Investigating Potential Wetland Development in Aging Kansas Reservoirs

By

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Abstract

Reservoirs around the world are losing their storage capacity due to sediment infilling; and with this infilling, the quality or value of some reservoir uses such as boating, fishing and recreation are diminishing. However, the sediment accumulating in the upper ends of reservoirs, particularly around primary inflows with well-defined floodplains, could potentially be developing into wetland ecosystems that provide services such as sediment filtration, nutrient sequestration, and habitat for migratory birds and other biota. The objectives of this study are as follows: 1) use water level management data and topography to delineate the primary zone of potential wetland formation around the reservoir perimeter, 2) examine the relationship between ground slope in this area and wetland delineations found in the U.S. Fish and Wildlife Service National Wetlands Inventory (NWI), and 3) investigate if these potential wetland locations have water quality, sediment, and vegetation that indicate a wetland ecosystem. To achieve these objectives, high quality LiDAR elevation data and bathymetry data were used to create reservoir basin topography for 20 large federally operated reservoirs in the state of Kansas located in the central U.S. Historical reservoir water surface elevation data were used to determine the water level and inundation extents associated with the 25th (dry), 50th (normal) and 75th (wet) water surface elevation percentiles for each reservoir, and the area between the 50th and 75th percentile boundaries was used to define the zone of potential wetland formation. Field work was also conducted to collect water, sediment, and vegetation samples from these potential wetland areas. Results showed that using the median slopes of the NWI yielded potential wetland development areas within the upper fluctuation zone (the area between the 50th -75th percentiles excluding areas of zero slope) that were comparable to the NWI coverages for each reservoir, and that slopes between 4.9 and 7.7 produced similar NWI coverages throughout this dataset. The results

also indicated that four water quality variables (Total Nitrogen, Total Suspended Solids, Volatile Suspended Solids, and Turbidity) were unique in the riverine sites compared to the main basin sites, and when all the variables were analyzed cumulatively the main basin sites and the riverine sites grouped together, except for two main basin sites and two riverine sites.

Acknowledgments

First and foremost, I'd like to thank my family and friends. For without their love and support this thesis would never have been written. I'd also like to thank my committee members along with Debbie Baker, Jude Kastens, Bob Everhart and the rest of the Kansas Biological Survey for all the guidance and support throughout this endeavor. Lastly, I'd like to thank the EPA Region 7 office and the Kansas Water Office for supporting and funding this research.

Dedication

I'd like to dedicate this thesis to my loving parents. To my Mom, who thinks that wetlands are defined by crackly dirt, and to my Dad, whose eyes glaze over whenever I start talking about statistics. Love you guys more than I love water!

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Chapter 1: General Introduction

Water is a global resource that is slowly becoming depleted and defiled. There are many facets of life that drive the use and management of water including demographic, economic, technological, social, and environmental. However, one thing is certain; the use and management of water should be examined in an all-inclusive manner, as to use this resource sustainably in the best possible way for everyone (Gallopín & Rijsberman, 2000). Climate change has been recognized as changing freshwater systems around the world by more extreme precipitation and droughts, faster snowmelt, and more evaporation; all which impact infrastructure, the availability of water, and aquatic ecosystems (Brooks et al., 2011). It is recognized that climate change has caused an increase of water temperatures, a decrease of dissolved oxygen, and the increase of pollutant toxicity within freshwater systems (Ficke et al., 2007).

Unfortunately, our current water problems not only impact the environment. Water is also incredibly important for humans, from activities ranging from direct consumption, energy production, agriculture, industrial use, and many others. The stress on our water resources will only increase while population growth, power generation, and climate variability increase; with lasting effects on infrastructure, water availability, and aquatic ecosystems (Roy et al., 2012). This growing demand for water calls for increased protection and regulation of freshwaters; including water in streams, natural lakes, man-made reservoirs, and other impoundments.

During the 1950s and 1960s, the United States began building reservoirs across the country mainly for flood control. However, due to population growth, increased water needs, climate change, and less water availability, the purpose of these reservoirs changed to include irrigation, public water supply, hydro-power, and recreation, in addition to flood control. These reservoirs were built with a life-span of approximately 150-200 years; however, due to sedimentation and eutrophication, these reservoirs are now expected to last from 50 to 150 years (William L Hargrove et al., 2010). Many states in the United States have made reservoirs a priority, to decrease the sedimentation deposition and to improve the water quality for the future for economic, public health, environmental, and social concerns (William L Hargrove et al., 2010).

Sedimentation in reservoirs is a problem because reservoirs are losing their storage capacity at an alarming rate, which causes concern for reservoir management practices. Research has shown that sedimentation occurs in all reservoirs, and it is unrealistic to believe that all sedimentation can be avoided; therefore, better management practices must be developed (Haregeweyn et al., 2006). Reservoirs around the world have such high sedimentation rates that they are losing storage capacity as quickly as 1 percent per year (Mahmood, 1987). The worst country for lost storage capacity in reservoirs is China; with approximately 82,000 reservoirs losing their storage capacity at a rate of 2.3 percent per year (Zhou, 1993). Other regions in the world, such as Northern Ethiopia, have serious soil erosion occurring from watersheds that are draining into the reservoirs. For example, the Camaré reservoir in Venezuela only took 15 years to completely lose all storage capacity due to sedimentation (Fan & Morris, 1997; Haregeweyn et al., 2006). Most reservoirs start to be negatively impacted when half of its original volume has been filled by sedimentation, but problems can occur with only a small fraction of lost storage capacity.

Sedimentation not only affects the storage capacity of the reservoirs, but also affects the nutrient budgets within the reservoirs (Fan & Morris, 1997). Eutrophication is a problem because it changes the ecology and functionality of freshwaters and reduces the water quality.

Eutrophication is mostly due to increased levels of nitrogen and phosphorus, which are more abundant in areas of agricultural cropland, although occur with other land use types as well.

All reservoirs are aging and posing management challenges for the future because of sedimentation and eutrophication. The upper most riverine zones of reservoirs are characterized as shallow, light-limited and high nutrient zones, whereas the deeper, clearer water area near the dam function more similarly to natural lakes (Kennedy et al., 1985). Shallow water and high nutrient concentrations are two parameters often found in wetland ecosystems, and it is proposed that as these reservoirs are experiencing increased sediment infilling rates that they are changing hydrologically and ecologically. Based on chemical and physical processes, these riverine areas could be adapting into different ecosystems. The fact that these areas are becoming shallower and have high nutrient concentrations flowing into them could potentially indicate a change from a reservoir ecosystem to a wetland ecosystem.

Wetlands are incredibly diverse and dynamic ecosystems that are often difficult to define. With the lack of one standard definition for a wetland, conservation and management efforts are often quite varied. However, one of the most widely used definitions was written by Lewis Cowardin of the U.S. Fish and Wildlife Service (Cowardin et al., 1979). He states that a wetland is any "land that is transitional between terrestrial and aquatic systems where the water table is usually at or near the surface or the land is covered by shallow water". For the purposes of his particular classification, wetlands must have one or more of the following three attributes: (1) at least periodically, the land supports predominantly hydrophytes, (2) the substrate is predominantly undrained hydric soil; and (3) the substrate is nonsoil and is saturated with water or covered by shallow water at some time during the growing season of each year" (Cowardin et al., 1979).

Wetlands have many ecological goods and services and as a part a reservoir system could potentially improve the overall water quality of the whole reservoir. Wetlands are ecosystems that are dependent on soils, climate, broad-scale and local hydrology, and other physical and biological factors. Unfortunately, before people understood the importance of wetland ecosystems, many important habitats were destroyed and drained, farmed, or converted to other land uses. It has been estimated that more than half of all wetland acreage in the contiguous United States has been lost before any type of wetland regulation was passed (Kusler et al., 1994).

In a literature review conducted by Leira & Cantonati (2008), the effects of water-level fluctuations on lakes were studied, with an emphasis on four main points: 1) physical environment effects, 2) lake biota effects, 3) effects on the ecosystem, and 4) modeling studies. A total of 243 papers on the effects of water level fluctuations published in 130 different journals were included in this study. According to the papers studied that discussed the effects of water level fluctuations on the physical environment, water level changes had drastic effects on lake morphometry and altered the characteristics of the areas in which sedimentation occurred. As drawdown occurred, sediment erosion increased, which may cause sediments to become resuspended, causing potential remediation concerns. Another main change in the physical environment during drawdown is the amount of light penetration. As lakes or reservoirs become shallower, light can reach further into the water column increasing the potential for macrophyte growth and the transition between pelagic primary production into littoral ecosystems.

The effects of water level fluctuations on the ecosystem are most widely seen in shallow water and littoral zones. In these areas, slight changes in water level can change the environment in which sediments are exposed to the air or inundated. Furthermore, the fluctuations of water level in these ecosystems are the main influence that determines the diversity and status of wetland plant communities (Leira & Cantonati, 2008). In Leira and Cantonati (2008), the papers that pertained to the effects on biota were further divided into studies that impacted flora and those that impacted fauna. The studies on flora described the effects found in littoral zones, shallow lakes, and wetlands. The results mostly pertained to the distribution of aquatic macrophytes; with macrophytes being unable to tolerate high water levels, while low water levels promote emergent macrophytes to re-establish. According to these studies, the changes in aquatic vegetation always precipitated changes in fauna. The studies on fauna indicated that water level fluctuations did not directly impact fauna, but impacted fauna more indirectly as habitats changed, especially due to macrophyte changes and sedimentation/resuspension.

Coops et al. (2003) summarizes a workshop that was held in Balatonfured, Hungary in May 2002. This workshop discussed the impact of water level fluctuations on shallow lakes in several countries worldwide. The conclusions of this workshop indicated that water levels in shallow reservoirs are a big factor in how the lake functions. For example, small changes in the water regime can shift the distribution of plant communities and extreme changes in water level could even exceed the limits of biota, which provides the opportunity for different ecosystems to develop. However, if water levels become too low, eutrophication, loss of fishery potential, and contamination from chemicals could become major problems for the lake, unless some sort of management strategy is used. (Coops et al., 2003).

Another example of the effects of water levels on lakes and reservoirs is the Three Gorges Reservoir, in the Chongqing municipality in China. This reservoir faces the same problems as most other reservoirs worldwide, including increased erosion, sedimentation, and eutrophication. However, scientists have noticed that the drawdown of the water level create conditions suitable for wetland development and have thus implemented a large 30-meter water level drawdown between the summer and winter months as part of its flood control management strategy (Willison et al., 2013). These wetlands within the Three Gorges Reservoir are reducing erosion, sedimentation, and improving the water quality of the reservoir and suggest that wetland ecosystems within drawdown zones of reservoirs can provide more benefits than originally thought (Willison et al., 2013).

It appears that the fluctuation of water levels is incredibly important in lake, shallow water, and wetland communities. The effects of water level changes are seen across hydrological, physical, and chemical characteristics. Based on water level fluctuations, along with other factors already discussed (sedimentation and eutrophication), we propose that there is a potential that wetland ecosystems can develop in the upper end riverine zones of the reservoirs. Chapter 2 of this thesis explores the hydrological characteristics of these reservoirs to determine the most probable location for wetlands to develop using the 25th, 50th, and 75th percentiles of historical water level elevations. Chapter 3 of this thesis discusses whether the proposed potential wetland areas determined in chapter 2 are functioning as wetland ecosystems using water quality, sediment, and vegetation data.

With the completion of this current study we hope to increase our understanding of the hydrological, physical, and chemical environments of the study reservoirs, and of the ecosystems found within them. We explore where wetland ecosystems are likely to develop within these reservoirs, as well as identify the characteristics of these potential development areas. With this increased understanding, we hope that better informed decisions can be made about how to manage these systems to make our water supply more sustainable.

Chapter 2: Identifying potential wetland development areas around managed reservoirs using LiDAR

Introduction

While man-made reservoirs and natural lake ecosystems are similar in their water usage, they differ in characteristics such as drainage area and age (Cooke et al., 2016). Soballe and Kimmel (1987) found that the ecological structure and function of rivers, river impoundments, and natural lakes on a broad scale varied along a composite gradient that changed with water residence time, drainage area, water depth, flow, and water clarity. Lakes and rivers occupied opposite ends of this spectrum with reservoirs typically occupying an intermediate position. Biologically it has been noted that impoundments often have fish communities with greater proportions of pollution tolerant species and higher percentages of non-native species than natural lakes (Whittier et al., 2002).

It is generally understood that in natural lakes the spatial variation of many physio-chemical and biological factors are related to shoreline length, depth, and wind-driven currents (Thornton et al., 1981). By contrast, these attributes appear to be of less importance in reservoirs, where the prominent determinants of observed spatial gradients in physio-chemical and biological conditions are large river inflows and depth gradients characteristic of damming the river channel to create an impoundment (Lehner et al., 2011). These upstream-downstream gradients in depth and flow often result in measureable gradients in turbidity, mixing depths, nutrient concentrations, primary production, and fish standing stocks along with other factors (Kennedy et al., 1982; B. Kimmel et al., 1990; B. L. Kimmel & Groeger, 1984; Lind, 1984; Thornton et al., 1981). The uppermost riverine zones of the reservoirs are characterized as shallow, light-limited and high nutrient zones, whereas the deeper, clearer water area near the dam functions more similarly to natural lakes (Kennedy et al., 1985).

As impoundments age, a frequent problem impacting reservoir management and sustainability is sediment infilling (Anton J. Schleiss, 2014). This is particularly problematic in regions where precipitation, soil properties, topography, and land use all contribute to high levels of soil erosion (García-Ruiz et al., 2015; Ziadat & Taimeh, 2013). One type of soil erosion specifically affected by climate change is gully erosion, in which increased and more frequent runoff events create favorable conditions for gully development (Nearing et al., 2004; Poesen et al., 2003). Gully erosion from agricultural lands has been acknowledged as a major supplier of the total sediment loads flowing into reservoirs, indicating the importance of land use on sedimentation rates (Fox et al., 2016). For example, in the Midwestern United States, urban stormwater runoff from a 0.11 km^2 commercial area in Ohio could bring in 64 $*10^3$ kg km^{-2} yr^{-1} of sediment (Piest et al., 1975) , while a 0.30 km^2 corn-cropped landscape in Iowa could contribute 7,499 $*10^3$ kg km^{-2} yr^{-1} of sediment to the riverine zones of reservoirs(Piest et al., 1975; Weidner et al., 1969).

The disproportionately large amount of sediment brought in from cropland is especially important where farming communities make up a substantial portion of a reservoir's watershed. One such area is the state of Kansas located in the central United States, where cropland (approximately 50%) and grassland (approximately 42%) dominate the landscape (Peterson et al., 2004). With this land cover composition, and with its many large impoundments, Kansas is a good candidate for studying the impact of sedimentation on potential wetland formation at the upper ends of reservoirs. These reservoirs have begun to lose more of their water storage capacity due to both the increased rate of sediment infilling resulting from more frequent and larger storm events (deNoyelles & Kastens, 2016; Rahmani, Hutchinson, Hutchinson, et al., 2015) and years since impoundment (deNoyelles & Kastens, 2016; Rahmani, Hutchinson, Hutchinson, et al., 2015). With reduced water storage capacity and high sediment rates in the uppermost ends of many Kansas reservoirs, other impacts such as habitat loss (or gain) and water quality changes are being examined (William Leonard Hargrove, 2008; Juracek, 2015).

One impact of sediment infilling is the increased area of shallow water and total amount of nutrient-rich sediments that go along with it (Cooke et al., 2013). Shallow water and increased nutrient loads are two parameters often found in wetland ecosystems. As these reservoirs are experiencing increased sediment infilling rates due to climate and land use changes, their hydrological and ecological characteristics may change (Raje & Mujumdar, 2010; Singh et al., 2014; Soundharajan et al., 2016).

In order to maintain the storage capacity of the reservoirs, the most immediate management plan is dredging, which is significantly costly. For example, it cost $20 million to remove 2.3 million cubic meters of sediment from John Redmond Reservoir in 2016, which added approximately 3 more years of life for the reservoir (KWO, 2016a). The cost of disposing dredged sediment can also be high, especially if the sediment contains harmful chemicals or trace metals such as arsenic, copper, lead, or mercury (William L Hargrove et al., 2010). A possible alternative management strategy for these reservoirs whose upper end riverine zones are becoming shallower could be to manage these areas as wetlands if wetland characteristics exist. By filtering inflow through wetlands, in addition to improving water quality, this approach could slow the transmission of sediment into the main body of the reservoir where the storage capacity is most needed.

Kansas has lost about half of its natural wetland habitats due to the conversion of wetland areas to agricultural fields, along with surface and groundwater reductions (Fretwell et al., 1996). This project aims to identify potential wetland areas along the perimeter of federal reservoirs in Kansas with the intent of conserving and managing these potential wetland areas for their ecological goods and services to society. The United States Army Corp of Engineers (USACE) offers legal protection to natural wetlands under Section 404 of the Clean Water Act if the wetland has hydrological characteristics, hydrophytes, and hydric soils (Johnson, 1992).

The objectives of this study are to 1) use water level management data and topography to delineate the primary zone of potential wetland formation around the reservoir perimeter, and 2) examine the relationship between ground slope in this area and wetland delineations found in the U.S. Fish and Wildlife Service (USFWS) National Wetlands Inventory (NWI).

