Photograph looking downstream from Gillham Dam,
taken by Victor Kuykendall (U.S. Army Corps of Engineers).

COSSATOT RIVER AND GILLHAM LAKE, ARKANSAS

Sustainable Rivers Program 2022

Abstract

This report details the current available data and literature for the Cossatot River to identify flow-dependent fish, mussels, and other species in the River, examine changes in these species over time, and look at alterations in the flow regime that potentially could have caused these changes.

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Contents

Acknowledgments

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Introduction

It is well documented in numerous scientific publications that dams disrupt the natural flow regime, disrupting native species life cycles, reducing species diversity and quantity, causing loss of connection of the river to its floodplain, and encouraging the encroachment of exotic and/or invasive species (Risley et al., 2010; Chen and Olden, 2017; Warner et al., 2014; Richter and Thomas, 2007). Dams, among other anthropogenic activities (altering land use, water withdrawals, etc.,) can cause hydrologic alterations that reduce peaks, prolong baseflows, smooth the hydrograph, produce unseasonably high flows, and impact water quality, in particular, water temperature and dissolved oxygen (DO). The U.S. Army Corps of Engineers (USACE) and The Nature Conservancy (TNC) along with several sponsors in the State of Arkansas [Arkansas Game and Fish Commission (AGFC), Arkansas Natural Heritage Commission (ANHC), and Arkansas Department of Environmental Quality (ADEQ)] have joined efforts in order to recommend a dam reoperation plan for Gillham Dam, an impoundment of the Cossatot River in southwest Arkansas, through the Sustainable Rivers Program (SRP; USACE, 2011).

In 2020, the Cossatot River was added to the SRP. This report details the current available data and literature for the Cossatot River to identify flow-dependent fish, mussels, and other species in the river, examine changes in these species over time, and look at alterations in the flow regime that potentially could have caused these changes. As with other SRP projects (see for example, “Environmental Flows Science” at https://www.hec.usace.army.mil/sustainablerivers/), once the degree of flow alterations has been determined and experts have developed recommendations to restore eco-hydrological function, USACE will examine possibilities for reservoir management modifications within the range of authorized releases that would meet expert recommendations to benefit the Cossatot River ecosystem and its biota. Initial coordination efforts, including identification of partners and issues are described in Appendix I.

History of environmental flows

Environmental flows have been defined in the Brisbane Declaration (2007) as “the quantity, timing, and quality of water flows required to sustain freshwater and estuarine ecosystems and the human livelihoods and well-being that depend on these ecosystems”. There is little information regarding environmental flows in the Cossatot River and surrounding streams, necessitating the need to develop and compile such information. However, environmental flow information has been compiled by various entities in Oklahoma for streams near the Cossatot, which may allow for the transfer of data and methodologies to occur between basins. The most notable publications and accompanying data are summarized in the following paragraphs.

The Oklahoma Department of Wildlife Conservation conducted instream flow modeling for mussels and fishes of southeastern Oklahoma (Jones and Fisher, 2005), which included the Kiamichi River and the Little River and its tributaries. The report documents findings and conclusions about “instream flow modeling for mussel beds in the Kiamichi River, Oklahoma, and determination of instream flow recommendations for four streams in southeastern Oklahoma”. In order to have the least impact on endangered mussel habitats, some of the more notable recommendations included: 1) providing optimal withdrawal rates for water-use at various locations in the Kiamichi River; 2) providing instream flows for the Kiamichi River at various locations that mimic a historic, unaltered flow regime; and 3) operation guidelines for Sardis Lake, a reservoir located on Jackfork Creek, a tributary to the Kiamichi River. The four southeastern Oklahoma streams included in the instream flow recommendations were the Kiamichi River, Little River, Glover River and Mountain Fork Creek. The study used a “proportional analysis method, which was developed for use in warm water streams and utilizes historic streamflow information, median streamflow values and species suitability curves based on macrohabitat variables to provide recommended streamflow values.” The results of the study provided estimated streamflow values to support populations of seven fish species at three sites. However, results may be used to estimate flows to support populations of these species at any location on each of the rivers.

The Oklahoma Water Resources Research Institute compiled "An Assessment of Environmental Flows of Oklahoma" report (Turton et al., 2009) that provided environmental flow recommendations as part of the updating process for the Oklahoma Comprehensive Water Plan. Within this report, 88 sites were used to classify Oklahoma streams. Of these 88 sites, 12 were located in Arkansas and included the two USGS stream gaging sites located on the Cossatot. Using the Hydroecological Integrity Assessment Process (HIP), each stream was classified based on flow regime. The first step in the HIP process requires a principal component analysis (PCA) to identify the hydrologic indices that contain the most information about the flow regime. The second step uses a cluster analysis to classify and group each streamflow gaging site based on similarity of flow regime. Six principal components were selected for high information indices explaining 77 percent of the total variation. The PCA resulted in 27 streamflow indices capturing all five components of the flow regime. The cluster analysis used multivariate statistical methods to identify patterns between many sites using many variables. Three distinct groups emerged from the cluster analysis, a two-cluster classification, a four-cluster classification, and a six-cluster classification. The two-cluster classification that included both the Cossatot River near DeQueen and the Cossatot River near Vandervoort, Arkansas, sites had higher mean flows with higher flow during low flow periods. The four-cluster classification resulted in a more regional distribution of the 88 sites with a distinct cluster in the southeast, which included all the streams in the vicinity of the Cossatot River, including Cossatot River near DeQueen, Arkansas, and excluding Cossatot River near Vandervoort, Arkansas; this site was grouped into a different cluster. The group containing the Cossatot River near DeQueen, Arkansas, site had more frequent and less variable high flow events, significantly higher mean flows, and a higher magnitude of maximum flows in April and were classified as perennial flashy streams. The Cossatot River near Vandervoort, Arkansas, site had lower mean flows with relatively stable flows and was classified as a perennial run-off stream. Finally, the six-cluster classification further refined all the streams in the vicinity of the Cossatot River near DeQueen, Arkansas, site, breaking those sites into two distinct groups notably defined by basin size.

The above "Assessment of Environmental Flows of Oklahoma" report was followed by a USGS Open-File Report (OFR) that conducted a "Biological Assessment of Environmental Flows for Oklahoma" (Fisher et al., 2012) using the environmental flow analyses conducted in the previous report. The USGS OFR concluded that reservoir construction and operations significantly altered fish assemblage structure, particularly in the Kiamichi River in southeastern Oklahoma. Using the four general types of flow regimes, describing a wide range of flow conditions (perennial runoff, perennial flashy, stable groundwater, and intermittent), described in the Turton et al. (2009) analyses, the level of alteration was determined and related to changes in biotic communities. The same 27 streamflow indices that were determined from the PCA developed by Turton et al. (2009) were used for this analysis. Using 28 fish sampling sites, two nonmetric multidimensional scaling (NMS) ordination axes (2D) were identified that explained 85.8 percent of the variation among the fish sampling sites. NMS axis 1 (x axis) represented changes in species composition along a gradient of decreasing variability in daily flows (MA4), variability in May flows (MA28), frequency of low flow spells (FL3), high flood pulse count (FH4), variability in annual maximums of 90-day means of daily discharge (DH10), constancy (TA1), and fall rate (RA3). NMS axis 2 (y axis) represented changes in species composition related to increasing variability in annual maximums of 3-day means of daily discharge (DH7), DH10, and RA3. The changes in flow environments were associated more with geography rather than flow alteration. However, eastern Oklahoma sites (the area most similar to the Cossatot) showed differences between altered and reference samples along a trajectory correlated with increasing MA4, MA28, FL3, FH4, and TA1. The PCA of fish functional group proportions identified two significant principal components that explained 63.2 percent of the variation. There were nine indices positively correlated with PC1 (MA4, MA28, specific mean annual maximum flows (MH20), FL3, flood frequency (FH5), number of zero-flow days (DL18), DH10, TA1, RA3, and five indices negatively correlated with PC2 [mean daily flow (MA1), mean minimum January flow (ML1), mean maximum April flow (MH4), and no daily rises (RA5)]. Overall, there were similar relationships in the PCA compared to the NMS ordination analysis.

Finally, Leasure et al. (2016) classified and quantified the natural flow regime for the Ozark-Ouachita Interior Highlands region of Arkansas, Missouri and Oklahoma. Using 64 reference streams, daily streamflow records of at least 15 consecutive years, and a mixture model cluster analysis, Leasure et al. (2016) identified seven natural flow regimes. The Cossatot River near Vandervoort, Arkansas, site was indicated as a reference gage, by the methods further explained in Leasure et al. (2016), whereas the Cossatot River near DeQueen, Arkansas, site was not considered a reference site and therefore not included in the cluster analysis. Using the methods defined by Leasure et al. (2016), the Cossatot River near Vandervoort, Arkansas, site was classified as a groundwater flashy site and the stretch of stream below Gillham Dam was classified as runoff flashy. Groundwater flashy streams were defined as having less daily flow variability than any runoff-dominated streams. These streams never dried up completely, although their flow was sometimes less than 5 percent of the mean daily flow. Runoff flashy streams receded slower than intermittent runoff streams, which were common in the Ouachita Mountains, had fewer days of no flow and less flow variability. Furthermore, Leasure et al. (2016) provided a set of nonredundant flow metrics to represent ecologically important components for each of the defined seven natural flow regimes.

Basin characteristics and water management

The Cossatot River, located entirely in southwest Arkansas, is one of five major tributaries to the Little River, which, subsequently, is a major tributary to the Red River (Figure 1). The Cossatot River (herein referred to as simply “the Cossatot”) is impounded by Gillham Dam to form Gillham Lake. The Cossatot upstream of Gillham Lake is part of the National Wild and Scenic Rivers system and, along with its tributary Caney Creek, is designated as an Extraordinary Resource Water. Additionally, the Cossatot upstream of Gillham Lake and its tributary Brushy Creek are designated as a Natural and Scenic Waterway (ANRC, 2018).

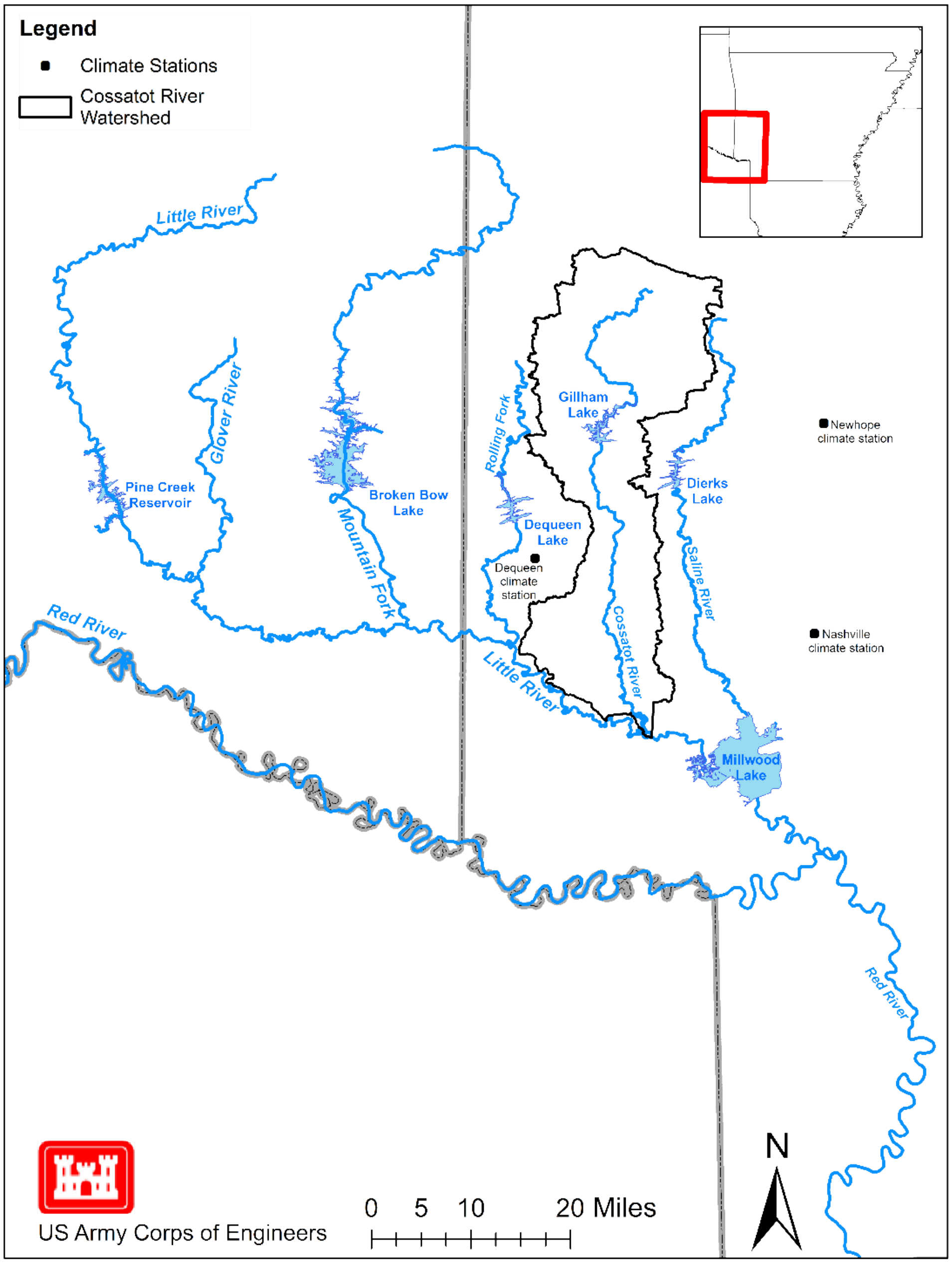

Figure 1. Map of study area and location of climate stations.

Gillham Dam is located in Howard County, Arkansas, at mile 49 of the 89-mile-long Cossatot, approximately 6 miles northeast of Gillham, Arkansas. Eighty-five percent of Gillham Lake is located in Howard County and the remainder in Polk County. The reservoir has a storage capacity of 23,000 acre-feet (at a conservation pool elevation of 502 feet), provides 36 miles of shoreline, and drains approximately 273 square miles. There are two U.S. Geological Survey (USGS) stream gaging sites located on the Cossatot: 07340300 (Cossatot River near Vandervoort, Arkansas) and 07340500 (Cossatot

River near DeQueen, Arkansas) (Figure 2), that measure daily mean stream discharge, stream stage, precipitation, and the Cossatot River near Vandervoort site also measures water temperature.

Figure 2. Physiographic sections and locations of streamflow gaging stations and additional water quality sampling sites.

Basin climate and physiography

The Cossatot watershed can generally be broken into and defined by a northern portion and a southern portion. The watershed is split approximately in half by two physiographic provinces (Figure 2) that generally align with two Environmental Protection Agency (EPA) Level III Ecoregions (Figure 3). This "splitting" of the watershed generally aligns with separate, unique characteristics for the climate and physiography of the watershed.

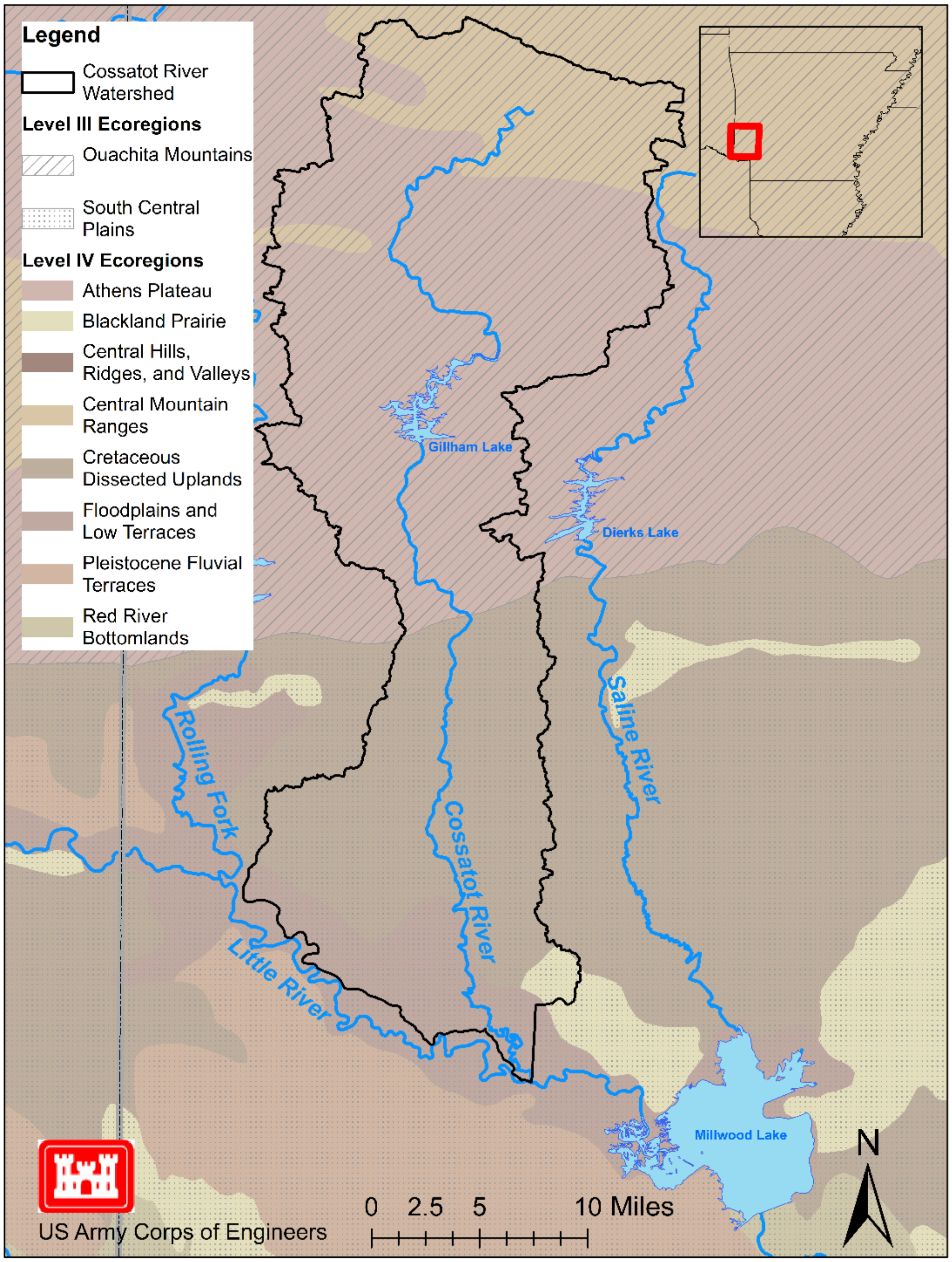

Figure 3. Environmental Protection Agency Level III and Level IV Ecoregions for the Cossatot River Watershed.

Climate

The climate for the Cossatot watershed is predominantly influenced by its proximity to the Gulf of Mexico and is classified as humid subtropical according to the Köppen climate model (Kottek et al., 2006). A humid subtropical climate is characterized by hot, usually humid summers and mild to cool winters. Warm, humid, subtropical air that is generated by the Gulf of Mexico can lead to heavy precipitation under certain large-scale pressure patterns. The moist warm air meets with cold dry air from the west and this combination creates an environment of high instability and wind shear (Perica et al., 2013). These fronts tend to have a north-south alignment but can also shift east-west, can occur any time of year and can generate heavy precipitation (or annual maxima) for daily or longer durations (Perica et al., 2013). The watershed is also susceptible to tropical systems which account for the majority of the extreme rainfall events (Perica et al., 2013).

Mean daily temperatures for the study area range from a high of 73 degrees Fahrenheit (°F) to a low of 48 °F with July being the hottest month, with a mean daily high of approximately 92 °F, and January being the coldest month, with a mean daily low of approximately 29 °F. Average first and last frost typically falls between November 1 through November 10 and April 1 through April 10, respectively. The area averages 4.5 inches of rainfall monthly, receiving the majority in spring and averaging 55 inches annually. On average, the area in the vicinity of Gillham Lake receives precipitation 92 days per year in the form of rain and snow, sleet, or hail (approximately 2 inches, annually, of frozen precipitation). Evaporation from Gillham Lake, over the past 30 years (1989 -2019), averages 282 acre-feet annually, with the majority of evaporation occurring in late July at 19 acre-feet per day and the lowest between December and January at 2.9 acre-feet per day.

Precipitation Trends

An understanding of historical trends in climate is needed to explain historical changes in hydrology. This includes variables of precipitation, temperature, evaporation, wind speed, and relative humidity; however, only a simplification of the precipitation trends for the Cossatot River near Vandervoort, Arkansas, and DeQueen, Arkansas, gaging sites are discussed. Overall, precipitation amounts for both the Vandervoort and DeQueen gaging sites have increased through time. Using monthly Parameter-elevation Regressions on Independent Slopes Model (PRISM) climate data (PRISM, 2020), total precipitation amounts have been calculated for the watersheds draining to each of the respective streamflow gaging sites on the Cossatot. Annual precipitation data extend back to 1895 through 2019, with each year being broken into seasonal data as well. Figures 4-8 display the data for the Cossatot River gaging site near DeQueen, Arkansas, and Figures 9-13 display the data for the Cossatot River gaging site near Vandervoort, Arkansas.

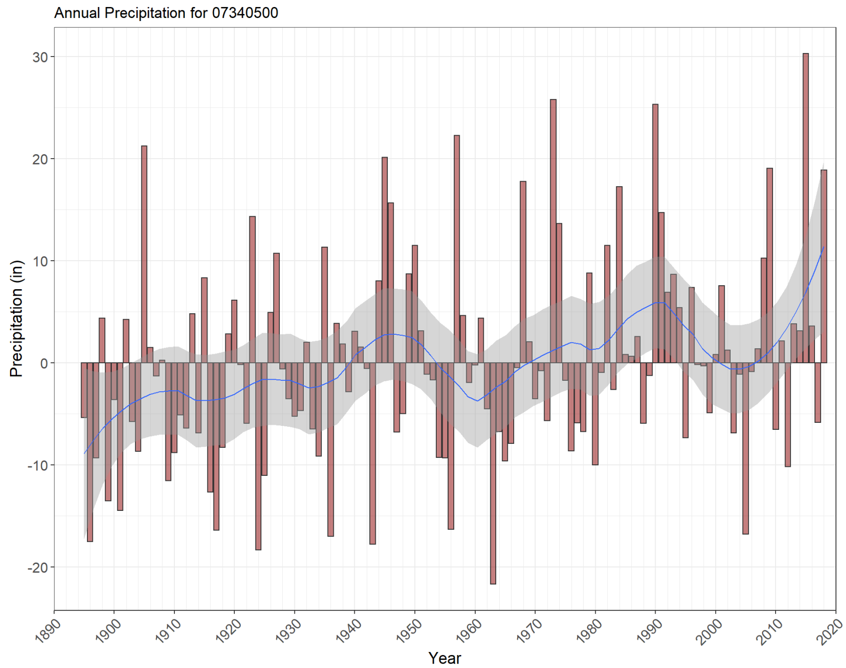

Figure 4. Annual precipitation departure from the mean precipitation for the period of record for Cossatot River near DeQueen, Arkansas (blue line is a LOESS curve with 90% confidence limits (grey area)).

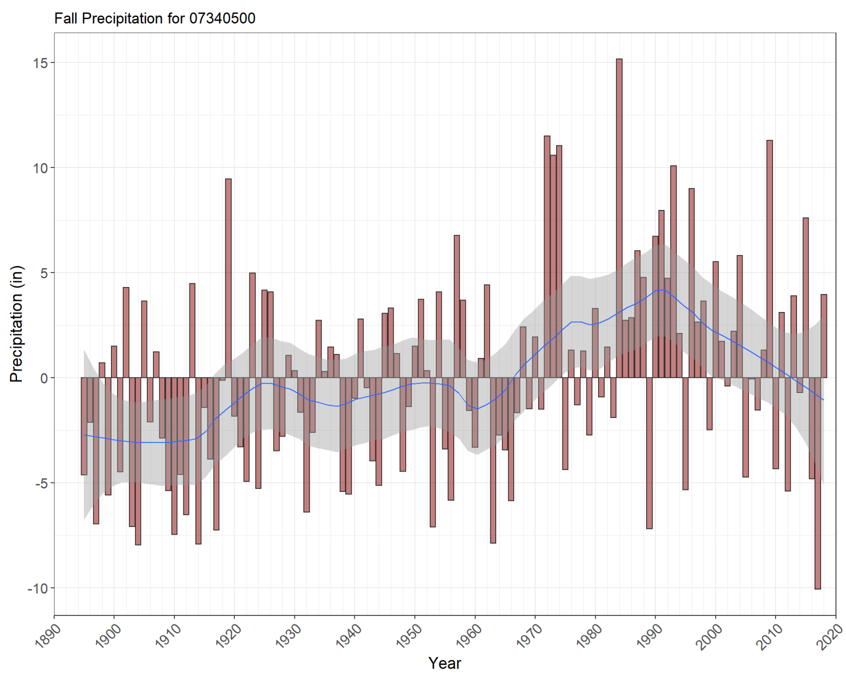

Figure 5. Fall precipitation departure from the mean precipitation for the period of record for Cossatot River near DeQueen, Arkansas (blue line is a LOESS curve with 90% confidence limits (grey area)).

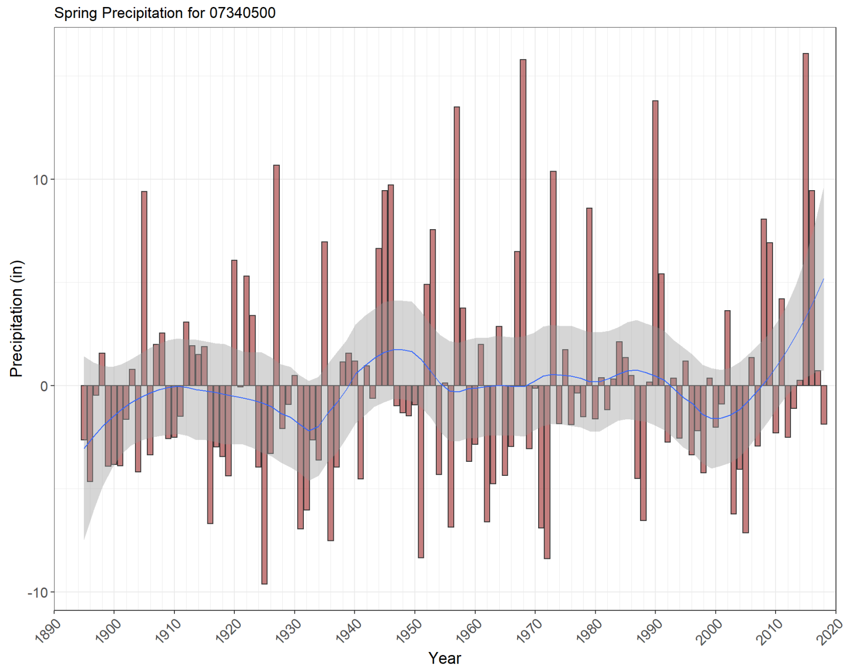

Figure 6. Spring precipitation departure from the mean precipitation for the period of record for Cossatot River near DeQueen, Arkansas (blue line is a LOESS curve with 90% confidence limits (grey area)).

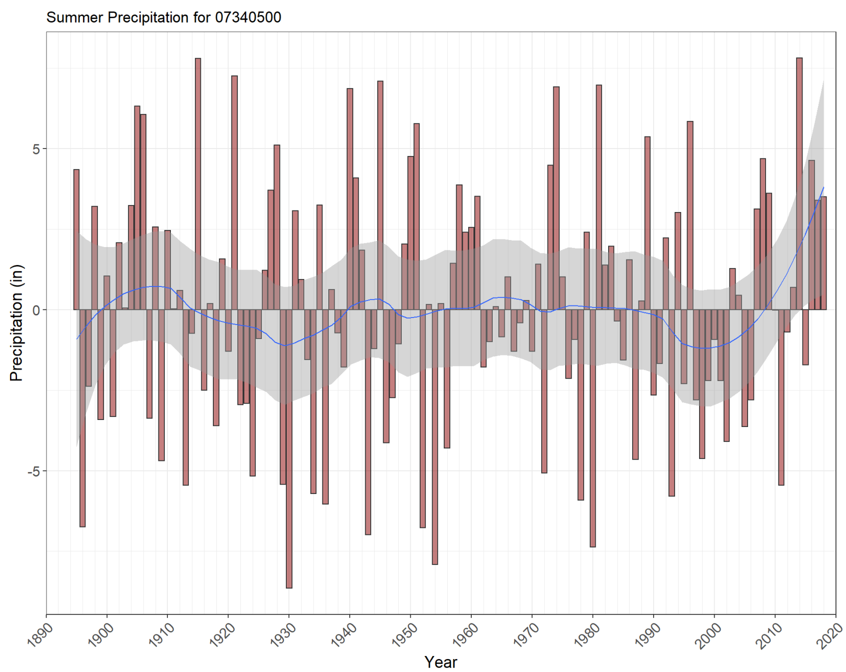

Figure 7. Summer precipitation departure from the mean precipitation for the period of record for Cossatot River near DeQueen, Arkansas (blue line is a LOESS curve with 90% confidence limits (grey area)).

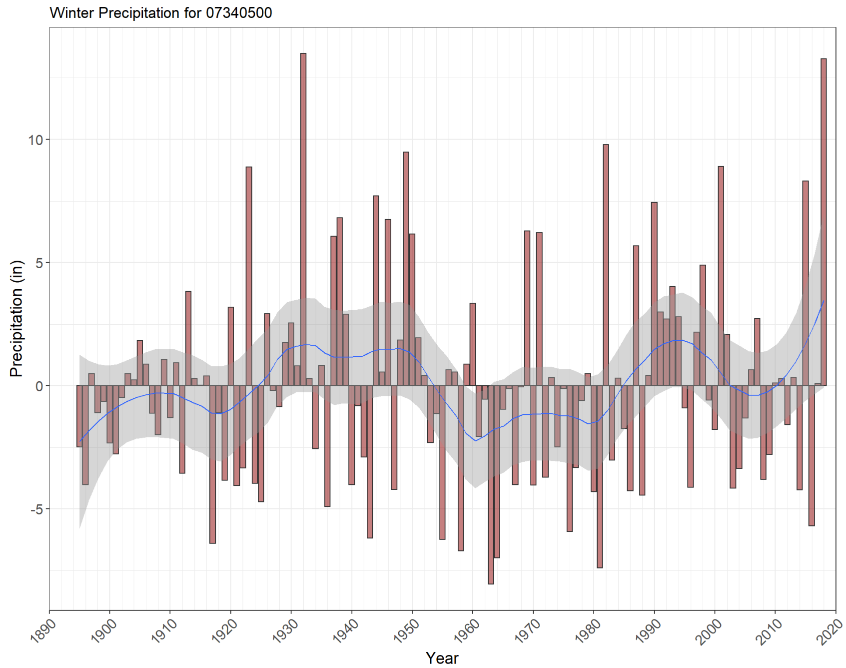

Figure 8. Winter precipitation departure from the mean precipitation for the period of record for Cossatot River near DeQueen, Arkansas (blue line is a LOESS curve with 90% confidence limits (grey area)).

