

Distribution and Population Demographics of Invasive Bigheaded Carp in the Lower Red River Basin



2024 ANNUAL TECHNICAL REPORT

October 1, 2020 through December 31, 2024

Project Title:

Distribution and Population Demographics of Invasive Bigheaded Carp in the Lower Red River Basin

Geographic Location:

Lower Red River Basin in Oklahoma, Texas, and Arkansas

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Funding Acknowledgments:

Funding for this project was provided by the USFWS Fish and Wildlife Management Assistance grants program and the ODWC State Wildlife Grants program.

Cover Photo Credit:

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Statement of Need:

Freshwater ecosystems are among the most biodiverse systems on earth; however, they may also be the most endangered (Reid et al. 2019). Despite covering only 2.3% of the Earth's surface, freshwater ecosystems account for 9.5% (126,000 species) of described animal species (Balian et al. 2008). Dudgeon et al. (2006) lists over-exploitation, flow modification, water pollution, habitat-degradation, and invasive species as the five major threats to biodiversity. Invasive species, or introduced non-native species that are able to survive to recruitment, reproduce across a variety of habitats, and expand their ranges to locations outside of where they were first introduced are of particular concern (Blackburn et al. 2011). Invasive species are of concern because they alter food web interactions, compete with other species for space and resources, and can ultimately change native species assemblage structure (Carey and Wahl 2010). As such, there is a need to understand population demographics of invasive species and the spatial and temporal extent to which they occur.

Two species emblematic of the concerns caused by invasive species are Bighead Carp *Hypophthalmichthys nobilis* and Silver Carp *Hypophthalmichthys molitrix* (hereafter, bigheaded carp). Bigheaded carp pose significant ecological (Schrank et al. 2003; Irons et al. 2007; Sampson et al. 2009), economic (Lovell et al. 2006), and safety (Vetter et al. 2015) threats in freshwaters around the globe, including in the Mississippi River Basin. These planktivores were introduced into the United States in the 1970s to help control nuisance algae in aquaculture ponds. During seasonal flooding, it is thought that these species escaped from ponds into the Mississippi River. Since their detection in the 1970's (Freeze and Henderson 1982; Kelly et al. 2011), bigheaded carp have proliferated and been reported in 23 states (Kolar et al. 2005). Currently, self-sustaining populations now exist in several large tributaries of the Mississippi River (e.g., Tucker et al. 1996; Fuller et al. 1999). In some cases, local populations of bigheaded carp increased exponentially (Chick and Pegg 2001; Irons et al. 2007; Sass et al. 2014), indicating their populations have great potential to rapidly increase in a short period of time.

One of the reasons bigheaded carp have been so successful is because they are filter feeders (Williamson and Garvey 2005), and both species have been linked to declines in phytoplankton and zooplankton abundances (Irons et al. 2011; Sass et al. 2014; Cooke 2016). Bigheaded carp affect fish populations through interspecific competition and depletion of

resources (Schrang et al. 2003; Sampson et al. 2009). As a result, Bigheaded carp are often linked to declines in native fish diversity and densities (Kolar et al. 2007) including the recruitment of native juvenile fishes (Chick et al. 2020b). There is concern that the continued spread of planktivorous bigheaded carps will result in the continued degradation of aquatic food webs due to their ability to efficiently consume and alter riverine planktonic communities (Xie and Yang 2000; Lu et al. 2002; Sass et al. 2014; Collins and Wahl 2017; Collins and Wahl 2018), which are important food resources for larval, juvenile, and adult native fishes (Fletcher et al. 2019). Recent evidence suggests that the reduction in plankton by bigheaded carp may negatively affect native fish communities where bigheaded carp populations become established (Irons et al. 2007; Solomon et al. 2016; Pendleton et al. 2017).

In addition to their ecological effects, bigheaded carp also cause economic declines. For example, the bigheaded carp invasion in Lake Michigan is projected to result in a 7 billion dollar loss via commercial fisheries revenue (Buck et al. 2010). Lastly, bigheaded carp pose threats to human safety due their penchant to launch themselves out of the water often causing serious injuries to boaters (Spacapan et al. 2016). The detection, capture, and removal of bigheaded carp from river ecosystems has been a management priority over the last decade in the Mississippi River basin (Collins et al. 2015, 2017; ACRCC 2016; Rodgers 2019).

Invasive bigheaded carp research activities have primarily focused on large floodplain rivers of the upper Mississippi River basin (e.g., Illinois, Mississippi, and Missouri rivers) where substantial advances in understanding their ecology have been achieved. However, substantially less is known about bigheaded carp populations in tributaries of the lower Mississippi River Basin where they have been studied less frequently (Chapman and Hoff 2011; Ochs et al. 2019). The presence of self-sustaining populations has been noted across the lower Mississippi River basin for a while (Tucker et al. 1996, Fuller et al. 1999, Thomas et al. 2011, Rodgers 2019) and sampling and landings data suggest their prevalence is increasing in the Lower Red River basin (TPWD, ODWC, AGFC, unpublished data). However, there is a general lack of information regarding the population dynamics of invasive bigheaded carps and their effects on native fish communities of the lower Mississippi River basin, especially the Arkansas-Red-White subbasin.

The climate of the Great Plains ecoregion is relatively extreme, fluctuating between floods and droughts; thus, providing a unique opportunity to study species assemblage structure and population dynamics of both native and invasive fishes. The Red River basin is characterized

by extreme floods and droughts (Matthews and Marsh-Matthews 2007), and large conductivity fluctuations (Hargrave and Taylor 2010). Bigheaded carp occur in the lower Red River basin; however, there has never been extensive sampling targeting bigheaded carp or many native fishes. Therefore, documenting bigheaded carp population demographics and distribution along with native fishes is a necessary first step to determining how best to manage the expansion of non-native fishes in the lower basin (i.e., how far have they spread, at what times of year, and what species may be affected).

Understanding occupancy by bigheaded carps in the basin, both spatially and temporally, is vital for directing management actions. There also exists a great need to understand the trajectory of the bigheaded carps' invasion to predict their influences on native fish assemblages within these large tributary basins and associated reservoirs and to collect baseline data on native fish assemblages to facilitate future assessments of impacts. Furthermore, there exists a need for a better understanding of the movements of bigheaded carps to enhance monitoring, guide any potential future control efforts, and better understand potential spawning movements.

The objectives of this project, as outlined in this document, are aligned with and support the goals and objectives of the Lower Mississippi River Asian Carp Control Strategy Framework, particularly Goal 2 – Monitoring and Population Status and Goal 4 – Understanding Impacts and Research. This project has initiated a surveillance effort for bigheaded carp at a broad geographic scale across the Lower Red River Basin as well as establishing baseline data for native fish assemblages potentially impacted by bigheaded carp needed for future evaluations of deleterious impacts. This project addresses the general lack of knowledge of bigheaded carp population dynamics and native fish assemblages in this sub-basin. The intentions of this effort are to aid in early detection, assess population distribution and status, and facilitate future evaluations of deleterious impacts to native fishes. Data on bigheaded carp populations would also inform potential removal efforts. Collaborative efforts of multiple partners and agencies (state, federal, and university) will be implemented to accomplish the project goals and objectives. This project will provide an ongoing, coordinated effort to evaluate bigheaded carp distribution and status in the Lower Red River Basin that will contribute to a better understanding of the status of this species in the Mississippi River Basin as a whole.

Objectives:

1. Determine the spatial and temporal distribution and adult population demographics of invasive bigheaded carp (Silver Carp and Bighead Carp)
2. Establish baseline native and invasive fish assemblage and habitat association data
3. Determine movement patterns of bigheaded carp in the Red River using telemetry

Highlights:

- In spring 2021, we began a fisheries assessment investigating the current population status of Bighead Carp and Silver Carp in the lower Red River catchment, while compiling information on native fishes within the basin. Our study objectives were to: 1) conduct an invasive carp population assessment, 2) conduct a native species assemblage assessment, and 3) beginning in 2023, conduct a movement/telemetry assessment of non-native Bighead Carp and Silver Carp.
- Bigheaded carp were observed or captured across the Red River catchment using a variety of gears. A total of 413 bigheaded carp were collected from May 2021 through November 2023. We captured 305 Silver Carp (180 males, 116 females, 9 unsexed, 1.5:1.0 sex ratio) and 108 Bighead Carp (67 males, 32 females, 9 unsexed, 2.1:1.0 sex ratio) throughout the lower Red River catchment (includes USFWS Carp donated by bow fisherpersons). Most bigheaded carp captured in the mainstem Red River were sampled from reaches with backwater habitat. Bighead Carp and Silver Carp in the Red River catchment appear to live longer and grow larger than other populations. Silver Carp and Bighead Carp in the lower Red River had a theoretical maximum length of 920 and 1,348-mm TL, respectively. The oldest sampled Silver Carp and Bighead Carp were age 14 and 17, respectively.
- Bigheaded carp were captured in the Sulphur River and adjacent waters only in 2022 and 2023. In 2022, one Silver Carp was collected in the Sulphur River just upstream of the confluence and another nearby in the Red River near a wing dam structure. In 2023, six

additional Silver Carp were captured at this Red River site. No Bighead Carp were collected and no Silver Carp were detected farther upstream in the Sulphur River.

- Bighead and Silver Carp growth were positively associated with warmer air temperatures and negatively associated with discharge variability. However, Silver Carp growth was also positively related to high discharge conditions and the variability of air temperature. Silver Carp annual mortality was relatively low and recruitment into the population appeared steady. It appears that bigheaded carp are likely coming from another catchment, have only limited or periodic successful reproduction in the study area, or spawn downriver in Louisiana.
- No juvenile bigheaded carp or individuals under the age of three have been collected during the project, although females of both Bighead and Silver carp were observed with ovaries occupying much of the body cavity and containing developed eggs. The average monthly GSI was highest in May for Bighead Carp and in June for Silver Carp. A number of Silver Carp and Bighead Carp sampled in May 2024 contained post-ovulatory follicles indicating at least one spawning event had occurred relatively recently prior to capture. These data reflect spawning is occurring within the basin.
- We used data associated with the entire catchment to model the occupancy of adult and juvenile fishes including both bigheaded carp species. Bigheaded carp tended to occupy reaches with the presence of slackwater habitat, that were deeper and narrower (lower habitat complexity), with higher discharge conditions, and Silver Carp were positively associated with chlorophyll-a concentrations.
- In the lower Red River catchment, a total of 86 species and 48 genera of fishes were identified during sampling. Of the 86 fish species, four were non-native including Bighead Carp, Silver Carp, Common Carp, and Grass Carp. The most abundant small-bodied species was Red Shiner, followed by Bullhead Minnow, Western Mosquitofish, Threadfin Shad, Chub Shiner, and Mississippi Silverside. The most abundant large-bodied species was River Carpsucker, followed by Smallmouth Buffalo, Bigmouth Buffalo, Freshwater Drum, Longnose Gar, and Black Buffalo. In the Sulphur River catchment, a total of 52 fish species from 17 families was documented throughout the project period. Of the 52 species, three were non-native including Silver Carp, Common Carp, and Grass Carp. The Sulphur River fish community was primarily comprised of

Dorosomatidae, Centrarchidae, Catostomidae, and Lepisosteidae. Log-transformed length-weight relationships were calculated for the most prevalent large-bodied species.

- We tagged 25 Bighead Carp and 25 Silver Carp at multiple locations within the lower Red River catchment with acoustic transmitters and conducted passive and active tracking. We detected 41 of our 50 tagged fish on 221 occasions using active tracking and 1,413 detections via our SUR array; the greatest number of detections came from our SURs located in oxbows and tributaries. Some of the tagged fish have made extensive movements (e.g., 336.5 km), whereas others spend long periods of time in the same general reach. Bighead Carp had a mean movement distance of 2.37 km (SD: 18.05, range: 0-336.57) and mean movement rate of 0.46 km/d (SD: 1.8, range: 0-37.89). Silver Carp had a mean movement distance of 1.01 km (SD: 9.61, range: 0-206.99) and mean movement rate of 0.36 km/d (SD: .95, range: 0-18.02). Both species moved at greater distances at higher CV of discharge values. There was an interactive effect between water temperature variability and mean discharge that showed varying effects of discharge at different water temperatures variability. During low and medium discharge conditions, water temperatures variability had a minimal effect on movement distances. During high discharges, movement distances increased at greater water temperature variability values.
- With the introduction of invasive Bighead Carp and Silver Carp in the catchment, concerns over changes in occupancy or condition of native fishes may be warranted (Schrunk et al. 2003). In other catchments, there is evidence that changes to the fish assemblage occur as densities of bigheaded carp increase (Carey and Wahl 2010; DeBoer et al. 2018). Having baseline data on the assemblage of juvenile and large-bodied native fishes will be important for monitoring changes in these populations over time and evaluating future management actions.
- Moreover, if the goal is to greatly reduce or eliminate bigheaded carp, then strategies that prevent further immigration before reproduction occurs or becomes more successful would be ideal. Targeted removal may then be useful for reducing numbers already in the catchment; however, there are also oxbow lakes that contain bigheaded carp but appear only connected to the river during high flows (i.e., possible source locations).

Methods – Red River

Objective 1. Invasive bigheaded carp population assessment

Juvenile bigheaded carp sampling

Although some juvenile sampling occurred in the winter, our occupancy modeling design included sampling conducted during a period that we defined as the juvenile warm-water season from May through early October. Our sample season was chosen to detect both juvenile fishes of bigheaded carp, if present, and to meet the closure assumption (i.e., if a species was detected, it was assumed present at that site for the duration of the season). Our sample reaches (hereafter sites) were distributed across the mainstem Red River (Appendix A, Figure 1) and were designed to target juvenile bigheaded carps. Sites were selected based on river access, proximity to U.S. Geological Survey (USGS) stream gages, and the likelihood of detection of the target species. Our sites were selected approximately 25-100 km downriver of major dams and confluences because this is the suggested length of river needed to allow bigheaded carp eggs to develop and hatch while in suspension (Kolar et al. 2007; Garcia et al. 2015). Our sample sites included slackwater habitats such as forewaters, backwaters, side channels, sandbars, and pool complexes. Slackwater habitats are thought to be important nursery areas for a variety of juvenile fishes including Bighead Carp and Silver Carp (Jurajda 1999; Love et al. 2017; George et al. 2018). Furthermore, discharge and temperature conditions are similar across these areas, and the areas are large enough to be considered closed to species immigrations (i.e., not individuals) during our sampling periods.

Our goal was to sample age-0 bigheaded carps using three different gear types during daylight hours. Using a combination of gears diminishes some of the sample bias associated with a single gear approach (Clark et al. 2007). For example, passive gears tend to target more active individuals (Fago 1998). At each site, we set mini-fyke nets, sampled using beach seines, and conducted larval tows (see Appendix A, Table 1 for gear descriptions). First, we set 3 mini-fyke nets in <2 m of water at locations adjacent to the shoreline to target small-bodied fishes (Eggleton et al. 2010). Mini-fyke nets are commonly used to sample age-0 bigheaded carps (Wanner and Klumb 2009; Gibson-Reinemer et al. 2017; Williams 2020) and sometimes capture high numbers compared to other gears (Collins et al. 2017). Next, a beach seine was used to

sample wadeable habitat across the reach using a modified version of the encirclement technique (Bayley and Herendeen 2000). Transects were established throughout wadeable habitat at each site and seine hauls were completed across each transect. Seine hauls were limited to 25 m to maintain the efficiency of the gear because longer hauls are less efficient (Lombardi et al. 2014). We quantified total seine distance, seine width, and maximum depth for each haul to estimate the area sampled. We completed a sub-surface larval tow at representative locations of deeper water (i.e., where we could not seine or place fyke nets). Each tow was pulled for 10 min and the volume of water sampled was quantified using a flow meter (General Oceanics Mechanical Flowmeter Model 2030R) attached to the mouth of the net. We standardized larval tows based on the volume of water filtered by the net. Any samples that could not be identified in the field were preserved in 70% ethanol and brought back to the lab for processing.

Juvenile bigheaded carp habitat

We quantified the physicochemical factors that may be related to bigheaded carp or native fish distributions across multiple spatial scales (i.e., reach, segment, and catchment). The physicochemical factors are divided into detection (Appendix A, Table 2) and occupancy (Appendix A, Table 3) covariates. Stream habitat use by fishes is hierarchical where finer levels of organization are nested within coarser landscape constraints (Frissell et al. 1986; Imhof et al. 1996). Coarse scale (e.g., segment and catchment) habitat factors are applied to multiple reaches that occur within the same stream segment or catchment (i.e., nested). For example, finer-scale channel unit conditions (i.e., pH and substrate) used by fish are often influenced by coarse factors (i.e., drainage area and geology) of the surrounding watershed (Mollenhauer et al. 2019). Including coarse-scale habitat factors helps explain fish distributions and account for pseudoreplication inherent in the nested structure of sampling riverine sites (i.e., sites closer in proximity are naturally more similar than sites further away).

We measured several factors across each sample site that described the general water-quality conditions. First, we collected temperature and dissolved oxygen samples at 0.5 m below the water's surface for each site using a multi-parameter water-quality meter (YSI ProDSS). We collected salinity from a well-mixed location of each site approximately 0.5m below the surface. We also measured water clarity using a Secchi disk, because turbidity can influence resource use,

foraging success, and even provide shelter from predators (Zamor and Grossman 2007; Reichert et al. 2010). To characterize the general conditions of each site, we measured all water-quality parameters three times in each site and averaged these values.

We also quantified the proportion of select channel unit features in each site. Because forewater and backwater habitat are often important nursery habitat for many large river fishes (Galat et al. 2004), we quantified the area of each using a meter tape or rangefinder (Simmons Volt 600 Laser Rangefinder) to measure length and average width. Other slackwater areas such as pools offer low-velocity areas in the main channel (Schwartz and Herricks 2005); therefore, we measured pool area using side-scan sonar (Humminbird Helix 12). The proportion of each of the slackwater channel units was expressed as a proportion of the available habitat in each site. Because age-0 bigheaded carp are associated with large woody debris in some systems (George et al. 2018), we also used side-scan sonar to quantify the percentage of large woody debris following the methods of Gordon et al. (1992).

We quantified several hydraulic variables to describe the fluvial dynamics of our sampling sites. Species often use specific depths within a water column (Lamouroux et al. 1998); therefore, we quantified the average thalweg depth by measuring depth at 10-m increments along the thalweg of the site using side-scan sonar. Further, because the shape of the channel dictates habitat availability (Thomson et al. 2001), we quantified width to depth ratios in each site. We measured three representative wetted width measurements using a rangefinder. The average thalweg depth of the site was then divided by the average widths. We also obtained discharge data from the nearest USGS stream gages to apply to sampling sites within the same stream segments to examine both detection and occupancy.

Some habitat metrics were quantified using existing geospatial data. At the reach-scale, we quantified distance to the nearest dam by measuring the distance from the most downstream point of our sites to the nearest upstream using National Hydrology Dataset (NHDplus) flowlines and ArcMap spatial analyst. We also measured the distance from our sites and the nearest upstream 5th-order tributary. Areas below dams and major tributary confluences are potential spawning locations for bigheaded carp species (Kolar et al. 2007; George et al. 2018; Camacho et al. 2020).

At the stream segment scale, we calculated stream sinuosity and slope. Sinuosity (i.e., channel migration of meandering rivers) affects fish habitat use including choice of spawning location (Fukushima 2001; Lazarus and Constantine 2013) and was calculated by dividing the thalweg length by the straight line distance of the segment. (Camana et al. 2016). We calculated river slope using ArcMap spatial analysis to determine the change in elevation between the upstream and downstream points of each stream segment and divide by the thalweg length (i.e., channel distance measured down the middle of the channel, Bain and Stevenson 1999).

We also measured several habitat variables that may affect fish distributions at the catchment scale. We measured the drainage area (km²) upstream of each site (i.e., catchment draining to each site) using NHDplus flow lines to determine the size and relative position of sites within the network. Because catchment lithology controls many local physicochemical conditions (Frissell et al. 1986; Stevenson 1997), we quantified the dominant lithology (e.g., limestone) upstream of each site. We also quantified landscape disturbance (hereafter LDI) following Brown and Vivas (2005) using the 2021 National Land Cover Dataset (further NLCD; Dewitz 2023) and a modification of Mouser et al. (2019) (see below). Human land-use modifications can disproportionately affect the quality and quantity of riverine nursery habitat (Schlosser 1995; Rochette et al. 2010; Britton and Pegg 2011). However, land-cover types tend to be multicollinear because they sum to 100% (Ainiyah et al. 2016); thus, combining land cover into a single index is helpful when analyzing data using multiple regression scenarios (Genovese et al. 2001). Therefore, we characterized the level of LDI following a modification of Brown and Vivas (2005) provided by Mouser et al. (2019).

Adult bigheaded carp sampling

Although sampling was conducted year-round, our data collected during the cold-water season (October through March) were insufficient for occupancy modeling due to the limited number of repeated surveys as low water levels made access extremely difficult. As such, our occupancy modeling for adult bigheaded carp included only data from surveys conducted during the adult warm-water season (April through September of 2021 and 2022). Thus, for bigheaded carp occupancy modeling we included data collected from 43 unique reaches, comprising 137 surveys. Each reach was approximately 1.5 to 2.0 river km (rkm) (hereafter sites) and sampled 1-3 times. Access can be problematic on the Red River and thus, sites were selected based on

accessibility (i.e., access to private lands and conditions conducive to boat launching) (Appendix A, Figure 2).

We sampled fishes using a combination of gillnets, hoop nets, and electrofishing because they have been shown useful for sampling both Bighead and Silver Carps in perceived low-density environments (Norman and Whitley 2015; Butler et al. 2019). Three experimental sinking gillnets (54.8-m long for mainstem and 30.5-m long for tributary sampling with 8.9, 10.16, and 10.8-cm bar-length mesh panels) and three hoop nets (4.88-m long with a 1.2-m diameter opening) were placed throughout each site (Appendix A, Table 1). Gillnets were deployed perpendicular to the shoreline with one placed near each end of the reach and the third net placed in the middle of the reach at the narrowest portion of the channel to restrict bigheaded carp movement. Hoop nets were placed parallel to the shoreline with the opening facing downstream in locations that included channel edges and channel crossovers but lacking extensive woody debris. After net placement, we electrofished using an 80-amp Midwest Lakes Electrofishing Systems shocking unit (Polo, Missouri). We used standard AFS electrofishing settings based on conductivity (though we tried several others- see below). Water conductivity in the tributaries was much lower than the mainstem Red River. As such, voltage was set to high range (pulsed DC current, >300 volts, 60Hz) for tributaries and low range (pulsed DC current, <300 volts, 60Hz) for the main stem Red River sites. Beginning at the upriver end of the site, the boat traversed downstream in a cloverleaf pattern with electrical current applied for 10-sec with 5-sec “off peddle” intervals to increase the effectiveness of capturing Silver Carp and to attempt to drive fish into the nets and shoreline (Bouska et al. 2017). Electrofishing continued until the entirety of the reach was sampled.

Before we established our electrofishing protocol, we used several electrofishing settings at sites where bigheaded carp were observed on previous occasions. During experimental electrofishing trials, we used pulsed DC current at both low and high frequencies, with Hz ranging from 15 to 60 and a target amperage of 4 and 20, respectively. Boat electrofishing was also used to drive bigheaded carp into set nets. Both gillnets and hoop nets were then removed after six hours post-placement. All bigheaded carp collected during our sampling events were euthanized. Total length (mm, +/- 1 mm), and weight (g, +/- 10 g), were recorded for captured bigheaded carp, except for a few captured while our scale was malfunctioning.

Additionally, Bois d'Arc Lake was also sampled for Silver Carp and Bighead Carp via electrofishing and gillnets in June 2021, March 2022, October 2022, and February 2023. During the June and March sampling, three 55 m long gillnets were set in the reservoir and left to soak for 6 h. While the nets soaked, we conducted 30 electrofishing transects (20 in June in conjunction with Texas Parks and Wildlife, 10 in March) that were each 15 minutes long. During the October and February sampling, three sampling locations were chosen to cover a variety of habitats and included one site at the bottom of the reservoir (near the dam), one site in the middle of the reservoir, and one at the top of the reservoir (near the creek mouth). The bottom and middle sites were approximately 1 km long. We set three 55 m long gillnets strategically throughout the bottom and middle sites and attempted to use the boat electrofisher to drive and corral any potential Silver Carp or Bighead Carp to the nets. We spent one hour electrofishing around the nets at each site. After electrofishing, the gillnets were pulled. The top site near the creek mouth was too shallow and contained too much woody debris to set gillnets. At this site, we electrofished for approximately one hour in a zigzag pattern attempting to drive any potential bigheaded carp to the banks.

Adult bigheaded carp habitat

We quantified the physicochemical factors that may be related to bigheaded carp distributions across multiple spatial scales. We quantified habitat factors at the catchment, segment, and reach scales. The habitat factors were either collected in the field or obtained using existing geospatial data (Appendix A, Table 4). We assessed habitat use using an occupancy modeling framework. Our warm-water season was defined as April through September where we could reasonably assume each site (sampling reach, defined as a 1.5 – 2.0 rkm section) was closed to changes in Silver Carp or Bighead Carp occupancy (i.e., if the species was present, then it was assumed present for the season, though individuals may move back and forth from the site) (Mackenzie et al. 2005). We defined the season using the species' biology and associated water temperature. Silver Carp remain relatively stationary during the summer months (Coulter et al. 2016a) and are hypothesized to spawn at water temperatures above 18 °C (Nico et al. 2022). Therefore, we established the season as April through September based on historical water temperature trends (Appendix A, Figure 3). We conducted repeated fish surveys using multiple gears where our surveys were temporally replicated over the warm season of 2021 – 2023.

The habitat factors operating at the catchment scale that may be related to bigheaded carp occurrence were drainage area, disturbance, and lithology (Appendix A, Table 4). Drainage area (km^2) is a coarse scale habitat factor that influences fish distributions, assemblage structure, and species richness (Newall and Magnuson 1999; Osborne and Wiley 1992; Griffiths 2018). We used the National Hydrography Database Plus (NHDplus) (<https://apps.nationalmap.gov/downloader/#/>) flow lines in ArcGIS Pro (version 3.0.1, Esri, Redlands, CA) to delineate each catchment (i.e., the entire upstream area that drains to the site) using the watershed tool and quantified the area of each catchment. Disturbance can affect assemblage structure and distribution by altering nutrient flow and habitat availability, and lead to decreased diversity throughout multiple trophic levels (Scrimgeour et al. 2008; Wang et al. 2008; Johnson and Angeler 2014). We used ArcGIS Pro to quantify the area of each land use type in each catchment using the National Land Cover Database (NLCD) and previously calculated drainage areas. Each land type was assigned the corresponding disturbance value from the Landscape Development Index (LDI) (Brown and Vivas 2005). However, in instances where the land-cover type applied to multiple LDI coefficients (e.g., multiple types of agriculture land), we calculated the average of the relative LDI coefficients. We multiplied the proportion of each land type in the catchment by the assigned LDI value to quantify the overall disturbance factor for each land type. We then summed the coefficients of the disturbance factors within each catchment to characterize the disturbance level for the catchment. For example, if a catchment was 50% woodland pasture and 50% row crop then the pastureland was assigned an LDI coefficient of 2.02 and the row crop was assigned an LDI coefficient of 4.45 resulting in an overall disturbance factor of 3.23. Lastly, lithology is related to sedimentation, pH, and controls the macro and micronutrient cycling load within a catchment (Sarkar et al. 2007; Zeng et al. 2011; McDowell et al. 2013; Glaus et al. 2019). Sandstone contains high quantities of silica, which leads to predominately neutral or slightly acidic environments because soluble silica forms orthosilicate acid (Worden and Morad 2000; Belton et al. 2012). Catchments with lower percentages of sandstone will likely have higher pH than those with higher percentages of sandstone. We quantified the percentage of sandstone for the drainage area of the catchment using the U.S. Geological Survey's (USGS) National Geologic Map Database (<https://mrdata.usgs.gov/geology/state/>) and the identify tool in ArcGIS Pro.

Habitat factors operating at the segment scale that may be related to bigheaded carp occurrence were sinuosity, slope, and discharge (Appendix A, Table 4). Segments were classified by 5th-order tributary confluences. Stream sinuosity, the ratio of the straight-line segment of the river to the channel distance (Rowe et al. 2009), is associated with habitat complexity (e.g., woody debris, canopy cover) and floodplain connection (Nagayama and Nakamura 2018). Sinuous reaches in a river are important for certain species reproduction (e.g., Sakhalin Taimen *Hucho perryi*; Fukushima 2001), and bigheaded carp in the Missouri River spawned larger quantities of eggs in more sinuous river segments (Deters et al. 2013). Sinuosity was calculated by dividing the river kilometer (rkm) distance by the straight-line distance of the segment using the distance tool in ArcGIS Pro. Slope can affect species distributions by influencing water velocity, channel morphology, and substrate, which are often correlated with the stream gradient (Camana et al. 2016). Stream gradient may alter the availability of low-velocity habitat associated with bigheaded carp presence. We quantified slope using spatial analysis in ArcGIS Pro by dividing the change in elevation from the upstream to downstream end of the segment by the segment length (rkm). Lastly, discharge (m³/s) affects fish density and occurrence, habitat associations, recruitment success, and can be altered for mitigation purposes (Valdez et al. 2001; Work et al. 2017; Love et al. 2017; Bašić et al. 2018). Silver Carp in the Illinois River were positively associated with discharge but avoided main channel habitats during high discharge (Coulter et al. 2017). We obtained discharge data from the USGS streamgage of the segment or from Stream Stats (<https://streamstats.usgs.gov/ss/>) in instances where USGS streamgages were not available. We calculated the median discharge during the season (i.e., occupancy) and divided by the drainage area of the segment to standardize discharge across rivers for comparability (i.e., Red River, Kiamichi, Blue River, etc.).

At the reach scale, we hypothesized that distance to the nearest upstream dam, percent backwater, width-to-depth ratio, salinity, and chlorophyll-*a* were related to bigheaded carp presence. Dam construction changes both biotic and abiotic riverine attributes (Catalano et al. 2007). For example, flow alteration caused by dam construction in the Yangtze River has led to reduced recruitment for both Bighead Carp and Silver Carp (Duan et al. 2009). Bighead Carp and Silver Carp are thought to require an estimated 100 km of free-flowing river to successfully spawn (Kolar et al. 2007). We used NHDplus flowlines and ArcPro GIS spatial analyst to quantify the distance from the downstream end of each site to the nearest upstream dam.

Backwaters (i.e., a specific slackwater type) are off channel, relatively shallow, low-velocity areas, relative to the main flow thread within the channel (Vietz et al. 2013). These locations are often used as a refuge by juvenile fishes due to forage availability and growth potential (Humphries et al. 2006). Backwater habitats are also used by adult bigheaded carp as refuge areas during higher discharge conditions (Coulter et al. 2017; MacNamara et al. 2018) and may offer higher forage potential (Williamson and Garvey 2005). We calculated the percent backwater for the reach by measuring the channel width and length within each backwater using a handheld rangefinder (Simmons VLRF 600, Overland Park, KS, +/- 1 m), and then expressed backwater area as a percent of the total reach area. Width-to-depth ratios describe the general structure of a stream channel where increasing ratios describe wider and shallower channels (Gordon et al. 1992; Dunham et al. 2002). We collected 3 channel width measurements with a handheld rangefinder and three corresponding channel depths with a boat equipped depth finder (Humminbird Helix 10, Rane, WI) at three locations of each reach to determine a mean reach ratio. Fishes have different salinity tolerances and will use habitat within their salinity tolerances over appropriate dissolved oxygen and temperature conditions (e.g., Shortnose Sturgeon *Acipenser brevirostrum*; Farrae et al. 2014). Inappropriate salinity environments can hinder reproduction and in extreme instances lead to poor osmoregulation and eventual death (Oto et al. 2017; Neves et al. 2019). We collected three salinity measurements (ppt) at the upper, middle, and bottom portions of each reach using a Yellow Springs Instrument (YSI pro dds, Yellow Springs, Ohio). Chlorophyll-*a* (chl-*a*) concentration is widely used as a surrogate for productivity and algal biomass (Pinder et. al 1997). Bigheaded carp are omnivores, consuming both zooplankton and phytoplankton (Calkins et al. 2012) and may be associated with varying chl-*a* densities in the catchment. A water sample was collected using an integrating tube sampler to sample the top 2-m of the water column at the most downstream end of the reach (Raikow et. al. 2004). The water was stored in containers and transferred to the laboratory. Within 24 h of water collection, three 250-mL subsamples were placed into a 47-mm diameter filter tower (PALL, Port Washington, New York) and filtered through a 1- μ m glass fiber filter (PALL, Port Washington, New York). The filter was then placed into a light-proof container and frozen for later laboratory analysis. In the laboratory, chl-*a* was extracted from the filters using 90% ethanol, filtered a second time, then estimated using a Trilogy Laboratory Fluorometer (Turner Designs, San Jose, California) (Sartory and Grobbelaar 1984).

At the reach scale, we quantified water temperature, turbidity, discharge, and sampling effort to relate to bigheaded carp detection (Appendix A, Table 4). Sullivan et al. (2017) found that increased catchability of Silver Carp occurred at higher water temperatures during the summer months (e.g., July and August) in the Des Moines River, Iowa. We measured water temperature (°C) at a well-mixed location of the upper, middle, and bottom portions of the reach using a YSI and calculated the mean during the survey to relate water temperature to bigheaded carp detection. Turbidity can affect the visual and chemical acuity of fishes thereby reducing growth and recruitment because of reduced foraging or successful spawning (Järvenpää et al. 2019; Korman et al. 2021). Turbidity also affects detection (Figueroa-Pico et al. 2020; Bunnell et al. 2021). We collected three visibility measurements (i.e., Secchi depth, +/- 1 cm) as a surrogate for turbidity at the upper, middle, and bottom portions of the reach. Discharge can affect the detection of fishes. For example, Zentner et al. (2021) found that detection of sucker species with passive integrated transponders (PIT) in streams was negatively associated with increasing discharge. We obtained discharge data from the nearest USGS stream gage and calculated the mean discharge for the day of each survey and standardized by the drainage area of the segment to compare discharge across rivers (i.e., Red River, Kiamichi, Blue River, etc.). In instances where USGS streamgages were not available, we used the median discharge value of the segment for the month in which the survey occurred using Stream Stats (U.S. Geological Survey, 2019, The StreamStats program, online at <https://streamstats.usgs.gov/ss/>, accessed on April 10, 2023). Sampling effort can affect the detection of fishes (Reid and Haxton 2017), so we calculated the electrofishing effort (i.e., seconds) for the survey.

Adult Bigheaded Carp data analyses

An occupancy model (OM) is useful for delineating factors related to occupancy probabilities while accounting for incomplete gear detection (Mackenzie et al. 2002). The four assumptions of an occupancy model are: 1) the occupancy state must be “closed” (i.e., to the species and not individuals), 2) there is no unexplained heterogeneity in detection, 3) there is no unexplained heterogeneity in occupancy, and 4) the sites are independent of each other (Bailey and Adams 2005). We met the assumption of species’ closure by establishing a season (April – September) and by having a sufficient reach size (1-2 rkm). The second and third OM assumptions were met with the inclusion of both detection and occupancy covariates to explain variation in detection or

occupancy probabilities (Mackenzie et al. 2002). We met the final assumption by spacing our sites at least 1.5-2 rkm apart so surveying one site did not influence detection at an adjacent site. Determining detection probability is essential because it affects our ability to infer occupancy (Benoit et al. 2018). Estimates of detection account for potential species presence at a site even if the sites were not sampled (i.e., false absence, Royle and Kéry 2007; Kéry et al. 2010). We quantified the probability of detection using temporally replicated surveys during our warm-water season (Mackenzie et al. 2002). The detection history (i.e., 1 if present, and 0 if absent) was modeled with covariates using a logit function to explain heterogeneity of detection because detection covariates varied across surveys (Mackenzie et al. 2002). Probability of detection was then used to estimate the probability of occupancy. The relationship between detection probability and occupancy was modeled as two Bernoulli distributions. Occupancy was modeled using covariates hypothesized to be related to species presence to explain the heterogeneity in occupancy (Mackenzie et al. 2002). However, we first had to ensure our model met the assumptions associated with regression.

Prior to model construction, we transformed our data if skewed or had natural breaks in the data, checked for multicollinearity, and standardized our remaining covariates. We log transformed percent sandstone, slope, discharge, width-to-depth, and chlorophyll-a because these data were skewed. We made drainage area categorical (where 0 was low, 1 was high, and 1 was the reference) and a natural break occurred in our data at 80,000 km² (33% of observations were less than this value). We also made percent backwater categorical (0 = absence, 1 = present, where 1 was the reference) and a natural break occurred in our data at 1% backwater (57% of observations less than this value). Next, we conducted a Pearson's pairwise correlation analysis on our continuous covariates to check for correlations. If our continuous covariates were multicollinear ($|r| > 0.6$), then we selected the covariate that had the greatest number of correlations or chose continuous covariates over categorical covariates. We removed drainage area from the analysis because it was highly correlated to width-to-depth and slope. We also removed slope and percent sandstone from the analysis because they were highly correlated with width-to-depth ratio ($r = -0.63$) and discharge ($r = 0.78$), respectively. Finally, we standardized all continuous covariates to a mean of zero and a standard deviation of one.

We examined the range of our covariates and removed one due to limited variation among sites. Disturbance was relatively constant throughout all catchments ranging from 1.40 to

2.53. The LDI for tributaries ranged from 1.40 to 2.53 and was more limited in the mainstem Red River (1.91 – 2.00). Therefore, we removed this variable from consideration prior to model building.

We evaluated several multi-species, single-season occupancy models in a Bayesian framework using JAGS (Just Another Gibbs Sampler, Plummer 2003) and Program R (version 4.2.2). We hypothesized different combinations of covariates would be important for occupancy by both species but held detection covariates constant for each hypothesis. We tested different combinations of occupancy variables to support overarching hypotheses related to factors supporting either bigheaded carp growth or spawning (Appendix A, Tables 5-6). The most complex growth model contained sinuosity, width-to-depth ratio, chlorophyll-a, discharge, and reaches with the presence of backwater (Appendix A, Tables 5-6). The most complex spawning model contained discharge, salinity, distance to dam, and reaches with the presence of backwater (Appendix A, Tables 5-6). We included reaches with the presence of backwater and discharge in both model frameworks as previous research indicates that bigheaded carp were associated with the presence of backwater and discharge which may be associated with higher forage potential, warmer water temperatures for bioenergetics, decreased energy expenditure, staging locations for spawning and adequate flow for spawning (Williamson and Garvey 2005; Coulter et al. 2017, Song et al. 2018) (Appendix A, Table 7). All models contained grouping factors for year and river (i.e., Red River, Kiamichi, etc.) where multiple sites were nested within river (i.e., to account for pseudoreplication, Wagner et al. 2006). Broad normal priors were used for the coefficients, with gamma priors for standard deviations and uniform priors for occupancy and detection probabilities. All models were run with 3 chains in parallel beginning with a 1,000 iteration adapt phase, a 30,000-iteration burn-in, and a total of 150,000 iterations thinning every 3 iterations using the jagsUI package (Kellner 2015).

We ranked our models using the Watanabe-Akaike information criterion (WAIC) with the NIMBLE package (de Velpine et al. 2022) and selected the models with a delta WAIC score less than 2 as models with equal support (i.e., top-ranked models) (Watanabe 2010; Vranckx et al. 2021). WAIC is considered a Bayesian model selection criterion because it samples from the entirety of the posterior distribution compared to other model selection methods such as the deviance information criterion (DIC) and has been demonstrated to perform better than other

model selection methods for complex Bayesian hierarchical models (Luo 2021; Vranckx et al. 2021).

For our top ranked models, we calculated the mode estimates, 90% highest density intervals (HDI), and estimated detection and occupancy probabilities for the retained covariates. We then predicted the occupancy probability and detection probability for each covariate in our final models within their observed range in the catchment (while holding the other model covariates at mean levels).

We evaluated model convergence and model fit of our top ranked models. We used the Brooks-Gelman-Rubin statistic ($\hat{R}$) to assess model convergence, where an $\hat{R}$ value < 1.1 indicates adequate convergence (Gelman et al. 1992; Gelman et al. 2000). Finally, we assessed model fit with the Bayesian p-value where a value between 0.05 and 0.95 indicates adequate model fit (Kéry and Royle 2016).

Adult bigheaded carp otolith extraction, processing, ageing, and growth

We removed lapilli otoliths for age and growth analyses following Seibert and Phelps (2013) and following our own preliminary assessment (See Birdsall et al. 2024, included as Attachment A). Briefly, the lapilli otoliths, located at the posterior of the skull, were accessed using a hacksaw. Otoliths were then removed using forceps and placed into coin envelopes marked with an individual fish number for later laboratory analyses. In the laboratory, otoliths were sectioned and prepared for age estimation. First, we marked the nucleus on the exterior of the otolith with a ballpoint pen. We then placed the otolith in epoxy resin (West System 105-A) and allowed it to harden for 24 hours. After hardening, the otolith was sectioned using an isomet saw (Buehler IsoMet Low Speed Precision Cutter, Lake Bluff, Illinois) and a single 0.5 to 0.6-mm cross-section was removed from the center of the otolith ensuring the inclusion of the nucleus. We polished the sectioned otolith for 1.5 min on each side with 3- μ m diamond lapping paper (Diamond Lapping Film, 203-mm diameter, plain backing, Electron Microscopy Sciences, Hatfield, PA). Subsequently, we mounted the sectioned otolith onto a slide using thermoplastic cement. The slide was then placed under a dissecting microscope equipped with a light source and imaged with a digital camera (Luminera Infinity 2, Tyledyne Luminera, Ontario). The images were saved for later growth analyses.

Age and growth of Bigheaded Carp

Two readers separately enumerated the annuli of the sectioned otolith to age each fish using transmitted light under a dissection microscope. An annulus was defined as a pair of translucent and opaque bands that continued uninterrupted around the nucleus (Dzul et al. 2012). The edge was counted as an annulus for fish captured prior to April 1st because an annulus was presumed to be created during the spawning season (Minard and Dye 1998; Ericksen 1999). There was no prior knowledge of the fish's length, weight, or age to avoid reader bias. If there was no consensus on the age of a fish, then the readers discussed how they derived the age, and a consensus was obtained.

We quantified the proportional growth of bigheaded carp to determine how growth related to discharge and temperature patterns and fish length (see Adult Bigheaded Carp Data Analyses). The annuli and edge were analyzed for proportional growth using Infinity Analyze 7 software (Tyledyne Luminera, Ontario) (Quist and Isermann 2017). Otoliths were measured for incremental growth along the midventral axis. The focus was identified, and then individual radii distances were recorded from the focus longitudinally to the outside edge of each opaque band to determine individual year growth (Weisberg et al. 2010). The distance from the focus to the edge was used to relate incremental growth to fish length.

Body condition and fecundity

Body condition and fecundity of Bighead and Silver Carp were analyzed for bigheaded carp captured in the mainstem Red River and its major tributaries from June 2021 through November 2023. For body condition, we calculated the relative weight (W_r) of individual fishes using standard weight (W_s) equations described by Lamer (2015).

For Bighead Carp, the W_s equation is:

$$\log_{10} W_{s(g)} = -4.65006 + 2.88934 (\log_{10}(tl(mm)))$$

For Silver Carp, the W_s equation is:

$$\log_{10} W_{s(g)} = -5.15756 + 3.06842 (\log_{10}(tl(mm)))$$

These equations were developed such that a W_r value of 100 indicates that a fish is in average condition (Lamer 2015). Typically, W_r is correlated with growth, but growth measures (above) would be more direct. Moreover, population data should represent the entire geographic range of the species to avoid misinterpretation of the growth form (Murphy et al. 1990). We calculated the gonadosomatic index (GSI) of a subset of female bigheaded carp and estimated fecundity. GSI, a ratio of gonad weight to body weight, is a commonly used indicator of reproductive periods. The reliability of GSI in determining reproductive status has varied among species with different reproductive strategies and is most useful when fish species spawn once annually. Although results are mixed in intermittent spawners GSI can sometimes be used to identify spawning peaks (see Brewer et al. 2008). GSI of female fish was calculated as (gonad weight/total body weight) * 100 (Strange 1996). Fecundity was estimated based on density-weight relationships where a sample was taken from each ovary, enumerated, and then multiplied by the weight of the ovaries (Crim and Glebe 1990). We began by taking the total weight (g, +/- 1 g) of the ovary. We then took subsamples (0.3 – 0.5 g) from the anterior, middle, and posterior of the ovary and enumerated the eggs for each subsample. From these enumerated subsamples, we then estimated the average eggs per gram and extrapolated that to the respective ovary weight.

We processed a subset of carp ovaries to determine if successful spawning occurred within either carp species in the lower Red River. Carp ovarian tissues samples were taken from the central portion of the ovarian lobe and were fixed in 10% neutral buffered formalin (NBF) for a minimum of five days to ensure proper fixation. Samples were rinsed to remove excess fixative and a 10 x 10 x 5-mm subsample of tissue was removed and placed into a microcassette for tissue processing. The samples were then dehydrated through a series of graded ethanol concentrations (80% to 100%) and cleared with xylene to facilitate paraffin infiltration (Fournie et al. 2000; Mumford 2004). Tissue dehydration was automated process using an Eprelia STP120 Tissue Processor and embedded using an Eprelia Histostar embedding station. Ovarian samples were embedded in stainless steel molds (i.e., 10 x 10-mm or 25 x 25-mm) and cooled on a cryotable before sectioning. Embedded samples were sectioned at 5- μ m thickness using a rotary microtome (Leica RM2125 RTS). Sections were transferred to a heated water bath (45°C), mounted on glass slides, and dried overnight. Slides were stained and coverslipped using an automated process (SAKURA Tissue-Tek, Torrance, CA) following the National Wild Fish

Health Survey Laboratory Procedures Manual. The process involved deparaffinization with xylene, rehydration with distilled water, staining with hematoxylin, nuclear bluing in Scott's Water, counterstaining with eosin, dehydration (80% to 100% ethanol); lastly, they were cleared with xylene and coverslips added.

Ovarian sections were examined using transmitted light on a microscope (4x-10x magnification) to classify ovarian developmental stage based on the most advanced oocyte present within the sample following descriptions within Brown-Peterson et al. (2011). For each female, we classified the most advanced oocyte stage present, identified the presence or absence of atresia, but our primary interest was identifying the presence of post ovulatory follicles which reflects recent spawning. We quantified oocyte stages based on seven development stages: primary growth (small, angular, and darkly stained), cortical alveolar (larger, circular, with a peripheral white ring), primary vitellogenesis (yolk vesicles appear in a ring near the periphery), secondary vitellogenesis (yolk globules filling inward towards nucleus), tertiary vitellogenesis (large oocyte full of yolk globules), oocyte maturation (germinal vesicle migration or breakdown), and post-ovulatory follicles (collapsed follicular wall after oocyte is released). Our primary interest was in the presence of post-ovulatory follicles as that is a direct indication that spawning has occurred.

Growth, mortality, and recruitment analyses

We calculated the mean back-calculated length-at-age for all ages to be used in a growth model. Back calculation for length-at-age was conducted using the Dahl-Lea method because of the lack of a known biological intercept (Francis 1990; Quist and Isermann 2017). We fit a von Bertalanffy growth model (vBGM) to bigheaded carp using the previously collected back-calculated length-at-age data. We used a vBGM for bigheaded carp because it is widely used for comparing growth between fish populations (Quist and Isermann 2017) and can elucidate important population growth parameters, such as the average asymptotic length (L_{∞}) and the population growth coefficient (k). These parameters can then be compared post mitigation if management practices aim to reduce fish growth.