Methods and Materials

Study area

The study sites consisted of low slope areas between the 50^{th} and 75^{th} water level elevation percentiles for 20 federally operated reservoirs in the state of Kansas that had both LiDAR elevation data (to represent topography outside the reservoir) and bathymetry data (to represent lake-bottom topography inside the reservoir) (Figure 1). Reservoir watersheds for these 20 reservoirs consisted mostly of grassland (approximately 58% of watershed area) and cropland (approximately 37% of watershed area) (KBS, 2014). Table 1 summarizes general characteristics of these reservoirs.

Table 1: Summary of characteristics for the 20 reservoirs in Kansas examined in this study (KBS, 2014)

Parameter	Maximum	Minimum	Mean	Median
Surface area (km^2)	63.0	4.5	25.0	23.2
Volume (km^3)	0.46	0.02	0.14	0.10
Age (yr)	69	36	51	52
Max depth (m)	23.8	3.6	13.2	12.8
Shoreline length (km)	161.1	28.8	75.3	69.5

Two distinct climates cover Kansas, ranging gradually from a humid climate in the east to a semi-arid climate in the west. Average precipitation rates vary from approximately 1150 mm in the southeastern portion of the state to approximately 500 mm in the far west (Rahmani, Hutchinson, Jr., et al., 2015) The average temperature patterns go from the warmest in the southeast to the coldest in the northwest, with a statewide average low of 0° C in January and an average high of 27° C in July (Goodin, 1995; Rahmani, Hutchinson, Jr., et al., 2015). All of the study reservoirs are located in the central and eastern part of the state, where precipitation ranges from approximately 600-1000 mm annually, the annual average low temperature ranges from 2-8 degrees Celsius, and the annual average high temperature ranges from 11-14 degrees Celsius. However, the watersheds for the central reservoirs extend into the western portion of Kansas where there is less precipitation annually (Goodin, 1995).

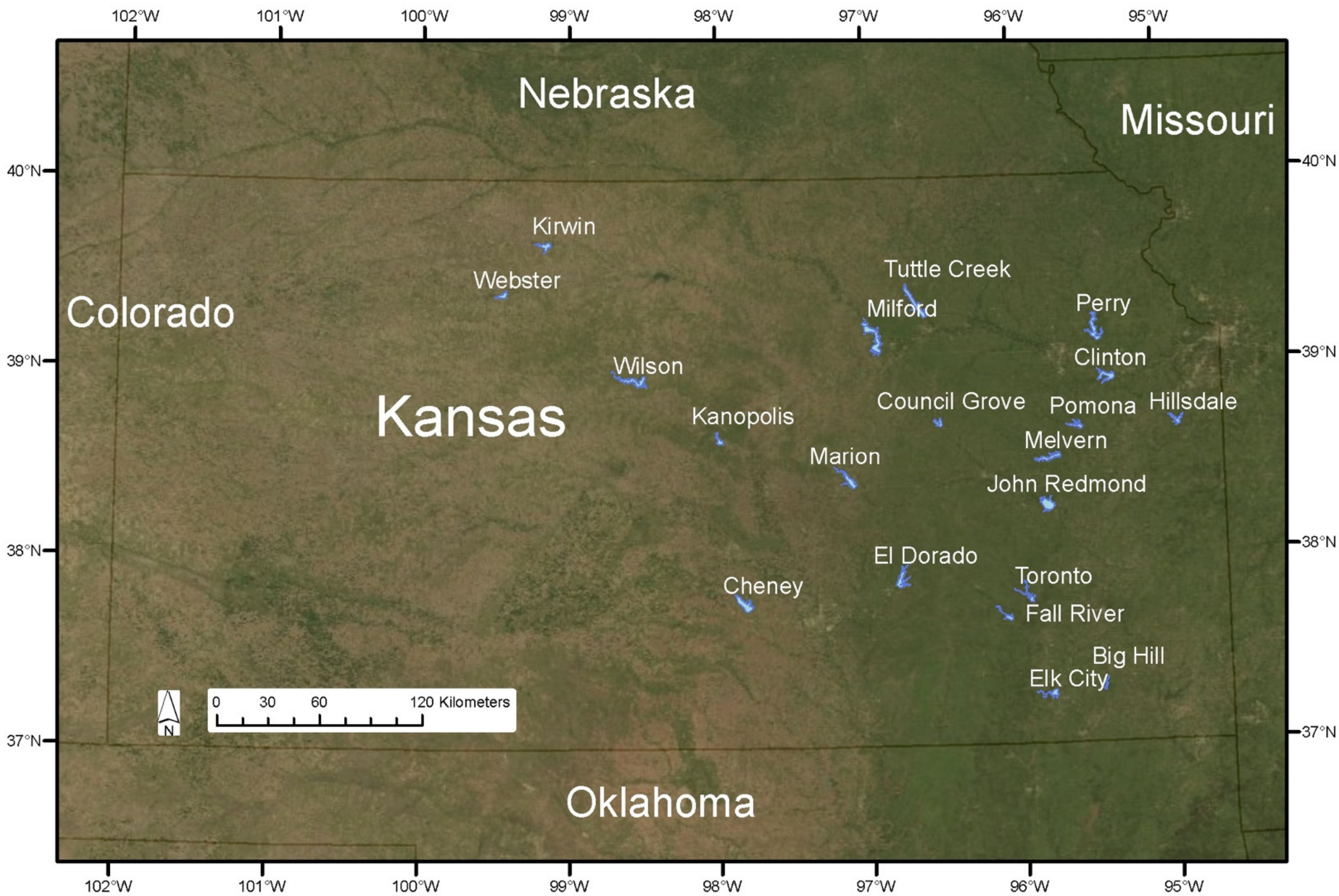

Figure 1: Locations for the 20 study federally operated reservoirs in Kansas. Service Layer Credits: ESRI

The first step was to use historic water level elevation data and reservoir basin topography (a blend of LiDAR digital elevation data and bathymetry information) to determine the zone around each reservoir that captures the range of extents between typical dry conditions (25^{th} percentile water level) and typical wet conditions (75^{th} percentile water level), which we term the *fluctuation zone*. Within the *upper fluctuation zone* (the region between the 50^{th} and 75^{th} percentile water levels) we then compared ground slope values against wetland delineations from the US Fish & Wildlife Service's National Wetland Inventory dataset (NWI; https://www.fws.gov/wetlands/). The second step was to use a slope threshold determined from NWI to identify locations within the upper fluctuation zone with the same or smaller slope and determine if these corresponded with high-likelihood potential wetland development areas. The historical reservoir water level data provided the range of typical water level elevations during wet and dry periods. These data were requested or retrieved from the USACE (USACE, 2017a, 2017b), the U.S. Geological Survey (USGS, 2017), and the U.S. Bureau of Reclamation (USBR, 2017). Data were used starting the day that the reservoir first reached its regulation level through 2015. The reservoir regulation level is defined as the maximum water level elevation during normal operating conditions, without considering flood control storage (NOAA, 2017).

The daily water elevation levels were ordered from lowest to highest, and from this list we calculated the 25^{th}, 50^{th}, and 75^{th} percentile water level elevations over the time period. These calculations provided reasonable estimates for water level during dry, normal, and wet periods, respectively. More explicitly, the 25^{th} percentile lake level corresponds with maximum area that is inundated at least 75% of the time (typical low-water condition), while the 75^{th} percentile lake level corresponds with maximum area that is inundated at least 25% of the time (typical high-water condition). The 50^{th} percentile water level elevation was compared to the regulation level for each reservoir to determine if the median observed lake elevation was near the regulation level. Median lake levels for reservoirs in the eastern part of Kansas (Big Hill, Cheney, Clinton, Council Grove, El Dorado, Elk City, Fall River, Hillsdale, John Redmond, Marion, Melvern, Milford, Perry, Pomona, Toronto, and Tuttle Creek) were found to be close to their regulation levels (namely, within 0.3 m), but the lake levels for a few reservoirs located in the central part of the state (Kanopolis, Kirwin, and Webster) with the exception of Wilson, were considerably lower than their regulation level (0.45 to 4.27 m). Wilson reservoir seems to have a higher average inflow than the other central reservoirs, likely due to groundwater contributions to the Saline River, which allows the water level to be kept near regulation level. Water levels for all of these federal reservoirs are highly managed; however, the reservoirs in the central part of Kansas are less likely to reach their regulation level due to the lack of rainfall and insufficient inflow.

LiDAR and bathymetry analysis

For this study, we used 2-m LiDAR raster elevation data obtained from the State of Kansas GIS Data Access and Support Center (DASC; https://www.kansasgis.org/resources/lidar.cfm). These data are at least Quality Level 3 and have general vertical accuracy of ±18.5 cm with 95% confidence (Heidemann, 2014). Gridded bathymetry data were supplied from studies conducted by the Kansas Biological Survey, USACE, and USGS for the 20 study reservoirs (reports for most can be found at http://www.kwo.org/Reservoirs.html). Reservoir footprints for the study sites were identified in the LiDAR data as hydroflattened extents (which are artificially leveled water surfaces, a common LiDAR post-processing enhancement) and replaced with bathymetrically derived lake bottom elevation so that the elevation dataset represented empty basin conditions. Figure 2 shows an example of the process for Clinton Lake.

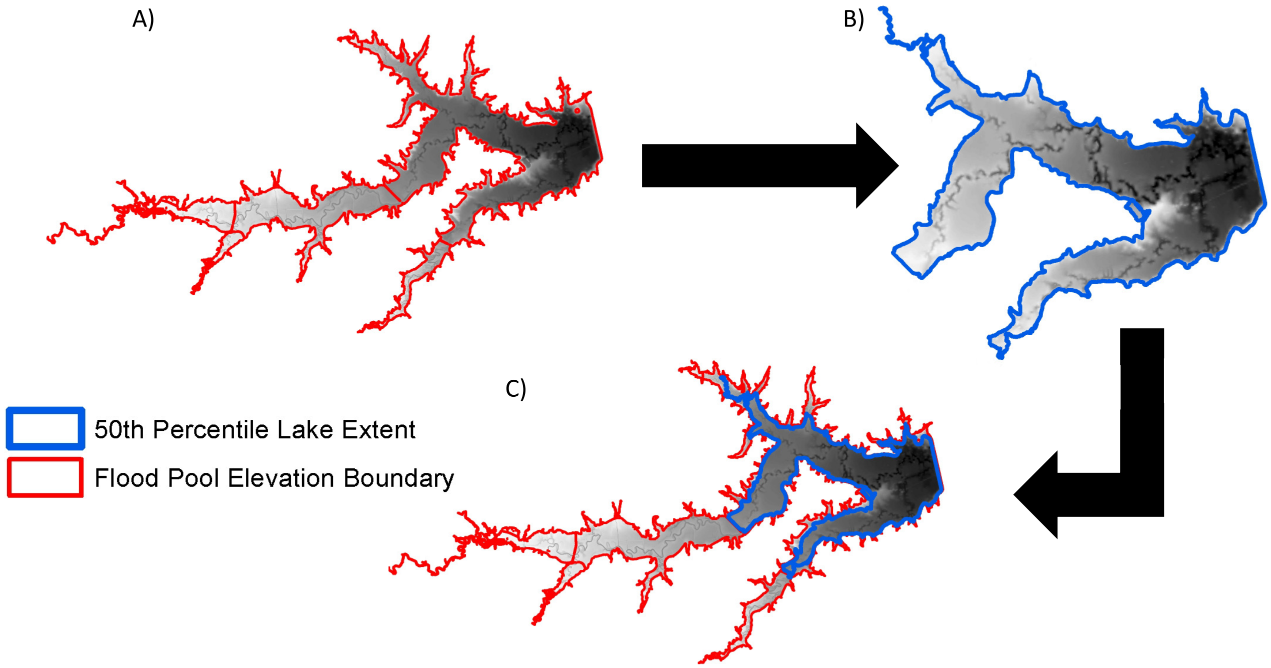

Figure 2: A) 2 meter LiDAR dataset for Clinton Lake, B) Bathymetry dataset for Clinton Lake, and C) Merged LiDAR and bathymetry datasets to create the empty basin topography for Clinton Lake

For each reservoir, a flood pool boundary was developed based on the design specifications for the maximum supported water level (the primary purpose for all of the study reservoirs is flood control). All further analysis was restricted to the interior of these polygons. The hydrological data were then used to create additional boundaries within this basin corresponding with the 25th, 50th, and 75th percentile water level elevations.

Potential wetland development areas

Using ESRI ArcMap 10.3 software, percent-slope values were calculated for the LiDAR topographic datasets and then restricted to the area within the upper fluctuation zones. These areas were further reduced by removing pixel footprints with 0 slope values, which almost exclusively corresponded with hydroflattened locations and are thus unreliable. NWI features were clipped to these same modified upper fluctuation zones, and for each reservoir, the median slope of pixels underlying NWI features was calculated for the whole flood pool boundary (Figure 3).

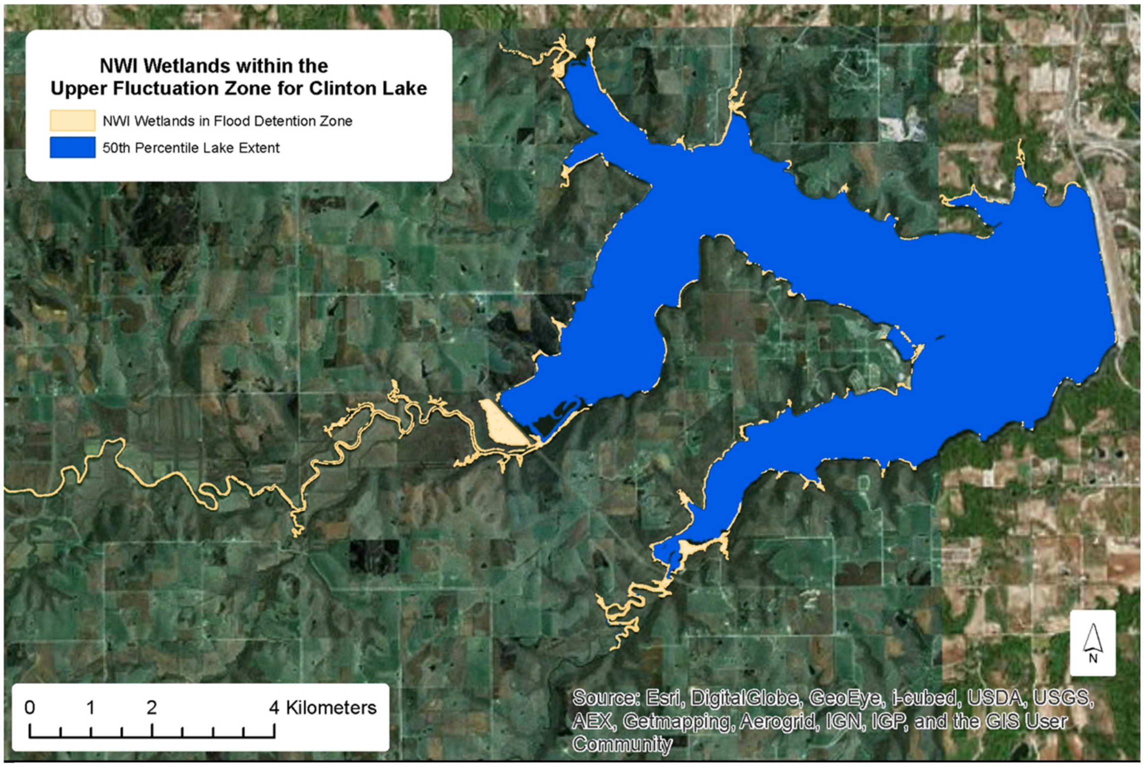

Figure 3: NWI wetlands within the upper fluctuation zone for Clinton Lake, Douglas County

Each median slope provided a threshold value that was used to identify potential wetland areas within a reservoir's modified upper fluctuation zone. Specifically, pixels within this zone that had a slope value less than or equal to the NWI median slope value were retained as potential wetland areas. To determine if there was a more general slope that could be applied across all of the study sites, we calculated the median and average NWI slopes from the modified upper fluctuation zones from all 20 reservoirs and then repeated the above thresholding procedure.

Results and Discussion

Water level fluctuations

The 25th and 75th percentile lake elevations were converted into boundary layers using the same process as the 50th percentile boundary layer. These boundary layers show the coverage areas under typical low-water and high-water conditions, respectively. These boundaries were used to calculate the regime-specific coverage area for each reservoir. Figure 4 presents the coverage areas for Tuttle Creek and Webster reservoirs as two examples of the 25th, 50th, and 75th percentile boundary layers.