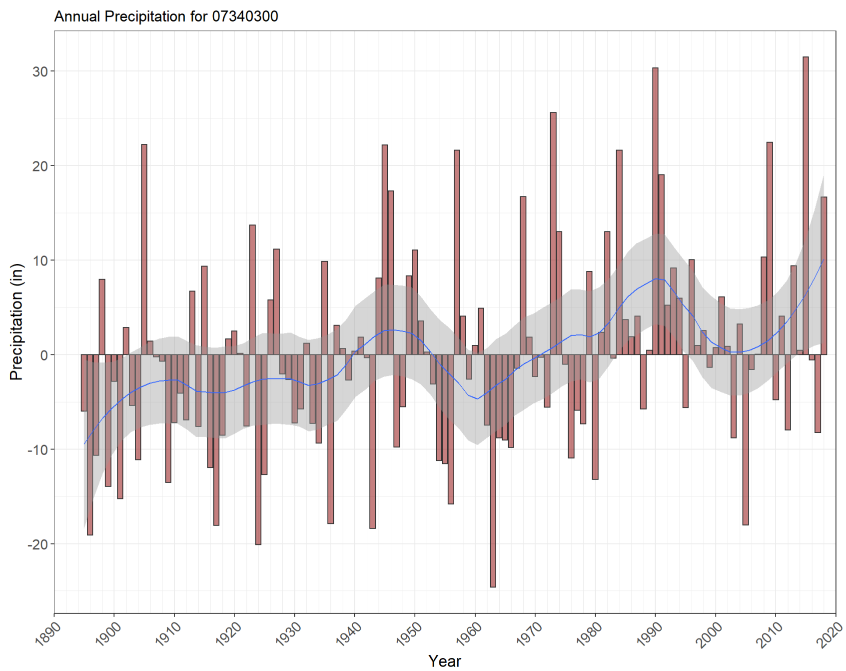

Figure 9. Annual precipitation departure from the mean precipitation for the period of record for Cossatot River near Vandervoort, Arkansas (blue line is a LOESS curve with 90% confidence limits (grey area)).

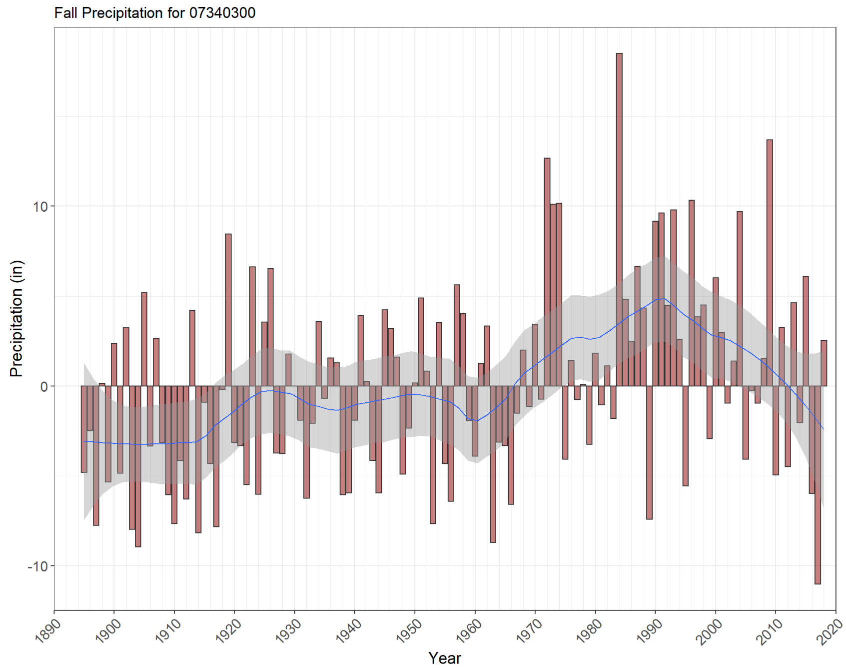

Figure 10. Fall precipitation departure from the mean precipitation for the period of record for Cossatot River near Vandervoort, Arkansas (blue line is a LOESS curve with 90% confidence limits (grey area)).

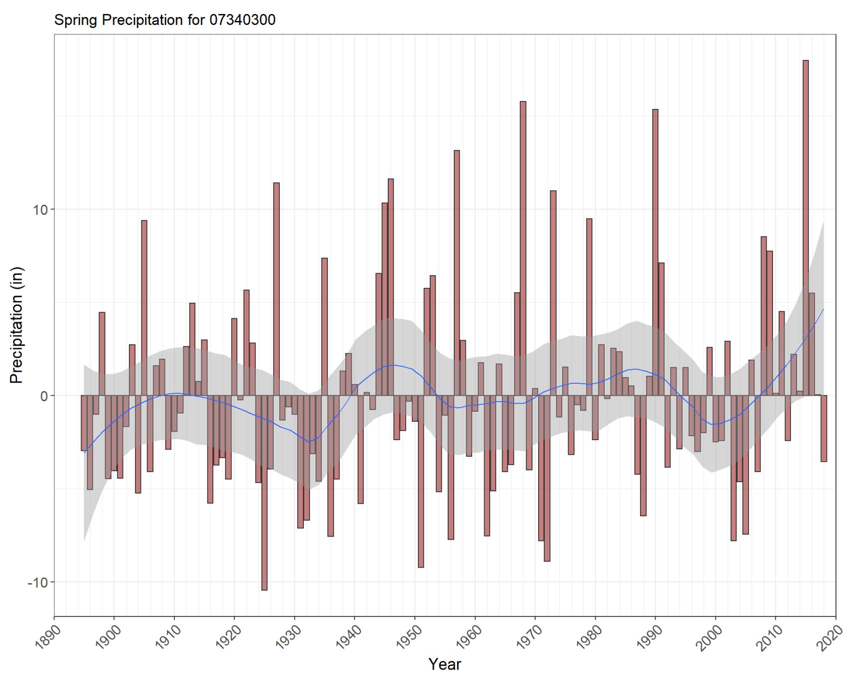

Figure 11. Spring precipitation departure from the mean precipitation for the period of record for Cossatot River near Vandervoort, Arkansas (blue line is a LOESS curve with 90% confidence limits (grey area)).

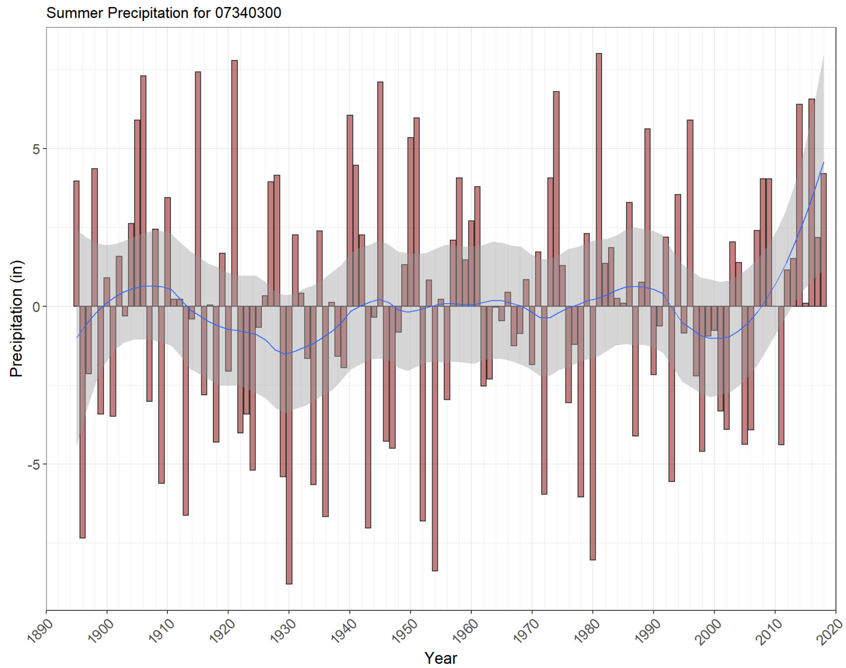

Figure 12. Summer precipitation departure from the mean precipitation for the period of record for Cossatot River near Vandervoort, Arkansas (blue line is a LOESS curve with 90% confidence limits (grey area)).

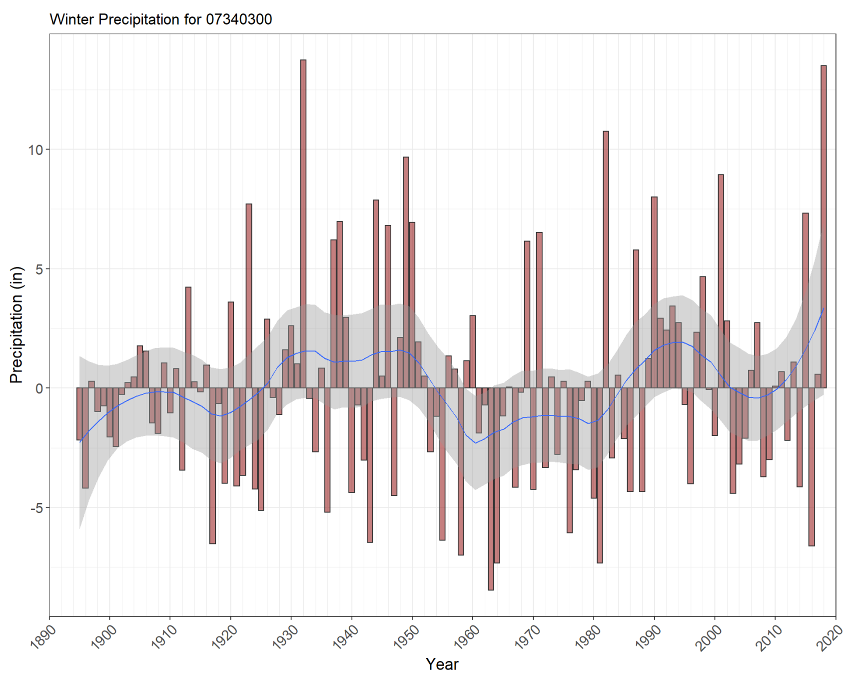

Figure 13. Winter precipitation departure from the mean precipitation for the period of record for Cossatot River near Vandervoort, Arkansas (blue line is a LOESS curve with 90% confidence limits (grey area)).

In general, for the period of record and for both sites, precipitation has increased annually, as well as seasonally. For the winter, spring and summer months, precipitation has increased substantially for approximately the last decade. The only season that shows a marked decrease in precipitation for approximately the last 30 years is the fall. However, overall, there has still been an increasing trend in fall precipitation for the entire period of record for both sites (1895-2019) (Figure 5 and Figure 10).

Land use

Land use within the Cossatot watershed, from the 2016 National Land Cover Database (NLCD; Jin et al., 2019), is predominantly forest (59 percent), particularly in the northern portion of the watershed. The southern portion of the watershed is a mix of agriculture (pasture, approximately 14 percent for the entire watershed) and forest (Figure 14). The watershed contains only approximately 5 percent of developed land and approximately 1 percent water. The remaining land cover within the watershed is equally split at 7 percent each between shrub and scrub, herbaceous, and wetlands (Figure 14). Shrub and scrub and herbaceous occur largely in regenerating clearcuts/young pine plantations.

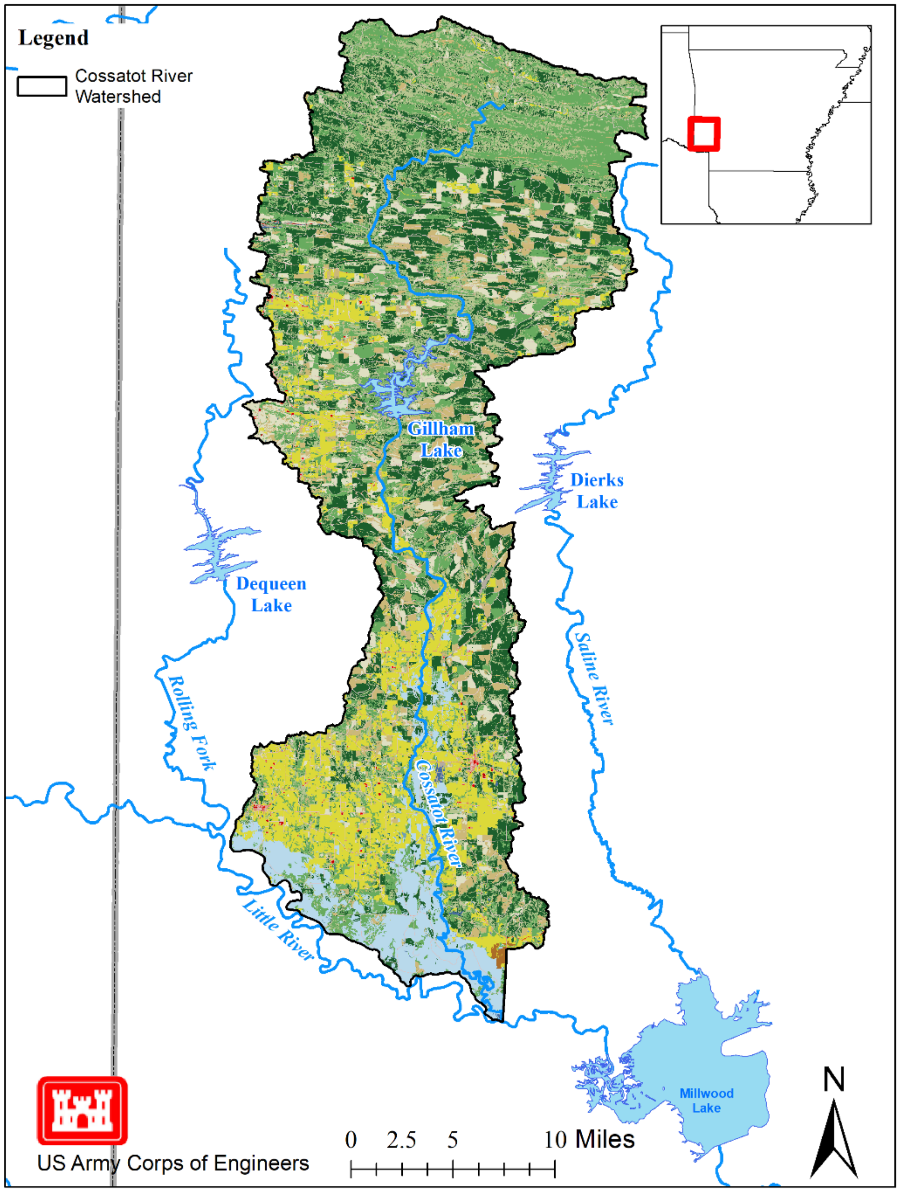

Figure 14. Land use in the Cossatot River Watershed based on 2016 data.

Most of the land use within the Cossatot watershed would be wooded under natural conditions. Currently within the basin, much of the acreage is held in extensive tracts for commercial wood crop production (loblolly pine plantation), while more permanently cleared acreage occurs as small, scattered fields on the wider parts of the ridgetops and in the stream valleys. These cleared areas are primarily fields for small farms and many of these farms produce poultry or are cleared to produce forage for or to raise livestock (Woods et al., 2004).

Physiography and geology

The Cossatot watershed is located in both the Ouachita Mountains physiographic section and the West Gulf Coastal Plain physiographic section (Fenneman, 1946; Figure 2). Fenneman (1946) broke out each of these broad-scale divisions based on geomorphology, i.e., terrain texture, rock type, and geologic structure and history. Therefore, the geology within the watershed is split approximately in half, at the "Fall Line", with the northern portion of the watershed comprising the Ouachita Mountains and the southern portion comprising the West Gulf Coastal Plain. Elevation within the watershed ranges from 2,326.9 feet above mean sea level (msl) at the northern end of the watershed to 258.7 feet above msl at the southern end of the watershed to (Figure 15). The relatively large change in elevation within the watershed is indicative of the physiography and geology within the watershed. The Ouachita Mountains consist of a series of east-west trending ridges and valleys and are composed of Early Ordovician through Middle Pennsylvanian age rocks. The valleys primarily consist of shales while the ridges primarily consist of competent sandstone, chert, and novaculite. The West Gulf Coastal Plain (Figure 2) consists of Cretaceous through Quaternary age sedimentary deposits of sand, gravel, and clay.

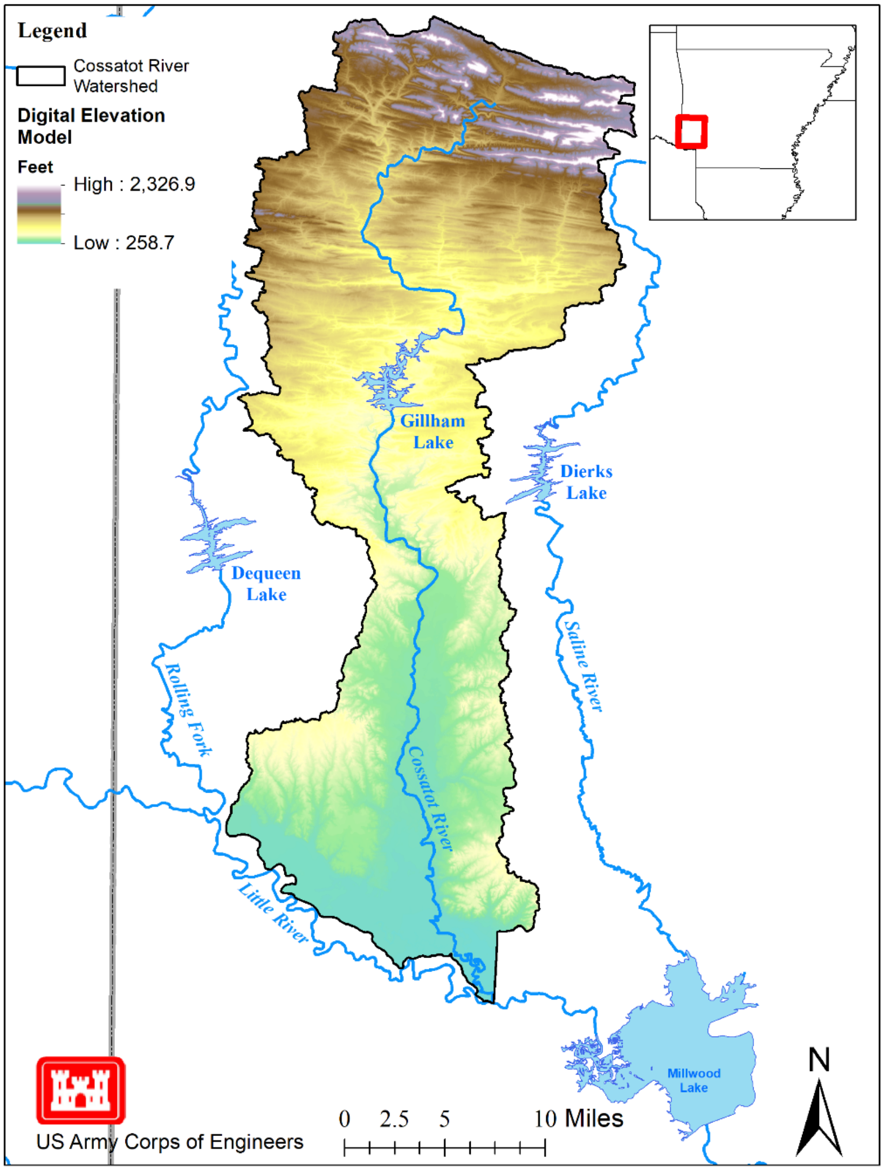

Figure 15. Digital Elevation Model for the Cossatot River Watershed and locations of the streamflow gaging stations within the watershed.

Soils

The Cossatot headwaters are in the south-central portion of Polk County, which is characterized by large east to west linear mountains intermingled with large cone-shaped hills, smaller dissected mountains, and narrow valleys. Soils in this area consist primarily of the Yanush-Bigfork-Bengal series, which are characterized as being very deep and moderately deep to deep, gently sloping to very steep, well drained, very cobbly, very stony, and clayey soils that formed in colluvium and residuum of novaculite and shale (Soil Survey Staff, 2020). This series is primarily located in mountainous and hilly areas of the uplands. Major land uses include woodland, pasture and wildlife habitat.

Upon leaving Polk County, the Cossatot drains the extreme northwest portion of Howard County. The soil formations in this part of the county are primarily the Hanceville series, located in the Athens Plateau region. These are rough stony lands occupying steep slopes and sharp ridges and these soil types are primarily used for pasturing cattle and hogs. Timber has value as well on these soil types (Soil Survey Staff, 2020).

The Cossatot flows two miles downstream of Gillham Lake before entering Sevier County. The River flows generally in a southerly direction through the east central portion of Sevier County to its confluence with Little River. Soils in the northern portion of Sevier County lie within the Ouachita Mountains Major Land Resource Area, which is characterized by tilted, folded, and fractured layers of shale, sandstone and quartzite (Soil Survey Staff, 2020). The softer, less resistant shale and impure sandstone are more susceptible to erosion and form most of the basins, valley floors and lower hills. The harder, more resistant, relatively pure sandstone and quartzite form the larger hills and ridges. The Bismarck-Littlefir-Nashoba soils complex is common in this land area (Soil Survey Staff, 2020). The southern portion of Sevier County lies within the Cretaceous Western Coastal Plain Major Land Resource Area, which is characterized by heavily dissected areas of deep marine sediments that were deposited during the Cretaceous age. The sediments are unconsolidated and range from clayey to loamy in texture. Antoine, DeQueen, Peanutrock, and Pikecity soils dominate the upper portion of these sediments. The DeAnn, Japany and Sumter soils dominate the clay, marl, and land-chalk areas (Soil Survey Staff, 2020).

Reservoir history, operations, and pertinent data

Gillham Lake has a surface area of approximately 3.4 square miles (sq mi) and is operated by the USACE. Gillham Lake is one of six reservoirs used for flood risk management and water supply in the Little River basin above Millwood Lake (Figure 1). Gillham Lake was authorized for construction by the Flood Control Act of 1958 (Public Law 85-500, 85th Congress, S. 3901) as a modification of Millwood Reservoir, authorized by the Flood Control Act of 1946 (Public Law 526, 79th Congress, Chapter 596, 2d Session, H.R. 6597). Project purposes of Gillham Lake, other than flood risk management and water supply, are environmental stewardship and recreation; however, pool storage is not allocated for these latter two purposes, and although recreation was not approved as a specific project purpose, it was included as incidental to the basic authorized purposes.

Gillham Dam construction was started in June 1963 with work on the right access road. The first concrete in the spillway was poured in November 1968. Work was halted in February 1971 but resumed in August of 1972 and the dam began storing water 8 May 1975.

Gillham Dam has a length of 1,750 feet and consists of a rockfill with a rolled earth core. The Dam reaches a maximum height of 160 feet at an elevation of 586 feet. The outlet works consist of a single, 10-foot diameter, concrete-lined conduit through the right abutment. The conduit has a discharge capacity of 2,240 cubic feet per second (cfs) when the lake elevation is at the top of the inactive pool (464.5 feet). The discharge capacity of the conduit is 3,468 cfs when the lake elevation is at the top of the conservation pool (502 feet) (Table 1). Although it is physically possible to obtain more flow out of Gillham Lake at certain pool levels and gate openings, a maximum release of 3,000 cfs is used due to downstream regulation constraints and would only be exceeded in a surcharge operation.

Low flow releases from Gillham Lake are made through a multilevel intake tower equipped with two 30 inch by 48 inch slide gated ports at elevations of 472 feet and 487 feet, respectively. Releases from the tower are controlled by a hydraulically operated butterfly valve in the 30-inch low flow discharge pipe. The capacity of the low flow discharge pipe is approximately 150 cfs at the top of conservation pool (elevation of 502 feet) and approximately 245 cfs at the top of the flood control pool (elevation of 569 feet) (Table 1).

Table 1. Pool elevations and corresponding outflows when intake 1 and intake 2 and the low flow intake are implemented.

Pool elevation (ft)	Intake 1 at elevation 472 feet (flow values in cfs)	Intake 2 at elevation 487 feet (flow values in cfs)	Low Flow intake (cfs)
502	2,690	3,468	150.9
514	2,951	3,770	167.7
527	3,208	4,068	185.9
545	3,533	4,449	211.1
558	3,750	4,703	229.3
569	3,923	4,892	244.7
583	4,112	5,109	263.0

Recreation

Gillham Lake has five recreation areas that are managed by the USACE. Three of the five offer class A camping and the others vary with hiking trails, boat ramps, swimming and other amenities. Northeast of the main body of the Lake, on the Cossatot, is Cossatot State Park Natural Area, which are lands leased from the USACE and managed by the State. The State Park and Natural Area extend for 12 miles along the Cossatot. The River forms Cossatot Falls, a rocky canyon with Class IV/Class V rapids for experienced kayakers and canoeists, when local rainfall increases in the watershed. Brushy Creek Recreation Area offers picnic sites, restrooms, and river access. Tent sites without water and electric hookups are located at the Cossatot Falls Area, Sandbar Area, and Ed Banks Area, for a total of 23 sites. A primitive camping area, with no water or electricity, accommodates group camping with tent sites, a pavilion, restroom, fire pit, grill, and river access all available by reservation. There are four scenic trails spanning nearly 20 miles and a visitor center featuring exhibits and a wildlife observation room. Interpretive programs are offered year-round (ArkansasStateParks.com).

The word Cossatot is French and means “crushed head” and adequately describes what the National Park Service postulates as “probably the most challenging” white-water float in the state. The Cossatot River State Park-Natural Area preserves a 12-mile stretch of the stream that includes the much-photographed Cossatot Falls area (Arkansas.com). The Cossatot travels through the Ouachita National Forest, along-side a wilderness area, and over and around upended layers of jagged bedrock. This last characteristic is what gives the stream its Class IV/Class V rating among white-water enthusiasts as well as its National Wild and Scenic River designation (Rivers.gov, 2020). Below the park, the stream borders the Howard County Wildlife Management Area where it is impounded to form Gillham Lake. Below the lake, the Cossatot continues south past De Queen before joining Little River just above Millwood Lake. Anglers including fly fishers will find smallmouth, largemouth, and spotted bass, crappie, channel and flathead catfish and various species of sunfish in the river below the lake, the stream's warmer waters (Arkansas.com 2020).

Hydrology

As previously stated, there are two USGS stream gaging sites on the Cossatot: 07340300 Cossatot River near Vandervoort, Arkansas, (hereafter referred to as the Vandervoort site or gage) and 07340500 Cossatot River near DeQueen, Arkansas, (hereafter referred to as the DeQueen site or gage) (Figure 2). The Vandervoort gage is located approximately 20.5 miles upstream of Gillham Dam at an elevation of 786 feet above msl and the DeQueen gage is located approximately 14.9 miles below Gillham Dam at 381 feet above msl (Figure 2). The Vandervoort gage captures the drainage area of approximately 89.6 square miles and the DeQueen gage approximately 361 square miles. The Vandervoort gage has mean daily streamflow values dating back to 1967 and, although the DeQueen gage has mean daily streamflow values dating back to 1938, the gage has missing streamflow data beginning in October of 1980 through September of 2012.

Because of the location of the Vandervoort site versus the DeQueen site, above the Dam and below the Dam, respectively, the DeQueen site was the only site used in the analysis of streamflow statistics and the comparison of historical (prior to dam construction) to current (after dam construction) streamflow characteristics. However, streamflow trends will still be discussed for the Vandervoort site, as well.

Historical

Streamflow statistics for the DeQueen site were calculated for the historical streamflow data, which is considered the time period from when the DeQueen gage began collecting data in 1 October 1938 to 30 September 1967 right before the first concrete was poured for the spillway.

Current

Using the release data from Gillham, a drainage area ratio of 1.32 was applied to fill in the missing streamflow data at the DeQueen gage beginning after the construction of Gillham Dam. Streamflow statistics for the observed streamflow were calculated using the combined current streamflow data (1975-2019) and the surrogate release data from Gillham Lake. For the DeQueen site, current streamflow data is considered the time period from when outflow data below the dam began being collected, 1 October 1975 to 31 September 2019.

Daily streamflow

The extremes in daily streamflow data have been altered at the DeQueen site since the construction of Gillham Dam. By looking at the daily streamflow data for the DeQueen site (Figure 16), peak streamflows have been reduced (less “dark blue”), on any given day, beginning after the mid-1970s which coincides with the construction of the dam. Additionally, there have been fewer low flow days (less “red”), during the same time period. However, for the Vandervoort site (Figure 17), daily streamflow data have not changed much, on any given day, for the period of record.

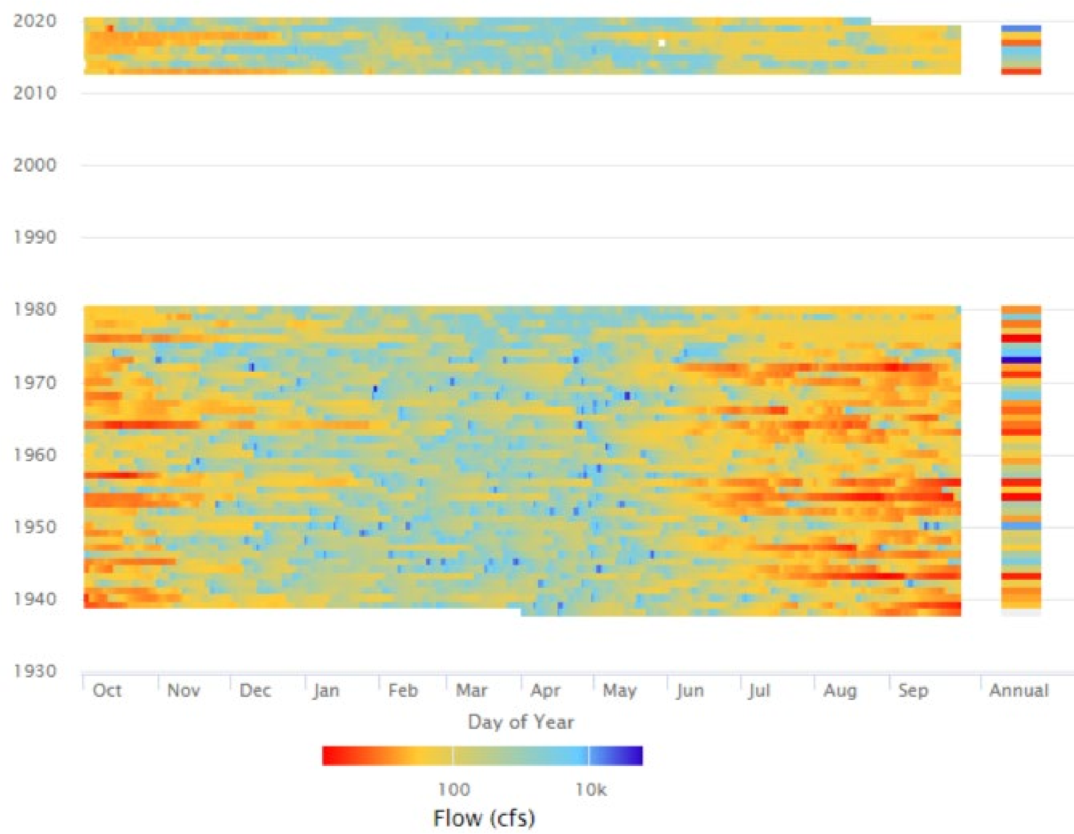

Figure 16. Cossatot near DeQueen, Arkansas, daily streamflow data. Plot obtained from the National Weather Service River Forecast Center.