We used a mixed-effects model, described by Weisberg et al. (2010), to relate Silver Carp and Bighead Carp growth to environmental conditions of the lower Red River catchment. It can be difficult to relate growth to the environment because growth is correlated with fish age,

fish length, and fish from the same cohort because cohorts can display higher growth rates than others (Watkins et al. 2017). Advances in mixed-effects growth models have permitted us to account for the age, length, and interactions between individual fish during a given year to assess the effects of environmental factors on growth (Weisberg et al. 2010). We modeled age, discharge, and water temperature as fixed effects while year and fish were random effects. This catchment experiences relatively high annual weather fluctuations including longer periods of flood and drought (see Mollenhauer et al. 2022).

We hypothesized that both Bighead Carp and Silver Carp growth were related to discharge and water temperature conditions. We created species-specific models relating the 75th percentile of discharge (m^3/s) (i.e., relatively high flows), the coefficient of variation (CV) of discharge (i.e., flow variability), the 75th percentile of air temperature ($^{\circ}\text{C}$), and the CV of air temperature to fish growth from April through September across the catchment. We used air temperature as a surrogate for water temperature due to the lack of consistent water temperature data for all the years considered, and water temperature is highly related to air temperature throughout the catchment (Morrill et al. 2005; Adlam et al. 2022). The oldest fish in our sample (e.g., 17) would have been recruited in 2004, however, because no fish younger than age 3 were observed in the lower Red River catchment we truncated our data to model growth from age 3 through the maximum age. Thus, we collected discharge and temperature data from 2007 through 2019 and calculated the 75th percentile and CV for the season (April 1st = September 30th).

We used Akaike's information criterion corrected for small sample size (AICc) to rank several models (Sugiura 1978). We constructed the following models: random effects (i.e., year, fish) and fish length with no environmental factors, all combinations with random effects, and a global model. We conducted model averaging for models that had an Akaike's difference (ΔAIC) less than two (Burnham and Anderson 2002). We then calculated the marginal R^2 and the conditional R^2 for both fixed and random effects, respectively, for the averaged models (Nakagawa and Shielzeth 2013). We used the "lme4" (Bates et al. 2015), "AICcmodavg" (Mazerolle 2020), and "MuMIn" (Barton 2022) packages for our analyses.

We used a catch curve to analyze mortality and recruitment of Silver Carp. We used a Chapman-Robson peak-plus catch-curve corrected for overdispersion to estimate mortality and recruitment variability via the recruitment variability index (RVI) (Isermann et al. 2002) for

Silver Carp only due to the small sample size for Bighead Carp (Smith et al. 2012). Peak plus denotes that the first age class used in the analysis is the age following the age with the largest quantity (Smith et al. 2012). Catch-curves for estimating mortality and recruitment are susceptible to bias when age classes are missing from these data (Catalano 2009), however all age classes (> 3) were present for Silver Carp.

Objective 2. Native fish species assemblage assessment

Native fish sampling

For native fish sampling, we used methods and gears described for Objective 1 (see Appendix A, Table 1 for gear specifications). Briefly, sites targeting juvenile and smaller-bodied fishes were sampled using three gear types: mini-fyke nets, beach seines, and larval tows. Mini-fyke nets were set in 1-2 m of water for approximately 6 h during daylight. Beach seining was conducted within areas of the site that allowed for seining (i.e., depths < 1 m). Sub-surface larval tows were conducted by towing an ichthyoplankton net upstream for approximately 10 min at each site at a representative location of deeper water (i.e., where we could not seine or set fyke nets). Each tow was completed for approximately 10 minutes and the volume of water filtered was quantified using a flow meter (General Oceanics Mechanical Flowmeter Model 2030R) attached to the mouth of the net (in case we captured large numbers of larval carp). Identifiable species were enumerated and recorded for each gear used. All larval individuals and unknown species were preserved in a 70% ethanol solution for later identification in the lab. At sites targeting larger-bodied fishes, we conducted electrofishing and net surveys. Three gill nets and three hoop nets were placed throughout each site to soak for approximately 6 h. Following net placement, the site was sampled via boat electrofishing. All sampled fish were identified to species, and the sampling method associated with each catch was recorded.

Native fish habitat

At each site, we quantified the physicochemical factors that may also be related to native fish distributions as described for Objective 1. Briefly, we collected both detection and occupancy covariates. For juvenile and smaller-bodied fishes we quantified: water temperature ($^{\circ}\text{C}$),

dissolved oxygen (mg/L), turbidity (cm), discharge (m³/s), salinity (ppt), average depth (m), width-to-depth ratio (m), zooplankton biomass (µg), large woody debris (%), forewater/backwater (%), and pools (%). For adult and larger-bodied fishes we quantified: salinity (ppt), water temperature (°C), water visibility (cm), discharge (m³/s), width-to-depth ratio (m), and backwater (%). We also quantified several geospatial covariates: distance from dam, elevation, sinuosity, slope, drainage area, and lithology.

Native fish data analyses: juvenile and adult occupancy modeling

We built two multispecies single-season occupancy models (MSOM) to 1) quantify juvenile nursery habitat by native fishes, and 2) quantify habitat associations of large-bodied adult fishes (Mackenzie et al. 2002). The juvenile occupancy model included data collected from 2021 and 2022 and adult occupancy model included data collected from 2021, 2022, and 2023. An occupancy model allows for the estimation of a probability of occurrence while accounting for incomplete detection by the sampling gears. Variation in both detection and occupancy is explained by collected environmental covariates (Mackenzie 2006). We built occupancy models (OM) using temporally replicated surveys at sites to create a detection history (1 if the species is detected, and 0 if it is not). Repeated surveys allow for the model to create estimates of both a detection probability (p_i) and an occupancy probability (ψ_i) (Kéry and Royle 2016). We were able to meet all four of the assumptions for OM (see objective 1 for description). We met the assumption of species' closure by establishing a season (i.e., May – October for juveniles and April – September for adults) during the spawning period of many native fishes of the catchment (e.g., after the water has reached >18°C). Our season ended while juvenile fishes were still using nursery habitat but before water temperatures declined appreciably during late autumn, and before adult fishes moved to over-wintering habitats. The second and third OM assumptions were met with the inclusion of both detection and occupancy covariates to explain variation in detection or occupancy probabilities (Mackenzie et al. 2002). We met the final assumption by spacing our juvenile sites at least 250 m apart and our adult sites at least 1 km apart so surveying one site did not influence detection at an adjacent site. Lastly, we included grouping factors to account for the nested nature of river systems and to account for pseudoreplication in these data.

We transformed and standardized data prior to model development. For juvenile sites, we first began with the detection covariates and any covariates that were not normally distributed

were transformed. Dissolved oxygen, visibility, effort, and discharge were log-transformed in the juvenile data set due to their right-skewedness. Next, we checked detection covariates to ensure they were not multicollinear ($|r| > 0.50$; Roever et al. 2014) using Pearson's correlation coefficient. All detection variables had $|r| \leq 0.35$ and were therefore, retained for the model building process. We completed the same process for occupancy covariates. The percent of limestone lithology, slope, LWD, thalweg depth, W:D, and zooplankton counts were all log transformed due to skewed distributions. Additionally, drainage area, percent of deep pools in the reach, and percent of slackwater in the reach were transformed into categorical variables based on natural breaks in these data (i.e., bimodal). Categorical transformation of drainage area represented either high ($>50,000 \text{ km}^2$) or low ($<50,000 \text{ km}^2$) drainage areas, whereas deep pools and slackwater represented either presence or absence. Categorical covariates were tested for independence by evaluating frequency at which they occurred together at each site. The W:D was multicollinear with salinity ($|r| = 0.53$) and LDI ($|r| = 0.52$). Further, median discharge was multicollinear with zooplankton ($|r| = -0.63$). Slope was also highly negatively correlated with sinuosity ($|r| = -0.53$). We retained W:D, median discharge, and slope for model development. Lastly, all continuous covariates were standardized to a mean of zero and a standard deviation of one to improve model convergence and interpretation (Mackenzie and Royle 2005; Mackenzie et al. 2017). For adult fish, we used that same approach with our covariates that was used in Objective 1 for adult bigheaded carp occupancy modeling. However, we included 2023 data in our adult species model and excluded chlorophyll-a as that was not collected during 2023 sampling.

We built occupancy models using covariates to inform the variation in both detection and occupancy. We built the detection component of the model by choosing two covariates that were hypothesized to share relationships among fishes and gear detection (i.e., not species specific) so more emphasis could be placed on the occupancy portion of the model. Discharge and water temperature were retained for juvenile fish and are commonly used to explain detection (Maire et al. 2019; Carpenter-Bundhoo et al. 2023). For detection of adult fishes, we retained electrofishing time and water clarity as these are also commonly used to explain detection. The detection component of the model is expressed as:

$$\text{logit}(p_{ijk}) = a_i + \beta_1 X_{1[ij]} + \beta_2 X_{2[ij]},$$

$$\text{for } i = 1, 2 \dots i, \text{ for } j = 1, 2 \dots j, \text{ for } k = 1, 2 \dots k,$$

$$a_i \sim t(\mu, \sigma^2, \nu),$$

$$\beta_m \sim t(\mu, \sigma^2, \nu),$$

where p_{ijk} is the detection probability during survey j at site i for species k , a_i is the detection intercept, β_1 & β_2 are the coefficients for the two covariates, and X_1 & X_2 are the detection covariates.

The occupancy portion of the models were built similarly to the detection portion, except we fit species-specific relationships using the covariates. The detection component was held constant as the occupancy component was fit. For juvenile fishes, we fit the occupancy component with the presence of slackwater in the reach, the presence of deeper-water pools in the reach, high or low drainage area, and the continuous covariates of thalweg depth, W:D, LWD, distance to the nearest upstream dam, median discharge, slope, and percent limestone lithology. For adult fishes, we fit the occupancy component with high or low drainage area, the presence of backwater, and the continuous covariates of elevation, sinuosity, distance to dam, W:D, median discharge, and salinity. Each species was modeled around the group mean, hyperparameter μ . The interpretation is similar to a random-slopes model where individual species are treated as random intercepts rather than focusing on interspecies differences. The resulting occupancy probabilities are interpreted similar to individual models but with the power of a single model (Kéry and Royle 2016). We also included grouping factors for both segment and sample year to account for any unexplained variability within the model. The inclusion of grouping factors within the model also accounts for pseudoreplication and spatial correlation created by the nested site study design (Wagner et al. 2006). The occupancy component of the model is expressed as:

$$\text{logit}(\psi_{ik}) = a_i + \beta_{1[i]}X_{1[k]} + \beta_{2[i]}X_{2[k]} \dots + \beta_{N[i]}X_{N[j]} + \gamma_{k[s]} + \gamma_{k[y]},$$

$$\text{for } i = 1, 2, \dots, i, j = 1, 2, \dots, j,$$

$$a_i \sim t(\mu, \sigma^2, \nu),$$

$$\beta_{N[i]} \sim t(\mu, \sigma^2, \nu),$$

$$\gamma_{k[s]} \sim t(\mu, \sigma^2, \nu), \text{ for } s = 1, 2, \dots, 3,$$

$$\gamma_{k[y]} \sim t(\mu, \sigma^2, \nu), \text{ for } y = 1, \dots, 2$$

where ψ_{ik} is the species probability of occurrence at site i for species k , a_i is the occurrence intercept, β_N are the coefficients for the occurrence covariates X_N for species k at site i , $\gamma_{k[s]}$ is

the segment grouping factor, $\gamma_{k[y]}$ is the year grouping factor, and μ is the mean hyperparameter of the assemblage, and σ is the variance around hyperparameter μ .

We used vague priors to calculate the posterior distributions. When informative prior information is not available, vague uninformative priors are used to give the model a starting point for estimating parameters with minimal effect on the model results (Kruschke 2014; Kéry and Royle 2016). Vague truncated normally distributed priors (i.e., t-distribution) were given to main effects, and vague gamma priors were applied to their standard deviations. The t-distribution adds a normality parameter ν which accounts for heavy tails and can improve model fit (Kruschke 2014). Lastly, uniform priors were used for both the detection and occurrence intercepts to aid in model convergence.

We assessed the posterior distribution of the model and covariates using Markov Chain Monte Carlo (MCMC) simulations (Marjoram et al. 2003). Due to the large number of covariates included in the model, 150,000 iterations were run on 3 chains with a burn-in of 10,000 and thinning of 5. The model was fit using the package jagsUI (Kellner 2015) and the program JAGS (Plummer 2003) within the statistical computing software R (Version 4.2.2, R Core Team 2022). The back transformed logit parameter was used to calculate the detection and occurrence probabilities. Model convergence was evaluated using the Brooks-Gelman-Rubin statistic $\hat{R}$ (Gelman et al. 1992; Gelman et al. 2000), where parameter estimations, $\hat{R} < 1.1$, indicate appropriate mixing of chains. Lastly, we used an omnibus goodness-of-fit test (i.e., evaluating chi-squared discrepancies; Mackenzie and Bailey 2004), where $\hat{c}$ values within 1.00 to 1.02 are considered to have adequate dispersion (Kéry and Royle 2016). Additionally, the Bayesian p-value also provides a posterior predictive check, where values near 0.5 (i.e., values that are not close to 0 or 1) are considered to fit the observed data (Kruschke 2014; Kéry and Royle 2016; Conn et al. 2018).

Objective 3. Invasive Bigheaded Carp Movement

Fish tagging and SUR deployment

We captured 25 Bighead Carp and 25 Silver Carp for subsequent tagging with acoustic transmitters (V16-4X, InnovaSea Inc) at multiple locations within the lower Red River

catchment (Appendix A, Table 8). We targeted areas where we were able to capture bigheaded carp during earlier sampling events. Tagging occurred in March and April of 2023. We captured both species using sinking gillnets and electrofishing. We used gillnets that were 54.8-m long for the mainstem Red River and 30.5-m long for the tributaries and backwaters. Both sizes were 3.65-m tall with 8.9, 10.16, and 10.8-cm bar-length mesh panels. We used an 80-amp Midwest Lakes Electrofishing Systems shocking unit using DC electrofishing (Midwest Lakes; Polo, Missouri) to conduct the electrofishing. We used standard AFS electrofishing settings based on conductivity and power (Miranda 2009). Gillnets were set throughout an area and then electrofishing was conducted throughout the area. We then pulled gillnets and collected any captured Bighead Carp or Silver Carp. No more than 12 bigheaded carp were tagged in a single location (Appendix A, Figure 4). We tagged 29 males (16 Bighead and 13 Silver carps) and 21 female bigheaded carp (9 Bighead and 12 Silver carps) (Appendix A, Figure 5). The average length of tagged Silver Carp and Bighead Carp were 871 (+/- 93mm) and 1131 mm (+/-104 mm) and, on average, weighed 9,632 g (+/- 3822 g) and 21,698 g (+/- 5953 g), respectively.

We used an electrosedation table to anesthetize both species. We used the electrosedation table until fish reached stage IV of sedation (Summerfelt and Smith 1990; Kim et al. 2017). Briefly, Stage IV sedation causes the fish to have total loss of equilibrium, muscle tone, and responsiveness to visual and tactile stimuli, however maintaining a slow opercular movement. We then measured the total length (TL, mm), and mass (g) of each fish. We only tagged bigheaded carp large enough to meet the minimum tag burden (i.e., transmitter was < 2% of total body weight) (Winter 1996). We then made a 5-6-cm incision on the left-ventral side of the body in between the anal and pelvic fins. We determined sex based on visual observation of gonads or the presence of sperm when handling a bigheaded carp. We inserted the transmitter into the coelomic cavity of the bigheaded carp and then closed the incision with barbed sutures (PDO-Adjustable Loop, 0 Violet 60cm, CP-2 Rev Cut 26mm 1/2C, Corza Medical, Westwood, MA, USA) by passing the suture needle through the skin on either side of the incision, ensuring the sutures were secured and closed the opening completely. We disinfected all surgical instruments between surgeries with 90% ethanol while in the field. We placed a looped FT-4 Lock-on external tag (Floy Tag & Mtg Inc, Seattle, Washington) at the posterior end of the dorsal fin (Coulter et al. 2022). These tags were placed to allow external identification by the researchers and the public. Once tagged, the fish were held in a recovery tank until the bigheaded carp

resumed normal swimming. We then released tagged fish within 50 meters of their captured location. We exclude all detections of tagged bigheaded carp for 14 days after tagging due to the possibility of movements being related to the stress of handling rather than typical behavior (Frank et al. 2009).

Submersible Ultrasonic Receivers (SURs) were deployed in areas that were necessary to determine if tagged fish left the study area or entered/left particular areas (Appendix A, Table 39). We placed two SURs together (hereafter gates) spaced 0.5-3.0 rkm apart for each gate. This configuration prevented simultaneous detection on paired SURs, increased the probability of detecting tagged carp by one of the SURs in a gate, and provided information on movement direction (i.e., by passing two SURs). Each SUR continuously scanned for transmissions and collected the identification number and time stamp of tagged carp when detected. Fifteen of our SURs were placed in tributaries and backwaters which were presumed habitats for feeding and refuge (Birdsall 2023; Werner et al. 2023) and would be important for our state-space model (see movement pattern analyses). For tributary or backwater locations, we placed SURs at least 0.8 rkm away from the mainstem river to avoid detection of tagged carp passing the area in the mainstem river.

We moored SURs and performed range testing to ensure they were in appropriate locations to ensure detection at baseflow and moderate flows. We moored receivers in place using a 72-kg concrete anchor with a 0.6-meter rebar post placed in the middle of the anchor. The rebar post was then attached to a surface buoy by a 6.35-mm stainless steel cable. The receiver was then attached to a rebar bracket and threaded onto the rebar anchor post. A secondary rope was then attached from the receiver bracket to the surface buoy. The water depth of SUR locations varied by site and season, but locations were chosen to ensure SURs were always submerged. Once deployed, we tested receivers on multiple occasions using a range testing tag to analyze detection ranges of receivers following the manufacturers guidelines (<https://support.fishtracking.innovasea.com/>). The distance at which detection probability falls below 50% in poor conditions or 75% in good conditions was ruled as the estimated detection range (Innovasea Inc). This process is important in understanding the range in which tagged fish can be detected. Detection range can be influenced by water depth, water temperature, current velocity, and other environmental conditions (Kessel et al. 2014). The detection range of our SURs ranged from 700 m during tests at base flow conditions to 98 m during higher discharge

conditions (i.e., we expect to miss fish detections when the Red River is at high flows). The average detection range of our SURs during testing was 238 m. The detection range varied with seasonal environmental changes; thus, carp detection was less than 100% (as with any sampling). We marked the GPS coordinates of each SUR and monitored them every 2-4 weeks to download data and maintain the moorings.

Active Tracking

Along with passive tracking from our SUR array, we actively tracked tagged bigheaded carp. We actively tracked weekly from March through August 31st of 2023 and 2024 and several occasions during autumn. Descriptions of our active tracking reaches are identified in Appendix A, Table 42. All tracking was conducted between ~0700-1900 hours. We tracked multiple sections of mainstem river and tributaries within the lower Red River catchment in Arkansas, Oklahoma, and Texas (Appendix A, Figure 6). Tracking sections varied in length and distribution within the catchment due to limited boat access and varying water levels throughout the study period (i.e., some cannot be accessed at very low flows). We tracked 3-5 days a week from March to August and 1-3 times monthly in September-December. We conducted tracking by boat or raft, moving in a downstream direction at approximately 7-9 km/h. We towed a VHTx—69k transponding omni-directional hydrophone behind the boat which listened for tagged bigheaded carp. The hydrophone was paired with a VR100-300 manual receiver to detect the presence of tagged bigheaded carp. Using the omni directional hydrophone, we approached a fish's location until decibels read ≥ 75 (Coulter et al. 2016). We then used a directional hydrophone to triangulate and confirm the fish's location (Calkins et al. 2012). Once a location was determined, we recorded the GPS coordinates, date, and several habitat characteristics.

Movement pattern environmental covariates

We quantified coarse-scale physicochemical conditions we hypothesized to relate to both Bighead Carp and Silver Carp. We defined movements as the distance traveled between consecutive detections and rate as the distance traveled per day between consecutive detections. Our covariates included average daily discharge per movement ($1.0 \text{ m}^3/\text{s}$), average discharge of the three days prior to each movement ($1.0 \text{ m}^3/\text{s}$), average water temperature of the three days

prior to each movement (1.0°C), day of study (1 day), average daily water temperature per movement (1.0°C), coefficient of variation (CV) of daily water temperature per movement, CV of daily discharge per movement, and average daily photoperiod per movement (1.0 min) (Appendix A, Table 9). We took environmental measurements at coarser scales (i.e., catchment and segment) due to both species' capacity for long-distance movements and the positioning of USGS stream gages (i.e., similar discharges over space and time).

We obtained average daily discharge and water temperature from several locations in the lower Red River catchment. We collected mean daily discharge measurements from USGS stream gages near Denison Dam TX 07331600, Arthur City TX 07335500, De Kalb TX 07336820, Index AR 07337000, Fulton AR 07341500, and Spring Bank AR 07344370. We also collected water release data from Hugo Dam in Oklahoma and Millwood Dam in Arkansas (Appendix A, Figure 7). We compiled discharge data from the closest location to the most recent carp detection for each movement made by an individual carp. We obtained water temperature data from our SURs placed across the catchment (Appendix A, Figure 5). SURs recorded water temperature every hour starting at the time of deployment. We calculated the mean daily water temperature from each SUR, and compiled water temperature data from the closest SUR to the most recent carp detection.

We developed three hypotheses (and associated flow metrics) about how flow patterns might be related to carp movements. We hypothesized that carp movement distances and rates (i.e., km per day) would be positively related to more variable (i.e., CV of discharge) and increased discharges (i.e., mean daily discharge) and that movements would be biased in an upstream direction. The importance of discharge variability has been described for other species of pelagic broadcast spawning species (e.g., Southern Great Plains minnows, Matthews 1988; Worthington et al. 2018; Wedgeworth et al. 2023). Although minnows are diminutive compared to carp, we hypothesized variability to be important to carp given the shared reproductive traits. We calculated average daily discharge and CV of discharge between redetections of tagged carp. Carp made longer movements under higher discharge conditions in the Wabash and Illinois rivers (Peters et al. 2006; DeGrandchamp et al. 2008; Coulter et al. 2016). Higher daily discharge conditions can initiate spawning migrations, cue fish to use different habitats, and are associated with successful reproduction in pelagic broadcast spawners (Schrack et al. 2001; Durham and Wilde 2009). Lastly, fish often do not react to changing environmental conditions immediately

(Forsythe et al. 2012); therefore, examining conditions prior to a movement likely better associate as a cue. To capture conditions leading up to a movement, we averaged the discharge of the three days before each movement following Forsythe et al. (2012) and Vine et al. (2019).

We developed three hypotheses about how our temperature metrics would relate to carp movements. First, we hypothesized that carp movement distances and movement rates would have a quadratic relationship with average daily water temperature. We anticipated that carp would move upstream as temperatures increased and downstream as temperatures cooled due to the associations with upstream movements during spawning migrations and need for refuge habitat during colder seasons (Hintz et al. 2017a; Vallazza et al. 2021). We calculated the mean daily water temperature over the period between consecutive fish detections to relate to carp movement. We hypothesized that carp would move more under moderate temperatures that would correspond to spawning (i.e., $> 18^{\circ}\text{C}$, Jennings 1988; Nico et al 2005) and that fish would begin to move less as temperatures continued to warm because of the tradeoff between energy expenditure for movement and feeding (Prechtel et al. 2018; Glubzinski et al. 2021). Second, we hypothesized carp would have a positive linear relationship with water temperature variability (i.e., CV). Variation in water temperature may cue carp to find spawning habitat (Olden and Naiman 2010). Lastly, because conditions leading up to fish movements may have a greater relationship with movement distances and rates than the conditions during a movement, we calculated the mean water temperature three days prior to each movement following Forsythe et al. (2012).

We hypothesized carp movement rates and absolute distances would have a positive relationship with mean photoperiod. Photoperiod is known to influence the timing of large movements through the increase in hormone production in male and female fish (Jonsson 1991; Norberg et al. 2004). We recorded the photoperiod of each day through sunrise and sunset timing using Astronomical Applications Department of the U.S. Naval Observatory (<http://aa.usno.navy.mil>) data, using Idabel, OK as the reference location. We took the difference between the two times of each day to obtain a quantitative metric for the length of photoperiod per day (Munz and Higgins 2013). We then calculated the mean photoperiod of the period between consecutive detections for each fish.

We also hypothesized that time of year, sex, and size of fish would be related to carp movements. We recorded the date of each movement and used calendar day to represent the

timing of each movement. We hypothesized carp would move more in the spring and autumn as was found in the Illinois and Wabash rivers (DeGrandchamp et al. 2008; Coulter et al. 2022b). Several fishes exhibit sex-based movement trends related to spawning and dispersal (e.g., Cichlids, Stiver et al. (2007); Brook Trout, Hutchings and Gerber (2002); and Coho Salmon, Rodnick et al. (2008)). We hypothesized female carp would move greater distances and at faster rates than males because of their role in seeking out spawning habitat (Waters and Noble 2004; Miller and Scarnecchia 2011). Regardless of sex, larger individuals may be able to swim greater distances or seek different habitats compared to smaller individuals (Radinger and Wolter 2014).

We hypothesized Bighead Carp would move greater distances and at faster rates than Silver Carp. Although both Bighead Carp and Silver Carp are assumed to hold similar trophic levels and have similar habitat needs (DeGrandchamp et al. 2008; Yao et al. 2016), many other species sharing those traits exhibit different movement dynamics with some being more mobile than others (e.g., Rock Bass *Ambloplites rupestris*, Redbreast Sunfish *Lepomis auritus* compared to Largemouth Bass *Micropterus salmoides*, Bluegill *Lepomis macrochirus*, Gatz and Adams (1994)).

Movement pattern analyses

We used detection locations from both active and passive tracking to model the variation in movement patterns of Bighead Carp and Silver Carp. We analyzed movement patterns by adding each detection locations onto a map of the study area using ArcGIS Pro (ESRI, Redlands, CA, USA). We calculated absolute distance, movement rate, and direction (i.e., upstream or downstream) between fish detection. Only one detection per day per fish was analyzed. Multiple detections within the same day occurred regularly. We used the detection which indicated the greatest movement from the last known location. Multiple movements could be made between SURs within a gate on the same day, but we only used one movement to minimize autocorrelation. We calculated daily movement rate (km/d) by dividing the distance moved between two consecutive detections by the number of days the fish was at large. These movements are conservative estimates as the fish likely moved nonlinearly between detections (Coulter et al. 2016). We measured movement direction by assessing the order in which

detections occurred (i.e., detections occurring upstream or downstream compared to the last detection).

We standardized and transformed our data as needed and checked multicollinearity among our covariates prior to model development. We log transformed mean discharge, CV of discharge, three days before movement discharge, and CV of water temperature due to right-skewness. This promotes convergence of the model and ensured we met regression assumptions of linearity and homoscedasticity. We then tested for multicollinearity among variables using Pearson's correlation coefficient. Any variable set with a $|r| > 0.6$ were considered multicollinear; thus, we proceeded using only one of the variables in the set for our model development following Roever et al. (2014) (Appendix A, Table 40). Multicollinearity was a concern with several variable sets; thus, we retained mean discharge, mean water temperature, CV of discharge, CV of water temperature, and carp weight for model building. All continuous variables were then standardized to have a mean of 0 and standard deviation of 1. We used the software R (version 4.4.1 R Core Team 2024) to conduct all analysis.

We had three hypotheses that we examined related to Bighead Carp and Silver Carp movement magnitudes. First, we hypothesized that movement would be primarily influenced by individual traits, as body size and sex can affect fish movement patterns (Miller and Scarnecchia 2011; Okamura et al. 2014; Sánchez-González and Nicieza 2021). Larger individuals and females were expected to move more, and we anticipated an interactive effect between species and discharge, with Silver Carp exhibiting greater movements at higher discharges (Hoover et al. 2017). Individual movement differences may also result from interactions between environmental conditions and intrinsic traits (Radinger and Wolter 2014; Lamarins et al. 2022). Second, we hypothesized that movement would be related to season and direction, given that seasonal changes in temperature, discharge, and photoperiod can influence fish behavior (Winemiller and Jepsen 1998; Taylor and Cooke 2012; Cooke et al. 2022; Schlosser 1991; Norberg et al. 2004; Ferguson et al. 2019). Carp typically exhibit large-scale spawning migrations in response to these cues (Peters et al. 2006; Hintz et al. 2017a; Coulter et al. 2022b), with most dam passages occurring from April to July (Vallazza et al. 2021). Movement direction could also be related to movement magnitude, as greater upstream migrations have been observed under spawning conditions (Peters et al. 2006; Erickson et al. 2016). We expected increased upstream movements with rising discharge and temperature and greater downstream

movements under lower or stable conditions (Yu et al. 2018; Vallazza et al. 2021). Lastly, we hypothesized that movement would be related to variability in environmental conditions, as fluctuations in discharge and temperature, particularly during spring flood pulses, serve as key spawning cues (Moore and Thorp 2008; Taylor and Cooke 2012; Coulter et al. 2016; Hintz et al. 2017a). We hypothesized movement magnitudes would increase when both discharge variability and mean temperature were high, as seen in other cyprinids (Li et al. 2013; Lubejko et al. 2017; Lenaerts et al. 2023a; Alexandre et al. 2016)

We used a generalized additive mixed-effect model (GAMM) to determine the relationships between our covariates and carp movement distances and rates (i.e., two models). We used the hypotheses above to create several subsets of main effects and interactions. All models contained a random effect of individual. Using a random effect of individual accounts for the repeated observations of tagged fish (Otis and White 1999). Both species were included in all analyses as a binary fixed effect. We fit models using the mgcv package in R (Wood 2023). We used thin-plate regression splines and a restricted-maximum likelihood (REML) procedure (Wood 2017).

Model 1: We modeled our covariates related to absolute movement distance (i.e., distance traveled by a single carp between consecutive detections).

Distance model:

$$y_i = f_1(x_{i1}) + f_2(x_{i2}) + \dots + f_p(x_{ip}) + Z_i b_i + \epsilon$$

y_i = response variable for observation i

$f_1(x_{i1}) + f_2(x_{i2}) + \dots + f_p(x_{ip})$ = series of smoothing functions for each predictor variable

$Z_i b_i$ = random effect of individual

ϵ = error term

Model 2: We modeled our covariates related to movement rate (i.e., distance traveled by a single carp divided by the number of days between consecutive detections).

Movement rate model:

$$y_i = f_1(x_{i1}) + f_2(x_{i2}) + \dots + f_p(x_{ip}) + Z_i b_i + \epsilon$$

y_i = response variable for observation i

$f_1(x_{i1}) + f_2(x_{i2}) + \dots + f_p(x_{ip})$ = series of smoothing functions for each predictor variable

Zibi = random effect of individual

ϵ = error term

We first evaluated the assumption of temporal independence of residuals by comparing simple models with and without autocorrelation structures. We fit movement rate and distance models with day of study only to evaluate if movements were temporally autocorrelated. We then compared movement distance and rate models using Akaike Information Criterion corrected for small sample size (AICc) (Burnham and Anderson 2002).

Next, we examined the appropriateness of including smoothing terms. We created a series of six models that contained the main effects of mean water temperature, mean discharge, CV of discharge, CV of water temperature and body weight. Within the base model, all variables had a smoothing term. We used an iterative process by removing the smoothing term from a different variable in each model to examine any possible linear relationships (Hunsicker et al. 2016; Wood 2017). We then compared all candidate models using Akaike Information Criterion corrected for small sample size (AICc) (Burnham and Anderson 2002). The decision to characterize a term as linear or non-linear in future models was also based on the effective degrees of freedom (EDF). Terms with an EDF of one are linear whereas terms further from one tend to be more non-linear (Wood 2017; Wood 2023).

Next, we built hypothesis-based models to investigate movement patterns by carp that mirrored the hypothesized models with movement distance (i.e., changing the response variable; Appendix A, Table 41). We compared all models using AICc (Burnham and Anderson 2002) to determine which model had the greatest support among those considered. The model with the lowest AICc score was designated as our top model. All models with $\Delta\text{AICc} < 2$ were considered to have equal empirical support. If models had $\Delta\text{AICc} < 2$, the simpler models were favored due to simplicity (Burnham and Anderson 2002). We assessed model validation by performing a 10-fold cross-validation using the CVgam function of the gamclass R package (Robertson et al. 2015; Maindonald 2023; Mazziotta et al. 2024). This approach generates a scale parameter (GAMscale) based on the complete data, and an estimate of mean squared error scale parameter from cross-validation (CV-mse-GAM). Models were considered to have good predictive capacity if the mean squared error estimate are slightly larger than the scale estimate (Robertson et al. 2015; Mazziotta et al. 2024)

Multistate analysis

We used detection locations from both active and passive tracking to develop a multistate model (Laake and Rexstad, E. 2008; Lebreton et al. 2009; Kéry and Schaub 2011) to estimate transition probabilities of Bighead Carp and Silver Carp between habitat types. We used the GPS coordinates of each detection from individual carp to determine the date (i.e., period) and habitat (i.e., state) where fish were located. This detection history represented the observation process of individual fish (i.e., where and when we detected a tagged carp). Within our multistate model, carp could move large magnitudes but remain in the same habitat. Our multistate analysis provides additional information because it emphasizes transitions among habitats rather than movement magnitudes. Understanding the probability of carp transitioning to and from habitats during specific time periods can improve management efforts.

We defined three states based on habitats hypothesized to be important to Bighead Carp and Silver Carp. We defined our states based on observed coarse habitat used by carp during the study. We limited our analysis to three states to avoid having an overly complicated model because the number of parameters being estimated increases exponentially with the number of states which quickly becomes prohibitive (White et al. 2006). Our states included backwater habitat (B), tributaries (T), and mainstem river (M). Backwater habitats may be important areas where carp congregate during certain times of year (MacNamara et al. 2018; Prechtel et al. 2018) and may be important for removal as gear recapture probability is higher in backwaters compared to mainstem habitats (Birdsall 2023). Likewise, tributaries have been identified in some rivers as congregation areas (Chapman 2006; Camacho et al. 2023), whereas mainstem habitats could be important for carp dispersal and spawning (Coulter et al. 2016; Werner et al 2023). Understanding conditions that may be related to transitions among major habitats is important because they can inform managers when removal effort may be most effective (Ridgway et al. 2021).

We defined six irregularly spaced time periods based on water temperature and discharge patterns throughout the study period to reflect unique environmental conditions between each time period (hereafter the time periods are t1, t2, t3, t4, t5, and t6) (Appendix A, Table 10;

Appendix A, Figure 8). Briefly, time intervals between t1 to t2, t4 to t5, and t5 to t6 were periods of increasing and variable discharge with increasing water temperatures. Time intervals t2 to t3 and t3 to t4 were periods of decreasing water temperatures and consistently low discharges with single increases in discharge magnitude above the mean for the study period ($170 \text{ m}^3/\text{s}$). We hypothesized some periods would have a greater effect on the probability that carp transition among habitats. We hypothesized that during the time intervals t4 to t5, the probability of transitioning would be higher than the probability of remaining in a single state. This period encompassed environmental conditions that have shown to cue carp to move greater distances and seek different habitat (Jennings 1988; Schrank et al. 2001; Nico et al 2005). We also hypothesized that during conditions associated with lower movement (t2 to t3 and t3 to t4) the probability of remaining in a state would be greater than transitioning to other states.

Multistate models are a generalized form of Cormack-Jolly-Seber (CJS) models (Cormack, 1964; Jolly, 1965; Seber, 1965). The general assumptions of a CJS model that are shared by multistate models include tagging does not affect survival, all individuals have an equal probability of recapture, tagging marks are permanent, sampling occasions are instantaneous, populations are open with emigration considered permanent (Kéry and Schaub 2011). CJS models estimate the probability of apparent survival (s) across open periods and recapture (p) within each period after the initial capture period. Where transitions are constrained to either an “alive” or “dead” latent state. Estimates of apparent survival are not estimates of mortality due to emigration and mortality being confounded with each other. Multistate models introduce a transition probability parameter (ψ) for multiple “alive” states, where individuals can transition back and forth unlike the “dead” state which is permanent. ψ is also estimated across open periods. Our primary interest was in transition probabilities between habitats of carp among discrete time periods. The estimation of ψ across periods allows us to extend traditional mark-recapture models to investigate carp transitions between habitats. An assumption of multistate models is that individuals cannot occupy multiple states within the same sampling period and will transition among states during open periods. Within our detection history a single individual violated this assumption. We only retained the detection of this carp in the first state to avoid violation of the model assumption (Perry et al 2010; Labuzzetta et al. 2024; Van Vleet et al. 2024).

We adjusted our detection history to accommodate shed or missing tags. Some individuals either died or shed their transmitters during our study. We could not distinguish between shed tags or dead fish within our study. For these individuals, a distinction was made in our detection history that censored future detections. This removed the individual from contributing to parameter estimations after it was presumed to have died or shed its tag. This censoring avoided biasing recapture probabilities. Six of these individuals died or shed their tag before the first period. We removed these individuals from the detection history because they provided no information to the model. A major assumption of the multistate framework is that all individuals in a state have the same likelihood to transition and be detected at each time period (Kéry and Schaub 2011). Throughout the study, we made needed adjustments to our SUR array. Some SURs were lost at various times and 10 additional SURs were added to our array in winter 2024 (see Appendix A, Table 39 for SUR locations). However, due to the mobile capabilities of both species and our wide distribution of active tracking reaches (Appendix A, Figure 6), we believe this assumption was reasonably met for the different state groups (i.e., no one group was more likely to be detection than another group). When constructing our detection history, there were multiple instances in which individuals were not found during a closed period but were detected in close temporal proximity to a closed period (i.e., in an open period). These instances result in the model not capturing transitions that occurred in the previous open period. For these instances, we moved individual detections into the closest closed period if the detection was <14 days from the closed period. There were three individuals that were never detected in any closed period but were detected during at least one open period. We moved these detections into closed periods to avoid biasing our survival and recapture estimates. The focus of our analysis was on habitat transitions over coarse time periods, thus moving individual detections allowed us to retain as much information as possible. This technique has been used in other multistate studies (Melnychuk et al. 2017; Coulter et al. 2018a).

We first fit a multistate model using all three states, where transitions probabilities did not vary across the six time periods.

State process:

$$Z_{i,t} = f_{S_i}$$

$$z_{i,t+1} | z_{i,t} \sim \text{categorical}(\Omega_{z_{i,t},1 \dots S_{i,t}})$$

Where $z_{i,t}$ is the true state of an individual i at time t . The state process represents the actual transitions of an individual (Kéry and Schaub 2011).

Observations process:

$$y_{i,t} | z_{i,t} \sim \text{categorical}(\Theta_{z_{i,t},1 \dots O,i,t})$$

Where $y_{i,t}$ is the observed state of individual i at time t . The observed process represents the transitions that are observed and cannot be the same as the state process due to imperfect detection (Kéry and Schaub 2011).

Estimated parameters : $s(B,.)$, $s(M,.)$, $s(T,.)$, $\Psi(B-B)$, $\Psi(B-M)$, $\Psi(B-T)$, $\Psi(M-B)$, $\Psi(M-M)$, $\Psi(M-T)$, $\Psi(T-B)$, $\Psi(T-M)$, $\Psi(T-T)$, $p(B,t)$, $p(M,t)$, $p(T,t)$, where s is survival probability, Ψ is habitat-transition probability, and p is recapture probability.

Derived parameters: $\Phi(B,t)$, $\Phi(M,t)$, $\Phi(T,t)$, where Φ is cumulative survival probability.

Model estimates are reported as the mode (most likely value) with 95% credibility intervals from the posterior distribution (Marjoram et al. 2003). These models indicated transitions between tributaries and backwater habitats were rare. Recapture and survival estimates were also similar between backwater and tributary states (Appendix A, Table 11; Appendix A, Figure 9). This indicated that minimal information would be lost by combining the backwater and tributary states. This is consistent with our observed data, where most transitions were associated with the mainstem. Combining the two states also allowed for time varying estimates of transitions probabilities to be made because there were fewer parameters. The conditions of the lower Red River also support combining tributary and backwater states as they both provided lower-velocity habitats where carp congregated (MacNamara et al. 2018; Birdsall 2023). We made time varying transition probabilities a random effect to generalize our findings to a broader temporal scale (Wagner et al. 2006). We defined the backwater/tributary state as state “A” The two-state model can be expressed as:

Estimated Parameters:

$$\begin{aligned} &s(A,.) , s(M,.) , \\ &p(A,t) , p(M,t) , \\ &\Psi_{A-Mt} \sim \text{Normal} (, , \end{aligned}$$

$$\Psi_{M-At} \sim \text{Normal}(\mu, \sigma^2)$$

Derived Parameters: $\Psi(M-Mt)$, $\Psi(A-At)$,

Where:

$z_{i,t}$ = the true state of an individual i at time t

$y_{i,t}$ = the observed state of individual i at time t

s = survival probability

p = the time-varying recapture probabilities

Ψ = time-varying transitioning probabilities,

μ = the group means associated variance σ^2 .

We fit the multistate model using the direct, state-space formulation described by Kéry and Schaub (2012). We estimated distributions with Markov chain Monte Carlo (MCMC) methods using the program JAGS (version 4.3.1, Plummer 2003) via the package jagsUI (Kellner 2015) in the statistical software R. All probability estimates were given broad uniform priors between 0–1. Adequate MCMC convergence (i.e., chain mixing) was achieved using four chains of 35,000 iterations each ran in parallel after a 10,000-iteration burn-in phase (thinning = 100). We considered convergence a potential scale reduction factor $\hat{R} < 1.05$ (Brooks and Gelman 1998).

Methods – Sulphur River

Objectives 1 & 2. Invasive bigheaded carp population and native fish assemblage assessments

Sampling area

To address the project objectives, we sampled multiple sites across the 72-km segment of the lower Sulphur River waterway between Wright Patman Lake and the confluence of the Red River (Appendix B, Figure 1). Sampling locations along the Sulphur River were selected to maximize the likelihood of capturing bigheaded carp and included the main river channel as well as dam tailwaters, backwaters, and deeper pool environments when available. Additional sampling occurred within Wright Patman Lake to evaluate the reservoir fish assemblage in

relation to the river assemblage and due to concerns that invasive carp could have been introduced into the lake.

River hydrology

River hydrology was examined in relation to fish data. Hydrographs of the Sulphur River and mainstem Red River were constructed for all of 2021, 2022, and 2023. River flow conditions were examined to better understand how dam operations may impact invasive species management and the native fish community. River stage heights were plotted from USGS gauging stations at the tailwater of Wright Patman Lake (USGS station: 07344210) and at the confluence of the Sulphur and Red Rivers (USGS station: 07344370). Visual examination of data was sufficient to characterize periods of flooding and low flow conditions. Although characterization of Sulphur River hydrology was not an explicit objective, it is integral to each objective.

Fish sampling

Unlike the Mississippi and its large tributaries, the Sulphur River is a mid-sized river, and thus, comparatively smaller in channel width, discharge, and other features. When sampling, we targeted the river channel as well as tailwater, oxbow, and confluence habitats when available to increase the likelihood of capturing bigheaded carp. In large rivers with complex river-floodplain environments, bigheaded carp can rapidly move between habitats making their capture more difficult. In smaller systems like the Sulphur River, such lateral movements of bigheaded carp are limited due to the limited availability of side channels and floodplain lakes. To sample fishes along the river, a combination of gill nets, pulsed-DC electrofishing, herding methods, mini-fyke nets, and hoop nets were used to capture bigheaded carp and other fishes (Appendix B, Table 1). During each sampling event, environmental conditions including water temperature (°C), conductivity ($\mu\text{S}/\text{cm}$), and Secchi depth (m) were recorded at each site (Appendix B, Table 2).

Pulsed-DC boat electrofishing (Smith-Root Inc, Apex control box) was conducted at each sampling location (minimum 3 transects per site; 10 mins. pedal time per transect) to capture

fishes. In the Upper Mississippi River basin, transects can extend to 15 mins for monitoring large rivers like the Mississippi River and Illinois River (Gutreuter et al. 1995, 1997). We opted for a shorter duration because the Sulphur River is considerably smaller. We also wanted to minimize the holding and handling times to minimize stress on captured fishes. Pulsed-DC electrofishing has been the most effective means of fish sampling bigheaded carp across the Mississippi River basin (Collins et al. 2015; Culver and Chick 2015). For each transect, the boat traversed the river channel to cover dominant habitats. In doing so, riverbank, channel margin, and thalweg habitats were sampled in an integrative manner. Transect lengths (m) varied depending on habitat conditions but conservatively ranged from 0.8 to 1.06 km depending on boat speed (3-5 miles per hr.). Nonetheless, sampling effort was standardized per unit time. Electrofishing settings ranged from 100-375 volts, 25% duty cycle, 25% frequency, 60 pulses/s, with amps ranging from 4-10. Voltages were adjusted based on river conductivity.

Entanglement gears are effective at catching carp. However, gill nets are not always suitable for sampling during flood conditions because drifting logs can quickly destroy them. Indeed, several nets were damaged or destroyed during early sampling during the project. Recent methodological developments have shown that hybrid herding approaches can be effective at catching and detecting bigheaded carps (Butler et al. 2019; Ridgway et al. 2021). Herding approaches use sound or electricity to drive fish into a gill net. For this project, gill nets were deployed for 10 mins during herding sampling whereas traditional gill net sets can sit for >2 hours. The 10-minute herding duration was used to match the 10-minute electrofishing transects so that effort was consistent between approaches. To date, herding approaches have been used in larger river settings (Butler et al. 2019; Ridgway et al. 2021). For example, Butler et al. (2019) used 5-minute herding to enhance catch rates and detection rates of fishes in the Illinois River.

Our assessment evaluated whether this approach is useful in a mid-sized river setting. When flow and habitat conditions permitted, experimental AFS gill nets (monofilament; multi-panel, varied meshes; Duluth Nets; specifications in Appendix B, Table 1) were deployed and the boat herded fish into nets following procedures described in Butler et al. (2019). Because of the shorter duration in the water, there is a reduced chance that the net will be destroyed by drifting logs. However, the shortened duration may limit gill net effectiveness. Thus, TTU will assess differences in catch rates, species captured, and sampling precision. Finally, when river conditions were suitable, we deployed gill nets for longer durations to capture fishes. Gill nets

were set on the backside of wing dams, within oxbow lakes, and near dam tailwaters when conditions were suitable.

Entrapment nets were also used to capture river fishes. Hoop nets were set parallel to the shoreline to capture fish moving upriver. All hoop nets were deployed in the water for 3-hour periods to minimize reptile by-catch mortality. Additionally, floats were inserted into hoop nets to create space for reptiles to breathe. Sites with relatively shallow depths (1-2 m) with current were selected for all hoop net sets. Hoop nets were spaced approximately 250-m apart to avoid potential gear interference. Additionally, mini-fyke nets were used later in the season (August – October), when flows permitted, to capture small-bodied fishes including juvenile bigheaded carp (Collins et al. 2017).

All individual fish were identified and measured for total length (mm) and weight (g or kg). Length-weight data were recorded for all fish to determine average body size and to construct baseline length-mass relationships to track changes in body condition of important native species (Bigmouth Buffalo *Ictiobus cyprinellus*, Smallmouth Buffalo *I. bubalus*, Black Buffalo *I. niger*, Longnose Gar *Lepisosteus osseus*, Blue Sucker *Cycleptus elongatus*, Flathead Catfish *Pylodictis olivaris*, River Carpsucker *Carpionodes carpio*, Gizzard Shad *Dorosoma cepedianum*). These values can be used in the future to track changes in relative weight values for each species. All fish were released (except bigheaded carp) after processing. If captured, bigheaded carps were retained for collection of fin clips or liver samples so they could be archived for potential future DNA analysis to assess the degree of hybridization within the sampled population.