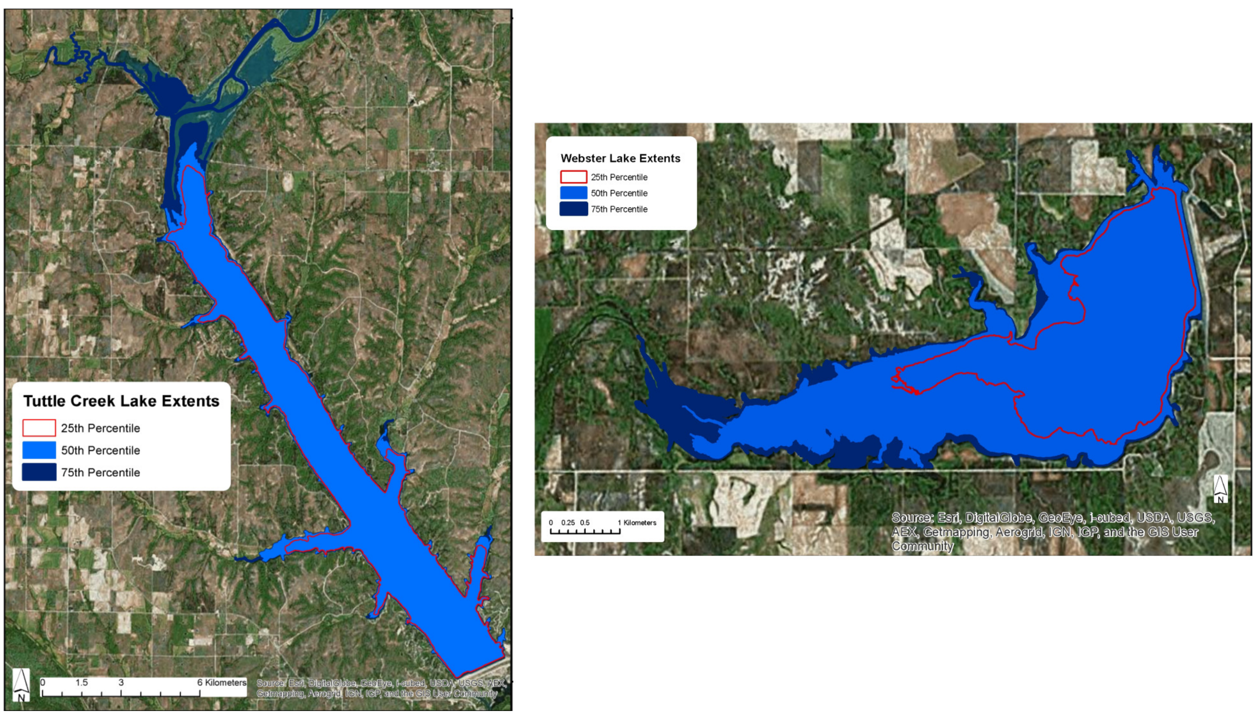

Figure 4: The 25th, 50th, and 75th percentile lake extents for Tuttle Creek and Webster Reservoirs. Source: ESRI

Table 2 shows the coverage area of each of the reservoirs, along with the percentage of area gained or lost relative to the 50th percentile area. The smallest difference between the 25th and 50th percentile lake extent was less than 23 hectares in Fall River Reservoir, while the largest difference was 621 hectares in Kirwin Reservoir. Likewise, the smallest difference between the 50th and 75th percentile lake extent was 2 hectares in Cheney Reservoir, and the largest difference was 727 hectares in Tuttle Creek Reservoir.

The differences seen here between the 25th and 50th and the 50th and 75th percentile water level elevations can be explained by the inflows to the reservoirs and the ground topography. For example, Fall River Reservoir is in the southeastern portion of the state where the highest amount of precipitation occurs. Since the reservoir receives plenty of inflow, it is frequently at its regulation level, causing the 25th percentile water level elevation to also be near the 50th percentile water level elevation. By contrast, Kirwin Reservoir in western Kansas does not receive a lot of precipitation but has high evapotranspiration. Therefore, there are large fluctuations of water level elevations causing a substantial difference between the 25th and 50th percentile water level elevations. Generally, if water is plentiful, maintaining regulation level is easier and high-water fluctuations are more varied. If water is scarce, maintaining regulation level is more difficult and low-water fluctuations are more varied. Other factors such as topography explain deviation from these precepts. As an example of the influence of ground topography, Cheney Reservoir is situated in a steeper-sloped valley than most of the other reservoirs, which makes for a more compact range of lake extents.

Across all the study sites, the ratio of area gained or lost relative to the 50th percentile indicates that from less than 2.5% (Fall River Reservoir) to 45.1% (Webster Reservoir) of area is lost during dry conditions (25th percentile) while from 0.1 % (Cheney Reservoir) to 21.7% (Toronto Reservoir) of area is gained during wet conditions (75th percentile).

Table 2: Calculated coverage for the 25th, 50th, and 75th percentile lake extents and the difference between each of the lake extents for the 20 study federally owned Kansas reservoirs, as well as the percent area lost or gained between each of the lake extents.

Reservoir	Area of 25^{th} Percentile Lake Extent (ha)	Area of 50^{th} Percentile Lake Extent (ha)	Area of 75^{th} Percentile Lake Extent (ha)	Area between 25^{th} and 50^{th} Percentile Lake Extents (ha)	Area between 50^{th} and 75^{th} Percentile Lake Extents (ha)	Percent Area Lost	Percent Area Gained
Big Hill	439	466	470	27	4	5.8	0.9
Cheney	3953	4085	4088	132	2	3.2	0.1
Clinton	2873	2959	3156	86	197	2.9	6.6
Council Grove	1097	1156	1173	59	17	5.1	1.5
El Dorado	2981	3150	3180	170	29	5.4	0.9
Elk City	1317	1426	1616	109	190	7.6	13.3
Fall River	914	937	1004	23	67	2.5	7.2
Hillsdale	1654	1713	1964	59	251	3.4	14.7
John Redmond	3634	3789	3995	155	206	4.1	5.4
Kanopolis	1232	1269	1534	37	264	2.9	20.8
Kirwin	1244	1865	2077	621	212	33.3	11.4
Marion	2519	2646	2665	126	19	4.8	0.7
Melvern	2465	2565	2705	100	140	3.9	5.5
Milford	5864	6213	6444	349	231	5.6	3.7
Perry	4028	4142	4603	114	461	2.8	11.1
Pomona	1309	1390	1535	81	145	5.9	10.4
Toronto	905	934	1137	29	203	3.1	21.7
Tuttle Creek	4079	4363	5090	285	727	6.5	16.7
Webster	688	1254	1471	566	216	45.1	17.2
Wilson	3424	3629	3664	205	36	5.6	1.0

The fluctuation of water levels is one of the most important factors in the establishment of wetlands and a dominant force in controlling littoral lake ecosystems (Poff et al., 1997; Wilcox et al., 1992). Water level variations impact the type of vegetation that is present in the wetland, which also affects the soils found there. Water depth, vegetation, and soil type are the three main features that distinguish wetlands from other types of ecosystems

For example, different types of vegetation are able to thrive, or at least grow, in different zones of flooding (Keddy & Fraser, 2000). Constant flooding in a wetland is not ideal because emergent plants cannot grow and only submergent plants will be present. The opposite is also true; not enough water and only upland vegetation will be present. Therefore, a mix between constant flooding and no flooding will yield the most biodiversity in a wetland (Keddy & Fraser, 2000). Yearly water level fluctuations create vegetative biodiversity by killing off low-water shrubs during times of high water and allowing dormant species to regenerate from the wetland's seed bank (Keddy & Fraser, 2000). In addition, the extent of water level fluctuations, as well as their frequency and duration, influences the physical processes of a lake; specifically, erosion and sedimentation (Leira & Cantonati, 2008). Since water level fluctuations have such a profound impact on lake ecosystems, it is important to understand how human impacts, such as reservoir water level regulation, affect the reservoir ecosystem as a whole (Coops et al., 2003; Coops & Havens, 2005).

National Wetlands Inventory (NWI)

The NWI was used for the initial identification of wetland habitats within each reservoir's flood pool boundary. NWI polygons were predominately forested/shrub wetlands, followed by emergent wetlands, riverine wetlands, and lastly ponds, all of which are freshwater (Table 3).

Table 3: Percentages of freshwater emergent, freshwater forested/shrub, freshwater pond, and riverine wetlands for the entire flood pool (no parenthesis), the upper fluctuation zone (in parenthesis), and the dominant reservoir groups based on wetland coverage within the upper fluctuation zone (bold) for each study reservoir according to the NWI

Reservoir	Freshwater Emergent	Freshwater Forested/Shrub	Freshwater Pond	Riverine
Big Hill	7.1 (22.2)	81.4 (**74.1**)	11.5 (3.7)	0.0 (0.0)
Cheney	14.3 (**83.3**)	37.7 (16.7)	0.8 (0.0)	47.2 (0.0)
Clinton	61.9 (26.3)	6.9 (10.5)	2.6 (0.0)	27.0 (63.1)
Council Grove	23.0 (23.1)	74.3 (**76.9**)	0.5 (0.0)	2.1 (0.0)
El Dorado	9.1 (22.6)	89.9 (**75.4**)	0.3 (1.8)	0.8 (0.3)
Elk City	6.3 (2.7)	30.9 (50.3)	16.0 (20.1)	46.9 (26.8)
Fall River	7.4 (31.7)	70.0 (**63.5**)	2.8 (0.0)	19.8 (4.8)
Hillsdale	3.4 (7.0)	95.3 (**91.1**)	1.3 (1.9)	0.0 (0.0)
John Redmond	15.2 (0.9)	69.7 (**99.1**)	3.4 (0.0)	11.7 (0.0)
Kanopolis	9.3 (0.0)	55.2 (**75.0**)	2.1 (0.0)	33.5 (25.0)
Kirwin	39.9 (12.1)	59.6 (**87.7**)	0.5 (0.2)	0.1 (0.0)
Marion	21.8 (19.2)	76.7 (**80.8**)	1.5 (0.0)	0.0 (0.0)
Melvern	41.1 (**50.0**)	31.5 (25.0)	6.4 (25.0)	21.0 (0.0)
Milford	17.9 (**64.4**)	14.9 (29.7)	0.6 (1.0)	66.5 (5.0)
Perry	46.9 (22.5)	34.6 (35.4)	7.8 (0.3)	10.8 (41.8)
Pomona	24.0 (4.5)	48.9 (**90.9**)	1.9 (0.0)	25.2 (4.5)
Toronto	5.7 (0.1)	86.6 (**99.0**)	7.1 (0.9)	0.6 (0.0)
Tuttle Creek	35.7 (**80.5**)	22.3 (8.3)	2.1 (0.03)	39.9 (11.1)
Webster	35.0 (**60.1**)	58.6 (38.0)	2.7 (0.01	3.7 (1.8)
Wilson	83.3 (**93.8**)	6.2 (6.3)	9.7 (0.0)	0.8 (0.0)
Average	25.4 (31.4)	52.6 (56.7)	4.1 (2.7)	17.9 (9.2)

These four general classes are defined as follows:

1. Emergent – Wetlands characterized by rooted, erect hydrophytes (≈ macrophytes) most of which are perennial species that are present throughout most of the year. These include all types of water regimes except subtidal and infrequently and irregularly exposed systems. The emergent vegetation adjacent to rivers and lakes is often referred to as "the shore zone" or the "zone of emergent vegetation" (Reid & Wood, 1976), and is most often considered separately from the river or lake.
2. Forest/shrub – A complex dominated by water-tolerant shrubs and trees typically located in the flood plain. Palustrine forested and/or palustrine shrub wetlands that in our studies often were dominated by willow (*Salix spp.*) and buttonbush (*Cephalanthus occidentalis*)**.** Cowardin et al. (1979) lists two separate wetland classifications, forested wetland and shrub-shrub wetland; however, the NWI combined these two classifications.
3. Riverine – Includes all wetlands and deepwater habitats contained within a channel, except for wetlands dominated by trees, shrubs, persistent emergents, emergent mosses, or lichens, and saltwater habitats (marine-derived salts > 0.5 ‰). This restrictive system excludes floodplains adjacent to the channel. A channel that conveys water all or some of the time or links two waterbodies can be a natural or artificially created feature.
4. Ponds – Typically small palustrine wetlands with an unconsolidated bottom or palustrine aquatic bed wetlands. Ponds can be natural or man-made and includes farm ponds, stock ponds, small tanks; (generally below 8 ha).

The NWI polygons were restricted to the upper fluctuation zone and showed similar results for the dominant wetland types (Table 3). The largest wetland polygon within the upper fluctuation zone was a 15 ha freshwater forested/shrub wetland polygon in Kirwin Reservoir, and the smallest mapped wetland within this dataset was 0.1 acres, which was found in most of the mapped reservoirs describing different wetland types. We chose to use the NWI because it is currently the only source of delineated wetlands in Kansas.

The exclusion of NWI polygon areas that occurred outside the 75% high water zone resulted in two notable shifts within wetland categories. Both freshwater pond and riverine category percentages were almost reduced by one half indicating that the more upland ponds and the upstream riverine areas were occurring in less frequently flooded regions associated with these reservoir systems. These reductions caused an increase in both the freshwater emergent and forest/shrub categories. The biggest increase was in the freshwater emergent category which seems to comprise the “shoreline” wetland community as these polygons tend to follow the bathymetric/elevational contours. A second wetland cover pattern was also observed in the NWI wetland polygons that occur within the upper fluctuation zone. Based on the dominance of wetland categories there appear to be two groups of reservoirs; 1) reservoirs with predominantly forest/shrub wetlands, and 2) reservoirs with emergent wetlands which appear to be mostly shoreline communities (Table 3). However, a dendrogram created in NCSS of the NWI classes suggests that these 20 reservoirs separate out into potentially four classes (Figure 5).

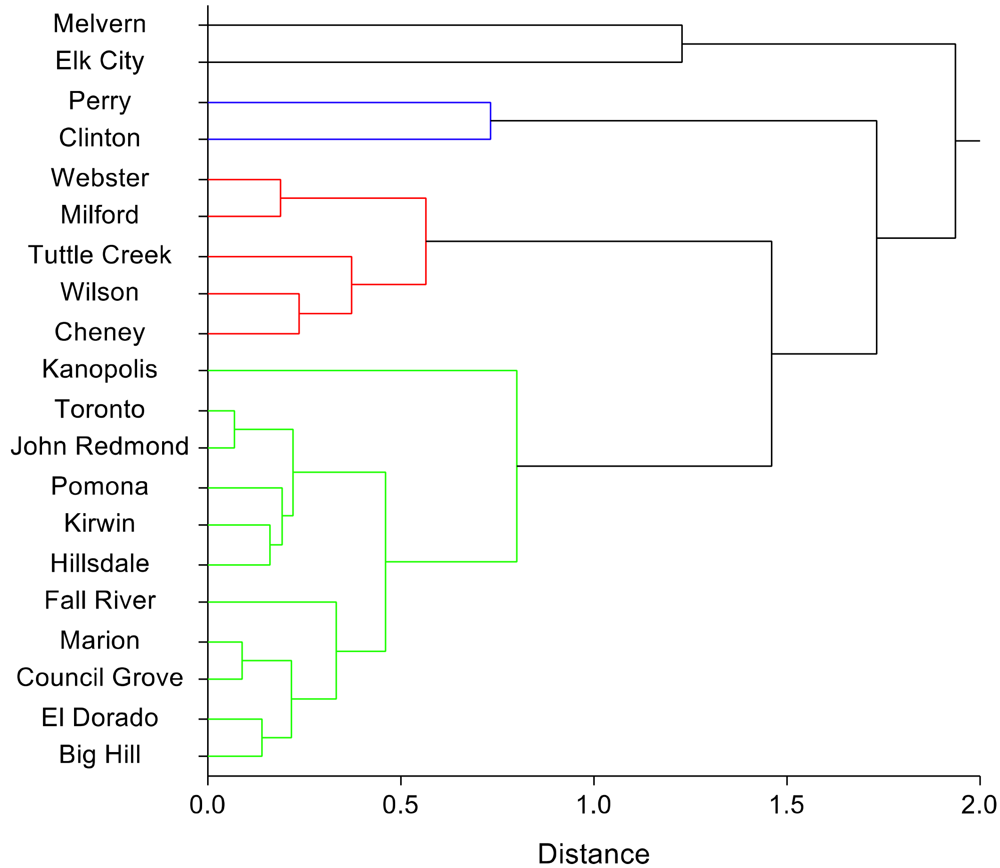

Figure 5: A dendrogram clustering of the NWI for the 20 study reservoirs

It is unclear why the majority of these large reservoirs tend to be dominated by either emergent or forest/shrub wetland communities as they show no obvious ecoregion, basin, spatial, age or gradient/slope pattern. Three reservoirs (Clinton, Elk City and Perry) did not fit well within the emergent and forest/shrub groupings where there was a clear pattern of dominance. Clinton and Perry reservoirs tended to have a dominance of riverine wetlands (63 and 42%, respectively) with the second most common wetland category being emergent for Clinton and forest/shrub for Perry. Elk City Reservoir could have been included with the forest/shrub grouping having just over 50% forest/shrub coverage but it was unique in that sub-dominate categories were pond (20%) and riverine (27%). Lastly there were just four reservoirs that had very limited emergent wetland communities (i.e. shoreline communities). John Redmond, Kanopolis and Toronto reservoirs had less than one percent emergent wetland coverage while Pomona reservoir had less than five percent coverage. Again, there were no apparent reservoir attributes that might help explain coverage extents of these wetland categories.

Potential wetland development areas based on LiDAR

We used the NWI dataset to test if there was a simple way to identify high-likelihood potential wetland areas in the modified upper fluctuation zones using topographical slope. For the reservoirs used in this study, we found that by using the median slope calculated from the NWI wetlands from the whole flood pool boundary as a threshold, the NWI covered an average of 89% of our calculated low slope areas (Table 4), suggestive of a low commission error rate for this approach. From this outcome we could predict locations within an upper fluctuation zone where wetlands are likely to develop based on water level fluctuations and slope. Altogether, the median and average slopes of the NWI wetlands for these 20 reservoirs were 4.9% and 7.7% respectively. The results of using each of these slopes for all the reservoirs yielded similar results to when we used the individual slopes to calculate the percent of low slope area that was covered by the NWI (Table 4). This finding supports the notion that a spatially generalized maximum-slope threshold can be applied to identify high-likelihood potential wetland areas within a reservoir's upper fluctuation zone.