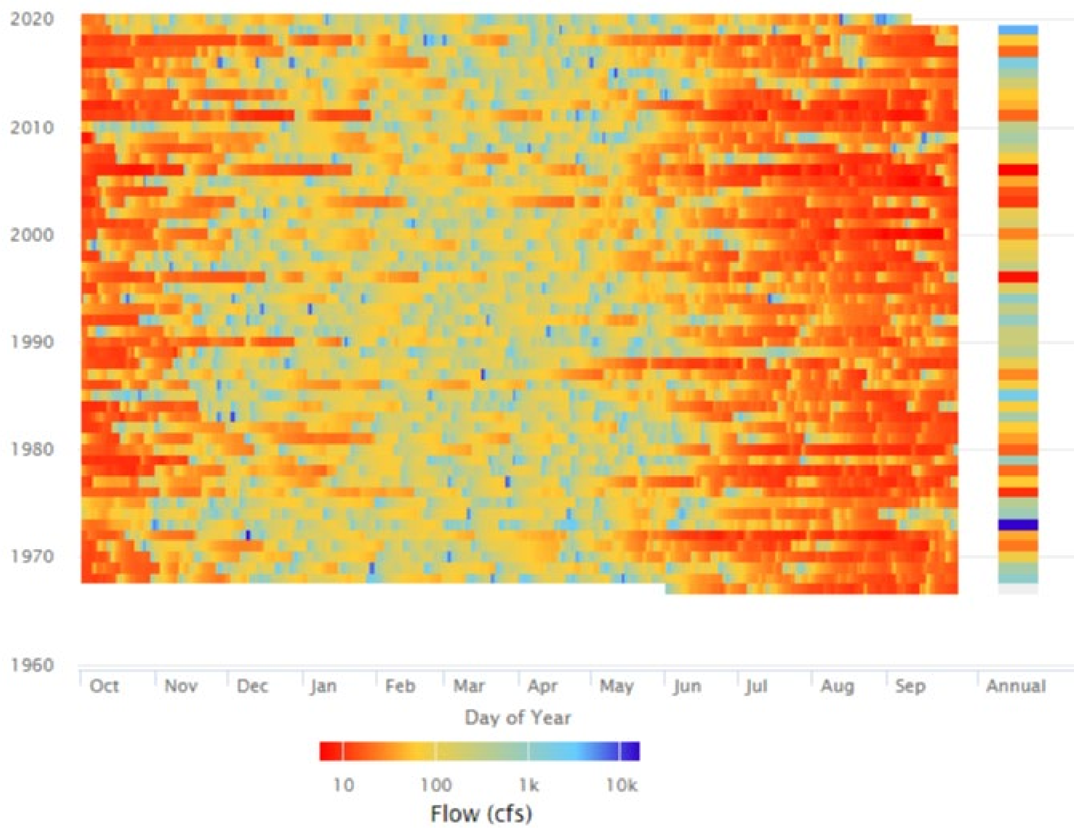

Figure 17. Cossatot near Vandervoort, Arkansas, daily streamflow data. Plot obtained from the National Weather Service River Forecast Center.

Flood frequency and peak flows

The frequency of large floods has been reduced since the construction of Gillham Dam. All of the major floods occurred before 1971 below Gillham Dam as recorded at the DeQueen site (Figure 18 and Figure 19). The largest flood occurred in 1968 with every other flood after 1971 being below the 2-year recurrence interval. However, for the Vandervoort site (Figure 20 and Figure 21), large floods (50-year recurrence interval) have occurred as recently as 2009. Although flood events are directly related to precipitation location and intensity, prior to dam construction, the same flood events can be seen between the two Cossatot sites, for example, 1961, 1968, 1969, and 1971. However, since 1971, there have not been any large events at the DeQueen site, but there have been several at the Vandervoort site (Figure 18 and Figure 20, respectively). This indicates there more than likely is a hydrologic disconnect between what is occurring upstream of Gillham Dam and what is occurring downstream. In other words, Gillham Dam was constructed for flood risk management and is removing all the large floods from the system downstream.

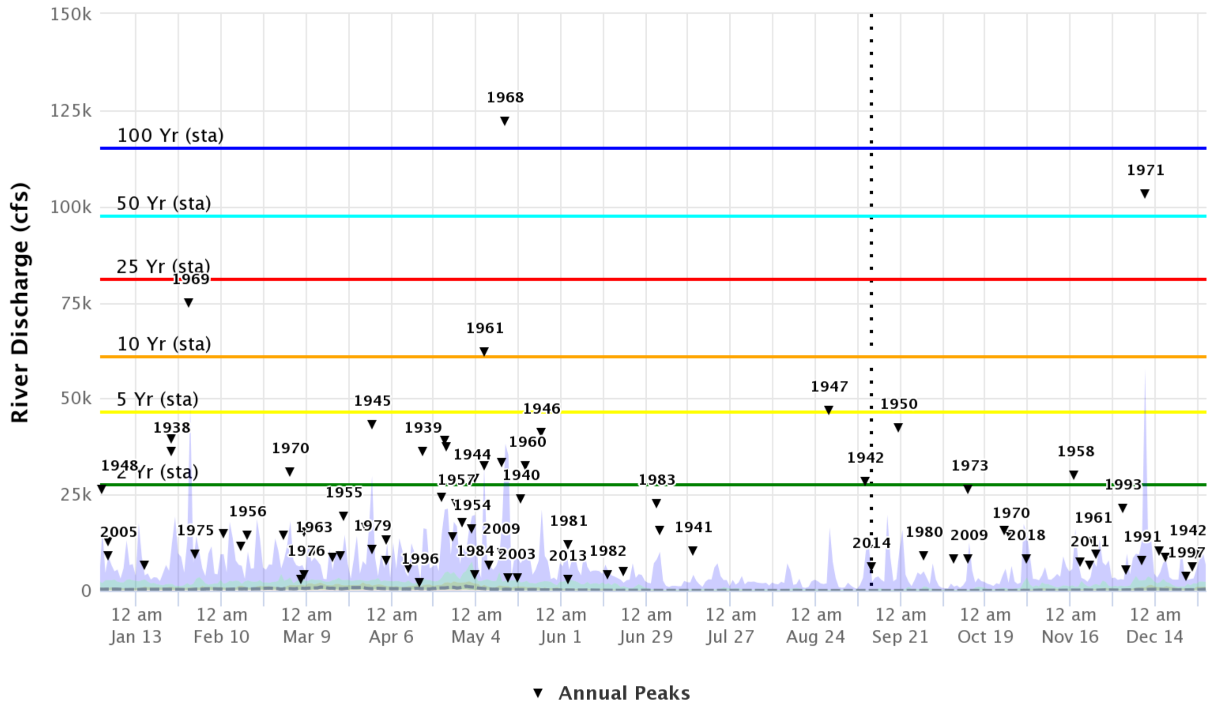

Figure 18. Cossatot near DeQueen, Arkansas, flood frequency data. Plot obtained from the National Weather Service River Forecast Center. Note that the latest date for an event larger than 2-year recurrence interval (above the green line) is 1971. (Dotted vertical line represents the date the data was retrieved.)

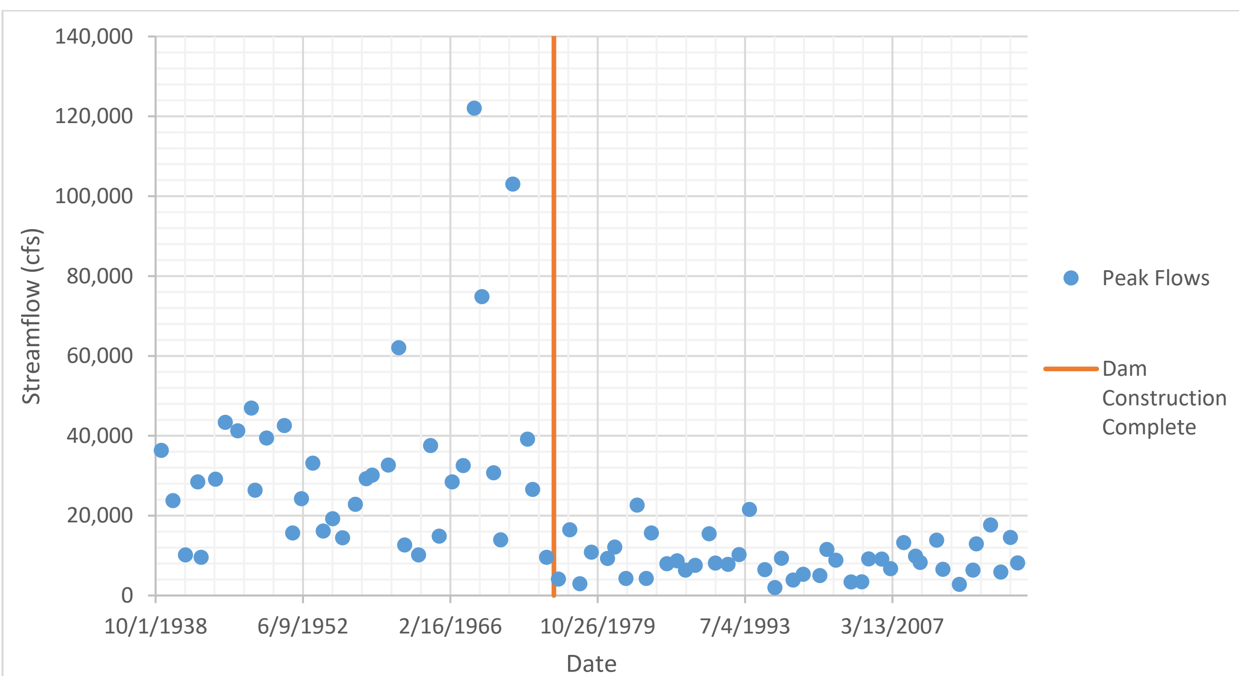

Figure 19. Cossatot near DeQueen, Arkansas, peak flow data.

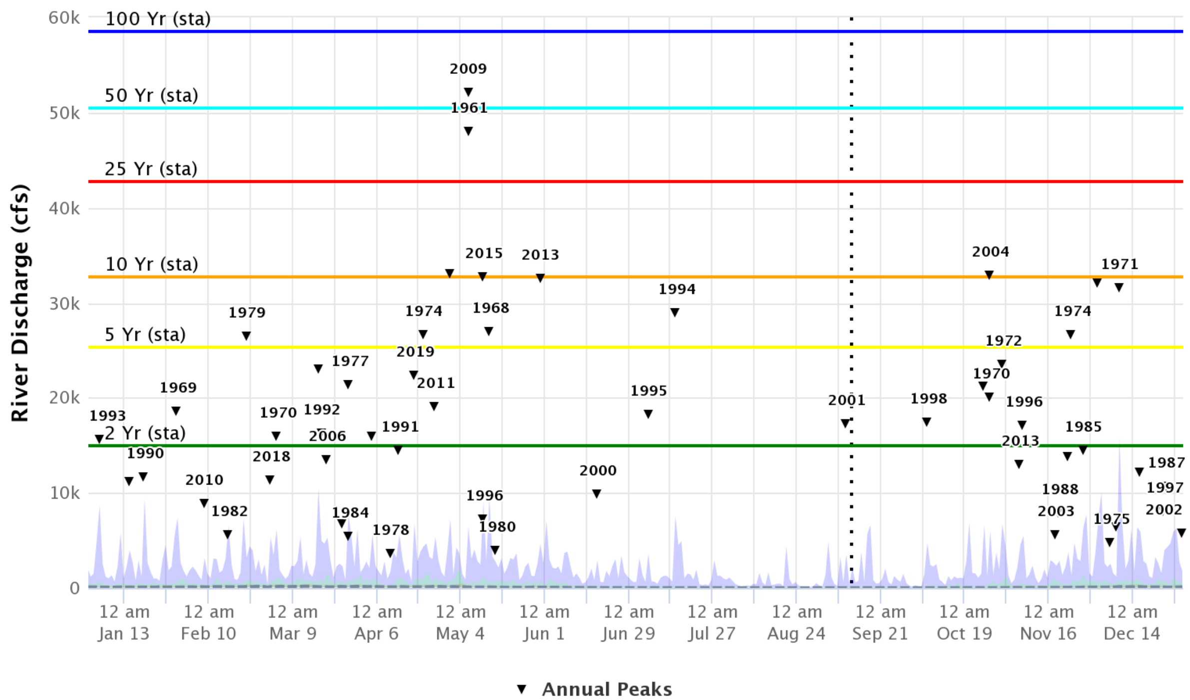

Figure 20. Cossatot near Vandervoort flood frequency data. Plot obtained from the National Weather Service River Forecast Center. Note that large flood events have occurred as late as 2009. (Dotted vertical line represents the date the data was retrieved.)

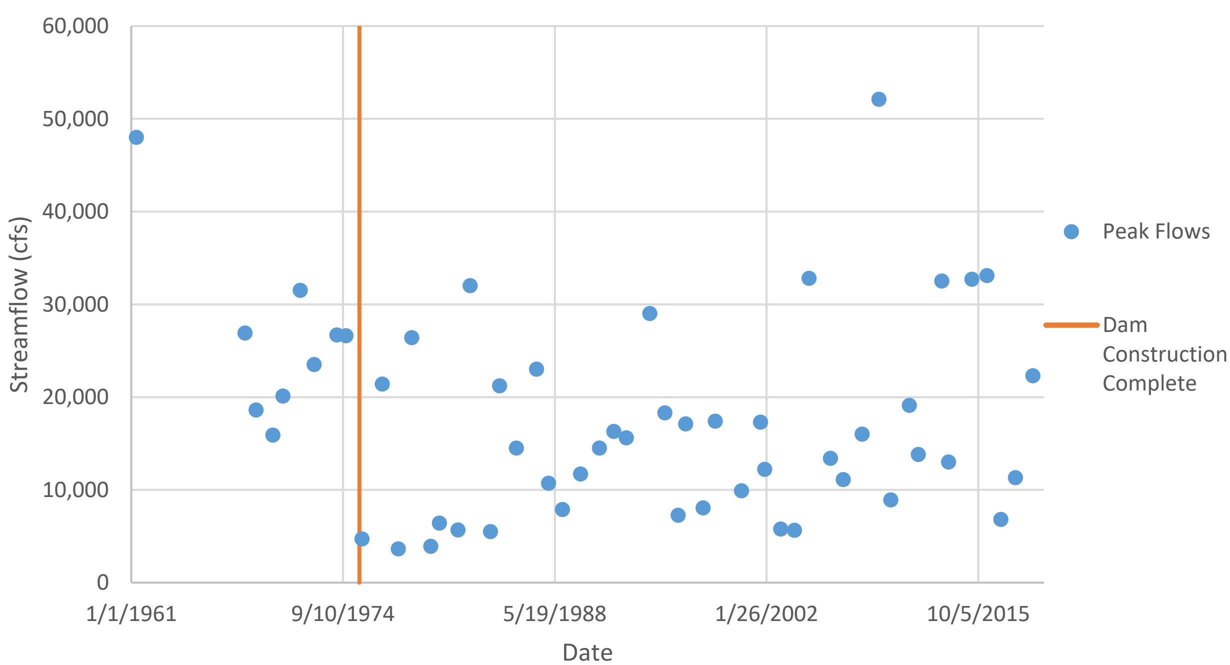

Figure 21. Cossatot near Vandervoort, Arkansas, peak flow data.

Low flows

Along with high flows and peak flows, low flow natural variability, in terms of magnitude, frequency, and duration, have important ecological functions. Baseflow and extreme low flow values are unique to each stream and these hydrologic flow regimes are needed to sustain good habitat and maintain suitable water quality (Yin et al., 2010). The variability of low flows has decreased since the construction of Gillham Dam (Figure 22). Prior to the construction of Gillham Dam, the minimum 1-day streamflow values ranged from 1.2 to 38.0 cfs with an average of 9.8 cfs. Since the construction of the Dam (1 October 1975) the minimum 1-day streamflow values ranged from 0.2 to 41.6 cfs with an average of 17.8 cfs (Figure 22). Furthermore, an increase in the minimum low flows can be seen in the 3-day, 7-day, and 30-day minimum streamflow values beginning in the mid-1970s, coinciding with the construction of Gillham Dam (Figure 23).

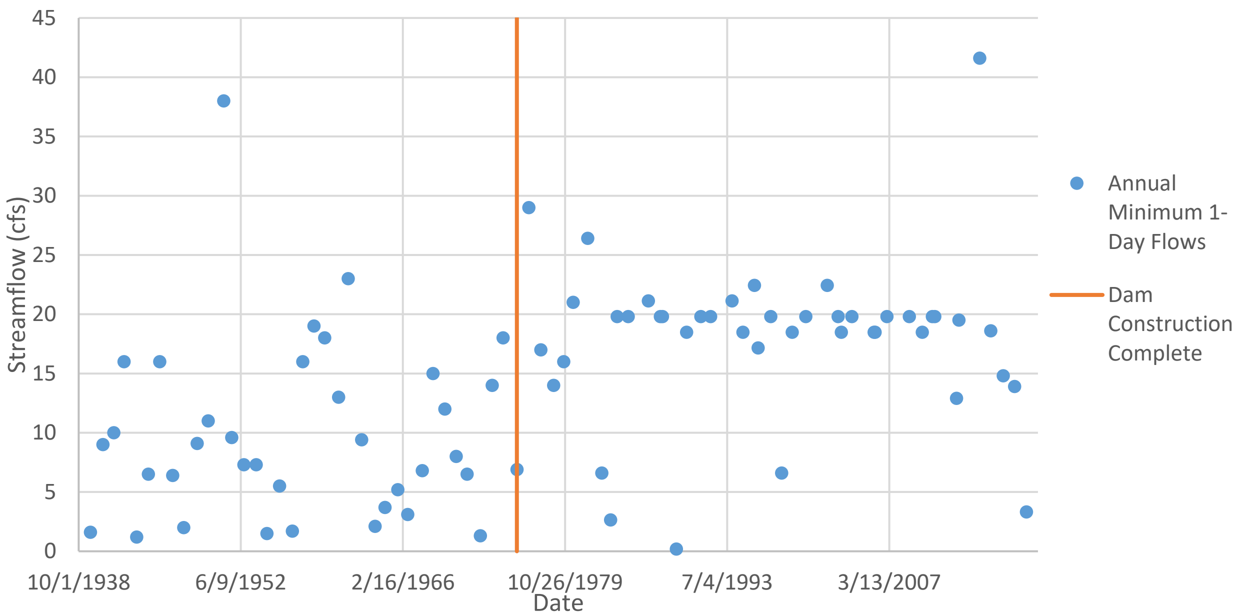

Figure 22. Cossatot near DeQueen, Arkansas, annual minimum 1-day flow data.

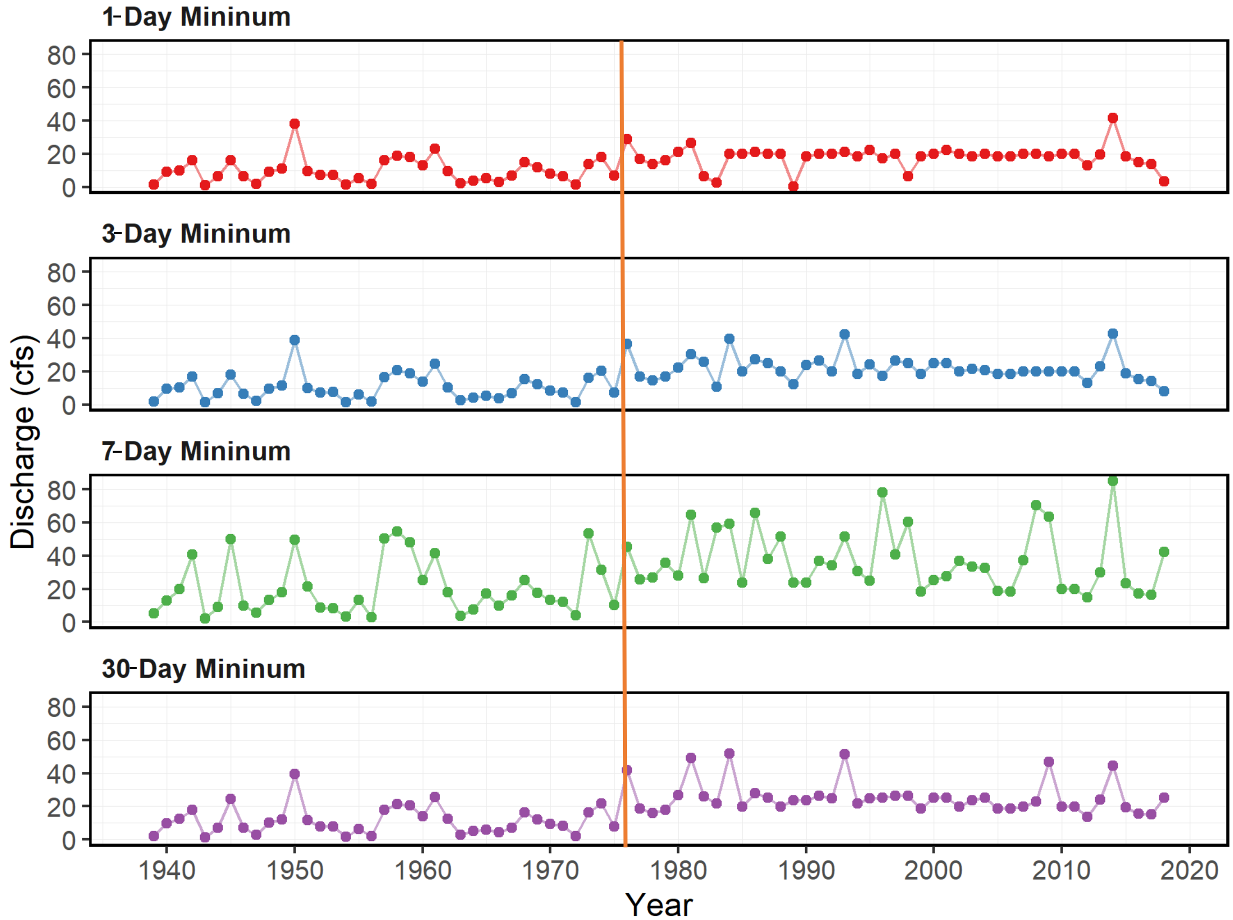

Figure 23. Minimum flows prior to and after Gillham Dam construction for Cossatot near DeQueen, Arkansas (vertical orange line is approximate date of dam completion (October 1, 1975)).

Gillham Lake

Gillham Lake water-level elevation needs to be considered to maintain healthy ecological functions. For example, the only known population for the Leopard Darter in the Cossatot is located upstream of Gillham Lake. Extreme inundation from high lake elevations could have detrimental impacts to this Leopard Darter habitat. Figure 24 provides the percent exceedance for Gillham Lake elevation based on historical elevation data and Figure 25 provides the annual exceedance probability, again based on Gillham Lake historical elevation data.

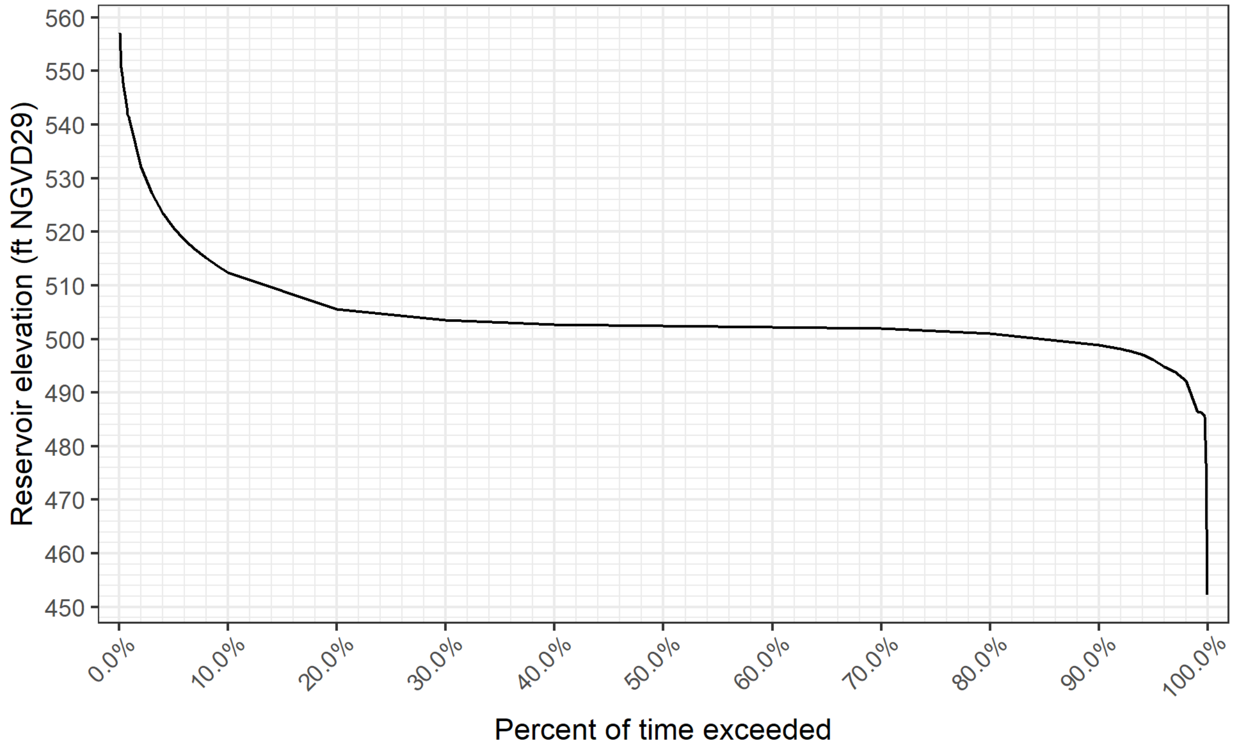

Figure 24. Gillham Lake elevation percent exceedance.

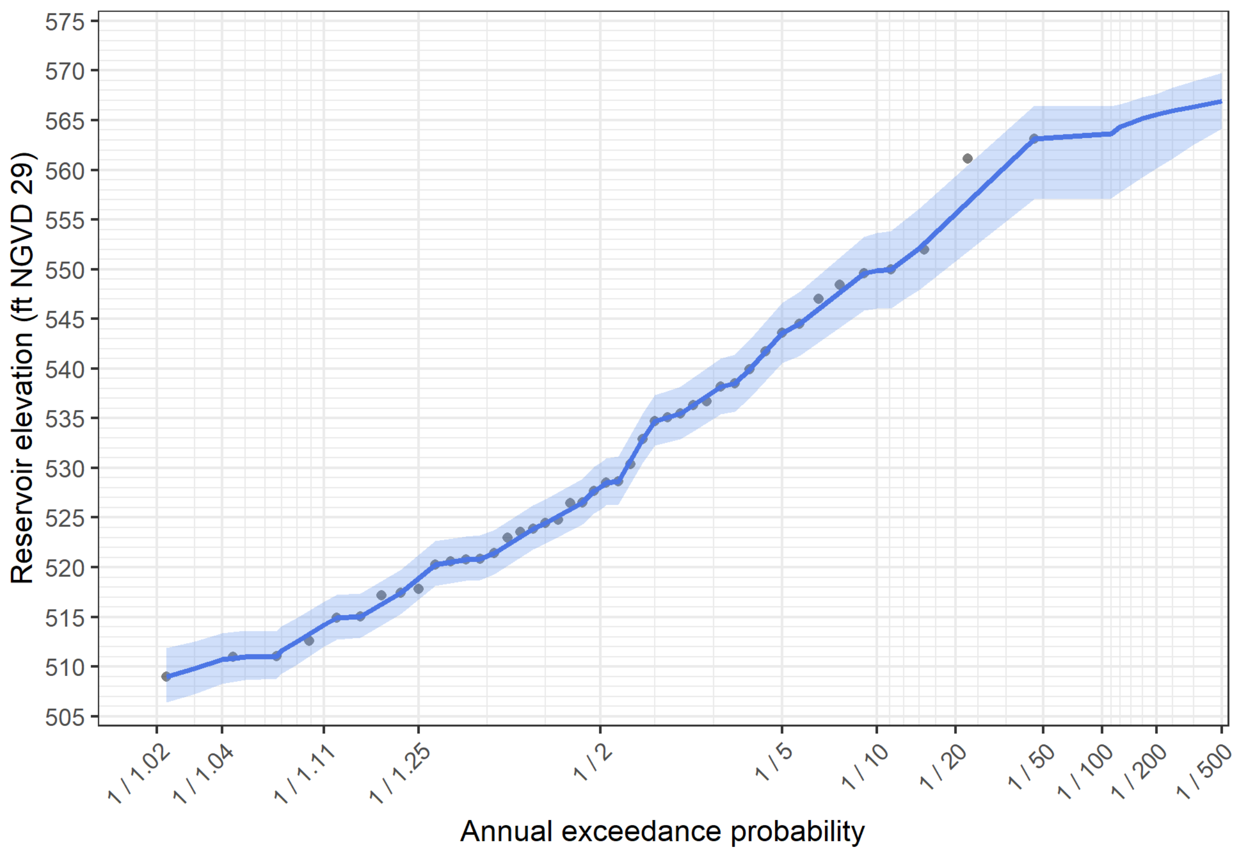

Figure 25. Gillham Lake elevation annual exceedance probability.

Change in flow regime
Regulated and unregulated flows
To determine the change in flow regime, comparison of regulated to unregulated flows needed to be conducted. Unregulated flows were developed for the DeQueen site using the R language for statistical computing (R Core Team, 2020), and multiple machine learning models within R (Breaker, 2020). Regression analysis was performed using Random Forest (RF; Liaw & Wiener, 2002) through model training. RF regression methods (Kuhn and Johnson, 2016) were used to model the relationship between explanatory variables of climate and land use and daily mean streamflow for the unregulated period of streamflow at Cossatot near DeQueen based on reduction in Root Mean Square Error (RMSE). The model produced daily mean streamflow for the unregulated period, prior to dam construction (pre-1975) and the regulated period, after dam construction (post-1975). For a detailed description of RF, see Cutler et al. (2007) and Liaw and Wiener (2002). The final RF model used 800 trees to grow and 24 variables that were randomly sampled at each split (Figure 26).

