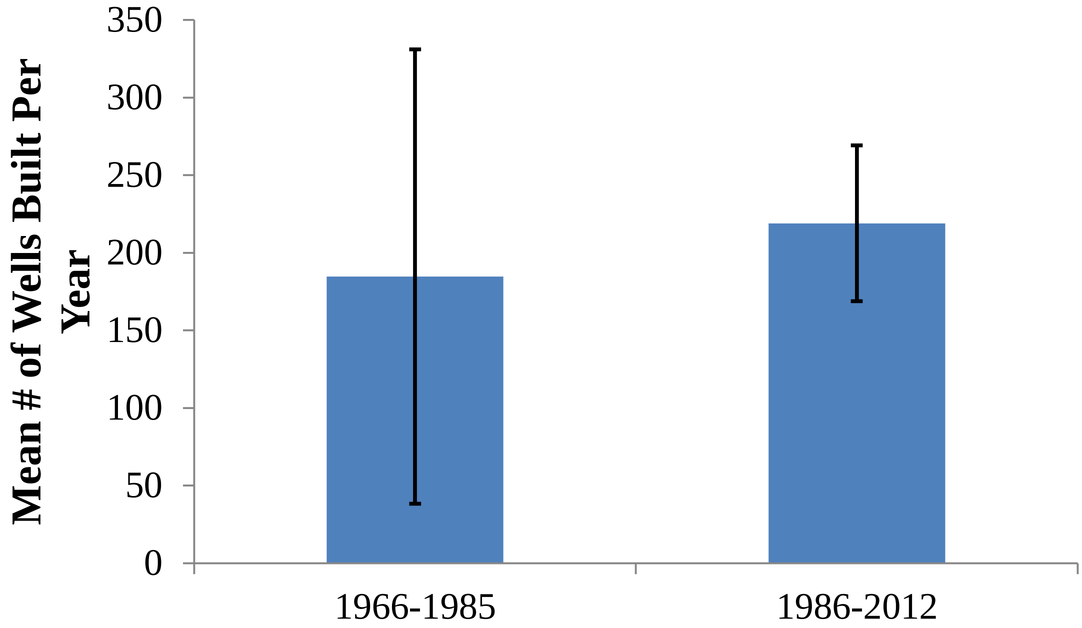

Figure 9. Mean number of wells built per year upstream of Forgan, Oklahoma, within the upper Cimarron River watershed for the pre and post impact periods of discharge represented in Figure 2. Error bars are ± 1 standard deviation.