Table 4: Summary of NWI and upper fluctuation zone areas and coverages for all study reservoirs. Areas with 0% slope values were excluded from the calculations.

Reservoir	Median NWI slope (percent rise)	Area of NWI (ha)	Area with slope ≤ median NWI slope (ha)	% of low-slope area (≤ NWI median) covered by NWI	% of low-slope area (≤ 4.9%) covered by NWI	% of low-slope area (≤ 7.7%) covered by NWI
Big Hill	34.7	2	2	60	90	67
Cheney	10.7	15	8	100	100	100
Clinton	5.2	114	63	90	90	89
Council Grove	0.5	6	3	100	100	100
El Dorado	15.1	21	12	83	100	86
Elk City	4.4	174	89	99	100	99
Fall River	4.1	59	30	100	100	100
Hillsdale	5.6	105	63	84	84	83
John Redmond	5.1	205	103	100	100	100
Kanopolis	2.8	66	35	94	90	89
Kirwin	4.2	76	109	35	34	35
Marion	16.9	15	9	89	100	100
Melvern	1.2	115	59	100	99	92
Milford	4.6	168	96	89	88	86
Perry	4.5	182	103	88	89	86
Pomona	2.4	116	59	98	99	98
Toronto	11.9	151	76	100	100	100
Tuttle Creek	1.9	330	178	96	95	95
Webster	5.6	163	107	78	77	76
Wilson	13.1	22	12	100	89	91
Average	7.7	105	60	89	91	89

Summary and Conclusion

Hydrological characteristics of 20 federally operated reservoirs in Kansas were used to determine potential wetland development areas associated with the upper fluctuation zone of each reservoir. We defined the spatial zones between the lake footprints given by the 50^{th} and 75^{th} water surface elevation percentiles as the active, upper region of water level fluctuations using historic daily lake level data and high quality LiDAR topographic data. Within the upper fluctuation zones, we used NWI features to compute maximum slope thresholds useful for identifying high-likelihood potential wetland areas. Two generalized regional thresholds were estimated using all the sites, and both were found to generate little commission error when applied to all of the sites. Thus we conclude that identification of low-slope areas within a reservoir's fluctuation zone can indicate where wetlands are likely to develop. Specifically, we recommend that one use a slope cutoff of 4.9%, but any slope between 4.9% and 7.7% should yield similar results.

Our method for determining areas of potential wetland development is most likely underestimating in some reservoirs and overestimating in others, based upon the range of median NWI slope values within the upper fluctuation zone. The NWI also has some shortcomings of its own. For example, the wetland polygons that we used in this analysis were hand delineated in 1985, and have likely changed over the years. However, despite these shortcomings, our findings suggest that this method is a good way to predict where potential wetlands are highly likely to occur within reservoir ecosystems. Further testing is required to establish if these areas are developing into wetland ecosystems. The next step to advance this study would be to collect water, sediment, and vegetation from the main basin and the fluctuation zones from of these reservoirs. The comparison between these two sites could determine whether the fluctuation zones are indeed developing into wetland ecosystems.

Results from this study and other studies indicate that the methods can be used in other regions to recognize wetland areas. For example, Baker et al. (2006) and Maxa & Bolstad (2009) have indicated that using remote sensing data such as Landsat ETM+ imagery and LiDAR elevation are valuable tools for recognizing wetland ecosystems within landscapes. According to Maxa & Bolstad (2009), LiDAR data for a 63.4 km^2 study area in Wisconsin yielded a 74.5% accuracy in wetland classification, compared to a 56% accuracy of the Wisconsin Wetland Inventory (WWI; http://dnr.wi.gov/topic/wetlands/inventory.html). Most of the misclassification in the WWI occurred while identifying upland and wetland classifications, such as lowland coniferous, evergreen shrub, and moss ecosystems.

Chapter 3: Characterizing potential developing palustrine wetlands within Reservoirs Using Water, Sediment, and Vegetation Data

Introduction:

Reservoirs are highly regulated man-made ecosystems managed to often provide multiple uses such as drinking water, recreation, and flood control (Juracek, 2015). They typically do not function as natural lake ecosystems because of their extensive drainages, artificial geological placement and strict management goals, that result in highly manipulated fluctuations of the reservoir water levels, such that they are classified as their own type of aquatic ecosystem. Reservoirs are important to society and will continue to be important as the water supply decreases and the demand increases. Unfortunately, increased sedimentation into reservoirs are causing these reservors to be unsustainable water sources (Fan & Morris, 1997). Most man-made reservoirs are aging around the world and are experiencing considerable changes in storage capacity, bathymetry and water quality due to sedimentation (Kummu & Varis, 2007). Many of the changes we are seeing within aging reservoirs are most pronounced in the regions of the reservoir near the inflows (i.e. riverine segments). Sediment flowing into these reservoirs are considered a pollutant on its own, but it can also cause eutrophication due to nitrogen and phosphorus adsorbing onto the sediment particulates (Morgan, 2005). Eutrophication is a problem that impacts the entire reservoir, but is specifically noticeable in the shallow inflow regions (Smith et al., 1999).

Sediment loading and accumulation can change the ecology of the reservoir. If sediment increases in these areas, then the open-water habitats will begin to transition into wetland ecosystems, and ultimately into upland habitats (Fan & Morris, 1997)

In regions such the state of Kansas which are mostly comprised of agricultural land and grasslands, it is often difficult to differentiate sedimentation from nutrient loading because both are closely related to these land use types (Carney, 2009). As of 2012, 20 of the federally-operated reservoirs in the state of Kansas had a lost storage capacity from around 2 to 43% (Juracek, 2015). Due to sediment infilling and the general aging processes, we propose that the upper end, riverine areas within the federally-operated reservoirs in Kansas and others could be evolving into different ecosystems based on their changing biological, chemical, and physical processes. We hypothesize that due to sedimentation and eutrophication, these riverine areas are transitioning and may in some cases eventually fully develop into distinct wetland ecosystems within the reservoir ecosystem itself.

The objective of this paper is to determine if the shallower (normally ≤ 1 m) riverine segments of reservoirs are functioning differently than the reservoir's main basin with regards to 1) water quality and 2) sediment quality and 3) plant communities. If portions of these riverine segments are structurally and functionally different than the main basin, then what type of ecosystem are they most similar to? We have identified a number of physical, chemical, and biological variables by which to characterize these shallow riverine reaches as well as main basins attributres in a number of Kansas reservoirs to illustrate possible differences between open-water reservoir conditions (i.e. deep-water lacustrine systems) and the infilling shallow-water riverine reaches. We hypothesize that in many cases these riverine segments are, in part, developing into palustrine wetland systems with distinctly different structural and functional characteristics and features. Currently we have sampled a subset of eight of the federally operated reservoirs in Kansas for various water quality, sediment and vegetation constituents and these were used in the analysis of this chapter (Figure 6).

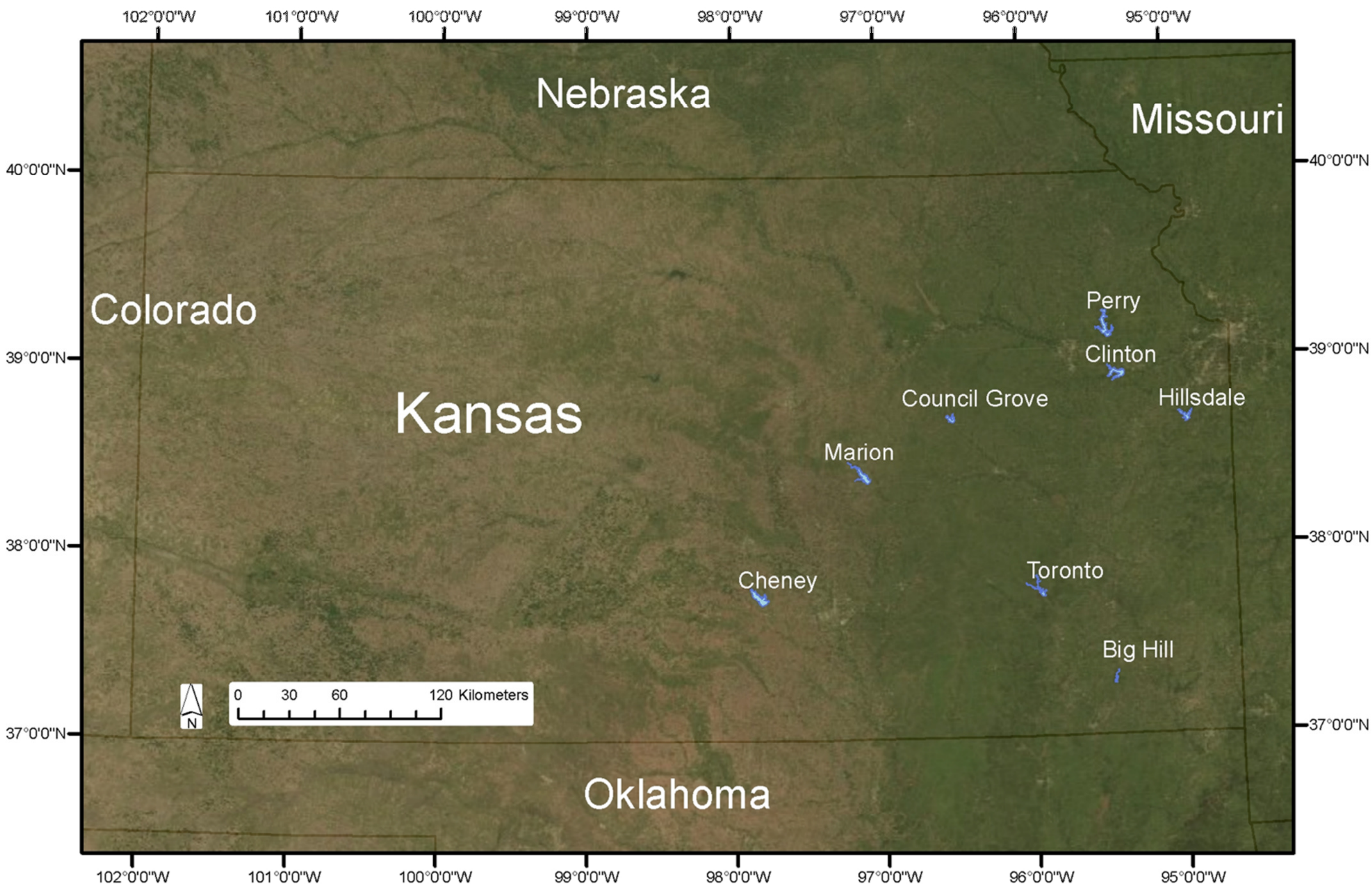

Figure 6: The locations of the eight federally operated Kansas reservoirs used as a subset for this study. Service Layer Credits: ESRI.

Methods:

Water and Sediment:

All water quality and sediment data were collected between September and November 2016. Five subsamples were collected at each sampling site (either a riverine segment or the main basin) based on a double transect configuration with samples taken at the tails and center point of the crossing (Figure 7). The five subsamples for sediment and water quality (except for in situ measurements) were composited to provide one sample per site to reduce sample cost yet still produce a more representive whole site sample. In situ water quality parameters were taken with a Horiba u-52, and the sediment cores were obtained from the top 5 cm of sediment taken with a Wildco® liner-type Hand Corer.

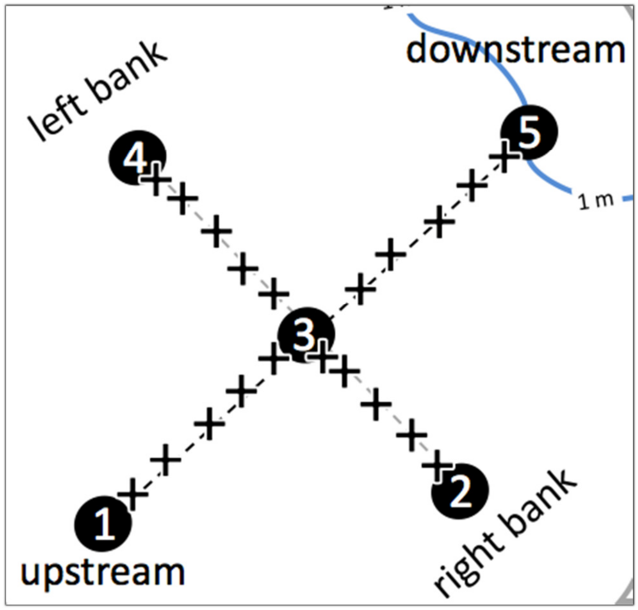

Figure 7: Diagram of the sampling site transects used for water, sediment, and vegetation sampling

Water samples were placed in a 1-liter container, put on ice, and transported back to the Kansas Biological Survey for analysis of total suspended solids (TSS) and volatile suspended solids (VSS) (APHA, 2005). Water samples were also sent to the Kansas State Soil Laboratory for analysis of total phosphorus and nitrogen (RFA Methodology no. A303-S170 and A303-S200-13), the University of Kansas Tertiary Oil Recovery Project (TORP) Laboratory for total organic carbon (TOC) (Standard Method 5310B), and the State Hygienic Laboratory of Iowa for chlorophyll (Standard Method 10200H).

Sediment samples were separately bagged, put on ice, and sent to the University of Kansas Pedology Laboratory for analysis of soil particle size classes (Hydrometer Method), bulk density (Core Method) and total organic carbon (Coulometric Titration Method). Sediment samples were then sent to the Kansas State Soil Laboratory where they were composited and analysed for total phosphorus and nitrogen (Salicylic-Sulfuric Acid Digestion Method). See QAPP for details (http://biosurvey.ku.edu/development-wetlands-aging-reservoirs-opportunities-enhance-wetland-capacity-and-improve-water).

The water quality and sediment data were first averaged by each site to determine a composite sample. The data was then graphically assessed with bar graphs to compare riverine segments to the main basin for each reservoir. Regressions were then run on each parameter comparing the main basin values (i.e. determinate variable) to the mean of the riverine values for

$$r = \frac{\Sigma\ (X_i' Y_i')}{\sqrt{\Sigma({x_i'}^2)\Sigma({Y_i'}^2)}}$$

transformation was conducted to normalize the data. If the data was still not normally distributed after the log transformation, the non-parametric Mann-Whitney test was conducted to see if the main basin and the riverine segments were functioning as two, separate, independent groups (Equation 2).

$$U = n_1 n_2 + \left(\frac{n_2(n_2+1)}{2}\right) - \sum_{i=n_1+1}^{n_2} R_i$$

Where U is Mann-Whitney u test; n_1 is sample size one, n_2= Sample size two, and R_i is rank of the sample size. If the data was normally distributed, an Analysis of Variance (ANOVA) test was run using a General Linear Model (GLM) techique to test for significantdifferences our two sample populations (riverine verses main basion sites) (Dobson & Barnett, 2008). ANCOVAs were also run with total distance from the main basin sites to riverine sites as a covariant to assess for potential influence of water distance as a surrogate for time-in-travel between basin and riverine sites.

Vegetation

The vegetation sampling method used was based on the Kansas Department of Health and Environment's "point quadrant method" (KDHE, 2014). To sample for submerged vegetation, a rake head attached to a rope was thrown over the side of the boat, dragged approximately 5-6 meters and retrieved, following the 2 transect lines depicted in Figure 1. This method is described by Madsen (1999) as a combined point and line intercept method where points (10/transect) were sampled along each of the two crossing transects for a total of 20 points samples per site. Aquatic vegetation was recorded to the lowest taxonomic level in the field, placed on ice, and returned to the Kansas Biological Survey for further identification, when necessary. Vegetative detritus and debris was noted on field sheets as terrestrial vegetation and returned to the water. The results of the vegetation counts were used to calculate percentages for three categories (aquatic vegetation, vegetative detritus and debris, and none).

Results and Discussion:

Water Quality

The ANOVA analysis indicated four water quality variables (total nitrogen, turbidity, TSS, and VSS that were statistically significantly different between the riverine sites and the main basin sites based on their mean values. Figures 8 and 9 indicate the average values of these four water quality variables. The total nitrogen, turbidity, TSS, and VSS values were higher in the riverine sites than the main basin sites for each of the eight sampled reservoirs. The average total nitrogen is more than double, the average turbidity is approximately 2.5 times higher, the average TSS is more than 1.5 times larger, and the average VSS is twice as large in the riverine sites than in the main basin sites. Since the TSS values were on average approximately 3.5 times larger than the VSS values (average TSS =29.19 mg/l, VSS = 7.65 mg/l), it was apparent that nearly all of the TSS consisted of inorganic suspended solids (ISS). Another way to illistrate the significant differences shown in the ANOVAs is by compairing box plots of the data (Figures 10-11). Again riverine sites have noticeably higher TN and turbidy values than main basin sites especially once outliers are accounted for in the box plots. While ISS values for both riverine and basin sites are highly variable, VSS (i.e. suspended organic matter) values are clearly higher in the riverine segments (Figures 9 and 11).