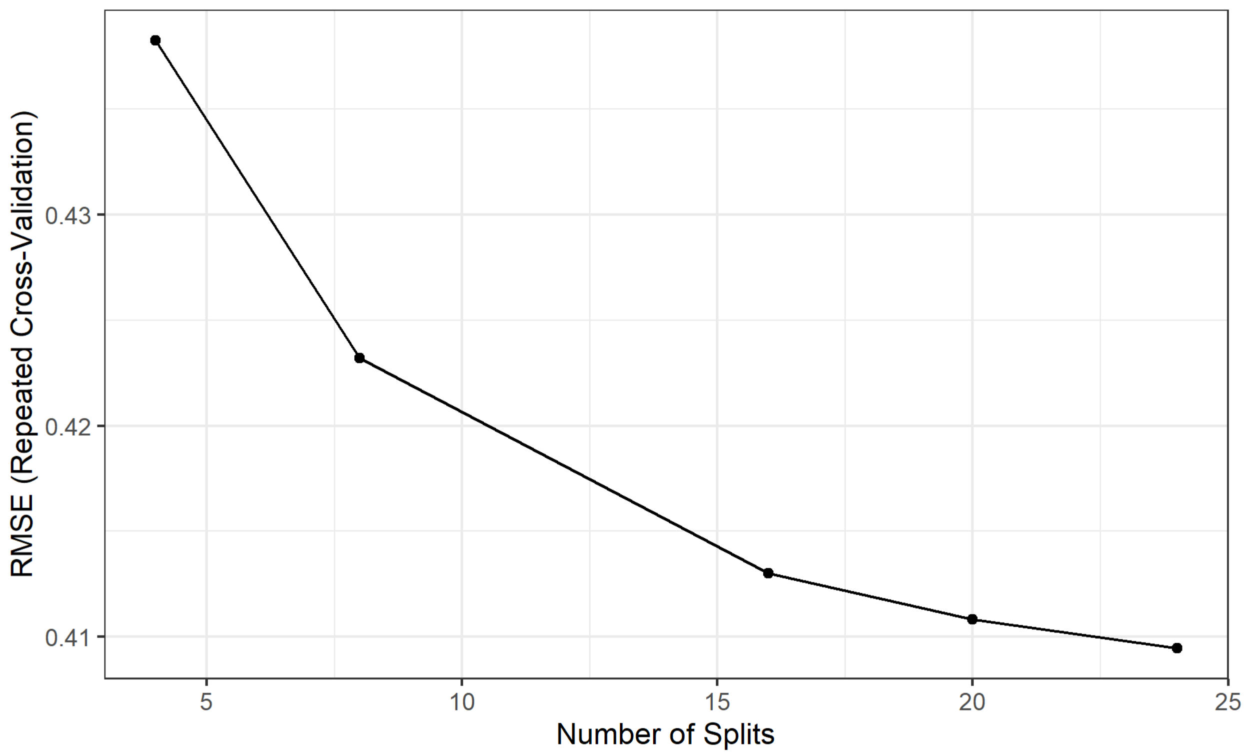

Figure 26. Graph showing reduction in RMSE for training Random Forest model for Cossatot River near DeQueen, Arkansas.

Prior to any machine learning modeling, climate and land use data had to first be aggregated. The daily climate data were obtained from NOAA's National Climate Data Center (NCDC) for the DeQueen, Arkansas, station, the Newhope, Arkansas, station, and the Nashville, Arkansas, station (Figure 1). The entire period of record was obtained for each climate station and later trimmed to match the period of record representing the most complete climate dataset (water years 1941-2019). Precipitation and minimum and maximum temperature were obtained for the DeQueen climate station and only precipitation was obtained for the Nashville and Newhope climate stations. A Fourier series, using the "Forecast" package (Hyndman and Khandakar, 2008) within R, transformed the observed temperature data for both minimum and maximum into a sinusoid for prediction of the missing temperature values. Both the missing temperature and precipitation data for the DeQueen climate station were then predicted using a generalized additive model (GAM) within R (Wood and Augustin, 2002). The Nashville and Newhope climate stations were in relatively close proximity to the DeQueen climate station (Figure 1), and, therefore, were good candidates for predicting precipitation at DeQueen using machine learning. Additionally, the Hargreaves method was used to calculate evapotranspiration (ET) based on data from the DeQueen climate station. Furthermore, the previous 1 through 7 days precipitation for the DeQueen, Nashville, and Newhope climate stations, previous 1 through 7 days ET values for DeQueen, and the total precipitation for all three climate stations for the previous 3 through 5 days were also determined. In the end, a complete record of climate was developed for the period consisting of water years 1941-2019.

The historical and current land use data were obtained from the USGS Earth Resources Observation and Science (EROS) Center. EROS provides yearly land use data extending back to 1938 to present (2019) (Sohl et. al, 2018). The 14 land use categories ultimately were grouped into 4 categories representing forest, developed, agriculture, and wetlands with percentages of each category within the watershed calculated for each year (Figure 27). As can be seen in Figure 27, forest is the predominant land use type within the watershed. There is a sharp decrease in the percent of forested land in 1992, but this is a relic of the way the land use data were modeled and not representative of what actually occurred in the watershed (see Sohl et. al, 2018, for further explanation). However, there is a notable increase of agricultural land in the late 1960s and a subsequent decrease in forest, which coincides with other studies in Arkansas (Clark and Hart, 2009).

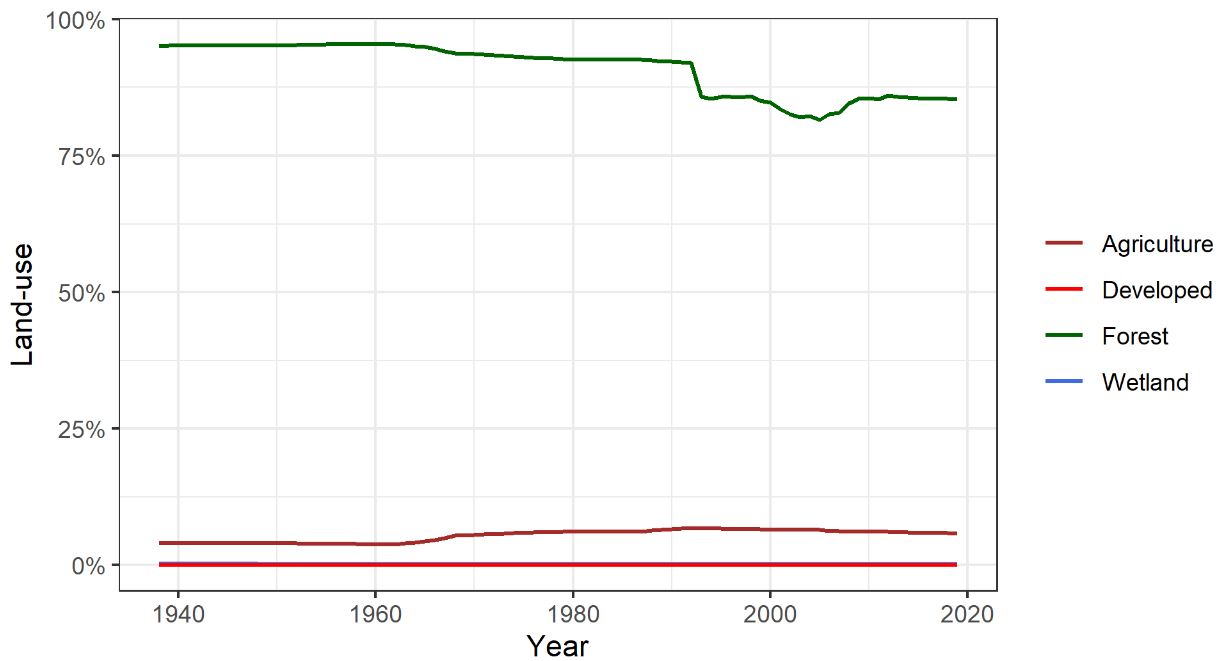

Figure 27. U.S. Geological Survey Earth Resources Observation and Science Center aggregated historical and current land use for the Cossatot watershed.

For the explanatory variables, as previously described, climate explained a majority of the importance for the RF model. The total five-day precipitation for the DeQueen climate station was the greatest variable of importance, followed by the total five-day precipitation for the Nashville climate station, the previous 7-day ET value, the total five-day precipitation for the Newhope climate station, the total four-day precipitation for the DeQueen climate station, previous 0-day ET value, and the minimum temperature value for the DeQueen climate station (Figure 28). Land use (Forest) did not become an important variable until the 10th variable (Figure 28).

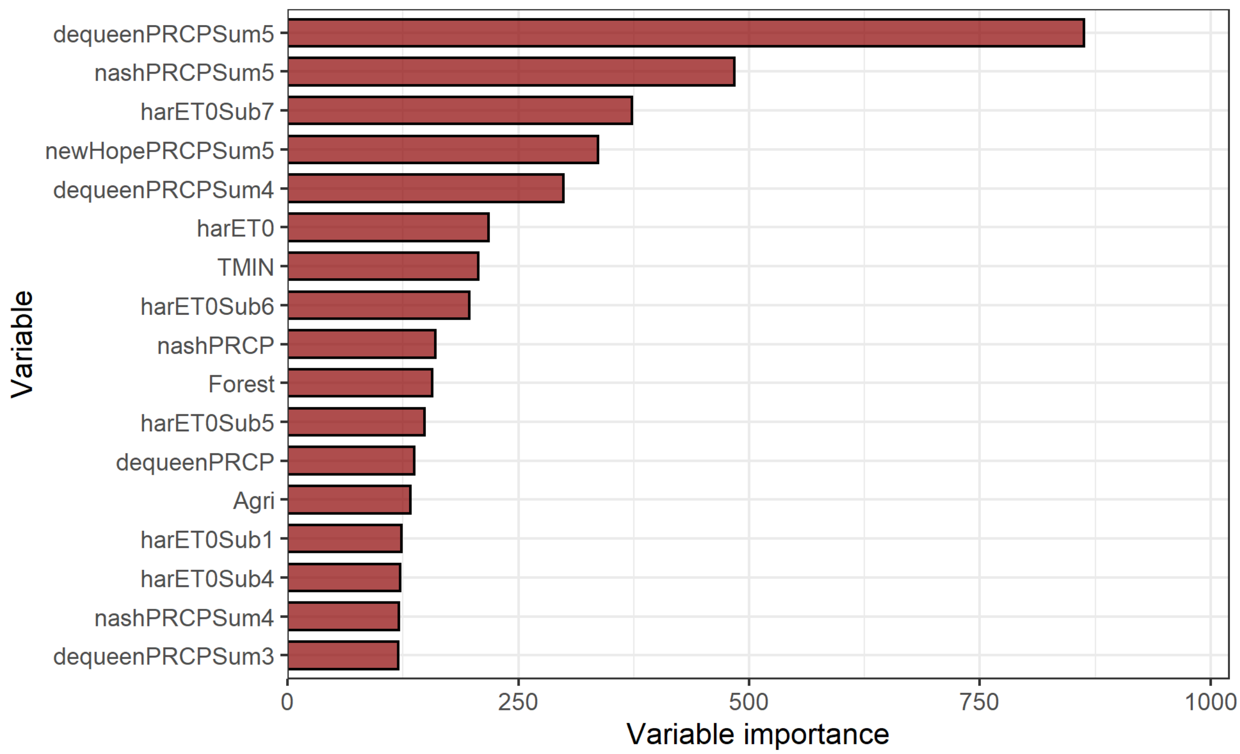

Figure 28. Variables of importance for determining unregulated daily streamflow for Cossatot near DeQueen gaging station from the random forest modeling.

After completion of the RF modeling, GAMs were used with the explanatory variables and predicted streamflow values to correct bias observed at the tails of the RF regression estimates and to reduce variability in the predicted values of streamflow (Breaker, 2020). All the 800 trees for each of the RF predictions resulted in 800 predicted values for each of the daily time steps. An iterative process was used within R to find the quantile of the 800 predicted values which was closest to the observed value. The quantile was then compared to the predicted values and GAMs were developed to predict new quantiles from the 800 tree values to estimate a new predicted streamflow value (Breaker, 2020).

Together, the RF and GAM modeling proved to be an excellent means to develop unregulated streamflows for Cossatot near DeQueen. The predicted streamflows matched the observed flows for the unregulated period extremely well. Table 2 provides the goodness of fit statistics for the bias corrected unregulated observed versus predicted streamflow values. Because the predicted streamflows matched the observed flows for the unregulated period so well (Figure 29), there is high confidence in the unregulated streamflow predictions for the regulated time period (Figure 30). Therefore, there is high confidence in the ability to compare streamflow characteristics between the observed regulated streamflow and the predicted unregulated streamflow.

Table 2. Bias corrected goodness of fit statistics for the observed versus predicted unregulated streamflows for Cossatot River near DeQueen, Arkansas.

Site	Mean Absolute Error (MAE)	Normalized Root Mean Square Error (NRMSE)	Percent bias (PBIAS)	RMSE-observations standard deviation ratio (RSR)	Nash Sutcliffe Efficiency (NSE)	Modified NSE (mNSE)	Coefficient of Determination (R^2)	R^2 multiplied by the slope of the linear regression between 'sim' and 'obs' (bR^2)
Cossatot near DeQueen	0.02	6.1	0.4	0.06	1	0.97	1	0.99

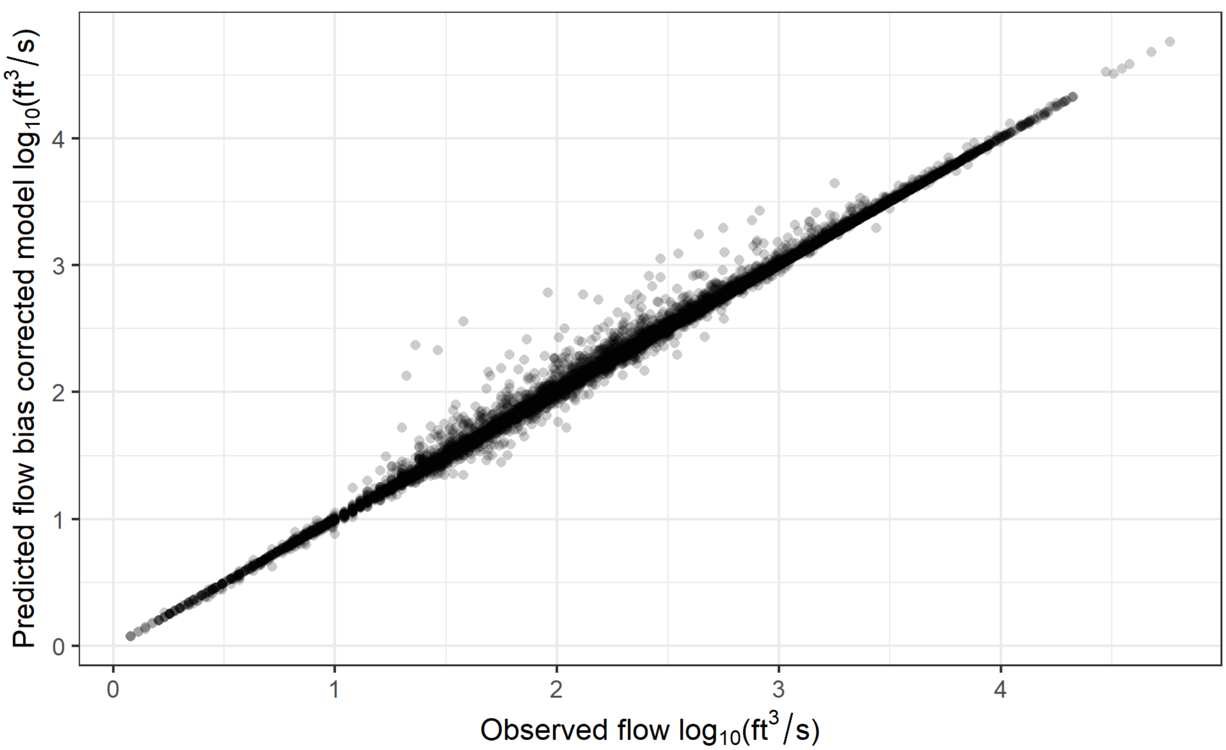

Figure 29. Bias corrected predicted streamflow versus observed streamflow for Cossatot River near DeQueen, Arkansas.

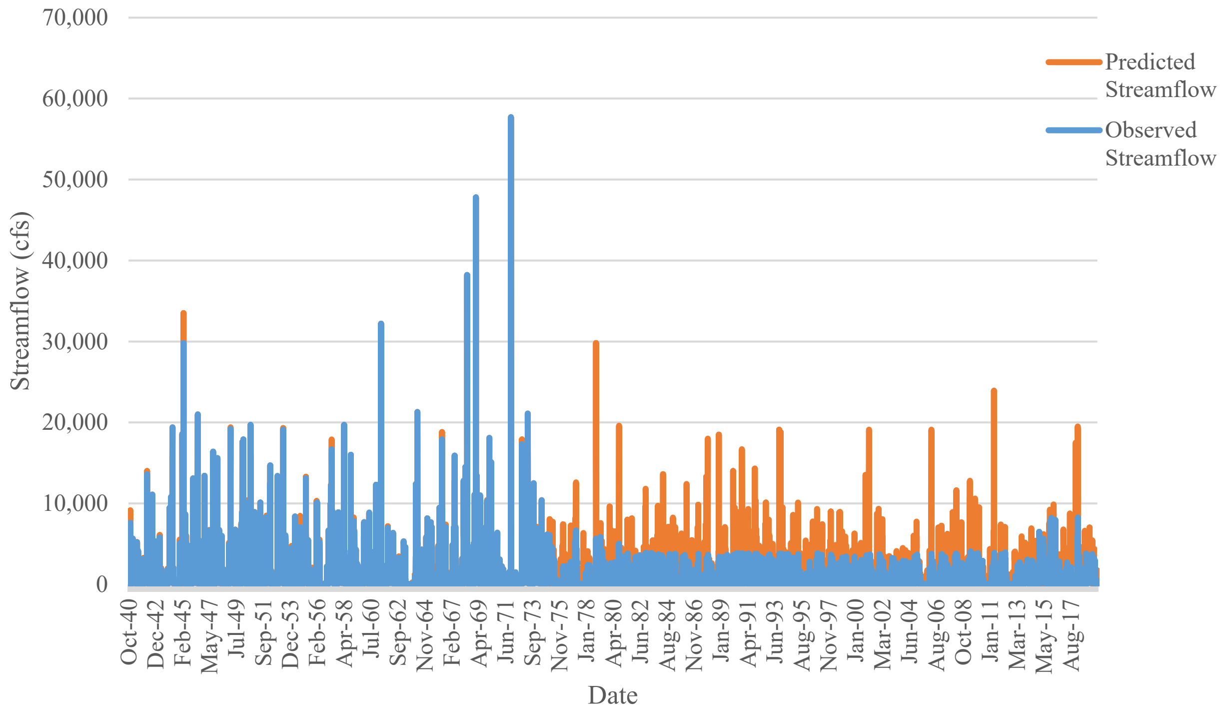

Figure 30. Observed and predicted streamflow for Cossatot River near DeQueen, Arkansas.

Streamflow characteristics

Ecologically important streamflow characteristics that make up the flow regime include seasonal patterns of flow; timing of extreme flows; the frequency, predictability, and duration of floods, droughts, and intermittent flows; daily, seasonal, and annual flow variability; and rates of change (Olden and Poff, 2003), and assessment of these characteristics is essential for understanding the biological impact of both natural and altered flow regimes on riverine biota (Olden and Poff, 2003). Defining changes in the flow regime can help guide restoration of highly flow-altered rivers concerned with the restoration of key ecologically and socially relevant flow characteristics that have been lost through regulation (Poff et al., 2017). To date, 171 hydrologic indices have been developed to characterize each of the streamflow characteristics within the flow regime to help define biological impact. A subset of these available hydrologic indices was developed in order to adequately describe the critical attributes of the flow regime for the Cossatot. Given that there are 171 different hydrologic indices, Olden and Poff (2003) have provided statistically sound recommendations on which hydrologic indices can be used to adequately characterize flow regimes in a non-redundant manner for a particular stream type. Therefore, the hydrologic indices that were deemed the most hydroecologically important for the Cossatot were based on the stream type categorized (as described above) by Turton et al. (2009) and are the most desirable in terms of being unique or non-redundant as determined by Olden and Poff (2003). The DeQueen site was classified as a perennial flashy stream and the Vandervoort site as a perennial run-off stream (Turton et al., 2009). For the purposes of determining hydrologic indices, perennial flashy and perennial runoff are considered in the same stream classification as described in Olden and Poff (2003). The final selected streamflow characteristics are common indices used to represent biologically relevant streamflow attributes and are considered suitable specifically for perennial stream types with a few exceptions considered suitable for all stream types (Olden & Poff, 2003; Table 3). Also included are hydrological indices that have ecological significance or just represent generalized streamflow characteristics, such as mean annual streamflow. The descriptions and results of the selected hydrological indices used to evaluate the different streamflow characteristics are presented in Table 3.

Table 3. Streamflow statistics of importance for Cossatot River near DeQueen, Arkansas.

Statistic	Description	Observed value prior to dam construction	Predicted value prior to dam construction	Hydrologic alteration factor prior to dam construction	Observed value after dam construction	Predicted value after dam construction	Hydrologic alteration factor after dam construction
Magnitude							
MA1	Mean daily flow values for the entire flow record (cfs)	588.93	601.97	-0.02	672.12	719.66	-0.07
MA2	Median of the daily mean flow values for the entire flow record (cfs)	146.00	152.00	-0.04	190.00	288.00	-0.34
MA5[1]	Skewness in daily flows: Computed as the mean for the entire flow record (MA1) divided by the median (MA2) for the entire flow record (dimensionless)	4.03	3.96	0.02	3.54	2.50	0.42
MA10	Spread in daily flows: Computed using the 20th and 80th percentiles (dimensionless)	0.62	0.63	0.00	0.60	0.49	0.23
MA26	Variability in March flow values: Variability (coefficient of variation) of monthly March flow values (dimensionless)	102.65	102.58	0.00	77.20	105.28	-0.27
MA41	Mean annual runoff: The mean annual flow divided by the drainage area (cfs per square mile)	1.63	1.67	-0.02	1.86	1.99	-0.07
ML6	Minimum June streamflow: Minimum June streamflow across the period of record (cfs)	38.29	37.92	0.01	64.48	26.41	1.44

ML9	Minimum September streamflow: Minimum September streamflow across the period of record (cfs)	16.13	15.93	0.01	44.96	14.84	2.03
ML14	Mean of annual minimum flows: Mean of the lowest annual daily flow divided by median annual daily flow averaged across all years (dimensionless)	0.06	0.05	0.02	0.12	0.03	3.10
ML16	Median of annual minimum flows: Median of the lowest annual daily flows divided by median annual daily flows averaged across all years (dimensionless)	0.05	0.05	0.02	0.09	0.03	2.41
ML17	Baseflow index: Seven-day minimum flow divided by mean annual daily flows averaged across all years (dimensionless)	0.02	0.02	0.00	0.04	0.02	1.05
MH4	Mean maximum monthly flow for April: The mean of all April maximum flow values over the entire record (cfs)	6,820.52	6,927.19	-0.02	2,664.87	4,377.57	-0.39
MH8	Mean maximum monthly flow for August: The mean of all August maximum flow values over the entire record (cfs)	1,366.79	1,353.37	0.01	583.76	1,173.43	-0.50
MH14	Median of annual maximum flows: Median of the highest annual daily flow divided by the median annual daily flow averaged across all years (dimensionless)	81.40	77.21	0.05	17.92	32.90	-0.46

MH15	High flow discharge: The 1-percent exceedance value divided by the median flow for the entire record (dimensionless)	51.55	50.45	0.02	19.43	20.77	-0.06
MH16	High flow-discharge: The 10-percent exceedance divided by the median flow for the entire record (dimensionless)	9.11	8.95	0.02	12.32	6.46	0.91
MH23	High flow volume: The average volume for flow events above 7 times the median flow for the entire record (days)	63.50	60.32	0.05	39.39	19.81	0.99
Frequency							
FL1	Low flood pulse count: The number of annual occurrences during which the flow is below the 25th percentile (low pulse) of all daily values for the time period (number of events/year)	4.89	4.96	-0.01	7.84	13.09	-0.40
FL2	Variability in low-pulse count: Coefficient of variation for the number of annual occurrences of daily flows less than the 25th percentile (FL1) (dimensionless)	31.25	30.85	0.01	34.35	19.83	0.73
FL3	Frequency of low-pulse spells: The average number of flow events with flows below a threshold equal to 5% of the mean flow value for the entire flow record (average number of events/year)	4.70	4.82	-0.02	3.91	10.77	-0.64

FH1	High flood pulse count: The number of annual occurrences during which the flow is above the 75th percentile (high pulse) of all daily values for the time period (number of events/year)	11.89	12.30	-0.03	11.98	17.39	-0.31
FH4	High flood pulse count: The upper threshold is defined as 7 times median daily flow (number of events/year)	46.78	46.19	0.01	67.48	33.98	0.99
FH6	Flood frequency: Mean number of high flow events per year above a threshold equal to 3 times the median flow value for the entire flow record (number of events/year)	11.82	12.22	-0.03	12.59	16.84	-0.25
FH7	Flood frequency: Mean number of high flow events per year above a threshold equal to 7 times the median flow value for the entire flow record (number of events/year)	11.00	11.22	-0.02	10.43	10.73	-0.03
FH11	Flood frequency: The average number of events with flows above a threshold equal to flow corresponding to a 1.67-year recurrence interval (average number of events/year).	0.70	0.70	0.00	12.39	17.41	-0.29
Duration							
DL1	Mean of the annual minimum 1-day average flows (cfs)	8.50	8.52	0.00	18.14	8.43	1.15
DL4	Mean of the annual minimum 30-day average flows (cfs)	18.10	18.02	0.00	33.09	34.28	-0.03

DL6	Variability of annual minimum daily average flow: Coefficient of variation in DL1 (dimensionless)	69.10	68.69	0.01	33.83	31.15	0.09
DL16	Low flow pulse duration: Mean duration of FL1 (days)	15.83	16.43	-0.04	9.80	6.64	0.48
DL17	Variability in low pulse duration: Coefficient of variation in DL16 and calculated as the standard deviation for the yearly average low-flow pulse durations (daily flow less than the 25th percentile) (days)	61.66	61.71	0.00	53.88	30.37	0.77
DH1	Mean of the annual maximum 1-day moving average flow for the entire record (cfs)	15,281.85	15,592.22	-0.02	3,977.14	11,577.50	-0.66
DH4	Mean of the annual maximum 30-day moving average flow for the entire record (cfs)	2,281.95	2,322.21	-0.02	2,238.10	2,178.97	0.03
DH13	Mean of the annual maximum 30-day moving average flow divided by the median for the entire record (dimensionless)	15.63	15.28	0.02	11.78	7.57	0.56
DH15	High flow pulse duration: Mean duration of FH1 (days)	0.70	0.70	0.00	7.48	5.13	0.46
DH16	Variability in high flow pulse duration: Coefficient of variation in DH15 (dimensionless)	23.32	21.74	0.07	32.38	16.17	1.00
DH20	High-flow duration: The 75th percentile value for the entire flow record or the average duration of flow events with flows above a threshold equal to the 75th percentile value for the median annual flows (days)	7.68	7.44	0.03	7.70	5.27	0.46

DH24	Flood-free days: Computed as the flood threshold as the flow equivalent for a flood recurrence of 1.67 years (days)	292.30	292.30	0.00	98.98	78.98	0.25
Timing							
TA1	Constancy (dimensionless)	0.36	0.36	0.00	0.43	0.43	0.01
TA3	Seasonal predictability of flooding: Maximum proportion of all floods over the period of record that fall in any one of six 60-day 'seasonal' windows days (dimensionless)	0.14	0.14	0.00	0.01	0.01	0.30
TH1	Average Julian date of the annual maximum flow for the entire record (Julian day)	102	102	0.01	49	18	1.67
TH3	Seasonal predictability of non-flooding: The maximum proportion of a 365-day year that the flow is less than the 1.67-year flood threshold over the entire period of record (dimensionless)	0.26	0.26	0.00	0.02	0.01	0.14
Rate of Change							
RA6	Change of flow: Median of difference between natural logarithm of flows between two consecutive days with increasing flow (cfs)	0.27	0.24	0.12	0.23	0.29	-0.18
RA7	Change of flow: Median of difference between natural logarithm of flows between two consecutive days with decreasing flow (cfs)	0.15	0.15	-0.01	0.14	0.26	-0.48

RA8	Reversals: Number of negative and positive changes in water conditions from one day to the next (number of events/year)	75.30	83.67	-0.10	73.89	151.07	-0.51
RA9	Variability in reversals: Coefficient of variation in RA8 (dimensionless)	10.35	8.10	0.28	43.02	7.19	4.98

Suitable for perennial flashy or runoff stream types (Olden and Poff, 2003)
Suitable for all stream types (Olden and Poff, 2003)
'General' stream statistics
Ecologically specific statistic -- see 'Notes' (Olden and Poff, 2003)

Notes:
[1]May be a particularly important measure of daily flow conditions for certain riverine taxa, e.g., examining the response of fish assemblages to erratic water releases below hydroelectric dams (Kinsolving and Bain, 1993).
[2]Especially good candidates for riparian studies since model predictions suggest that annual flood duration may be critical in determining the ability of riparian forest patch types to persist within their natural range of abundance (Richter and Richter, 2000).

Streamflow statistics representing the varying streamflow characteristics were only calculated for Cossatot at DeQueen since this is the site impacted by the construction of Gillham Dam. For the statistical analyses, the streamflow data were separated into two time periods, as described above, into historical, i.e., prior to dam construction, and current, i.e., after dam construction. Streamflow statistics were calculated for each time period for both the observed and predicted streamflow values. The hydrologic alteration factor (observed minus predicted divided by predicted) was then determined in order to evaluate the change in flow regime (Richter et al., 1998). A factor close to 0 indicates little to no change for that particular streamflow characteristic; the further the value deviates from 0, the greater the change. A comparison of the observed to predicted for each steamflow statistic was completed for the historical period, i.e., prior to dam construction, in order to demonstrate the accuracy of the predicted values. Streamflow characteristics representing each of the five flow regimes are described in further detail in the following sections.