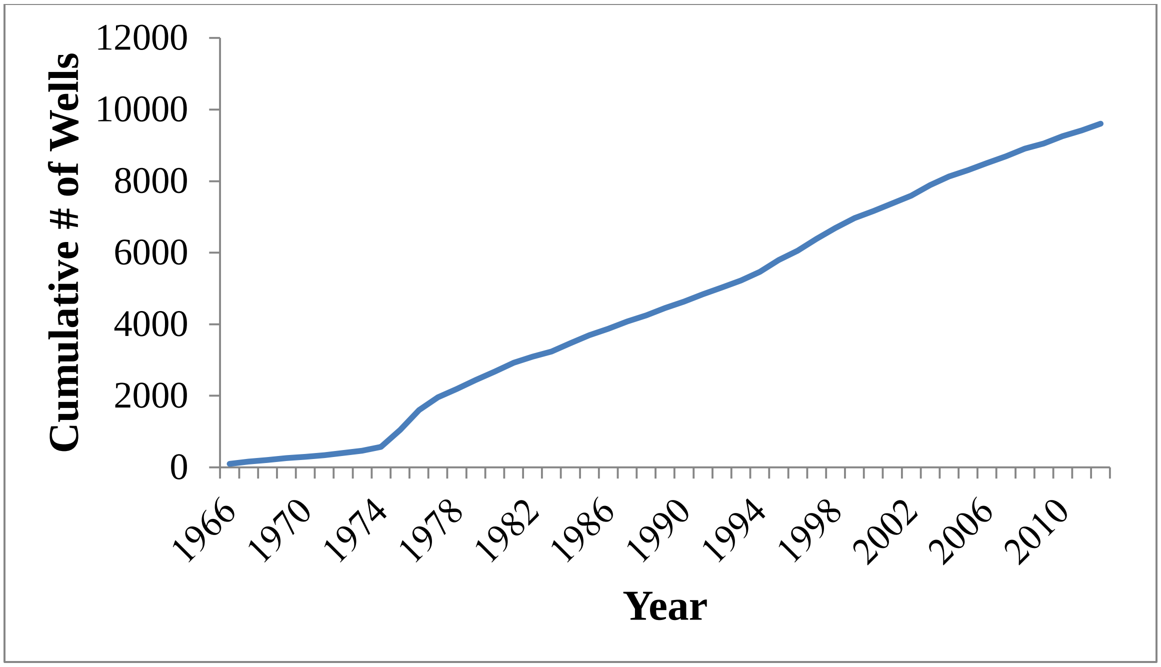

Figure 10. Cumulative number of wells built by year within the upper Cimarron River watershed.

CHAPTER III

THE EFFECTS OF STREAM DIVERSION ON FISH ASSEMBLAGE COMPOSITION IN THE UPPER CIMARRON RIVER

INTRODUCTION

The diversion of water into irrigation canals can have immediate and substantial effects to native fish population. Native fish may become entrained into the canal system with few individuals able to return to the river (Baumgartner *et al.* 2007, Carlson and Rahel 2007, King and O'Connor 2007). Irrigation canals rarely contain natural habitat structure needed to support native fish assemblages (Baumgartner *et al.* 2007, King and O'Connor 2007) and entrained fish typically die as water flows recede during drier months (Carlson and Rahel 2007). Juvenile and larval life stages may be especially susceptible to entrainment into canals because the reproductive cycle of many species relies on passive downstream drift and juvenile life stages generally have poorer swimming capabilities (Baumgartner *et al.* 2007, King and O'Connor 2007). Because canals do not provide suitable habitat for developing drift larvae (Baumgartner *et al.* 2007), entrainment may result in decreased recruitment as young life stages fail to reach nursery habitat (Humphries *et al.* 2002). Entrained fish that survive through an agricultural season in an irrigation canal may then be forced to find refuge in homogenous habitat as water begins to subside during the drier months (King and O'Connor 2007). Additionally, irrigation canals can lead to the proliferation of non-native fish species that do better in these habitats than native species (Cowley *et al.* 2007).

The Cimarron River has been routinely described as a large, undammed river (Reash and Pigg 1990), but this is not entirely accurate. Between 1893-1905, the Settler's Milling Canal and Reservoir Company dug a 22.53 km long canal in what would become Harper County, Oklahoma, to irrigate nearly 2,428 hectares of farmland (USNPS 1982) (Figure 1). The canal is typically 3.7 meters wide at the base, 18.3 meters wide at its banks and fed by an earthen dam that diverts the entire surface flow of the Cimarron River into the canal. The dam is composed primarily of sand and must be rebuilt by bulldozers after floods. This canal is still in use today, with an allocation of surface water permitted by the Oklahoma Water Resources Board (OWRB), and was placed on the National Register of Historic Properties in Oklahoma. The surrounding area fed by this canal is commonly known as "Ditch Valley."

Near Ditch Valley, the Cimarron River is classified as an intermittent stream with high rates of evaporation, infiltration, and dissolved solids (Pigg 1988, Reash and Pigg 1990). It is a relatively unshaded, shallow river with a shifting sand substrate (Pigg 1988, Reash and Pigg 1990) and three main habitat types: long shallow runs, shallow pools, and backwaters with emergent vegetation at the margins (Pigg 1988). During drier months, subsurface flow creates isolated pools and narrow shallow runs that support aquatic organisms (Hargett et al. 1999). In contrast, the canal consists of mud substrate that is relatively homogenous with few pool habitats and virtually no backwaters.

Ditch Valley provides an ideal setting to analyze differences in fish assemblages because it is comprised of four distinct aquatic areas divergent in habitat and water quantity. These include (1) the area upstream of the Old Settler's Irrigation Canal Dam, (2) downstream of the dam within the diverted section of the river, (3) the irrigation canal that receives the diverted stream flow, and (4) the downstream portion of the mainstem Cimarron River after the return from the canal (Figure 1). I expected water flow to be substantially different in each of these areas and that this would result in identifiable differences among the fish assemblages. My objective was to compare the fish assemblages of the four different sections of Ditch Valley to elucidate the effect of water quantity.