Data analysis

Catch data (counts; average lengths; catch per unit effort) were summarized for the 2021, 2022, and 2023 seasons. Catch rates (CPUE, # per 10 mins) of bigheaded carp and native fishes were calculated for the Sulphur River, Red River, and Wright Patman Lake for each gear type. Additionally, a summary of the total body lengths (average $\pm$ SD) was conducted to characterize the range of sizes encountered for each species. Additionally, total catch rates were contrasted among gears to identify the most effective gear for sampling the Sulphur River (Kruskal-Wallis test, $\alpha = 0.05$). Pairwise comparisons were then made to determine which gears differed (Wilcoxon method). Comparing gears with different forms of effort can be challenging (Collins

et al. 2015; Lederman et al. 2024). Yet, fisheries scientists are often tasked with identifying the most effective and reliable means of sampling fish. These statistical comparisons were made with the intention of identifying the sampling gear that captured the most fish. Our reasoning is that it would give the best representative sample of the fish community and also identify the gear most likely to capture bigheaded carp.

Electrofishing data (CPUE) was used to analyze the structure of the Sulphur River fish assemblage. Specifically, a non-metric multidimensional scaling (NMDS) analysis was used to examine compositional differences in the fish community across sample sites and months (PC-ORD 7; Bray-Curtis distance). In addition, river flow conditions (categorical: flooding, low flow) were included in the analysis to account for potential drivers of fish community changes. To determine whether the structure of the fish assemblage differed between high and low flow conditions, we applied the multi-response permutation procedure (MRPP; bootstrapped 5000 iterations) to test if groups differed statistically ($\alpha = 0.05$).

Fish body condition is a useful and simple indicator for tracking responses of individual fish to some form of change or disturbance (Irons et al. 2007). For example, changes in body condition (relative weight) of Bigmouth Buffalo and Gizzard Shad have been used to document the negative effects of bigheaded carps in the La Grange pool of the Illinois River (Upper Mississippi River; Irons et al. 2007). Using data from 2021 and 2022, we developed separate linear models of length-weight relationships for Bigmouth Buffalo, Smallmouth Buffalo, Black Buffalo, Longnose Gar, Blue Sucker, Flathead Catfish, River Carpsucker, and Gizzard Shad. Length (mm) and weight (g) measurements were $\log_{10}$ transformed prior to analysis. Each model is specific to the Sulphur River and can be used to evaluate changes in relative weight of individual fish if bigheaded carp populations rapidly increase.

Results – Red River

We completed sampling at 136 sites, 71 where we targeted juvenile bigheaded carp and other small-bodied fishes and 65 sites where we targeted adult bigheaded carp and large-bodied fishes in the lower Red River catchment through September 2023. We conducted 330 surveys at our juvenile sampling sites and 318 surveys at our adult sampling sites. Of the 134 sites, 40% were in the Oklahoma portion of the catchment (hereafter OK Red) ($n = 54$), 35% in the Arkansas

portion of the Red River (hereafter AR Red) ($n = 47$), and 25% were in associated tributaries in Oklahoma ($n = 24$) and Texas ($n = 11$). As expected, gillnets and electrofishing were most effective at capturing larger-bodied fishes, whereas fyke nets and seine hauls collected mainly smaller-bodied native fishes. Hoop nets and larval tows were not as effective at collecting fishes as other gear types. Although we did not meet the assumption of catch-per-effort sampling, we provided CPUE for the agencies in Appendix A Table 37.

The experimental electrofishing settings were not as effective at collecting Carp or getting Carp to jump as the standard settings used during the initial fish assemblage shocking events. When Carp were observed jumping, we were somewhat able to manipulate their swimming direction by using the electrofisher. On several instances, we were able to observe the wakes of bigheaded carp being driven towards set gillnets as they attempted to escape the electric field. However, most bigheaded carp that were actively driven towards the nets would either jump the net upon reaching it or turn around and swim away from it and around the electrofishing boat. We attempted to set our gillnets parallel to the bank to electrofish between the net and bank as was suggested by the Arkansas Game and Fish Commission (Jimmy Barnett, Arkansas Game and Fish Commission, oral communication, 2021). This did not result in any noticeable differences in our catch. We also baited hoop nets with cattle cubes as was suggested by commercial fishermen in Arkansas. However, neither modification improved bigheaded carp catch. Additionally, we did not observe any bigheaded carp while conducting surveys on Bois d'Arc Lake.

Objective 1: Invasive bigheaded carp population assessment

Bigheaded carp sampling and collection

All Carp modeling and analysis was completed using sampling data collected from both tributary sites and mainstem Red River sites in all three states (AR, OK, and TX). Silver and Bighead Carp were detected in both the mainstem Red River and tributaries within the lower Red River basin during adult surveys. No juvenile or age-0 bigheaded carp were sampled or observed in the study area. Silver Carp were detected at 26 of the 49 mainstem Red River adult sites and 13 of the 16 adult tributary sites with an overall naïve occupancy of 0.65. Bighead Carp were detected

at 11 of the 49 mainstem Red River adult sites and 10 of the 16 adult tributary sites with an overall naïve occupancy of 0.32.

Bigheaded carp were observed or captured across the catchment (Appendix A, Figure 10) using a variety of gears. A total of 413 bigheaded carp were collected from May 2021 through November 2023, of which 145 were captured via electrofishing, 240 were captured using gillnets, and the remaining fish were either provided by bow fisherpersons (via the USFWS) or jumped in our boat during sampling (all Silver Carp). We captured 305 Silver Carp and 108 Bighead Carp throughout the lower Red River catchment (this number includes USFWS Carp donated by bow fisherpersons) (Appendix A, Table 12). Although carp were distributed across the study area, most bigheaded carp captured in the mainstem Red River were sampled from reaches with adjacent backwater habitat, with some difference in catch between the two species (Appendix A, Figures 11 and 12). Bigheaded carp were visually confirmed (i.e., observed jumping during sampling but not netted) during 61 surveys (Appendix A, Table 13). For Silver Carp throughout the lower basin, 52% (160 of 305) were captured in gillnets, 41% (125 of 305) were captured from electrofishing, and the remainder were fish that jumped into the boat while sampling or were captured by bow fisherpersons. For Bighead Carp throughout the lower basin (AR, OK, TX), 74% (80 of 108) were captured in gillnets and 18% (20 of 108) were captured using electrofishing, and the remainder were captured by bow fisherpersons.

Bigheaded carp occupancy, detection, and habitat

The occupancy models that had the most support for both species (i.e., WAIC difference <2, Vranckx et al. 2021) included the covariates: presence of backwater in the reach, sinuosity, width-to-depth ratio, and chlorophyll-a ($\mu\text{g/L}$) (Appendix A, Tables 14-15). All top ranked models included the detection covariates of water temperature ($^{\circ}\text{C}$), Secchi depth (cm), discharge, and electrofishing effort (s) (Appendix A, Table 16).

Detection varied with environment indicating catch-per-unit effort (CPUE) would not be adequate to represent bigheaded carp abundance trends (but may reflect places simply easier to sample carp). Detection probability, with our occupancy covariates held at mean levels, ranged from 0.39 to 0.40 for Bighead Carp and 0.60 to 0.63 for Silver Carp (Appendix A, Table 17). Bighead and Silver Carp detection was positively associated with increasing water temperature (Appendix A, Figure 13), and electrofishing effort (Appendix A, Figure 14) and negatively

associated with discharge (Appendix A, Figure 15) and Secchi depth (Appendix A, Figure 16). Given variation in detection, CPUE data are not appropriate for use as trend data, but are provided in Appendix A, Table 37.

Occupancy probability, with our detection covariates held at mean levels, ranged from 0.53 to 0.78 for Bighead Carp and 0.78 to 0.85 for Silver Carp (Appendix A, Table 17). Bigheaded carp occupancy was positively related to reaches with the presence of backwater habitat and negatively associated with sinuosity (Appendix A, Figure 17). Both species of bigheaded carp were also negatively associated with width-to-depth ratio (Appendix A, Figure 18) indicating bigheaded carp used reaches with narrower and deeper channels. Silver Carp occupancy was positively associated with chlorophyll-a (Appendix A, Figure 19), whereas Bighead Carp occupancy had no relationship with chlorophyll-a.

Our top-ranked models converged and had adequate model fit. Our final models achieved convergence as evidenced by all parameters having $R\text{-hat}$ values < 1.1 and visual assessment of the Markov chains (Appendix A, Tables 15-16) (Kéry and Royle 2016). The Bayesian p-values for models with equal support ranged from 0.275 to 0.292 and the $c\text{-hat}$ values ranged from 1.094 to 1.114 indicating adequate model fit (Kéry and Royle 2016).

Bigheaded carp population demographics

A total of 305 Silver Carp (180 males, 116 females, 9 unsexed, 1.5:1.0 sex ratio) and 108 Bighead Carp (67 males, 32 females, 9 unsexed, 2.1:1.0 sex ratio) were sampled or donated by bow fisherpersons throughout the lower basin (Appendix A, Table 12). Silver Carp tended to be smaller and younger, on average, compared to Bighead Carp though Silver Carp tended to grow faster early in life (Appendix A, Table 18). On average, the Silver Carp we collected were 879-mm TL (range: 532-1,100-mm TL), whereas Bighead Carp were 1,105-mm TL (range: 868-1,382-mm TL). The mean age of Bighead Carp estimated using otoliths was 10.4 years, whereas Silver Carp mean age was lower (6.8 years). The oldest sampled Silver Carp and Bighead Carp were age 14 and 20, respectively (Appendix A, Figure 20). Silver Carp were larger (i.e., TL) than Bighead Carp, on average, until age 5. Silver Carp and Bighead Carp mean back-calculated lengths at age 5 were 743 and 748-mm TL, respectively. L_{∞} for both species was relatively high (SVC = 935-mm, Bighead Carp, BHC = 1218-mm), whereas growth rate (k) was higher for Silver Carp ($k = 0.31$) compared to Bighead Carp ($k = 0.17$) (Appendix A, Figure 21). Relative

weights of Bighead and Silver Carp collected in the study area tended to be slightly above that of an average fish. For Bighead Carp captured in the lower Red River catchment, W_r ranged from 94.48 to 107.82, with an average of 101.13 ($n = 102$, $sd: 2.25$). For Silver Carp, W_r ranged from 90.05 to 106.33, with an average of 100.81 ($n = 298$, $sd = 1.58$).

Silver Carp mortality was relatively low and recruitment into the population appeared steady. Our catch-curve for Silver Carp was fit using ages 6 through 14 because age 5 fish had the highest count in our sample. The instantaneous mortality estimate (Z) was 0.35, conferring an annual mortality rate of 0.30 for Silver Carp ages 6 through 14 (Appendix A, Figure 22). As with mortality, only age-6 fish and older were used to calculate Silver Carp recruitment variability index, which was 0.87, indicating recruitment was relatively stable.

Air temperature, discharge variability, and high discharge conditions were related to growth of Silver Carp and Bighead Carp. We model-averaged five Weisberg models associated with Silver Carp growth and three models associated with Bighead Carp growth that had a delta AIC score less than 2 to reduce model bias and address uncertainty (Appendix A, Tables 19-20) (Kruse et al. 2022). Bighead Carp growth was positively associated with warmer air temperatures (75th percentile of air temperature) and higher discharge (75th percentile of discharge) and negatively associated with air temperature variability (CV of air temperature). Similarly, Silver Carp growth was positively associated with the warm air temperature (75th percentile of air temperature) and higher discharge (75th percentile of discharge). However, Silver Carp growth was also positively related to the variability (CV) of air temperature (i.e., as a surrogate for water temperature) and negatively associated with discharge variability (i.e., CV of discharge).

Our fixed and random effects explained a large portion of the variability in our growth models. The marginal R^2 s for our Silver Carp models having equal support ranged from 0.57 to 0.52. Including random effects explained 18% to 22% more variability in our data (R^2 - 0.69 to 0.73). The fixed effects in our top-ranked Bighead Carp models with equal support explained 49% of the variation in our data (marginal R^2 - 0.49). Including the random effects of year and individual fish explained an additional 111% of the variation in growth (conditional R^2 - 0.60).

We examined fecundity of both species by macroscopic observations of ovaries, gonadosomatic index (GSI) calculations, and egg counts estimates of female bigheaded carp. For both Bighead and Silver Carp, we observed ovaries occupying much of the body cavity and

containing developed eggs (i.e., oocytes occupy most of the coelomic cavity) throughout the year. The average monthly GSI was highest in May for Bighead Carp (GSI: 14.95) and in June for Silver Carp (GSI: 20.39) (Appendix A, Figures 23 and 24). GSI values for Bighead Carp ranged from 1.72 to 21.21, with an average of 10.23. GSI values for Silver Carp ranged from 3.87 to 25.75, with an average of 15.68. Egg count estimates for Bighead Carp ranged between 254,817 and 1,406,850 with an average of 774,458. Egg count estimates for Silver Carp ranged between 233,740 and 2,510,504 with an average of 1,494,604.

We successfully processed and quantified 9 Bighead Carp ovaries, and 22 Silver Carp ovaries collected from May 2023 through May 2024 in the lower Red River catchment. Oocyte development varied across time within the catchment. We observed stage six oocytes in May, June, and October of 2023 for Silver Carp and May 2023 and May 2024 for Bighead Carp. Interestingly, the smallest female we collected (i.e., 532-mm, collected October 2023) reflected stage three as the most advanced oocyte. We observed the presence of atretic oocytes (i.e., alpha atresia, Corriero et al. 2021) in five female carp. Most interesting is that two Silver Carp and two Bighead Carp (sampled May 1, 2024) contained post-ovulatory follicles indicating at least one spawning event had occurred relatively recently prior to capture Appendix A, Figure 25 and 26). We also processed carp collected later in May that also had post ovulatory follicles. Combined, these data reflect spawning is occurring within the basin.

Objective 2. Native fish species assemblage assessment.

Native fish sampling

A total of 284,194 fishes, comprising 86 species and 48 genera, were identified during sampling of the lower Red River catchment. Vouchered fishes were reviewed in the laboratory. Of the three river sections sampled, 74 species were sampled from the Red River in Arkansas, 69 species were sampled from the Red River in Oklahoma, and 70 species were sampled in the tributaries (Appendix A, Tables 22-24). The most abundant small-bodied species was Red Shiner (124,248), followed by Bullhead Minnow (49,809), Western Mosquitofish (16,187), Threadfin Shad (13,211), Chub Shiner (13,066) and Mississippi Silverside (12,047). The most abundant large-bodied species was River Carpsucker (2,963), followed by Smallmouth Buffalo (2,158), Bigmouth Buffalo (1,228), Freshwater Drum (1,092), Longnose Gar (802), and Black Buffalo

(758). Conversely, the rarest species with only one or two individuals collected were American Eel, Blackspotted Topminnow, Bowfin, Chestnut Lamprey, Flier, Grass Pickerel, Highland Stoneroller, Mooneye, Redspot Darter, Redspotted Sunfish, Ribbon Shiner, Western Starhead Topminnow, and Yellow Bullhead. Of the 86 fish species, four were non-native including Bighead Carp, Silver Carp, Common Carp, and Grass Carp. The genus that contained the most species collected was *Lepomis* (Appendix A, Table 25). Common and associated scientific names are provided in Appendix A (Table 38). Log-transformed length-weight relationships were developed for 18 species of fishes (see Attachment B, Vilchez et al. 2024)

Juvenile nursery habitat

The final model converged and had adequate fit (see Ramsey et al. 2024, included as Attachment C). All model parameters displayed appropriate chain mixing with $\hat{R} < 1.1$ (Kéry and Royle 2016). The OM displayed adequate dispersion of posterior values and adequate goodness-of-fit.

The probability of detection and occupancy varied by species; however, some relationships with covariates were shared though there were differences in effect sizes (see Ramsey et al. 2024, included as Attachment C). Species detection increased with increasing water temperatures, and discharge conditions. All 38 juvenile species had positive occupancy relationships with reaches having deep pools and slackwater habitats present, and the distance from the nearest upstream dam. Lastly, all species had a negative occupancy relationship with deeper thalwegs and the percentage of limestone within the catchment. Although species had the same relationship with thalweg depth, the effect size of these relationships differed.

Several nursery habitat relationships were species specific (see Ramsey et al. 2024, included as Attachment C). The occupancy relationships with drainage area, segment slope, amount of LWD, W:D ratio, and seasonal median discharge were variable among species and taxonomic groups.

Adult fish habitat

For modeling adult river fishes, we included data collected from 46 sites, 190 sampling events, and 25 large-bodied species in the model. We omitted some sites due to missing physiochemical covariate information and omitted some of the large-bodied species that had extremely low naïve occupancy (e.g., Grass Pickerel). The species that were included in the model included the

families Acipenseridae, Catostomidae, Centrarchidae, Cyprinidae, Ictaluridae, Lepisostedae, Moronidae, Polyodontidae, and Sciaenidae. The model successfully converged and displayed adequate fit (Appendix A, Table 25). All model parameters displayed appropriate chain mixing with $\hat{R} < 1.1$. The OM displayed adequate dispersion of posterior values ($\hat{c}$ of 1.004), and adequate goodness-of-fit with a Bayesian p-value of 0.471.

Large-bodied fishes displayed variability in both the probability of detection and occupancy. Species detection probability ranged from 0.17 to 0.90 with a group mean of 0.51 (Appendix A, Tables 26-27; Appendix A, Figure 33). As expected, species' detection increased with both increasing water clarity (secchi) and electrofishing effort. The group mean occupancy probability was higher for large-bodied fishes at 0.71, with individual species' occupancy ranging from 0.19 to 0.97 (Appendix A, Tables 26-27).

Similar to juvenile fishes, some occupancy relationships were shared between large-bodied fishes; however, others varied by species (Appendix A, Table 27; Appendix A, Figures 34-37). All species were negatively related to increasing drainage area, distance from dam, and sinuosity. Large-bodied species displayed variable relationships with salinity, elevation, width-to-depth ratio, presence of blackwater in the reach, and discharge. All species other than Silver Carp were negatively related to increasing salinity. All species other than Bighead Carp and Paddlefish were negatively related to increasing elevation. Most species were negatively related to presence of backwater within a reach; however, all three buffalo species, all non-native Carp (Bighead, Common, Grass, Silver), Green Sunfish, Orangespotted Sunfish, Paddlefish, River Carpsucker, and Shovelnose Sturgeon were positively associated with reaches that contained >1% backwater habitat. Species' relationship with channel shape tended to be evenly split with.

However, some species within the same genus exhibited variable relationships. For example, Spotted Gar, Bigmouth Buffalo, and Smallmouth Buffalo were associated with narrower deeper channels, whereas Longnose Gar, Shortnose Gar, and Black Buffalo tended to be more associated with shallower, wider channels. Blue Sucker, Channel Catfish, Flathead Catfish, Shovelnose Sturgeon, and many *Lepomis* and piscivore species were also associated with shallower, and wider channels. Most species were positively related to median discharge of the season. However, Alligator Gar, Bighead and Silver Carp, all three buffalo species, Blue Catfish, Paddlefish, and Spotted Bass were negatively associated with discharge. Lastly, the

grouping factors of segment and year accounted for additional variance (0.239 and 1.234, respectively).

Adult fish habitat

For modeling adult river fishes, we included data collected from 46 sites, 190 sampling events, and 25 large-bodied species in the model. We omitted some sites due to missing physiochemical covariate information and omitted some of the large-bodied species that had extremely low naïve occupancy (e.g., Grass Pickerel). The species that were included in the model included the families Acipenseridae, Catostomidae, Centrarchidae, Cyprinidae, Ictaluridae, Lepisostedae, Moronidae, Polyodontidae, and Sciaenidae. The model successfully converged and displayed adequate fit (Appendix A, Table 26). All model parameters displayed appropriate chain mixing with $\hat{R} < 1.1$. The OM displayed adequate dispersion of posterior values ($\hat{c}$ of 1.004), and adequate goodness-of-fit with a Bayesian p-value of 0.471.

Large-bodied fishes displayed variability in both the probability of detection and occupancy. Species detection probability ranged from 0.17 to 0.90 with a group mean of 0.51 (Appendix A, Tables 27-28; Appendix A, Figure 27). As expected, species' detection increased with both increasing water clarity (secchi) and electrofishing effort. The group mean occupancy probability was higher for large-bodied fishes at 0.71, with individual species' occupancy ranging from 0.19 to 0.97 (Appendix A, Tables 27-28).

Similar to juvenile fishes, some occupancy relationships were shared between large-bodied fishes; however, others varied by species (Appendix A, Table 28; Appendix A, Figures 28-31). All species were negatively related to increasing drainage area, distance from dam, and sinuosity. Large-bodied species displayed variable relationships with salinity, elevation, width-to-depth ratio, presence of blackwater in the reach, and discharge. All species other than Silver Carp were negatively related to increasing salinity. All species other than Bighead Carp and Paddlefish were negatively related to increasing elevation. Most species were negatively related to presence of backwater within a reach; however, all three buffalo species, all non-native Carp

(Bighead, Common, Grass, Silver), Green Sunfish, Orangespotted Sunfish, Paddlefish, River Carpsucker, and Shovelnose Sturgeon were positively associated with reaches that contained >1% backwater habitat. Species' relationship with channel shape tended to be evenly split with. However, some species within the same genus exhibited variable relationships. For example, Spotted Gar, Bigmouth Buffalo, and Smallmouth Buffalo were associated with narrower deeper channels, whereas Longnose Gar, Shortnose Gar, and Black Buffalo tended to be more associated with shallower, wider channels. Blue Sucker, Channel Catfish, Flathead Catfish, Shovelnose Sturgeon, and many *Lepomis* and piscivore species were also associated with shallower, and wider channels. Most species were positively related to median discharge of the season. However, Alligator Gar, Bighead and Silver Carp, all three buffalo species, Blue Catfish, Paddlefish, and Spotted Bass were negatively associated with discharge. Lastly, the grouping factors of segment and year accounted for additional variance (0.239 and 1.234, respectively).

Environmental conditions

Environmental conditions varied among sites, regions, and years. River conditions were drastically different between 2021, 2022, and 2023 (Appendix A, Figure 38). Discharge was much higher at many of our sites in 2021 for much of the year compared to 2022 and 2023 (i.e., a drought years). Physicochemical conditions varied between sites and regions (i.e., OK Red, AR Red, and tributaries), with the strongest differences between the tributaries and the mainstem Red River (both AR and OK sections) (Appendix A, Table 29). As expected, drainage area increased, on average, from upstream to downstream portions of the basin from tributaries (3,280 km²) through the OK Red (115,170 km²) to the AR Red (128,723 km²). During the occupancy season, sites in the Oklahoma portion of the Red River had a shallower maximum depth (2.53 m), when compared to both the AR Red (3.82 m), and tributaries (3.09 m). Tributaries generally comprised deeper, narrower river channels, whereas the mainstem Red River had wider and shallower channels. Additionally, the Arkansas portion of the Red River was deeper and narrower when compared to Oklahoma. The tributary reaches comprised higher limestone composition (9.46 %) than both portions of the Red River (OK: 2.46% and AR: 2.48%). Tributaries had lower average salinity levels (0.31 ppt) when compared to the Red River (OK – 0.56 ppt and AR – 0.38 ppt). On average, sinuosity and LDI were more similar among

regions than other environmental conditions. Detection conditions were also more similar between regions than occupancy covariates. However, water clarity was much higher in the Oklahoma portion of the Red River than both the Arkansas portion and tributaries.

Objective 3. Invasive bigheaded carp movement

Active and passive tracking

We detected 82% of our tagged carp. We conducted 171 active tracking events and obtained 1,519 detections from Bighead Carp and 2,596 detections from Silver Carp using active and passive telemetry. We detected 41 individuals at least twice (i.e., a minimum of two detections were needed to record a movement by an individual for the GAMM analysis). Eight of our tagged fish were presumed to have died or shed their tag (Appendix A, Table 8).

Bighead Carp and Silver Carp exhibited variability in their movement magnitudes. Bighead Carp had a mean movement distance of 2.37 km (SD: 18.05, range: 0-336.57) and mean movement rate of 0.46 km/d (SD: 1.8, range: 0-37.89). Silver Carp had a mean movement distance of 1.01 km (SD: 9.61, range: 0-206.99) and mean movement rate of 0.36 km/d (SD: .95, range: 0-18.02). The greatest movement distance (336.57 km) was recorded by a female Bighead Carp during our summer season (Appendix A, Table 30).

Detections of tagged fish varied across our SUR array. The greatest number of detections came from our SURs located in oxbows and tributaries (i.e., especially those areas close to the OK-AR border, Kiamichi River, and Muddy Boggy Creek). Several SURs had no detections during their deployment including the gates located at the upper extent of the catchment about 17 km downriver of Denison Dam, the Sulphur River, and Big Pine Creek. This does not mean that these areas are not used, but rather our tagged fishes were not using these areas. SURs located at the southern extent of our study area (i.e., AR-LA border) recorded only a few detections, but we acknowledge some tagged fish could avoid detection at higher flows.

Movement pattern analysis

Our initial model comparisons indicated support for not including autocorrelation structures, whereas including smoothing terms for our parameters was supported. For the distance model, the simplest model with no autocorrelation structure ($AICc = 21181.78$, log-likelihood = -10586.89, $df = 4$) had the most support compared to the same model with an autocorrelation structure ($AICc = 21190.59$, log-likelihood = -10590.29, $df = 5$). For our movement rate model, the simpler model ($AICc = 6698.372$, log-likelihood = -3345.181, $df = 4$) also had more support than the more complex model ($AICc = 6697.675$, log-likelihood = -3343.83, $df = 5$). Although the $\Delta AICc$ was <2 between the two models, we choose the simpler model with no correlation structures for all future models (Wood 2017). For both movement models, exploratory models with all terms containing a smoothing term had the greatest support compared to other models (Appendix A, Table 31).

Our top-ranked model of movements by carp contained an additive effect of CV of discharge and an interactive effect between CV of water temperature and discharge (Appendix A, Table 32 and Table 33). Our top ranked model explained 71% of the deviance in these data. The random effect of individual explained 9% of the deviances in these data. Our model validation indicated reasonable results with ($GAMscale = 162.52$) and ($CV-mse-GAM = 165.13$). This similarity in estimates indicated that the model generalized well to new data and is therefore, not overfit. The model was $>2 \Delta AICc$ away from all other models indicating this model had the greatest support among those considered. Both species moved at greater distances at higher CV of discharge values (Appendix A, Figure 32). The interactive effect between water temperature variability and mean discharge showed varying effects of discharge at different water temperatures variability. We divided discharges based on two standard deviations below and above the mean to show the effect of CV of water temperature on movement distances at low, medium, and high discharge. During low and medium discharge conditions, water temperatures variability had a minimal effect on movement distances. During high discharges, movement distances increased at greater water temperature variability values (Appendix A, Figure 33).

Our top ranked movement rate model included an additive effect of CV of discharge and an interactive effect between CV of water temperature and discharge (Appendix A, Table 33 and Table 34). This model explained 39.2% of the deviance in these data. The random effect of

individual explained 12% of the deviance in these data. Our model validation indicated reasonable fit (GAMscale = 1.6819) and (CV-mse-GAM = 1.6880). The similarity in estimates indicated that the model generalizes well to new data. This model was $>2 \Delta AICc$ away from all other models that we considered. Carp moved at higher rates during higher values of CV of discharge (Appendix A, Figure 34). The interactive effect between CV of temperature and mean discharge indicated varying effects water temperature variability at different discharge conditions. We divided discharges based on two standard deviations below and above the mean to show the effect of CV of water temperature on movement distances at low, medium, and high discharge. During low discharges, movement rates were higher at low to intermediate values of variability, during average discharges, water temperature had little effect on movement rate, and during high discharges, movement rates were also greater at low to intermediate values of water variability. (Appendix A, Figure 35).

Multistate analyses

The transition probabilities varied between our two states and over time. Transition probabilities from state A (tributary/backwater) to state M (mainstem river) were the highest between t1 and -t2 and t4 and -t5 and the lowest between t2 and -t3, t3 and -t4, and t5 and -t6 (Appendix A, Figure 36). Transition probabilities from state M to state A were highest between t1 and t2, t4 and -t5, and t5 and -t6 and lowest between t2 and -t3 and t3 and -t4 (Appendix A, Figure 37). The level of uncertainty was higher for transitions estimates from state M to state A. This uncertainty also occurred in the probability of remaining in state M (Appendix A, Table 35). Average apparent survival in state A was higher than in state M (Appendix A, Table 35). Recapture probability of our tagged fish was also variable through time. Recapture probability of our tagged carp was lower in state M with higher levels of uncertainty.

Results – Sulphur River

Sulphur River hydrology and environmental variables

Water quality conditions were generally similar among waterbodies within each year with respect to temperature and water clarity (Appendix B, Table 3). Water was generally turbid

across all waterbodies and ranged from 0.25 m in the Red River to 0.49 m in Wright Patman Lake (Appendix B, Table 3). Likewise, water temperatures were within a few degrees among sites while sampling (Appendix B, Table 3). Notably, conductivity in the Red River was more than double that of the Sulphur River and Wright Patman Lake (Appendix B, Table 3).

The flow regime of the lower Sulphur River was highly impacted by dam operations at Wright Patman Lake. The Sulphur River existed primarily in high (flood) or low flow conditions during the 2021 sampling season (Appendix B, Figure 2). During the high and low flow periods, gauge heights were relatively stable because of the dam. The transition from spring high flow to summer low flow conditions in 2021 was accompanied by a four-fold reduction in stage height, dropping from ~30 ft to ~7 ft over several days (Appendix B, Figure 3, note differences in panels). Along the Sulphur River, we noted that side channel habitats and access to the dam emergency spillway channel were disconnected from the main channel during low flow conditions (Appendix B, Figure 4, contrast a & b; Appendix B, Figure 4, contrast c & d). During the 2022 sampling season, the Sulphur River was in low flow conditions for a majority of the sampling season. Low flows were likely the result of drought conditions and maintenance on the dam. Flow conditions in 2023 were more varied and included high and low flows throughout the year (Appendix B, Figure 2).

Objective 1. Invasive bigheaded carp population assessment

The Sulphur River was sampled using a combination of gear types that varied based on river flow conditions during 2021, 2022, and 2023. A total of 22.3 (2021), 26.5 (2022), and 23.2 (2023) hours of electrofishing pedal time were spent capturing fishes. An additional 4.6 (2021), 7.3 (2022) and 4.6 (2023) hours of pedal time were spent herding fishes into gill nets with the electrofishing boat. A small number of gill nets were set across both years when river flows were suitable (8 in 2021; 16 in 2022; 18 in 2023). Other gill net sets were damaged or lost during high flows. Hoop net sampling started during periods of intermediate and low flows during both years. A total of 684 hours (2021, 284 hrs; 2022, 400 hrs) of soaking time was accumulated across both years. Hoop nets were not deployed in 2023 because they were found to be ineffective in previous years. In the latter half of the season (August – October), mini-fyke nets were deployed at a subset of sites to capture small-bodied fishes that were otherwise missed in hoop and gill nets because of size selection.

Capture of bigheaded carp occurred only in 2022 and 2023 (Appendix B, Table 4). Two Silver Carp were collected via boat electrofishing in June and September of 2022 (Appendix B, Table 4). The first Silver Carp was collected in the Sulphur River upstream of the confluence. The second Silver Carp was collected in the Red River near a wing dam structure just downstream of the Spring Bank boat ramp (downstream of Sulphur-Red confluence). During 2023, six additional Silver Carp were captured at the Red River sampling site downstream of the Spring Bank boat ramp (Appendix B, Table 4). All Silver Carp were large bodied and averaged 929 (SD, ± 76) mm and 9.0 (± 1.8) kg. Moreover, all Silver Carp were adults with ages ranging from 8 to 14 years old (Appendix B, Table 4). Throughout the project period, we observed multiple Silver Carp jump out of the water near the Sulphur-Red River confluence—in both rivers—in 2021, 2022, and 2023. No jumping behavior was observed further upriver at the other sites. No juvenile bigheaded carp were captured while electrofishing or mini-fyke netting during the project period (Appendix B, Table 5-9). Additionally, no Bighead Carp have been captured or observed. Because all Silver Carp were captured via boat electrofishing, comparison of catches with other gears was not possible.

Objective 2. Native fish assemblage assessment

The Sulphur River had a diverse community of fishes that included a wide range of feeding guilds (e.g., planktivores, benthic invertivores, generalist omnivores, predators) and body sizes (Appendix B, Figure 5; Appendix B, Tables 5 - 9). We documented a total of 52 fish species from 17 Families throughout the project period. The fish community was primarily comprised of Dorosomatidae (shads, 24% of all individuals), Centrarchidae (sunfishes, 19.1%), Catostomidae (catfishes, 19.0%), and Lepisosteidae (gars, 11.5%). Additionally, we regularly captured invasive Common Carp ($n = 102$) and occasionally captured Grass Carp ($n = 8$) along the Sulphur River during the study. Most Common Carp and Grass Carp (94%) were captured while boat electrofishing. Across all years, we found that pulsed-DC electrofishing ($n = 51$ species) captured the most fish species, followed by gill netting ($n = 21$ species), herding ($n = 20$ species), and hoop netting ($n = 14$ species). These differences among gears were due, in part, because electrofishing was more effective at capturing smaller bodied fishes that would not be captured in the larger mesh panels of the gill net or hoop net (Appendix B, Figure 6). For example, pulsed-DC electrofishing captured 18 fish species <100 mm that were not observed in gill nets,

herding, or hoop nets, indicating size-selective capture bias (Appendix B, Figure 7).

Electrofishing was able to capture fish >1000 mm (e.g., Grass Carp, Longnose Gar, American Paddlefish) as well as fish <100 mm (e.g., minnows, shiners). Notably, any fish species captured by herding, gill netting, or hoop netting were also captured by boat electrofishing (Appendix B, Figure 7).

Catch rates (CPUE) varied among gear type and species throughout the project (Appendix B, Figure 8; Appendix B, Table 10). We observed pronounced differences in total catch rates among gears. For example, pulsed-DC boat electrofishing consistently captured greater total numbers of fish when compared to herding, gill netting, and hoop netting in the Sulphur River. Although sampling effort differs between active and passive approaches, our findings indicate that electrofishing appears to provide a better representative sample of the fish community by providing higher catches of greater numbers of species across a broader body size spectrum. Moreover, this trend was consistently observed during all sampling events (Appendix B, Figure 8). Comparison of total catch rates confirmed that pulsed-DC electrofishing captured greater numbers of fish when compared to gill nets, herding, and hoop nets (Appendix B, Figure 9; Kruskal-Wallis Test; Chi-square = 163.2, $p < 0.0001$). Post hoc pairwise comparisons further revealed that electrofishing was greater than all approaches (Wilcoxon method, $p < 0.05$), hoop netting had lower catches than all other gears (Wilcoxon method, $p < 0.05$), and gill netting and herding had similar catch rates (Wilcoxon method, $p = 0.32$). Mini-fykes were not included in the comparison because they focused on small-bodied fish taxa. Catches in mini-fykes were highly variable among sets and dominated by Mosquitofish, *Lepomis* spp., Emerald Shiner, and Inland Silversides.

Given the strong differences in river flow conditions, we analyzed the 2021 catch data because it gave the cleanest distinction between low and high flows. We did this to examine the potential impacts of dam operations on the Sulphur River fishery. We specifically contrasted the period of high flows (May – July) to the period of low flows (Aug. – Nov.). We observed that the fish community composition and structure exhibited a pronounced shift between the two flow conditions (Appendix B, Figure 10), and that these shifts were significant (MRPP, $p < 0.001$). Differences are likely a combination of capture/detection efficiency, seasonality, and river flow regulation.

Relative weight indices like body condition are useful for tracking responses of fish species. We developed linear models of length-mass relationships for Gizzard Shad, Bigmouth Buffalo, Smallmouth Buffalo, Black Buffalo, Longnose Gar, River Carpsucker, Flathead Catfish, and Blue Sucker (Figure 11). Additionally, we validated the Bigmouth Buffalo and Gizzard Shad models by comparing our results with the models reported in Irons et al. (2007). We observed that predictions from the Bigmouth Buffalo models were very similar, deviating by only 0.25%, on average. The relationship between models was highly significant, with a slope of 1.0028, indicating strong congruence between models. We observed similar findings for Gizzard Shad, however predictions from each model were slightly more variable, differing by about 1.19%. Overall, fish body conditions for each species were normal and did not indicate poor health (i.e., condition values lower than expected).

Discussion

Objective 1. Invasive bigheaded carp population assessment

The detection, capture, and removal of bigheaded carp has been a management priority over the last decade in the Mississippi River Basin (Collins et al. 2015; 2017; ACRCC 2016; Rodgers 2019; Lederman et al. 2024). Assessing invasive species at their ‘invasion front’ is challenging but necessary because such information is needed to inform management decisions regarding monitoring and targeted removal efforts.

Juvenile bigheaded carp

Many age-0 fishes are difficult to detect in large river systems (Brewer and Ellersieck 2011), including Bighead and Silver Carp (Roth et al. 2020). Bigheaded carp are extremely difficult to sample (Wanner and Klumb 2009; Bouska et al. 2017; Roth et al. 2020) and detection was reported at approximately 38% in the presumably highly populated Illinois River basin (Coulter et al. 2018). We selected sampling gears following Collins et al. (2017), who found both minifyke nets and beach seines to be the most efficient for capturing age-0 bigheaded carp. However, we did not capture any age-0 bigheaded carp either due to extremely low sampling detection (i.e., possibly due to very wet conditions in 2021), or other influences. Camacho (2016), Collins et al. (2017), and Chick et al. (2020a) have reported stark differences in the successful collection

of larval and juvenile bigheaded carp in successive years. For example, Collins et al. (2017) collected 39,398 Silver Carp in 2014; however, they collected only 116 in 2015. During the same years, Camacho (2016) captured a higher density of eggs and larval fish in 2014 than in 2015. Our 2021 (i.e., extremely wet), 2022 (i.e., extremely dry), and 2023 (i.e., also dry) sampling seasons may be emblematic of extremely low capture years where adults chose not to reproduce (or reproduced further downriver). Because bigheaded carp in the lower Red River basin have not been documented at densities as high as the Upper Mississippi River, sampling inefficiencies may be exacerbated.

Sand-bed streams of the Central Great Plains, including the Red River are extremely dynamic and continuously shift over time (e.g., a backwater may be present during the wet months and absent during the dry months). Due to the constant shifts and extreme conditions associated with sand-bed streams, detection of fishes is quite variable and often imperfect (Mollenhauer et al. 2018). The extensive high-flow events observed in 2021 may have influenced our ability to successfully detect juveniles of both species of bigheaded carp. Alternatively, the low-flow conditions of 2022 and 2023 may have not been favorable conditions for bigheaded carp spawning. In July 2021, Red River median discharge was 318 m³/s (11,231 ft³/s), roughly 195 m³/s (6,800 ft³/s) higher than the 79-year median (USGS gage 07335500) (U.S. Geological Survey 2023). However, in July 2022 and 2023, Red River median discharge was 72 m³/s (2,546 ft³/s) and 89 m³/s (3,160 ft³/s), which is roughly 51 m³/s (1,813 ft³/s) and 34 m³/s (1,200 ft³/s) lower than the 79-year median (USGS gage 07335500), respectively. Discharge is assumed to be a spawning cue for bigheaded carp and both our seining efficiency and mini-fyke net effort may have been affected by high flows (though it is unlikely we would not have detected a single juvenile). Moreover, because bigheaded carp are pelagophils, their eggs may have washed much further downriver during these extremely high flows. Another possibility is the abnormally high and low flows created unfavorable spawning conditions. Lastly, some investigators have suggested water hardness may relate to eggs bursting under some conditions, but this idea has been discounted by others (Chapman and Deters 2009; Rach et al. 2010). Interactions with water hardness and other environmental factors on successful reproduction may be possible.

Adult bigheaded carp

Occupancy by both Bighead Carp and Silver Carp in the Red River reflects a catchment that has been invaded for quite some time. Typically, Bighead Carp is the first to invade followed by Silver Carp which then outcompete the former. Silver Carp occupancy was relatively higher (0.78 – 0.85) across the catchment when compared to Bighead Carp (0.53 – 0.78). These occupancy rates indicate that Carp likely inhabit reaches across the majority of the lower Red River catchment (i.e., though first reported in 2012, Patton and Tackett 2015). However, very few bigheaded carp were captured in the Sulphur River, despite past reports of Bighead Carp as far upstream as the Wright Patman Lake dam in Texas, and we hypothesize that most bigheaded carp move through the Red River rather than stray into the Sulphur River tributary. Estimating species distributions is an important aspect of fisheries management as it can be used to identify important locations for conservation or rehabilitation of imperiled species, or locations for targeted mitigation for invasive species (Anderson et al. 2012). Unfortunately, some of the same features leading to homogenization of the fish assemblage in the lower Red River (Mollenhauer et al. 2022) are also features that appear to benefit invasive bigheaded carp.

Although catchment-level, land-use disturbance was relatively constant across our study area, both species of bigheaded carp were associated with several instream habitat features that may reflect local disturbances. Across a broader geographic area, more cosmopolitan fish species in the basin were associated with land-use disturbances and altered flow regimes (Mollenhauer et al. 2022). We did not examine longer-term flow patterns due to the temporal scale of our study, and we did not relate bigheaded carp occupancy to land-use disturbances because the variability was minimal across our study area. However, several of the attributes we found related to bigheaded carp occupancy are related to local disturbances. Lower sinuosity reaches, for example, can reflect channelization or other degradations that result in a less complex channel (Lennox and Rasmussen 2016) and channel incision (i.e., deeper and narrow channels) (Rowe et al. 2009). Habitat complexity typically declines in areas where sinuosity is low and width-to-depth ratios reflect narrower and deeper stream channels. Degradation of natural riparian vegetation, bridge construction, and scouring associated with dams can cause erosion or armoring of stream banks, thereby increasing channel depth and these conditions tend to be associated with invasive species (Beschta and Platts 1986; Chen et al. 2010; Stein et al. 2013; Bueno et al. 2023). Altered flow regimes, common in the catchment (Mollenhauer et al. 2022),

also lead to degradation of instream habitat over time where complex, braided channels tend to become greatly miniaturized and disconnected from the floodplain (Brewer et al. 2016). The lower Red River has also been regulated to some degree using wing dikes and other structures to direct flow and increase channel depth (Matthews et al. 2005). Calkins et al. (2012) found that Silver Carp used river reaches with wing dikes and avoided those lacking wing dikes likely due to the creation of deeper water, but also the velocity refuges formed behind the dikes (Braun et al. 2016). Ironically, these human alterations are found lower in the catchment, but we did show some correlation between width-to-depth ratio and drainage area. Higher in the stream network, most of the major tributaries are dammed or have deep incised channels associated with erodible lands (Powers 2015). These areas are not managed using environmental flows and thus, except for periods when flood flows are released, several of the tributaries provide slow-moving, warm water that may provide important bigheaded carp refuge and feeding areas.

The disconnection between the floodplain and main channel in many reaches of the Red River catchment likely exacerbates the importance of tributary habitat and reaches containing backwaters to both invasive Bighead Carp and Silver Carp. We found Silver Carp to be positively correlated with chlorophyll-a concentrations, which may relate to their feeding strategy. Silver Carp are considered obligate phytoplanktivores, incidentally consuming zooplankton (Li et al. 2013; Ochs et al. 2019). Although variability in our measured chlorophyll-a concentrations was high, some of highest densities of chlorophyll-a concentrations in the lower Red River catchment were observed in tributaries (e.g., Choctaw Creek, Bois d'Arc Creek) (though not highly correlated with backwater reaches). Williamson and Garvey (2005) found that Silver Carp predominately consumed phytoplankton in the Mississippi River and proposed that Silver Carp used low-velocity habitats to maximize foraging opportunities. Both the lower tributaries in our study area and backwater habitat provide low-velocity habitats that would facilitate foraging opportunities during the warm-water period. Bigheaded carp association with low-velocity and off-channel habitats during the warm-water periods is common in many documented areas of the United States (e.g., Illinois River, DeGrandchamp et al. 2008; Wabash River, Coulter et al. 2016a). However, DeGrandchamp (2006) found Bighead Carp and Silver Carp avoided backwater habitats of the Illinois River and instead used main-channel margins during summer and autumn. Effectively monitoring these habitats over time will be beneficial to understanding future population changes.

Bigheaded carp population demographics

Both Silver Carp and Bighead Carp in the Red River catchment have body sizes (i.e., length-at-age) that are commonly associated with relatively recent or continued population invasions. No individuals of either species younger than 3 years of age were collected; however, the younger fish were relatively large with a mean back-calculated TL of 589 mm for Silver Carp and 557 mm for Bighead Carp at age 3. Coulter et. al (2018) found that individuals with greater body condition are more likely to be located on the fringe of the species distribution and are primarily responsible for expanding the species range. River fishes with higher body condition are generally more mobile (Kanno et al. 2023). Furthermore, rivers with robust populations of Silver Carp have relatively smaller fish. For example, Sullivan et al. (2021) found that the mean TL for Silver Carp ranged from 532 – 737 mm in the Missouri, Mississippi, Wabash, and Illinois rivers, whereas the mean TL was 879 mm in our samples from the lower Red River catchment. Additionally, TL for newly established populations of Silver Carp in the Mississippi River and Bighead Carp in the Missouri River ranged from 600 to 800 mm and 450 to 1,099 mm, respectively (Schrack and Guy 2002; Williamson and Garvey 2005).

We are now aware that carp can spawn in the Red River and recruitment appears similar to other systems. Silver Carp recruitment variability was relatively stable (RVI of 0.72), which is comparable to what is observed in other catchments such as the Missouri, Mississippi, De Moines, and Wabash rivers (RVI 0.66 – 0.95, Sullivan et al. 2021). This may be due to both fish consistently recruiting to the catchment from other river systems (i.e., Atchafalaya River) and recruitment in the Red River. Although we did not capture juveniles in 2021-2022 (Ramsey 2023), nor was it documented during our 2023 juvenile surveys, we did find that adults were successfully spawning (though we do not know what survival is post spawn). Fertilization rates by bigheaded carp can be quite low (e.g., 37%, Gonzal et al. 1987; Lenaerts et al. 2023). If sex ratios are skewed (as we found), fertilization rates may be low, but they are still reproducing. Moreover, bigheaded carp exhibit schooling behaviors (Murchy et al. 2017), and chemical cues associated with schools may be necessary for attracting females. If the populations are relatively low density compared to other populations, then they may currently lack emergent properties that facilitate high levels of spawning (though this is clearly unknown).

Bighead Carp and Silver Carp in the Red River catchment appear to live longer and grow larger than other populations. Silver Carp L_{∞} in the Missouri and Mississippi rivers ranged from 691 to 802-mm TL and Bighead Carp L_{∞} was 983-mm in the Mississippi River (Tsehaye et al. 2013; Ridgeway and Bettoli 2017), whereas Silver Carp and Bighead Carp in the lower Red River had a L_{∞} of 935 and 1218-mm TL, respectively (values reported are for bigheaded carp collected through 2022). This may be because older age classes were present in the lower Red River population, as Silver Carp maximum age was much higher in the lower Red River (i.e., 14 years old) than that typically seen in the Mississippi River basin (i.e., 7 years old) (Schrack and Guy 2002; Williamson and Garvey 2005). This is further highlighted by Silver Carp growth coefficient (k). The growth coefficient represents the speed at which fish length approaches the L_{∞} , with a higher k indicating faster growth (Quist and Isermann 2017). Although Silver Carp L_{∞} was higher than other populations, the rate of growth ($k = 0.31$) was like that of populations in the Mississippi and Illinois rivers (0.23 – 0.445, Tsehaye et al. 2013, Sullivan et al. 2021), whereas Bighead Carp growth rate ($k = 0.17$) was slower relative to Mississippi River populations (0.433, Tsehaye et al. 2013). However, several of the previous studies conducted on bigheaded carp in the Mississippi and Illinois rivers used different ageing structures (i.e., fin rays) which may underage bigheaded carp compared to lapilli otoliths (Birdsall et al. 2024, Appendix B). This may bias growth estimates, because growth models estimate parameters such as L_{∞} and k from length-at-age estimates.