Magnitude

Magnitude is an important aspect of the flow regime as it has frequently been linked to ecological impairment (Poff and Zimmerman 2010, as referenced in Carlisle et al., 2010) and has clear implications for water management (Postel and Richter 2003, as referenced in Carlisle et al., 2010). Streamflow characteristics pertaining to average magnitude values demonstrated some degree of change; however, the most notable changes in magnitude for Cossatot near DeQueen between the observed and predicted streamflows after dam construction occur in both the low and high magnitude streamflow characteristics (Table 3). Some of the largest changes are for minimum June (ML6) and September (ML9) flows (Figure 31), minimum mean and median annual flows (ML14 and ML16, respectively; Figure 32), mean maximum August (MH8) flows, and median of annual maximum streamflows (MH14) (Table 3). The minimum June and September streamflows and the minimum mean and median annual streamflows are much greater than what should be expected for Cossatot near DeQueen since dam construction (Table 3; Figure 31 and Figure 32, respectively). Conversely, the mean maximum August and the median of annual maximum streamflows are less than what is expected (Table 3).

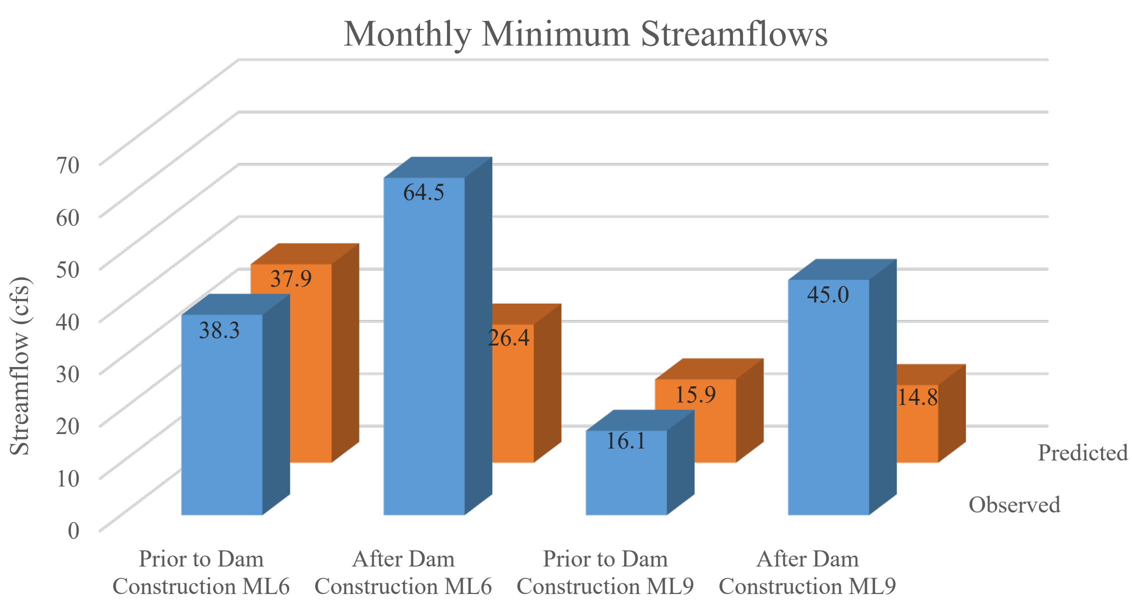

Figure 31. Minimum June streamflow (ML6) and minimum September streamflow (ML9) prior to and after dam construction for Cossatot River near DeQueen, Arkansas (see Table 3 for streamflow characteristic description).

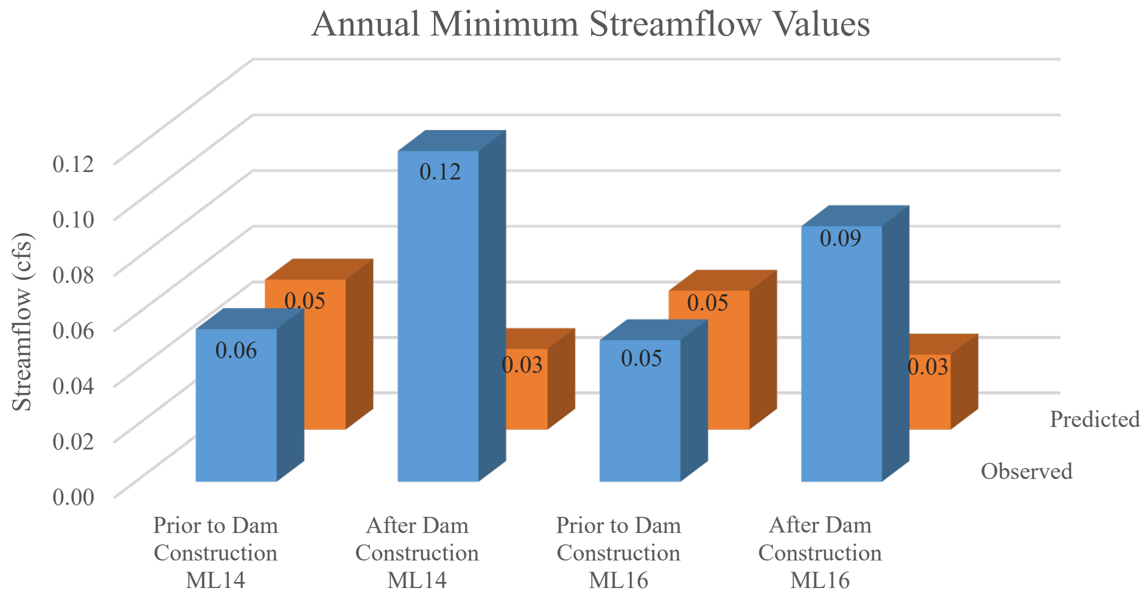

Figure 32. Mean of annual minimum streamflows (ML14) and median of annual minimum streamflows (ML16) prior to and after dam construction for Cossatot River near DeQueen, Arkansas (see Table 3 for streamflow characteristic description).

Frequency

There was less change in frequency of streamflows between observed and predicted for Cossatot near DeQueen than there was for magnitude. And, in fact, a majority of the observed streamflow characteristics that were analyzed more closely match the historical (prior to dam construction) time period than what should be expected (Table 3; Figure 33). This is more than likely due to controlled dam releases, whereas the predicted values coincide with current changes in climate.

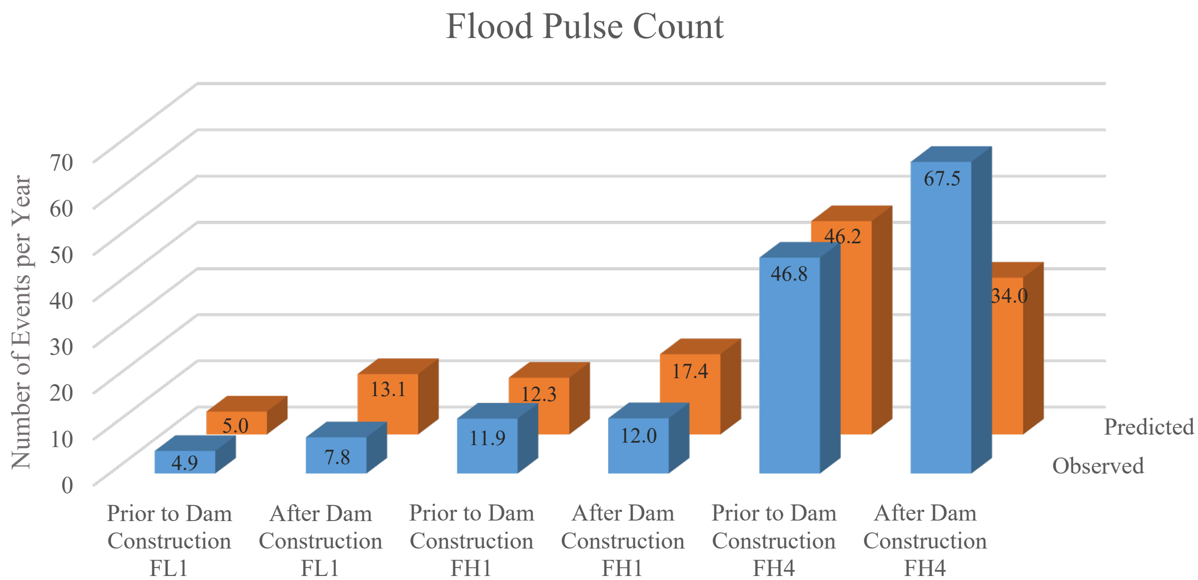

Figure 33. Low flood pulse count (FL1) and high flood pulse counts (FH1 and FH4) prior to and after dam construction for Cossatot River near DeQueen, Arkansas (see Table 3 for streamflow characteristic description).

Duration

For a majority of the selected duration streamflow statistics, the predicted value was less than the observed. Some of the more notable differences occur for the minimum 1-day average flow (DL1), low flow pulse duration (DL16) and its variability (DL17), and high flow pulse duration (DH15) and its variability (DH16) (Table 3). For the minimum 1-day average flow, the discharge is greater for the observed than what has occurred historically and what has been predicted (Table 3). The number of days for the low flow pulse duration (Figure 34) and its variability are less for predicted streamflows than what has occurred historically and what has been observed. This is likely indicative of the increase in overall precipitation. Subsequently, the number of days for the high flow pulse duration (Figure 34) and its variability are greater for the observed and predicted for the current (after dam construction) time period than what has occurred historically (Table 3).

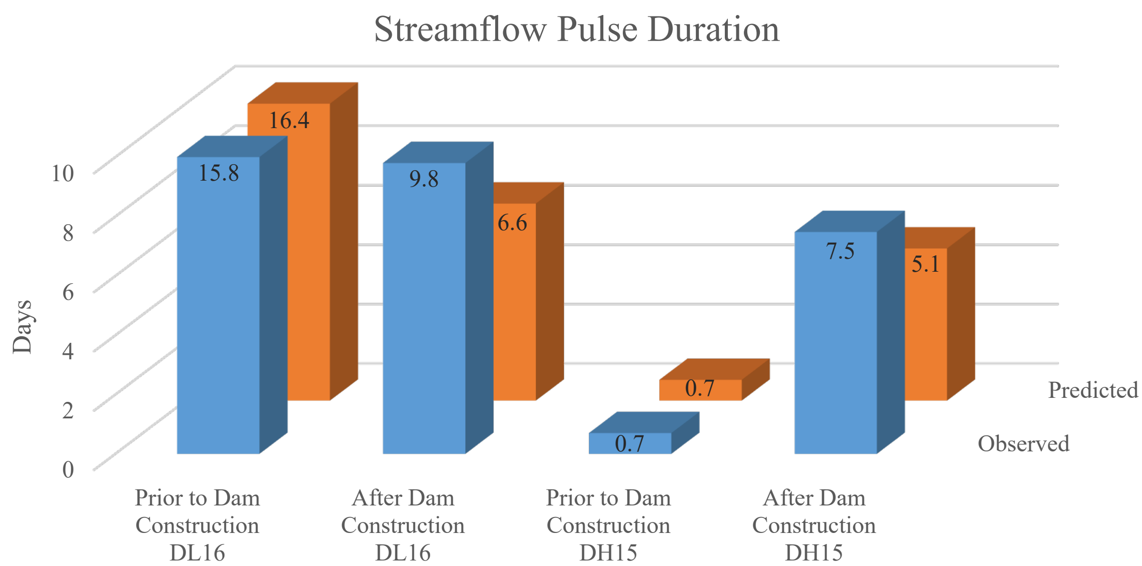

Figure 34. Low flow pulse duration (DL16) and high flow pulse duration (DH15) prior to and after dam construction for Cossatot River near DeQueen, Arkansas (see Table 3 for streamflow characteristic description).

Timing

Although there is little change between most observed and predicted timing characteristics for the current time period, there are notable differences between the current and historical time periods for both the observed and the predicted. However, there is one exception, the average Julian day for the annual maximum flow (TH1). This date has occurred earlier in the year for the current time period as compared to the historical (April 12, Julian Day 102) for the observed (February 18, Julian Day 49) and the predicted (January 18, Julian Day 18) (Figure 35). For all other selected timing characteristics, the observed and predicted matched well between each time period; however, the seasonal predictability for flooding (TA3) and non-flooding (TH3) differed greatly between the historical and current time periods, the proportion of each have decreased between the two time periods (Table 3).

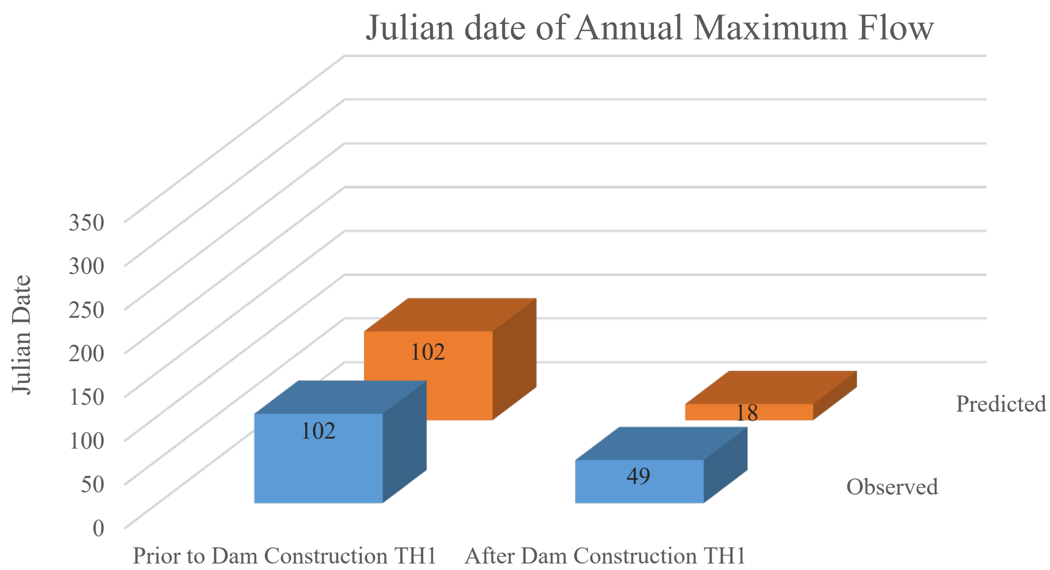

Figure 35. Julian date of annual maximum flow (TH1) prior to and after dam construction for Cossatot River near DeQueen, Arkansas (see Table 3 for streamflow characteristic description).

Rate of change

The largest changes in rate of change streamflow statistics for Cossatot near DeQueen occurred between the change of flow between two consecutive days with decreasing flow (R7), the number of negative and positive reversals in water conditions from one day to the next (R8) and its variability (R9) (Table 3). Although the observed current reversals closely matched historical reversals, they differed greatly from the predicted number of reversals that should have occurred (Figure 36). Similar results occurred between the observed current and predicted and historical values for the change of flow between two consecutive days with increasing (RA6) and decreasing flow, also known as the rise and fall rate of streamflow (Table 3; Figure 37).

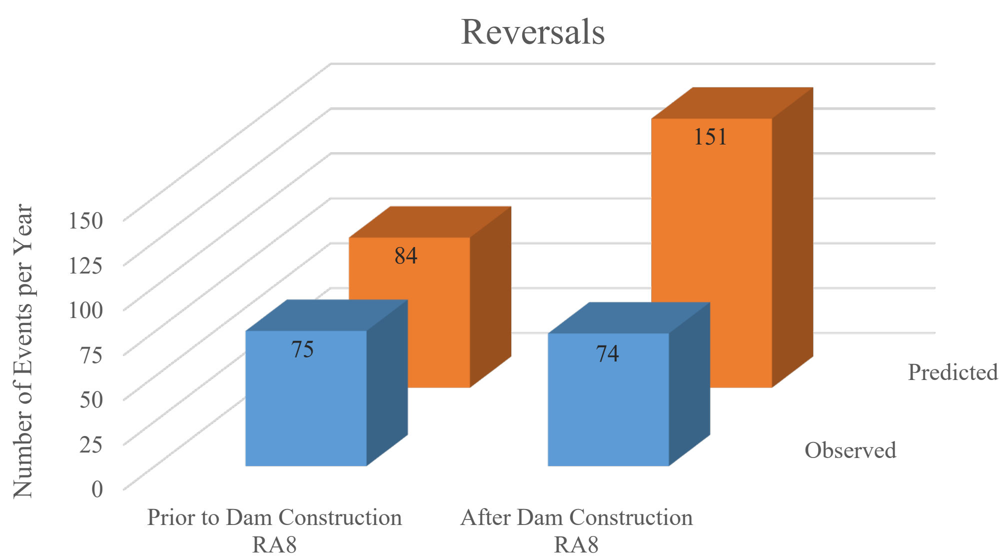

Figure 36. Number of reversals from one day to the next (RA8) prior to and after dam construction for Cossatot River near DeQueen, Arkansas (see Table 3 for streamflow characteristic description).

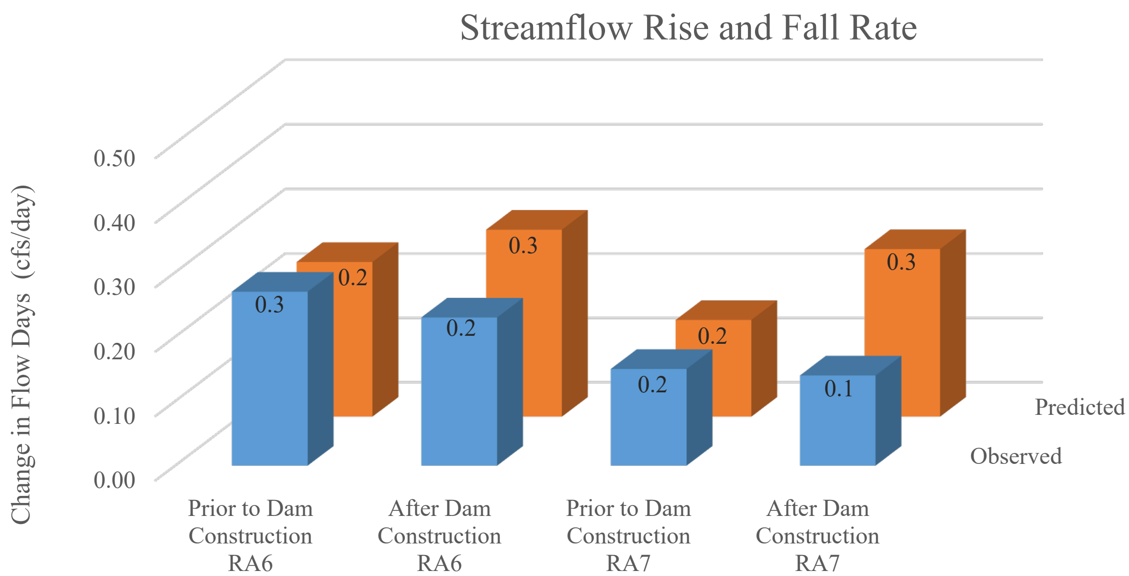

Figure 37. Change of flow between two consecutive days with increasing flow (RA6) and change of flow between two consecutive days with decreasing flow prior to and after dam construction for Cossatot River near DeQueen, Arkansas (see Table 3 for streamflow characteristic description).

Range of Variability

The Range of Variability Approach (RVA; Richter et al., 1997) has been used to guide the design of river management studies in order to attain protection of natural ecosystem functions as primary river management objectives. RVA uses the unregulated flows as a reference for defining the extent to which natural flow regimes have been altered and can also be used as a basis for defining initial environmental flow goals (The Nature Conservancy, 2009). Richter et al. (1997) suggested that water managers should strive to keep the distribution of natural streamflow variation as close to the prior to dam construction (pre-alteration) streamflow distributions as possible. As shown in Figure 38 and Figure 39, the range in variability between all values less than or equal to the 33rd percentile and all values greater than the 67th percentile, have changed prior to dam construction versus after dam construction. The median streamflow values between the two time periods have also shifted upwards.

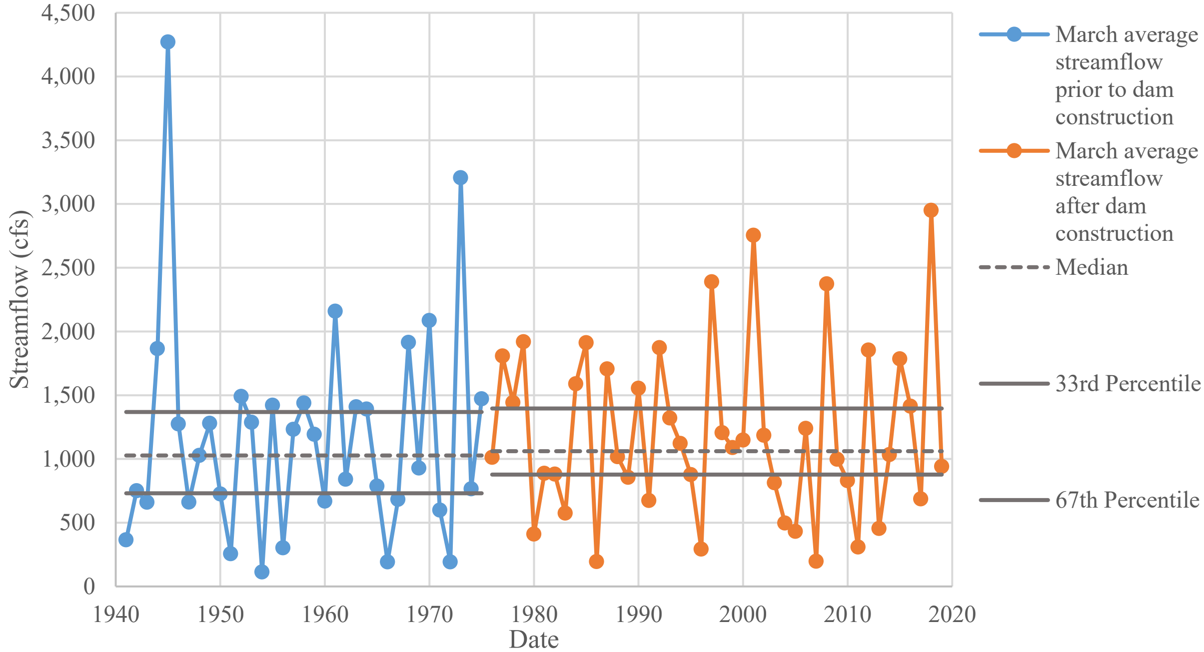

Figure 38. Range of Variability Approach analysis for March average streamflows prior to and after dam construction for Cossatot River near DeQueen, Arkansas.

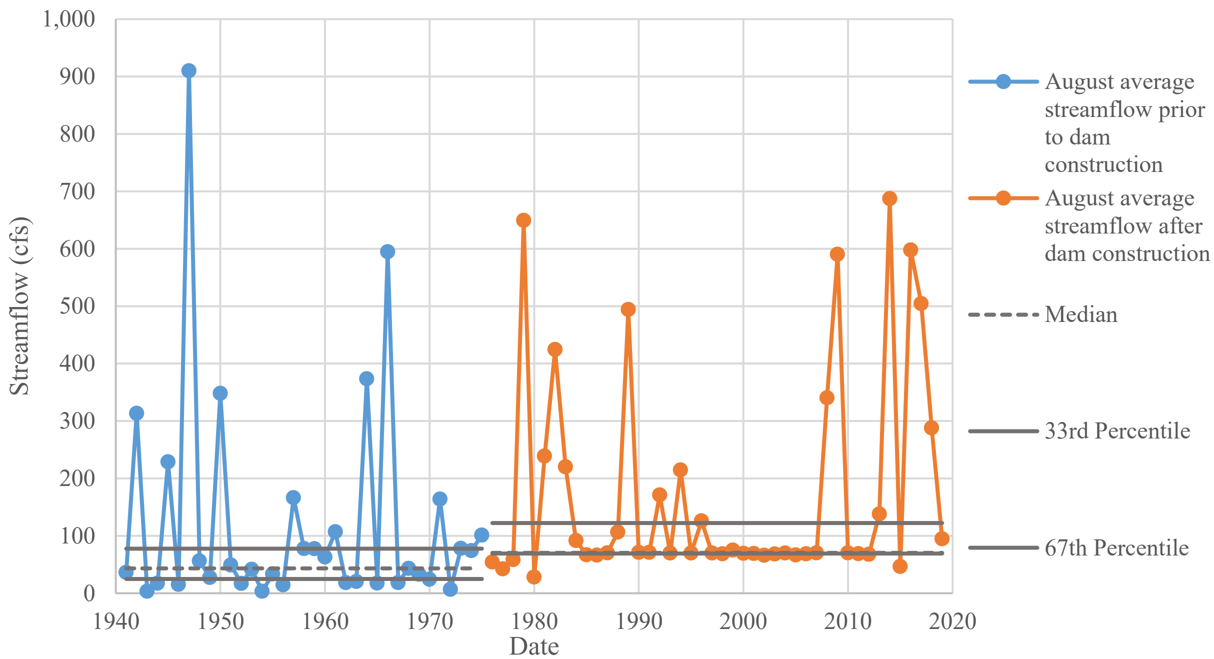

Figure 39. Range of Variability Approach analysis for August average streamflows prior to and after dam construction for Cossatot River near DeQueen, Arkansas.

Water quality

Regional water quality is influenced by lithology, soil composition, and land use activities. In most reaches of the Cossatot, overall, water quality is good (ADEQ, 2016). However, ANRC (2018) determined that the Lower Little River Watershed, the larger watershed containing the Cossatot, was identified for the first time as a “priority watershed” in their 2018 Nonpoint Source Pollution Management Plan. Sections of the Cossatot above Gillham Lake and directly below Gillham Dam are listed as impaired waters and are on the 303(d) List for Dissolved Oxygen (DO) (ADEQ, 2016). Though water temperature has not been listed on the 303(d) list for impaired waters, temperature is also a water quality concern for the Cossatot below Gillham Dam. Furthermore, FTN and Associates (2016) ranked all the 12-digit Hydrologic Unit Code (HUC) subwatersheds within the Little River watershed for sediment and pathogen inputs. For all the Cossatot HUC-12 subwatersheds, the overall sediment load ranked low to medium. For further discussion on how the subwatersheds were ranked, see FTN and Associates (2016). Finally, for the Cossatot River, the antidegradation policy of the Arkansas water quality standards states, “For outstanding state or national resource waters, those uses and water quality for which the outstanding waterbody was designated shall be protected” (FTN Assoc., 2016).

Research revealed little information regarding water quality data for Gillham Lake. The U.S. Fish and Wildlife Service (USFWS) published results of a water quality collection effort during a wet and dry period for Gillham Lake and its tailwater and the Cossatot (Smith and Moen, 1984). As expected, the results indicated higher inorganic and organic matter at the upper end of the Lake where the Cossatot comes in, compared to the lower end of the Lake by the dam. Additionally, the USGS made a concerted effort to collect water quality data beginning in 1981 and ending in 1995. The USGS collected data at four separate locations on Gillham Lake and collected numerous water quality constituents including physical parameters, metals, nutrients, and bacteria. All the data can be found in the USGS National Water Information System. Furthermore, ADEQ collected profile data from lower Gillham Lake from 2011 to 2018 on a quarterly basis, which is briefly summarized in the following sections.

Temperature

Though there was little recent data found regarding water quality for Gillham Lake, one thing to note is that the USACE Water Control Plan for Gillham Lake provides directive on regulating for temperature in order to reduce sudden temperature changes on the downstream fishery. Water temperature in Gillham Lake will be an important consideration when developing potential reservoir management modifications aimed to benefit the Cossatot River ecosystem and its biota below Gillham Dam. Based on temperature profile data compiled from ADEQ (https://www.adeq.state.ar.us/techsvs/env_multi_lab/water_quality_station.aspx), collected from 2011 to 2018, Gillham Lake begins to stratify as early as March (Figure 40, an example from two “typical” climatic years (2016 and 2017)) and generally experiences turnover in November (based on other data not shown). Water temperature at the bottom of the lake ranged between approximately 50 °F to 62 °F for 2016 and 2017 (Figure 40). Water temperature at an elevation of 488 feet (approximate elevation of the low flow pipe) was generally between 50 °F to 82 °F, again, for 2016 and 2017 (Figure 40).

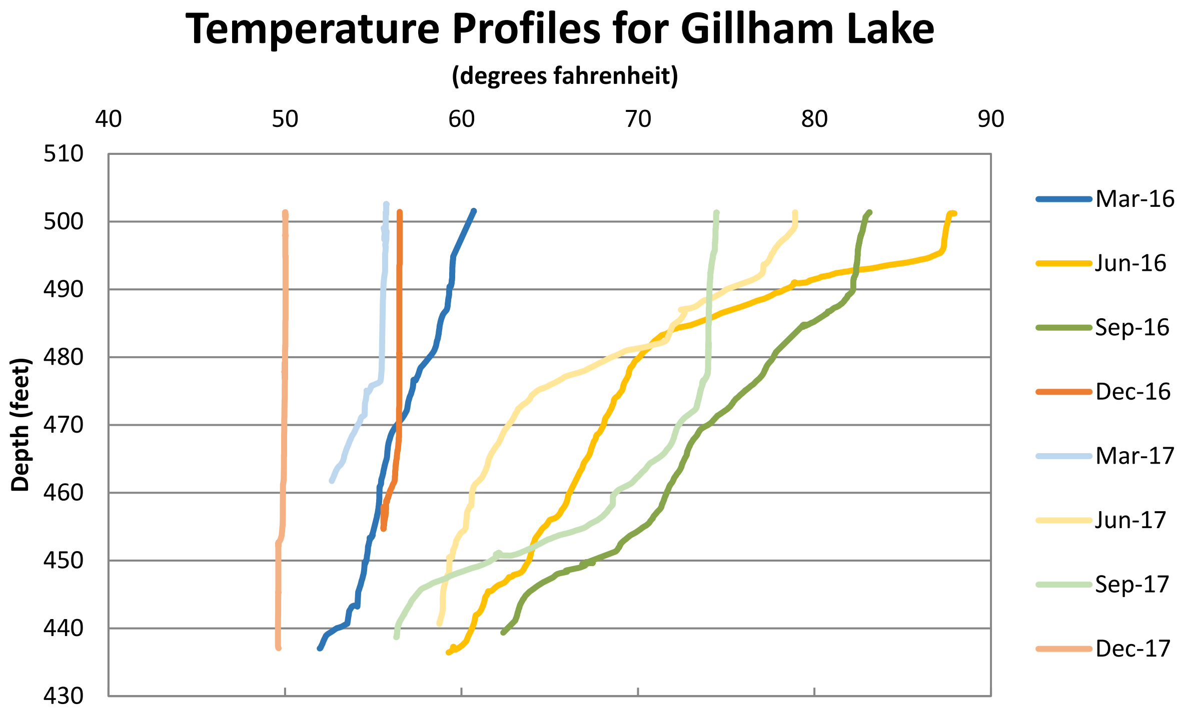

Figure 40. Temperature profile plots for Gillham Lake recorded quarterly during 2016 and 2017.