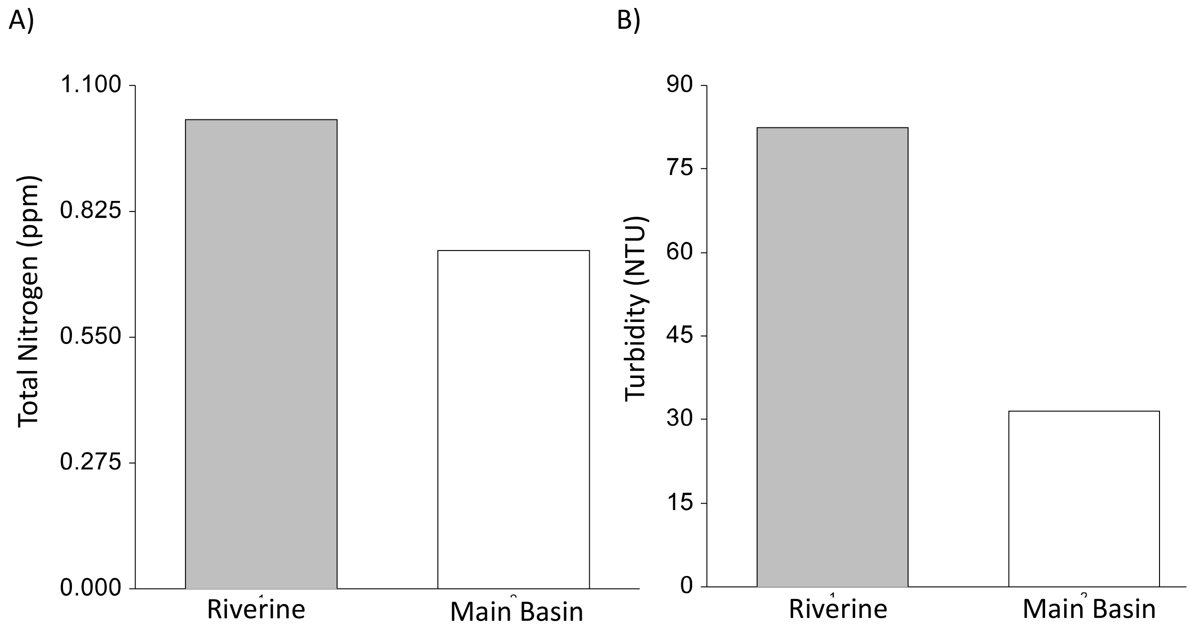

Figure 8: Average values for a) total nitrogen and b) turbidity for the main basin and riverine site

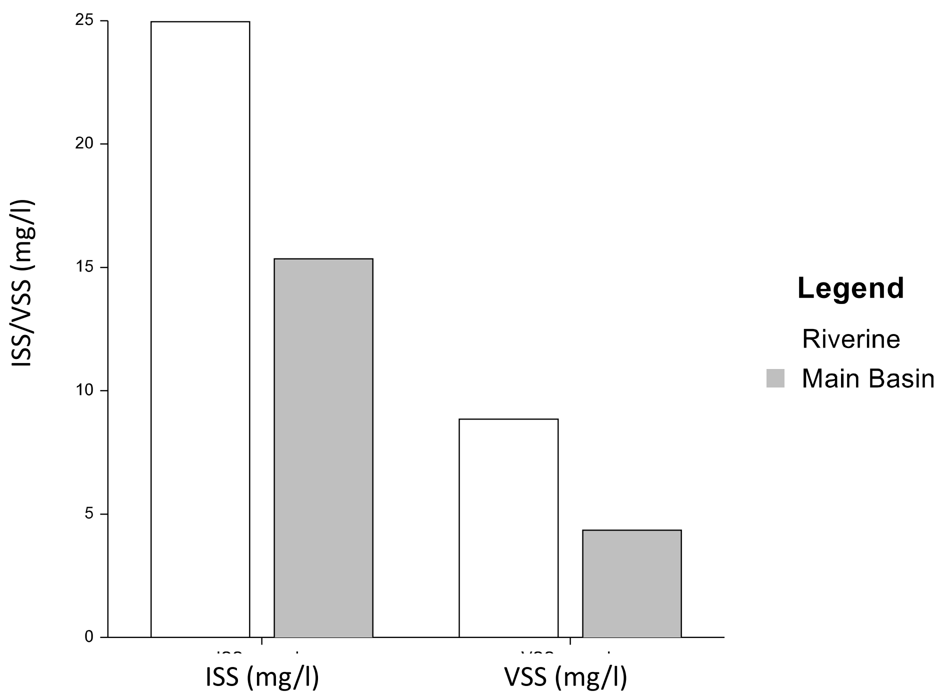

Figure 9: Average values for ISS and VSS values for the main basin and the riverine sites

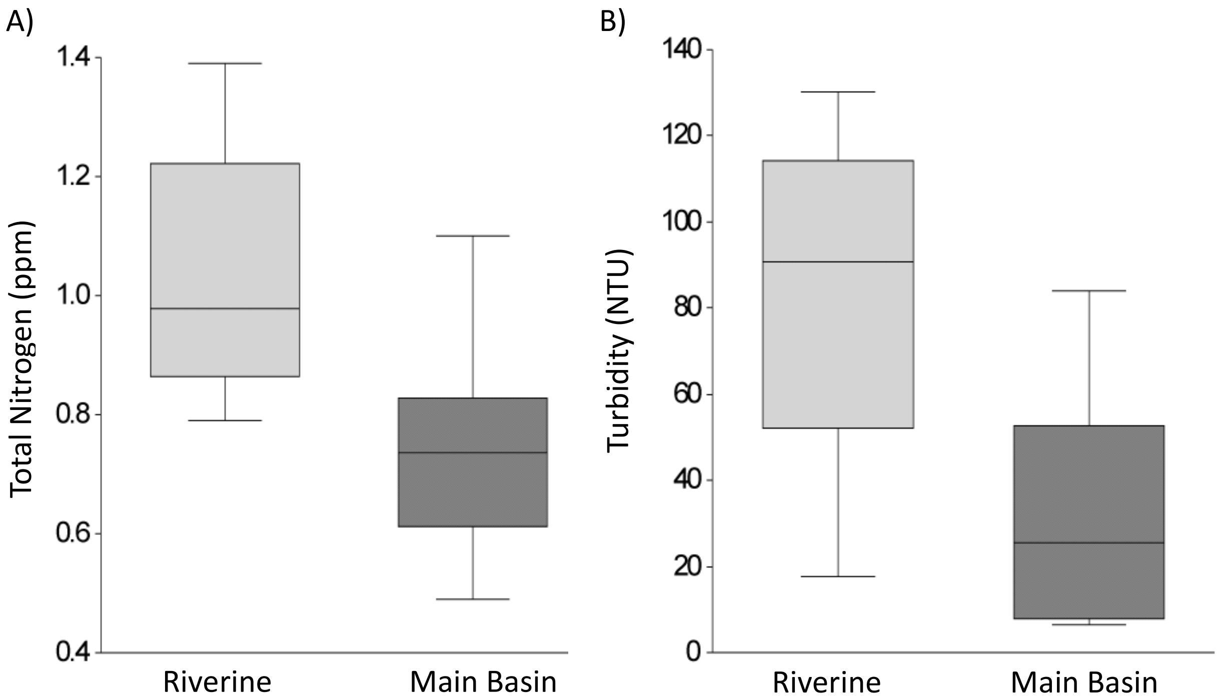

Figure 10: Boxplots of the a) total nitrogen values and b) turbidity values for the riverine and the main basin sites

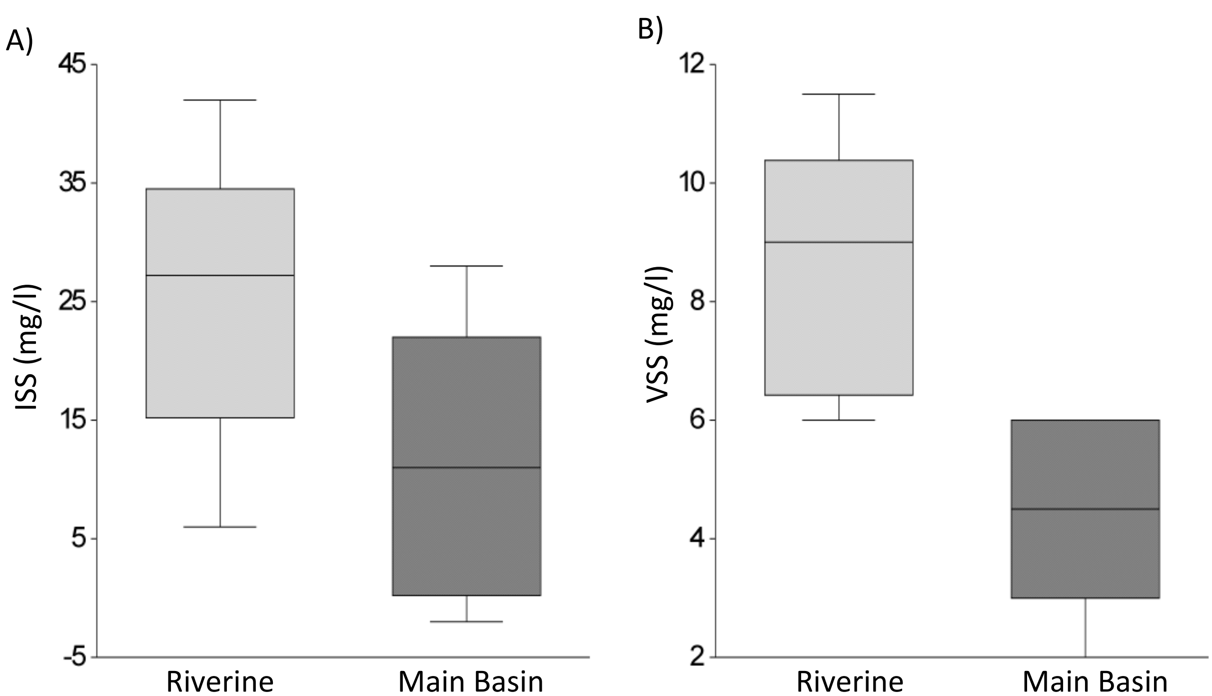

Figure 11: Boxplots of the a) ISS values and b) VSS values for the riverine and the main basin sites

Three out of the four previously mentioned variables (ISS, total nitrogen, and turbidity) had r^2 values of greater than 0.5, and p-values less than 0.05 (Table 5). The total nitrogen regression (p-value= 0.0011) with a r^2 value of 0.85 suggests that as total nitrogen values in the riverine sites increase, total nitrogen values in the main basin sites also increase.

Table 5: Summary of regression analysis for water quality variables. The dependent variables were the main basin values and the independent variables were the riverine site values for each variable

Variable	r^2 value	p-value	Slope
Dissolved Oxygen	**0.53**	**0.0405**	0.98
ISS	**0.62**	**0.0199**	0.85
ORP	0.49	0.0551	0.25
pH	0.24	0.2221	0.35
TDS	**0.96**	**0.0000**	0.96
Temperature	**0.92**	**0.0001**	0.99
Total Nitrogen	**0.85**	**0.0011**	1.06
TOC	**0.84**	**0.0014**	1.16
Total Phosphorus	0.16	0.3189	0.24
Turbidity	**0.52**	**0.0444**	0.98
VSS	0.03	0.6705	0.25

The increased levels of total nitrogen in the riverine segments were as expected; however, one would assume that total phosphorus levels would have also been higher in the riverine sites due to the agricultural runoff that is common to Kansas waters (Nelson et al., 2006). The nitrogen/phosphorus ratio (N:P ratio) is a ratio that helps determine whether nitrogen or phosphorus is the limiting nutrient in a system, or if the two nutrients are co-limiting. N:P ratios less than 14 are said to be nitrogen limiting, ratios greater than 16 are phosphorus limiting, and ratios between 14 and 16 are co-limiting between these two nutrients. (Bedford et al., 1999; Guildford & Hecky, 2000; Smith, 2003) The calculated N:P ratios in our study were not statistically different between the main basin sites and the riverine sites (N:P ratios of 8 and 6, respectively), indicating that both the riverine sites and the main basin sites are potentially limited by the amount of total nitrogen in the system. These results were somewhat unexpected because previous general studies have established that phosphorus is typically the limiting nutrient in lakes and reservoirs (Hecky & Kilham, 1988; Schindler, 1977). However, it has been determined that many Kansas lakes and reservoirs are actually co-limited by both Nitrogen and Phosphorus (Wang et al., 2003). Therefore, we hypothesize that determining the limiting nutrient in a aquatic ecosystem is more complex than just the total nitrogen and total phosphorus values, and that nutrient interactions are occuring within the system.

The high ISS, VSS, and turbidity values in the riverine sites are probably related to the high turbidity and TSS concentrations associated with the rivers and streams that flow into these areas, especially from extreme flooding events (Dodds & Whiles, 2004). As water from the inflowing rivers and streams contribute to the waters of the riverine sites of the reservoir, total riverine concentrations increase or are maintained at high levels while reduced velocities within the reservoir proper allow more sedimentation to occur, thus reducing TSS and turbidity in the main basin sites.

Water chemistry can vary both within and between aquatic ecosystems (e.g. lakes, reservoirs, streams, wetlands); sometimes being similar, sometimes being distinct, or often times as a continum of values and concentrations changing with time and space. Water quality can also differ within ecosystems types because of geology, land use, and land cover. Therefore we used typical ranges of total nitrogen, ISS/TSS, VSS, and turbidity for Kansas streams, Kansas farm ponds, and Kansas reference wetlands for comparison purposes to the riverine sites, collected from studies conducted by the Kansas Department of Health and Environment (KDHE) and the Kansas Biological Survey (KBS) (C. E. Carney, 2002; E. Carney, 2002; Huggins et al., 2017) (Table 2). ANOVA's run comparing the total nitrogen, ISS/TSS, VSS, and turbidity values between the riverine sites and the four reference ecosystems mentioned above yielded contrasting results. The results of these ANOVA's indicated that the total nitrogen concentrations were similar to the stream ecosystems, the VSS values fell within typical wetland ecosystem values, and the TSS and turbidity values were not similar to any of the reference sites (Table 6). However, these results are based on a small sample size (approximately eight values) where the temporal variations could not be totally accounted for.

Table 6: Range of total nitrogen, TSS, VSS, and turbidity values comparing the tested riverine sites to Kansas streams, farm ponds, and reference wetlands.

	Number of Samples	TN (ppm)	TSS (mg/l)	VSS (mg/l)	Turbidity (NTU)
Riverine Sample Sites	8	0.6-1.39	4-42	2-11.5	6.5-130
Streams	8	0.6-3.0	40-150	NA	19-62
Farm Ponds	8	NA	NA	NA	10-85
Reference Wetlands	7	0.9-2.9	2-23	1-22	2.5-16

Of the water quality variables that were not statistically different between the riverine sites and the main basin sites based on the ANOVA analysis, dissolved oxygen (DO) did appear to be consistantly higher in the riverine sites if outliers were not considered (Figure 12).

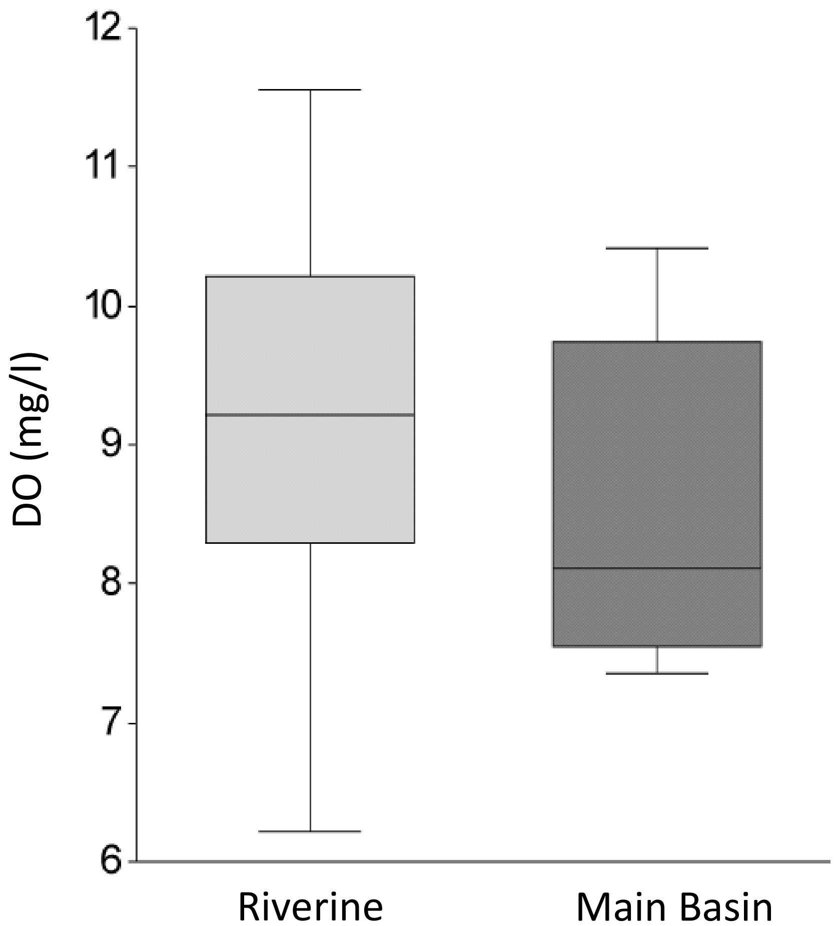

Figure 12: Boxplots of dissolved oxygen for the riverine and main basin sites

The riverine sites have a larger range of DO values than the main basin sites (5.34 mg/l and 3.06 mg/l), but when the DO values are averaged, the extreme low value in the riverine sites dataset draws the mean value to be lower than the median value. This causes the ANOVA analysis for DO to be negative (the means are similar), while the boxplots indicate a difference between these two datasets. The regression of the DO levels of the Main Basin and the Riverine sites also indicated a significant regression with an r^2 value of 0.530 (Table 1).