Our results indicate lapilli otoliths for ageing and monitoring populations of both Bighead Carp and Silver Carp would be the best choice among hard structures even though between-reader-agreement (BRA) was lower than found in other fishes. Proper age estimates are critical for assessing any of these rates (Koenigs et al. 2013; Anderson et al. 2023). Determining the accuracy of an ageing structure can be difficult for invasive species using known-age fish or marginal increment analysis (Rugg et al. 2014; Anderson et al. 2023). Precision estimates can be used as a surrogate to determine the best structure to age fish when no structure has been validated (Campana 2001). Common precision metrics include BRA and the mean coefficient of variation (CV), where the highest BRA and lowest mean CV indicate the highest precision (Seibert and Phelps 2013). Between-reader-agreement was relatively low for lapilli otoliths of carp compared other species such as Walleye *Stizostedion vitreum* (BRA = 0.98), Largemouth Bass *Micropterus nigricans* (BRA = 0.91), Smallmouth Bass *Micropterus dolomieu* (BRA =

0.94), Yellow Perch *Perca flavescens* (BRA = 0.98), and Brown Bullhead *Ameiurus nebulosus* (BRA = 0.92) (Isermann et al. 2003; Maceina and Sammons 2006). Longer lived fishes are inherently more difficult to age compared to fishes with shorter life spans due to crowding of annuli, especially in warm-water systems when growth is more consistent (Quist and Isermann 2017). For example, Dunton et al. (2016) found that BRA for Atlantic Sturgeon *Acipenser oxyrinchus oxyrinchus* was 63% for fin spines and Labay et al. (2011) found that BRA for Blue Sucker was 50% for fin-rays.

Factors that increase water temperatures and stabilize flows may positively affect growth and recruitment for both species of bigheaded carp; however, pressures on water resources and declines in precipitation reducing flows may negatively affect growth by both species. Additionally, factors that stabilize discharge may negatively affect Silver Carp whereas factors that stabilize water temperature would inversely affect both species. Climate models predict that air temperatures will increase over the next several decades (Dixon et al. 2020; Pörtner and Roberts 2022). These increasing water temperatures throughout the catchment may lead to an environment that fosters increased growth and an extended spawning period for both bigheaded carp species (once successful). Based on a series of predicted models and reviewed data, feeding was observed by Silver Carp at 15-30 °C (Kolar et al. 2007; Cooke and Hill 2010), Bighead Carp at 20-30 °C, and observed or predicted spawning temperatures ranged 14-30 °C (see Table 2 of Cooke 2016). Pease and Paukert (2014) found that Smallmouth Bass growth would increase with warming water temperature due to climate change. Furthermore, McCann et al. (2018) found that Sea Lamprey *Petromyzon marinus* spawning occurred earlier in the year due to increased stream water temperature resulting in possible increased growth and survival of juveniles in the Great Lakes basin. The combination of warming water temperatures increasing bigheaded carp growth (assuming available food) and their observed tendency to supplant native species may exacerbate the invasive capabilities of these species. Additionally, growth for both species of bigheaded carp was negatively associated with discharge variability. Major impoundments exist on the mainstem Red River (i.e., Denison Dam) and many of the tributaries (i.e., Kiamichi, Muddy Boggy, Sulphur River) which lead to stabilized flows (Gibson et al. 2005; Wang et al. 2016; Zhang et al. 2017). Additional impoundments have recently been constructed or are planned in the catchment (e.g., Bois'd Arc Creek) (Payne et al. 2021), which may further decrease flow variability and lead to decreased growth for Silver Carp. Flow variability is also positively

associated with occupancy of several native species (Mollenhauer et al. 2022). However, the taxing of water resources in the Southern Great Plains and a slight reduction in precipitation is projected to decrease the overall duration and magnitude of flows (Brikowski 2008; Dixon et al. 2020; Pörtner and Roberts 2022). For example, Dallas, TX requires additional water resources from the Red River catchment, and Oklahoma City will also be diverting additional water from a tributary of the Red River (i.e., Kiamichi River) (Burch et al. 2020; Payne et al. 2021). This may result in a decrease in the consistency of year-to-year growth for bigheaded carps punctuated by increased growth during flood years in the lower Red River catchment.

Bigheaded carp growth and low mortality may be related to low fish density, high food availability, and decreased fishing mortality in the lower Red River. For example, Lorenzen and Endberg (2002) found that asymptotic length for nine teleost populations had an inverse relationship with species specific biomass density. Additionally, the lower Red River catchment may offer abundant forage which facilitates increased growth. Our chlorophyll-*a* concentrations were on average 32.97 µg/L in the Red River, whereas chlorophyll-*a* levels in the Mississippi River from 1998 to 2018 were over 20 µg/L only 12% of the time (Turner et al. 2022). Silver Carp exhibited lower natural mortality (0.33) than populations in the Mississippi River basin (0.68, Tsehaye et al. 2013). The demographic data described by Tsehaye et al. (2013) was derived using pectoral fin spines, which may have led to underestimating fish age and possibly overestimating mortality (Koenigs et al. 2013). Additionally, we calculated mortality for Silver Carp ages 6 through 14 whereas Tsehaye et al. (2013) mortality estimates were for fishes up to 7. However, the higher mortality observed in the Mississippi River basin may be related to density dependent mortality or lower fishing mortality compared to other river catchments. For example, Matte et al. (2020) found that mortality of Brook Trout *Salvelinus fontinalis* was positively associated with density. Bigheaded carp densities are currently perceived to be lower than many other rivers (though sampling indicates otherwise at some locations) and lower densities may improve overall survival. A commercial fishery persists in the Arkansas portion of the lower Red River, with incidental bigheaded carp bycatch. However, commercial harvest is not permitted in the Oklahoma or Texas portions of the catchment which may alleviate harvest pressure for these bigheaded carp populations (but also on native fishes as bycatch). High fishing mortality from commercial harvest and mitigation efforts persists in the Missouri, Mississippi, and Illinois rivers. However, in many cases, there is very limited evidence that removal efforts have resulted

in any change in overall population abundance or if they alter the reproductive potential in those populations (i.e., compensatory response).

Objective 2. Native fish species assemblage assessment.

Fish sampling

Throughout the course of this study, we documented 86 fish species throughout the lower Red River catchment, exclusive of the Sulphur River, and 52 fish species in the Sulphur River. Relatively few sampling efforts covering this spatial extent have been devoted to collecting data on the native fish assemblage within the lower Red River basin. From 1995 to 2001, Buchanan et al. (2003) sampled the Arkansas portion of the Red River and reported the collection of 72 fish species. Of the 72 species collected from 1995 to 2001, we documented 67 throughout our study area. In addition to the 67 species caught from Buchanan et al. (2003), we documented 19 unique species including: American Eel, Bigeye Shiner, Bighead Carp, Bluntnose Minnow, Bowfin, Chestnut Lamprey, Flier, Grass Pickerel, Highland Stoneroller, Mooneye, Plains Killifish, Quillback, Redspot Darter, Ribbon Shiner, Sand Shiner, Scaly Sand Darter, Silver Carp, Slenderhead Darter, and Weed Shiner. We did not detect five species that were reported in Buchanan et al. (2003), however, those five species were described by the authors as “Uncommon” or “Rare” relative to other species (Banded Pygmy Sunfish, Blackside Darter, Creole Darter, Freckled Madtom, and Mud Darter). One of the sampling techniques used by Buchanan et al. (2003) included rotenone application. Differences in the sampling efficiency between our study and that of Buchanan et al. (2003) may be due either to the latter’s use of rotenone, a method we did not use, or to the simple fact that these species are relatively rare. In the event of rapid population growth by bigheaded carp, our baseline data can be used to describe the proportions of native and invasive fishes in terms of relative abundance and biomass (Coulter et al. 2018). As part of the ecological baseline, we summarized their catch rates and

sizes to track how their populations may change in response to bigheaded carp populations through time.

This baseline native fish assemblage data will be important for monitoring changes in species assemblage related to bigheaded carp over time. In the event of rapid population growth by bigheaded carp, our baseline data can be used to describe the proportions of native and invasive fishes in terms of relative abundance and biomass (Coulter et al. 2018). As part of the ecological baseline, we summarized their catch rates and sizes to track how their populations may change in response to bigheaded carp populations through time. Body condition indices were developed for several recreational and ecologically important fish species (e.g., Bigmouth Buffalo, Smallmouth Buffalo, Black Buffalo, Blue Sucker, Flathead Catfish, River Carpsucker, Longnose Gar, and Gizzard Shad) during this study (for the Red River, see Vilchez et al. 2024, included as Attachment B; for the Sulphur River see Appendix B). The length-mass models provide a convenient baseline for tracking potential effects of invasive bigheaded carp if their populations increase and spread from the Red River. Interestingly, body condition data from the Sulphur River were surprisingly similar to the “pre-invasion” models developed and used to quantify reductions in body condition of Bigmouth Buffalo and Gizzard Shad (Irons et al. 2007). Our comparison indicates that these length-mass models developed for the Red and Sulphur Rivers over the course of this study should be sufficient to detect changes in native fish body condition if bigheaded carp reduce plankton resource availability in the future.

Juvenile nursery habitat

Our results from juvenile modeling indicate that nursery habitats in large rivers are largely context dependent, even for closely related species (see Ramsey et al. 2024 for more in-depth discussion; Attachment C). Nursery habitats in the lower Red River can generally be described as reaches containing off-channel slackwater habitat, having deep pools, with shallow average thalweg depths, further away from dams with lower percentages of limestone geology. Although taxonomically similar species are often thought to use similar habitats that is not always the case (Lowe-McConnell 1987). For example, we found that Green Sunfish and Redear Sunfish were positively associated with wider, shallower channels, whereas Bantam Sunfish, Bluegill, Longear Sunfish, and Orangespotted Sunfish tended to occur in reaches with narrower and deeper channels. Although these species are not of conservation concern, it demonstrates the perils of

assuming closely related species share habitat choices because they have other shared traits (e.g., body morphology, feeding strategies). Changes in channel slope also appeared to provide context dependency to nursery habitats where fishes in the genera *Ictalurus*, *Ictiobus*, *Pomoxis*, *Lepomis*, and *Dorosoma* all had species with opposing relationships with segment slope. Increased slope can lead to stronger water velocities (Gordon et al. 1992), create more heterogeneous water depths (Troutman et al. 2007), and diversify the channel units within the river segment (Harvey and Bencala 1993). It appears that more common species may be more tolerant of homogeneous conditions (e.g., Spotted Bass and Bluegill); however, rarer species (e.g., Skipjack Herring) may benefit from the higher water velocity that creates more diverse habitats (Marchetti and Moyle 2001; Walters et al. 2003). Although the mechanisms for these associations are unknown, these varying relationships within closely related species indicate that river slope and width-to-depth ratio relate to different nursery habitat for assemblage members (see Ramsey et al. 2024, included as Attachment C).

Adult fish habitat

Although our results from modeling large-bodied fish habitat indicate there is variation in habitat associations among species, we observed some relationships that were shared between sympatric species. Fish species are often aggregated into guilds when conducting assemblage studies to simplify analyses assuming that species within the same guild will have similar responses (Benoit et al. 2021). Although we did not run our analyses by aggregating fish species into guilds, we observed that some species within similar functional groups tended to have similar habitat associations (though certainly not all of them). For example, planktivores and detritivores were more associated with low-velocity backwater habitat, whereas invertivores, omnivores, and piscivores tended to be associated with wider, shallower, and presumably higher velocity habitat within the main river channel. All three buffalo species, River Carpsucker, and Paddlefish were positively associated with sites containing more backwater. This is also the habitat that all four of the non-native species (Bighead Carp, Common Carp, Grass Carp, and Silver Carp) were most positively associated with. Orangespotted Sunfish and Green Sunfish were the only invertivores that had a positive association with backwater, though the effect size for Green Sunfish was nearly zero (0.009). Backwaters provide access to new food resources and can allow fishes to escape swift currents and limit energy expenditure (Junk et al. 1989; Power et al. 1995;

Williamson and Garvey 2005). Paddlefish and Bigmouth Buffalo filter feed in the water column and frequently use backwater habitats where it is hypothesized that they can feed more efficiently (Minckley et al. 1970; Sampson et al. 2009). The substrate in these backwater areas consists of fine silt and clay deposits that may offer foraging resources that are less available in the main channel for benthic detritivores such as Smallmouth Buffalo and River Carpsucker (Quist and Spiegel 2012). Backwater habitats are also probably used by some of these fishes for spawning and subsequent nursery habitat (Quist and Spiegel 2012; Dutterer et al. 2013) as we found both adult and juvenile Green Sunfish and Orangespotted Sunfish to be associated with them. Although we sampled Alligator Gar at sites with more backwater in the mainstem Red River, they had weak negative association with them. Instead, they were commonly sampled in tributary sites (observed at 61% of all tributary sites included in the model vs 45% Red River sites). Alligator Gar often occupy tributary habitat in other systems, and it is especially important during the spawning season when they were found to use tributary habitat more often than mainstem or backwater oxbow habitats (Buckmeier et al. 2013; Kluender et al. 2017). Although Black Buffalo was associated with sites containing backwaters, the only buffalo species positively shallower, wider reaches. Black Buffalo diet is more diverse than Bigmouth Buffalo and Smallmouth Buffalo, consuming more clams and crustaceans (Pflieger 1997). Black Buffalo occur more often in strong currents than the other buffalo species, (Pflieger 1997), which is consistent with our findings of them also having a positive association with shallow, wide habitat where the current is presumably stronger. Shovelnose Sturgeon, similarly, was weakly associated with reaches containing backwater habitat but more strongly related to wide, shallow reaches that had higher discharge. Numerous other species also had a positive association with these areas, including nearly all of the piscivore and *Lepomis* species, and Blue Sucker, another fluvial specialist. Shovelnose Sturgeon use shallow (1.0 - 2.0 m) water depths, over sand substrate, and relatively low velocities in the Kansas River at certain times of the year (Quist et al. 1999). We observed similar behavior with Shovelnose Sturgeon in the lower Red River basin, as cross-sectional depths at sites where we detected Shovelnose Sturgeon averaged between 0.8 and 2.6 m. Blue Suckers are also associated with shallow water depths and areas of swift water velocities in other systems (Acre et al. 2021; Neely et al. 2009). Channel Catfish and Flathead Catfish were observed more frequently in shallower habitats of other rivers, despite their reputation of seeking out the deep pools (Daugherty and Sutton 2005; Braun and Phelps 2016).

The high degree of habitat heterogeneity in portions of the lower Red River offers a unique opportunity to study a complex of niches and the species that occupy them. The river is typified by both pools within the thalweg throughout the year as well as sections of shallow braided channels during low flow. In some areas, there are more homogenous habitats where abundant wing dikes and rip-rap lined banks direct flow to maintain deeper pools, while also creating backwater areas behind them. Additionally, large oxbow lakes are also present that become laterally connected to the mainstem river during high flow periods, allowing for faunal exchange between the two habitats. Our results and others indicate that these areas are particularly important for juvenile fishes and adult planktivore and detritivore species (Minckley et al. 1970; Sampson et al. 2009; Quist and Spiegel 2012; Dutterer et al. 2013). However, all bigheaded carp species had similar positive associations with these slackwater habitats and native fishes occupying these areas could be more susceptible to competition or displacement (Wang et al. 2018). Other stretches of the Red River catchment contain little to no artificial channelization, allowing for more dynamic habitat that is typically shallower with a wider channel (i.e., closer to a more natural channel in the Southern Great Plains). These areas were particularly important for fluvial specialists such as Shovelnose Sturgeon and Blue Sucker, but were also important for adult *Lepomis* species, catfish, White and Spotted Bass, and Freshwater Drum. Maintaining these areas where disturbance (e.g., channelization) is limited and the habitat is more indicative of a natural Southern Great Plains stream could facilitate high native species diversity (Galat et al. 2005). By quantifying these reach scale habitat parameters, along with coarse scale metrics, we were able to identify numerous associations between habitat and fishes in the lower Red River basin that may be important for species conservation when considering the overlap between invasive bigheaded carp habitat use.

Accounting for incomplete detection is particularly important to assess changes in distributions or occupancy over time, both of which are important when invasive species that may compete for food sources have been introduced. Accounting for detection is also important when surveying for smaller-bodied, rarer, and cryptic species within aquatic ecosystems (Albanese et al. 2011; Schloesser et al. 2012; Wedderburn 2018), but may also help understand fish-habitat relationships of more common species (Sliwinski et al. 2016; Guillera-Arroita 2017). In fact, some species are quite difficult to detect, but are quite common across a catchment (Mollenhauer et al. 2022). Sampling fish assemblages is increasingly difficult as river size, flow,

and turbidity increase (Flotemersch et al. 2006). Thus, the importance of accounting for detection when sampling juvenile and adult native fishes in the lower Red River basin is evident from the low detection rates of some species with relatively high occupancy. (e.g., Alligator Gar, Silver Chub, Spotted Gar, Logperch, and Redear Sunfish). Without accounting for detection probability, occupancy estimates would have been much lower than the modeled outcome (Mackenzie et al. 2009) and relationships with the habitat parameters would be altered (see also Gerber et al. 2020). By accounting for detection, we were able to produce a less biased estimate of true occupancy within the lower Red River catchment.

Objective 3. Invasive bigheaded carp movement

The variation among movement patterns in our tagged carp indicates there is a subset of highly mobile fish and many that move very little, and these two movement characteristics may be displayed by the same individual over time. We found both species' average movements were <3 km over our study period indicating a relatively low-movement population. Whereas other studies found carp to move at larger magnitudes, on average. For example, mean movement distances of 12.4 km and 4.13 km for Silver Carp and Bighead Carp, respectively, in the Wabash and Illinois rivers (Peters et al. 2006; Coulter et al. 2016). However, a subset of our tagged fish also made long distance movements (maximums were 336.57 km and 206.99 km for Bighead Carp and Silver Carp, respectively). Among the six individuals that had mean distances above 10 km, all recorded minimum distances <1 km and four had mode movement distances of <5 km. The variability within individual movements indicates the ability to be mobile or relatively sedentary may not be static within the population (Peters et al. 2006; Booth et al. 2014). This contrasts with other studies that found individual carp to have relatively consistent mobile or sedentary characteristics (Coulter et al. 2022b). Successful invasive species typically have highly mobile capabilities which allow them to disperse into new environments and compete with native species (Lockwood et al. 2005; Roy et al. 2013; Prechtel et al. 2018).

Many movement studies show that individual variability is important. Individual variation in fish movement behaviors is assumed to contribute to population-level robustness and adaptability (e.g., gene flow, recolonization, species turnover; Albanese et al. 2004; Radinger and Wolter 2014). It is not uncommon to have a large range of variability in movement studies

(Dance and Rooker 2015; Becker et al. 2016; Hughes et al. 2022; Pedaccini et al. 2023). Our individual variability accounted for only 9% of deviance in fish movement distance model and 12% of deviance in our movement rate model. Harrison et al. (2019) demonstrated by tracking five species of large-bodied fishes that individual movements within a species were often more variable than movements among species. Other studies on invasive species tend to demonstrate highly variable movements among individuals (Peters et al. 2006; Rasmussen et al. 2012; Juetten et al. 2014). Other studies of Bighead Carp and Silver carp have shown large variations in individual movement tendencies. For example, most individual Silver Carp had consistent total home range sizes of <26 km or >102 km in the Wabash River (Prechtel et al. 2018). Also on the Wabash River, Coulter et al. (2022) found mobile individuals consistently moved at greater distances than sedentary individuals. They found that mobile individual's movements were more related to environmental conditions compared to sedentary individuals. In addition to low individual variability among our tagged fish, there were also only a few that made larger movements.

The spatial patterns of tagged carp in our study indicated minimal large-scale migrations by individuals, especially upriver. The positioning of our SUR array allowed us to detect possible large-scale migration to the outer extents of our study area. SUR gates at both boundaries of our study area recorded zero or few detections. Varying detection efficiency and range of SURs due to different environmental conditions and mooring designs could explain the lack of detection at these gates (Winter et al. 2021; Carlson et al. 2023). In other words, it is possible that lower detection on SURs at higher flows missed fish that entered or returned from these areas. However, given the duration that our gates were deployed and the detection of 82% of our tagged carp within the study area, it is unlikely that large numbers of tagged fish went undetected at these gates. Our SUR gate ~17 rkm below Denison Dam recorded no detections throughout the study period though we know that carp are captured above that location. Individuals may develop their own home ranges where they spend most of their time and these differences may relate to where the fish were tagged (i.e., fish tagged at the dam would be more likely to stay near the dam). Carp have the capacity to travel large distances especially when making spawning migrations (Lubejko et al. 2017; Whitley et al. 2019) meaning this SUR gate was within a reasonable distance from all tagging locations. Large movements were recorded by some individuals, but we did not record any carp moving further upstream than Bois d'Arc Creek

(but with the same caveats). Past studies in the same study area found both species occupy the entire catchment (Birdsall 2023), and USFWS has tagged carp in the upper part of the basin that we also never detected moving downstream past our SUR gate. The most upstream fish that we tagged were located in Choctaw Creek, just downriver of our uppermost SUR. Interestingly, we never detected those two fish again and never detected any fish moving upstream of that location. Collectively, this suggests that carp may settle into specific areas and limit their overall home range. Smaller home ranges used by more sedentary individuals may reflect ideal habitat in a smaller area (Prechtel et al. 2018). Stanton et al (2024) and Coulter et al. (2018a) both found carp were less likely to move from the pools they were originally tagged in within Illinois River; this type of movement that is isolated to species locations has been found for other riverine species and in other geographic locations (e.g., Flathead Catfish, Vokoun and Rabeni 2005; Silver Carp; Werner et al. 2023; White Perch, McGrath and Austin 2009).

Our results contrast with other studies that show intense directional movements during spawning conditions (Lubejko et al. 2017; Vallazza et al. 2021). Our SUR gate at the lower extent of our study area (AR-LA border) only recorded three detections from two individuals. Although our study area is not biologically meaningful to fishes (i.e. political boundary), it appears that the fish we tagged may have home ranges that are primarily within the study area. The proportions of resident and immigrant fish varies in other populations and is thought to be influenced by habitat and limitations to movement. Werner et al. (2023) found 54% of adult Silver Carp were resident fish that were born in and had never left the Kansas River, whereas transient individuals mostly traveled short durations into the Missouri River. Barshinger et al. (2024) found that Silver Carp in the Arkansas River had minimal reproductive contributions to the lower Mississippi River catchment indicating large movements were made from the White River and Mississippi River into the Arkansas River for reasons other than spawning. Although successful hatch has not been documented in the lower Red River (Ramsey et al. 2024), we did determine both Bighead Carp and Silver Carp were successfully spawning (i.e., present of empty follicles in ovaries) in the study area via histological analyses (Brewer et al. Unpublished data). This is supported by the moderate movements during conditions that have been associated with spawning and the minimal emigrations out of the study area; behaviors that suggest both species are likely attempting to spawn within the study area. Future efforts would benefit from tagging carp across a broader range of the Red River to determine population home ranges. Moreover, a

longer tracking period may reveal that these fish move under certain environmental conditions outside of what we experienced over our study extent (e.g., following a drought when recolonization would be needed).

Our top movement distance and movement rate model contained relationships consistent with our variability hypothesis. Our results indicated nonlinear relationships between movement and these abiotic conditions. Discharge and temperature variability has been related to fish movement in other studies (Taylor and Cooke 2012; Coulter et al. 2016; Olden and Naiman 2010; Morash et al. 2021). Discharge variability is a key aspect of the flow regimes for many large river fishes (Reinfelds et al. 2013; Carpenter-Bundhoo et al. 2023). Within their native ranges, flow is very important for carp to complete their life cycles (Duan et al. 2009; Li et al. 2013; Fang et al. 2022). Specific flow and temperature conditions are needed for both species to spawn in native ranges (Chapman 2006; Chapman et al. 2016). Changes to these thermal and flow regimes have negatively affected their abundances in the Yangte River (Yu et al. 2018). Changes to dam operations on the mainstem Red River and major tributaries could affect flow variations and thus limit carp movement but would also likely negatively affect the native fishes with similar flow relationships (i.e., Cyprinids Worthington et al. 2018; Mollenhauer et al. 2022; or Paddlefish Tripp et al. 2019).

Our multi-state analysis results point to important periods when carp transition between major habitats. Carp were more likely to transition into different habitats across time intervals with increasing water temperatures and more variable discharge. Intervals that contained decreasing water temperatures and few instances where discharges rose above the mean for the study period resulted in low probability of transitioning into different habitats. The seasonal association with different habitat and movement magnitude have been seen in other studies (Lubejko et al. 2017; Vallazza et al. 2021; Coulter et al. 2022b). Interestingly, carp transitions to and from mainstem and backwater or tributary habitat were high between the same periods (i.e., t4-t5). This time interval was characteristic of environmental conditions associated with spawning by carp (Lubejko et al. 2017; Erickson et al. 2016; Whitledge et al. 2019). Bighead Carp and Silver Carp typically move into higher velocity habitats such as the mainstem to successfully spawn (Deters et al. 2013; Li et al. 2013). Similar probabilities could be caused by individuals being at different stages in the spawning process resulting in similar transition rates as some individuals are moving into the mainstem to possibly spawn and some are moving into

backwater and tributaries to rest and feed (Lubejko et al. 2017; Erickson et al. 2016; Yu et al. 2018). Large individual variations in spawning timing is not uncommon (Coulter et al. 2013; Tucker et al. 2020; Lenaerts et al. 2023a). Carp could also be using backwater and tributary habitats as velocity refugia along their spawning movements (Werner et al. 2023) or while waiting for more suitable spawning conditions in the mainstem (Calkins et al. 2012; Coulter et al. 2016)

Recapture probabilities may be influencing the uncertainty in our model estimates. The lower recapture probabilities within the mainstem translated to higher levels of uncertainty in our estimates of transition probabilities and survival in the mainstem river. The high level of uncertainty in our recapture probabilities within the mainstem was not surprising. In the Red River, discrete sampling of carp using electrofishing found similar detection discrepancies between the mainstem and other habitats (Birdsall 2023). Using an occupancy framework, Birdsall (2023) found detection was negatively associated with discharge indicating carp were more likely to be detected in lower-velocity habitats. Conditions including velocity, substrate, vegetation, and receiver orientation can limit the detection range of acoustic receivers (Kessel et al. 2014; Stott et al. 2021; Carlson et al. 2023). The wide channel, increased velocity, and shifting sand substates of the Red River likely contribute to low and uncertain detection probabilities within the mainstem state. Stanton et al. (2024) found detection probabilities of tagged fish were lower in larger pools with more complex habitat in the Illinois River. Carlson et al. (2023) found that detection probability was higher for receiver mooring designs that protected the receiver from higher velocities compared to designs that were not protected from flows. Weinz et al (2021) found that submerged vegetation also reduced detection efficiency. A greater number of SURs within backwater and tributary habitats could also be contributing to the differences in recapture probabilities. To offset that possibility, we actively tracked more frequently in the mainstem. Thus, if this study continues, active tracking would be needed to avoid bias toward SUR locations; alternatively, more SURs could be placed in the mainstem river but would require more effort for downloading and maintenance.

Our results indicate that environmental conditions were more important to carp movement patterns than individual traits. Our individual trait hypothesis models had less relative support than other models. This is likely due to the low amount of variance explained by our individual random effects. This indicates that all fish movements regardless of individual traits

were more related to our environmental covariates. This is in contrast with other studies that found relationships between fish weight and movement magnitude (Mollenhauer et al. 2013; Radinger and Wolter 2014; Cooke et al. 2022). Other studies have also found sex to be related to fish movement (Hutchings and Gerber 2002; Stiver et al. 2007). Bighead Carp did, on average, move at greater distances and rates compared to Silver Carp. However, species' interactive effect with discharge was not retained in our top model. This could be a result of both species having very similar life-histories and habitat needs (Li et al. 2013; Yao et al. 2016).

Our results also indicate apparent survival was higher in backwater and tributary habitats compared to mainstem habitats. Commercial fishing and bowfishing in the mainstem could be influencing the lower survival in our mainstem state. Bow fishers regularly target Bighead Carp and Silver Carp in their invaded range (York et al. 2022). Commercial fishing is only open in the Arkansas portion of the Red River, which is open year-round, and may influence carp survival. Carp may be better protected from fishing mortality in tributaries and backwaters that have more turbid water, complex habitat, and cannot be accessed by boat during parts of the year. This may also relate to some private landowners who was observed to block backwater areas from access via the main river during hunting season.

Our results better inform managers of which habitats are being selected by Bighead Carp and Silver Carp. In a unique catchment like the Red River, where the main channel and floodplain are largely disconnected, and dynamic water levels are frequent, carp may be challenged to find suitable habitat (Erickson et al. 2016; Lu et al. 2020). Other parts of their invaded range such as the Illinois River contain large amounts of pool-like habitat (Stuck et al. 2015) which are more conducive to carp feeding (Sass et al. 2010; Calkins et al. 2012) and more closely reflect their native ranges (Chen et al. 2007; Lu et al. 2020). Unlike other catchments, the Red River has relatively few velocity refuges. This may provide a management opportunity given the limited areas that carp seem to use. Management of these species in the Red River would be made more efficient if control efforts were conducted in velocity refuges comprising of deeper water. These habitats provide relatively ideal conditions for broad-scale removal efforts (Ridgway et al. 2023a); however, fish are also spawning in the river and have access from other basins so these efforts could be offset by continued immigration and spawning within the catchment.

Recommendations

Management of invasive bigheaded carps in the Upper Mississippi River benefitted immensely from multiple long-term monitoring programs on the Mississippi and Illinois Rivers because data spanned ‘pre’ and ‘post’ invasion phases. Several long-term river monitoring programs exist within the Upper Mississippi River Basin (e.g., LTRMP, state agency datasets, commercial harvest records). These programs have tracked biotic and abiotic changes for varying durations—some of which started in the 1990’s prior to the rapid expansion and growth of bigheaded carp populations (USFWS 1992). Complementary programs in the Lower Mississippi River are less common. Continued monitoring and establishing of ecological baselines are needed if bigheaded carp populations rapidly increase like in the UMR.

Future monitoring strategies would benefit from consideration of gear detection and the use of multiple sampling gears. Not accounting for incomplete gear detection can lead to the underestimation of a species’ distribution and management strategies that do not have the desired outcomes due to consideration of incorrect underlying ecological relationships (Mackenzie et al. 2002; Anderson et al. 2012). For example, ecological relationships could be inferred with discharge that are a function of detection probability where fish are simply more likely to be captured at lower discharge locations. We found detection probability for Bighead Carp was relatively low (average was 0.39 – 0.40), whereas detection for Silver Carp was higher (average was 0.60 – 0.63). However, we incorporated visual confirmations of Silver Carp into our estimates; otherwise, detection of Silver Carp would have been similar to that of Bighead Carp (0.36). Our results indicate that sampling both Bighead Carp and Silver Carp during warmer water temperatures during relatively low discharge would maximize detection, particularly if the river is turbid. Detection was also lower in the mainstem river. Detection probability of fishes in large rivers is commonly affected by water temperature, discharge, and clarity (Gwinn et al. 2016; Mollenhauer et al. 2018; Zentner et al. 2021). Bigheaded carp display schooling behavior during warm-water periods which may increase sampling detection (Sullivan et al. 2017). Silver Carp are commonly observed avoiding sampling gears (Williamson and Garvey 2005; Irons et al. 2007). With low detection probabilities, agencies would benefit from either accounting for detection or completing multiple surveys during the season if monitoring for species presence or abundance. In our study area, Bighead Carp could be present at 10 sites but only detected at less

than half if we relied on a single survey. This underestimation would be exacerbated if sampling were conducted with a single gear. Moreover, use of multiple gears is necessary if agencies are concerned about monitoring both species at different life stages (Wanner and Klumb 2009). If bigheaded carp become more abundant in the Red River catchment, then sampling efficiencies may increase over time (Sullivan et al. 2017), but perhaps at the expense of ecological consequences.

As Bighead Carp and Silver Carp occupy the Red River catchment for longer periods of time, management strategies aimed at preventing their spread and exploiting their vulnerabilities will be key to population control. It would be beneficial for agencies to consider restrictions on locations for anglers to obtain bait if concerned about bigheaded carp spreading to new systems. Collecting live bait from one waterway and transferring it to another can aid the spread of bigheaded carp to nearby reservoirs or river locations above large dams. Although there is currently no documentation of juvenile carp in the Red River upstream of the LA-AR border (Ramsey 2023), we did document successful spawning by adults, and regular recruitment may also be occurring in adjacent basins or reaches further downriver (i.e., several large river fishes have been observed to not spawn each year (e.g., White Sucker *Catostomus commersonii*, Quinn and Ross 1985; see also review by Rideout et al. 2005). If fish recruit from downriver areas, determining actions that prevent movements upstream from locks and dams may be beneficial (e.g., water movement strategies or barriers at the locks, Moy et al. 2011; Hasler et al. 2019; Cupp et al. 2021). Zielinski et al. (2018) found that alterations to lock-and-dam flows via gate operation could reduce bigheaded carp passage while maintaining native fish passage. Interestingly, Bighead Carp have low salinity tolerances during their early life stages (Garcia et al. 1999) which may be useful information for determining possible spawning and rearing locations. For example, average survival time of 11-day post-hatch fry was only 3 days at 4‰ salinity but increased to 96 days at 35 days post hatch (Garcia et al. 1999). However, it is unlikely that salinity will limit reproduction by Silver Carp (larvae tolerance of 6,000–12,000 mg/L CaCO₃, Abdusamadov 1986) which appear to be more common in the catchment than Bighead Carp (i.e., based on counts and similar detection probabilities). Targeted removal efforts at locations associated with both species (e.g., reaches with backwaters, near wing dikes, at tributary confluences) may be beneficial in reducing bigheaded carp numbers, though changes in resulting population abundances have not been demonstrated in open systems, to our knowledge.

Moreover, caution should be taken with removal efforts as we commonly sampled native big river fishes of concern in the same habitats associated with bigheaded carp (e.g., Paddlefish, Alligator Gar).

As Silver Carp and Bighead Carp continue to expand their invasion front, proper assessment and management of these populations will be beneficial if the goal is to reduce their numbers or overall body size. Experimental flows are a mitigation tool that may be used to reduce bigheaded carp growth and overall body size via increased discharge variation. For example, Oliveira et al. (2020) found that experimental flows increased body condition of a barbell *Luciobarbus bocagei* in the Vouga River basin. Additionally, Kelly et al. (2017) found that Longnose Dace *Rhinichthys cataractae* and Slimy Sculpin *Cottus cognatus* mortality increased with flow alterations. Altering hydrographs to increase flow variability could negatively affect bigheaded carp growth and survival while benefiting some native fishes (see Mollenhauer et al. 2021). However, Silver Carp recruitment has been positively related to flow variability in their native ranges (Coulter et al. 2016). Therefore, caution is warranted when devising experimental flows with goals related to invasive species as they are sometimes met with unintended consequences. If bigheaded carp are not currently successfully recruiting (even if spawning) in the lower Red River catchment, then focusing control efforts on immigration points may be a useful strategy. Moreover, further examination of reproduction over multiple years will be useful to determine when reproduction can occur, particularly if the population continues to grow. Efforts to understand survival of spawned eggs would also be prudent.

Invasive bigheaded carp in this catchment are likely to increase without mitigation efforts. Implementing commercial harvest or other removal efforts could increase annual mortality of these populations (though we are unaware of this inducing population collapse or documented declines over large rivers); however, this could harm species of concern (i.e., Alligator Gar and Paddlefish) which share habitat with these invasive fishes and may have limited population level effects on bigheaded carp. Novel strategies for attracting bigheaded carp, even to artificial habitat, during specific times of the year when native fish mortality would be lower (i.e., cooler water) or timing mitigation efforts when native species densities are lower in these habitats (i.e., backwaters) would seem prudent to reduce the associated risk to native species.

Our study results provide useful direction when determining when and where to attempt carp removal efforts in the lower Red River. Carp removal success can vary greatly on timing and location of these efforts (Norman and Whitley 2015; Altenritter et al. 2022). Understanding when carp may be in specific habitats can help fine tune removal efforts to slow down invasion or reduce carp biomass. We found that carp were more likely to transition into different habitats during increased discharge and warming water temperatures. During these conditions, carp may not be as congregated, resulting in less effective capture. This is especially true given that transitions appeared to be in both directions (i.e., to the mainstem river and into backwater and tributaries) as opposed to movement into one area that could be targeted. Targeting carp during times associated with lower movement magnitudes and lower likelihood of transitions between habitats (i.e., winter months during lower discharges) would increase sampling efficiency (and rely on the information associated with detection probability to maximize capture using gears). These results should be considered context dependent as managing carp across their invaded range in the same way may not reach management goal. Whereas many have suggested that carp removal in known spawning habitats during the spring and summer months could reduce reproductive success of both species (Tsehaye et al. 2013; Cupp et al. 2021; Glubzinski et al. 2021), it is unlikely to be successful using traditional gears on the Red River. Moreover, not sampling during this time would also be less disruptive to spawning by native large river fishes. Most of our tagged carp appeared to remain in the study area indicating carp may not be emigrating out of the study area in great numbers (but they have largely spread throughout the basin except for a few areas which may not be suitable). Future studies evaluating the use of barriers lower in the catchment (i.e., Louisiana reaches of the Red River) could also be a strategy worth exploring to ensure that any removed fish are not simply replaced by new fish immigrating into the middle basin of AR, OK, and TX. However, the behavior of fish can change via a compensatory response to some stressors. Thus, continued evaluation of carp movements may be warranted if the agencies decide to undergo large-scale removal operations. As we point out earlier in the report, ensuring SURs are maintained and making use of active tracking is necessary to reduce inherent bias of SUR locations across such a large basin. Given the costs and risks associated with removal efforts, continued evaluation of both native fishes and carp may be necessary to justify such an approach without barrier implementation.

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Appendix A – Red River Figures and Tables

Appendix A, Table 1. The dimensions of each sampling net used for Silver and Bighead Carp studies in the Red River basin, Arkansas. The target life-history stage is indicated.

Gear	Length	Height	Mesh size	Target stage
Gillnet	100'	12'	3.5", 4", 4.25"	Adult
Gillnet	180'	12'	3.5", 4", 4.25"	Adult
Hoop net	16'	4'	3"	Adult
Seine	15'	6'	1/8"	Juvenile
Seine	11'	6'	1/32"	Juvenile
Mini-fyke net	4'	2'	1/8"	Juvenile
Larval tow	1.65m	0.5m	500µm	Juvenile

Appendix A, Table 2. Detection covariates with their associated spatial scale, resolution, and a description of the ecological importance (Justification) for juvenile fishes. Bold covariates were retained for model building after consideration of correlations and effect sizes.

Scale	Covariate	Justification
Reach	Calendar day (24 h)	As fish grow larger and increase in abundance during the season, they are easier to detect ¹
	Temperature (1.0 °C)	Fish move more and grow larger in warmer conditions making them easier to detect. ^{1,2}
	Clarity (1.0 cm)	Higher clarity water may allow fish to more easily evade gears. ³
	Dissolved oxygen (1.00 mg/L)	Decreased dissolved oxygen levels can make fish harder to detect. ⁴
	Seine effort (1.0 m ²)	Higher sampling effort can increase species detection. ⁵
Segment	Discharge (m ³ /s)	High flows can reduce gear efficiency, making fish more difficult to detect. ^{6,7}

1. (Brewer and Ellersieck 2011) 2. (Coutant 1976) 3. (Zamor and Grossman 2007) 4. (Tyler and Targett 2007) 5. (Pritt et al. 2014) 6. (Nunn et al. 2012) 7. (Love et al. 2017)

Appendix A, Table 3. Occupancy covariates with their associated spatial scale, resolution, and a description of the ecological importance (Justification) for juvenile fishes. Bold covariates were retained for model building after consideration of correlations $|r| < 0.50$. Parameters with * indicate they were transformed to categorical covariates due to the distribution of these data. LDI indicates landscape disturbance index, LWD indicates large woody debris, and Dam indicates the distance from the nearest upstream dam.

Scale	Covariate	Justification
Reach	Salinity (1.0 ppt)	Salinity levels in the Red River basin are highly variable and may influence occupancy. ¹
	Zooplankton (1.0 #)	Increased zooplankton densities may increase juvenile fish occupancy because they are the primary food source. ²
	Thalweg depth (1.0 m)	Juvenile fishes may be negatively associated with deeper channel depths. ³
	Width-to-depth (1.0 m)	Wider, shallower channels may be more positively associated with nursery habitat. ⁴
	LWD (1.00 %)	Juvenile fish may be positively associated with LWD because they use it as shelter. ^{4,5}
	* Slackwater (1.00 %)	Juvenile fish likely occupy reaches containing slackwaters because they are important nursery habitats for large river fishes. ⁶
	* Deep pools (1.00 %)	Pools offer low-velocity areas within the main channel and can positively influence occupancy. ⁷
	Dam (1.0 km)	Dams are potential spawning locations of migratory species and affect flow regimes. ^{7,8}
Segment	Discharge (m ³ /s)	Reaches experiencing lower discharges may be beneficial for juvenile species. ^{9,10}
	Sinuosity (1.0 index)	More sinuous stretches of river may contain more habitat complexity that can be used by juvenile fishes. ¹¹

	Slope (1.00%)	Higher stream gradients have higher water velocities which may negatively influence juvenile species occupancy. ¹²
Catchment	*Drainage area (1.0 km ²)	Juvenile fish may occupy nursery habitats within tributaries more strongly than the mainstem river. ¹³
	LDI (1.0 index)	Human disturbance can degrade nursery habitat negatively influencing occupancy. ¹⁴
	Limestone (1.00%)	Limestone composition controls local pH levels which can affect egg survival. ^{15,16}

1. (Hargrave and Taylor 2010b) 2. (Fernando 1994) 3. (Lamouroux et al. 1998) 4. (Thomson et al. 2001) 5. (Everett and Ruiz 1993) 6. (Galat et al. 2004) 7. (Schwartz and Herricks 2005) 7. (Poff et al. 1997) 8. (Soares et al. 2022) 9. (Nunn et al. 2012) 10. (Love et al. 2017) 11. (Warfe and Barmuta 2006) 12. (Camana et al. 2016) 13. (Pracheil et al. 2009) 14. (Schlosser 1995) 15. (Frissell et al. 1986) 16. (Swain et al. 2020)

Appendix A, Table 4. Covariates used to estimate occupancy probability (Ψ) and detection (p) hypothesized to be related to Carp and native fish distributions in the lower Red River catchment with the corresponding state (occupancy [Ψ], and detection [p]), scale, data source, unit, URL, and citation.

Habitat factor	State	Scale	Data source	Unit	URL
Drainage area ^[1]	Ψ	Catchment	NHD+/Stream Stats	km ²	https://apps.nationalmap.gov/downloader/#!/
Disturbance ^[2]	Ψ	Catchment	NLCD	LDI	https://apps.nationalmap.gov/downloader/#!/
Lithology ^[3]	Ψ	Catchment	U.S. Geological Survey	% limestone	https://mrdata.usgs.gov/geology/state/
Sinuosity ^[1]	Ψ	Segment	ArcPro GIS		https://apps.nationalmap.gov/downloader/#!/
Slope ^[1]	Ψ	Segment	ArcPro GIS	%	https://apps.nationalmap.gov/downloader/#!/
Discharge ^[4]	Ψ	Segment	U.S. Geological Survey	m ³ /s	https://waterdata.usgs.gov/nwis/rt
Distance to Dam ^[1]	Ψ	Reach	ArcPro GIS	rkm	https://apps.nationalmap.gov/downloader/#!/
Percent backwater	Ψ	Reach	Field collection	%	
Width to depth	Ψ	Reach	Field collection		
Salinity	Ψ	Reach	YSI pro dds	ppt	
Chlorophyll-a	Ψ	Reach	Water sample	mg/L	
Temperature	P	Reach	Field collection	°C	
Discharge ^[4]	P	Segment	U.S. Geological Survey	m ³ /s	https://waterdata.usgs.gov/nwis/rt
Secchi depth	P	Reach	Field collection	cm	
Electrofishing effort	p	Reach	Field collection	S	

Appendix A, Table 5. Covariate combinations (backwater [Bck], discharge [Q], chlorophyll-*a* [Chla], width-to-depth ratio [W:D], sinuosity [Sin], distance to dam [Dtd], and salinity [Sal]) for the two overarching hypothesized models (growth and spawn) related to Carp occupancy in the Red River basin.

Model framework	Model combinations
Growth	Bck
	Q
	Bck + Q
	Chla
	W:D
	Bck + Chla
	Bck + Sin
	Bck + W:D
	Q + Chla
	Q + Sin
	Q + W:D
	Sin + Chla
	Sin + Chla
	W:D + Chla
	W:D + Sin
	Bck + Q + Chla
	Bck + Q + Sin
	Bck + Q + W:D
	Bck + Sin + Chla
	Bck + W:D + Chla
	Bck + W:D + Sin
	Bck + Q + W:D + Chla
	Bck + Q + W:D + Sin
	Bck + W:D + Sin + Chla

Spawn

$Q + W:D + \text{Sin} + \text{Chla}$

$\text{Bck} + Q + W:D + \text{Sin} + \text{Chla}$

Bck

Q

$\text{Bck} + Q$

Dtd

Sal

$\text{Bck} + \text{Dtd}$

$\text{Bck} + \text{Sal}$

$Q + \text{Dtd}$

$Q + \text{Sal}$

$\text{Sal} + \text{Dtd}$

$\text{Bck} + Q + \text{Dtd}$

$\text{Bck} + Q + \text{Sal}$

$\text{Bck} + \text{Sal} + \text{Dtd}$

$\text{Sal} + \text{Dtd} + Q$

$\text{Bck} + Q + \text{Sal} + \text{Dtd}$

Appendix A, Table 6. Model combinations for evaluating the relationship between Silver Carp and Bighead Carp growth and environmental factors. Model combinations for Weisberg models: model intercept [B0], fish age [A], coefficient of variation (CV) of discharge [CV.Q], CV of air temperature [CV.T], discharge [Q], and air temperature [T]. Random effects (i.e., fish and year) were included in all models.

Models
~ B0 + A
~ B0 + A + CV.Q
~ B0 + A + CV.T
~ B0 + A + Q
~ B0 + A + Q + CV.Q
~ B0 + A + Q + CV.T
~ B0 + A + Q + CV.T + CV.Q
~ B0 + A + Q + T
~ B0 + A + T
~ B0 + A + T + CV.Q
~ B0 + A + T + CV.T
~ B0 + A + T + CV.T + CV.Q
~ B0 + A + T + Q
~ B0 + A + T + Q + CV.Q
~ B0 + A + T + Q + CV.T
~ B0 + A + T + Q + CV.T + CV.Q

Appendix A, Table 7. Overarching hypothesized model (growth and spawn) with associated covariates (backwater [Bck], discharge [Q], width-to-depth ratio [W:D], sinuosity [Sin], chlorophyll-*a* [Chla], salinity [Sal], and distance to dam [Dtd]) and the corresponding hypothesis of their relationship to occupancy by Silver Carp and Bighead Carp.