For the Lower Little River watershed, the numeric water quality criteria for temperature, as set by Arkansas Pollution Control and Ecology Commission, is 86 °F. This all-flow criteria for water temperature should not be exceeded in more than 10 percent of all samples collected over an entire year (Arkansas Pollution Control and Ecology Commission, 2014 as referenced in FTN Assoc., 2016). Based on the antidegradation policy of the Arkansas water quality standards, "For potential water quality impairments associated with a thermal discharge, the antidegradation policy and implementing method shall be consistent with Section 316 of the Clean Water Act", Gillham Lake would fall under this provision (FTN Assoc., 2016). Furthermore, temperature has been listed as a water quality impairment for the Little River (FTN Assoc., 2016).

Water temperature data upstream and downstream of Gillham Lake has been collected by the AGFC beginning in October 2015 extending through June 2016 (Figure 41). The Ladd Rd and 80,000 Rd sites are located downstream of Gillham Lake Dam and the State Park site is located above Gillham Lake (Figure 2). The data indicate that temperatures do not differ much from above the dam to below the dam during the fall and spring months and differ by approximately 5 degrees Celsius (°C) during the start of the summer months and approximately 6 °C during the winter months. Furthermore, water temperatures are consistently higher in winter and consistently lower in the late spring to early summer months (Figure 41). Water temperature variability is greater between the upstream (State Park site) and the downstream sites (Ladd Rd and 80,000 Rd sites) with the downstream sites having less range in water temperature, particularly during the winter months (Figure 41).

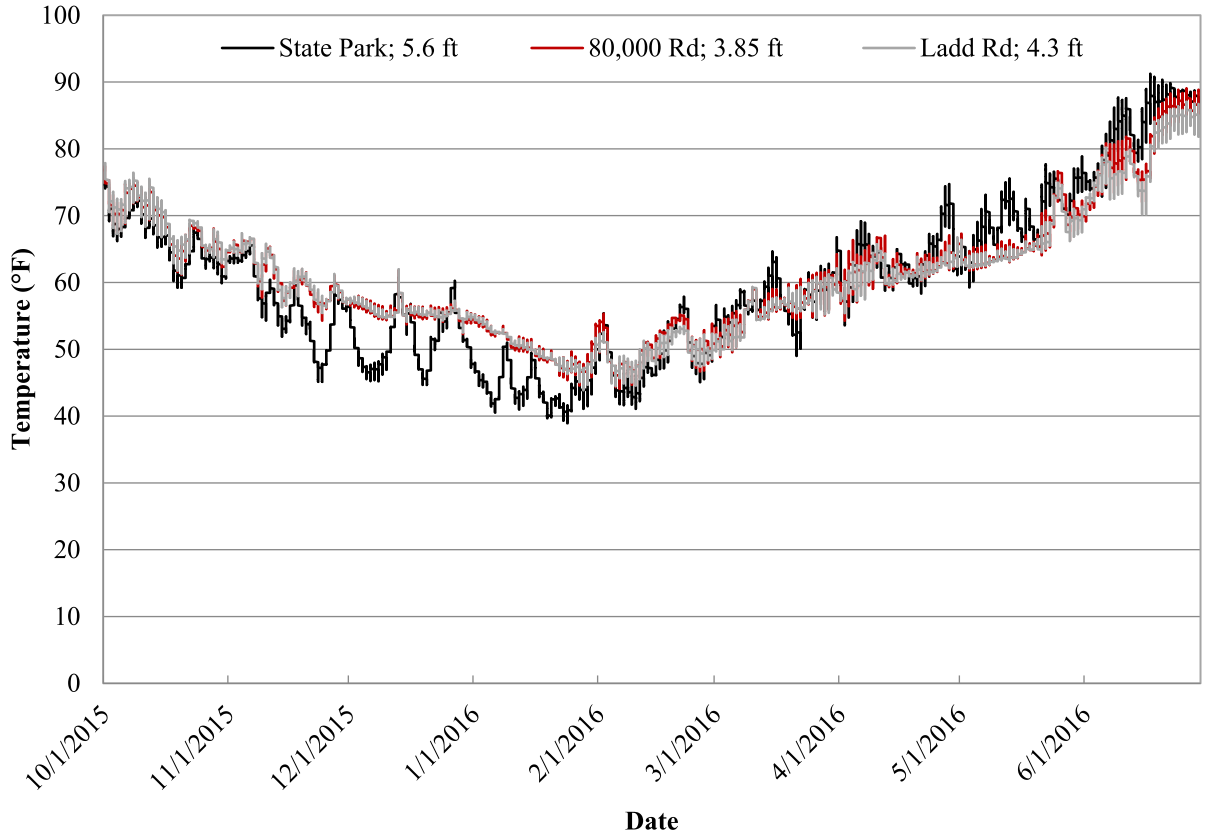

Figure 41. Water temperature comparison at Cossatot River water quality sampling sites. The State Park site is in the unregulated reach above the dam; Ladd Rd is approximately 5.3 river miles below the dam; 80,000 Rd is approximately 9.3 river miles below the dam (see Figure 2 for site locations; sensor depths are given in feet after the site name).

Dissolved Oxygen

DO profile data was also compiled from ADEQ (https://www.adeq.state.ar.us/techsvs/env_multi_lab/water_quality_station.aspx), for Gillham Lake collected from 2011 to 2018 (Figure 42, an example from two "typical" climatic years (2016 and 2017)). DO at the bottom of the lake ranged from near zero milligrams per liter (mg/L) to approximately 2 mg/L from June to November for 2016 and 2017 (Figure 42). At a depth of approximately 30 feet, DO was mostly less than 3.0 mg/L from July to November for 2016 and 2017. At a depth of 15 feet (elevation 488, approximate elevation of the low flow pipe), DO ranged between less than 5.0 mg/L to approximately 7.0 mg/L from July to November for 2016 and 2017 (Figure 42).

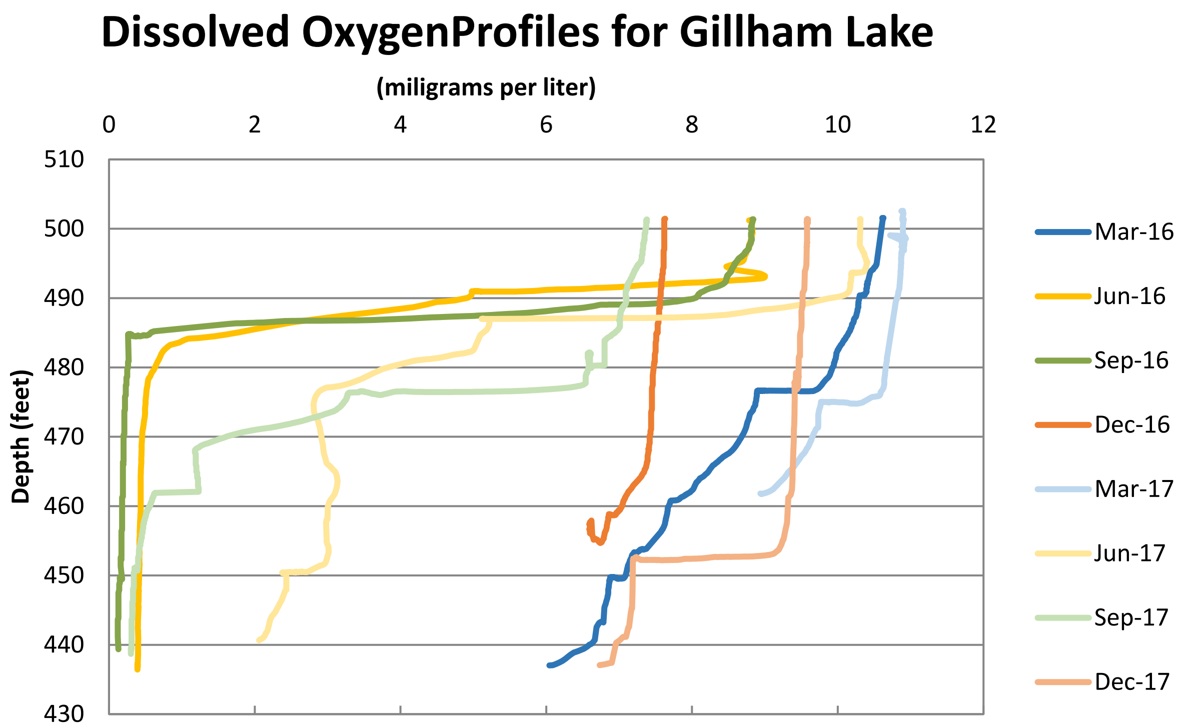

Figure 42. Dissolved Oxygen profile plots for Gillham Lake recorded quarterly during 2016 and 2017.

The Cossatot has been listed on the 303(d) list for impaired waters for DO because the designated use for aquatic life is not being supported by the waterbody or stream segment (ADEQ, 2016). Additionally, ADEQ (2016) ranks the Cossatot as a medium priority rank—"a ranking of waters in order of need for corrective action taking into account the severity of the pollution and designated uses of the waters". A medium priority is considered moderate risk to public health, welfare or to aquatic life (ADEQ, 2016).

DO data for the Cossatot was downloaded from EPA's Water Quality Portal (WQP) (https://www.waterqualitydata.us/#mimeType=csv&providers=NWIS&providers=STEWARDS&providers=STORET). The closest downstream site to the dam, with a sufficient amount of data, was the Cossatot River west of Lockesburg site (approximately 21 miles downstream; Figure 2). DO ranged at this site from 4.3 mg/L to 14.1 mg/L with an average of 8.7 mg/L (Figure 43). This site is located in an area dominated primarily by agriculture (Figure 14) and is too far downstream of the dam to analyze any effects from dam releases, but, instead, represents conditions that currently exist within the watershed.

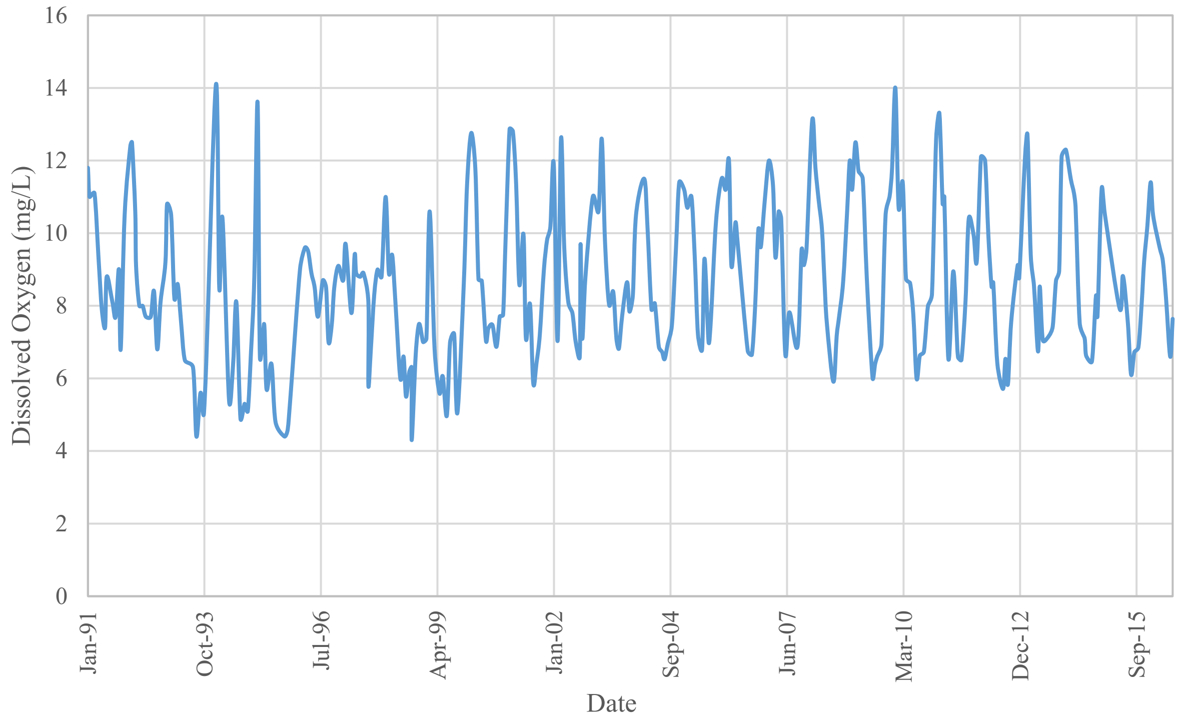

Figure 43. Dissolved Oxygen concentration for Cossatot River west of Lockesburg site (see Figure 2 for site location).

Biological and ecological conditions

The Cossatot has been identified as an Ecologically Sensitive Waterbody by ADEQ, which identifies stream segments known to provide habitat for and be within the geographic range of threatened, endangered, or endemic species of aquatic or semi-aquatic life forms (ADEQ, 2016). Routine biological monitoring and inventory does not occur at Gillham Lake with any frequency. Nevertheless, operational civil works projects administered by USACE are required, with few exceptions, to prepare an inventory of natural resources. The basic inventory required is referred to within USACE regulations (ER and EP 1130-2-540) as a Level One Inventory. This inventory includes the following: vegetation in accordance with the National Vegetation Classification System through the sub-class level; assessment of the potential presence of special status species including, but not limited to, federal and state listed endangered and threatened species, migratory species, and birds of conservation concern listed by the USFWS; land (soils) capability classes in accordance with the Natural Resource Conservation Service (NRCS) criteria; and wetlands in accordance with the USFWS Classification of Wetlands and Deepwater Habitats of the United States.

Regional habitat types

The northern portion of the watershed is contained within the Ouachita Mountains Level III Ecoregion and the southern portion in the South-Central Plains Ecoregion (Figure 3). The Level III Ecoregions are further subdivided into Level IV Ecoregions with the majority of the northern portion of the watershed located in the Athens Plateau Level IV Ecoregion and the southern portion located in the Cretaceous Dissected Uplands Ecoregion, while the headwaters of the Cossatot are located in the Central Mountain Ranges Ecoregion (Figure 3).

Terrestrial and riparian communities and habitat types

The wildlife population is quite diverse in and around the Gillham Lake area, which includes armadillo, swamp and cotton-tailed rabbits white-tailed deer, black bear, coyote, bobcat, gray fox, gray squirrel, turtles, mink, eastern fox squirrel, wild turkey, American Kestrel, wood thrush, whippoorwill, yellow-billed cuckoo, snakes, and salamanders.

Avian species that can be affected by changes in streamflows include chimney swift that nest in large old hollow trees found in riparian forests and backwater sloughs; yellow-billed cuckoo that nest in riparian and other mesic forest habitat; Willow flycatcher that nest in riparian forest (particularly river front willows); rusty blackbird that winters in wet riparian forest and backwater; wood thrushes that nest in riparian and other mesic forests; the Swainson's warbler that nest in dense riparian habitats; and the American woodcock that nest and winters in riparian forests (Doug Zollner, The Nature Conservancy, 2020, pers. comm.). Colonial wading/diving birds including the anhinga, black and yellow crowned night heron, little blue and tricolored heron, American and least bittern could also be affected by changes in streamflow.

The area surrounding the lake is forested and can be described as an upland mixed pine-hardwood closed canopy forest. Four vegetation subclasses are present at Gillham Lake: deciduous closed tree canopy, evergreen forests, mixed evergreen-deciduous closed tree canopy and grassland-herbaceous vegetation. Trees, understory vegetation and shrub on site will include upland oak and hickory species, persimmon, sweet gum, black gum, loblolly and short-leaf pines, winged and American elm, dogwoods, buttonbush, maples, sassafras and hornbeams. Ground covers consist of greenbrier, wild blueberries, farkleberries, and other herbaceous vegetation.

Five major terrestrial communities make up the Ouachita Mountains. These large patches make up most of the uplands that are still in natural cover (a large part of the headwaters area in the National Forest and within the State Park-Natural Area). These terrestrial communities are distributed according to slope and aspect. Cultivated forests are the dominant type in the lower watershed outside of conservation ownership. Embedded in these large patches are smaller areas that occur where specific geology, topography, hydrology, slope, and aspect occur and account for a large portion of the species of conservation concern, especially plants. Pastures and ponds in the Ouachita Mountains are typically manmade and placed accordingly. Areas of "natural" aquatic habitats exist primarily along the lower part of the Cossatot and along the smaller creeks and the upper Cossatot (ANHC, 2020, pers. comm.) (Table 4).

Table 4. Terrestrial and aquatic communities in the Cossatot watershed.

Terrestrial (large patch) Communities	**Terrestrial communities (small patch) that are embedded in the large patch communities**	**Aquatic Communities**
Ozark-Ouachita Dry Oak and Pine Woodland	Interior Highlands Calcareous Glade and Barrens	Ponds, Lakes, and Water Holes
Ozark-Ouachita Dry-Mesic Oak Forest	Ozark-Ouachita Cliff and Talus	Ozark-Ouachita Large Floodplain
Ozark-Ouachita Pine-Oak Forest/Woodland	Ozark-Ouachita Forested Seep	Ozark-Ouachita Riparian
Ozark-Ouachita Mesic Hardwood Forest		
Cultivated Forest (Pine Plantation)		

Threatened & endangered species

There are many species in the Ouachitas that are on the federal and state list of threatened and endangered species. Species become imperiled for a variety of reasons including over-hunting, overfishing, and habitat loss as a result of human development and pollution. Of these, habitat loss is the main contributor that imperils most of the species listed in Table 5. A threatened species is one that is likely to become endangered within the foreseeable future. An endangered species is one in danger of extinction throughout all or a significant portion of its range.

At Gillham Lake there are eight species that are listed as either threatened or endangered and have the potential to occur at the lake project according to the USFWS. The Bald Eagle (*Haliaeetus leucocephalus*) is common during the winter months and a few stay year-round. Although the Bald Eagle was delisted by the USFWS in 2007 due to recovery of the species, both the Bald and Golden Eagles are still protected in accordance with the Bald and Golden Eagle Protection Act.

The range of the Northern Long-Eared Bat (*Myotis septentrionalis*) extends to the Ouachita Mountains and Gillham Lake. USACE works closely with the USFWS, AGFC, and Arkansas State Parks (ASP) to protect and manage project lands and waters of Gillham Lake in order to protect this bat's habitat. Transient populations of Gray, Indiana, and other bat species have not yet been documented; results of the 2020 bat surveys are still pending.

Gillham Lake may also be home to the Leopard Darter (*Percina pantherina*) an threatened darter that is found in intermediate to larger streams. Habitat disruptions including sediment and erosion may affect the substrate or water quality for Leopard Darter, affecting feeding or reproduction. Five federally listed mussels have the potential to be found at Gillham Lake; Ouachita Rock Pocketbook (*Arcidens wheeleri*), Pink Mucket (*Lampsilis abrupta*), Rabbitsfoot (*Theliderma cylindrica)*, Scaleshell (*Leptodea leptodon)* and the Winged Mapleleaf (*Quadrula fragosa)* and, if present, all are dependent on adequate water flows and all are susceptible to sedimentation and poor water quality.

One federally listed plant species, Harperella (*Ptilimnium nodosum*), is known from rocky stream channels in high gradient streams in the Ouachita Mountains. Habitat in other streams where it occurs, including the Fourche LaFave River, South Fourche LaFave River, Mountain Fork (of the Little River), and Irons Fork (of the Ouachita River) is very similar to that on the Cossatot River upstream from Gillham Lake. Furthermore, many of the associated species of Harperella are common on the Cossatot.

The species listed in Table 5 are from the USFWS federally classified status list of species and the ANHC datasets, which have been reported and identified on project lands. Additionally, Table 6 shows a list of species that are State species of concern, as well as Federal Special Status Species of Interest (USFWS, 2020).

Table 5. Federally threatened and endangered species with the potential of occurring at Gillham Lake (E = endangered and T = threatened).

Scientific Name	Common Name	Federal Status
Ptilimnium nodosum	Harperella	E
Percina pantherina	Leopard Darter	T
Myotis septentrionalis	Northern Long-Eared Bat	T
Arcidens wheeleri	Ouachita Rock Pocketbook	E
Lampsilis abrupta	Pink Mucket (pearlymussel)	E
Theliderma cylindrica	Rabbitsfoot	T
Leptodea leptodon	Scaleshell Mussel	E
Quadrula fragosa	Winged Mapleleaf	E

Table 6. State and Federal Special Status Species and Species of Interest in the study area (LT = listed threatened species; S1 = critically imperiled in Arkansas; S2 = imperiled in Arkansas; S3 = vulnerable in Arkansas (ANHC, 2020).

Scientific Name	Common Name	Federal Status	State Status
Animals-Invertebrates			
Gomphurus ozarkensis	Ozark Clubtail		S1
Lampsilis spA cf hydiana	Arkoma Fatmucket		S3
Speyeria diana	Diana Fritillary		S2S3
Villosa sp. cf lienosa	Little Spectaclecase		S2S3
Animals-Vertebrates			
Hemidactylium scutatum	Four-toed Salamander		S2
Lythrurus snelsoni	Ouachita Shiner		S2
Myotis septentrionalis	Northern Long-Eared Bat	LT	S1S2
Notropis atrocaudalis	Blackspot Shiner		S3
Percina pantherina	Leopard Darter	LT	S1
Plethodon caddoensis	Caddo Mountain Salamander		S2
Plants-Vascular			
Acer saccharum var. leucoderme	Chalk Maple		S2S3
Amorpha ouachitensis	Ouachita Indigo-Bush		S3
Cardamine angustata	Slender Toothwort		S2
Carex gracilescens	Slender Wood Sedge		S2
Carex latebracteata	Waterfall's Sedge		S3
Carex timida	Timid Sedge		S2S2
Erythronium mesochoreum	Prairie Tout-Lily		S1S2
Euphorbia ouachitana	Ouachita Spurge		S3

Gratiola brevifolia	Sticky Hedge-Hyssop		S3
Houstonia ouachitana	Ouachita Bluet		S3
Hydrophyllum brownei	Browne's Waterleaf		S2
Ilex longipes	Georgia Holly		S3
Liatris compacta	Ouachita Blazing-star		S3
Stachys iltisii	Ouachita Hedge-nettle		S3
Streptanthus squamiformis	Ouachita Twistflower		S2
Valerianella palmeri	Palmer's Cornsalad		S3
Vernonia lettermannii	Letterman's Ironweed		S3
Special Elements-Natural Communities			
Ouachita Shale Glade and Barrens	Ouachita Shale Glade and Barrens		S2

Invasive species

Invasive species and non-indigenous organisms are not new to Gillham Lake. Table 7 notes the non-indigenous species that have been identified at Gillham Lake by the ANHC and the USACE. Invasive species, both terrestrial and aquatic, have the potential to displace native species because many grow and reproduce rapidly, creating competition for resources. Some can also easily adapt to a variety of habitat conditions, reducing habitat available to native species. Invasive species reduce forest health and productivity and alter ecosystems and their processes because they lack natural enemies and pests like those that keep native species in check. The costs associated with managing invasive species and the harm they can levy on property values, agricultural productivity, public utility operations, native fisheries, tourism, and outdoor recreation can be minimized if efforts to detect and prevent the spread of the species begin early. Furthermore, changes in water regimes can worsen or curb the problem depending on the species and the water management regime.

Table 7. Nonindigenous and nonindigenous invasive species found at Gillham Lake (ANHC, 2019; USACE OMBIL, 2018). The extent to which each species is a problem has not been determined at Gillham Lake except for a few.

Common Name	**Scientific Name**	**Source**
Annual Hair Grass	*Aira caryophyllea var. capillaris*	ANHC
Silver Hair Grass	*Aira caryophyllea var. caryophyllea*	ANHC
Mimosa/Silk Tree	*Albizia julibrissin*	USACE
Field Garlic	*Allium vineale*	ANHC
Spiny Amaranth	*Amaranthus spinosus*	ANHC
Annual Vernal Grass	*Anthoxanthum aristatum*	ANHC
Early Yellow-Rocket	*Barbarea verna*	ANHC
Meadow Brome	*Bromus commutatus*	ANHC
Hairy Bittercress	*Cardamine hirsuta*	ANHC
Gray Mouse-Ear Chickweed	*Cerastium brachypetalum*	ANHC
Sticky Mouse-Ear Chickweed	*Cerastium glomeratum*	ANHC
Watermelon	*Citrullus lanatus var. lanatus*	ANHC
Wild Basil	*Clinopodium gracile*	ANHC
Asiatic Dayflower	*Commelina communis*	ANHC

Spreading Dayflower	*Commelina diffusa var. diffusa*	ANHC
Piedmont Bedstraw	*Cruciata pedemontana*	ANHC
Deptford Pink	*Dianthus armeria subsp. armeria*	ANHC
Southern Crab Grass	*Digitaria ciliaris*	ANHC
Jungle-Rice	*Echinochloa colona*	ANHC
Goose Grass	*Eleusine indica*	ANHC
Cudweed	*Gamochaeta coarctata*	ANHC
Indian Heliotrope	*Heliotropium indicum*	ANHC
Japanese Bush-Clover	*Kummerowia striata*	ANHC
Purple Deadnettle	*Lamium purpureum*	ANHC
Sericea lespedeza	*Lespedeza cuneata*	ANHC
Common Privet	*Ligustrum sinense*	USACE
Rye Grass	*Lolium perenne*	ANHC
Japanese Honeysuckle	*Lonicera japonica*	USACE
Carpetweed	*Mollugo verticillata*	ANHC
Bahia Grass	*Paspalum notatum*	ANHC
Bristly Lady's-Thumb	*Persicaria longiseta*	ANHC
English Plantain	*Plantago lanceolata*	ANHC
Annual Blue Grass	*Poa annua*	ANHC
Purslane	*Portulaca oleracea*	ANHC
Peach	*Prunus persica*	ANHC
Callery Pear	*Pyrus calleryana*	USACE
Curly Dock	*Rumex crispus*	ANHC
Yellow Bristle Grass	*Setaria pumila subsp. pumila*	ANHC
Field-Madder	*Sherardia arvensis*	ANHC
Prickly Sida	*Sida spinosa*	ANHC
Red Imported Fire-ants	*Solenopsis invicta*	USACE
Lawn Burweed	*Soliva sessilis*	ANHC
Johnson Grass	*Sorghum halepense*	USACE
Common Chickweed	*Stellaria media*	ANHC
Feral Hog	*Sus scrofa*	USACE
Common Dandelion	*Taraxacum officinale*	ANHC
Field Hedge-Parsley	*Torilis arvensis*	ANHC
Hop Clover	*Trifolium campestre*	ANHC
Red Clover	*Trifolium pratense*	ANHC
White Clover	*Trifolium repens*	ANHC
Moth Mullein	*Verbascum blattaria*	ANHC
Woolly Mullein	*Verbascum thapsus*	ANHC
Corn Speedwell	*Veronica arvensis*	ANHC
Common Vetch	*Vicia sativa*	ANHC

Aquatic and wetland communities

Wetlands around Gillham Lake are mostly riverine systems with shallow to deep water habitats containing a channel, and dominated by trees, shrubs, and persistent emergent plants. In general, the wetland types include freshwater ponds, riverine systems, and freshwater forested shrub wetlands (Figure 45).

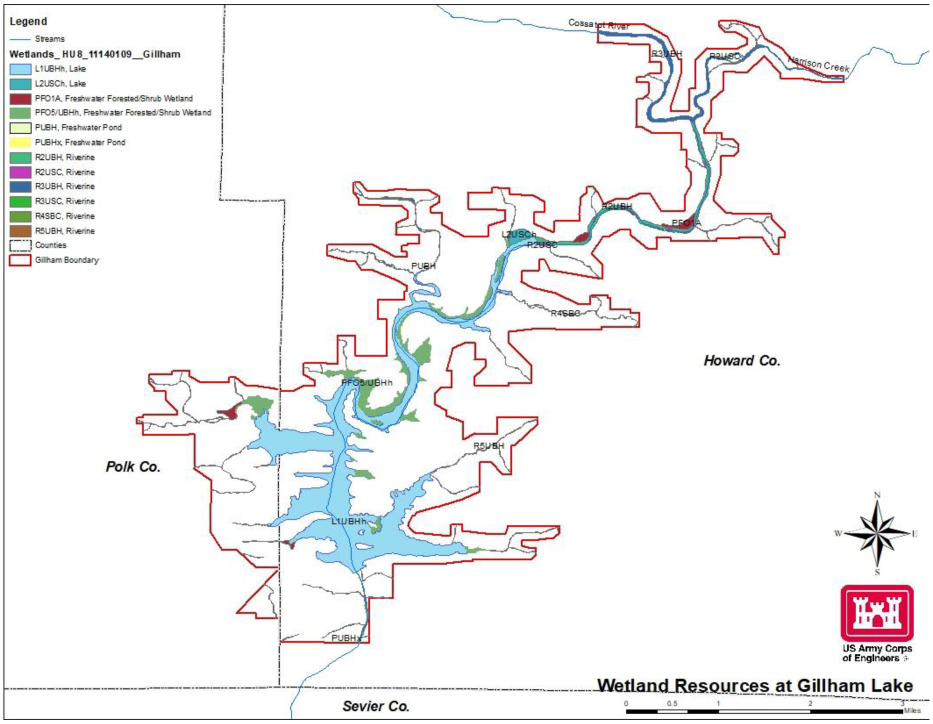

Figure 45. Wetlands around Gillham Lake.

In terms of the larger watershed, wetlands upstream from Gillham Lake are mostly linear features associated with stream channels (and occasionally with abandoned channel scar features on terraces of larger streams such as the Cossatot River), groundwater-fed seepage wetlands, or artificial (impounded) freshwater ponds. Downstream from Gillham Lake are wider, lower gradient stream channels with more backwater and abandoned channel features as well as artificial (impounded) freshwater ponds. There is also a large (26,879 acre) wetland complex of bottomland hardwoods comprised of forested overflow bottoms and riparian forests in the Pond Creek National Wildlife Refuge (NWR), which is located in the floodplain at the confluence of the Little River and the Cossatot. The refuge is approximately 95 percent forested, with small areas of open water, shrub swamps, beaver ponds, open marsh, and roads (USFWS 2014a as referenced in FTN Assoc., 2016).

Fish Collection

Pre-impoundment fish communities were documented for the Cossatot River by several investigators with varying degrees of effort, and almost all used seining as the fish collection method. It is believed the first collection of fish from the Cossatot basin was made on 30 June 1927 by Hubbs and Ortenburger (1929) at a site located seven miles northeast of DeQueen and included eight species that are cataloged at the Sam Noble Museum at the University of Oklahoma. Black (1940) collected 22 species from the basin in 1938 and 1939 and many of those specimens are at the University of Michigan's museum. Also noteworthy was that Black (1940) collected *Hybognathus nuchalis*, a species that has never been detected in any other surveys of the Cossatot River. Black (1940) also collected *Notropis ortenburgeri* (UMMZ, catalogue

128067). Robison and Buchanan's (1988) collection maps only show two pre-1960 collections from the Cossatot.