METHODS

Except for the upstream section, three 100-m sampling sites were selected in each of the distinct sections (upstream, diverted river, canal, and downstream) (Figure 1). Restrictions in land access allowed only one sampling site in the upstream section. Each site was sampled in May, July, September, and December, 2012, and in June, 2013. Within each site, habitats were visually identified as backwater (BW), shallow-fast (SF), or shallow-slow (SS) (Utrep and Fisher 2006). I use a 3.05 x 1.22-m seine comprised of 3.81-cm delta mesh to sample eight different habitats within each sampling site (Utrep and Fisher 2006). Each habitat sampled received two consecutive seine hauls to ensure that rare species were captured (Utrep and Fisher 2006). The eight habitats sampled included all BW habitats with the remainder of the sampling distributed among the other represented habitat types (SF and SS). Captured fish were preserved in 50% isopropyl alcohol, transported to the lab, identified to species, and counted.

Discharge, water temperature, salinity and dissolved oxygen (DO) were measured at each site by using a YSI model 85 handheld oxygen, conductivity, salinity, and temperature system. Discharge (Q) was calculated as Q=V x A where V is the velocity and A is the cross-sectional area of the feature. Cross-sectional area was determined by recording water depth and velocity every meter across the wetted width then multiplying the water depth by wetted width. Velocity (V) was measured with a Marsh and McBirney model 201 portable water current meter.

RESULTS

The upstream sampling site had water more often than other sites and in larger quantities (Figure 2). Water was present in the upstream site four of five sampling months with all others having water in no more than two months (canal = 1, diverted = 2, and downstream = 1). All sampling sites were dry in July 2012 and at no time did all sampling sites have water concurrently.

Temperature increased as water quantity decreased across sampling sites (Figure 3). Except for December 2012, an increase in salinity occurred as water quantity decreased (Figure 4). Water quantity had no apparent effect on DO (Figure 5).

Due to multiple dry periods and a discrepancy in number of sampling sites upstream compared to other sections, sampling sites were aggregated into a single collection record for each section. Trends in fish assemblage mirrored those of water quantity. The upstream site had the highest number of species (9), followed by the canal (6), downstream (4), and the diverted section (3) (Figure 6). Red River pupfish (*C. rubrofluviatilis*) and plains killifish (*F. kansae*) were present in all sample sites when any fish were caught, with the exception of the diverted section in December 2012, when only plains killifish were caught (Tables 1-4).

DISCUSSION

Sampling sites were dry more than 50% of the time, resulting in limited fish collections. In 2012, Kansas and Oklahoma were in the worst drought in 56 years, the 5th worst on record (Masters 2012). While this made it difficult to catch fish at many sites, it improved collection efficiency, making comparisons easier. Furthermore, it identifies a worst case scenario that is becoming increasingly prevalent in this area of the United States due to loss of stream flow (Chapter 2).

Diversion of water into the canal clearly had an effect on water quantity and fish assemblage composition, but less than expected from simple water availability. For example, during May 2012 water was present only in the upstream segment and the canal. Discharge in the canal was less than half of discharge in the upstream segment of the river, suggesting that approximately half of the water was lost to either seepage or water withdrawal before discharge was measured at the first sampling site 6.5 km downstream. Only half of the species captured were shared between the upstream and canal sampling sites, with suckermouth minnow only found in the canal and red shiner, western mosquitofish, fathead minnow, and bullhead minnow found only in the upstream site.

Although habitat and habitat-mediated predator-prey dynamics may partially explain differences in fish presence between the canal and river, differential sampling efficiency likely also played a role. The canal is wedge-shaped with deep (0.5 m) unconsolidated silt substrates and nearby riparian vegetation that can serve as a source of food inputs. In contrast, the river is broad and shallow (<0.1 m) with shifting sand substrates and limited access to riparian vegetation. These differences in habitat also affect sampling efficiency. It was relatively easy to move a seine rapidly through river sections. Fish trying to move ahead of the net were trapped, whereas the sediment layers of the canal prevent rapid movement and allowed many fish to escape capture.