DO levels are important for fish and aerobic organisms and indirectly indicate if there is some sort of pollution in the water. Too much pollution causes lower DO levels which causes fish kills and dead zones. DO depends on water temperature, dissolved salts, atmospheric pressure, reducing compounds, suspended matter, and living species (Ibanez et al., 2008). These higher DO vlaues could be the results of high reaeration enhansed by high mixing and wave action that is characteristic of these shallow water areas. These riverine sites tend to have higher amounts and concentrations of organic materials which could lend to higher biological oxygen demand (BOD) which could result in lower DO values but that is not the case and thus physically reaeration seems more likely why riverine levels of oxygen are higher than main basin sites, at least during the day time.

Of the data that was not deemed statistically significant by the ANOVA or the boxplot analyses, three parameters had an r^2 value of greater than 0.6 (total dissolved solids, temperature, and total organic carbon) (Table 5). Regression analysis of TDS and conductivity showed a significant positive relationship between the two variables with a r2 value of 0.99 (p-value= 0.000) (Figure 13). This is not suprising in that TDS is also referred to as "filterable residue" and is the concentration of dissolved mineral and organic substances in water, whether in ionic form or not (Helmer, 1999). Thus TDS is often the major large contributor to conductivity measures which is simply the ability of a solution to conduct an electric current. However conductivity is also dependent of the amount of colloidal suspensions and does not account for dissolved silica and undissociated ions that don't carry a charge (McNeil & Cox, 2000). Since these two measures were so strongly related (almost unity, 0.99) we choose to limit all further analyses to TDS.

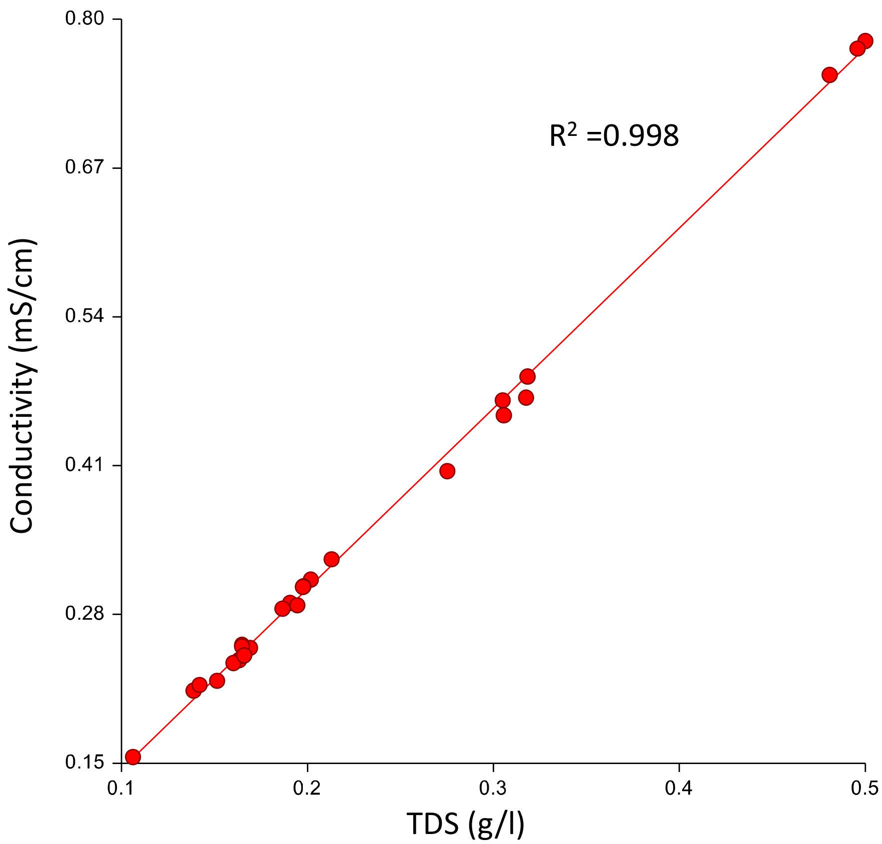

Figure 13: Regression of TDS and conductivity

The values of TOC ranged from 6.7 to 22 mg/l (Tables 7-8); however, due to lab errors, TOC results for three out of the eight reservoirs were discarded, which limited the veracity of these TOC analysis.

The riverine sites had higher temperatures than the main basin sites for all reservoirs, except for Big Hill Reservoir. The temperature values for the riverine and main basin sites also had a significant, positive, linear regression (r^2 of 0.92) indicating that these shallow segements were more influenced by warmer summer air temperature (Figure 14).

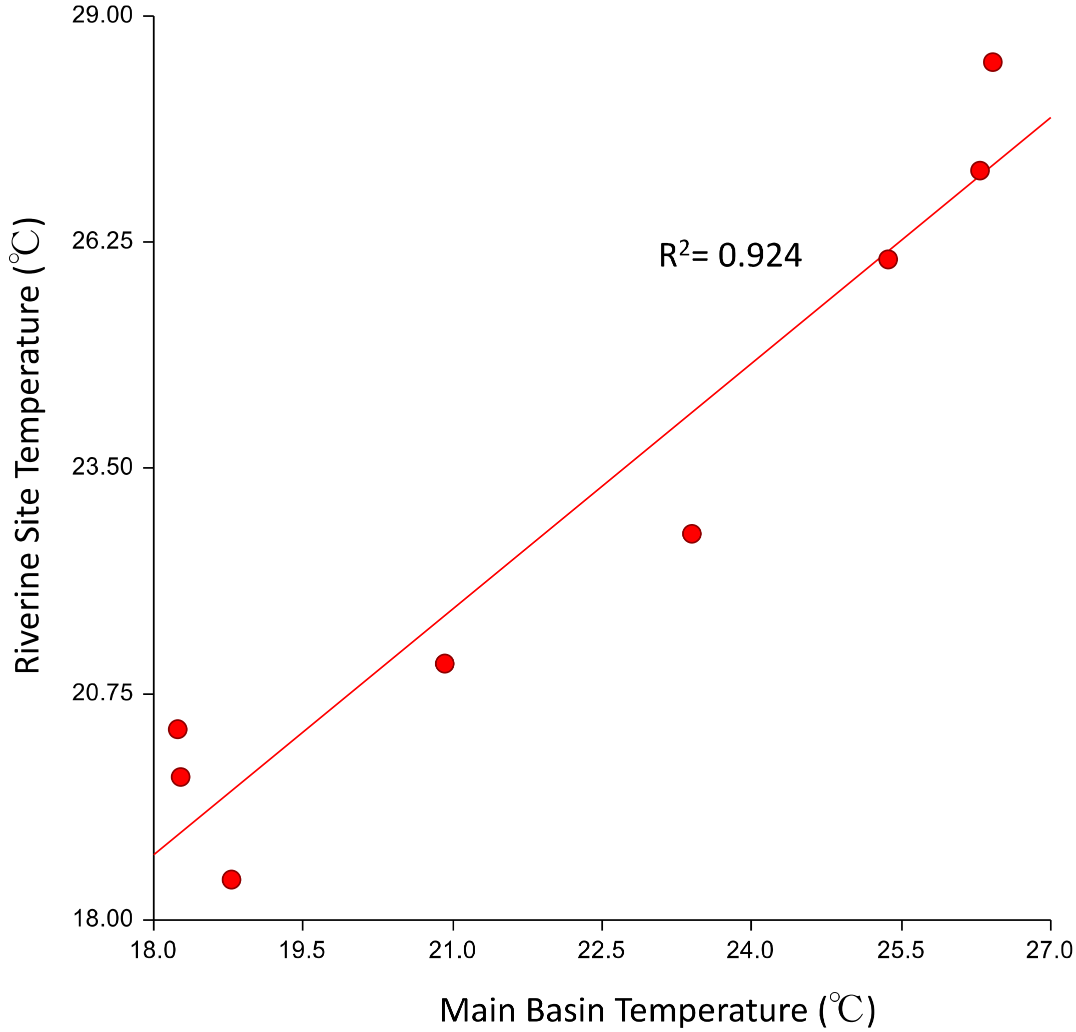

Figure 14: Regression between the temperature of the Main Basin and the temperature of the Riverine Sites

Table 7: Main Basin Water Quality Summary

Parameter	Maximum	Minimum	Mean	Median
Total Nitrogen (ppm)	1.10	0.49	0.74	0.73
Total P (ppm)	0.45	0.03	0.15	0.09
TOC (mg/l)	16	6.7	12	14
TSS (mg/l)	34.0	1.0	15.5	15.0
VSS (mg/l)	6.0	2.0	4.3	4.5
Water Temp	26.42	18.23	22.21	22.15
pH	8.71	7.34	8.09	8.26
ORP (mV)	269.2	112.8	206.2	216.2
Conductivity (mS/cm)	0.7514	0.2134	0.3375	0.2521
Turbidity (NTU)	83.90	6.46	31.46	25.42
DO (mg/l)	10.42	7.36	8.50	8.11
TDS (g/l)	0.480	0.139	0.221	0.167

Table 8: Riverine Site Water Quality Summary

Parameter	Maximum	Minimum	Mean	Median
Total Nitrogen (ppm)	1.47	0.60	1.04	0.98
Total P (ppm)	1.01	0.07	0.23	0.17
TOC (mg/l)	22	7.0	13	14
TSS (mg/l)	58.0	12.0	35.0	36.5
VSS (mg/l)	14.0	5.0	9.1	9.5
Water Temp	29.75	18.00	23.17	21.54
pH	8.89	7.77	8.40	8.49
ORP (mV)	253.8	187.8	220.9	219.1
Conductivity (mS/cm)	0.781	0.155	0.362	0.297
Turbidity (NTU)	161.92	17.58	87.21	80.28
DO (mg/l)	11.91	6.21	9.25	9.18
TDS (g/l)	0.500	0.106	0.239	0.196

No significant site differences or trends were observed or found between oxidation reduction potential (ORP), pH, and the total phosphorus (TP) with regards to ANOVA, box plot, and regression analyses. Also, distance from the main basin was not a significant covariant for any of the tested parameters suggesting that distance traveled was not important. However, if these areas are transitioning into wetland ecosystems based on the water quality results, along with the increased nutrients, one would also expect differences between the riverine sites and the main basin regarding the oxidation-reduction potential (ORP). The ORP is a measurement of the availability of electrons, influenced by pH and temperature, which quantifies the ability of nutrient reduction in a solution. Oxidation is said to be occuring when oxygen is taken up or hydrogen is removed. The process of reduction is the opposite; the removal of oxygen and the gain of hydrogen. The gain and loss of oxygen and hydrogen are important in wetland ecosystems because it dictates the breaking down and recycling of all nutrients that macrophytes can then use (Mitsch & Gosselink, 2007). The fact that the ORP results do not show any differences between these sites does not necessarily mean that the riverine sites are not similar to or transitioning to wetland ecosystems, but that our one time sampling regime could not capture the diurnal or seasonal changes that occur with variables such as pH, DO, ORP and temperature.

Sediment

The ANOVA analysis of the sediment parameters between the reservoir's main basin and the reservoir's branches were not statistically significant for any of the measured variables TN, TP, bulk density, particle size composition, and total organic carbon (TOC)). The ANOVA analysis for TN and TP was limited to just seven reservoirs since results of Big Hill were discarded due to lab errors. However, some differences were noted between riverine and main basin sediments.

Bulk density, percent silt, and percent clay showed differences between their 25th, median, and 75th percentile values between the main basin and the riverine sites (Figures 15-16)

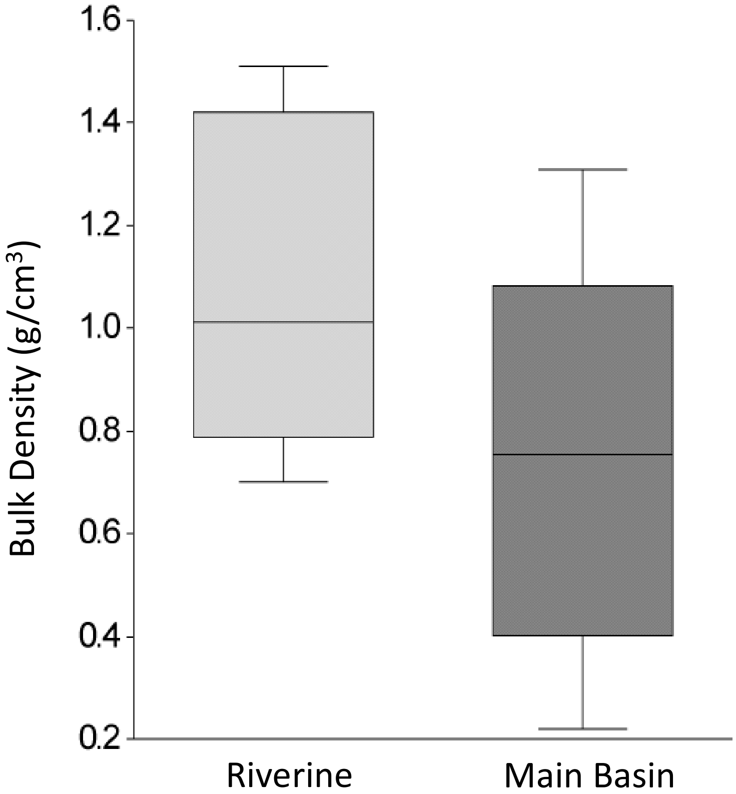

Figure 15: Boxplots of bulk density for the Main Basin and the Riverine Sites

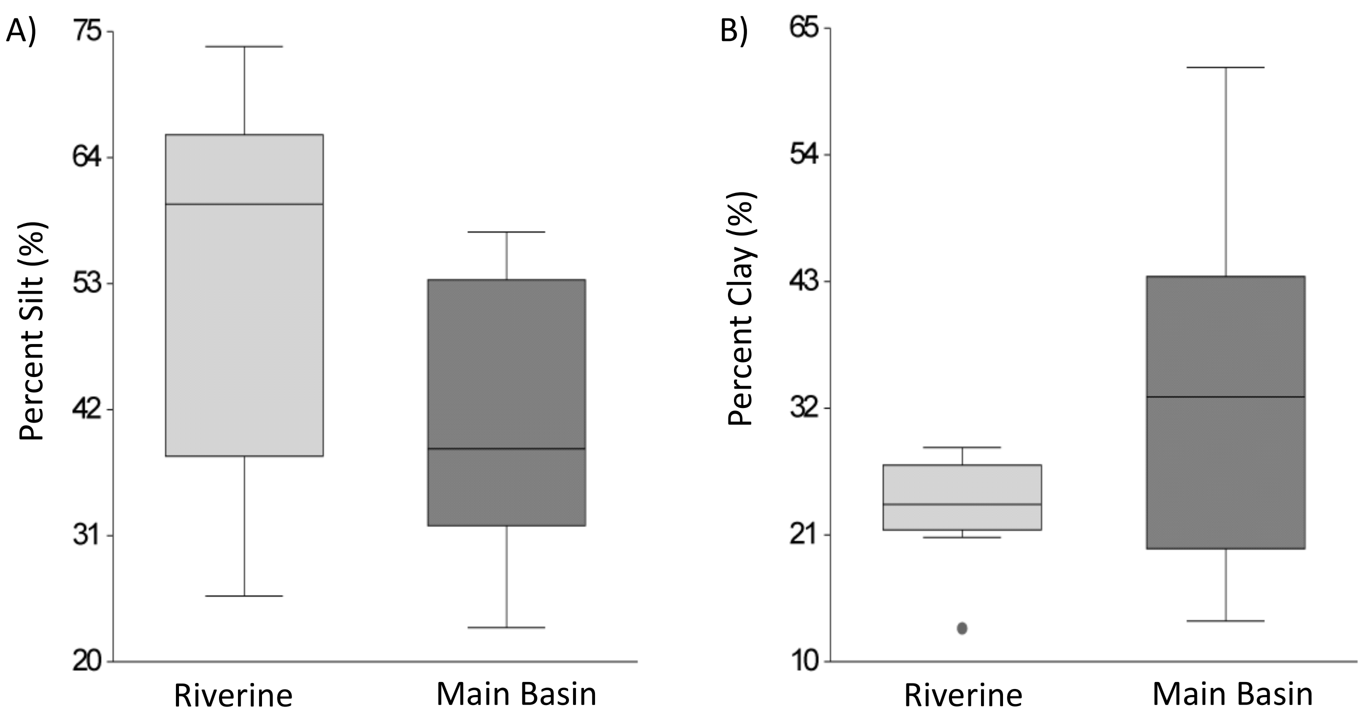

Figure 16: Boxplots of the Main Basin and Riverine Sites for A) silt and B) clay

It was hypothesized that as the sediments enter the reservoirs from these inflow regions, the larger, more dense particles would settle out of the water column and compact into the upper riverine segments, causing higher averages of bulk density compared to the bulk density values in the main basin. The regression analysis on the sediment variables indicated that both percent sand and bulk density were statistically significant and had a positive linear regression (Table 11 and Figure 17).