The USACE's Environmental Impact Statement (EIS) for Gillham Dam provided an overview of some of the historical fish collections (USACE, 1971). The USACE collected 887 fish representing 29 species in 1971; Dr. Rudy Miller identified the specimens. The EIS indicated that the USACE collected Leopard Darter (*Percina pantherina*) from the Gillham Dam site and Ladd Rd site (Highway 380; Figure 2). Pre-impoundment surveys of the Cossatot below Gillham Dam were also made (Hubbs and Ortenburger, 1929; Robison 2005; Robison and Buchanan, 1988; Robison and Buchanan, 2020).

Cloutman and Olmsted (1974) collected fish presence data at 19 sites during 5 collection trips from spring 1971 to summer 1972, but they released most specimens and the counts of the numbers of individuals collected is unknown. They collected 53 species and documented that 62 species were known from the basin. They noted the gradient of the river changes from 13 feet per mile (ft/mile) in the Ouachita Mountains to 4 ft/mile in the Gulf Coastal Plain and the substrate changed from bedrock and boulders to gravel. Close to the mouth with Little River, the river becomes turbid and is swampy with cypress trees. Several migratory large river species, Cloutman and Olmsted (1974) reported, were likely extirpated due to the construction of the downstream Millwood Dam (e.g., American Eel, Paddlefish). They noted that only 5 species were found in the most upstream headwaters and 32 species were found at the most downstream site.

The most comprehensive pre-impoundment evaluation of Cossatot fishes was the unpublished thesis research of Ethridge (1974). Ethridge (1974) examined 20,128 specimens from 57 collections at 24 sites between July 1972 and September 1974. She reported the presence of 75 species from the Cossatot basin. Her thesis included collections in the lowest reaches of the Cossatot near the mouth, and the Ladd Rd site (Figure 2) was sampled on 11 dates during the study. Her collections were largely deposited in the fish collection at Northeast Louisiana University at Monroe (NLU). The NLU collection lost funding support and the specimens from this collection are currently at Tulane University awaiting shipment to the Arkansas State University Museum in Jonesboro. The NLU database indicates that approximately 8,900 of the specimens were held in the NLU museum.

Walberg et al. (1983) documented fish, macroinvertebrates, and water quality in multiple tailwaters below dams, including the Cossatot. This report compared biota among cold and multi-level warm water release dam tailwaters, including Gillham tailwaters. Walberg et al. (1983) sampled fish at one site at the 80,000 Rd below Gillham Dam (Figure 2). Additional electrofishing and rotenone data were collected by Fritsche (1982) from 80,000 Rd (Figure 2). Frietsche (1982) published the Cossatot electrofishing data in his evaluation of the cool Lake Greeson tailwater.

AGFC biologists have sampled the Cossatot for sport fish on several occasions with most surveys performed in the upper watershed above Gillham Lake. Fishes were collected with boat electrofishing from 1-4 August 1989, at four sites above Gillham Dam. This survey reported 9 sport fish species, including 22 Smallmouth Bass (2.96 fish per hour (fish/hr)). During 15-16 August 1990, samples were collected below Gillham Dam at Mize Crossing and above the Lake at U.S. Forest Service (USFS) Road 31 (the first bridge upstream of Hwy 246). The USFS Road 31 site survey included 5 sportfish species. The sample at Mize Crossing contained an unusually large number of Bluegill (*Lepomis macrochirus*) among the 19 species collected. The Cossatot was sampled at two locations above Gillham Lake during 1993, including Highway 278 (old Highway 4 bridge) and the USFS Road 31 pool and step run. AGFC documented 28 fish species including Leopard Darter at Highway 278. The sample at Highway 278 also included collection of habitat data and mark-recapture population estimates; however, estimates generally were imprecise and biased with less than 6 recaptures for every species. Furthermore, AGFC sampled at the USFS Road 31 on 21 August 2009 for sport fishes and captured 18 Smallmouth Bass and a nongame fish collection which included 13 *Lythrurus snelsoni*.

Zale (1994) provides the most comprehensive published study of the distribution of the threatened Leopard Darter. Zale (1994) surveyed for Leopard Darters using snorkeling at 15 sites along the Cossatot during July and August 1986, including 7 sites below Gillham Dam. Leopard Darter was detected at three sites which appear to correspond to locations from Cow Creek through Highway 278 and was not detected below Gillham Dam or above the Cossatot Falls. Additionally, this study documented that Leopard Darter experienced a 90% reduction in range due to impoundment of Gillham Lake.
The USFWS Tulsa office and the USFS have cooperatively implemented a long-term snorkeling monitoring program for the threatened Leopard Darter from 1998 to present (Fenner 2012). Data include multiple snorkeling counts (e.g., 5) at each site and sampling at 17 fixed permanent sites including two sites at Robison Fork, one at the Cossatot, and four sites at the Little, Mountain Fork, and Glover Rivers. Also included are approximately 150 temporary sites that are sampled on a rotating basis at a rate of approximately 25 sites per year. These samples include limited covariates such as horizontal secchi disk visibility, conductivity, water temperature, turbidity, and TDS. Mann-Kendall tests generally failed to detect significant biological trends in this highly variable data set, as coefficient of variation values ranged from 37 to 176 percent.

The most comprehensive post-impoundment survey of the Cossatot River basin was the unpublished study by Fluker and Lee (2018). As part of their study of Leopard Darter Environmental DNA (eDNA), they made seining collections in the Cossatot River system from 2015 to 2017. They seined nine sites over six seasons at the Cossatot ranging from Hwy 246 upstream of Gillham Lake to the Hwy 24 bridge downstream in the Gulf Coastal Plain. Sampling was conducted for 45 to 60 minutes at each site and included riffles, runs, and pools. They collected 10,087 fish during their survey, including data from the Robinson Fork sites (12 sites total).

Additionally, several studies have collected fishes in the upper Cossatot and tributaries to understand timber management practices (Williams et al. 2002; 2003). Furthermore, Robison and Buchanan (2020) report that 131 species are known from the Red River basin in Arkansas, and AGFC tabulated that 101 species have been reported from the Cossatot.

Defining ecosystem flow alterations and restoration needs

An objective of the SRP is to determine the ecological, as well as social goals for the flows within the Cossatot, upstream and downstream of Gillham Dam. Ecological goals that stakeholders may identify include re-establishing physical processes that create and connect in-channel and off-channel habitats, recruiting and maintaining floodplain vegetation, reestablishment of the Leopard Darter, conserving other endangered and threatened species, and mitigating thermal impacts of the dam releases. Social goals that stakeholders may identify include Gillham Lake project purposes, downstream water withdrawals, recreation, and indigenous cultural values. In order to address the full complement of ecological and social goals, we suggest using a ‘holistic approach’, as described by Arthington et al. (2003) and Poff et al. (2017), to address the water requirements of the entire “riverine ecosystem”. The holistic approach is underpinned by the concept of the “natural flows paradigm” (Poff et al., 1997, as referenced in Arthington et al., 2003) and basic principles guiding river corridor restoration (Ward et al., 2001; Uehlinger et al., 2001, as referenced in Arthington et al., 2003). Holistic methodologies share a common objective - to maintain or restore the flow related biophysical components and ecological processes of in-stream and groundwater systems, floodplains and downstream receiving waters (Arthington et al., 2003). They seek to protect or restore a diverse set of socially and ecologically important river resources and processes across the full spectrum of low flows to flood events characterizing a river's flow regime within and between years (Opperman et al., 2018). Additionally, holistic methodologies can be enhanced by integrating modeled responses of river ecosystems to regulated flow change, and, therefore, environmental water requirements would be defined. Alternative water resource developments would then be evaluated by describing relationships between hydrology and the flow-related ecological processes governing biological diversity and river ecosystem integrity (Arthington et al., 2003). A fundamental aspect of holistic approaches has been to use a long-term hydrologic time series of daily or monthly flows to derive a set of static flow metrics (as detailed above) that quantify various aspects of the magnitude, frequency, timing, duration, and rate-of-change in discharge (Poff et al., 2017).

As part of the holistic approach, the concept of ‘designer flows’, described by Chen and Olden (2017), uses a multi-objective optimization framework to design dam operation releases that balance human water needs with the dual conservation targets of benefiting native fishes while disadvantaging invasives. Furthermore, Chen and Olden (2017) demonstrated that designer flows consistently outperformed natural flow mimicry in simultaneously meeting human water needs and promoting ecological goals. Additionally, reliance on the assumption of historical flow conditions is no longer feasible for long-term planning in most systems due to rapid climate change and other sources of change, including population growth and increasing water demand (Acreman et al., 2014a; Kopf et al., 2015; Poff et al., 2016, as referenced in Poff et al., 2017). The beauty of designer flows is that they can be engineered to meet human water demands while simultaneously incorporating multiple species associations with the entire hydrologic regime thereby taking advantage of mismatches between native and nonnative species responses to flow. These mismatches provide an opportunity to allocate water for dam releases that deliver multiple ecological outcomes, i.e., supporting native species conservation and nonnative species control (Chen and Olden, 2017). In this regard, rather than altering the water control releases below Gillham Dam to match “natural” flows, we propose the consideration of using the designer flow concept to define the hydrologic conditions, which, more than likely, will deviate from the natural flows concept, in order to promote the biological outcomes of interest for the species of conservation concern while maintaining the authorized purposes for Gillham Lake. Even though the flow regime more than likely will not match a natural hydrology, in order to find an optimal solution, the flow regime will likely include some elements of the natural hydrology critical to maintaining some of the ecological goals. A close-to-natural hydrology optimizes habitats within and among years. It is often better to provide each species excellent conditions every now and again than to provide every species mediocre or poor conditions all the time (which can be a consequence of simple threshold-based flow management). Furthermore, as many of the developed ‘designed’ flows will have high degrees of uncertainty, the need to be able to collect data to update these designs as the flow strategies are implemented will become critical in order to adjust for optimal management action.

Relationships between flow alteration and ecological response

Some understanding is needed of the relationship between changes in the flow regime and ecological response in order to set specific restoration targets; therefore, critical thresholds must be set above or below key functions or elements of the ecosystem (Poff et al., 2017). However, Yarnell et al., (2020) point out “eco-hydrological data are seldom available at sufficient density and biotic assemblages for which data are available may not broadly represent the ecology of the entire stream ecosystem”. Therefore, Yarnell et al., (2020) proposes a “functional flows approach” to manage for key flow components that will “preserve the necessary hydrologic signals upon which biophysical processes and native biological communities depend". These functional flows approach include "focusing on elements of the natural flow regime known to sustain important ecosystem processes" and "offering a pathway for linking the understanding of ecosystem processes with discrete, quantifiable measures of the flow regime for a broad range of native taxa and assemblages" (Yarnell et al., 2020).

There are several species of conservation concern within the Cossatot River corridor. These species occupy the instream, riparian, and floodplain portions of the River corridor and include both animals and plants. Of these species of conservation concern, several are more notable and are currently being surveyed by the AGFC. These species of conservation concern include the Leopard Darter (*Percina pantherina*), the Rabbitsfoot mussel (*Quadrula cylindrica cylindrica*) and the Ouachita Rock Pocketbook (*Arkansia wheeleri*). The Leopard Darter and Rabbitsfoot mussel are currently (2020) listed by the state as "Threatened" species, while the Ouachita Rock Pocketbook is listed as an "Endangered" species. There are few data with direct measures of flow requirements for the listed biota of concern for the Cossatot River watershed; however, the State sponsors, particularly AGFC, are currently (2021) collecting data (occurrence, abundance, and detection) on these threatened and endangered species under different flow conditions.

Aquatic Species of Greatest Conservation Need (SGCN)

Fish SGCN

Of the 60 fish species of greatest conservation need (SGCN) in the State of Arkansas, 23 percent (14 species) have been detected within the Cossatot River basin (Table 8). Some of these are known only from historic records that represent species likely extirpated from the system, and others are known from very few collecting events in the watershed and may be so rarely encountered or difficult to detect that inferring environmental flow-biological response relationships for the Cossatot based on these rare species may not be feasible. The Paddlefish (*Polyodon spathula*) and American Eel (*Anguilla rostrata*) were historically known from the lowest reaches of the Cossatot near its confluence with the Little River (Cloutman and Olmstead, 1974), but it has been decades since either of these species were recorded in the watershed and both are likely extirpated. Other species known from very few collection events (historic or recent) in the Cossatot include Lake Chubsucker (*Erimyzon sucetta*), Bluehead Shiner (*Pteronotropis hubbsi*), Western Sand Darter (*Ammocrypta clara*), Crystal Darter (*Crystallaria asprella*), Swamp Darter (*Etheostoma fusiforme*), and Goldstripe Darter (*Etheostoma parvipinne*).

Aspects of the life histories for the remaining six SGCN species could inform potentially important flow-response hypotheses for the Cossatot River. They are the threatened Leopard Darter (*Percina pantherina*), Ouachita Mountain Shiner (*Lythrurus snelsoni*), Blackspot Shiner (*Notropis atrocaudalis*), Kiamichi Shiner (*Notropis ortenburgeri*), Rocky Shiner (*Notropis suttkusi*), and Western Starhead Topminnow (*Fundulus blairae*) (Table 9).

Table 8. Aquatic Species of Greatest Conservation Need (SGCN) in the Cossatot River watershed. State Conservation Ranks: S1=critically imperiled in Arkansas, S2=imperiled in Arkansas, S3=vulnerable in Arkansas, S4=apparently secure in Arkansas, SNR=not ranked in Arkansas. S5=secure in Arkansas. Global Conservation Ranks: G1=critically imperiled globally, G2=imperiled globally, G3=vulnerable globally, G4=secure globally, G5=critically imperiled globally.

Common Name	Scientific Name	State Conservation Status	Global Conservation Status	Federal Status	Documented within last 30 years
		FISH			
Paddlefish	*Polyodon spathula*	S3	G4	-	No
American Eel	*Anguilla rostrata*	S3	G4	-	No
Ouachita Mountain Shiner	*Lythrurus snelsoni*	S2	G3G4	-	Yes
Blackspot Shiner	*Notropis atrocaudalis*	S3	G4	-	Yes
Kiamichi Shiner	*Notropis ortenburgeri*	S3	G3	-	Yes
Rocky Shiner	*Notropis suttkusi*	S2	G3G4	-	Yes
Bluehead Shiner	*Pteronotrpis hubbsi*	S3	G3	-	
Lake Chubsucker	*Erimyzon sucetta*	S3	G5	-	Yes
Western Starhead Topminnow	*Fundulus blairae*	S2	G4	-	Yes
Western Sand Darter	*Ammocrypta clara*	S3	G3	-	No
Crystal Darter	*Crystallaria asprella*	S2	G3	-	No
Swamp Darter	*Etheostoma fusiforme*	S3	G5	-	Yes
Goldstripe Darter	*Etheostoma parvipinne*	S3	G4G5	-	Yes
Leopard Darter	*Percina pantherina*			Threatened	Yes
		MUSSELS			
Rabbitsfoot	*Theliderma cylindrica*	S3	G3G4	Threatened	Yes
Little Spectaclecase	*Villosa lienosa*	S3	G5	-	Yes
Ouachita Kidneyshell	*Ptychobranchus occidentalis*	S3	G3G4	-	Yes
Red River Fatmucket	*Lampsilis sp B cf hydiana*	S2	GNR	-	Yes
Scaleshell	*Leptodea leptodon*	S2	G1G2	Endangered	No
Ohio Pigtoe	*Pleurobema cordatum*	S3	G4	-	Yes
Louisiana Pigtoe	*Pleurobema riddellii*	S1	G1G2	-	Yes

Round Pigtoe	*Pleurobema sintoxia*	S3	G4G5	-	Yes
Southern Mapleleaf	*Quadrula apiculata*	S3	G5	-	Yes
Winged Mapleleaf	*Quadrula fragosa*	G1	S1	Endangered	Yes
Texas Lilliput	*Toxolasma texasiense*	G4	S3	-	No
		OTHER TAXA			
Little River Creek Crayfish*	*Faxonius leptogonopodus*	S2S3	G3	-	Yes
Ozark Clubtail	*Gomphurus ozarkensis*	S1	G4	-	Yes
Ouachita Diving Beetle	*Heterosternuta ouachita*	S2	GNR	-	No
Ouachita Shorebug	*Pentacora ouachita*	S1	GNR	-	No

*Provisional SGCN status and state/global conservation status ranks pending completion of Arkansas crayfish conservation status reassessment (Wagner and Lynch, 2020)

Table 9. Life history, trophic ecology, feeding, and habitat traits for six fish Species of Greatest Conservation Need (SGCN) in the Cossatot River. B=Benthic, W=Water Column, S=Surface (Frimpong, 2009, Robison and Buchanan, 2020).

Common Name	Scientific Name	Feeding Zone	Reproductive Ecology	Lifespan (years)	Fecundity (count)	Spawning Season	Above/Below Fall Line	Above/Below Gillham Lake
Leopard Darter	*Percina pantherina*	B	non-guarding, brood hider, lithophilic (rock-gravel), serial	2	410	Mar-May	above	above
Ouachita Mountain Shiner	*Lythrurus snelsoni*	B/W/S	unknown	2	200	May-July	above	above/below
Blackspot Shiner	*Notropis atrocaudalis*	B/W/S	non-guarding, open substrate, litho-pelagophilic, serial	2	1,044	Mar-June	below	below
Kiamichi Shiner	*Notropis ortenburgeri*	B/W/S	unknown	unknown	unknown	unknown	above/below	below
Rocky Shiner	*Notropis suttkusi*	B/W/S	non-guarding, brood hider, lithophilic (rock-gravel)	3	1,500	Mar-Aug	below	below
Western Starhead Topminnow	*Fundulus blairae*	W/S	non-guarding, open substrate, phyto-lithophilic, serial	2.5	800	April-Aug	below	below

Leopard Darter

The Leopard Darter (*Percina pantherina*), endemic to the Little River watershed in southeastern Oklahoma and adjacent southwestern Arkansas, is by far the most imperiled fish species in the Cossatot and only found above Gillham Lake. This species was listed by the USFWS as threatened under the Endangered Species Act primarily due to the impacts of dams within its limited range (USFWS 1978). This Ouachita Mountain species is primarily an inhabitant of pools during the summer and fall but moves into riffles during the spring breeding season (Jones et al., 1984; James et al., 1991; Robison and Buchanan, 2020). Movement into riffles begins in late February and spawning occurs from mid-March through mid-May at water temperatures of 54 to 68 °F (James et al., 1991). Spawning typically occurs at depths of 12 to 35 inches over fine gravel substrate at water velocities of 0 to 1,765 feet/second (ft/s), primarily in the tail-waters of riffles (James et al., 1991). This species is categorized as a non-guarding, brood-hiding, lithophilic (rock-gravel) spawner (Frimpong, 2009), and the species is known to bury their eggs in fine gravel (James et al., 1991). This is a short-lived species with a typical lifespan of around 18 months; most individuals spawn only once during a lifetime (James et al., 1991; Zale et al., 1994). While the Leopard Darter is adapted to living in pools during much of the year, it is intolerant of reservoir conditions (Robison and Buchanan, 2020). Preferred non-spawning substrates range from a mixture of gravel, cobble and rubble to boulder (Jones et al., 1984; James et al., 1991; Zale et al., 1994). This is a strictly invertivorous species that consumes the larvae of midges, mayflies, black flies, and other insects (Robison, 1978; James et al., 1991; Williams et al., 2006).

Fragmentation of the species' range due to impoundment of reservoirs, such as Gillham Lake in Arkansas and others in Oklahoma, have eliminated habitat and reduced gene flow in the species, which was declared federally threatened in 1978. The species persisted a short while below Gillham Lake after construction was completed, having last been detected there in 1979 (ANHC, 2020). The Leopard Darter is now thought to remain at very low densities in the upper Cossatot above Gillham Lake at only a handful of isolated pools, where it has been detected in extremely small numbers in recent surveys (Quinn et al., 2019). Recent surveys of the Robinson Fork and in the lower Cossatot in 2019 failed to detect the species and it is now thought to likely be extirpated from those systems (Morris et al., 2020). Echelle et al. (1999) analyzed allele frequencies in the species throughout its range and found that the species consisted of three primary clades – one in the Little and Glover Rivers in Oklahoma, one in the Mountain Fork River in Oklahoma and Arkansas, and one in the Cossatot and Robinson Fork. Schwemm (2013) found that the Cossatot population had the lowest genetic diversity and he considered the population in the Cossatot near extinction from the genetic effects of extremely low population size. The USFWS and partners are currently considering a plan to genetically rescue the Cossatot population by augmenting it with individuals transferred from the more western populations. Increasing water temperatures in the Cossatot may also be adversely affecting the species, which is known to seek thermal refuge in deeper waters when temperatures exceed 84 °F (Schaefer et al., 2003). This species is considered globally imperiled and critically imperiled in Arkansas (ANHC, 2020).

Ouachita Mountain Shiner

The Ouachita Mountain Shiner (*Lythrurus snelsoni*) is another Little River watershed highland endemic found in the upper Cossatot. This species prefers small to medium-sized high elevation streams with clear water and high gradient, particularly in pools along stream margins lined with Water Willow (*Justicia americana*) and boulder-dominated substrate (Taylor and Lienesch, 1995; Robison and Buchanan, 2020). A well-developed riffle-pool sequence providing high variability in stream velocity may be important to this species (Taylor and Lienesch, 1995). Taylor and Lienesch (1996) considered *L. snelsoni* parapatric with the related Redfin Shiner (*Lythrurus umbratilis*), which is common in the lower Cossatot below the Fall Line, although Robison and Buchanan (2020) do not consider the two entirely parapatric. This species feeds at both the surface and in the water column on a variety of invertebrate prey, including midges, black flies, and mayfly larvae (Miller and Robison, 2004; Robison and Buchanan, 2020).

Terrestrial insects dominate its diet during summer months (Robison and Buchanan, 2020). Reproduction occurs from May to late July. In Arkansas this species is restricted to portions of the Cossatot, Mountain Fork, and Rolling Fork Rivers above the Fall Line (Robison and Buchanan, 2020). It has been collected occasionally in reservoirs, including Gillham Lake (Buchanan, 2005). It is considered imperiled in Arkansas by the Arkansas Natural Heritage Commission (2020). Robison and Buchanan (2020) regard *L. snelsoni* as threatened in Arkansas due to its restricted distribution and as a result of threats including clear-cutting and reservoir construction.

Blackspot Shiner

The Blackspot Shiner (*Notropis atrocaudalis)* is more widespread than the previous two species, but is uncommon in Arkansas, where it is restricted to lower reaches of the Little River system and Red River tributaries (Robison and Buchanan, 2020). It is also found in adjacent portions of Oklahoma, Texas, and Louisiana. Unlike the previous two species which are restricted to the Ouachita Mountain portion of the Cossatot, this species is found in the watershed only below the Fall Line in the Gulf Coastal Plain. This species prefers smaller streams and headwaters (Robison and Buchanan, 2020). It is a benthic feeder (Bean et al., 2010) with a diet made up largely of aquatic insects, including larval dipterans, mayflies, caddisflies, and beetles (Robison and Buchanan, 2020). Bean et al. (2010) hypothesized that this species may have a strong role in structuring stream communities in portions of its range where it is common. Reproduction occurs from March through June. This species is categorized as a non-guarding, open substrate, litho-pelagophilic spawner (Frimpong and Angermeier, 2009). It is characterized by early maturation, a relatively short life span, extended spawning periods, and downstream drift of eggs and larvae (Bean et al., 2010), traits which are typically associated with stream fishes adapted to variable systems. Robison and Buchanan (2020) consider this species endangered in Arkansas due to its small range and few known occurrences. It is considered vulnerable in Arkansas by the Arkansas Natural Heritage Commission (2020).

Kiamichi Shiner

The Kiamichi Shiner (*Notropis ortenburgeri*) is a regional endemic found only in Arkansas and Oklahoma, where it has a somewhat patchy distribution in the Ouachita, Red, and Arkansas River basins (Robison and Buchanan, 2020). It is rare in Arkansas. This is a pool-dwelling species that has an affinity for clear upland streams with gravel, rubble, or boulder substrates (Robison, 2005) where it may favor the ends of pools at the beginnings or ends of riffles (Black, 1940). It is often found near beds of Water Willow at the edges of pools where stream velocity is slight (Robison and Buchanan, 2020) and may be intolerant of turbidity and ecological perturbations that result from clear-cutting, as well as reservoir conditions (Robison, 2005). In the Cossatot, this species is known from above and below the Fall Line but has not been collected above Gillham Lake. Life history information on this species is scarce, but the species appears to be an invertivore (Robison and Buchanan, 2020) that feeds primarily on the substrate but may also take some prey (such as winged terrestrial insects) at the surface. In addition to mayfly nymphs, dipteran larvae, and other benthic invertebrates, this species may consume detritus, microcrustaceans, and diatoms (Robison and Buchanan, 2020). Reproductive biology of this species is poorly known. Robison and Buchanan (2020) consider *N. ortenburgeri* threatened in Arkansas, and it is considered vulnerable both in Arkansas and globally by the Arkansas Natural Heritage Commission (2020).

Rocky Shiner

The Rocky Shiner (*Notropis suttkusi*) is a species with a very small overall range, found only in Red River tributaries from south-central Oklahoma to extreme southwest Arkansas (Miller and Robison, 2004; Robison and Buchanan, 2020). In Arkansas it is known only from lower reaches of the Rolling Fork, Cossatot, Saline, and Little Rivers (Robison and Buchanan, 2020). In the Cossatot, it has not been collected above the Fall Line. This species prefers clear streams with moderate to high gradient and gravel or rubble substrate (Humphries and Cashner, 1994) where it is typically found in areas of moderate streamflow at the periphery of riffles. *N. suttkusi* is occasionally collected in habitats with low to moderate turbidity (Robison and Buchanan, 2020). It may utilize deep pools (depth greater than 6 ft) as thermal refugia during spring and summer. This species tends to avoid headwater environments where it may be replaced by *Lythrurus snelsoni* (Robison and Buchanan, 2020). This species apparently feeds throughout the water column but much of its diet comes from items taken at the water surface, dominated by winged adult insects, including dipterans, beetles, and odonates, although it also consumes aquatic insect larvae (Robison and Buchanan, 2020). The reproductive season begins in late March and continues through early August (Robison and Buchanan, 2020). Assuming a similar mode of reproduction as the closely related *Notropis rubellus* and *Notropis percobromus*, this species is expected to be categorized as a non-guarding, brood-hiding, lithophilic (rock-gravel) spawner (Frimpong and Angermeier, 2009). This species is considered imperiled in Arkansas by the Arkansas Natural Heritage Commission (2020). Robison and Buchanan (2020) regard the species as threatened, and possibly endangered, in Arkansas.

Western Starhead Topminnow

The Western Starhead Topminnow (*Fundulus blairae*) is found primarily in the Red River basin in Arkansas, Oklahoma, Texas, and Louisiana, and disjunctly, in other lower Mississippi River tributaries in southwestern Mississippi, as well as Gulf Slope drainages in Florida, Alabama, and Texas. Populations tend to be sporadically distributed and small (Robison and Buchanan, 2020). In Arkansas, the species' range is limited to the southwestern corner of the state in the lower sections of the Saline, Cossatot, Rolling Fork, and Little Rivers, although it is also found in oxbow lakes and Red River backwaters (Robison and Buchanan, 2020). This is a lowland species of quiet, heavily vegetated waters and prefers clear water with a soft mud and detritus bottom. It is a surface-feeder whose diet is dominated by terrestrial dipterans, though it also consumes aquatic larval insects, as well as other terrestrial insects (Robison and Buchanan, 2020). This species has a long breeding season, extending from April through August, and is thought to be a serial spawner (Robison and Buchanan, 2020). Specific spawning behavior is not known for this species, but it is likely similar to the closely related Starhead Topminnow (*Fundulus dispar*) which replaces it eastward in southern Arkansas. That species is considered a non-guarding, open substrate, phyto-lithophilic spawner (Frimpong and Angermeier, 2009).

Fish species of interest that are not imperiled due to flow dependencies

Although to some extent a focus on aquatic SGCN in the development of hypotheses about flow-response relationships in the Cossatot is warranted, as it could help inform important management decisions, it may be worth broadening the scope of the study to include other aquatic species for a variety of reasons. SGCN are, almost by definition, species that are rare and may be seldom encountered on surveys. Low occupancy and detection probability among these species may pose a challenge in the development of solid flow hypotheses; however, the real challenge will be testing the developed hypotheses and updating them since ecological response data could be difficult to gather. Furthermore, some of the other more frequently encountered species in the Cossatot have well-established life history traits and appear to show strong trends when collection data between historic and more recent surveys are compared.

The Highland Stoneroller (*Campostoma spadiceum*) is a primarily algivorous minnow that is often the most abundant fish species in upland streams of the Ouachitas (Robison and Buchanan, 2020). Like other stoneroller species, it is a critically important component of aquatic communities in which it occurs, having been documented to have direct or indirect effects on important structural or functional properties of small stream ecosystems (e.g., Power et al., 1985), including algal community composition or productivity, uptake and dynamics of organic matter, invertebrate community composition and life history, movement of materials, carbon-nitrogen ratios, and standing crops of bacteria, as well as predator-driven cascades (Robison and Buchanan, 2020). Because of its abundance and the important role it plays in stream communities and ecosystems, flow-response relationship hypotheses focusing on this species could be important.

Four other cyprinid (minnow) species that occur in the Cossatot may be instructive in the formation of environmental flow hypotheses due to a large number of occurrences, and apparent trends in abundance when comparing historic data to more recent data. These are the Bigeye Shiner (*Notropis boops*), Steelcolor Shiner (*Cyprinella whipplei*), Redfin Shiner (*Lythrurus umbratilis*), and Ribbon Shiner (*Lythrurus fumeus*). All but the latter species are known from both above and below the Fall Line in the Cossatot watershed, while the Ribbon Shiner is a lowland species not known from the upper portion of the River. The Bigeye Shiner is considered a non-guarding, open substrate lithophilic (gravel-sand) spawner (Frimpong and Angermeier, 2009). Both *Lythrurus* species are considered non-guarding open substrate phyto-lithophlic spawners for whom aquatic vegetation may be important. Additionally, the Redfin Shiner is known to be a broadcast spawner (Robison and Buchanan, 2020) noted for dense spawning aggregations (Hunter and Hasler, 1965). Broadcast spawning species are predicted by Carlisle et al. (2009) to have an advantage over other species in systems experiencing both diminished maximum and inflated minimum flows. This species is also known to spawn in association with the nests of various sunfish species, whose nests provide a clean, silt-free substrate for egg development and protection from predators by male sunfish (Robison and Buchanan, 2020), so it could be hypothesized that flow conditions that favor sunfish nesting would favor proliferation of this species as well. The Steelcolor Shiner is considered a non-guarding, brood-hiding speleophil (cavity generalist) (Frimpong and Angermeier, 2009). This species deposits its eggs under loose bark or in crevices of submerged logs and tree roots in moderate to swift current (Pflieger, 1965). All four of these species do much of their feeding at the surface (Robison and Buchanan, 2020).