The presence of suckermouth minnows in the canal seems related more to prey availability than habitat suitability. Suckermouth minnows are typically found in riffles over sand or gravel substrate (Miller and Robison 2004, TSU 2013), which is more prevalent in the river. Substrate in the canal consisted of loose, unconsolidated silt that, in places, was approximately 0.5 m deep. More likely, the canal offers abundant food resources compared to the Cimarron River upstream. The confined channel, with proximity to abundant vegetation and input of agriculture waste, provides sufficient quantities of detritus on which chironomids and dipterans feed (Hammond 2009), which, in turn, are primary prey for suckermouth minnow (TSU 2013).

The unusually high percentage of dry sites does allow one to gain some perspective on fish re-colonization in this section of the Cimarron River, especially in light of the numerous water diversion structures. Typically, the entire flow of the Cimarron River is diverted into the irrigation canal, dewatering the diverted portion and blocking fish passage upstream. A series of concrete barriers in the return portion of the irrigation canal also prohibits upstream fish passage during base flow. During high flow events, however, the dam is washed out; renewing the hydrologic connection and allowing unrestricted fish passage. When the dam is rebuilt after a spate, re-colonization occurs from isolated habitats.

During 2012, the dam was intact for all samples. In June 2013, the dam had been recently breached and flow had returned to all sections of the river. Based on local hydrograph records, this breach most likely occurred on June 10, 2013 (USGS 2014). This was the only time fish were found in all sections of the river and all sites had similar communities, although the upstream site had the greatest amount of diversity in concert with water quantity. Interestingly, the downstream site contained more Red River pupfish than plains killifish, whereas the opposite trend was observed at the two upstream sites.

The plains killifish (*Fundulus zebrinus*) and the Red River Pupfish (*Cyprinodon rubrofluviatilis*) typically occur together, but their abundance varies based on salinity levels (Echelle *et al.* 1972). At lower salinity levels, the plains killifish abundance exceeds that of Red River pupfish, but as salinity levels rise, the Red River pupfish dominates the fish assemblage (Echelle *et al.* 1972). Because water quantity influences other properties of water, such as temperature (Mas-Martí *et al.* 2010) and salinity (Connor *et al.* 2012), changes in water quality can alter species interactions (Stromberg *et al.* 2007). The downstream site had less than a tenth of the water available at other sites, resulting in a substantial increase in salinity. Salinity levels most likely reached a threshold in which Red River pupfish were able to dominate the species assemblage.

In general, although hampered by the drought that limited water availability, I observed that species richness was positively correlated with water quantity. Moreover, water quality (e.g., salinity) as influenced by water quantity further affected fish assemblage structure. However, sampling efficiency complicated findings related to the canal and additional research on this topic in these systems would be beneficial.

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Tables and Figures

Table 1. Species abundance in the Cimarron River immediately upstream of the Old Settler Irrigation Canal Dam, Beaver County, Oklahoma 2012-2013. No fish were captured during July 2012, because the river was completely dry.

		2012				2013
Common Name	Scientific Name	May	July	Sept	Dec	June
central stoneroller	*Campostoma anomalum*	10				
red shiner	*Cyprinella lutrensis*	61				
Red River pupfish	*Cyprinodon rubrofluviatilis*	107		112	1	70
Arkansas darter	*Etheostoma cragini*	4				
plains killifish	*Fundulus kansae*	387		382	48	299
western mosquitofish	*Gambusia affinis*	27		8		21
sand shiner	*Notropis stramineus*	213				25
fathead minnow	*Pimephales promelas*	42				
bullhead minnow	*Pimephales vigilax*	3				
unknown YOY						8

Table 2. Species abundance in the Old Settler Irrigation Canal, Harper County, Oklahoma 2012-2013. No fish were captured during July, September, and December, 2012, as well as June, 2013, because the canal was completely dry.