Model	Covariate	Hypothesis
Growth	Bck	Backwaters can offer higher forage potential, growth potential because of warmer water temperature for bioenergetics, and decreased energy expenditure. ^[1,2,3]
	Q	Negatively associated because of increased energy expenditure and lower forage availability. ^[4, 5, 6, 7]
	W:D	Carp growth positively associated due to low-velocity habitats, increased forage, and decreased competitor species due to lower habitat complexity ^[8, 9, 10, 11]
	Sin	Increased growth because of decreased competitor species and decreased habitat complexity. ^[10, 11]
	Chla	Increased forage available for growth. ^[12, 13, 14]
Spawn	Bck	Possibly used as staging locations for spawning. ^[15, 16, 17]
	Q	Positively associated with discharge because of increased flow requirements for pelagic spawners and successful spawning associated with high discharge. ^[18, 19, 20]
	Sal	Improper salinity can hinder spawning. ^[21, 22]
	Dtd	Positively associated with presence because of minimum flow distance requirements for successful spawning and flow alteration can affect recruitment. ^[23, 24, 25]

^[1]Williamson and Garvey 2005, ^[2]Humphries et al. 2006, ^[3]Coulter et al. 2017, ^[4]Newbold et al. 2016, ^[5]Hoover et al. 2017, ^[6]MacNamara et al. 2018, ^[7]Pretchel et al. (2018), ^[8]Williamson and Garvey 2005, ^[9]Scheler et al. 2012, ^[10]Hasegawa and Maekawa (2008), ^[11]Alexander et al. (2015), ^[12]Calkins et al. 2012, ^[13]Li et al. 2013, ^[14]Ochs et al. 2019, ^[15]Junk et al. 1989, ^[16]Coulter et al. 2017, ^[17]Whitten et al. 2021, ^[18]Kolar et al. 2007, ^[19]Gibson-Reinemer et al. 2017, ^[20]Lenaerts et al. 2021, ^[21]Akimova et al. 2016, ^[22]Neves et al. 2019, ^[23]Duan et al. 2009, ^[24]Song et al. (2018), ^[25]Parkos III et al. 2021

Appendix A, Table 8. Summary of tagged fish demographics, tagging dates, shed tag or death, and number of detections. Tag Identification, date tagged, species ([Spp] Silver Carp (SVC) or Bighead Carp (BHC)), length (total length mm), weight (g), sex (male (M) or female (F)), number of detections of tagged carp, and shed tag or death are included. “Tag removed” indicates the date that a fish was estimated to have died or shed their tag. Individuals with no date were alive throughout the study.

Tag ID	Tag date	Spp	Length (mm)	Weight (g)	Sex	Number of detections	Tag removed
A69-1604-24204	3/14/23	SVC	900	8000	M	286	
A69-1604-24205	3/13/23	SVC	1052	15900	F	0	
A69-1604-24206	3/10/23	SVC	798	6000	M	6	
A69-1604-24207	3/22/23	BHC	974	14500	M	10	
A69-1604-24208	3/17/23	BHC	1055	16500	M	16	
A69-1604-24209	3/17/23	SVC	835	11000	F	1	
A69-1604-24210	3/24/23	SVC	682	5000	F	0	
A69-1604-24211	3/24/23	SVC	856	7500	M	10	2/12/2024
A69-1604-24212	3/22/23	BHC	1005	15000	M	315	
A69-1604-24213	4/4/23	SVC	991	16900	F	13	6/23/2023
A69-1604-24214	3/29/23	SVC	786	6750	M	42	
A69-1604-24215	3/24/23	SVC	838	7000	M	0	
A69-1604-24216	4/6/23	SVC	850	9000	F	3	
A69-1604-24217	4/6/23	SVC	795	6000	F	3	
A69-1604-24218	4/5/23	SVC	881	7000	M	0	
A69-1604-24219	4/26/23	BHC	1200	21800	F	1	
A69-1604-24220	4/18/23	BHC	1025	17000	F	3	

A69-1604-24221	4/27/23	BHC	1289	27000	M	3	
A69-1604-24222	3/3/23	SVC	850	7450	M	64	5/23/2024
A69-1604-24223	3/3/23	SVC	929	15000	F	365	
A69-1604-24224	3/3/23	SVC	1000	15500	F	401	
A69-1604-24225	3/3/23	SVC	882	9450	M	267	
A69-1604-24226	3/3/23	SVC	959	12750	M	5	5/27/2023
A69-1604-24227	3/3/23	SVC	978	15100	F	92	7/4/2023
A69-1604-24228	3/3/23	BHC	1221	24450	M	0	
A69-1604-24229	3/3/23	SVC	901	10000	M	313	
A69-1604-24230	3/3/23	SVC	892	9500	M	338	
A69-1604-24231	3/3/23	SVC	903	10500	M	238	
A69-1604-24232	3/3/23	BHC	1155	20750	M	138	
A69-1604-24233	3/9/23	BHC	1245	30750	F	9	
A69-1604-24234	3/9/23	BHC	1105	22000	M	24	
A69-1604-24235	3/9/23	BHC	1212	24500	M	1	
A69-1604-24236	3/9/23	BHC	1155	24500	M	5	7/31/2023
A69-1604-24237	3/9/23	BHC	1071	23600	F	38	
A69-1604-24238	3/9/23	BHC	1010	13500	M	94	
A69-1604-24239	3/9/23	BHC	1186	25000	M	2	
A69-1604-24240	4/11/23	BHC	1340	36000	F	18	
A69-1604-24241	4/18/23	SVC	800	5500	F	2	
A69-1604-24242	4/27/23	BHC	1279	26200	M	7	
A69-1604-24243	4/12/23	SVC	913	11000	F	9	

A69-1604-24244	4/24/23	BHC	1061	18500	M	214	
A69-1604-24245	4/11/23	SVC	609	2500	M	1	4/25/2023
A69-1604-24246	4/26/23	BHC	1158	20500	F	85	
A69-1604-24247	4/11/23	BHC	964	11750	M	270	
A69-1604-24248	4/26/23	BHC	1171	23300	F	124	
A69-1604-24249	4/27/23	BHC	1086	25000	M	52	
A69-1604-24250	4/18/23	BHC	1220	29500	F	5	
A69-1604-24251	4/24/23	BHC	1040	15750	F	85	
A69-1604-24252	4/26/23	BHC	1051	15100	M	3	8/10/2023
A69-1604-24253	4/12/23	SVC	912	10500	F	139	

Appendix A, Table 9. Variables, scale, unit, and gear or source used for determining movement trends of Bighead Carp and Silver Carp in the lower Red River. SVC = Silver Carp, BHC = Bighead Carp. M = male, and F =female, and SURs = Submersible ultrasonic receivers.

Variable	Scale	Unit	Gear/Source
Average water temperature	Catchment	°C	SURs
Three days before average water temperature	Catchment	°C	SURs
CV of average water temperature	Catchment	-	SURs
Average water temperature squared	Catchment	°C	SURs
Average photoperiod	Catchment	min	https://aa.usno.navy.mil/
Average discharge	Catchment	m ³ /S	USGS stream gage
CV of discharge	Catchment	-	USGS stream gage
Three days before average discharge	Catchment	m ³ /S	USGS stream gage
Day of study	NA	1 Day	Fish movement
Direction	Individual	1/0	Fish movement
Species	Individual	SVC/BHC	Morphology
Sex	Individual	M/F	Morphology
Weight	Individual	g	Scale
Total Length	Individual	mm	Measuring board

Appendix A, Table 10. Description of environmental conditions for each open period interval (i.e., time between each closed period, t) based on water temperature and discharge (see Figure 7).

Period intervals	Environmental conditions
t1 to t2	Discharges were highly variable with four increases above the mean discharge for the study period (170 m ³ /s). Water temperatures increased between these two periods.
t2 to t3	Discharges were lower than average between these periods with one increase in discharge magnitude above mean. Water temperatures decreased from the maximum for the year.
t3 to t4	Discharges were below the mean with one increase in magnitude above the mean. Water temperatures reached the minimum for the study period between these two periods.
t4 to t5	Discharges were highly variable with four increases in magnitude above the mean. Water temperatures steadily increased with moderate variability.
t5 to t6	Discharges were highly variable with extended periods above the mean and a decline below the mean. Water temperatures increased and reached the maximum for the year between these periods.

Appendix A, Table 11. Recapture probability estimates (p) for time periods (t) and time-constant average survival probability estimate (s) from the three-state transition model. Parameters = each estimated parameter where B is backwater, M is mainstem, and T is tributary, Mode = most likely value, C.I. = 95% credibility interval.

Parameter	Mode	C.I.
p(B,t2)	0.92	(0.69, 0.99)
p(B,t3)	0.93	(0.73, 0.99)
p(B,t4)	0.88	(0.64, 0.99)
p(B,t5)	0.82	(0.53, 0.98)
p(B,t6)	0.82	(0.54, 0.98)
p(M,t2)	0.56	(0.25, 0.92)
p(M,t3)	0.13	(0.02, 0.44)
p(M,t4)	0.13	(0.02, 0.44)
p(M,t5)	0.83	(0.56, 0.99)
p(M,t6)	0.26	(0.07, 0.62)
p(T,t2)	0.62	(0.35, 0.87)
p(T,t3)	0.59	(0.30, 0.89)
p(T,t4)	0.77	(0.41, 0.98)
p(T,t5)	0.82	(0.46, 0.98)
p(T,t6)	0.73	(0.33, 0.98)
s(B,.)	0.95	(0.95, 1)
s(M,.)	0.86	(0.67, 0.97)
s(T,.)	0.96	(0.87, 1)

Appendix A, Table 12. Demographic information of most Bighead Carp (BHC) and Silver Carp (SVC) collected from May 2021 through November 2023 during sampling events. The sample date, location, and gears used are provided. Total length (TL, mm), weight (W, g), and sex (male [M] or female [F]) of each fish are provided. The age estimates using otoliths are provided. These carp were sampled using gillnets (GN), electrofishing (EF), bow-fishermen (BF) which were received from the U.S. Fish and Wildlife Service or jumped in the boat during a survey (JM). The latitude and longitude were measured at the most downstream portion of each reach. These locations (and those lacking demographic data) have all been provided to the U.S. Geological Survey, NAS reporting page via communication with Dr. Matt Neilson (U.S. Geological Survey, written communication, 2023). Fin clips from a subset of these fish were mailed to the U.S. Geological Survey on August 9, 2022 (Stephen F. Spear, U.S. Geological Survey, 2022).

State	River	Date	Latitude	Longitude	Species	TL	W	Gear	Sex	Age	Eggs
TX	Choctaw	6/23/21	33.77368	-96.41828	SVC	745	4900	BF	M	3	-
AR	Red River	7/5/21	33.60848	-93.81358	SVC	710	3880	EF	F	4	233740
TX	Bois d'Arc	7/7/21	33.82851	-95.85503	BHC	1048	12840	GN	F	11	561374
AR	Red River	7/9/21	33.57763	-94.36778	SVC	897	7260	GN	M	-	-
AR	Red River	7/12/21	33.58165	-94.36528	SVC	912	7460	GN	M	6	-
OK	Kiamichi	7/15/21	33.96051	-95.29222	SVC	708	3850	GN	M	3	-
OK	Red River	7/16/21	33.63824	-94.58038	BHC	1240	-	GN	F	4	256314
TX	Choctaw	7/19/21	33.72074	-96.3769	SVC	910	9500	BF	M	9	-
TX	Choctaw	7/21/21	33.72004	-96.39876	SVC	850	8160	JM	M	7	-
TX	Bois d'Arc	7/23/21	33.82851	-95.85503	BHC	1245	-	GN	M	11	-
TX	Bois d'Arc	7/23/21	33.82851	-95.85503	BHC	1090	-	GN	F	13	618524
OK	Webb Creek	7/25/21	33.77368	-96.41828	SVC	720	4620	BF	M	3	-

AR	Red River	8/4/21	33.57763	-94.36778	BHC	1108	13670	GN	M	9	-
AR	Red River	8/4/21	33.57763	-94.36778	SVC	808	6460	EF	M	5	-
TX	Choctaw	8/10/21	33.71952	-96.3907	BHC	1097	14220	GN	F	12	722638
TX	Choctaw	8/10/21	33.71952	-96.3907	BHC	1100	13480	GN	M	11	-
TX	Choctaw	8/10/21	33.71952	-96.3907	BHC	1140	15180	GN	M	6	-
TX	Choctaw	8/10/21	33.71952	-96.3907	BHC	990	9260	GN	M	5	-
TX	Choctaw	8/10/21	33.71952	-96.3907	SVC	850	7600	GN	M	7	-
OK	Red River	8/10/21	33.77693	-96.47263	BHC	925	6350	BF	M	7	-
TX	Choctaw	8/11/21	33.72068	-96.39828	BHC	1069	12000	GN	M	11	-
TX	Choctaw	8/11/21	33.72068	-96.39828	SVC	851	8100	EF	M	8	-
TX	Choctaw	8/11/21	33.72068	-96.39828	SVC	882	8350	EF	F	3	1217828
OK	Red River	8/23/21	33.8032	-94.91955	BHC	1230	21500	GN	-	6	-
AR	Red River	8/24/21	33.57763	-94.36778	BHC	960	17500	GN	-	9	-
AR	Red River	8/24/21	33.57763	-94.36778	SVC	850	9000	EF	-	8	-
AR	Red River	8/24/21	33.57763	-94.36778	SVC	752	5020	EF	F	5	720804
AR	Red River	8/24/21	33.57763	-94.36778	SVC	783	6300	GN	-	4	-
OK	Red River	9/5/21	33.79629	-96.51525	BHC	1130	15600	BF	F	10	-
OK	Red River	9/6/21	33.79629	-96.51525	BHC	1130	19700	BF	F	-	-
OK	Red River	9/6/21	33.79629	-96.51525	BHC	1090	14600	BF	F	12	-
AR	Red River	9/21/21	33.58165	-94.36528	SVC	876	8500	JM	F	4	462370
AR	Red River	9/21/21	33.58165	-94.36528	SVC	752	4800	GN	F	3	308066
AR	Red River	10/24/21	33.58165	-94.36528	SVC	952	9500	GN	-	8	-

AR	Red River	10/24/21	33.58165	-94.36528	SVC	830	6000	JM	-	5	-
TX	Choctaw	11/16/21	33.71952	-96.3907	BHC	1205	18000	GN	M	13	-
TX	Choctaw	11/16/21	33.71952	-96.3907	BHC	1033	10025	EF	F	13	407264
TX	Choctaw	11/16/21	33.71952	-96.3907	SVC	932	10750	GN	F	3	1022782
TX	Choctaw	11/16/21	33.71952	-96.3907	SVC	765	6000	EF	F	5	381742
TX	Choctaw	11/16/21	33.71952	-96.3907	SVC	1020	12050	EF	F	10	1110148
OK	Webb Creek	12/1/21	33.7729	-96.41801	SVC	883	7940	BF	M	-	-
OK	Webb Creek	12/1/21	33.7729	-96.41801	SVC	864	8300	BF	F	8	-
TX	Choctaw	12/15/21	33.71952	-96.3907	BHC	1225	23000	EF	F	16	1322169
TX	Choctaw	12/15/21	33.71952	-96.3907	SVC	902	8000	GN	M	7	-
TX	Choctaw	1/4/22	33.71952	-96.3907	BHC	974	11000	EF	M	8	-
TX	Choctaw	1/4/22	33.71952	-96.3907	SVC	911	8500	EF	F	4	1333354
TX	Choctaw	1/5/22	33.71952	-96.3907	BHC	1252	-	EF	F	15	1406850
AR	Red River	1/6/22	33.05954	-93.82767	SVC	750	4750	GN	M	5	-
AR	Red River	1/6/22	33.05954	-93.82767	SVC	820	5500	GN	M	5	-
AR	Red River	1/12/22	33.58165	-94.36528	SVC	915	11000	EF	F	5	1271831
AR	Red River	1/12/22	33.58165	-94.36528	SVC	865	8600	EF	M	9	-
AR	Red River	1/12/22	33.58165	-94.36528	SVC	902	8600	EF	M	9	-
AR	Red River	1/12/22	33.58165	-94.36528	SVC	904	7000	EF	M	8	-
AR	Red River	1/12/22	33.58165	-94.36528	SVC	894	7000	EF	M	8	-
AR	Red River	1/12/22	33.58165	-94.36528	SVC	848	7000	EF	M	6	-
AR	Red River	1/12/22	33.58165	-94.36528	SVC	850	7700	EF	M	10	-

AR	Red River	1/12/22	33.58165	-94.36528	SVC	899	10000	EF	F	5	1665113
AR	Red River	1/12/22	33.58165	-94.36528	SVC	868	7000	EF	M	7	-
AR	Red River	1/12/22	33.58165	-94.36528	SVC	945	12600	EF	F	6	1938600
AR	Red River	1/12/22	33.58165	-94.36528	SVC	815	7500	EF	M	4	-
AR	Red River	1/12/22	33.58165	-94.36528	SVC	852	8000	EF	F	5	1371910
AR	Red River	1/12/22	33.58165	-94.36528	SVC	1090	15200	EF	F	12	2146966
AR	Red River	1/12/22	33.58165	-94.36528	SVC	842	7500	EF	F	5	1212280
AR	Red River	1/12/22	33.58165	-94.36528	SVC	926	11500	EF	M	5	-
AR	Red River	1/12/22	33.58165	-94.36528	SVC	915	11400	EF	F	13	1245502
AR	Red River	1/12/22	33.58165	-94.36528	SVC	1036	12900	EF	F	11	-
AR	Red River	1/12/22	33.58165	-94.36528	SVC	872	9500	EF	F	6	1347598
AR	Red River	1/12/22	33.58165	-94.36528	SVC	945	11800	EF	F	11	1526780
AR	Red River	1/12/22	33.58165	-94.36528	SVC	821	6250	EF	M	5	-
AR	Red River	1/12/22	33.58165	-94.36528	SVC	828	6750	GN	M	5	-
AR	Red River	1/12/22	33.58165	-94.36528	SVC	828	8000	GN	M	11	-
AR	Red River	1/12/22	33.58165	-94.36528	SVC	822	8200	GN	F	5	1543506
AR	Red River	1/12/22	33.58165	-94.36528	SVC	820	8750	GN	M	6	-
AR	Red River	1/18/22	33.33958	-93.69724	SVC	872	6750	EF	M	4	-
OK	Kiamichi	1/19/22	34.00923	-95.38224	BHC	1092	12400	EF	F	9	-
OK	Red River	2/8/22	33.77009	-96.42174	BHC	1020	11600	EF	M	8	-
OK	Red River	2/8/22	33.77009	-96.42174	BHC	1232	20450	GN	M	10	-
OK	Red River	2/8/22	33.77009	-96.42174	BHC	1152	17200	GN	M	5	-

OK	Red River	2/8/22	33.77009	-96.42174	SVC	928	10000	GN	M	8	-
OK	Red River	2/8/22	33.77009	-96.42174	SVC	834	7400	GN	F	5	1357569
OK	Red River	2/8/22	33.77009	-96.42174	SVC	878	7100	GN	M	4	-
OK	Red River	2/8/22	33.77009	-96.42174	SVC	892	8000	GN	M	9	-
OK	Red River	2/8/22	33.77009	-96.42174	SVC	920	8900	GN	M	9	-
OK	Red River	2/8/22	33.77009	-96.42174	SVC	798	6000	GN	M	4	-
OK	Red River	2/8/22	33.77009	-96.42174	SVC	828	6400	GN	M	5	-
OK	Red River	2/8/22	33.77009	-96.42174	SVC	780	6250	GN	F	5	440620
OK	Red River	2/8/22	33.77009	-96.42174	SVC	818	6000	GN	M	4	-
OK	Red River	2/8/22	33.77009	-96.42174	SVC	854	7600	GN	M	9	-
OK	Red River	2/27/22	33.82107	-96.56023	BHC	990	13050	BF	F	8	-
TX	Choctaw	3/2/22	33.72068	-96.39828	SVC	797	5750	GN	M	4	-
AR	Red River	3/15/22	33.57763	-94.36778	BHC	1052	17100	GN	M	15	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	938	9478	EF	M	10	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	870	6732	EF	M	5	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	898	8860	EF	F	5	1852392
AR	Red River	3/15/22	33.57763	-94.36778	SVC	829	4768	EF	M	7	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	811	6406	EF	M	5	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	910	8076	EF	M	7	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	888	8718	EF	F	6	1853165
AR	Red River	3/15/22	33.57763	-94.36778	SVC	920	8616	EF	M	9	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	919	9728	EF	F	4	1726246

AR	Red River	3/15/22	33.57763	-94.36778	SVC	813	6668	EF	M	9	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	939	9402	EF	F	7	1994196
AR	Red River	3/15/22	33.57763	-94.36778	SVC	1021	12646	EF	F	9	2117725
AR	Red River	3/15/22	33.57763	-94.36778	SVC	900	9776	EF	F	5	2219507
AR	Red River	3/15/22	33.57763	-94.36778	SVC	922	7674	EF	M	11	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	902	8484	EF	M	6	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	818	6486	EF	M	4	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	933	8404	EF	M	14	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	920	9034	EF	M	13	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	874	8328	EF	F	4	1531772
AR	Red River	3/15/22	33.57763	-94.36778	SVC	875	7622	EF	M	5	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	999	11980	EF	F	9	1754611
AR	Red River	3/15/22	33.57763	-94.36778	SVC	954	9654	EF	M	11	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	988	11412	EF	F	7	2427228
AR	Red River	3/15/22	33.57763	-94.36778	SVC	882	8256	EF	M	9	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	832	7998	GN	M	10	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	902	8340	GN	M	10	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	847	7836	GN	M	5	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	900	7878	GN	M	9	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	920	8904	GN	M	9	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	790	5890	GN	M	4	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	792	6700	GN	M	6	-

AR	Red River	3/15/22	33.57763	-94.36778	SVC	901	7256	GN	M	4	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	870	7832	GN	M	5	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	798	6592	GN	M	4	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	901	7518	GN	M	9	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	905	8166	GN	M	7	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	834	7080	GN	M	5	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	844	5888	GN	M	5	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	833	6996	GN	M	8	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	911	9292	GN	M	10	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	772	5470	GN	M	5	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	802	9546	GN	M	5	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	910	9098	GN	M	9	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	946	11584	GN	F	4	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	800	6306	GN	M	5	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	894	8016	GN	M	12	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	858	6208	GN	M	4	-
AR	Red River	3/15/22	33.57763	-94.36778	SVC	856	7390	GN	M	5	-
AR	Red River	3/23/22	33.58165	-94.36528	BHC	968	8870	GN	F	9	799297
AR	Red River	3/23/22	33.58165	-94.36528	SVC	858	5982	GN	M	7	-
AR	Red River	3/23/22	33.58165	-94.36528	SVC	862	7488	GN	M	9	-
AR	Red River	3/23/22	33.58165	-94.36528	SVC	874	9482	GN	M	12	-
AR	Red River	3/23/22	33.58165	-94.36528	SVC	912	9138	GN	M	7	-

AR	Red River	3/23/22	33.58165	-94.36528	SVC	854	7824	GN	F	4	1040701
AR	Red River	3/23/22	33.58165	-94.36528	SVC	740	-	EF	F	7	-
AR	Red River	3/23/22	33.58165	-94.36528	SVC	820	6300	GN	F	7	-
AR	Red River	3/23/22	33.58165	-94.36528	SVC	838	7134	EF	M	5	-
AR	Red River	3/23/22	33.58165	-94.36528	SVC	850	6974	EF	M	6	-
AR	Red River	3/23/22	33.58165	-94.36528	SVC	890	8000	GN	M	11	-
AR	Red River	3/23/22	33.58165	-94.36528	SVC	784	5300	EF	M	5	-
AR	Red River	3/23/22	33.58165	-94.36528	SVC	930	-	EF	F	10	1164252
AR	Red River	3/23/22	33.58165	-94.36528	SVC	808	5964	GN	M	6	-
AR	Red River	3/23/22	33.58165	-94.36528	SVC	1040	12200	EF	F	8	-
AR	Red River	3/23/22	33.58165	-94.36528	SVC	928	-	EF	F	5	1687332
AR	Red River	3/24/22	33.57763	-94.36778	BHC	1204	17600	GN	M	11	-
AR	Red River	3/24/22	33.57763	-94.36778	BHC	1200	18000	GN	M	8	-
AR	Red River	3/24/22	33.57763	-94.36778	BHC	1114	15500	GN	M	10	-
AR	Red River	3/24/22	33.57763	-94.36778	BHC	1180	16500	GN	M	9	-
AR	Red River	3/24/22	33.57763	-94.36778	BHC	1164	18500	GN	M	9	-
AR	Red River	3/24/22	33.57763	-94.36778	BHC	1142	15300	GN	M	9	-
AR	Red River	3/24/22	33.57763	-94.36778	BHC	1206	18300	GN	M	10	-
AR	Red River	3/24/22	33.57763	-94.36778	BHC	1148	16400	GN	M	11	-
AR	Red River	3/24/22	33.57763	-94.36778	BHC	1092	15400	GN	M	9	-
AR	Red River	3/24/22	33.57763	-94.36778	BHC	1050	13000	GN	M	4	-
AR	Red River	3/24/22	33.57763	-94.36778	BHC	1062	9784	GN	M	10	-

AR	Red River	3/24/22	33.57763	-94.36778	BHC	1090	14500	GN	M	13	-
AR	Red River	3/24/22	33.57763	-94.36778	BHC	1299	20000	GN	M	17	-
AR	Red River	3/24/22	33.57763	-94.36778	BHC	1123	14600	GN	M	5	-
AR	Red River	3/24/22	33.57763	-94.36778	BHC	1151	14600	GN	M	11	-
AR	Red River	3/24/22	33.57763	-94.36778	BHC	1210	16100	GN	M	10	-
AR	Red River	3/24/22	33.57763	-94.36778	BHC	1120	18400	GN	M	12	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	788	5850	GN	M	5	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	876	6502	GN	M	14	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	918	9408	GN	M	10	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	908	8700	GN	M	9	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	850	6914	GN	M	-	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	852	6302	GN	M	5	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	824	5912	GN	M	4	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	1070	15600	GN	F	10	2258958
AR	Red River	3/24/22	33.57763	-94.36778	SVC	1056	13250	GN	M	10	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	992	11288	GN	F	10	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	968	10756	GN	F	5	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	873	7524	GN	M	6	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	918	8322	EF	M	9	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	988	10432	EF	F	4	1642943
AR	Red River	3/24/22	33.57763	-94.36778	SVC	1050	13500	EF	F	10	1805216
AR	Red River	3/24/22	33.57763	-94.36778	SVC	886	9752	EF	F	6	1224470

AR	Red River	3/24/22	33.57763	-94.36778	SVC	966	10716	EF	F	5	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	924	9352	EF	M	9	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	830	6824	EF	M	5	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	838	7328	EF	M	5	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	976	12020	EF	F	4	2432568
AR	Red River	3/24/22	33.57763	-94.36778	SVC	874	9176	GN	F	5	1258424
AR	Red River	3/24/22	33.57763	-94.36778	SVC	878	6896	GN	M	8	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	960	10902	GN	F	5	1369413
AR	Red River	3/24/22	33.57763	-94.36778	SVC	936	11272	GN	F	5	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	794	5698	GN	M	5	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	998	10056	GN	F	6	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	1010	13400	GN	F	8	1906972
AR	Red River	3/24/22	33.57763	-94.36778	SVC	946	10834	GN	F	8	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	904	11096	GN	F	5	1878883
AR	Red River	3/24/22	33.57763	-94.36778	SVC	888	9218	GN	F	-	1561302
AR	Red River	3/24/22	33.57763	-94.36778	SVC	916	8822	GN	M	7	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	912	9860	GN	F	4	1756537
AR	Red River	3/24/22	33.57763	-94.36778	SVC	920	11484	GN	F	6	923649
AR	Red River	3/24/22	33.57763	-94.36778	SVC	856	8964	GN	F	5	2226944
AR	Red River	3/24/22	33.57763	-94.36778	SVC	938	12100	GN	F	7	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	948	11300	GN	F	9	1981819
AR	Red River	3/24/22	33.57763	-94.36778	SVC	885	9200	GN	F	5	1195327

AR	Red River	3/24/22	33.57763	-94.36778	SVC	875	9260	GN	F	12	766236
AR	Red River	3/24/22	33.57763	-94.36778	SVC	820	6000	GN	M	8	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	818	5858	GN	M	6	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	806	6158	GN	M	5	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	888	9212	GN	M	11	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	878	7626	GN	M	9	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	980	10894	GN	F	4	1790088
AR	Red River	3/24/22	33.57763	-94.36778	SVC	904	10266	GN	F	5	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	898	9604	GN	M	10	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	910	8956	GN	M	12	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	852	6174	GN	M	5	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	864	7476	GN	M	6	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	866	9756	GN	F	4	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	928	9302	GN	M	-	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	816	6510	GN	M	5	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	890	8332	GN	M	9	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	934	9078	GN	M	13	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	941	9136	GN	M	8	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	902	8780	GN	F	5	1527668
AR	Red River	3/24/22	33.57763	-94.36778	SVC	874	10392	GN	F	5	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	830	6382	GN	M	4	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	920	10268	GN	F	5	1685973

AR	Red River	3/24/22	33.57763	-94.36778	SVC	976	10612	GN	F	10	2187637
AR	Red River	3/24/22	33.57763	-94.36778	SVC	870	8194	GN	M	6	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	928	9964	GN	M	10	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	942	9370	GN	M	10	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	891	8850	GN	F	5	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	822	8978	GN	F	5	-
AR	Red River	3/24/22	33.57763	-94.36778	SVC	1042	13700	GN	F	11	-
OK	Kiamichi	4/1/22	34.00912	-95.38141	SVC	1040	13640	BF	F	13	-
AR	Red River	4/4/22	33.60848	-93.81358	SVC	891	9000	EF	M	7	-
TX	Choctaw	4/13/22	33.72068	-96.39828	BHC	1258	17000	GN	F	15	1291278
TX	Choctaw	4/13/22	33.72068	-96.39828	BHC	1152	12500	GN	F	10	-
TX	Choctaw	4/13/22	33.72068	-96.39828	SVC	842	7100	EF	M	5	-
OK	Red River	4/21/22	33.82131	-96.55203	BHC	1010	17000	BF	F	10	-
AR	Red River	4/29/22	33.58165	-94.36528	SVC	915	9000	EF	F	4	-
OK	Red River	5/4/22	33.8032	-94.91955	SVC	888	8000	GN	F	3	-
OK	Red River	5/13/22	33.91901	-95.07648	BHC	1063	10600	GN	M	9	-
OK	Garland Creek	5/13/22	33.92015	-95.07693	SVC	937	9400	EF	F	9	-
OK	Red River	5/18/22	33.82131	-96.55203	BHC	1095	19100	BF	F	8	-
OK	Kiamichi	5/26/22	33.9605	-95.25517	BHC	1050	9300	GN	M	11	-
OK	Kiamichi	5/26/22	33.9605	-95.25517	BHC	1068	11400	GN	M	8	-
OK	Kiamichi	5/26/22	33.9605	-95.25517	SVC	752	4750	EF	M	4	-
OK	Kiamichi	5/26/22	33.9605	-95.25517	SVC	887	7100	GN	M	6	-

OK	Kiamichi	5/26/22	33.9605	-95.25517	SVC	859	6500	GN	M	9	-
AR	Red River	5/28/22	33.58165	-94.36528	BHC	1004	11892	GN	M	9	-
AR	Red River	5/28/22	33.58165	-94.36528	BHC	1198	16750	EF	F	12	-
AR	Red River	5/28/22	33.58165	-94.36528	BHC	1350	27750	EF	F	12	-
AR	Red River	5/28/22	33.58165	-94.36528	SVC	789	4338	GN	M	8	-
AR	Red River	5/28/22	33.58165	-94.36528	SVC	912	8876	GN	M	6	-
AR	Red River	5/28/22	33.58165	-94.36528	SVC	813	6324	GN	M	5	-
AR	Red River	5/28/22	33.58165	-94.36528	SVC	886	8662	GN	F	4	-
AR	Red River	5/28/22	33.58165	-94.36528	SVC	919	11388	GN	F	4	-
AR	Red River	5/28/22	33.58165	-94.36528	SVC	850	8168	GN	M	5	-
AR	Red River	5/28/22	33.58165	-94.36528	SVC	869	8812	EF	F	4	-
AR	Red River	5/28/22	33.58165	-94.36528	SVC	616	3122	EF	M	5	-
AR	Red River	5/28/22	33.58165	-94.36528	SVC	850	10284	EF	F	10	-
AR	Red River	5/28/22	33.58165	-94.36528	SVC	921	12020	GN	F	4	-
AR	Red River	5/28/22	33.58165	-94.36528	SVC	907	9692	EF	F	4	-
AR	Red River	5/28/22	33.58165	-94.36528	SVC	891	9318	EF	F	7	-
OK	Muddy Boggy	6/1/22	33.94254	-95.59405	SVC	892	7600	GN	-	6	-
TX	Choctaw	6/3/22	33.72068	-96.39828	SVC	831	7100	JM	-	4	-
TX	Choctaw	6/4/22	33.71952	-96.3907	SVC	-	-	GN	-	8	-
TX	Choctaw	6/4/22	33.71952	-96.3907	SVC	-	-	GN	-	4	-
AR	Red River	6/5/22	33.56936	-94.06402	SVC	964	9500	JM	M	9	-
AR	Red River	6/5/22	33.56936	-94.06402	SVC	891	8000	GN	M	4	-

OK	Red River	6/6/22	33.8032	-94.91955	BHC	1298	-	EF	F	11	-
OK	Red River	6/6/22	33.8032	-94.91955	BHC	1016	-	GN	M	10	-
OK	Red River	6/16/22	33.63824	-94.58038	BHC	1050	16600	GN	-	-	-
AR	Red River	6/21/22	33.58165	-94.36528	BHC	1172	15250	EF	M	15	-
AR	Red River	6/21/22	33.58165	-94.36528	SVC	940	12000	EF	F	5	2293615
AR	Red River	6/21/22	33.58165	-94.36528	SVC	992	12250	EF	F	7	2000224
AR	Red River	6/21/22	33.58165	-94.36528	SVC	999	12250	EF	F	4	2510504
AR	Red River	6/21/22	33.58165	-94.36528	SVC	1014	13500	EF	F	7	1161526
AR	Red River	6/21/22	33.58165	-94.36528	SVC	985	8750	EF	F	4	733982
AR	Red River	6/21/22	33.58165	-94.36528	SVC	952	8250	EF	M	6	-
AR	Red River	6/21/22	33.58165	-94.36528	SVC	949	11000	JM	F	4	-
AR	Red River	6/21/22	33.58165	-94.36528	SVC	942	7500	EF	M	9	-
AR	Red River	6/21/22	33.58165	-94.36528	SVC	901	7400	JM	M	4	-
AR	Red River	6/21/22	33.58165	-94.36528	SVC	1062	13800	EF	F	8	1531954
AR	Red River	6/21/22	33.58165	-94.36528	SVC	849	7100	GN	M	3	-
OK	Webb Creek	6/21/22	33.77355	-96.41837	SVC	820	6500	BF	F	6	-
OK	Kiamichi	6/23/22	33.9605	-95.25517	BHC	1015	10300	GN	M	9	-
OK	Kiamichi	6/23/22	33.9605	-95.25517	BHC	1250	25250	GN	F	16	400378
OK	Kiamichi	6/23/22	33.9605	-95.25517	BHC	1048	11600	GN	M	11	-
OK	Garland Creek	6/24/22	33.92015	-95.07693	BHC	1122	14900	GN	F	5	924037
OK	Garland Creek	6/24/22	33.92015	-95.07693	BHC	1333	13700	GN	M	-	-
OK	Garland Creek	6/24/22	33.92015	-95.07693	BHC	949	11100	GN	M	4	-

OK	Red River	6/24/22	33.91901	-95.07648	SVC	1091	12000	EF	F	12	1698096
OK	Garland Creek	6/24/22	33.92015	-95.07693	SVC	928	-	EF	M	8	-
TX	Pine Creek	6/28/22	33.87272	-95.30441	BHC	952	10200	GN	M	9	-
OK	Red River	6/30/22	33.88492	-95.46896	SVC	900	-	GN	M	6	-
OK	Red River	6/30/22	33.82042	-96.56031	BHC	1040	14850	BF	M	7	-
OK	Muddy Boggy	7/5/22	33.94254	-95.59405	BHC	1033	12000	GN	M	9	-
OK	Muddy Boggy	7/5/22	33.94254	-95.59405	BHC	979	11900	GN	M	12	-
OK	Muddy Boggy	7/5/22	33.94254	-95.59405	BHC	1022	21000	GN	M	11	-
OK	Muddy Boggy	7/5/22	33.94254	-95.59405	BHC	1046	11400	GN	M	12	-
OK	Muddy Boggy	7/5/22	33.94254	-95.59405	BHC	1033	18000	GN	-	11	-
OK	Muddy Boggy	7/5/22	33.94254	-95.59405	SVC	792	6000	GN	M	3	-
OK	Red River	7/14/22	33.91901	-95.07648	SVC	875	7250	EF	M	6	-
TX	Choctaw	7/19/22	33.72068	-96.39828	BHC	1073	20500	GN	F	13	-
TX	Choctaw	7/19/22	33.72068	-96.39828	SVC	808	7000	EF	-	5	-
OK	Red River	7/22/22	33.6583	-94.54367	SVC	881	8500	EF	F	4	1827963
OK	Kiamichi	7/28/22	33.9605	-95.25517	BHC	1051	11600	GN	M	11	-
OK	Kiamichi	7/28/22	33.9605	-95.25517	BHC	1050	11100	GN	M	13	-
OK	Kiamichi	8/1/22	33.96159	-95.28264	BHC	1021	9500	EF	F	10	-
OK	Kiamichi	8/1/22	33.96159	-95.28264	BHC	1201	21100	GN	M	12	-
OK	Kiamichi	8/1/22	33.96159	-95.28264	SVC	748	5100	EF	M	6	-
TX	Bois d'Arc	8/2/22	33.82851	-95.85503	BHC	1000	12900	GN	M	10	-
TX	Bois d'Arc	8/2/22	33.82851	-95.85503	SVC	805	7000	JM	M	3	-

TX	Bois d'Arc	8/2/22	33.82851	-95.85503	SVC	853	7900	GN	M	8	-
TX	Choctaw	8/3/22	33.71952	-96.3907	BHC	1054	19500	GN	F	15	-
TX	Bois d'Arc	8/5/22	33.82851	-95.85503	BHC	1004	10000	GN	M	7	-
TX	Bois d'Arc	8/5/22	33.82851	-95.85503	SVC	814	2500	EF	F	5	-
TX	Bois d'Arc	8/5/22	33.82851	-95.85503	SVC	855	8000	GN	M	10	-
OK	Kiamichi	8/9/22	33.96159	-95.28264	SVC	861	6800	EF	F	7	475807
AR	Red River	8/10/22	33.56399	-94.00924	SVC	945	9000	EF	M	9	-
TX	Bois d'Arc	8/11/22	33.82851	-95.85503	BHC	1105	15500	GN	M	8	-
TX	Bois d'Arc	8/11/22	33.82851	-95.85503	BHC	1018	14500	EF	M	9	-
TX	Bois d'Arc	8/11/22	33.82252	-95.86404	BHC	868	9000	GN	M	11	-
TX	Bois d'Arc	8/11/22	33.82851	-95.85503	BHC	1061	15500	GN	M	12	-
TX	Bois d'Arc	8/11/22	33.82851	-95.85503	SVC	964	13500	EF	F	5	1483368
TX	Bois d'Arc	8/11/22	33.82851	-95.85503	SVC	906	10000	EF	F	9	1328391
TX	Bois d'Arc	8/11/22	33.82851	-95.85503	SVC	902	10000	EF	F	9	-
TX	Bois d'Arc	8/11/22	33.82252	-95.86404	SVC	904	9500	EF	M	6	-
OK	Red River	8/15/22	33.82042	-96.56031	SVC	860	9300	BF	F	8	-
OK	Red River	8/26/22	33.96024	-95.20688	SVC	894	8500	EF	M	7	-
AR	Red River	8/29/22	33.56399	-94.00924	SVC	855	7900	EF	M	5	-
OK	Red River	9/3/22	33.82042	-96.56031	SVC	920	9150	BF	F	8	-
OK	Red River	9/7/22	33.82042	-96.56031	SVC	850	7160	BF	M	10	-
OK	Red River	10/8/22	33.82147	-96.54313	SVC	916	8390	BF	M	3	-
AR	Red River	10/18/22	33.54988	-94.36266	SVC	740	4100	EF	M	3	-

AR	Red River	10/18/22	33.54988	-94.36266	SVC	841	7000	EF	M	5	-
AR	Red River	10/18/22	33.54988	-94.36266	SVC	825	7500	EF	M	9	-
OK	Kiamichi	11/3/22	33.00632	-95.37972	BHC	1020	11200	EF	F	-	507756
OK	Kiamichi	11/3/22	33.00632	-95.37972	BHC	1012	10500	EF	F	-	254817
OK	Cutoff Oxbow	11/7/22	33.75273	-94.75616	BHC	1360	35500	GN	F	-	1369726
AR	Red River	11/8/22	33.56936	-94.06402	SVC	796	5600	EF	M	-	-
AR	Red River	11/8/22	33.56936	-94.06402	SVC	835	6750	GN	M	-	-
AR	Red River	11/15/22	33.56399	-94.00924	SVC	861	7500	GN	M	-	-
AR	Red River	11/16/22	33.54689	-94.38066	BHC	1134	18600	GN	M	-	-
AR	Red River	11/16/22	33.54689	-94.38066	BHC	1133	14450	GN	M	-	-
AR	Red River	11/16/22	33.54689	-94.38066	SVC	844	7100	EF	M	-	-
AR	Red River	11/16/22	33.54689	-94.38066	SVC	801	5200	GN	M	-	-
AR	Red River	11/16/22	33.54689	-94.38066	SVC	753	5200	GN	M	-	-
AR	Red River	1/11/23	33.5515	-94.39453	SVC	920	11100	EF	F	7	-
AR	Red River	1/11/23	33.5515	-94.39453	SVC	942	9700	GN	M	8	-
OK	Garland Creek	2/13/23	33.92473	-95.08337	BHC	1114	22000	EF	-	-	-
OK	Garland Creek	2/13/23	33.92473	-95.08337	BHC	1222	26000	EF	-	-	-
OK	Red River	2/20/23	33.65393	-94.56868	BHC	1050	17000	GN	-	-	-
OK	Red River	2/21/23	33.65225	-94.59476	BHC	1382	37500	EF	-	-	-
OK	Red River	2/21/23	33.65225	-94.59476	BHC	1308	31000	GN	-	-	-
AR	Red River	3/3/23	33.56842	-94.38122	SVC	878	7250	GN	M	8	-
AR	Red River	3/3/23	33.56842	-94.38122	SVC	870	8000	GN	M	6	-

AR	Red River	3/3/23	33.56842	-94.38122	SVC	805	6000	GN	M	6	-
AR	Red River	3/3/23	33.56842	-94.38122	SVC	950	10100	GN	M	7	-
AR	Red River	3/3/23	33.56842	-94.38122	SVC	924	9550	GN	M	9	-
AR	Red River	3/3/23	33.56842	-94.38122	SVC	802	7100	GN	M	8	-
AR	Red River	3/3/23	33.56842	-94.38122	SVC	892	9000	GN	M	9	-
OK	Red River	3/9/23	33.65393	-94.56868	BHC	1240	15000	EF	M	17	-
OK	Red River	3/9/23	33.65393	-94.56868	BHC	1130	19000	GN	M	10	-
OK	Garland Creek	3/17/23	33.92473	-95.08337	SVC	770	3750	GN	M	5	-
AR	Little River	3/29/23	33.63875	-93.89177	BHC	1061	15000	GN	F	10	-
AR	Red River	4/17/23	33.55708	-94.04868	SVC	940	9500	EF	M	11	-
OK	Kiamichi	4/18/23	33.964097	-95.30325	SVC	742	4000	GN	F	5	-
OK	Kiamichi	4/18/23	33.964097	-95.30325	SVC	766	4600	GN	F	5	-
OK	Kiamichi	4/18/23	33.964097	-95.30325	SVC	663	2500	GN	F	4	-
AR	Red River	4/20/23	33.56842	-94.38122	SVC	900	9000	GN	F	8	-
AR	Red River	4/20/23	33.56842	-94.38122	SVC	1051	14000	EF	F	11	-
AR	Red River	4/20/23	33.56842	-94.38122	SVC	871	9000	EF	F	8	-
AR	Red River	4/20/23	33.56842	-94.38122	SVC	944	10500	EF	F	9	-
AR	Red River	4/20/23	33.56842	-94.38122	SVC	1010	13500	EF	F	13	-
OK	Muddy Boggy	4/24/23	33.938676	-95.61715	SVC	647	3000	GN	M	4	-
OK	Kiamichi	4/25/23	33.972609	-95.33212	SVC	804	5000	GN	M	7	-
OK	Red River	4/26/23	33.65393	-94.56868	SVC	996	10000	GN	F	13	-
OK	Kiamichi	4/27/23	33.95071	-95.24384	SVC	860	7100	EF	M	11	-

OK	Kiamichi	4/27/23	33.95071	-95.24384	SVC	742	3500	EF	M	6	-
OK	Kiamichi	4/27/23	33.95071	-95.24384	SVC	614	1800	EF	M	4	-
OK	Kiamichi	4/27/23	33.95071	-95.24384	SVC	870	6000	GN	M	5	-
OK	Kiamichi	4/27/23	33.95071	-95.24384	SVC	964	10000	GN	M	8	-
OK	Kiamichi	4/27/23	33.95071	-95.24384	BHC	1295	28000	GN	M	-	-
OK	Kiamichi	4/27/23	33.95071	-95.24384	BHC	1006	21900	GN	M	10	-
OK	Kiamichi	4/27/23	33.95071	-95.24384	BHC	1258	25100	GN	M	10	-
OK	Kiamichi	4/27/23	33.95071	-95.24384	BHC	1082	15000	GN	M	15	-
OK	Muddy Boggy	5/8/23	33.93833	-95.60911	BHC	1026	14500	GN	M	12	-
AR	Red River	5/10/23	33.60915	-93.8242	SVC	1046	13800	EF	F	10	-
OK	Garland Creek	5/16/23	33.92473	-95.08337	BHC	923	8500	GN	F	12	-
OK	Muddy Boggy	5/24/23	33.93833	-95.60911	BHC	1080	23000	GN	F	13	-
OK	Muddy Boggy	5/24/23	33.93833	-95.60911	BHC	1072	14500	GN	M	9	-
AR	Red River	5/31/23	33.5515	-94.39453	SVC	891	11000	EF	F	5	-
OK	Kiamichi	6/20/23	33.95071	-95.24384	SVC	810	7400	EF	M	5	-
OK	Kiamichi	6/20/23	33.95071	-95.24384	SVC	864	12500	EF	F	7	-
OK	Kiamichi	6/20/23	33.95071	-95.24384	BHC	1191	23000	GN	M	12	-
OK	Kiamichi	6/20/23	33.95071	-95.24384	SVC	835	5900	GN	M	6	-
AR	Red River	7/5/23	33.55708	-94.04868	SVC	962	10050	EF	M	6	-
AR	Red River	7/24/23	33.39703	-93.71171	SVC	921	9250	GN	F	-	-
OK	Kiamichi	7/26/23	33.95071	-95.24384	SVC	900	8900	GN	M	7	-
OK	Kiamichi	8/29/23	33.95071	-95.24384	SVC	825	7000	EF	M	7	-

OK	Kiamichi	10/2/23	33.95071	-95.24384	SVC	1100	14100	JM	F	-	-
AR	Red River	10/19/23	33.56936	-94.06402	SVC	817	7500	EF	M	-	-
AR	Red River	10/25/23	33.3659	-93.70329	SVC	532	1750	EF	F	-	-
OK	Kiamichi	11/9/23	33.00632	-95.37972	BHC	900	8250	EF	M	-	-
OK	Kiamichi	11/9/23	33.00632	-95.37972	BHC	952	9300	EF	F	-	-

Appendix A, Table 13. Carp visually confirmed (i.e., observed jumping or jumped in boat) from May 2021 through November 2023 within a site but not collected during fish sampling on the Red River and its tributaries. The observations indicate the state, date, location, and species observed (SVC =Silver Carp, BHC=Bighead Carp).