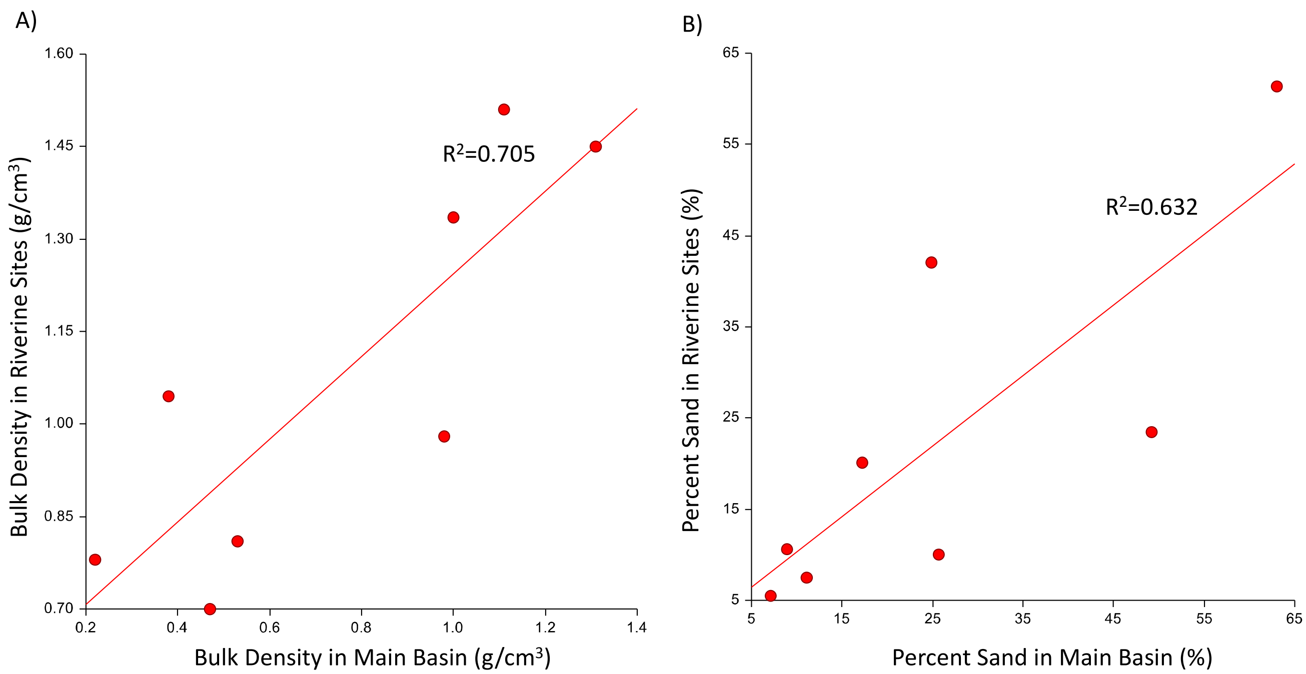

Figure 15: Regression between the Main Basin and the Riverine Sites for A) Bulk density and B) percent sand

The linear regression of percent clay versus total phosphorus indicate a positive statistically insignificant relationship between these to variables (r^2=0.229). This is surprising since clay is an important source of bond phosphorus (Figure 13).

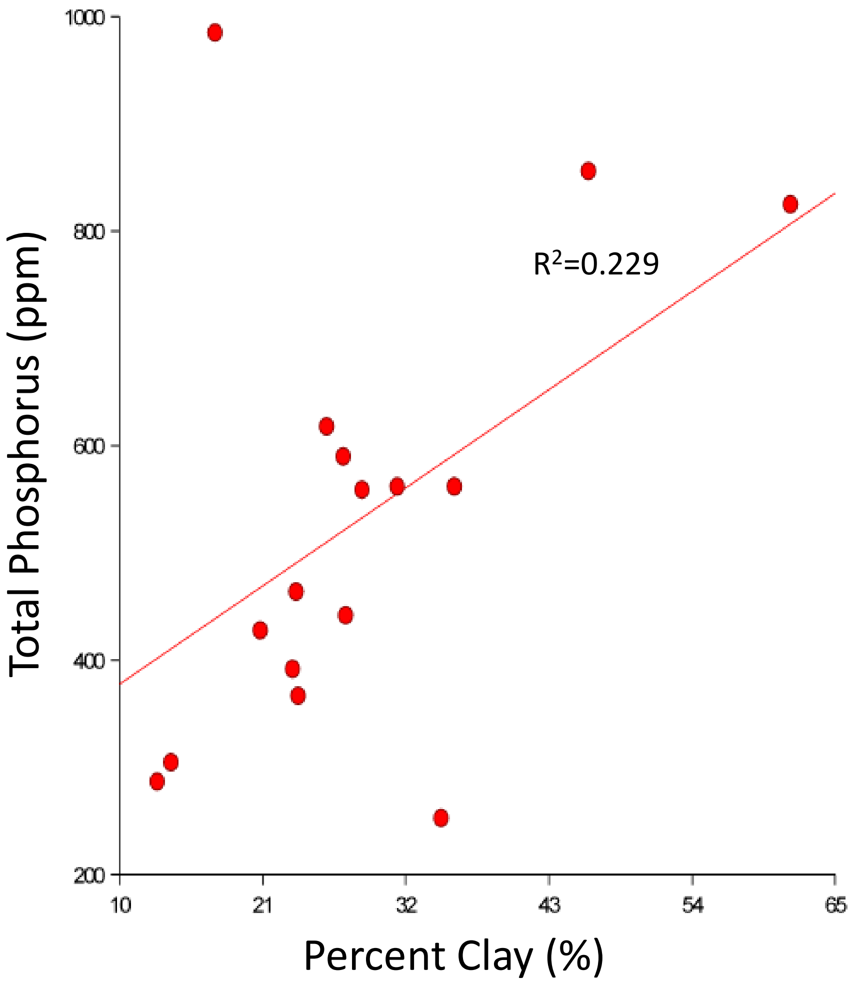

Figure 16: Regression of percent clay versus total phosphorus

This could be due to the sorption of phosphorus onto clay particles (Mitsch & Gosselink, 2007). Lower phosphorus levels in the riverine sites than in the main basin sites may be explained by the parallel differences in clay fraction of their sediments (see Tables 9 and 10). The higher percentage of clay than silt in the main basin sediments is probably due to clay's small size (< 2 µm verses 2-50 µm) and the resulting sedimentation velocities that result in clays being deposited well within the main basin (Hunter & Liss, 1979). Phosphorus attached to clay particles is important in wetland ecosystems because aquatic vegetation use the phosphorus bound in the clay particles for macrophyte growth (Mitsch & Gosselink, 2007).

A summary of the sediment results is shown in Tables 9-10 and a summary of the sediment regression analysis is shown in Tables 11.

Table 9: Maximum, minimum, mean, and median values for the sediment parameters in the main basin sites

Parameter	Maximum	Minimum	Mean	Median
% Clay	62	14	32	31
% Silt	58	23	41	40
% Sand	63	7	27	25
% Total Organic Carbon	2	1	1.6	1.4
Median Bulk Density (g/cm^3)	1.11	0.22	0.58	0.49
Total Nitrogen (ppm)	1755	250	1185	1369
Total Phosphorus (ppm)	825	259	517	562

Table 10: Maximum, minimum, mean, and median values for the sediment parameters in the riverine sites

Parameter	Maximum	Minimum	Mean	Median
% Clay	35	7	23	24
% Silt	76	15	56	63
% Sand	78	0.2	20	11
% Total Organic Carbon	3	0.5	1.6	1.5
Median Bulk Density (g/cm^3)	1.82	0.62	1.09	1.00
Total Nitrogen (ppm)	1607	672	1062	1029
Total Phosphorus (ppm)	680	191	458	435

Table 11: Summary of Regression Analysis for Sediment Quality Variables. The Main basin variable values were the dependent and the riverine values the independent variables.

Variable	r^2 Value	p-value	Slope
% Organic Carbon	0.21	0.2579	-0.42
% Clay	0.02	0.7663	0.04
% Sand	**0.63**	**0.0184**	**0.77**
% Silt	0.27	0.1840	0.73
Bulk Density	**0.70**	**0.0091**	**0.67**
Total Nitrogen	0.34	0.1703	0.17
Total Phosphorus	0.17	0.3608	0.20

The fact that only percent clay and bulk density which are highly correlated to each other show any site relationships is interesting. These results suggest that the concentrations of variables in Table 11 are being influenced by other factors that impact the potential relationships between upper incoming water parameters and main basins values. This may not be surprising since many other factors within the reservoir can influence changes in nutrients including carbon. Why riverine and main basin values of clay and silt are not related is unknown.

Vegetation:

At the main basin sites, 98% of the samples had no vegetation present while the remaining 2% had some vegetative detritus and debris (Figure 19). Within the riverine sites vegetative detritus and debris was collected at 56% of the sites, followed by no vegetation (43%), and 1% aquatic vegetation (Big Hill Reservoir) (Figure 20). It is clear that for the most part coarse particulate organic matter and fine particualte organic matter (CPOM and FPOM) enters into these reservoir systems from the tributaries and riverine segments and is apparently processed and as it moves through the system into the lacustrine zone nearest the dam. The small amount of vegetative detritus and debris found in the lacustrine areas could have orginated from shoreline areas adjacent to these lacustrine areas or from the near by transition zones or less likely from the upper most riverine reaches. The lack of wetland vegetation even within the shallow waters of the riverine zones was somewhat surprising, however other than shoreline vegetation there are no published reports of major aquatic and wetland vegetation communities being reported from any of these reservoirs. The reasons for the near absence of obligate aquatic plants in open water areas of these reservoirs are speculative.

Some relationships between vegetation categories and water and sediment quality were investigated with limited success. No water quality and plant category relationships were found but one significant sediment/plant relationship was noted.

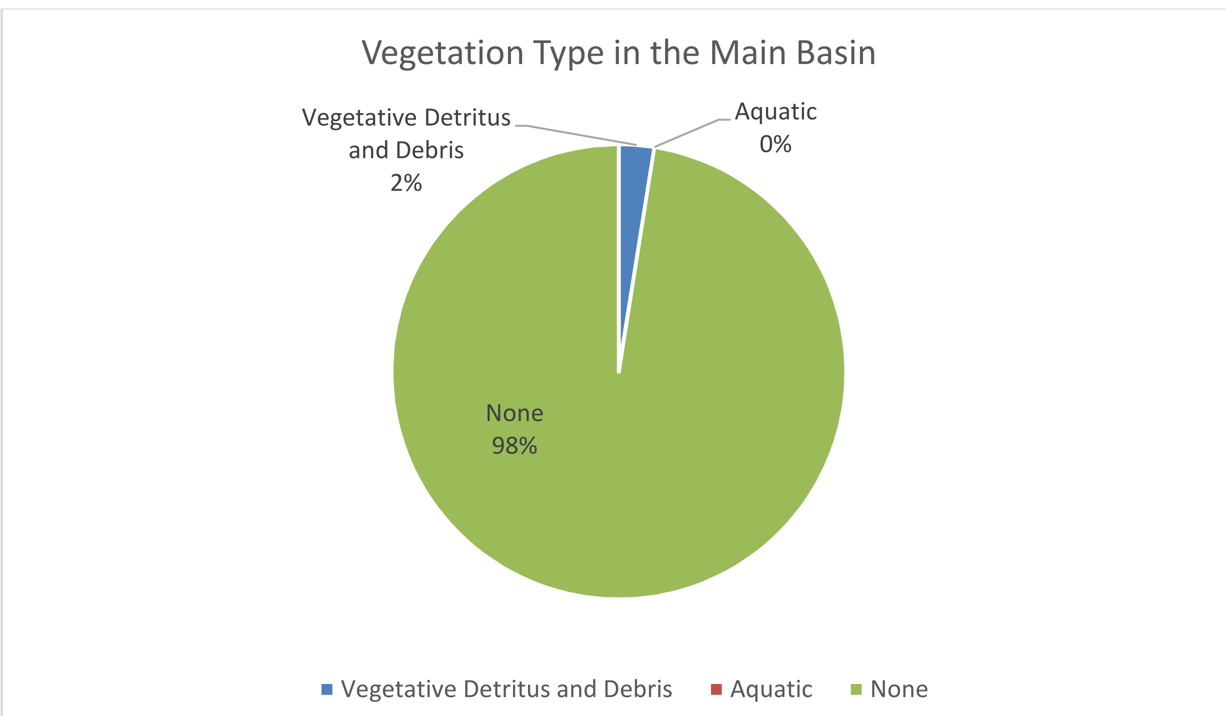

Figure 17: Percentages of vegetative detritus and debris, aquatic vegetation, and no vegetation in the Main Basins

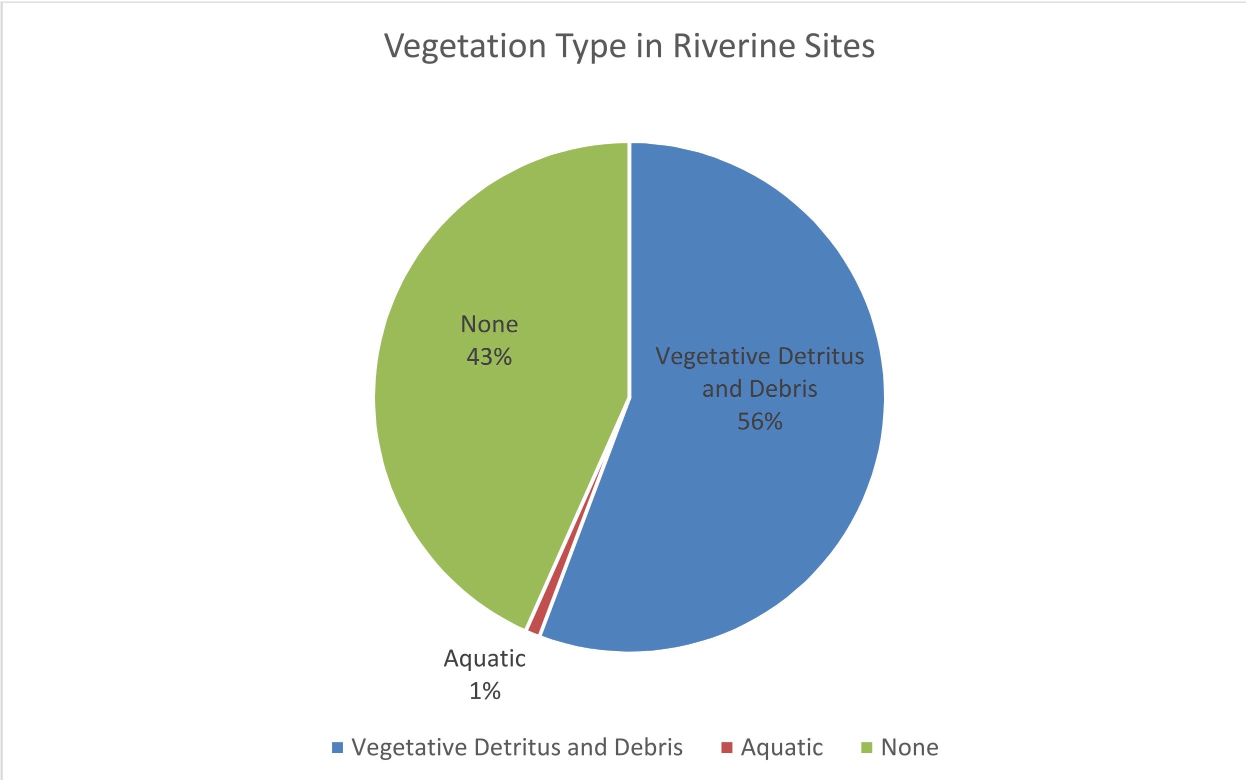

Figure 18: Percentages of vegetative detritus and debris, aquatic vegetation, and no vegetation in the Riverine Sites

Based on the riverine site data, it appears that the percent of vegetative detritus and debris is significantly ($p = 0.0036$) related to the concentration of total nitrogen found in the sediment (r^2 of 0.42, but not the total phosphorus ($r^2 = 0.11$, $p = 0.1706$) (Figure 21).

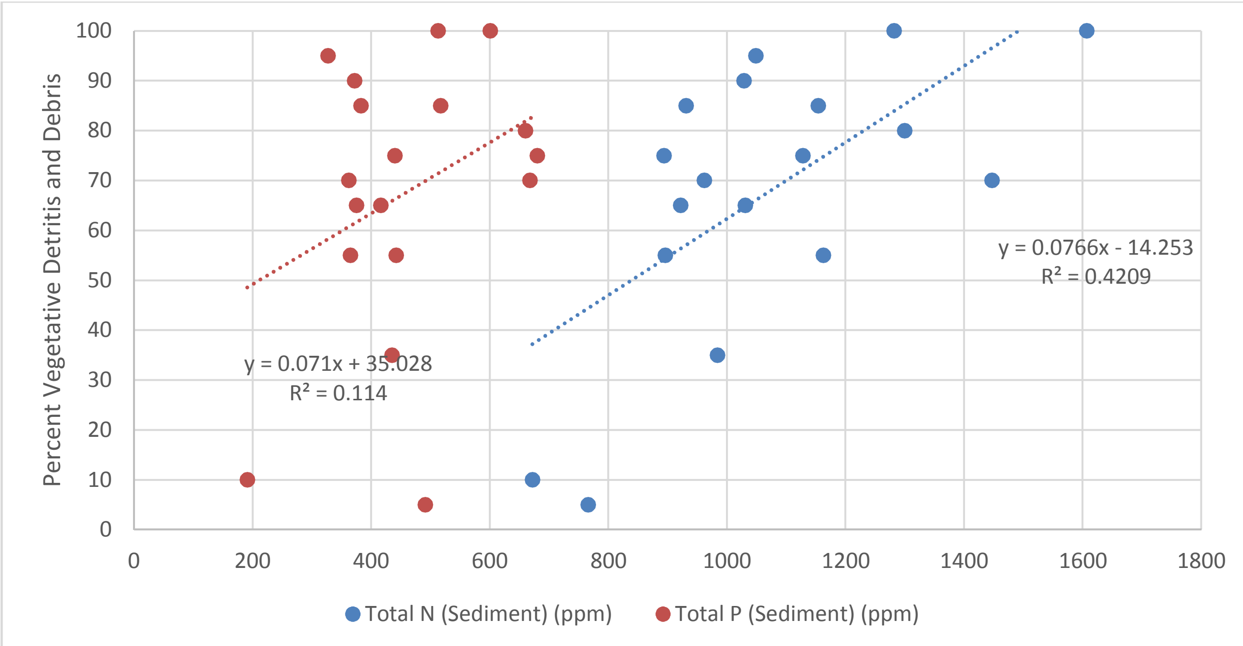

Figure 19: Regression between the TN and TP in the sediment and the percent of vegetative detritus and debris

The correlation between the nitrogen levels and the percent vegetative detritus and debris might be explained by nitrogen recycling as the vegetative detritus and debris break down. As the oxygen is removed from a system, nitrates take over as the electron acceptors leading to the further oxidation of organic matter (Mitsch & Gosselink, 2007). When the vegetative detritus and debris in the system begins to breakdown, organic nitrogen is released. All nitrogen in the system then go through the nitrogen cycle, leading to increased levels of nitrogen.