		2012				2013
Common Name	Scientific Name	May	July	Sept	Dec	June
central stoneroller	*Campostoma anomalum*	1				
Red River pupfish	*Cyprinodon rubrofluviatilis*	37				
Arkansas darter	*Etheostoma cragini*	1				
plains killifish	*Fundulus kansae*	53				
sand shiner	*Notropis stramineus*	30				
suckermouth minnow	*Phenacobius mirabilis*	5				
unknown YOY		3				

Table 3. Species abundance in the diverted section of the Cimarron River downstream of the Old Settler Irrigation Canal Dam, Beaver and Harper Counties, Oklahoma 2012-2013. No fish were captured during May, July, and September, 2012, because the river was completely dry.

		2012				2013
Common Name	Scientific Name	May	July	Sept	Dec	June
Red River pupfish	*Cyprinodon rubrofluviatilis*					553
plains killifish	*Fundulus kansae*				2	1112
western mosquitofish	*Gambusia affinis*					2
unknown YOY						69

Table 4. Species abundance in the Cimarron River downstream of the return of water from the Old Settler Irrigation Canal, Meade County, Kansas 2012-2013. No fish were captured during May, July, September, and December, 2012, because the river was completely dry.

		2012				2013
Common Name	Scientific Name	May	July	Sept	Dec	June
Red River pupfish	*Cyprinodon rubrofluviatilis*					2084
plains killifish	*Fundulus kansae*					1630
western mosquitofish	*Gambusia affinis*					39
sand shiner	*Notropis stramineus*					38

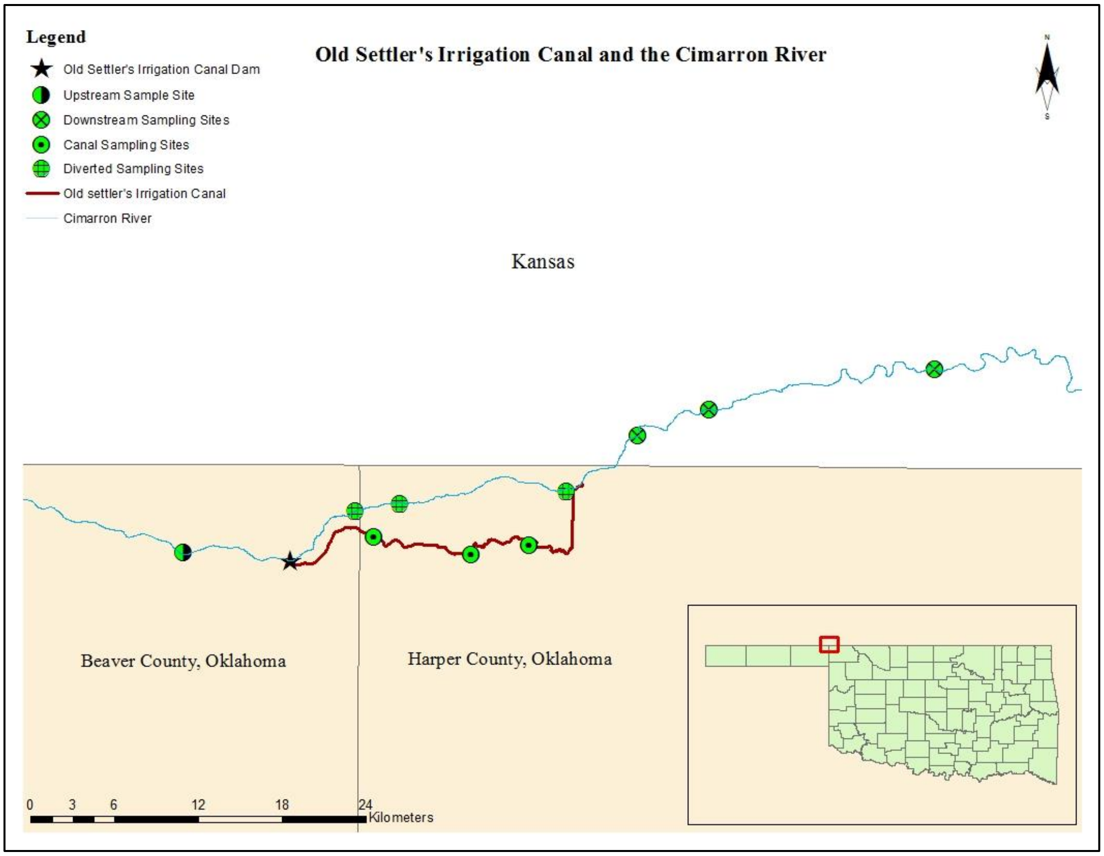

Figure 1. Map depicting sampling sites in the Ditch Valley portion of the Cimarron River and the Old Settler's Irrigation Canal, with legend.