River	State	Date	Latitude	Longitude	Species
Muddy Boggy	OK	7/2/21	33.94339	-95.60174	SVC
Muddy Boggy	OK	7/27/21	33.93557	-95.63493	SVC
Muddy Boggy	OK	7/28/21	33.92844	-95.65096	SVC
Red River	OK	7/29/21	33.65393	-94.56868	SVC / BHC
Pine Creek	TX	8/3/21	33.86477	-95.30788	BHC
Red River	AR	8/31/21	33.39703	-93.71171	SVC
Red River	AR	10/8/21	33.39703	-93.71171	SVC
Red River	AR	4/1/22	33.39703	-93.71171	SVC
Red River	AR	4/5/22	33.5515	-94.39453	SVC
Red River	OK	4/19/22	33.88111	-95.50545	SVC
Red River	OK	4/21/22	33.95053	-95.24028	SVC
Red River	AR	4/26/22	33.57537	-94.08128	SVC
Red River	AR	5/6/22	33.5515	-94.39453	SVC
Buzzard Creek	OK	5/9/22	33.90033	-95.05406	SVC
Red River	AR	5/12/22	33.13784	-93.82909	SVC
Garland Creek	OK	5/16/22	33.92473	-95.08337	SVC
Muddy Boggy	OK	5/31/22	33.92844	-95.65096	SVC
Red River	AR	6/7/22	33.57537	-94.08128	SVC
Red River	AR	6/8/22	33.60915	-93.8242	SVC
Red River	AR	6/13/22	33.13784	-93.82909	BHC
Pine Creek	TX	6/14/22	33.86477	-95.30788	SVC
Red River	AR	6/15/22	33.5998	-94.44686	SVC
Red River	AR	6/17/22	33.34793	-93.71021	SVC / BHC
Choctaw	TX	6/22/22	33.72223	-96.41024	SVC

Red River	AR	7/15/22	33.55708	-94.04868	SVC
Red River	AR	7/20/22	33.34793	-93.71021	SVC
Muddy Boggy	OK	7/21/22	33.93833	-95.60911	SVC
Red River	AR	7/25/22	33.60915	-93.8242	SVC
Choctaw	TX	8/3/22	33.72021	-96.37333	SVC
Red River	OK	8/4/22	33.96302	-95.22118	BHC
Red River	AR	8/23/22	33.59898	-93.81232	SVC
Red River	OK	8/26/22	33.96302	-95.22118	BHC
Kiamichi	OK	9/9/22	33.95095	-95.29142	SVC
Red River	AR	9/21/22	33.55718	-94.0195	SVC
Red River	AR	10/12/22	33.34793	-93.71021	SVC
Red River	OK	10/19/22	33.88111	-95.50545	SVC
Red River	AR	10/26/22	33.60915	-93.8242	SVC
Cutoff Oxbow 41	OK	2/9/23	33.74471	-94.7695	SVC / BHC
Garland Creek	OK	3/7/23	33.92473	-95.08337	SVC
Garland Creek	OK	3/17/23	33.92473	-95.08337	BHC
Red River	AR	4/3/23	33.5515	-94.39453	SVC
Red River	AR	4/4/23	33.56842	-94.38122	SVC
Red River	AR	5/4/23	33.5515	-94.39453	SVC / BHC
Red River	AR	5/15/23	33.34793	-93.71021	SVC
Garland Creek	OK	5/25/23	33.92473	-95.08337	BHC
Red River	AR	5/30/23	33.59898	-93.81232	SVC
Red River	AR	6/7/23	33.60915	-93.8242	SVC
Red River	AR	6/8/23	33.55718	-94.0195	SVC
Red River	AR	6/9/23	33.55708	-94.04868	SVC
Red River	AR	6/13/23	33.34793	-93.71021	SVC
Red River	AR	6/15/23	33.55718	-94.0195	SVC
Red River	OK	7/6/23	33.8849	-95.45556	SVC
Pine Creek	TX	7/13/23	33.86477	-95.30788	SVC
Pine Creek	TX	8/7/23	33.86477	-95.30788	SVC
Red River	AR	8/8/23	33.60915	-93.8242	SVC

Red River	AR	8/24/23	33.34793	-93.71021	SVC
Red River	AR	9/21/23	33.07839	-93.84704	SVC
Red River	AR	9/26/23	33.55718	-94.0195	SVC
Red River	AR	10/11/23	33.14741	-93.83134	SVC
Red River	AR	11/6/23	33.55718	-94.0195	SVC
Red River	AR	11/16/23	33.07839	-93.84704	SVC

Appendix A, Table 14. Occupancy model covariate combinations (width-to-depth ratio [W:D], sinuosity [Sin], backwater [Bck], chlorophyll-*a* [Chla], salinity [Sal], discharge [Q], and distance to dam [Dtd]) hypothesized to be related to Carp presence with the corresponding Watanabe-Akaike information criterion (WAIC) and Δ WAIC scores.

Model	WAIC	Δ WAIC
W:D + Sin	249.53	0
Bck	249.73	0.2
Bck + W:D + Sin	250.02	0.49
Bck + W:D	251.04	1.51
Bck + W:D + Chla	251.29	1.76
Bck + Sin	252.91	3.38
Bck + W:D + Sin + Chla	253.68	4.15
W:D	253.81	4.28
Sal	253.92	4.39
Q	254.26	4.73
W:D + Chla	255.03	5.5
Bck + Sin + Chla	255.16	5.63
Chla	255.27	5.74
Sin + Chla	255.71	6.18
Bck + Q	257.14	7.61
Bck + Sal	257.22	7.69
Bck + Chla	257.38	7.85
Q + Sin	258.2	8.67
Sin + Chla	258.39	8.86
Bck + Dtd	258.61	9.08
Bck + Q + W:D	260.02	10.49
Dtd	260.57	11.04
Bck + Q + Sin	260.99	11.46
Q + W:D	261.94	12.41
Bck + Q + Dtd	263.24	13.71
Q + Dtd	263.53	14

Sal + Dtd	265.59	16.06
Bck + Q + W:D + Chla	266.03	16.5
Bck + Sal + Dtd	266.71	17.18
Sal + Dtd + Q	267.02	17.49
Q + Sal	267.27	17.74
Q + Chla	269.1	19.57
Bck + Q + W:D + Sin	270.06	20.53
Bck + Q + Sal + Dtd	272.67	23.14
Bck + Q + Chla	273.31	23.78
Bck + Q + Sal	276.74	27.21
Q + W:D + Sin + Chla	277.31	27.78
Bck + Q + W:D + Sin + Chla	295.63	46.1

Appendix A, Table 15. The mode, 90% highest density interval (HDI), standard error (SE), and Rhat values for occupancy covariates (backwater [Bck], width-to-depth [W:D], chlorophyll-a [Chla], and sinuosity [Sin]) for the top ranked occupancy models for Bighead Carp and Silver Carp collected in the lower Red River catchment from May 2021 through September 2022.

Species	Model	Covariate	Mode	SE	90% HDI	Rhat
Bighead Carp	Bck	Bck	2.348	0.08	(0.04, 7.43)	1.004
	Bck + W:D	Bck	1.193	0.07	(-0.73, 3.21)	1
	Bck + W:D + Chla	Bck	1.203	0.07	(-0.90, 3.42)	1.001
	Bck + W:D + Sin	Bck	2.619	0.08	(-0.15, 6.72)	0.999
	Bck + W:D + Chla	Chla	0.004	0.09	(-1.37, 1.16)	1
	Bck + W:D + Sin	Sin	-1.748	0.06	(-3.28, -0.50)	0.999
	W:D + Sin	Sin	-1.218	0.06	(-2.15, -0.32)	1
	Bck + W:D	W:D	-1.638	0.08	(-3.06, -0.42)	1.001
	Bck + W:D + Chla	W:D	-1.784	0.08	(-3.39, -0.46)	1.001
	Bck + W:D + Sin	W:D	-2.217	0.10	(-4.12, -0.69)	0.999
	W:D + Sin	W:D	-2.051	0.10	(-3.77, -0.68)	1
Silver Carp	Bck	Bck	2.311	0.08	(-0.33, 7.67)	1.003
	Bck + W:D	Bck	1.159	0.07	(-1.01, 3.70)	1
	Bck + W:D + Chla	Bck	1.177	0.07	(-1.24, 3.90)	1.001
	Bck + W:D + Sin	Bck	2.42	0.08	(-0.37, 6.95)	0.999
	Bck + W:D + Chla	Chla	0.621	0.09	(-0.66, 2.00)	1
	Bck + W:D + Sin	Sin	-1.575	0.06	(-3.16, -0.30)	0.999
	W:D + Sin	Sin	-1.176	0.06	(-2.24, -0.25)	1
	Bck + W:D	W:D	-1.177	0.08	(-2.46, -0.02)	1
	Bck + W:D + Chla	W:D	-1.284	0.08	(-2.67, -0.02)	1
	Bck + W:D + Sin	W:D	-1.436	0.10	(-3.05, 0.12)	0.999
	W:D + Sin	W:D	-1.323	0.10	(-2.64, -0.02)	1

Appendix A, Table 16. The mode, 90% highest density interval (HDI), standard error (SE), and Rhat values for detection covariates (discharge [Q], electrofishing effort [Sec], Secchi depth [Secchi], and water temperature [Temp]) for the top ranked models (backwater [Bck], width-to-depth ratio [W:D], Chlorophyll-*a* [Chla], and sinuosity [Sin]) for Bighead Carp (BHC) and Silver Carp (SVC) collected in the lower Red River catchment from May 2021 through September 2022.

Species	Model	Covariate	Mode	SE	90% HDI	Rhat
BHC	Bck	Q	-0.418	0.03	(-0.83, -0.04)	1
	Bck + W:D	Q	-0.462	0.03	(-0.88, -0.06)	1
	Bck + W:D + Chla	Q	-0.465	0.03	(-0.89, -0.07)	1
	Bck + W:D + Sin	Q	-0.437	0.03	(-0.84, -0.05)	1
	W:D + Sin	Q	-0.48	0.03	(-0.90, -0.07)	1
	Bck	Sec	0.69	0.03	(0.24, 1.14)	1
	Bck + W:D	Sec	0.599	0.03	(0.13, 1.05)	1
	Bck + W:D + Chla	Sec	0.597	0.03	(0.14, 1.06)	1
	Bck + W:D + Sin	Sec	0.667	0.03	(0.22, 1.13)	1
	W:D + Sin	Sec	0.584	0.03	(0.10, 1.03)	1
	Bck	Secchi	-0.393	0.04	(-0.90, 0.13)	1.001
	Bck + W:D	Secchi	-0.403	0.04	(-0.90, 0.11)	1
	Bck + W:D + Chla	Secchi	-0.399	0.04	(-0.87, 0.12)	1
	Bck + W:D + Sin	Secchi	-0.457	0.04	(-0.94, 0.04)	1
	W:D + Sin	Secchi	-0.482	0.04	(-0.98, 0.01)	1
	Bck	Temp	0.818	0.04	(0.28, 1.41)	1
	Bck + W:D	Temp	0.736	0.03	(0.22, 1.31)	1
	Bck + W:D + Chla	Temp	0.725	0.03	(0.21, 1.29)	1.001
	Bck + W:D + Sin	Temp	0.836	0.04	(0.31, 1.43)	1
	W:D + Sin	Temp	0.749	0.03	(0.23, 1.33)	1
SVC	Bck	Q	-0.39	0.03	(-0.74, -0.03)	1
	Bck + W:D	Q	-0.418	0.03	(-0.79, -0.04)	1
	Bck + W:D + Chla	Q	-0.41	0.03	(-0.78, -0.04)	1
	Bck + W:D + Sin	Q	-0.408	0.03	(-0.78, -0.05)	1

W:D + Sin	Q	-0.421	0.03	(-0.81, -0.04)	1
Bck	Sec	0.795	0.03	(0.41, 1.21)	1
Bck + W:D	Sec	0.736	0.03	(0.33, 1.16)	1
Bck + W:D + Chla	Sec	0.744	0.03	(0.33, 1.16)	1
Bck + W:D + Sin	Sec	0.764	0.03	(0.38, 1.18)	1
W:D + Sin	Sec	0.743	0.03	(0.34, 1.16)	1
Bck	Secchi	-0.731	0.04	(-1.17, -0.31)	1
Bck + W:D	Secchi	-0.722	0.04	(-1.17, -0.31)	1
Bck + W:D + Chla	Secchi	-0.704	0.04	(-1.14, -0.28)	0.999
Bck + W:D + Sin	Secchi	-0.752	0.04	(-1.19, -0.33)	1
W:D + Sin	Secchi	-0.777	0.04	(-1.22, -0.36)	1
Bck	Temp	0.525	0.04	(0.13, 0.93)	1
Bck + W:D	Temp	0.534	0.03	(0.14, 0.94)	1
Bck + W:D + Chla	Temp	0.522	0.03	(0.12, 0.92)	1
Bck + W:D + Sin	Temp	0.535	0.04	(0.13, 0.95)	1.001
W:D + Sin	Temp	0.527	0.03	(0.12, 0.93)	1

Appendix A, Table 87. Occupancy and detection estimates and corresponding 90% highest density intervals (HDI) for the top ranked models (backwater [Bck], width-to-depth ratio [W:D], chlorophyll-a [Chla], and sinuosity [Sin]) for Silver Carp (SVC) and Bighead Carp (BHC) collected in the Red River catchment from May 2021 through September 2022.

Species	Model	Occupancy	90% HDI	Detection	90% HDI
Silver Carp	Bck	0.83	(0.51, 0.97)	0.6	(0.50, 0.70)
	Bck + W:D	0.78	(0.33, 0.96)	0.61	(0.51, 0.71)
	Bck + W:D + Chla	0.79	(0.32, 0.97)	0.61	(0.50, 0.71)
	Bck + W:D + Sin	0.8	(0.27, 0.98)	0.61	(0.51, 0.71)
	W:D + Sin	0.85	(0.43, 0.97)	0.63	(0.53, 0.72)
Bighead Carp	Bck	0.78	(0.39, 0.96)	0.39	(0.25, 0.54)
	Bck + W:D	0.61	(0.23, 0.91)	0.4	(0.27, 0.55)
	Bck + W:D + Chla	0.65	(0.23, 0.94)	0.4	(0.27, 0.54)
	Bck + W:D + Sin	0.53	(0.15, 0.91)	0.39	(0.27, 0.53)
	W.D + Sin	0.68	(0.29, 0.92)	0.4	(0.28, 0.55)

Appendix A, Table 18. Mean back-calculated length-at-age (mm) for Silver Carp and Bighead Carp collected in 2021-2022 in the lower Red River catchment.

Age (years)	Silver Carp	Bighead Carp
1	261	256
2	446	418
3	590	557
4	689	670
5	743	748
6	777	813
7	824	869
8	850	929
9	882	972
10	904	1009
11	920	1042
12	940	1077
13	917	1104
14	905	1159
15	-	1143
16	-	1175
17	-	1191
18		1122
19		1155
20		1184

Appendix A, Table 19. The top ranked models with the corresponding parameter number (K), Akaike information criterion corrected for small sample size ($AICc$), model difference (ΔAIC), and model weight for models that were averaged for Bighead Carp in the lower Red River catchment collected through October 2022. B0 is the model intercept, A is fish age, T is air temperature, Q is discharge and CV.T is the coefficient of variation of air temperature.

Model	K	AICc	ΔAIC	Weight
~ B0 + A + T	6	7353.49	0	0.53
~ B0 + A + Q + T	7	7354.89	1.40	0.27
~ B0 + A + T + CV.T	7	7355.44	1.96	0.20

Appendix A, Table 20. The top ranked models with the corresponding parameter number (K), Akaike information criterion corrected for small sample size (AICc), model difference (ΔAIC), and model weight for models included in the averaged Weisberg model for Silver Carp in the lower Red River catchment collected through October 2022. B0 is the model intercept, A is fish age, T is air temperature, Q is discharge, CV.T is the coefficient of variation of air temperature, and CV.Q is the coefficient of variation of discharge.

Model	K	AICc	ΔAIC	Weight
$\sim B0 + A + T + Q + CV.T + CV.Q$	9	12710.59	0	0.31
$\sim B0 + A + Q + CV.T + CV.Q$	8	12711.29	0.7	0.22
$\sim B0 + A + T + Q$	7	12711.4	0.81	0.21
$\sim B0 + A + T + Q + CV.T$	8	12712.21	1.62	0.14
$\sim B0 + A + QT + CV.Q$	7	12712.41	1.82	0.13

Appendix A, Table 9. Averaged model estimates for evaluating the relationship between Silver Carp and Bighead Carp growth and environmental factors. The final average Weisberg model estimates with the corresponding standard error (SE), p-value ($\text{Pr}>|z|$), and 90% confidence intervals (90% C.I.) for Bighead Carp and Silver Carp in the lower Red River catchment.

Species	Covariate	Estimate	SE	$\text{Pr}> z $	90% C.I.
Bighead Carp	Age	-9.73	0.50	0.00	(-10.56, -8.90)
	Air temperature	3.82	1.72	0.02	(1.00, 6.65)
	Discharge	0.62	1.80	0.73	(-2.34, 3.58)
	CV of temperature	-0.15	1.10	0.90	(-1.95, 1.68)
Silver Carp	Age	-15.23	0.52	0.00	(-16.09, -14.37)
	Discharge	11.47	7.52	0.13	(-0.90, 23.84)
	Air temperature	9.07	3.49	0.01	(3.33, 14.81)
	CV of temperature	2.81	4.19	0.50	(-4.09, 9.72)
	CV of discharge	-3.63	3.55	0.31	(-9.48, 2.21)

Appendix A, Table 22. The number of individuals, by species and by gear (EF=electrofishing, FN= mini-fyke net, GN = gillnet, HN=hoopnet, LT=larval tow, SE=seine) sampled from the Red River in Arkansas from 2021-2023. Mean total length (TL, mm) is reported for larger-bodied fishes sampled during adult surveys. (Bighead Carp and Silver Carp collections/observations are not included in this table).

Species	EF	FN	GN	HN	LT	SE	TL	Total
Alligator Gar	2	-	39	-	-	-	1348.7	41
American Eel	1	-	-	-	-	-	805.0	1
American Paddlefish	9	-	57	-	-	-	1305.9	66
Bantam Sunfish	-	4	-	-	-	17	-	21
Bigeye Shiner	-	1	-	-	-	1	-	2
Bigmouth Buffalo	175	-	274	1	-	3	614.6	450
Bigscale Logperch	-	30	-	-	-	18	-	48
Black Buffalo	124	-	185	2	-	-	680.9	311
Black Crappie	5	326	-	-	-	63	152.4	394
Blackstripe Topminnow	-	9	-	-	-	29	-	38
Blacktail Shiner	5	3	-	-	-	48	67.75	53
Blue Catfish	193	-	34	-	-	4	422.1	231
Blue Sucker	241	-	17	7	-	-	538.5	265
Bluegill	71	806	-	-	-	760	118.5	1624
Bluntnose Darter	-	3	-	-	-	4	-	7
Bowfin	1	1	-	-	-	-	-	2
Brook Silverside	6	7	-	-	-	98	78	111
Bullhead Minnow	97	3269	-	-	-	17997	64.85	21363
Channel Catfish	47	12	5	-	-	20	350.1	84
Chub Shiner	18	553	-	-	-	7303	60.5	7874
Common Carp	5	-	11	-	-	-	770.1	14
Dorosoma spp.	-	2	-	-	324	21	-	347
Dusky Darter	1	51	-	-	-	18	72.0	70
Emerald Shiner	152	3415	-	-	32	1773	60.2	5372

Flathead Catfish	227	1	2	-	-	2	214.5	232
Flier	1	1	-	-	-	-	42	2
Freshwater Drum	243	74	6	1	10	78	257.4	334
Ghost Shiner	-	14	-	-	-	6	-	20
Gizzard Shad	608	770	11	-	1	2010	202.6	3400
Golden Shiner	-	51	-	-	-	26	-	77
Golden Topminnow	-	17	-	-	-	11	-	28
Grass Carp	12	-	48	-	-	-	960.02	60
Grass Pickerel	1	-	-	-	-	-	104	1
Green Sunfish	38	19	-	-	-	65	99.5	122
Ictiobus spp.	-	-	-	-	3	-	-	3
Largemouth Bass	5	-	-	-	-	-	250	5
Lepomis spp.	1	477	-	-	1	558	84	1037
Longear Sunfish	162	445	-	-	-	1147	102.1	1754
Longnose Gar	145	1	92	4	2	15	908.3	259
Mississippi Silverside	48	1861	-	-	-	2448	81.25	4357
Mississippi Silvery Minnow	-	-	-	-	-	2	-	2
Mosquitofish	1	692	-	-	-	7223	23	7916
Orangespotted Sunfish	25	2164	-	-	-	2217	73.4	4406
Pallid Shiner	-	-	-	-	-	2	-	2
Pirate Perch	-	-	-	-	-	21	-	21
Pomoxis spp.	-	-	-	-	-	24	-	24
Pugnose Minnow	-	-	-	-	-	1	-	1
Quillback	1	-	-	-	-	-	200	1
Red Shiner	2297	15551	-	-	162	41132	51.8	59142
Redear Sunfish	1	5	-	-	-	20	105.0	26
Redspotted Sunfish	1	-	-	-	-	-	69	1
River Carpsucker	518	47	5	3	1	691	298.7	1265
River Darter	-	3	-	-	-	4	-	7
Sand Shiner	-	23	-	-	-	2	-	25
Shoal Chub	3	24	-	-	-	405	71	432

Shortnose Gar	89	79	6	1	-	1	624.8	176
Shovelnose Sturgeon	19	-	1	-	-	-	470	20
Silver Chub	26	290	-	-	-	562	87.1	878
Silverband Shiner	6	3	-	-	-	19	49	28
Skipjack Herring	1	2	-	-	-	4	316	7
Slenderhead Darter	-	-	-	-	-	1	-	1
Slough Darter	-	2	-	-	-	9	-	11
Smallmouth Buffalo	473	1	281	10	-	10	530.9	775
Spotted Bass	153	112	-	-	-	1062	186.6	1327
Spotted Gar	59	4	-	-	-	4	522.5	67
Spotted Sucker	-	2	-	-	-	4	-	6
Striped Bass	2	10	-	-	-	1	171.0	13
Tadpole Madtom	1	2	-	-	-	-	-	3
Threadfin Shad	358	2133	-	-	24	4275	71.03	6790
Warmouth	7	69	-	-	-	201	79.3	277
Western Sand Darter	-	3	-	-	-	17	-	20
Western Starhead Topminnow	-	2	-	-	-	-	-	2
White Bass	30	379	-	-	-	307	211.5	716
White Crappie	27	1478	-	2	-	316	206	1823
Yellow Bass	3	-	-	-	-	-	298.3	3
Yellow Bullhead	-	1	-	-	-	-	-	1

Appendix A, Table 23. The number of individuals, by species and by gear (EF=electrofishing, FN= mini-fyke net, GN = gillnet, HN=hoopnet, LT=larval tow, SE=seine) sampled from the mainstem Red River in Oklahoma from 2021-2023. Mean total length (TL, mm) is provided for larger-bodied fishes sampled during adult surveys. (Bighead Carp and Silver Carp collections/observations are not included in the table).

Species	EF	FN	GN	HN	LT	SE	TL	Total
Alligator Gar	4	-	37	-	-	-	1429.7	41
American Paddlefish	3	-	37	-	-	-	1338.2	40
Bigeye Shiner	-	-	-	-	-	4	-	4
Bigmouth Buffalo	56	1	153	-	-	4	610.2	214
Bigscale Logperch	-	7	-	-	-	11	-	18
Black Buffalo	46	1	152	1	-	-	647.6	200
Black Crappie	-	20	-	-	1	9	-	30
Blackspotted Topminnow	-	-	-	-	-	1	-	1
Blackstripe Topminnow	-	5	-	-	-	3	-	8
Blacktail Shiner	-	8	-	-	-	32	-	40
Blue Catfish	85	1	45	1	-	2	501.5	134
Blue Sucker	185	-	4	1	-	-	512.1	190
Bluegill	18	342	-	-	-	149	94.9	509
Bluntnose Minnow	3	-	-	-	-	-	40.3	3
Brook Silverside	-	7	-	-	-	10	-	17
Bullhead Minnow	38	3209	-	-	-	12536	68.3	15783
Carpionodes spp.	-	-	-	-	-	2	-	2
Catastomidae spp.	-	-	-	-	-	4	-	4
Channel Catfish	46	14	5	-	-	8	278.2	73
Chestnut Lamprey	1	-	-	-	-	-	220.0	1
Chub Shiner	3	740	-	-	2	4433	53.7	5178
Common Carp	2	-	4	-	-	28	633.9	34
Dorosoma spp.	-	6	-	-	2	270	-	278

Dusky Darter	-	-	-	-	-	2	-	2
Emerald Shiner	35	1171	1	-	-	901	64.7	2108
Flathead Catfish	76	1	6	-	-	-	305.1	83
Freshwater Drum	73	495	5	-	4	60	310.5	637
Ghost Shiner	1	348	-	-	-	359	35.0	708
Gizzard Shad	408	1417	2	-	-	770	246.5	2597
Golden Topminnow	-	4	-	-	-	2	-	6
Goldeye	2	-	-	-	-	-	177.5	2
Grass Carp	1	-	32	-	-	-	924.0	33
Green Sunfish	6	80	-	-	-	25	116.5	111
Hybrid Sunfish	-	1	-	-	-	-	-	1
Largemouth Bass	1	-	-	-	-	2	360.0	3
Lepomis spp.	-	151	-	-	-	251	-	402
Longear Sunfish	15	429	-	-	-	321	93.4	765
Longnose Gar	127	24	118	5	-	29	895.8	303
Mississippi Silverside	61	203	-	-	-	6602	66.0	6866
Mississippi Silvery Minnow	2	-	-	-	-	-	82.0	2
Mosquitofish	1	207	-	-	-	2236	25.0	2444
Orangespotted Sunfish	10	293	-	-	-	120	59.5	423
Pallid Shiner	-	-	-	-	-	2	-	2
Pirate Perch	-	2	-	-	-	1	-	3
Plains Killifish	-	-	-	-	-	4	-	4
Pomoxis spp.	-	-	-	-	-	2	-	2
Quillback	5	-	1	-	-	-	425.5	6
Red Shiner	369	16491	1	-	185	37920	56.2	54966
Redear Sunfish	1	24	-	-	-	6	80.0	31
Ribbon Shiner	-	-	-	-	1	-	-	1
River Carpsucker	275	127	5	-	17	727	285.4	1151
River Darter	-	-	-	-	-	1	-	1
Sand Shiner	-	9	-	-	-	76	-	85
Shoal Chub	-	13	-	-	-	760	-	773

Shortnose Gar	34	57	3	1	-	11	587.1	106
Shovelnose Sturgeon	11	-	1	-	-	-	581.5	12
Silver Chub	4	61	0	-	-	135	101.8	200
Silverband Shiner	-	2	-	-	-	3	-	5
Slough Darter	-	1	-	-	-	-	-	1
Smallmouth Buffalo	300	1	296	18	-	11	537.9	626
Spotted Bass	16	194	-	-	-	914	170.2	1124
Spotted Gar	32	6	-	1	-	1	534.3	40
Spotted Sucker	-	-	-	-	-	3	-	3
Striped Bass	6	-	-	-	-	-	267.0	6
Suckermouth Minnow	-	3	-	-	-	15	-	18
Tadpole Madtom	1	-	-	-	-	-	-	1
Threadfin Shad	363	2395	-	-	4	1171	69.2	3933
Warmouth	2	26	-	-	-	127	105.3	155
Weed Shiner	-	-	-	-	-	7	-	7
Western Sand Darter	-	1	-	-	-	48	-	49
White Bass	8	236	-	-	-	128	176.7	372
White Crappie	11	250	-	-	1	201	142.0	463
Yellow Bullhead	1	-	-	-	-	-	74.0	1

Appendix A, Table 24. The number of individuals, by species and by gear (EF=electrofishing, FN= mini-fyke net, GN = gillnet, HN=hoopnet, LT=larval tow, SE=seine) sampled from the tributaries of the Red River in Texas and Oklahoma from 2021-2023. Mean total length (TL, mm) is reported for larger-bodied fishes sampled during adult surveys. (Bighead Carp and Silver Carp collections/observations are not included in this table).

Species	EF	FN	GN	HN	LT	SE	TL	Total
Alligator Gar	2	-	25	-	-	-	1429.9	27
American Eel	1	-	-	-	-	-	768.0	1
American Paddlefish	28	-	123	-	-	-	1302.9	151
Bigeye Shiner	-	-	-	-	-	30	-	30
Bigmouth Buffalo	210	-	352	2	-	-	595.9	564
Bigscale Logperch	2	13	-	-	-	27	57.0	42
Black Buffalo	73	1	173	-	-	-	654.6	247
Black Crappie	2	13	0	-	-	12	174.5	27
Blackstripe Topminnow	2	4	-	-	-	29	-	35
Blacktail Shiner	32	247	-	-	-	1419	32.0	1698
Blue Catfish	36	-	61	-	-	1	642.6	98
Blue Sucker	3	-	3	1	-	-	598.3	7
Bluegill	56	266	1	-	-	146	87.0	469
Bluntnose Darter	-	1	-	-	-	1	-	2
Bluntnose Minnow	2	-	-	-	-	-	-	2
Brook Silverside	2	23	-	-	-	79	63.5	104
Bullhead Minnow	119	1241	-	-	-	11303	62.9	12663
Channel Catfish	13	9	16	-	-	18	505.7	56
Chestnut Lamprey	1	-	-	-	-	-	203.0	1
Chub Shiner	2	2	-	-	-	10	52.0	14
Common Carp	4	1	11	-	-	2	660.0	18
Dorosoma spp.	-	-	-	-	-	20	-	20

Dusky Darter	2	14	-	-	-	10	60.0	26
Emerald Shiner	104	18	-	-	-	128	61.3	250
Flathead Catfish	27	-	3	1	-	-	592.4	31
Freshwater Drum	83	3	24	-	-	11	433.8	121
Ghost Shiner	2	106	-	-	1	113	64.0	222
Gizzard Shad	564	36	58	-	63	501	234.5	1222
Golden Shiner	-	11	-	-	-	-	-	11
Goldeye	1	-	-	-	-	-	424.0	1
Grass Carp	9	-	44	1	-	-	946.6	54
Green Sunfish	6	17	-	-	-	18	69.0	41
Highland Stoneroller	-	-	-	-	-	1	-	1
Hybrid Sunfish	-	-	-	-	-	1	-	1
Largemouth Bass	10	1	-	-	-	2	263.0	13
Lepomis spp.	-	98	-	-	4	254	-	356
Longear Sunfish	97	243	1	-	-	464	85.4	805
Longnose Gar	176	10	39	2	1	12	835.3	240
Mississippi Silverside	47	43	-	-	-	734	-	824
Mooneye	1	-	-	-	-	-	-	1
Mosquitofish	2	209	-	-	-	5616	-	5827
Orangespotted Sunfish	16	189	1	-	-	222	50.7	428
Pallid Shiner	-	4	-	-	-	25	-	29
Pomoxis spp.	-	-	-	-	-	1	-	1
Pugnose Minnow	-	14	-	-	-	6	-	20
Red Shiner	589	1397	2	-	10	8152	47.9	10150
Redear Sunfish	-	-	-	-	-	5	-	5
Redspot Darter	-	-	-	-	-	2	-	2
River Carpsucker	428	7	4	1	-	107	287.1	547
River Darter	1	1	-	-	-	7	62.0	9
Sand Shiner	-	17	-	-	-	321	-	338
Scaly Sand Darter	-	-	-	-	-	4	-	4
Shoal Chub	-	-	-	-	-	1	-	1

Shortnose Gar	63	25	4	-	-	2	620.0	94
Silver Chub	6	4	-	-	-	23	95.5	33
Silverband Shiner	-	10	-	-	-	-	-	10
Slenderhead Darter	-	30	-	-	-	8	-	38
Slough Darter	-	3	-	-	-	50	-	53
Smallmouth Buffalo	285	-	435	6	-	31	533.4	757
Spotted Bass	66	39	-	-	-	666	166.2	771
Spotted Gar	89	3	2	-	-	3	543.6	97
Spotted Sucker	-	-	-	-	-	4	-	4
Striped Bass	-	-	1	-	-	-	122.0	1
Suckermouth Minnow	-	-	-	-	-	19	-	19
Tadpole Madtom	-	-	-	-	-	5	-	5
Threadfin Shad	627	240	-	-	2	1619	64.9	2488
Warmouth	8	14	-	-	-	21	66.3	43
Weed Shiner	-	1	-	-	-	21	-	22
White Bass	17	1	-	-	-	11	227.0	29
White Crappie	38	171	1	1	-	94	135.5	305
Mississippi Silvery Minnow	13	-	-	-	-	-	61.8	13
Yellow Bass	1	-	-	-	-	-	252.0	1

Appendix A, Table 25. List of genera, the number (#) of species within each genus, the total sampled, and the percent of total of all fishes sampled. Several species counts were removed prior to calculating percent of total due to the disproportionately high observations of them including Red Shiner (Cyprinella: 124,258 individuals), Bullhead Minnow (Pimephales: 49,809), Mosquitofish (Gambusia: 16,187), Threadfin Shad (Dorosoma: 13,211), Chub Shiner (Notropis: 13,066), Mississippi Silverside (Menidia: 12,047), Emerald Shiner (Notropis: 7,730), and Gizzard Shad (Dorosoma: 7,219).

Genus	# of species	Total collected	Percent of total
Alosa	1	7	0.02%
Ameiurus	1	2	<0.00%
Amia	1	2	<0.00%
Ammocrypta	3	75	0.19%
Anguilla	1	2	<0.00%
Aphredoderus	1	24	0.06%
Aplodinotus	1	1092	2.73%
Atractosteus	1	110	0.27%
Campostoma	1	1	<0.00%
Carpionodes	2	2970	7.42%
Centrarchus	1	2	<0.00%
Ctenopharyngodon	1	147	0.37%
Cycleptus	1	462	1.15%
Cyprinella	2	1791	4.48%
Cyprinus	1	66	0.16%
Dorosoma	2	-	-
Esox	1	1	<0.00%
Etheostoma	3	76	0.19%
Fundulus	4	120	0.30%
Gambusia	1	-	-

Hiodon	1	3	0.01%
Hiodontiformes	1	1	<0.00%
Hybognathus	1	17	0.04%
Hybopsis	1	33	0.08%
Hypophthalmichthys	2	413	1.03%
Ichthyomyzon	1	2	<0.00%
Ictalurus	2	676	1.69%
Ictiobus	3	4144	10.35%
Labidesthes	1	232	0.58%
Lepisosteus	3	1382	3.45%
Lepomis	8	13813	34.51%
Lythrurus	1	1	<0.00%
Macrhybopsis	2	3217	5.79%
Menidia	1	-	-
Micropterus	2	3243	8.10%
Minytrema	1	13	0.03%
Morone	3	1141	2.85%
Notemigonus	1	88	0.22%
Notropis	7	1506	3.76%
Noturus	1	9	0.02%
Opsopoeodus	1	21	0.05%
Percina	4	262	0.65%
Phenacobius	1	37	0.09%
Pimephales	2	5	0.01%
Polyodon	1	257	0.64%
Pomoxis	2	3069	7.67%
Pylodictis	1	346	0.86%
Scaphirhynchus	1	32	0.08%

Appendix A, Table 26. Model estimates from the final adult occupancy model. ψ (Psi) and p are the group mean occupancy and detection estimates within the study area respectively. $\hat{R}$ (R-hat) is the measure of model convergence. $\hat{c}$ (c-hat) is a measure of posterior dispersion. The Bayesian p-value represents the goodness-of-fit test for the model. Segment and year sigma are a measure of the variance captured by the grouping factors. Lower (LHDI) and upper (UHDI) 95% high density intervals.

Coefficient	Mean	LHDI	UHDI
ψ (group)	0.715	0.417	0.953
p (group)	0.516	0.417	0.617
$\hat{R}$	1.000	0.999	1.003
$\hat{c}$	1.004	0.942	1.070
Bayesian p-value	0.471	0.00	1.00
Segment - Sigma	0.239	0.000	0.549
Year - Sigma	1.234	0.373	2.457

Appendix A, Table 27. Detection model coefficients for adult species (probability scale) and covariates (logit scale) included in the final model, and their lower (LHDI) and upper (UHDI) 95% high density intervals. Water clarity (Secchi) was collected during each survey and effort is electrofishing time (seconds).

Coefficient	Median	LHDI	UHDI
Effort	0.376	0.281	0.474
Secchi	0.063	-0.031	0.157
Alligator Gar	0.334	0.231	0.455
Bighead Carp	0.421	0.300	0.552
Bigmouth Buffalo	0.799	0.716	0.866
Black Buffalo	0.742	0.666	0.804
Blue Catfish	0.605	0.515	0.689
Blue Sucker	0.551	0.441	0.655
Bluegill	0.482	0.348	0.620
Channel Catfish	0.457	0.343	0.574
Common Carp	0.188	0.098	0.328
Flathead Catfish	0.540	0.431	0.648
Freshwater Drum	0.662	0.567	0.747
Grass Carp	0.406	0.292	0.533
Green Sunfish	0.227	0.109	0.430
Longear Sunfish	0.571	0.442	0.695
Longnose Gar	0.796	0.727	0.851
Orangespotted Sunfish	0.266	0.156	0.420
Paddlefish	0.607	0.465	0.734
River Carpsucker	0.721	0.646	0.789
Shortnose Gar	0.583	0.476	0.683

Shovelnose Sturgeon	0.171	0.066	0.384
Silver Carp	0.541	0.425	0.651
Smallmouth Buffalo	0.900	0.849	0.937
Spotted Bass	0.576	0.451	0.697
Spotted Gar	0.424	0.325	0.525
White Bass	0.334	0.208	0.490

Appendix A, Table 28. Occupancy model coefficients for adult species (probability scale) and their covariates (logit scale) for the final model, and their lower (LHDI) and upper (UHDI) 95% high density intervals. The occupancy coefficient represents the probability of species occupancy within the study area. Continuous variables included were distance from the nearest upstream dam (Dam Distance), median discharge for the season (Discharge), the elevation of the segment, the sinuosity of the segment, average measured salinity (ppt), and the width-to-depth ratio of the reach (W:D). Categorical variables were 1) backwater: present was the reference and was considered when >1% the site area was backwater and 2) drainage area: where high drainage area was the reference (>80,000 km²).

Coefficient	Median	LHDI	UHDI
Alligator Gar - Occupancy	0.804	0.390	0.969
Bighead Carp - Occupancy	0.601	0.188	0.910
Bigmouth Buffalo - Occupancy	0.809	0.413	0.961
Black Buffalo - Occupancy	0.946	0.728	0.991
Blue Catfish - Occupancy	0.920	0.636	0.987
Blue Sucker - Occupancy	0.676	0.260	0.921
Bluegill - Occupancy	0.660	0.255	0.917
Channel Catfish - Occupancy	0.633	0.223	0.903
Common Carp - Occupancy	0.403	0.079	0.837
Flathead Catfish - Occupancy	0.704	0.277	0.926
Freshwater Drum - Occupancy	0.882	0.547	0.976
Grass Carp - Occupancy	0.736	0.305	0.946
Green Sunfish - Occupancy	0.302	0.056	0.748
Longear Sunfish - Occupancy	0.628	0.230	0.908
Longnose Gar - Occupancy	0.964	0.793	0.995
Orangespotted Sunfish - Occupancy	0.437	0.109	0.840
Paddlefish - Occupancy	0.545	0.153	0.878

River Carpsucker - Occupancy	0.941	0.698	0.991
Shortnose Gar - Occupancy	0.790	0.403	0.954
Shovelnose Sturgeon - Occupancy	0.193	0.028	0.664
Silver Carp - Occupancy	0.863	0.486	0.979
Smallmouth Buffalo - Occupancy	0.970	0.826	0.995
Spotted Bass - Occupancy	0.623	0.230	0.907
Spotted Gar - Occupancy	0.833	0.443	0.969
White Bass - Occupancy	0.407	0.096	0.800
Alligator Gar - Backwater	-0.249	-1.721	1.255
Alligator Gar - Drainage	-0.577	-2.007	0.967
Alligator Gar - Elevation	-0.056	-0.77	0.861
Alligator Gar - Sinuosity	-0.191	-0.697	0.209
Alligator Gar - Salinity	-0.090	-0.665	0.579
Alligator Gar - W:D	-0.180	-1.077	0.699
Alligator Gar - Dam distance	-0.244	-0.757	0.319
Alligator Gar - Discharge	-0.037	-1.068	0.677
Bighead Carp - Backwater	1.234	-0.235	2.823
Bighead Carp - Drainage	-0.886	-2.568	0.484
Bighead Carp - Elevation	0.109	-0.610	1.055
Bighead Carp - Sinuosity	-0.169	-0.628	0.230
Bighead Carp - Salinity	-0.241	-0.900	0.370
Bighead Carp - W:D	-1.235	-2.525	-0.144
Bighead Carp - Dam distance	-0.393	-1.06	0.141
Bighead Carp - Discharge	-0.668	-2.528	0.368
Bigmouth Buffalo - Backwater	1.535	0.202	2.931
Bigmouth Buffalo - Drainage	-0.872	-2.377	0.502

Bigmouth Buffalo - Elevation	-0.292	-0.933	0.321
Bigmouth Buffalo - Sinuosity	-0.104	-0.495	0.309
Bigmouth Buffalo - Salinity	-0.077	-0.588	0.554
Bigmouth Buffalo - W:D	-1.197	-2.156	-0.319
Bigmouth Buffalo - Dam distance	-0.169	-0.63	0.396
Bigmouth Buffalo - Discharge	-0.502	-1.994	0.371
Black Buffalo - Backwater	0.545	-0.893	2.184
Black Buffalo - Drainage	-0.539	-1.912	1.037
Black Buffalo - Elevation	-0.131	-0.785	0.664
Black Buffalo - Sinuosity	-0.203	-0.696	0.194
Black Buffalo - Salinity	-0.202	-0.833	0.431
Black Buffalo - W:D	0.112	-0.77	1.022
Black Buffalo - Dam distance	-0.361	-0.95	0.141
Black Buffalo - Discharge	-0.18	-1.177	0.551
Blue Catfish - Backwater	-0.602	-2.199	0.852
Blue Catfish - Drainage	-0.331	-1.741	1.576
Blue Catfish - Elevation	-0.279	-1.01	0.425
Blue Catfish - Sinuosity	-0.035	-0.449	0.574
Blue Catfish - Salinity	-0.399	-1.139	0.16
Blue Catfish - W:D	0.459	-0.491	1.446
Blue Catfish - Dam distance	-0.047	-0.596	0.746
Blue Catfish - Discharge	-0.347	-2.674	0.547
Blue Sucker - Backwater	-0.39	-1.807	0.973
Blue Sucker - Drainage	-0.615	-2.026	0.839
Blue Sucker - Elevation	-0.363	-1.087	0.256
Blue Sucker - Sinuosity	-0.177	-0.66	0.216

Blue Sucker - Salinity	-0.02	-0.573	0.659
Blue Sucker - W:D	2.123	0.967	3.41
Blue Sucker - Dam distance	-0.271	-0.807	0.297
Blue Sucker - Discharge	0.133	-0.787	1.261
Bluegill - Backwater	-0.287	-1.626	1.037
Bluegill - Drainage	-0.974	-2.692	0.342
Bluegill - Elevation	-0.311	-0.989	0.306
Bluegill - Sinuosity	-0.155	-0.576	0.231
Bluegill - Salinity	-0.356	-1.005	0.181
Bluegill - W:D	0.263	-0.569	1.098
Bluegill - Dam distance	-0.444	-1.034	0.021
Bluegill - Discharge	0.187	-0.448	0.955
Channel Catfish - Backwater	-0.473	-1.806	0.834
Channel Catfish - Drainage	-0.566	-1.919	0.889
Channel Catfish - Elevation	-0.097	-0.7	0.596
Channel Catfish - Sinuosity	-0.145	-0.558	0.247
Channel Catfish - Salinity	-0.346	-0.969	0.19
Channel Catfish - W:D	1.084	0.209	2.026
Channel Catfish - Dam distance	-0.316	-0.844	0.166
Channel Catfish - Discharge	0.343	-0.479	1.487
Common Carp - Backwater	1.347	-0.385	3.365
Common Carp - Drainage	-0.607	-2.079	0.899
Common Carp - Elevation	-0.288	-1.008	0.389
Common Carp - Sinuosity	-0.173	-0.668	0.233
Common Carp - Salinity	-0.169	-0.796	0.556
Common Carp - W:D	-0.499	-1.964	0.799

Common Carp - Dam distance	-0.354	-1.028	0.208
Common Carp - Discharge	0.223	-1.029	1.679
Flathead Catfish - Backwater	-1.123	-2.61	0.21
Flathead Catfish - Drainage	-0.469	-1.829	1.104
Flathead Catfish - Elevation	-0.204	-0.858	0.49
Flathead Catfish - Sinuosity	-0.173	-0.624	0.211
Flathead Catfish - Salinity	-0.064	-0.586	0.58
Flathead Catfish - W:D	1.489	0.531	2.531
Flathead Catfish - Dam distance	-0.381	-0.954	0.09
Flathead Catfish - Discharge	0.137	-0.637	1.067
Freshwater Drum - Backwater	-1.199	-2.632	0.127
Freshwater Drum - Drainage	-0.476	-1.79	1.074
Freshwater Drum - Elevation	-0.437	-1.155	0.166
Freshwater Drum - Sinuosity	-0.108	-0.501	0.312
Freshwater Drum - Salinity	-0.277	-0.87	0.267
Freshwater Drum - W:D	0.333	-0.468	1.167
Freshwater Drum - Dam distance	-0.224	-0.697	0.326
Freshwater Drum - Discharge	0.07	-0.628	0.768
Grass Carp - Backwater	0.125	-1.336	1.634
Grass Carp - Drainage	-0.718	-2.177	0.665
Grass Carp - Elevation	-0.296	-0.981	0.324
Grass Carp - Sinuosity	-0.122	-0.547	0.301
Grass Carp - Salinity	-0.174	-0.771	0.452
Grass Carp - W:D	-0.519	-1.851	0.597
Grass Carp - Dam distance	-0.248	-0.771	0.348
Grass Carp - Discharge	0.211	-0.986	1.305

Green Sunfish - Backwater	0.009	-1.702	1.655
Green Sunfish - Drainage	-0.798	-2.443	0.621
Green Sunfish - Elevation	-0.456	-1.321	0.234
Green Sunfish - Sinuosity	-0.133	-0.572	0.321
Green Sunfish - Salinity	-0.284	-1.016	0.328
Green Sunfish - W:D	0.312	-0.784	1.367
Green Sunfish - Dam distance	-0.502	-1.291	0.043
Green Sunfish - Discharge	0.009	-1.309	1.315
Longear Sunfish - Backwater	-0.449	-1.757	0.868
Longear Sunfish - Drainage	-0.914	-2.527	0.418
Longear Sunfish - Elevation	-0.368	-1.038	0.233
Longear Sunfish - Sinuosity	-0.132	-0.535	0.271
Longear Sunfish - Salinity	-0.437	-1.132	0.116
Longear Sunfish - W:D	0.323	-0.481	1.144
Longear Sunfish - Dam distance	-0.296	-0.777	0.203
Longear Sunfish - Discharge	0.438	-0.356	1.672
Longnose Gar - Backwater	-0.251	-1.763	1.25
Longnose Gar - Drainage	-0.462	-1.836	1.278
Longnose Gar - Elevation	-0.203	-0.969	0.614
Longnose Gar - Sinuosity	-0.152	-0.596	0.27
Longnose Gar - Salinity	-0.192	-0.785	0.451
Longnose Gar - W:D	0.240	-0.764	1.174
Longnose Gar - Dam distance	-0.236	-0.767	0.39
Longnose Gar - Discharge	0.577	-0.111	1.392
Orangespotted Sunfish - Backwater	0.381	-1.119	1.932
Orangespotted Sunfish - Drainage	-0.704	-2.197	0.709

Orangespotted Sunfish - Elevation	-0.285	-1.032	0.403
Orangespotted Sunfish - Sinuosity	-0.180	-0.664	0.210
Orangespotted Sunfish - Salinity	-0.37	-1.126	0.210
Orangespotted Sunfish - W:D	-0.066	-1.264	1.089
Orangespotted Sunfish - Dam distance	-0.384	-1.038	0.147
Orangespotted Sunfish - Discharge	0.359	-0.572	1.703
Paddlefish - Backwater	1.522	0.161	2.988
Paddlefish - Drainage	-0.996	-2.793	0.377
Paddlefish - Elevation	0.202	-0.533	1.137
Paddlefish - Sinuosity	-0.168	-0.601	0.217
Paddlefish - Salinity	-0.179	-0.722	0.416
Paddlefish - W:D	-0.492	-1.510	0.442
Paddlefish - Dam distance	-0.261	-0.751	0.278
Paddlefish - Discharge	-0.745	-2.986	0.361
River Carpsucker - Backwater	0.487	-1.085	2.373
River Carpsucker - Drainage	-0.321	-1.725	1.610
River Carpsucker - Elevation	-0.269	-1.016	0.478
River Carpsucker - Sinuosity	-0.199	-0.703	0.198
River Carpsucker - Salinity	-0.041	-0.621	0.708
River Carpsucker - W:D	0.298	-0.655	1.237
River Carpsucker - Dam distance	-0.324	-0.913	0.217
River Carpsucker - Discharge	0.473	-0.163	1.257
Shortnose Gar - Backwater	-0.272	-1.532	0.894
Shortnose Gar - Drainage	-0.632	-1.943	0.799
Shortnose Gar - Elevation	-0.106	-0.709	0.601
Shortnose Gar - Sinuosity	-0.099	-0.488	0.330

Shortnose Gar - Salinity	-0.116	-0.654	0.448
Shortnose Gar - W:D	0.106	-0.677	0.89
Shortnose Gar - Dam distance	-0.224	-0.686	0.308
Shortnose Gar - Discharge	0.692	-0.120	1.875
Shovelnose Sturgeon - Backwater	0.302	-1.458	2.083
Shovelnose Sturgeon - Drainage	-0.681	-2.238	0.884
Shovelnose Sturgeon - Elevation	-0.404	-1.309	0.333
Shovelnose Sturgeon - Sinuosity	-0.146	-0.616	0.280
Shovelnose Sturgeon - Salinity	-0.169	-0.85	0.559
Shovelnose Sturgeon - W:D	1.169	-0.035	2.482
Shovelnose Sturgeon - Dam distance	-0.329	-0.96	0.258
Shovelnose Sturgeon - Discharge	0.016	-1.397	1.372
Silver Carp - Backwater	0.977	-0.467	2.662
Silver Carp - Drainage	-0.614	-2.079	0.875
Silver Carp - Elevation	-0.260	-1.02	0.511
Silver Carp - Sinuosity	-0.134	-0.57	0.297
Silver Carp - Salinity	0.187	-0.491	1.091
Silver Carp - W:D	-0.657	-1.763	0.381
Silver Carp - Dam distance	-0.278	-0.793	0.272
Silver Carp - Discharge	-0.970	-3.53	0.370
Smallmouth Buffalo - Backwater	0.596	-0.915	2.34
Smallmouth Buffalo - Drainage	-0.575	-1.993	0.983
Smallmouth Buffalo - Elevation	-0.136	-0.808	0.661
Smallmouth Buffalo - Sinuosity	-0.162	-0.612	0.244
Smallmouth Buffalo - Salinity	-0.107	-0.696	0.620
Smallmouth Buffalo - W:D	-0.378	-1.412	0.643

Smallmouth Buffalo - Dam distance	-0.198	-0.715	0.434
Smallmouth Buffalo - Discharge	-0.208	-1.631	0.665
Spotted Bass - Backwater	-0.419	-1.659	0.812
Spotted Bass - Drainage	-0.813	-2.299	0.497
Spotted Bass - Elevation	-0.365	-1.004	0.212
Spotted Bass - Sinuosity	-0.156	-0.552	0.226
Spotted Bass - Salinity	-0.354	-0.972	0.183
Spotted Bass - W:D	0.378	-0.393	1.170
Spotted Bass - Dam distance	-0.549	-1.178	-0.053
Spotted Bass - Discharge	-0.218	-0.888	0.37
Spotted Gar - Backwater	-0.588	-2.019	0.769
Spotted Gar - Drainage	-0.594	-1.996	0.964
Spotted Gar - Elevation	-0.224	-0.923	0.505
Spotted Gar - Sinuosity	-0.148	-0.576	0.247
Spotted Gar - Salinity	-0.314	-0.999	0.25
Spotted Gar - W:D	-0.246	-1.098	0.571
Spotted Gar - Dam distance	-0.306	-0.838	0.223
Spotted Gar - Discharge	0.091	-0.813	0.931
White Bass - Backwater	-0.509	-2.103	0.974
White Bass - Drainage	-0.624	-2.031	0.854
White Bass - Elevation	-0.643	-1.604	0.07
White Bass - Sinuosity	-0.105	-0.512	0.355
White Bass - Salinity	-0.498	-1.327	0.129
White Bass - W:D	0.210	-0.771	1.234
White Bass - Dam distance	-0.443	-1.095	0.055
White Bass - Discharge	0.233	-0.687	1.424

Appendix A, Table 29. Mean values for several field-collected habitat variables from sites during the occupancy season by region (Tribes = tributaries): Temp. = water temperature (°C), D.O. = dissolved oxygen (ppt), max depth = maximum depth (m), and W:D = width-to-depth ratio.