Three other fish species commonly encountered in the Cossatot both above and below the Fall Line and Gillham Lake are the Longear Sunfish (*Lepomis megalotis*), Blackspotted Topminnow (*Fundulus olivaceus*), and Orangebelly Darter (*Etheostoma radiosum*). An abundance of occurrence data for these species, as well as apparent trends toward an increase in all three species when comparing historic to recent data suggest they may all be useful in formulating meaningful flow-response hypotheses for the Cossatot. These three species exhibit a variety of reproductive ecologies, trophic ecology traits, and life histories that could be used to develop flow-response hypotheses. All three species are generalist species that appear to respond positively to the altered flows within the River i.e., are tolerant to dams.

A final fish species, the Striped Shiner (*Luxilus chrysocephalus*) may also warrant inclusion in development of flow-response relationships due to its particular importance as a general host fish for several rare species of mussel species (e.g., Ford and Oliver, 2015). This species is also found in both the upland and lowland portions of the Cossatot.

Mussels SGCN

Life cycles and life history traits of freshwater mussels are extremely complex. The spawning strategy for freshwater mussels is unique in that males release sperm into the water column collected by females during normal siphoning. Upon fertilization, embryos develop into glochidia; this period of gravidity is also called brooding. Generally, mussels are categorized into short-term and long-term brooders. Typically, short-term brooders spawn, brood, and release glochidia over a 2 to 6-week time period in late spring or summer, while long-term brooders spawn in late summer or early fall, brood eggs over winter and release glochidia the following spring to summer.

Because, the life cycles and life history traits of freshwater mussels are extremely complex, they are vulnerable to hydrologic modification. Impoundments and water development projects are a primary cause of decline of freshwater mussels in many streams (Layzer et al., 1993; Vaughn and Taylor, 1999;

Moles and Layzer, 2008). Timing, duration, magnitude, frequency, and rate of change of appropriate environmental flows are paramount to support freshwater mussels during critical periods of gravidity, spawning, host fish infestation, and juvenile settlement (Neves and Widlak, 1987; Holland-Bartels, 1990; Layzer and Madison, 1995; Hardison and Layzer, 2001; Daraio et al., 2010). Increases in the magnitude of high flows may prevent juvenile mussels from settling in new habitat or dislodge newly settled juveniles (Neves and Widlak, 1987; Holland and Bartels, 1990; Layzer and Madison, 1995; Hardison and Layzer, 2001; Daraio et al., 2010). In contrast, variation in the timing of high and low flows may cause mussel beds to be exposed to altered temperature regimes (Galbraith and Vaughn, 2011) or indirectly affect mussels by preventing interactions between host fish species and mussels resulting in a reduction of or complete lack of glochidial attachment onto suitable hosts (Freeman and Marcinek, 2006, Gido et al., 2010). Additionally, hydrological impacts compounded with physiological limitations, such as limited mobility, disrupt feeding, survival, and reproduction (Yeager et al., 1993; Poff et al., 1997; Kat, 1982; Young and Williams, 1983; Ahlstedt and Tuberville, 1997; Hastie et al., 2001). Deleterious effects to mussel assemblages are further exacerbated by extreme hydrologic releases during spawning and larvae (glochidia) infestation periods, which can limit assemblage recruitment (Neves and Widlak, 1987; Layzer and Madison, 1995). Successful mussel reproduction is dependent on flow conditions that resemble the flow regime under which mussels and their host fish co-evolved (Barnhart et al., 2008). Mussels and their fish host species are intrinsically linked as glochidia are ectoparasites that require a host fish for attachment and metamorphosis. Therefore, any hydrologic modifications that limit host fish abundance will inevitably also alter mussel assemblages (Watters, 1993; Haag and Warren, 1998; Vaughn and Taylor, 1999).

Federally listed mussels of conservation concern in the Cossatot are Rabbitsfoot (*Theliderma cylindrica)*, Ouachita Rock Pocketbook (*Arcidens wheeleri*), Pink Mucket (*Lampsilis abrupta*), Scaleshell (*Leptodea leptodon)* and the Winged Mapleleaf (*Quadrula fragosa)*. The Threatened Rabbitsfoot mussel is known from six locations in the Cossatot (Bouldin et al., 2013). While the authors (K. Moles) provide the first continuous mussel survey of the Cossatot downstream of Gillham Lake, it is not considered exhaustive, as it was a qualitative survey and no excavation was conducted. Therefore, Rabbitsfoot mussels could occur at additional locations. The Rabbitsfoot mussel is a short-term brooder and host specialist that uses various minnows (cyprinids) as its host fishes. Spawning of Rabbitsfoot mussel has been observed in June (K.Moles, unpublished data). Confirmed host fish of the Rabbitsfoot in the Little River drainage are Striped Shiner (*Luxilus chrysocephalus*), Blacktail Shiner (*Cyprinella venusta*), Emerald Shiner (*Notropis atherinoides*), and Blackstripe Topminnow (*Fundulus notatus*) (Fobian, 2007) (Table 10).

While there are no records of the Endangered Ouachita Rock Pocketbook from the Cossatot River proper, it is assumed the species may be present in the lower reaches of the River since it is present in the Little River. Spawning of the Ouachita Rock Pocketbook is believed to occur in the first two weeks of October. Partially gravid females containing developing glochidia have been observed as early as October 3 (K. Moles, unpublished data). Ouachita Rock Pocketbook are long-term brooders remaining gravid until the following spring. The precise time of host fish infestation is unknown but is currently (2021) being investigated by AGFC personnel. Ouachita Rock Pocketbook are glochidial broadcasters and host fish generalists. Confirmed host fish of the Ouachita Rock Pocketbook in the Little River drainage include River Carpsucker (*Carpiodes carpio*), Longear Sunfish (*Lepomis megalotis*), Green Sunfish (*Lepomis cyanellus*), Bluegill (*Lepomis macrochirus*), Warmouth (*Lepomis gulosus*), Largemouth Bass (*Micropterus salmoides*), White Crappie (*Pomoxis annularis*), Black Crappie (*Pomoxis nigromaculatus*), Emerald Shiner (*Notropis atherinoides*), and Bleeding Shiner (*Luxilus zonatus*) (Seagraves, 2006) (Table 10).

The Endangered Winged Mapleleaf is known from only one location in the lower Cossatot (Bouldin et al., 2013). Winged Mapleleaf is a short-term brooder but unlike most other short-term brooders it spawns in the fall. Gravid females have been observed in Arkansas in late October and it is assumed the spawning occurs in early to mid-October. Confirmed host fish for the Winged Mapleleaf are Channel Catfish (*Ictalurus punctatus*) and Blue Catfish (*Ictalurus furcatus*) (Steingraeber et al., 2004) (Table 10).

While there are no records of the Endangered Pink Mucket from the Cossatot River proper, it is assumed the species may be present in the lower reaches of the river since it occurs in the Little River. The Pink Mucket is a long-term brooder and generally spawns in early October and releases glochidia the following spring and into early summer. Confirmed host fish for the Pink Mucket are Largemouth Bass (*Micropterus salmoides*), Spotted Bass (*Micropterus punctulatus*), Smallmouth Bass (*Micropterus dolomieui)*, and Walleye (*Sander vitreus*) (Barnhart et al., 1997) (Table 10).

The Endangered Scaleshell is known from only one site, near Lockesburg, in the Cossatot (Figure 2). As a result of their burrowing behavior, Scaleshell are usually underrepresented in qualitative mussel surveys. Scaleshell is a long-term brooder reported to spawn in August and brood glochidia until the following spring (Barnhart et al., 1998). The only confirmed host fish for the Scaleshell is the Freshwater Drum (*Aplodinotus grunniens*) (Barnhart et al., 1998) (Table 10).

Table 10. Critical reproductive periods and host fish for listed mussels in the Cossatot River.

Common Name	Species	Spawning Period	Glochidial Release	Host Fish	Juvenile Excystment
Rabbitsfoot	*Theliderma cylindrica*	June	June	Striped Shiner, Blacktail Shiner, Emerald Shiner, Blackstripe Topminnow	July
Ouachita Rock Pocketbook	*Arcidens wheeleri*	October	March/April	Carpsucker, Longear Sunfish, Green Sunfish, Bluegill, Warmouth, Largemouth Bass, White Crappie, Black Crappie, Emerald Shiner, Bleeding Shiner	May
Winged Mapleleaf	*Quadrula fragosa*	October	October	Channel Catfish, Blue Catfish	Not known
Pink Mucket	*Lampsilis abrupta*	October	March/April	Largemouth Bass, Spotted Bass, Smallmouth Bass, Walleye	May
Scaleshell	*Leptodea leptodon*	August	April/May	Freshwater Drum	Not known

Other aquatic SGCN

Fish and mussels comprise the majority of aquatic SGCN in the Cossatot but there are a few other SGCN from other taxa that have been documented in the watershed. While crayfish diversity is high in southwestern Arkansas, including a number of SGCN within the watershed, all of these are terrestrial (burrowing) species that primarily live below ground at the water table and do not inhabit the instream river channel. While flow alteration that potentially impacts the water table could affect these species, such relationships would be difficult to determine, in part because the life histories of these species are so poorly known. One stream-dwelling species, the Little River Creek Crayfish (*Faxonius leptogonopodus*), known from the Cossatot above Gillham Lake, is not currently considered a SGCN, but will be reassigned SGCN status in the near future (vulnerable to imperiled in Arkansas and vulnerable globally), given a recent reassessment of crayfish conservation status in Arkansas (Brian Wagner, Arkansas Game and Fish Commission, and Dustin Lynch, Arkansas Natural Heritage Commission, pers. comm.).

A handful of rare aquatic insect species (fully or semi-aquatic during at least some portion of their life histories) occur in the watershed. The Ozark Clubtail (*Gomphurus ozarkensis*), a dragonfly considered critically imperiled in Arkansas, is known from a single collection on a gravel bar in the river in 2004 (ANHC, 2020). The Ouachita Diving Beetle (*Heterosternuta ouachita*), an imperiled state endemic, has been collected in Harris Creek (a tributary of the Cossatot) and is known primarily from highland streams at scattered localities across Arkansas (ANHC, 2020). The Ouachita Shorebug (*Pentacora ouachita*) is a poorly known, critically imperiled state endemic known only from a few records in the Ouachita Highlands, including an individual collected near Cossatot Falls in 1974.

Dependencies between the ecological indicators and specific environmental flow components

Although there is little data relating specific ecological indicators for the Cossatot to specific environmental flow components, Carlisle et al. (2019) related biological integrity to certain streamflow characteristics at a regional scale. Carlisle et al. (2019) related individual indicators of streamflow modification, by taking the ratio of observed values to expected for each streamflow characteristic, to biological integrity for six regions nationally. The closer the value to 1, the less altered the stream. The majority of the Cossatot watershed lies within the central and southeast plains region as described by Carlisle et al. (2019); however, the northern portion lies within the east highlands region. Carlisle et al. (2019) determined that in general, biological community integrity is best explained by more than one dimension of streamflow modification. The central and southeast plains region most closely related biological impairment to duration of high flows combined with the frequency of high flows in April (spawning season for most fish), the annual variability of flow magnitude in January and the annual variability of flow magnitude in October (Carlisle et al., 2019). The east highlands region most closely related biological impairment to annual variability and the magnitude of maximum April flows (spawning season for most fish); magnitude of monthly average flows during winter combined with magnitude of spring flows and magnitude of maximum annual flows; variation of winter monthly flows combined with variation of high annual flows and flow reversals/rises; and timing of annual maximum flows combined with magnitude of monthly flows during summer (Carlisle et al., 2019). Using the combinations of streamflow characteristic modifications as explained by Carlisle et al. (2019), we can formulate empirically based hypotheses about the specific components of streamflow regimes that are critical to the aquatic communities within the Cossatot River.

Flow-response hypotheses for fishes in the Cossatot River

A strong negative relationship is expected between Leopard Darter abundance and the magnitude and duration that Gillham Lake exceeds 515 ft elevation. Leopard Darters are considered intolerant of lentic habitat conditions and recent increases in the magnitude and frequency of reservoir inundation associated with climate change may be a contributing factor to the decline of the population. Any measures that can be taken to reduce the magnitude and duration of lentic conditions in this "flood pool" may benefit this threatened species. Climate change may increase the frequency of large floods that may be associated with high reservoir elevations. As riffle obligate spawners, high lake levels (greater than 530 ft) that cause lentic conditions in spawning riffles during the March-May spawning season may have strong negative effects on reproductive success and could possibly lead to extirpation. Leopard Darter may be hypothesized as an opportunistic strategist (e.g., Mims and Olden, 2012), as these are generally r-selected species that are small, have early maturation, low juvenile survival, and they occupy rivers defined by frequent and intense disturbances such as floods. Thus, Leopard Darter likely require unpredictable disturbances such as flooding events to compete with other species. As brood hiders, they bury eggs in fine gravel possibly to protect the eggs from higher flows or from predators (McManamay and Frimpong, 2015).

During the fall, releases from storage impoundments have low DO levels. A strong negative relationship is expected between biological integrity or species richness and fall high flow releases of these low DO waters (< 4 mg/L) from Gillham Dam. High inflow years often present the greatest issue, as they add nutrients to the reservoirs that can lead to anoxic hypolimnetic conditions. There is a lack of data on inter-annual variability of DO for the lower Cossatot, but low DO could be a factor influencing fish assemblages in the vicinity of Gillham Dam. Therefore, it is hypothesized that fish Index of Biotic Integrity (IBI) scores will be lower near Gillham Dam and may partially recover downstream. In other words, and assuming no other major disturbances, i.e., major land-use changes, water withdrawals, in-stream mining, etc., fish populations generally recover with increasing downstream distance from a dam.

Reduction of high flow magnitudes and inflation of low flows below Gillham Dam are hypothesized to be associated with declines in imperiled species for this portion of the Cossatot. Carlisle et al. (2010) examined the biological alteration associated with stream flow alteration at 2,888 sites throughout the United States. This study reported 86 percent of sites had altered minimum and maximum flow magnitude. Diminished flow magnitudes were predictors of biological impairment (i.e., IBI scores), and systems with depleted flows tend to have generalist species that are tolerant of silt substrate and lentic habitats. Streams with diminished minimum or maximum flows shifted from simple nesting to nest-guarding or broadcast-spawning strategies and active swimmers replaced benthic-oriented streamlined fish species**.** For this particular study, sites were generally classified as "impaired" if observed magnitudes of maximum flow were less than 0.4 of expected natural flow magnitudes (ratio of observed to expected), if observed minimum flows were less than 0.4 of the expected flow, and if observed minimum flows were inflated to greater than 1.8 of the expected minima. Therefore, significant biotic recovery may be hypothesized to occur below Gillham Dam when high flow magnitude metrics are restored to greater than 0.4 of expected values and low flow metrics are less than 1.8 of expected minima. For the Carlisle et al. study, the lowest category of biological impairment had 0 to 25 percent streamflow deviation from the expected natural magnitude. Higher 1-day maximum flows have been associated with greater fish species richness in other studies (e.g., McManamay et al., 2013). Poff and Zimmerman (2010) in their review of ecological responses to altered flows also noted that fish diversity consistently declined where flow magnitudes exceeded 50 percent change.

Cravens et al. (2010) evaluated how flow components and species traits influenced young-of-the-year (YOY) fish abundance among three North American streams, including the Kankakee, Flint, and Tallapoosa rivers (regulated, hydropeaking projects). They concluded that YOY fish density was positively related to short-term (10-day maximum) high flows during the spawning season; YOY fish density was negatively associated to flow variability (10-day discharge standard deviation (SD)) during the rearing period; and flow component effects varied by species and life history traits. Hierarchical linear models were used to account for 61 percent of the variability in YOY fish density, and this best model included: (1) the maximum 10-day discharge during the spawning period, (2) broad-cast spawning life history, (3) the interaction of the two previous parameters, (4) cruiser locomotion, (5) the interaction of the minimum 10-day discharge SD during the rearing period by cruiser locomotion, and (6) adult/juvenile density in the previous year. Maximum 10-day discharge during the spawn was positively related to YOY fish density but was variable among species, and broadcast spawning and spawning duration accounted for 31 and 26 percent of the variation, respectively. The effect of high flows was weaker for species with a long (90-day) spawning duration or with broadcast spawning. YOY density was greater for species with a long (90-day) spawning duration and lower for broadcast spawners. They also found that low-flow discharge variability (minimum 10-day discharge SD) during the rearing period was negatively related to YOY density, but relationships varied by 82 percent among species (Cravens et al., 2010). While cruiser morphology traits had higher densities, a strong negative relationship was observed between YOY cruiser density and minimum 10-day discharge SD during the rearing period. This suggests that Centrarchids are highly influenced by discharge variability during rearing periods. A conclusion of this study was that short-term, 10-day maximum high flows strongly affect spawning success. These high flows maintain channel heterogeneity, flush fine sediments, and allow energy flow with floodplains. High flows are also needed as a cue to initiate spawning of some species. Another conclusion was that short-term variation in flows had great effects on YOY fish densities during the rearing period. Fish that swim in the water column (cruisers) were vulnerable to short-term variation in discharge. Spawning duration is an important fish trait and species with long spawning durations often had greater reproductive success and were less influenced by short-term high flows during the spawning season. The example given was Bluegill spawn over 4 months but spotted suckers spawn over 1.5 months and only once per year.

Previous studies that examine the effects of dams on fish assemblages may provide important insight into flow-ecology relationships for fishes. Larval fish abundance was more than three times higher in a non-regulated river (77 percent of fish collected) than a regulated, hydropeaking river in Alabama (Scheidegger and Bain, 1995). Minnows comprised 71 percent of larval fish in the reference stream but comprised only 12 percent at the regulated river. The regulated system had a much higher proportion of Centrarchidae (stronger, faster swimmers that can quickly find flow refuges behind logs, undercut banks, etc.) close to the dam (7.5 miles downstream) more than likely because of the hydropeaking nature of the system (sub-daily fluctuations), and proportional abundance declined far downstream of the dam (i.e., 38.5 miles downstream). The greatest concentration of larval fish was located in the river margin, and the vast majority of larva, especially cyprinids, were collected from shallow, slow, nearshore habitats. Microhabitat use showed that catostomid larvae were most abundant in shallow habitats adjacent to the stream banks with vegetation. Many fluvial specialist species are sensitive to the effects of dams (e.g., Bain et al., 1988; Quinn and Kwak, 2003), and many of the species use shallow, slow water areas (Bain et al., 1988; Aadland, 1993) that disappear with frequent discharge changes. In a companion study (Kingsolving and Bain, 1993), abundance of juvenile and adult fishes was greater than 6.5 times higher in the river with the natural flow regime compared to the regulated river with altered hydrology. This study noted a riverine recovery gradient along the regulated river with fluvial specialist species increasing with distance downstream. This study noted much lower abundance of several native minnows in the regulated river, including *Campostoma oligolepis*, *Cyprinella spp., Notropis ammophilus, Notropis volucellus, Pimephales vigilax.* In contrast, more *Lepomis macrochirus* and *Gambusia affinis* were collected in the regulated river. Several generalist species had similar abundances among rivers, including *Lepomis megalotis*, Black Basses, and *Fundulus olivaceus*.

Taylor et al. (2014) found dams on Bird Creek and the Kiamichi River, Oklahoma, altered hydrology differently and the fish assemblages showed different responses to impoundment. Skiatook Lake on Bird Creek converted the system from an intermittent regime to a stable flow regime and effects on the fish assemblage were substantial. In contrast, Sardis Lake on the Kiamichi system had modest declines in maximum flows and this system experienced less assemblage change. Poff and Zimmerman (2010) also noted how responses of biotic communities to altered discharge can be variable depending on system.

Eley et al. (1981) compared the fish assemblage in the Mountain Fork before and after Broken Bow Dam was constructed and noted a dramatic decline in native cyprinid species. The total number of species declined in the 21-mile stretch below the dam from 84 to 65 species. This river has cold-water hypolimnetic hydropower releases so the flow impacts of this impoundment were likely confounded with temperature changes. However, Longear Sunfish and Green Sunfish were still abundant below the dam. The Ribbon Shiner (*Notropis fumeus*), Western Creek Chubsucker, Slough Darter, and River Carpsucker were widespread and common species pre-impoundment, but were not detected after dam construction. Downstream of the Dam, Spotted Bass and Largemouth Bass replaced Smallmouth Bass as the dominant sport fishes, and Brook Silverside and Plains Darter were relatively common.

Walburg et al. (1983) compared water quality, macroinvertebrates, and fish downstream of seven USACE Dams, including two flood control projects with warm-water releases (Pine Creek Dam, Oklahoma, and Gillham Dam, Arkansas), two flood control dams with cold-water releases (Barren River and Green River Dams, Kentucky), and three hydropower facilities with cold-water releases (Beaver, Hartwell, and Narrows Dams). Macroinvertebrates were sampled with drift and Hess samplers. Gillham Lake established a thermocline in June at a depth of 13 ft, and hypolimnion oxygen levels were less than 1 mg/L. During 1979, pH downstream was below 6 on 4 occasions with a low of 5.7. Iron (Fe) was occasionally greater than 1 mg/L during 1979, presumably due to its interaction with pH. Ammonia was recorded up to 1.10 mg/L during October 1979. Walburg et al. (1983) noted that iron concentration exceeded EPA criterion at that time (1 mg/L) during late summer at both Pine Creek and Gillham Dam tailwaters. Macroinvertebrate densities just downstream of Gillham Dam were dominated by Chironomidae and Oligochaetes. At station 2 (Mize Crossing), Chironomidae, Oligochaeta, Trichoptera, and Hydrocarina were dominant. At Station 3 (80,000 Rd), Ephemeroptera (mayflies), Chironomidae, Plecoptera (stoneflies), and Trichoptera (caddisflies) were dominant. This suggests that a riverine recovery gradient exists below Gillham Dam, and sites upstream are dominated by tolerant taxa and downstream sites have more Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa. The drift near the dam was dominated by reservoir taxa (Chaoboridae). Walburg (1983) reported sunfishes, suckers, and catfishes dominated the Gillham Dam tailwater, and high abundance of reservoir species that pass through the dam are often found near the dam, including Bluegill and White Crappie. Frietsche (1982) studied the recovery gradient at the Little Missouri River below Narrows Dam at sites 0.3 miles, 6.5 miles, and 10 miles from the dam. Fish catch per hour was highest (251.8 vs. 195 to 196) at the most downstream site. The implications of this report are that there is a recovery gradient below Gillham Dam that may extend close to the fall line where the Ouachita Mountains meet the West Gulf Coastal Plain (Figure 2).

Fox (2020) examined flow-ecology relationships for the Ozark-Ouachita Interior Highlands and the West Gulf Coastal Plain region of Arkansas, Missouri, and Oklahoma. He applied gradient forest modeling to fish presence/absence data to determine non-linear gradients and thresholds where changes in species turnover occurs. Spatial variables had the highest cumulative importance for all flow regimes, and a threshold was apparent at a change in dam storage of approximately 68 million gallons per square mile. He found timing of high flows was a critical metric. The frequency and timing of low flows was more important than magnitude and duration. Furthermore, Fox (2021) determined Leopard Darter occurrence has a negative correlation to variability in high flow pulse duration (DH16), a negative correlation to variability across annual maximum flows (MH18), and a negative correlation to variability of annual maximum of 7-day moving average flows (DH8). Fox (2021) also determined that Leopard Darter occurrence has a positive correlation to the spread in daily flows with the ratio of the difference between the 25th and 75th percentiles (MA11) and a positive correlation to variability in base flow (ML18).

Aadland (1993) noted instream flow studies should target fish species that live in habitats most sensitive to flow changes. He noted that most YOY fishes used shallow pools less than 2 feet deep with less than 1 ft/s velocity, and this habitat was greatly reduced during high flow events during rearing periods. Low and high flows may eliminate the fast riffle habitat required by some darters.

Well-developed riffle-pool structure appears to be important to Leopard Darter, Ouachita Mountain Shiner, Kiamichi Shiner, and Rocky Shiner. In highly diverse systems, the riffle-run-pool structure (or in large rivers, the intermittent shoal and sandbar features), combined with naturally variable hydrology leads to habitat heterogeneity that is critical to maintaining a diverse community structure. Flow alterations that lead to a homogenization of this structure, such as inflated minimum and deflated maximum flows, could be predicted to adversely affect these species. Life history traits indicating an adaptation to highly variable systems in Blackspot Shiner would suggest that alterations reducing flow variability may adversely affect this species. Aquatic vegetation is crucial to Western Starhead Topminnow, so any flow alterations that scour vegetation or lead to increased turbidity may impact this species. Additionally, Leopard Darters appear to be susceptible to increasing water temperatures and, particularly in the upper Cossatot, have become heavily reliant on pools as thermal refugia during summer months. Flow alterations leading to increased water temperatures could negatively impact this species. None of the aforementioned species are known to show nest-guarding behavior, which may make them more susceptible to decreases in minimum flow than species that do show this behavior (Carlisle et al., 2009).

Aquatic community metrics and ongoing data collection

Numerous metrics may be useful for monitoring aquatic communities' restoration success. However, it is very important that any metric developed is directly tied to restoration objectives, which should be developed early in any flow management process. Similarly, the flow management alternatives that are chosen should be hypothesized to meet the restoration objectives. Monitoring should not be done for monitoring's sake, but to evaluate the strength of an environmental flow prescription and the effectiveness of river management actions for achieving restoration objectives. In light of this, the measures and metrics chosen need to be relevant to restoration objectives or to potential changes in system state resulting from a change in flow management.

Similar to discharge, fish communities are highly variable and dynamic so detecting trends in short-term datasets can be difficult. AGFC is collecting more fish data for pre-post dam comparisons and to ascertain how the fishery is impaired, as well as to determine the status of imperiled species in the basin. Changes in a particular metric may be useful to determine if the aquatic communities are responding positively to changes in river management. For example, the reproductive success of freshwater mussels could be evaluated through annual recruitment, or if population stability is determined as a resource objective, the population needs to be examined over a long-time frame of at least 5-10 years. Additionally, as previously noted, the Leopard Darter's current habitat exists above Gillham Lake; biologic and hydrologic focus needs to be maintained on this limited habitat range. Long-term Leopard Darter monitoring data need to be analyzed to determine the effects of flow components and temperature on Leopard Darter occupancy and detection probability from long-term monitoring data.

Conclusion

As pointed out by Warner et al. (2014), an important aspect of implementing an environmental flows prescription is to make sure the flow volume and timing adjustments associated with these environmental flows are within the USACE's range of authorized reservoir releases while also finding solutions to benefit the Cossatot River ecosystem. By using the designer flows concept to develop a set of environmental flow prescriptions, we have the potential to support the aforementioned freshwater conservation goals by mitigating dam-related impacts, while supporting current project authorizations. This also gets around the fact that Gillham Dam is restricted to, at most, 3,000 cfs being discharged. For example, an environmental flows prescription could include timing high-flow releases to coincide with high water events to help supplement flows and to achieve a higher flow volume in the mainstem. Warner et al. (2014) referred to this as "episodic implementation"; "that is, implementation is driven by changing hydrologic conditions in the watershed—such as a large storm event or an extended period of drought—that in turn allow for or require changes in reservoir releases." Furthermore, the designer flows concept can also be considered when reregulating to downstream temperature requirements for sensitive species, such as the Leopard Darter.

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Appendix I

Cossatot SRP Identification of Partners, Stakeholders, and Issues of Concern

Brief meetings with natural resource partners and stakeholders were conducted in April and May 2020 to garner input related to environmental flows and water level management strategies to be considered for the Water Control Plan. On 7 May 2020, the USACE Cossatot SRP team met virtually with stakeholders from several State agencies to provide an overview of SRP, an objective to the project, background information about the study area, and to layout the expectations from the stakeholders. There were representatives from the Arkansas Game and Fish Commission, Arkansas Department of Environmental Quality, and The Nature Conservancy. A brief presentation was given and a question-and-answer session were held. On 20 May 2020 the USACE Cossatot SRP team met virtually with the Arkansas Natural Heritage Commission. Again, a brief presentation was given to provide an overview of SRP, an objective to the project, background information about the study area, and to layout the expectations from the stakeholder. Both virtual meetings provided insight into potential partner involvement, verified commitment, and discussed roles of each agency.

Since the initial virtual meetings, several ad hoc meetings were held with each, individual agency to further discuss expectations about developing a quantitative set of flow recommendations and deliverables for the State of the Science report. Ultimately, the Arkansas Game and Fish Commission and the Arkansas Natural Heritage Commission were the agencies that had the greatest interest and provided the most data and input into the Cossatot SRP project.

The main issues of concern identified during the virtual meetings include:

1. Restoration of environmental flows and dissolved oxygen below Gillham Dam to support Smallmouth Bass, mussels, threatened Leopard Darter, and the entire native fish community (approximately 72 species). The Cossatot River is the southernmost stream for the species within its range. Species of interest includes multiple Species of Greatest Conservation Need, including Kiamichi Shiner (*Notropis ortenburgeri*), Ouachita Mountain Shiner (*Lythrurus snelsoni*), Brown Bullhead (*Ameiurus nebulosus*), Goldstripe Darter (*Etheostoma parvipinne*), Blackspot Shiner (*Notropis atrocaudalis*) and Bluehead Shiner (*Pteronotropis hubbsi*). There are several SGCN taxa that are in the nearby Little River and/or Rolling Fork rivers that possibly could be in the Cossatot River, including the Rocky Shiner (*Notropis suttkusi*), Colorless Shiner (*Notropis perpallidus*), and Slenderhead Darter (*Percina phoxocephela*).

2. How may expected climate change impact reservoir operations? There are concerns that the frequency of large rains may be increasing, especially since 2010. These large rains can cause Gillham Lake level to back up almost to the Cossatot Falls, and this can inundate all the habitat for the threatened Leopard Darter. Arkansas Game and Fish Commission would like to see the duration of any flooding minimized at the Cossatot River State Park area, especially during spawning season (late March to May). The paddling community and the Cossatot State Park would probably support this too.

3. Leopard Darter have been possibly extirpated from the lower Cossatot River. If a natural flow regime is restored with suitable DO levels, then we could consider reintroduction efforts. Arkansas Game and Fish Commission assessed the status of Leopard Darter in the lower Cossatot River during 2019 and they appear to be extirpated.

4. There is interest in preventing algae blooms within Gillham Lake that can cause fish kills. These algal blooms seem to occur during flash droughts or extended periods without rain and with hot temperatures.

5. Gillham Lake likely has a Largemouth Bass and Spotted Bass fishery, and downstream and upstream needs will need to be balanced with this fishery

6. The Cossatot River seems to be thermally impaired upstream of Gillham Lake although the watershed is 98 percent forested. It is not recognized as being impaired by Arkansas Department of Environmental Quality, but summer temperatures have been recorded to reach 95 to 98.6° F.

7. There is also interest in the possible effects of low pH, although national trends seem to be improving.