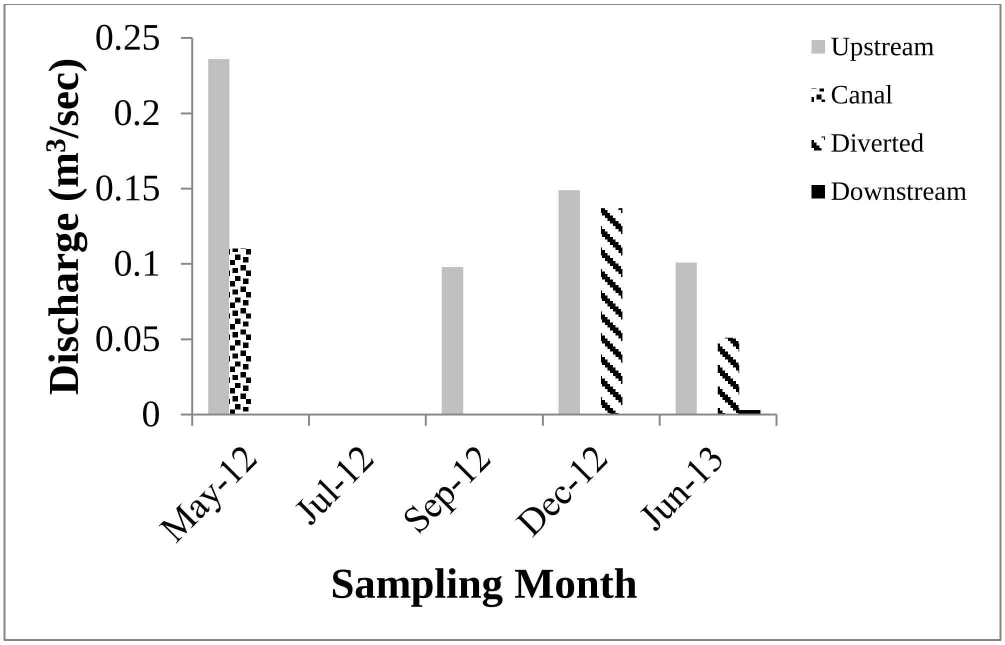

Figure 2. Water discharge among sampling sites in the Cimarron River and Old Settler's Irrigation Canal, 2012-2013.