Region	Temp.	Secchi	Salinity	Conductivity	Max depth	W:D	D.O.
AR	27.88	30.90	0.38	734.43	3.82	115.26	7.02
Red	(3.33)	(13.23)	(0.20)	(361.39)	(1.88)	(69.79)	(2.28)
OK	28.23	42.59	0.56	1069.73	2.53	137.59	7.78
Red	(2.57)	(31.29)	(0.24)	(372.52)	(0.78)	(63.71)	(3.40)
Tribes	27.96	35.65	0.31	580.91	3.09	26.87	6.95
	(3.31)	(15.47)	(0.27)	(490.81)	(1.30)	(11.26)	(2.86)

Appendix A, Table 30. Summary statistics of movement for Silver Carp (male (M) or female (F)) and Bighead Carp (male (M) or female (F)). Movement distances were calculated as the distance moved between consecutive detections. Movement rates were calculated as the movement distance divided by the duration of time between consecutive detections. Means, standard deviations ($\pm$), maximum distance, and maximum rate are reported.

Species	Sex	Mean distance (km)	Mean rate (km/d)	Maximum distance (km)	Maximum rate (km/d)
Silver	M	1.43 $\pm$ 0.31	0.40 $\pm$ 0.03	206.99	18.02
Silver	F	0.37 $\pm$.03	0.29 $\pm$ 0.02	26.39	3.73
Bighead	M	1.87 $\pm$ 0.43	0.46 $\pm$.04	319.19	25.70
Bighead	F	3.96 $\pm$ 1.35	0.45 $\pm$ 0.13	336.57	37.89

Appendix A, Table 31. Exploratory movement distance and rate models investigating linear and smooth terms for each covariate. MT = mean water temperature (°C), MQ = mean discharge (m³/s), CVT = CV of water temperature, CVD = CV of discharge, W = weight of individual carp (g), and s() denotes smoothed terms. A random effect of individual was included in all models.

Model description
$\sim s(\text{MT}) + s(\text{MQ}) + s(\text{CVT}) + s(\text{CVD}) + s(\text{W})$
$\sim \text{MT} + s(\text{MQ}) + s(\text{CVT}) + s(\text{CVD}) + s(\text{W})$
$\sim s(\text{MT}) + \text{MQ} + s(\text{CVT}) + s(\text{CVD}) + s(\text{W})$
$\sim s(\text{MT}) + s(\text{MQ}) + \text{CVT} + s(\text{CVD}) + s(\text{W})$
$\sim s(\text{MT}) + s(\text{MQ}) + s(\text{CVT}) + \text{CVD} + s(\text{W})$
$\sim s(\text{MT}) + s(\text{MQ}) + s(\text{CVT}) + s(\text{CVD}) + \text{W}$

Appendix A, Table 32. Summary of top movement distance model. Model = response variable, Covariates = covariates included in the model, CVD = coefficient of variation discharge, CVT= coefficient of variation of water temperature MQ = mean discharge (m^3/s), Individual = random effect of individual fish, and edf = effective degrees of freedom. s() denotes smoothed terms. X denotes an interactive effect.

Model	Covariate	edf
Distance	s(CVD)	7.9
	s(CVT x MQ)	27.39
	s(Individual)	35.55

Appendix A, Table 33. Summary of top movement distance and movement rate models. Model = response variable for each model, Model description = covariates included either additive (+) or interactive (x), CVD = coefficient of variation discharge, CVT= coefficient of variation of water temperature MQ = mean discharge (m³/s). s() denotes smoothed terms. The random effect of individual was included in all models. df = degrees of freedom, logLik = log-likelihood AICc = AIC corrected for small sample size, Δ AICc = the difference between each model and the top ranked model, w_i = Akaike weight, and Deviance explained = proportion of the deviance in residuals that is explained by fixed and random effects.

Model	Model description	df	logLik	AICc	Δ AICc	w_i	Deviance explained
Distance	~s(CVD) + s(CVT x MQ)	73	-3338.524	6825.8	0	1	71%
Rate	~s(CVD) + s(CVT x MQ)	68	- 1153.404	2446.2	0	1	39.2%

Appendix A, Table 34. Summary of top movement rate model. Model = response variable, Covariates = covariates included in the model, CVD = coefficient of variation discharge, CVT= coefficient of variation of water temperature MQ = mean discharge (m^3/s), Individual = the random effect of individual, and edf = effective degrees of freedom, s() denotes smoothed terms. X denotes an interactive effect.

Model	Covariate	edf
Rate	s(CVD)	5.45
	s(CVT x MQ)	26.3
	s(Individual)	34.27

Appendix A, Table 35. Probability of remaining in a given state (Ψ) and time constant average survival probability for each state (s) from our two-state model. Parameters = Probabilities of remaining in each state (A = backwater/tributary state, M = mainstem state) across each time period and time constant survival estimates for each state. Mode = most likely value, C.I. = 95% credibility intervals.

Parameter	Mode	C.I.
$\Psi(A-A, t1)$	0.62	(0.45, 0.77)
$\Psi(A-A, t2)$	0.95	(0.76, 1)
$\Psi(A-A, t3)$	0.95	(0.77, 1)
$\Psi(A-A, t4)$	0.63	(0.40, 0.79)
$\Psi(A-A, t5)$	0.89	(0.53, 1)
$\Psi(M-M, t1)$	0.41	(0.0, 0.94)
$\Psi(M-M, t2)$	0.96	(0.732, 1)
$\Psi(M-M, t3)$	0.97	(0.71, 1)
$\Psi(M-M, t4)$	0.61	(0.22, 0.9)
$\Psi(M-M, t5)$	0.23	(0.0, 0.61)
$s(A, .)$	0.98	(0.92, 1)
$s(M, .)$	0.85	(.72, 0.95)

Appendix A, Table 36. Recapture probability estimates (p) from the two-state model. Parameters = detection probabilities for each state (A = backwater/tributary state, M = mainstem state) within each time period (t2-t6), Mode = Most likely value, C.I. = 95% credibility intervals.

Parameter	Estimate	C.I.
p(A,t2)	0.90	(0.73, 0.99)
p(A,t3)	0.86	(0.68, 0.97)
p(A,t4)	0.89	(0.70, 0.99)
p(A,t5)	0.88	(0.62, 0.99)
p(A,t6)	0.67	(0.45, 0.95)
p(M,t2)	0.37	(0.17, 0.6)
p(M,t3)	0.10	(0.02, 0.3)
p(M,t4)	0.12	(0.02, 0.34)
p(M,t5)	0.79	(0.52, 0.96)
p(M,t6)	0.45	(0.09, 0.95)

Appendix A, Table 37. Catch-per-unit-effort data for Bighead Carp (BHC) and Silver Carp (SVC) sampled 2021-2023. Latitude (Lat.) and Longitude (Long.) represent the coordinates from the most upstream end of the sample site. Gillnet effort (GN) is the number of fish collected per hour set of our gillnet complex (three, 180ft long gillnets in the mainstem Red River and 3, 100ft long gillnets in the tributaries). Electrofishing effort (EF) is the number of fish collected per hour of button time. Season reflects our warm sampling season (April through September) and our cold sampling season (October through March). Rivers include Blue River (BR), Bois d’Arc Creek (BD), Buzzard Creek (BC), Choctaw Creek (CC), Garland Creek (GC), Kiamichi River (KR), Muddy Boggy Creek (MB), 41 Cutoff Oxbow Lake (OX), Pine Creek (PC), and the Red River (RR). Although we are providing these data, we did not meet the assumptions associated with catch-per-unit effort data (e.g., equal detection, see detection probability sections within the report; detection varies with the environment); thus, **these data should not be used as a comparison to future sampling efforts** (i.e., they do not reflect trends in fish abundance). However, these data may provide insight to locations where it is simply easier to catch fish under the sampling conditions at that time (e.g., a low flow period during the cold season).

River	Lat.	Long.	Season	BHC EF (fish/hr)	BHC GN (fish/hr)	SVC EF (fish/hr)	SVC GN (fish/hr)
BC	33.90033	-95.05406	Warm	0.00	0.00	0.00	0.00
BD	33.83864	-95.84481	Warm	0.38	0.27	1.53	0.08
BD	33.82314	-95.85532	Warm	0.00	0.06	0.69	0.00
BR	33.88604	-95.93205	Warm	0.00	0.00	0.00	0.00
OX	33.75583	-94.77789	Warm	0.00	0.00	0.00	0.00
CC	33.72021	-96.37333	Warm	0.00	0.22	0.38	0.13
CC	33.72223	-96.41024	Warm	0.00	0.19	2.39	0.00

GC	33.92473	-95.08337	Warm	0.00	0.15	0.84	0.00
KR	33.99742	-95.3722	Warm	0.00	0.00	0.00	0.00
KR	33.98678	-95.3671	Warm	0.00	NA	0.00	NA
KR	33.95071	-95.24384	Warm	0.21	0.17	0.86	0.10
KR	33.94832	-95.29562	Warm	NA	0.00	NA	0.19
KR	33.95095	-95.29142	Warm	0.40	0.05	0.80	0.00
MB	33.94339	-95.60174	Warm	0.00	0.25	0.00	0.06
MB	33.93557	-95.63493	Warm	0.00	0.00	0.00	0.00
MB	33.92844	-95.65096	Warm	0.00	0.00	0.00	0.00
PC	33.86477	-95.30788	Warm	0.00	0.03	0.00	0.00
RR	33.85966	-95.0235	Warm	0.00	0.00	0.00	0.00
RR	33.75426	-96.41081	Warm	0.00	0.00	0.00	0.00
RR	33.68924	-94.68239	Warm	0.00	NA	0.00	NA
RR	33.55708	-94.04868	Warm	0.00	0.00	0.59	0.02
RR	33.88111	-95.50545	Warm	0.00	0.00	0.00	0.00
RR	33.86761	-95.03386	Warm	0.00	0.00	0.00	0.00
RR	33.60915	-93.8242	Warm	0.00	0.00	0.58	0.00
RR	33.89959	-95.06521	Warm	NA	0.00	NA	0.00

RR	33.56842	-94.38122	Warm	0.00	0.16	4.90	0.16
RR	33.58881	-94.37804	Warm	0.98	0.03	4.56	0.28
RR	33.43524	-93.73965	Warm	NA	0.00	NA	0.00
RR	33.09082	-93.85964	Warm	0.00	0.00	0.00	0.00
RR	33.65393	-94.56868	Warm	0.00	0.07	0.00	0.00
RR	33.5515	-94.39453	Warm	0.00	0.00	0.24	0.00
RR	33.71146	-94.73273	Warm	NA	0.00	NA	0.00
RR	33.87724	-95.48534	Warm	0.00	0.00	0.00	0.03
RR	33.90802	-95.06658	Warm	0.00	0.04	1.21	0.00
RR	33.81967	-96.55652	Warm	0.00	0.00	0.00	0.00
RR	33.71665	-96.36472	Warm	0.00	0.00	0.00	0.00
RR	33.66246	-94.64803	Warm	0.00	0.00	0.00	0.00
RR	33.60932	-93.85986	Warm	0.00	0.00	0.00	0.00
RR	33.80257	-94.9285	Warm	0.64	0.12	0.00	0.06
RR	33.64846	-94.54315	Warm	0.00	0.00	0.56	0.00
RR	33.39703	-93.71171	Warm	0.00	0.00	0.00	0.02
RR	33.06602	-93.83293	Warm	0.00	0.00	0.00	0.00
RR	33.1568	-93.81832	Warm	0.00	0.00	0.00	0.00

RR	33.95053	-95.24028	Warm	0.00	0.00	0.00	0.00
RR	33.8897	-95.52022	Warm	0.00	0.00	0.00	0.00
RR	33.57537	-94.08128	Warm	0.00	0.00	0.00	0.00
RR	33.5998	-94.44686	Warm	0.00	0.00	0.00	0.00
RR	33.59526	-94.42342	Warm	0.00	0.00	0.00	0.00
RR	33.55718	-94.0195	Warm	0.00	0.00	0.45	0.00
RR	33.34793	-93.71021	Warm	0.00	0.00	0.00	0.00
RR	33.61833	-94.55481	Warm	0.00	0.00	0.00	0.00
RR	33.14741	-93.83134	Warm	0.00	0.00	0.00	0.00
RR	33.13784	-93.82909	Warm	0.00	0.00	0.00	0.00
RR	33.96302	-95.22118	Warm	0.00	0.00	0.57	0.00
RR	33.59898	-93.81232	Warm	0.00	0.00	0.00	0.00
RR	33.63781	-94.541388	Warm	0.00	0.00	0.00	0.00
RR	33.04394	-93.8302	Warm	0.00	0.00	0.00	0.00
RR	33.8849	-95.45556	Warm	0.00	0.00	0.00	0.00
RR	33.07839	-93.84704	Warm	0.00	NA	0.00	NA
BC	33.90033	-95.05406	Cold	0.00	0.00	0.00	0.00
OX	33.75583	-94.77789	Cold	0.00	0.27	0.00	0.00

CC	33.72021	-96.37333	Cold	2.14	0.14	1.60	0.28
CC	33.72223	-96.41024	Cold	0.00	0.00	0.00	0.08
GC	33.92473	-95.08337	Cold	2.71	0.00	0.00	0.17
KR	33.99742	-95.3722	Cold	1.28	0.00	0.00	0.00
KR	33.95071	-95.24384	Cold	0.00	0.00	0.00	0.00
KR	33.94832	-95.29562	Cold	0.00	0.00	0.00	0.00
RR	33.75426	-96.41081	Cold	0.00	0.00	0.00	0.00
RR	33.55708	-94.04868	Cold	0.00	0.00	0.81	0.04
RR	33.88111	-95.50545	Cold	0.00	0.00	0.00	0.00
RR	33.60915	-93.8242	Cold	0.00	0.00	0.00	0.00
RR	33.56842	-94.38122	Cold	0.00	0.22	23.21	1.07
RR	33.58881	-94.37804	Cold	0.00	0.09	10.47	1.11
RR	33.43524	-93.73965	Cold	NA	0.00	NA	0.00
RR	33.09082	-93.85964	Cold	0.00	0.00	0.00	0.00
RR	33.65393	-94.56868	Cold	1.76	0.33	0.00	0.00
RR	33.5515	-94.39453	Cold	0.00	0.06	1.17	0.08
RR	33.71146	-94.73273	Cold	0.00	0.00	0.00	0.00
RR	33.87724	-95.48534	Cold	0.00	0.00	0.00	0.00

RR	33.66246	-94.64803	Cold	0.00	0.00	0.00	0.00
RR	33.60932	-93.85986	Cold	NA	0.00	NA	0.00
RR	33.80257	-94.9285	Cold	0.00	0.00	0.00	0.00
RR	33.64846	-94.54315	Cold	0.00	0.00	0.00	0.00
RR	33.39703	-93.71171	Cold	0.00	0.00	0.00	0.00
RR	33.95053	-95.24028	Cold	0.00	0.00	0.00	0.00
RR	33.57537	-94.08128	Cold	NA	0.00	NA	0.00
RR	33.5998	-94.44686	Cold	0.00	0.00	0.00	0.00
RR	33.07597	-93.8387	Cold	0.00	0.00	0.00	0.11
RR	33.69385	-94.71692	Cold	0.00	0.00	0.00	0.00
RR	33.59526	-94.42342	Cold	0.00	0.00	0.00	0.00
RR	33.76464	-96.41476	Cold	0.00	0.00	0.00	0.00
RR	33.55226	-94.04026	Cold	0.00	0.00	0.00	0.00
RR	33.55718	-94.0195	Cold	0.00	0.00	0.00	0.06
RR	33.34793	-93.71021	Cold	0.00	0.00	0.43	0.00
RR	33.61833	-94.55481	Cold	0.00	0.00	0.00	0.00
RR	33.77695	-96.42174	Cold	1.26	0.50	0.00	2.50
RR	33.14741	-93.83134	Cold	0.00	NA	0.00	NA

RR	33.13784	-93.82909	Cold	0.00	0.00	0.00	0.00
RR	33.63781	-94.541388	Cold	0.00	0.00	0.00	0.00
RR	33.04394	-93.8302	Cold	0.00	0.00	0.00	0.00
RR	33.65225	-94.59476	Cold	1.67	0.17	0.00	0.00
RR	33.07839	-93.84704	Cold	0.00	NA	0.00	NA

Appendix A, Table 38. The common name with the corresponding scientific name for fish species sampled in the lower Red River catchment of Arkansas, Oklahoma, and Texas.

Common Name	Scientific Name
Alligator Gar	<i>Atractosteus spatula</i>
American Eel	<i>Anguilla rostrata</i>
American Paddlefish	<i>Polyodon spathula</i>
Bantam Sunfish	<i>Lepomis symmetricus</i>
Bigeye Shiner	<i>Notropis boops</i>
Bighead Carp	<i>Hypophthalmichthys nobilis</i>
Bigmouth Buffalo	<i>Ictiobus cyprinellus</i>
Bigscale Logperch	<i>Percina macrolepida</i>
Black Buffalo	<i>Ictiobus niger</i>
Black Crappie	<i>Pomoxis nigromaculatus</i>
Blackspotted Topminnow	<i>Fundulus olivaceus</i>
Blackstripe Topminnow	<i>Fundulus notatus</i>
Blacktail Shiner	<i>Cyprinella venusta</i>
Blue Catfish	<i>Ictalurus furcatus</i>
Blue Sucker	<i>Cycleptus elongatus</i>
Bluegill	<i>Lepomis macrochirus</i>
Bluntnose Darter	<i>Etheostoma chlorosomum</i>
Bluntnose Minnow	<i>Pimephales notatus</i>
Bowfin	<i>Amia calva</i>
Brook Silverside	<i>Labidesthes sicculus</i>
Bullhead Minnow	<i>Pimephales vigilax</i>
Channel Catfish	<i>Ictalurus punctatus</i>
Chestnut Lamprey	<i>Ichthyomyzon castaneus</i>
Chub Shiner	<i>Notropis potteri</i>
Common Carp	<i>Cyprinus carpio</i>
Dusky Darter	<i>Percina sciera</i>
Emerald Shiner	<i>Notropis atherinoides</i>
Flathead Catfish	<i>Pylodictis olivaris</i>

Flier	<i>Centrarchus macropterus</i>
Freshwater Drum	<i>Aplodinotus grunniens</i>
Ghost Shiner	<i>Notropis buchanani</i>
Gizzard Shad	<i>Dorosoma cepedianum</i>
Golden Shiner	<i>Notemigonus crysoleucas</i>
Golden Topminnow	<i>Fundulus chrysotus</i>
Goldeye	<i>Hiodon alosoides</i>
Grass Carp	<i>Ctenopharyngodon idella</i>
Grass Pickerel	<i>Esox americanus</i>
Green Sunfish	<i>Lepomis cyanellus</i>
Highland Stoneroller	<i>Campostoma spadiceum</i>
Hybrid Sunfish	<i>Lepomis spp.</i>
Largemouth Bass	<i>Micropterus salmoides</i>
Longear Sunfish	<i>Lepomis megalotis</i>
Longnose Gar	<i>Lepisosteus osseus</i>
Mississippi Silverside	<i>Menidia audens</i>
Mississippi Silvery Minnow	<i>Hybognathus nuchalis</i>
Mooneye	<i>Hiodon tergisus</i>
Mosquitofish	<i>Gambusia affinis</i>
Orangespotted Sunfish	<i>Lepomis humilis</i>
Pallid Shiner	<i>Hybopsis amnis</i>
Pirate Perch	<i>Aphredoderus sayanus</i>
Plains Killifish	<i>Fundulus zebrinus</i>
Pugnose Minnow	<i>Opsopoeodus emiliae</i>
Quillback	<i>Carpiodes cyprinus</i>
Red Shiner	<i>Cyprinella lutrensis</i>
Redear Sunfish	<i>Lepomis microlophus</i>
Redspot Darter	<i>Etheostoma artesia</i>
Redspotted Sunfish	<i>Lepomis miniatus</i>
Ribbon Shiner	<i>Lythrurus fumeus</i>
River Carpsucker	<i>Carpiodes carpio</i>

River Darter	<i>Percina shumardi</i>
Sand Shiner	<i>Notropis stramineus</i>
Scaly Sand Darter	<i>Ammocrypta vivax</i>
Shoal Chub	<i>Macrhybopsis hyostoma</i>
Shortnose Gar	<i>Lepisosteus platostomus</i>
Shovelnose Sturgeon	<i>Scaphirhynchus platyrhynchus</i>
Silver Carp	<i>Hypophthalmichthys molitrix</i>
Silver Chub	<i>Macrhybopsis storeriana</i>
Silverband Shiner	<i>Notropis shumardi</i>
Skipjack Herring	<i>Alosa chrysochloris</i>
Slenderhead Darter	<i>Percina phoxocephala</i>
Slough Darter	<i>Etheostoma gracile</i>
Smallmouth Buffalo	<i>Ictiobus Bubalus</i>
Spotted Bass	<i>Micropterus punctulatus</i>
Spotted Gar	<i>Lepisosteus oculatus</i>
Spotted Sucker	<i>Minytrema melanops</i>
Striped Bass	<i>Morone saxatilis</i>
Suckermouth Minnow	<i>Phenacobius mirabilis</i>
Tadpole Madtom	<i>Noturus gyrinus</i>
Threadfin Shad	<i>Dorosoma petenense</i>
Warmouth	<i>Lepomis gulosus</i>
Weed Shiner	<i>Notropis Texanus</i>
Western Sand Darter	<i>Ammocrypta clara</i>
Western Starhead Topminnow	<i>Fundulus blairae</i>
White Bass	<i>Morone chrysops</i>
White Crappie	<i>Pomoxis annularis</i>
Yellow Bass	<i>Morone mississippiensis</i>
Yellow Bullhead	<i>Ameiurus natalis</i>

Appendix A, Table 39. Coordinates of SURs used for passive tracking tagged carp.

Latitude	Longitude	Description
33.93851	-95.61553	Muddy boggy Creek
33.95471	-95.25122	Kiamichi River
33.11436	-93.86507	Sulphur River
33.96027	-95.26954	Kiamichi River
33.94137	-95.60529	Muddy boggy Creek
33.01086	-93.80905	AR-LA border
33.12278	-93.87703	Sulphur River
33.77541	-96.42056	Near dam top
33.57639	-94.37097	Spanish Bluffs oxbow
33.77184	-96.41735	Near dam
33.57149	-94.37942	Spanish Bluffs Oxbow
33.0142	-93.80605	AR-LA border
33.61217	-93.82195	Little River confluence
33.64075	-94.57764	Oxbow
33.83744	-95.84824	Bois D'arc Creek
33.6149	-93.85767	Little River
33.61578	-93.83864	Little River
33.60315	-93.8121	Little River confluence
33.65751	-94.5619	Mainstem river
33.83298	-95.85223	Bois D'arc Creek
33.86109	-95.03302	Big Pine Creek
33.86204	-95.03845	Big Pine Creek

Appendix A, Table 40. Pearson's Correlation matrix for movement distance and rate model covariates. Values $> |0.6|$ were considered multicollinear. Only one of each multicollinear pair was retained in future model building. MQ = mean discharge (m^3/s), MT = mean water temperature ($^{\circ}\text{C}$), MPP = Mean photo period (Min), DOS = day of study, CVT= coefficient of variation of water temperature, CVD = coefficient of variation discharge, MT3db = mean water temperature of the three days before a movement ($^{\circ}\text{C}$), MQ3db = mean discharge of the three days before a movement (m^3/s), W = weight of individual carp (g), and L = total length of individual carp (mm).

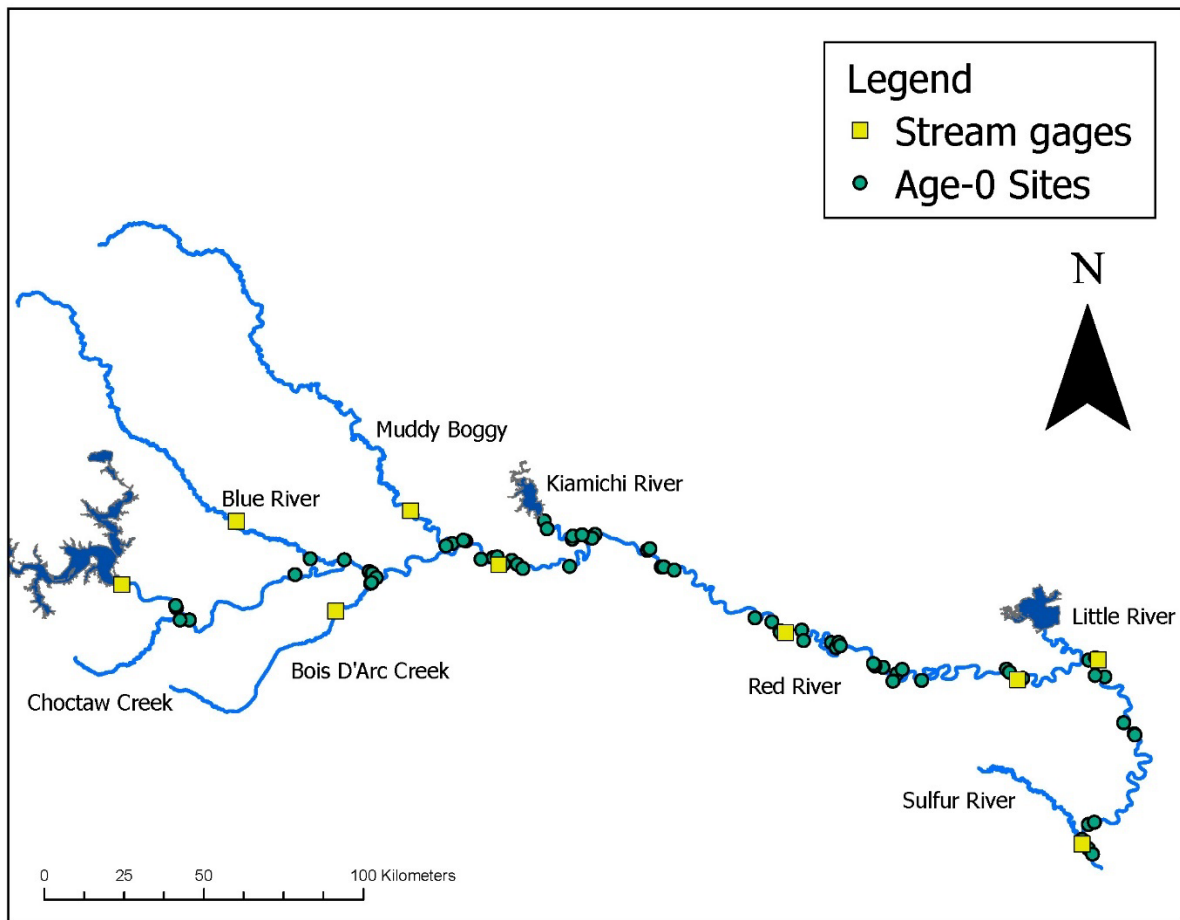
Covariates	MQ	MT	MPP	DOS	CVT	CVD	MT3db	MQ3db	W	L
MQ	1.00									
MT	0.06	1.00								
MPP	0.18	0.90	1.00							
DOS	0.11	0.78	0.70	1.00						
CVT	0.03	-0.49	-0.42	-0.32	1.00					
CVD	0.25	-0.11	-0.06	-0.13	0.23	1.00				
MT3db	0.05	0.99	0.88	0.78	-0.50	-0.13	1.00			
MQ3db	0.91	0.07	0.19	0.12	-0.03	0.17	0.06	1.00		
W	-0.02	-0.02	0.01	-0.01	0.08	0.03	-0.03	-0.02	1.00	
L	-0.10	-0.02	0.00	0.00	0.07	0.01	-0.02	-0.09	0.93	1.00

Appendix A, Table 41. Hypothesized movement distance and rate models. DOS = day of study, MT = mean water temperature (°C), MQ = mean discharge (m³/s), CVT= coefficient of variation of water temperature, CVD = coefficient of variation discharge, W = weight of individual carp (g), Species = Bighead Carp or Silver Carp, Sex = sex of individual, Season = my three season labels (Winter, Spring, and Summer), Direction = upstream or downstream movement based on the previous detection, Hypothesis = overarching hypothesis that may relate to carp movement patterns. s() denotes smoothed terms. The random effect of individual was included in all models. Preliminary models to assess temporal autocorrelation are denoted with “*”.

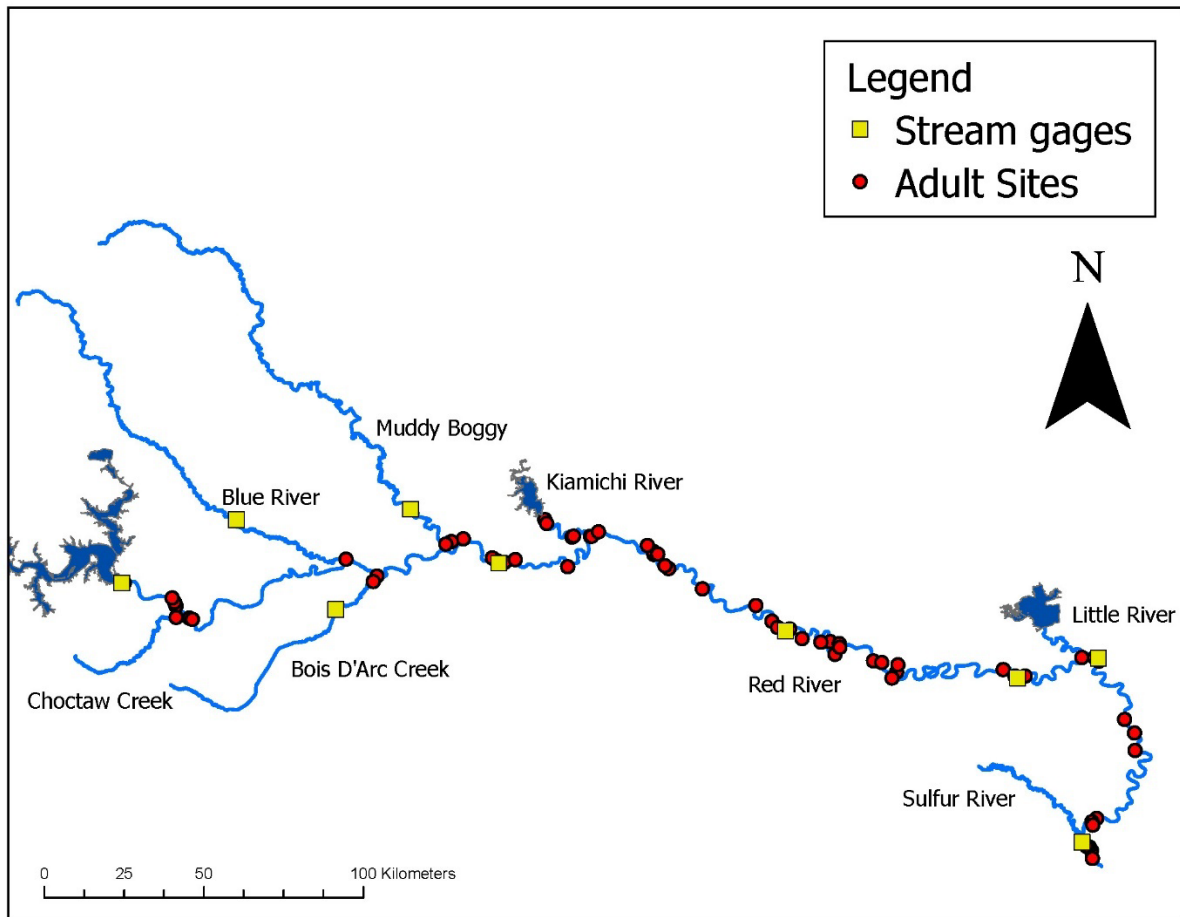
Model description	Hypothesis
~s(DOS)*	Exploratory
~s(DOS) + corARMA(~1 DOS)*	Exploratory
~s(W) + s(MQ) + Sex	Individual traits
~s(W) + s(MT) + Sex	Individual traits
~s(W) + s(MQ x Species)	Individual traits
~s(MQ x Season)	Season and direction
~s(MT x Season)	Season and direction
~s(MQ x Direction)	Season and direction
~s(MT x Direction)	Season and direction
~s(CVT) + s(CVD)	Variability
~s(CVD x Temp)	Variability
~s(CVT x MQ)	Variability

Appendix A, Table 42. Summary of all tracking reaches that were used to actively track tagged Bighead Carp and Silver Carp. Reach identification letter (Figure 6), length of the river reach (km), and the total number of river kilometers actively tracked within each tracking reach are included.

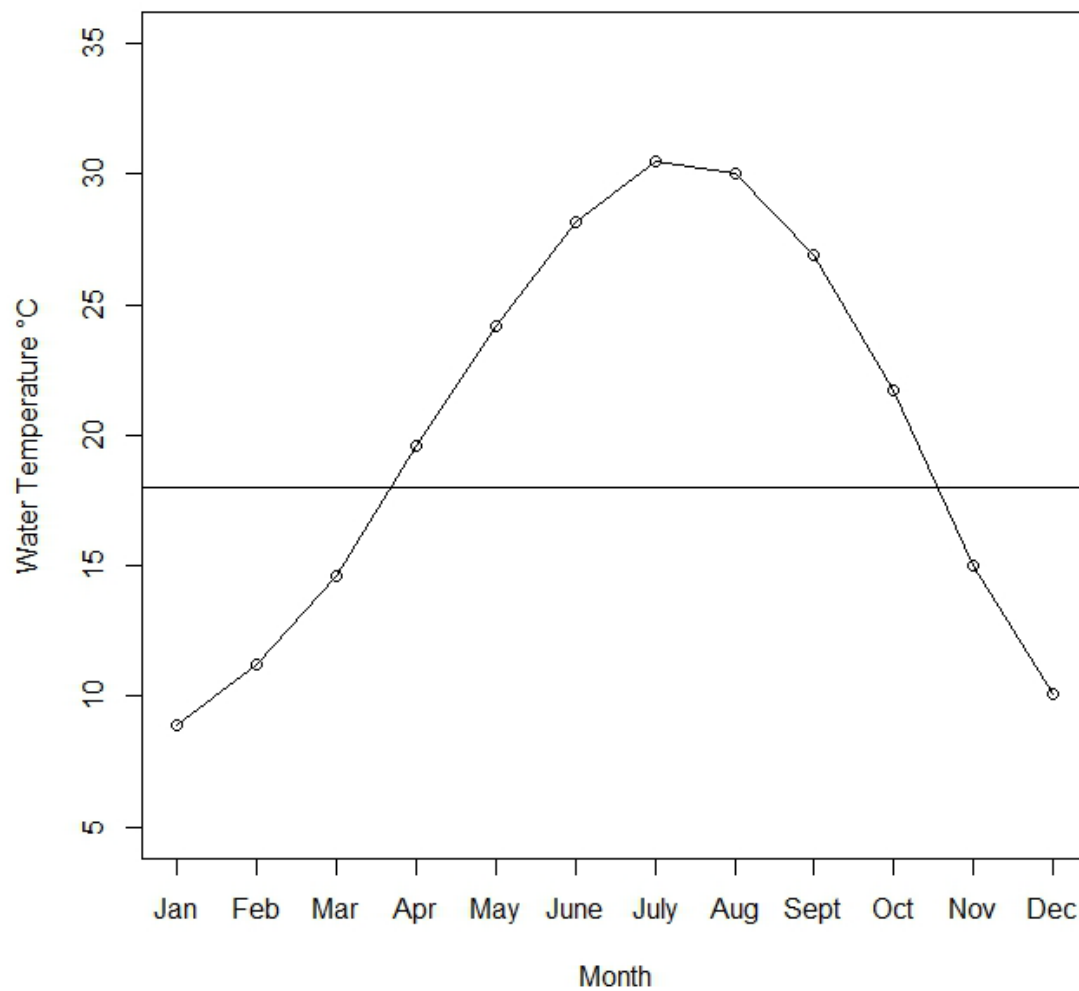
Reach	Length of reach	River km
J	24	504
G	23	552
F	31	682
K	48	576
A	25	25
I	35	385
C	22	352
E	33	495
H	21	567
D	37	370
B	20	240
Total	319	4748



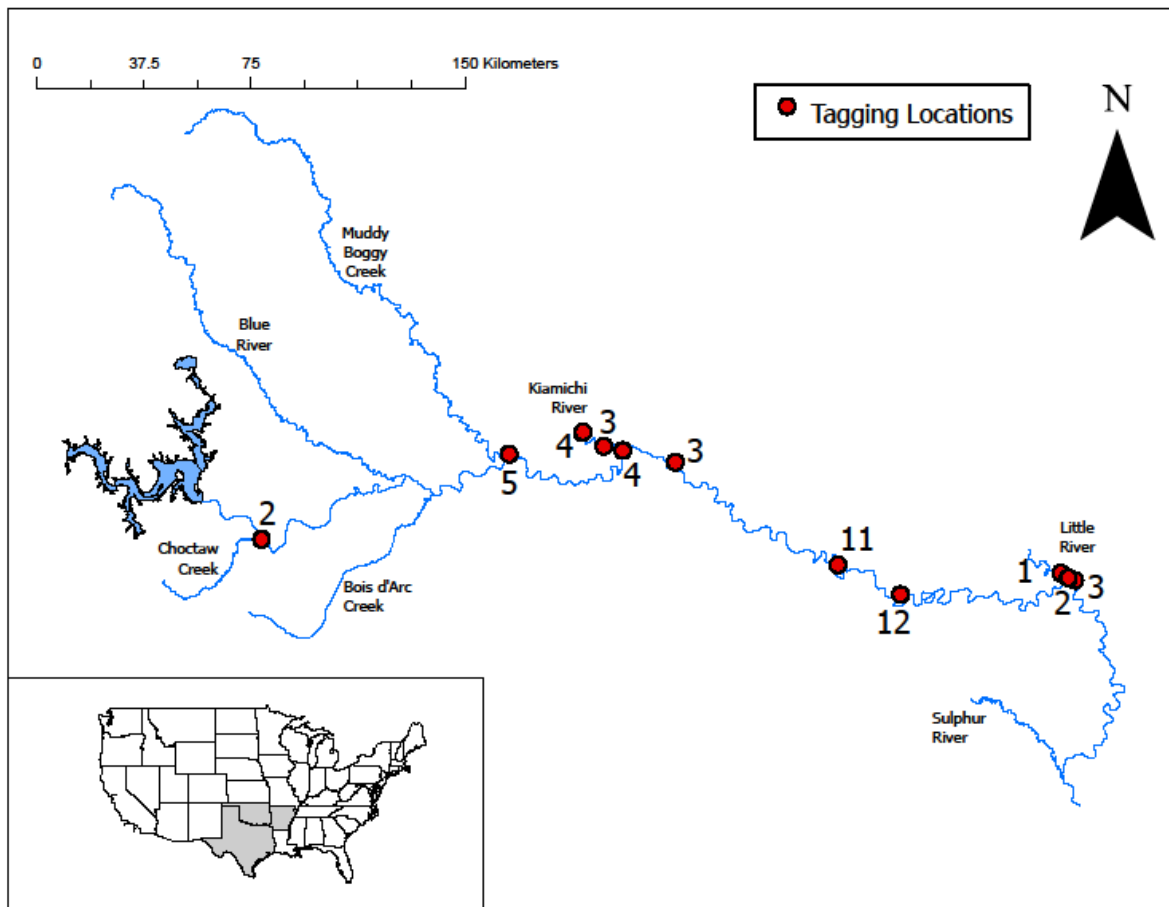
Appendix A, Figure 1. Age-0 fish sampling locations (circles) in the lower Red River basin. The green dots denote the study sites throughout the basin. The blue lines represent major rivers with yellow boxes denoting U.S. Geological Survey stream gages. Each sampling reach was sampled 1-3 times May through early October 2021-2022 using seines, mini-fyke nets, and larval tows.



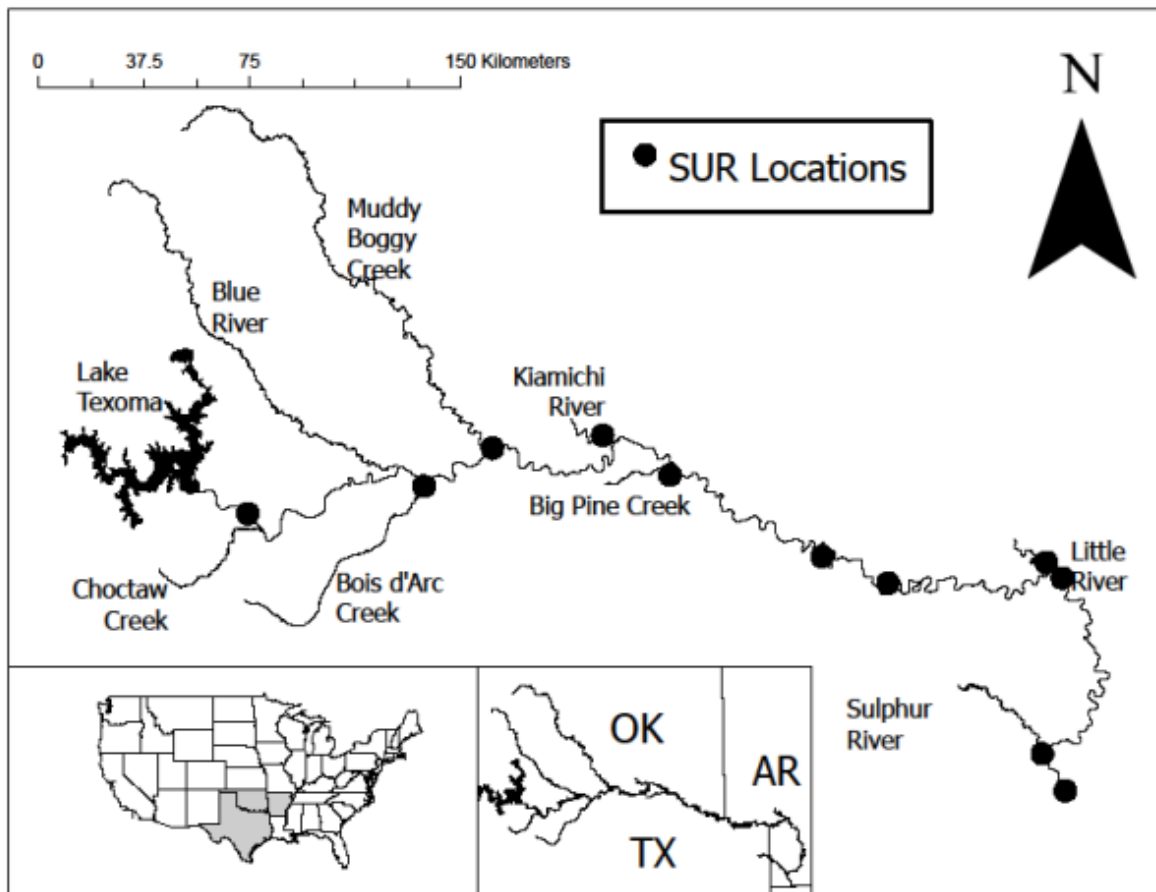
Appendix A, Figure 2. Adult fish sampling locations (red circles) in the lower Red River basin. The blue lines represent major rivers with yellow boxes denoting U.S. Geological Survey stream gages. Each site was sampled 1-3 times in April through September 2021-2023 using gillnets, electrofishing, and hoop nets.