Out of the 8 sampled reservoirs, only Big Hill Reservoir had aquatic vegetation. Only 20% of the rake hauls in the Big Hill River riverine segment yielded *Ceratophyllum demersum L.* (Common Hornwort). The Common Hornwort is a submergent plant that thrives in habitats of low light intensity (Rook, 2002a). In addition to the Common Hornwort, two other species of aquatic vegetation were found along the shoreline (*Spirodela polyrrhiza Schleid*. (Greater Duckmeal) and *Lemna minor L.* (Common Duckweed)). These two species of aquatic vegetation are often found free-floating in calm, quiet waters creating a vegetative covering on the water surface (Rook, 2002b; Rooks, 2002).

One would expect that more vegetation would be present due to the high nitrogen levels within these shallow riverine areas. One explanation as to the lack of open water macrophytes could be associated with sediment resuspension. Sediment resuspension in reservoirs increases turbidity and the amount of suspended large particulates both orgainc and inorganic. Wave action and bottom shear stress are the main components of sediment resuspension and their effects are most profound in the littoral zones of lakes (Bloesch, 1995).

Wind speed and direction have a large effect on the wave action and the shear stress of a specific area of a water body. For example, higher wind speeds that travel longer distances across the water surface (wind fetch) will create large waves in shallow waters which can create a large enough force to move the sediments on the bottom of the lake or reservoir (Laenen & LeTourneau, 1996). According to Laenen & LeTourneau (1996), wind speeds equal to or greater than 4.5 mph are necessary before sediment resuspension occurs in a reservoir; since Kansas has median wind speeds of 5-11 mph across the state, wave action and shear stress could be a deciding factor in whether aquatic vegetation can establish and maintain themselves in these shallow, riverine areas. Aside from increasing turbidity and thus decreasing light availability, shear stress across the bed sediments could dislodge and suspend the seed beds of rooted macrophytes and prevent successful germination and survival of seedlings. To determine what kind of effect the wind has on the bottom shear stress, we calculated the average distance wind can travel across water for each of the sampled riverine sites using ArcMap and calculated the bottom shear stress using the equations found in Laenen & LeTourneau (1996) (Table 12).

Table 12: Calculated wind fetch and bottom shear stress for the sampled riverine sites

Reservoir	Branch	Wind fetch (m)	Bottom shear stress (Pascals)
Big Hill	Big Hill Creek	19	2
Cheney	Eastern Branch	176	8
Cheney	North Fork Ninnescah	128	8
Clinton	Deer	23	12
Clinton	Rock Creek	27	3
Clinton	Wakarusa	69	1
Council Grove	Munker	303	8
Council Grove	Neosho	101	10
Hillsdale	Bull	72	2
Hillsdale	Niles	39	4
Hillsdale	Rock Creek	18	10
Marion	Cottonwood	103	10
Marion	French	34	3
Perry	Delaware	113	30
Perry	Slough	50	798
Perry	Rock Creek	30	4
Toronto	Eastern Branch	24	5
Toronto	Verdigris	126	8

The calculated bottom shear stress for these riverine sites are fairly consistant between reservoirs except for the Delaware and Slough branches of Perry Reservoir. The range of the bottom shear stress is 12 Pascals, the 25th percentile is 3 Pascals, the median value is 6 Pascals, and the 75th percentile is 8 Pascals, without the outliers of 30 and 798 Pascals.

In a study on wind induced resuspension conducted on Upper Klamath Lake, Oregon, a lake with a surface area of 208 km^2 and an average depth of 1.5 meters, the average shear stress was calculated as 2.7 dynes/cm^2 (0.27 Pascals) (Laenen & LeTourneau, 1996). As seen in table 8, the results of our study yielded larger values of shear stress in the riverine sites compared to the Upper Klamath Lake study. The differences in these results are most likely due to the average water depth (1 meter versus 1.5 meters) and the fact that the Upper Klamath Lake study averaged bottom shear stress values from across the whole lake in contrast to our study where we calculated the shear stress for each riverine site. Based on our calculated values of bottom shear stress and compairsons with shear stress values from literature, it seems highly likely that sediment resuspension is occuring in these riverine areas. While not calculated a high occurance rate of high winds along fetch lines could reduce or prevent the establishment of large communities aquatic vegetation in the shallow open waters of many reservoirs. The high turbidity and ISS levels found in the riverine sites support the resuspension/shear stress hypothesis.

Another explanation as to the lack of aquatic vegetation in our riverine sites could be due to the fact that these reservoirs are highly regulated. A study on the effects of regulating lakes regarding vegetation states that regulated waterbodies had less diversity, more non-native species, and lacked shoreline vegetation compared to waterbodies that have not been regulated. However, that study also states that regulated systems could be restored to a more natural vegetative community with an improved, more natural hydrological regime (Hill et al., 1998). The Kansas Water Office has produced a management plan for the 2017 water year for the reservoirs used in this current study, which includes water level fluctuations for the establishment of aquatic vegetation (KWO, 2016b).

Cumulative Effects

Dendrograms are a visual representation of a hierarchical cluster analysis commonly used in the biological sciences as a way to show similarities and differences between sets of data (Wilks, 2011). In this study, all water quality and sediment data were clustered using the Group

Average (Unweighted Pair-Group) and the Euclidean distance methods within NCSS. Based on the hierarchal analysis, there appear to be 3 distinct groups (Figure 22). The first group consists of 2 reservoirs (Cheney and Marion) that are distinct from the rest of the study sites. The second group consists of 6 out of 8 riverine sites, and the third group consists of 6 out of 8 main basin sites, plus one riverine site (Big Hill).

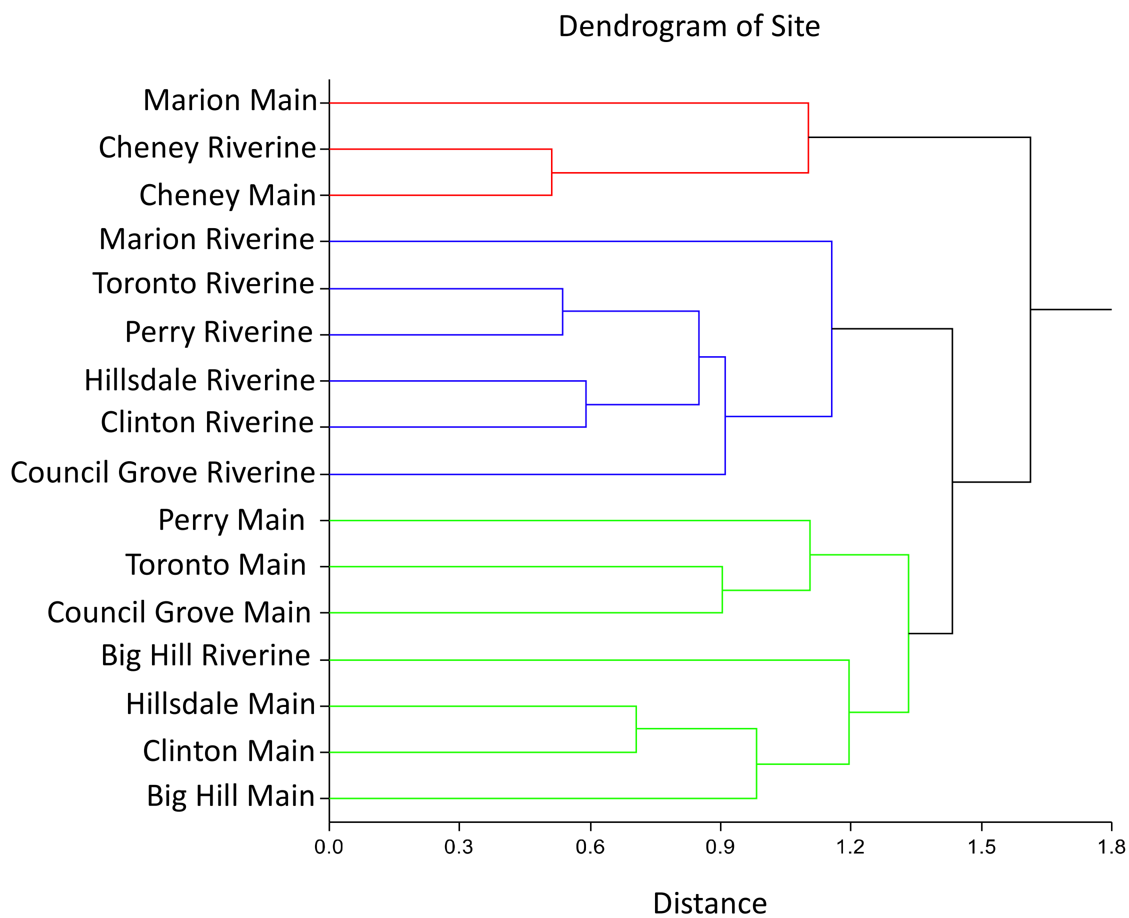

Figure 20: Dendrogram of all study sites based on water and sediment quality results

As indicated in the results section, only four water quality variables were deemed statistically different between the riverine and the main basin sites. However, when all the water quality and sediment parameters were clustered together, a different pattern emerged. It appears that the cumulative effect of the water quality and sediment variables indicate a collective difference exists between the main basin and riverine sites. This collective assessment of water quality characteristics was useful in identifing mainbasin verses riverine differences which was not seen in the results of individual water quality and sediment parameters. Based on the dendrograms, the riverine segments of six of the eight study reservoirs were more similar to each other than the main basins of these reservoirs. This results indicates that cumulatively the riverine segments of these reservoirs are transitioning from functioning as a reservoir ecosystem into some different ecosystem.

Conclusion

It was hypothesized that these upper riverine segments were functioning differently from a typical reservoir ecosystem due to sediment infilling and high nutrient levels. Four individual water quality parameters were statistically different from the main basin sites and fell into typical ranges of other ecosystems, which supports the hypothesis. However, very little aquatic vegetation was found at these riverine sites. Vegetative detritus and debris was found most often at these riverine sites and correlated fairly well with the concentration of total nitrogen.

When all the water quality and sediment parameters were analyzed cumulatively using a hierarchical cluster analysis the riverine sites and the main basin sites were separated into two fairly separate systems, with the exception of two reservoirs. This suggests that the riverine sites of these studied reservoirs are transitioning into a different ecosystem than the typical reservoir ecosystem, however, more analysis needs to be completed to determine exactly what type of ecosystem these riverine sites are transitioning into.

Chapter 4: Summary and Conclusion:

The objectives of this study were to identify areas of potential wetland development using LiDAR within Kansas Reservoirs, and to assess water quality, sediment, and vegetation to determine if these potential wetland areas were developing into wetland ecosystems.

We used Historical hydrological water level data to determine 25th, 50th, and 75th percentile water level boundaries for 20 federal reservoirs in the state of Kansas. We then calculated the median slope of the wetlands delineated by the NWI that were within the study area for each reservoir. Based on the median NWI slopes, we determined areas within the 50th–75th percentile water level boundaries that had slopes equal to or less than the median NWI slopes. Results indicated that the areas of low slope determined by the median NWI wetland slopes were mostly covered by already delineated wetland polygons (mean = 89%). We determined the area of low slope based on the median and the average slopes of the 20 reservoirs in our dataset. Results showed that using different slopes of low values yielded similar results in the coverage of the NWI compared to our calculated potential wetland areas.

We determined areas that typically have a water level depth of 0-1 meters based on the 50th percentile water level boundary as potential wetland development areas. Water, sediment, and vegetation were collected at each sampling sites (either a branch or main basin site). Results of the field sampling analysis showed that four water quality variables were statistically different from the main basin results (total suspended solids, volatile suspended solids, turbidity, and total nitrogen), but no sediment variables were distinct from the main basin data. Out of the eight reservoirs that were sampled, only one reservoir yielded some vegetation (Big Hill Reservoir), although many of the reservoirs showed wetland vegetation along the shoreline. When all the water quality and sediment variables were analyzed cumulatively the main basin sites grouped together and the branch sites grouped together, except for four sites (Cheney main basin and branch sites, Marion main basin site, and Big Hill branch site).

This study shows that the riverine sites are functioning similarly to each other and the main basin sites are functioning similarly to each other. However, currently, it is not well understood what ecosystems these riverine sites are functioning as. Based on comparing NWI and the reservoir boundaries during wet and dry periods in addition to the water quality results, wetlands may develop in the shallow water areas between the 50th and 75th percentile water level boundaries, but we hypothesize that wind and shear stress are preventing light penetration that is necessary for vegetation to grow in open water habitats. The zones located in the shallow areas are likely to have wetland characteristics or potentially have wetlands develop because of the fluctuations of water level depth, duration, and frequency that are driving factors of wetland ecosystems (Casanova & Brock, 2000; Gopal, 2016).

We completed this study to have increased understanding of the hydrological, physical, and chemical environments of the reservoirs in Kansas and to determine if these upper end riverine areas were transitioning into wetland ecosystems. With this increased understanding of the ecosystems within the reservoirs, better informed decisions can be made about how to manage these systems.

The EPA, USACE, the Kansas Department of Health and Environment, and the Kansas Water Office would benefit from this study to better assess, develop, and manage potential wetlands located in reservoir fluctuation zones to enhance nutrient processing, sediment entrapment, and other potential ecological goods and services. The Kansas Department of Agriculture could use this study to discern the use of water in the reservoirs, and the Kansas

Department of Wildlife, Parks & Tourism may use this study to help protect these potential wetland areas for wildlife habitat.

Some limitations of this study were the number of reservoirs sampled, the number of samples taken, as well as the timing of sampling. It was difficult to establish patterns from the data obtained from only eight reservoirs. The number of samples taken and the timing of sampling were a limitation because we potentially missed temporal changes occurring in these riverine segments. Time of sampling was also a limitation in regard to finding vegetation; perhaps multiple sampling times would have yielded more aquatic vegetation in these areas.

In the future, shoreline surveys should be included in potential wetland assessments as many of our upper riverine zones had shoreline vegetation. We established in this study that these reservoirs are not conducive to open water wetland habitats; however, palustrine wetlands appear to be present.

The shoreline development index (SDI) is a ratio used to measure the complexity of a lake or reservoir shoreline (Wetzel & Likens, 1979). The index shows the potential for the development of communities along the outside of the reservoir, with higher values corresponding with greater potential. Communities found within these littoral zones tend to be of high biological productivity (Wetzel & Likens, 1979). The SDI can potentially be used along with shoreline surveys to determine a relationship between shoreline complexity and palustrine wetland communities.

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Appendix 1: A table of the 20 federal reservoirs in Kansas, the number of each riverine site at each reservoir, and each riverine site name studied.

Reservoir	# Riverine Sites	Name of Riverine Site

Big Hill Lake	1	Big Hill Creek
Cheney Reservoir	2	North Fork Ninnescah
		Eastern Branch
Clinton Lake	3	Deer Creek
		Wakarusa River
		Rock Creek
Council Grove Reservoir	2	Neosho River
		Munkers Creek
El Dorado Lake	3	Cole Creek
		Satchel Creek
		Harrison Creek
Elk City Lake	3	Elk River
		Chetopa Creek
		Squaw Creek
Fall River Lake	2	Fall River
		Badger Creek
Hillsdale Lake	3	Bull Creek
		Niles Creek
		Rock Creek
John Redmond Reservoir	1	Jacob's Creek
Kanopolis Lake	2	Smoky Hill River
		Bluff Creek
Kirwin Reservoir	2	North Fork Solomon River
		Bow Creek
Marion Reservoir	2	North Cottonwood River
		French Creek
Melvern Lake	2	Marais des Cygnes
		Turkey Creek
Milford Lake	1	Republican River
Perry Lake	3	Delaware River
		Rock Creek
		Slough Creek
Pomona Lake	3	Coyote Creek
		Hundred and Ten Mile Creek
		Valley Brook
Toronto Lake	2	Verdigris River
		Eastern Branch
Tuttle Lake	1	Big Blue River
Webster Reservoir	1	Spring Creek
Wilson Lake	2	Saline River
		Hell Creek
Total	42	