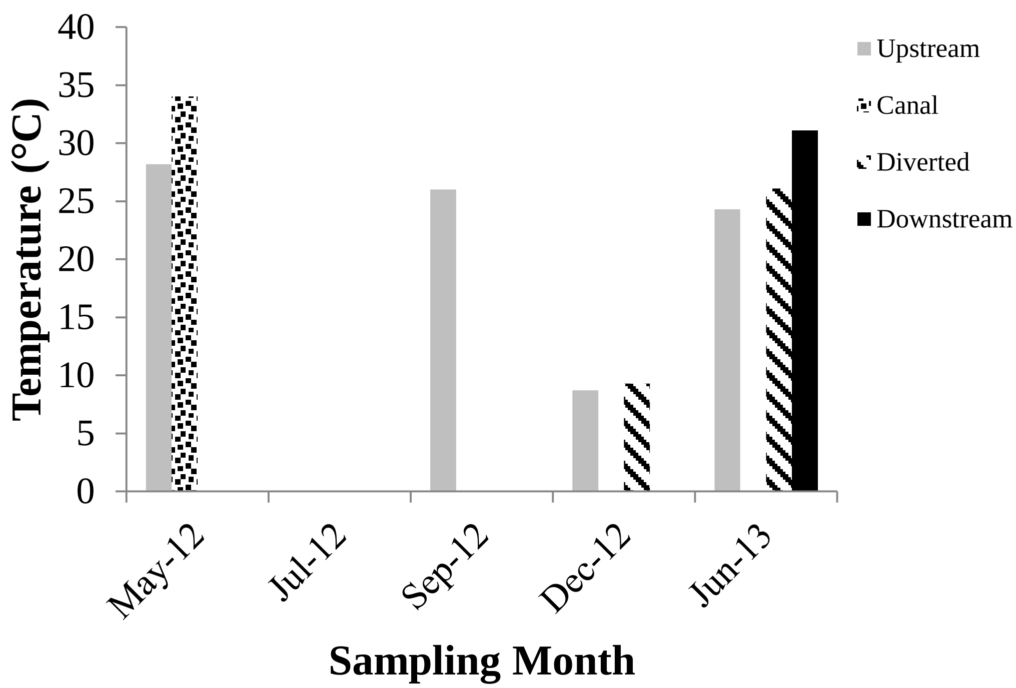

Figure 3. Water temperature among sampling sites in the Cimarron River and Old Settler's Irrigation Canal, 2012-2013.

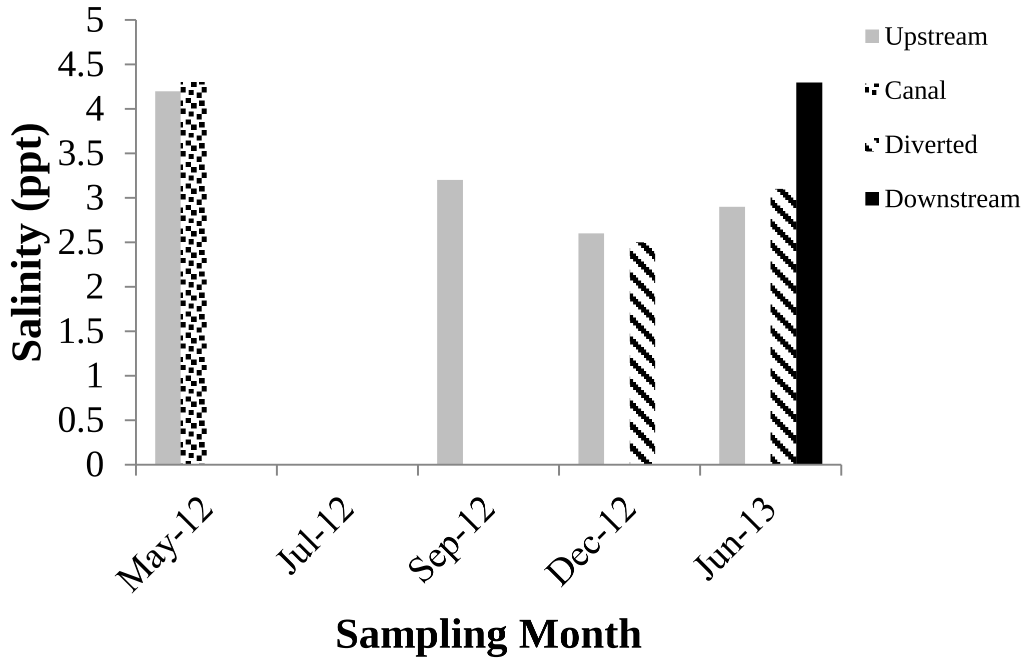

Figure 4. Water salinity among sampling sites in the Cimarron River and Old Settler's Irrigation Canal, 2012-2013.

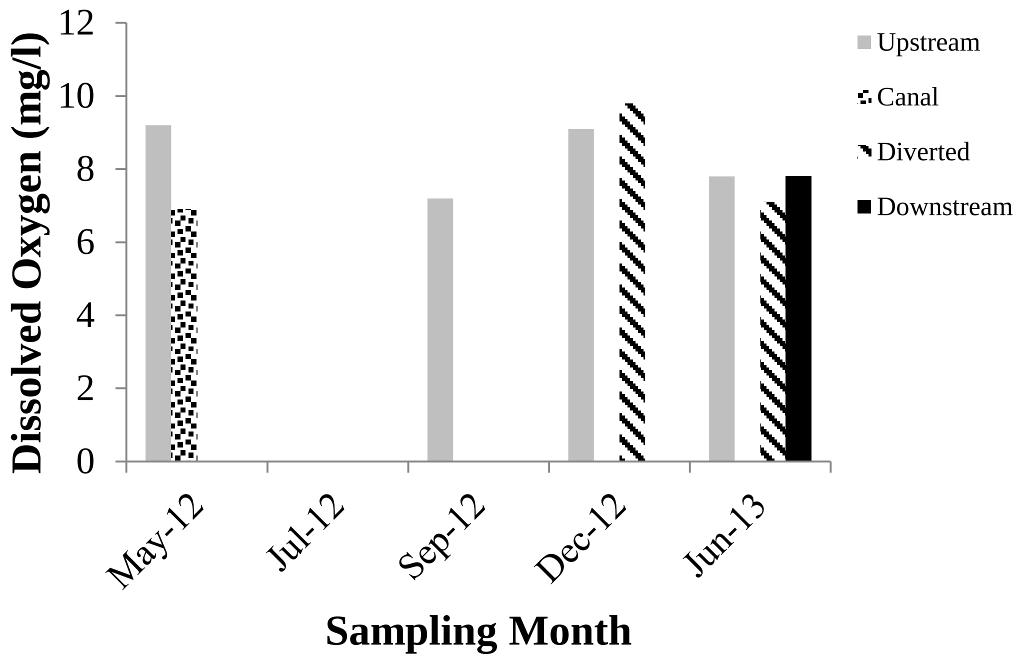

Figure 11. Dissolved oxygen content among sampling sites in the Cimarron River and Old Settler's Irrigation Canal, 2012-2013.

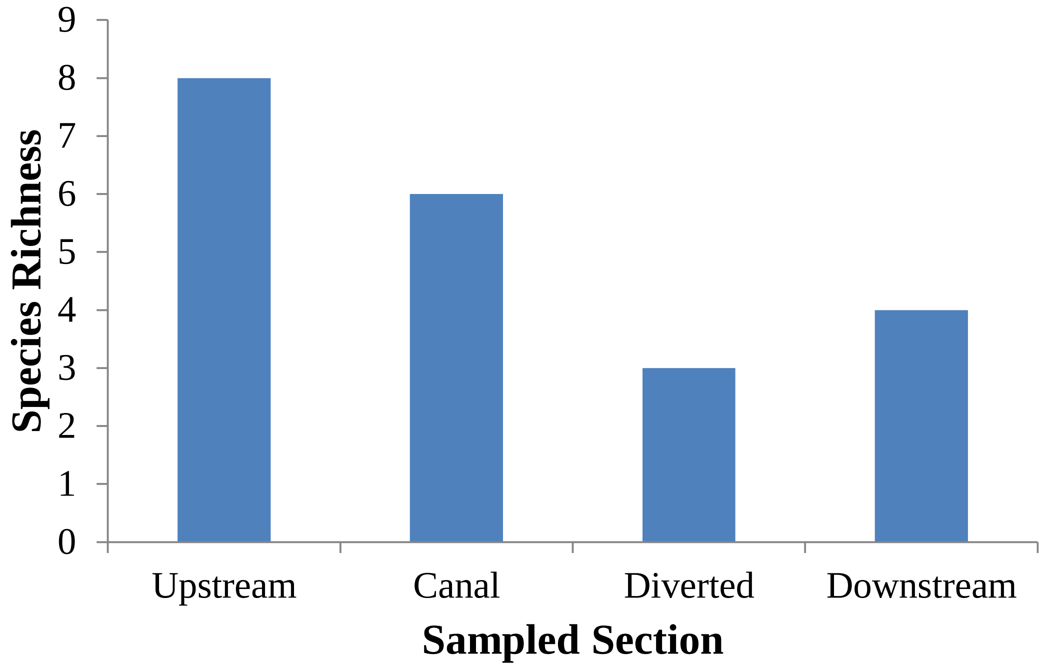

Figure 6. Species richness among sampling sites on the Cimarron River and in the Old Settler’s Irrigation Canal, 2012-2013.

VITA

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