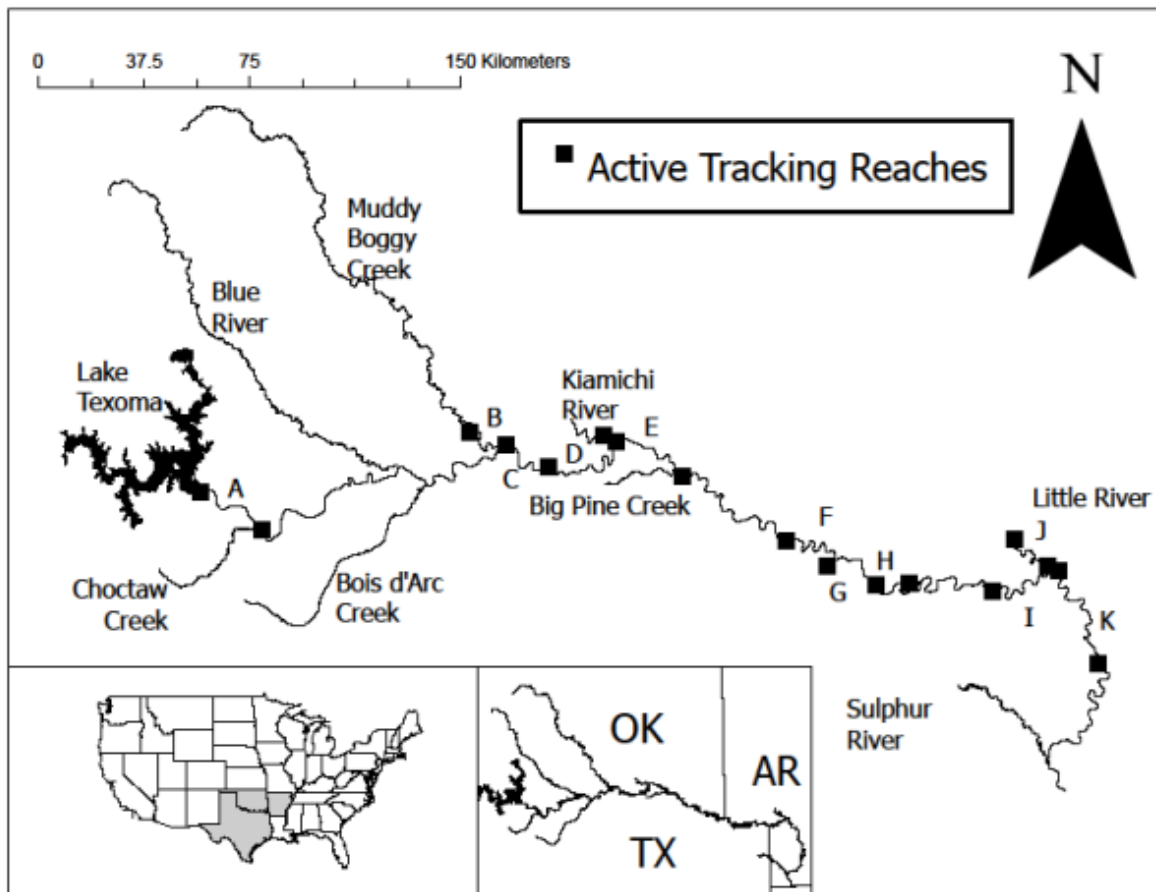
Appendix A, Figure 3. The mean monthly water temperature (°C) for the lower Red River (1997 to 2021) from the U.S. Geological Survey stream gage located near Index, AR (07337000). The horizontal line indicates 18 °C, which was hypothesized to be required for Carp spawning.



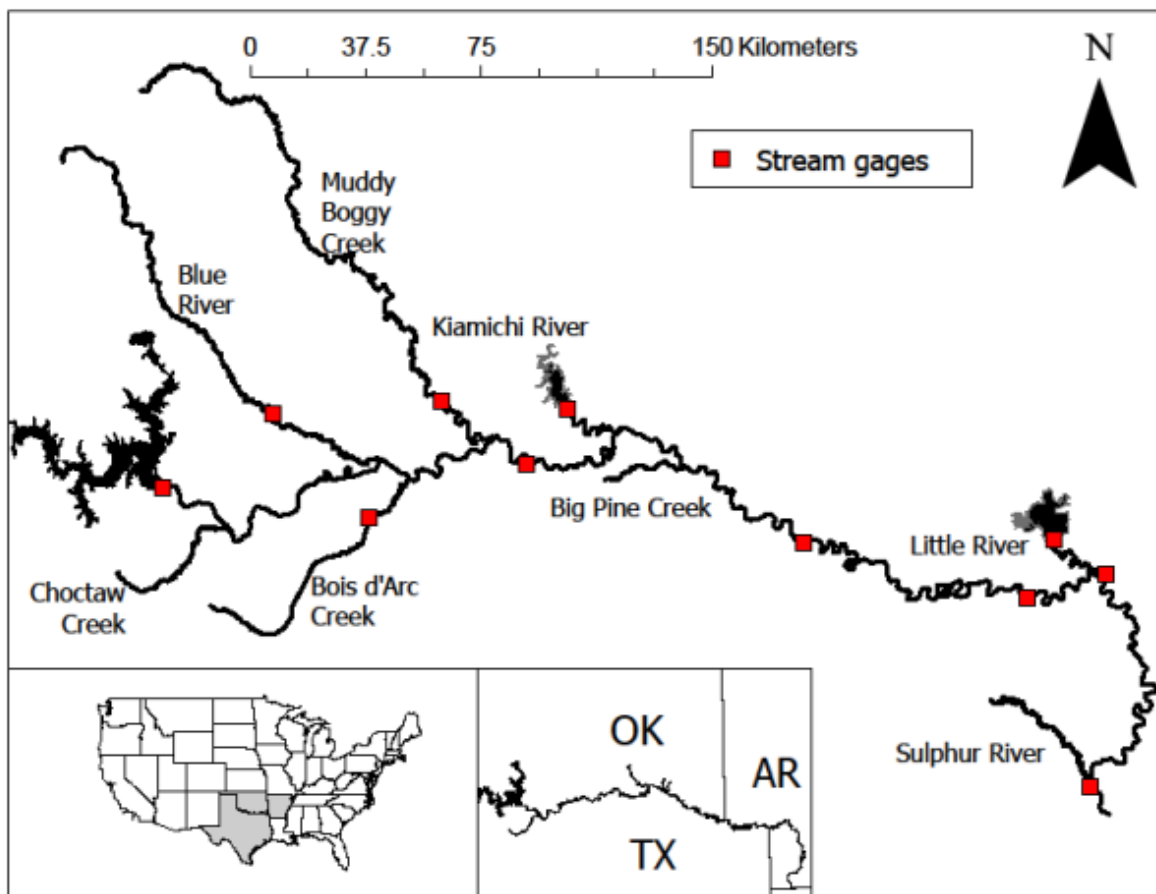
Appendix A, Figure 4. A map of locations where Bighead and Silver carp were captured and implemented with an acoustic transmitter. The numbers associated with each red circle indicate the number of carp that were tagged at that location.



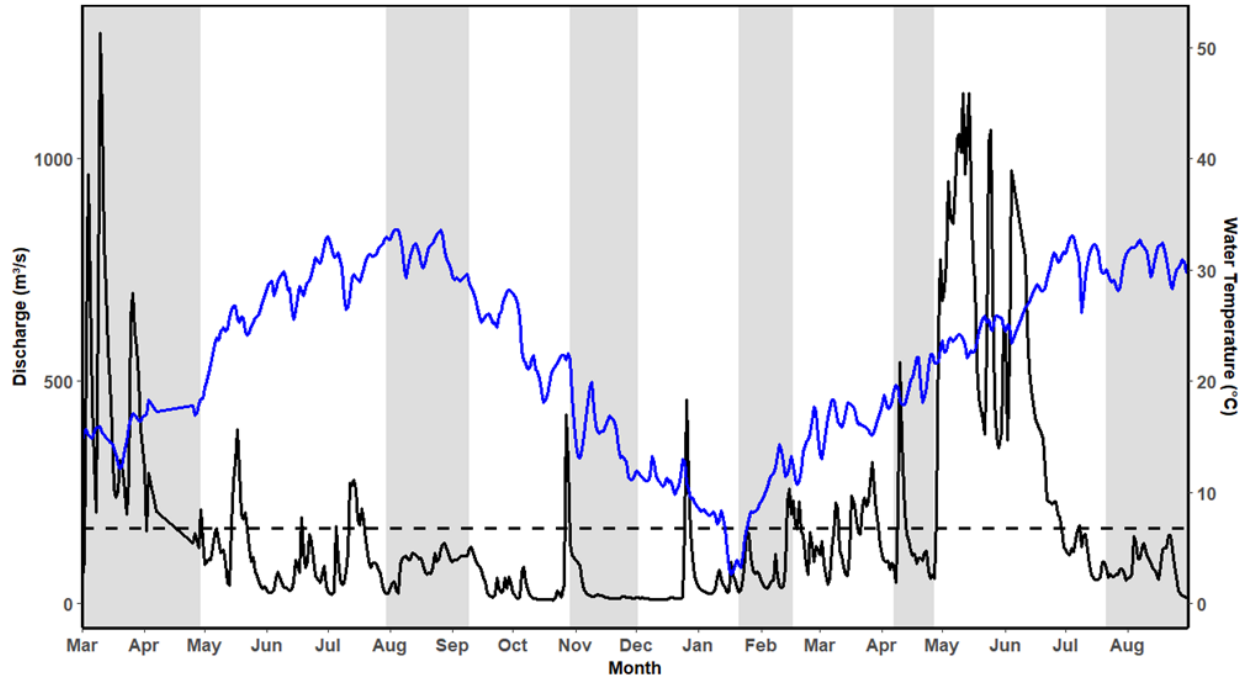
Appendix A, Figure 5. Map of the study area showing SUR locations (black circles) deployed in the lower Red River catchment. The coordinates of the SURs are provided in Appendix A, Table 39. The SURs remained deployed for use by USFWS at the end of our study.



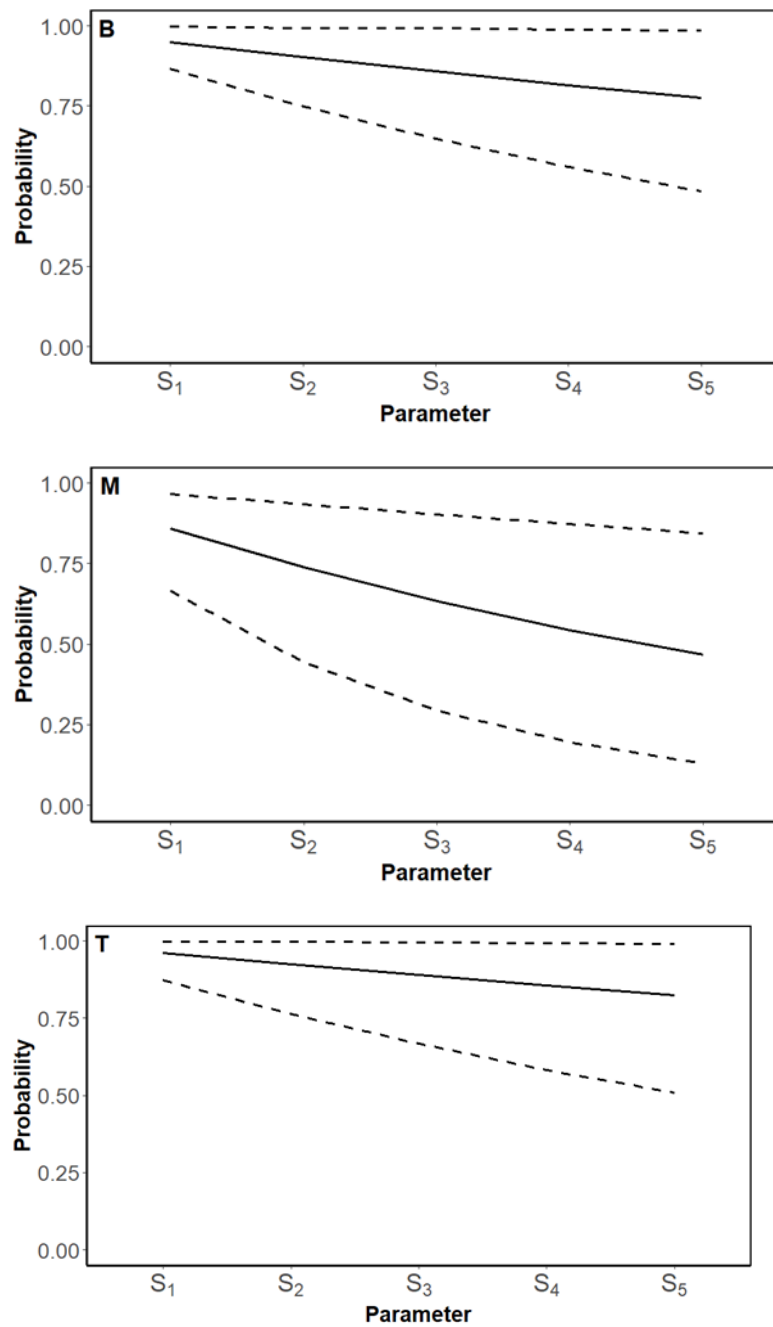
Appendix A, Figure 6. Map of the lower Red River catchment indicating sections of river that were actively tracked to detect tagged carp in 2023 and 2024. Each letter denotes an individual tracking section, and the black squares indicate both the start and end of the track. Descriptions of each tracked section are provided in Appendix A, Table 42.



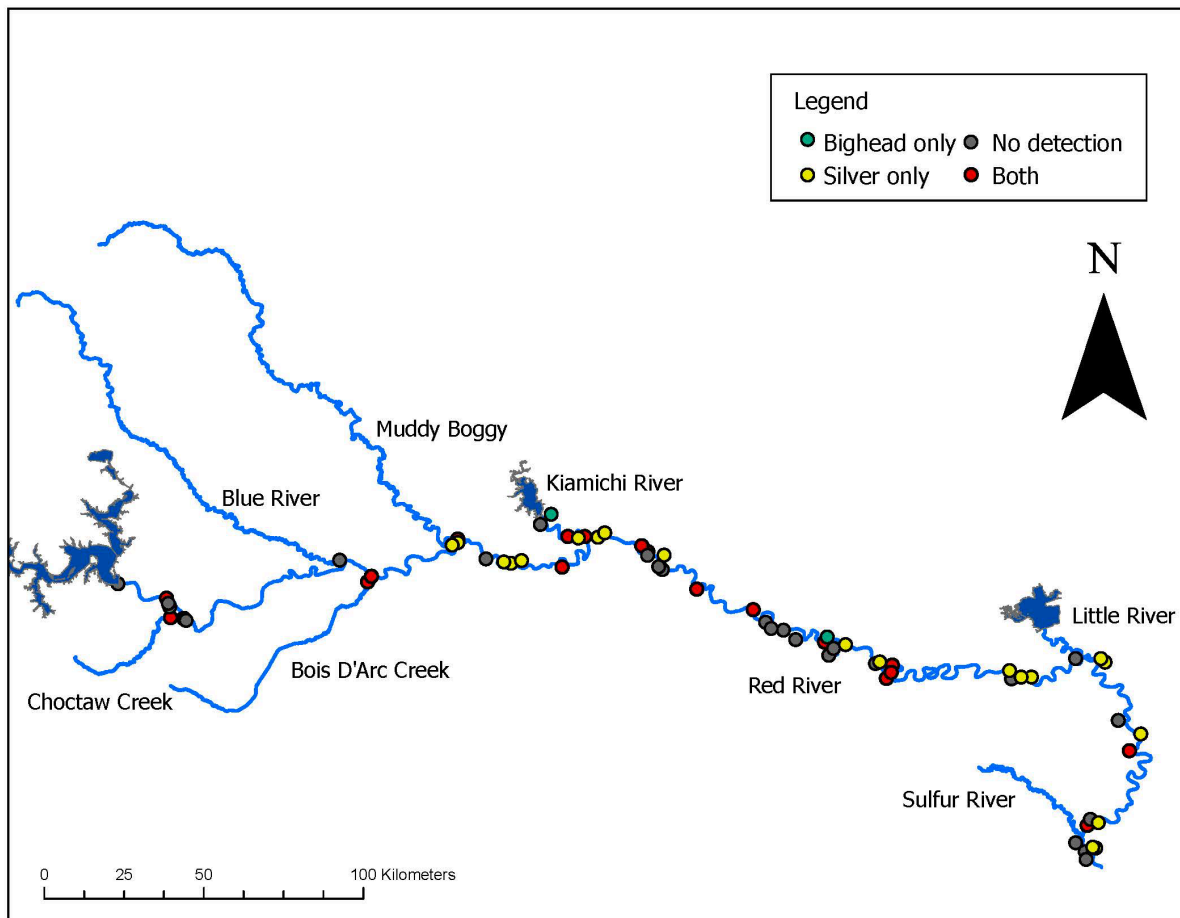
Appendix A, Figure 7. USGS stream gages within the lower Red River catchment. The red squares indicate each stream gage location from which discharge data were collected for our movement pattern analysis.



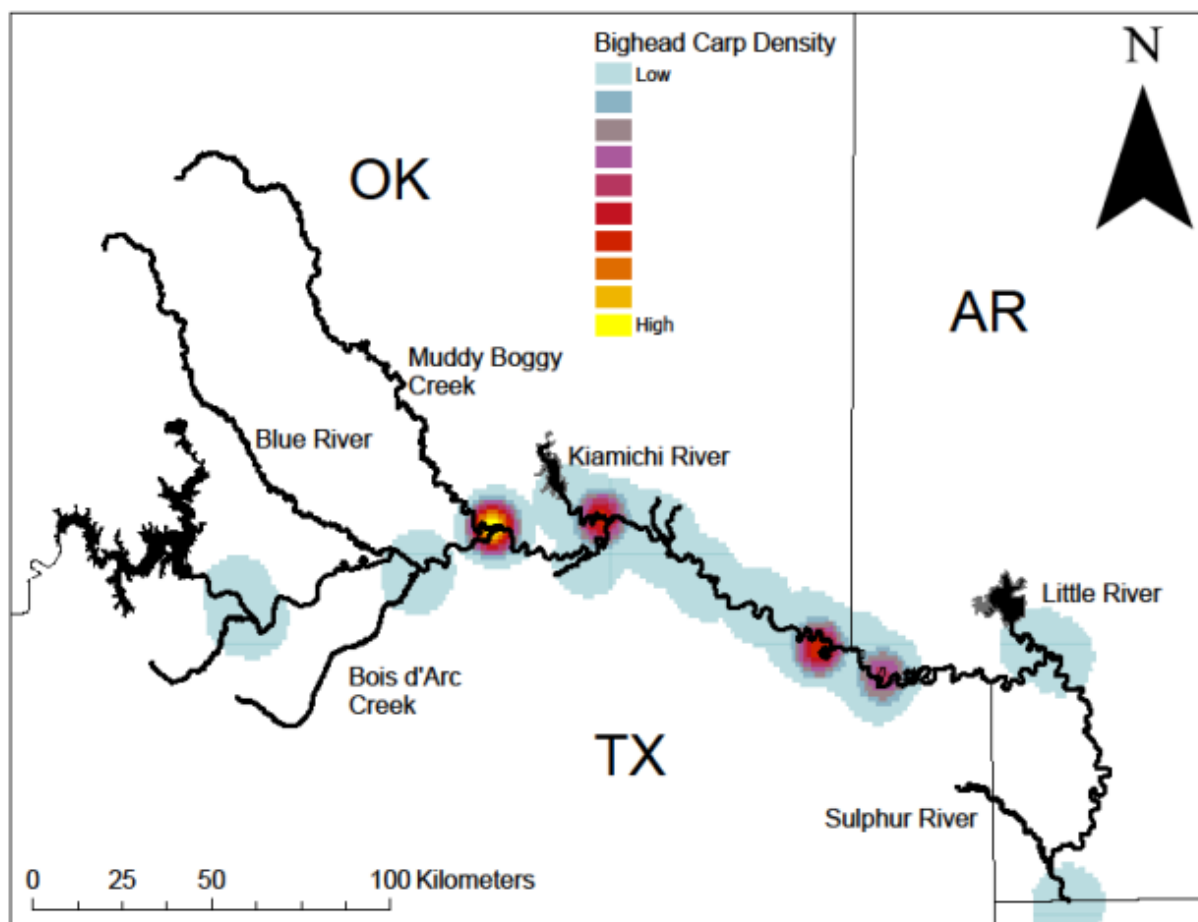
Appendix A, Figure 8. Environmental conditions for the lower Red River catchment during the study period. Discharge measurements (black line) were collected from the USGS stream gage near Arthur City TX 07335500. The dashed line represents mean discharge for the study period ($170 \text{ m}^3/\text{s}$). Water temperature measurements (blue line) were collected from the USGS stream gage near Index AR 07337000. Shaded bars indicate closed periods which were condensed to create the six time periods of our multistate model.



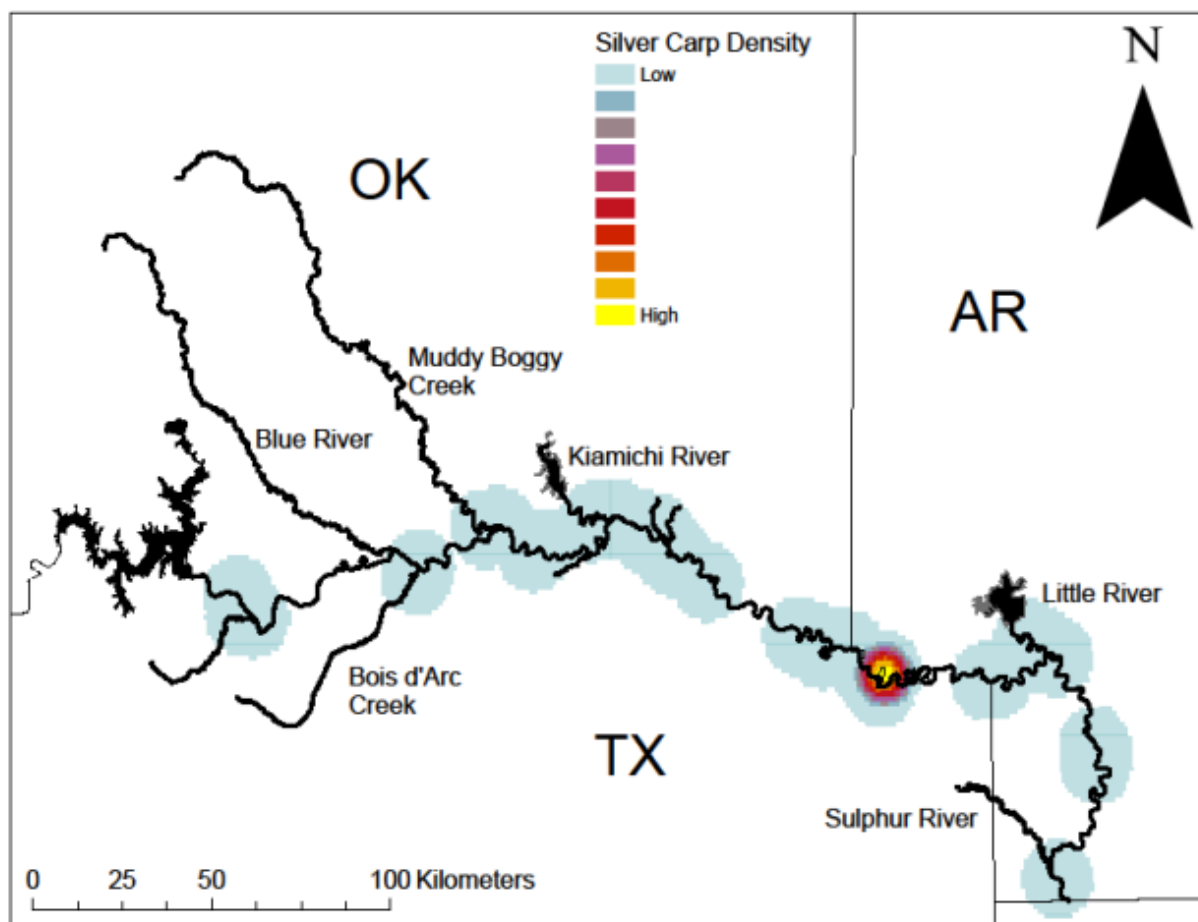
Appendix A, Figure 9. Cumulative apparent survival estimates for our three-state model. Panel B shows survival estimates in the backwater state, panel M shows the survival estimates in the mainstem state, and panel T shows survival estimates in the tributary state. The y-axis shows the probability of survival. The x-axis shows the survival notation (S) of each time period. The black dots indicate the mode estimate, and the black lines indicate the 95% credibility intervals.



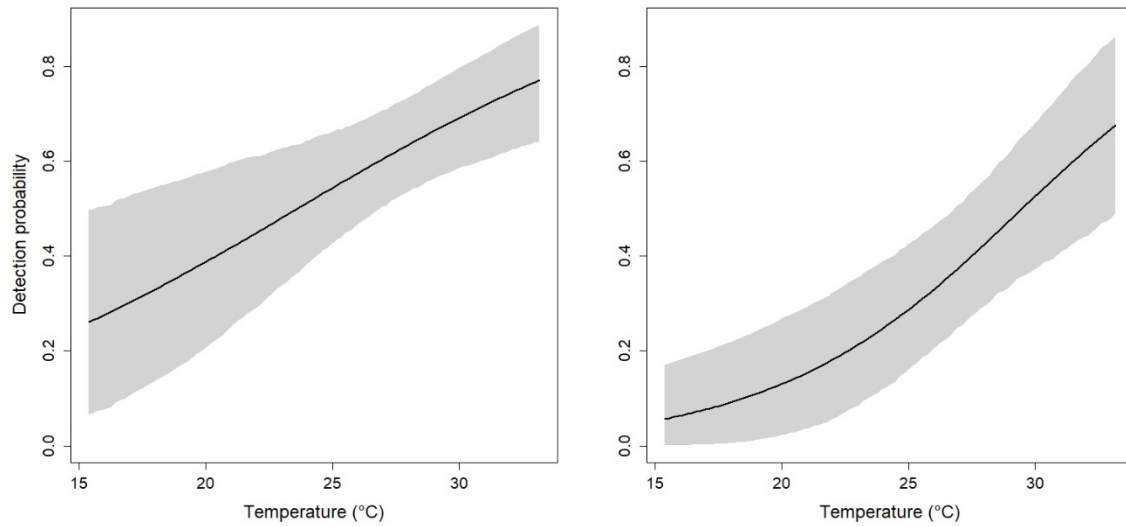
Appendix A, Figure 10. A map of sites in the lower Red River catchment from 2021-2023 where no carp were detected (gray circle), only Silver Carp was detected (yellow circle), only Bighead Carp (green circle), and where both Bighead Carp and Silver Carp were detected (red circle).



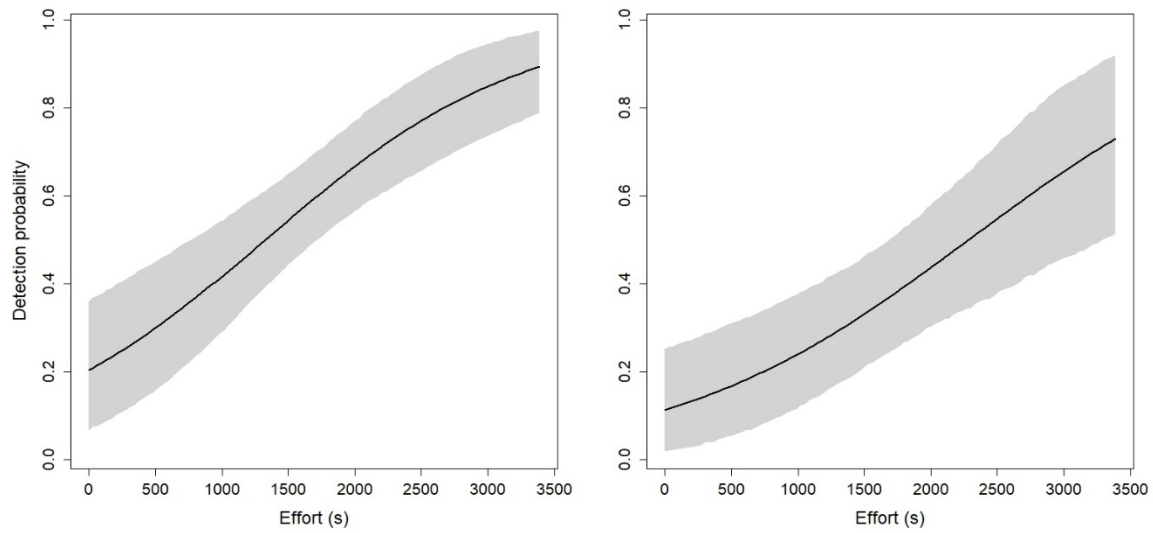
Appendix A, Figure 11. Aggregations of Bighead Carp that were captured or relocated in the lower Red River catchment from July 2021 to August 2024. Areas in blue represent lower carp densities and areas in Red and Yellow represent areas of higher carp densities.



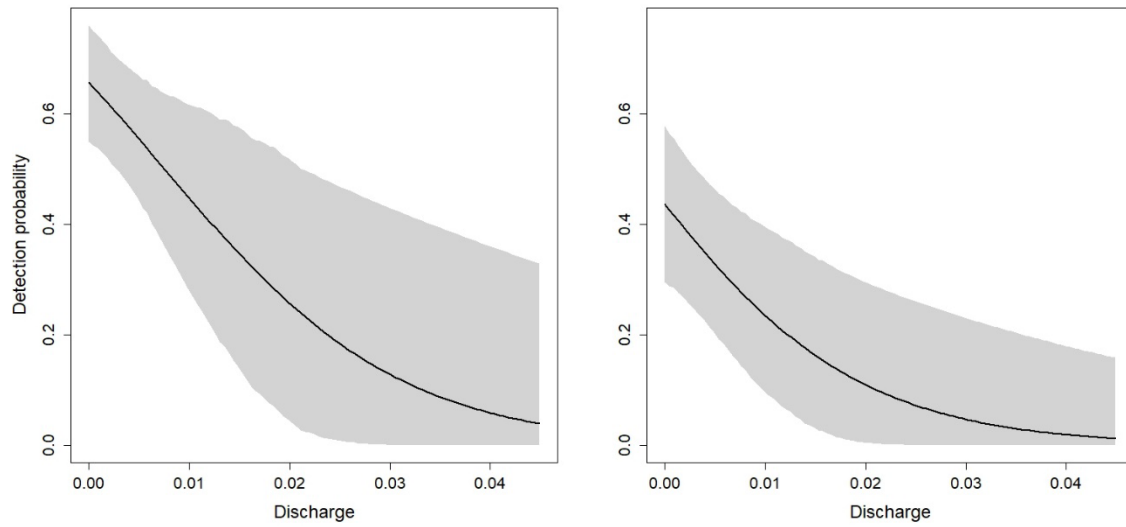
Appendix A, Figure 12. Aggregations of Silver Carp that were captured or relocated in the lower Red River catchment from July 2021 to August 2024. Areas in blue represent lower carp densities and areas in Red and Yellow represent areas of higher carp densities.



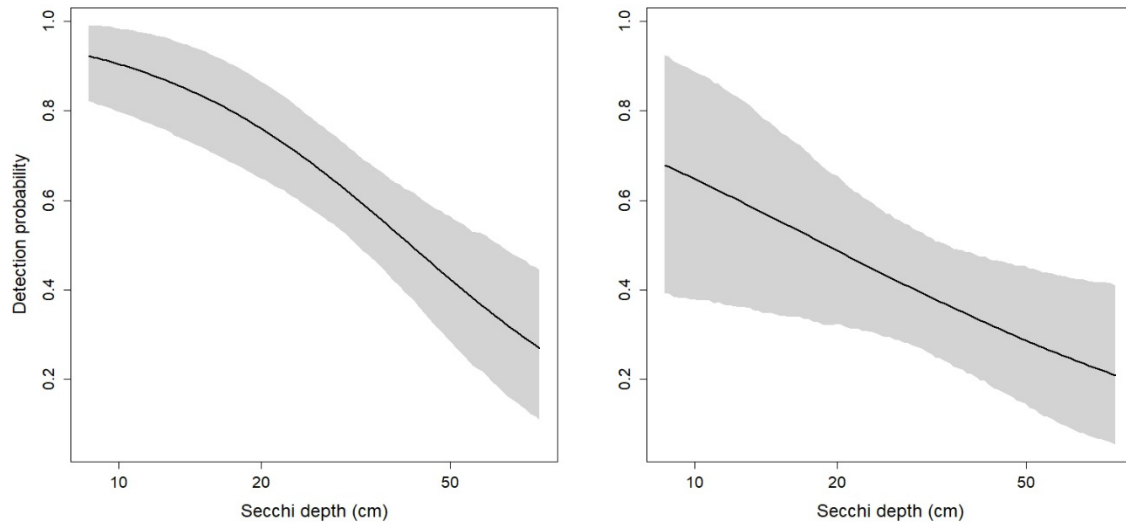
Appendix A, Figure 13. Silver Carp (left) and Bighead Carp (right) detection probability related to water temperature (°C) in the lower Red River catchment. The solid line is the mode estimate, and the gray polygon is the 90% highest density interval (HDI). The mode was estimated with all other model covariates held at mean values.



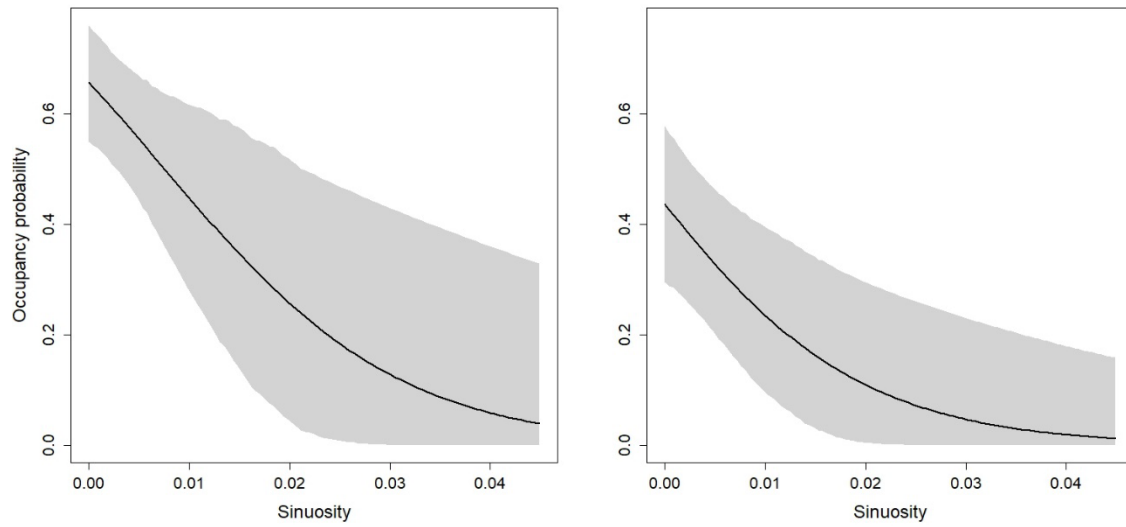
Appendix A, Figure 14. Silver Carp (left) and Bighead Carp (right) detection probability related to electrofishing effort (s) in the lower Red River catchment. The solid line is the mode estimate, and the gray polygon is the 90% highest density interval (HDI). The mode was estimated with all other model covariates held at mean value



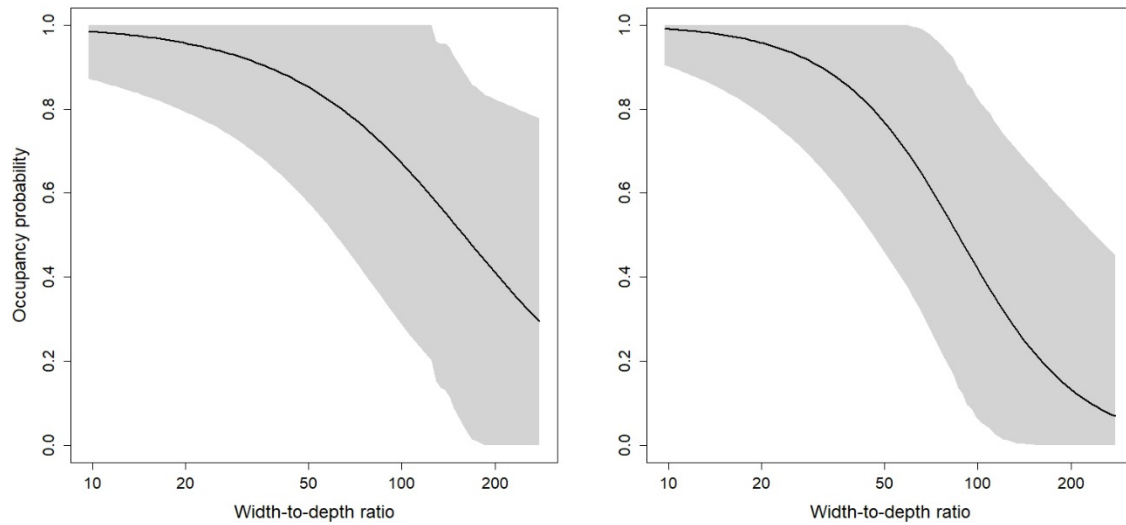
Appendix A, Figure 15. Silver Carp (left) and Bighead Carp (right) detection probability related to discharge (m^3/s) in the lower Red River catchment. The solid line is the mode estimate, and the gray polygon is the 90% highest density interval (HDI). The mode was estimated with all other model covariates held at mean values.



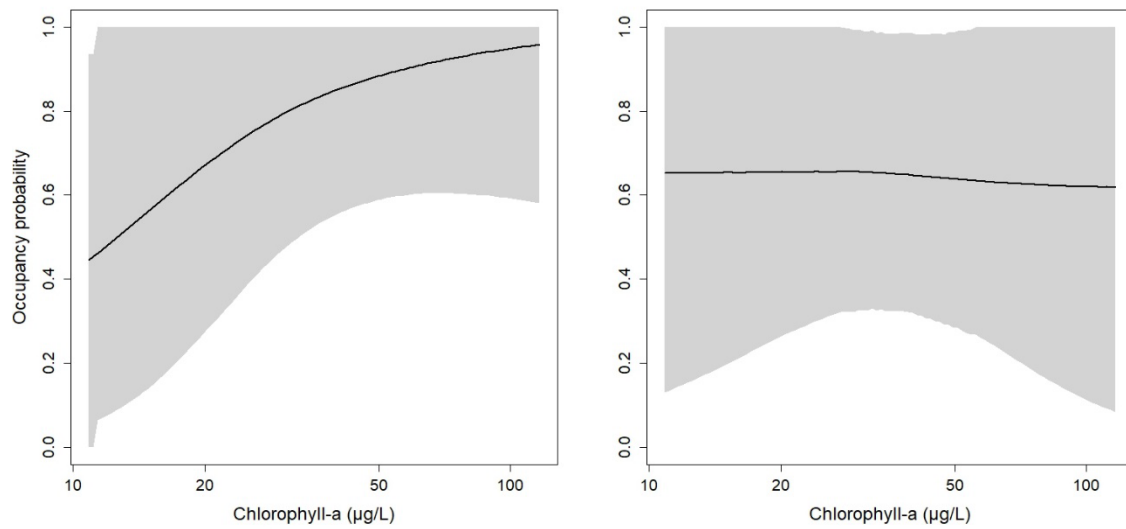
Appendix A, Figure 16. Silver Carp (left) and Bighead Carp (right) detection probability related to Secchi depth (cm) in the lower Red River catchment. The solid line is the mode estimate, and the gray polygon is the 90% highest density interval (HDI). The mode was estimated with all other model covariates held at mean values.



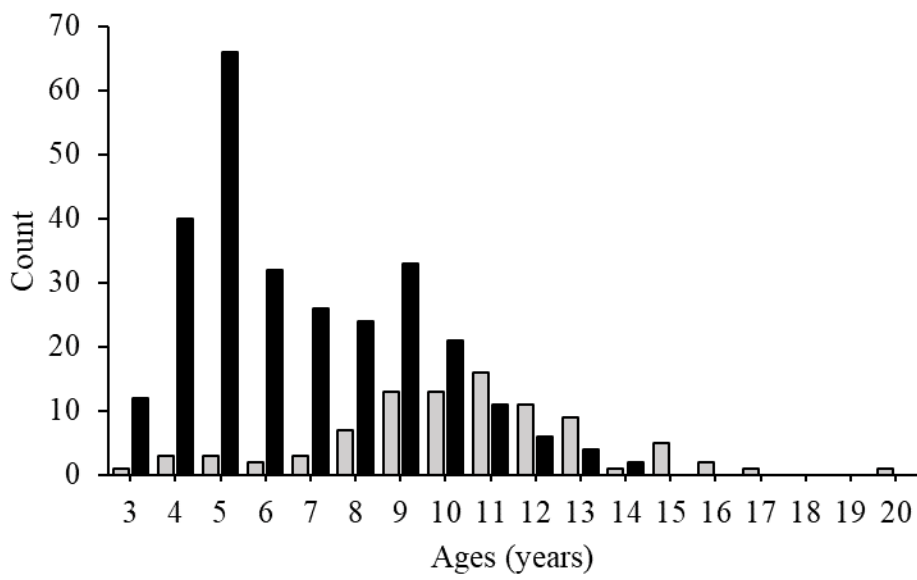
Appendix A, Figure 17. Silver Carp (left) and Bighead Carp (right) occupancy probability related to sinuosity in the lower Red River catchment. The solid line is the mode estimate, and the gray polygon is the 90% highest density interval (HDI). The mode was estimated with all other model covariates held at mean values.



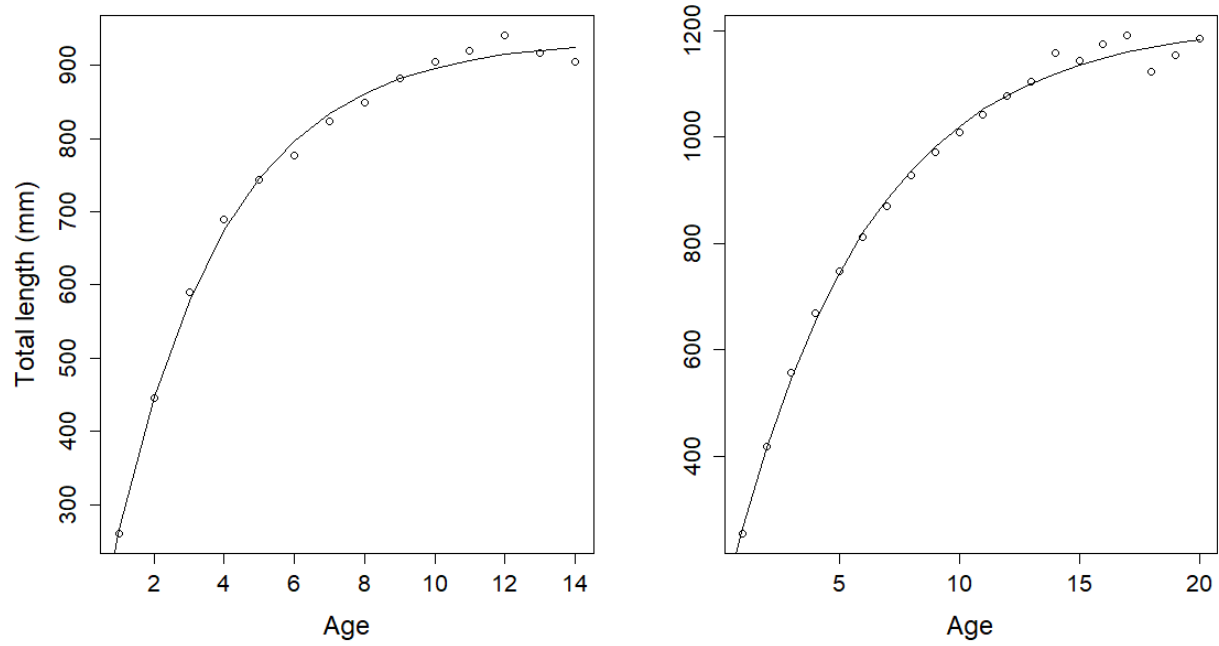
Appendix A, Figure 18. Silver Carp (left) and Bighead Carp (right) occupancy probability related to width-to-depth ratio in the lower Red River catchment. The solid line is the mode estimate, and the gray polygon is the 90% highest density interval (HDI). The mode was estimated with all other model covariates held at mean values.



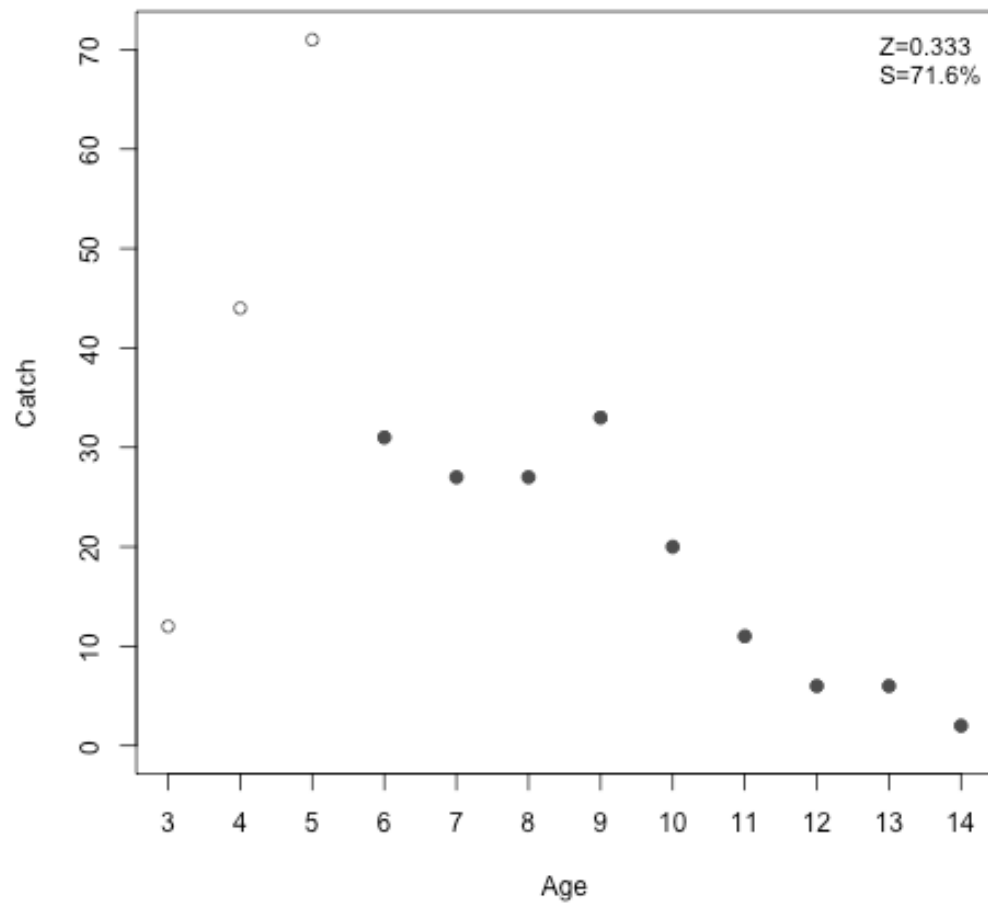
Appendix A, Figure 19. Silver Carp (left) and Bighead Carp (right) occupancy probability related to chlorophyll-*a* in the lower Red River catchment. The solid line is the mode estimate, and the gray polygon is the 90% highest density interval (HDI). The mode was estimated with all other model covariates held at mean values.



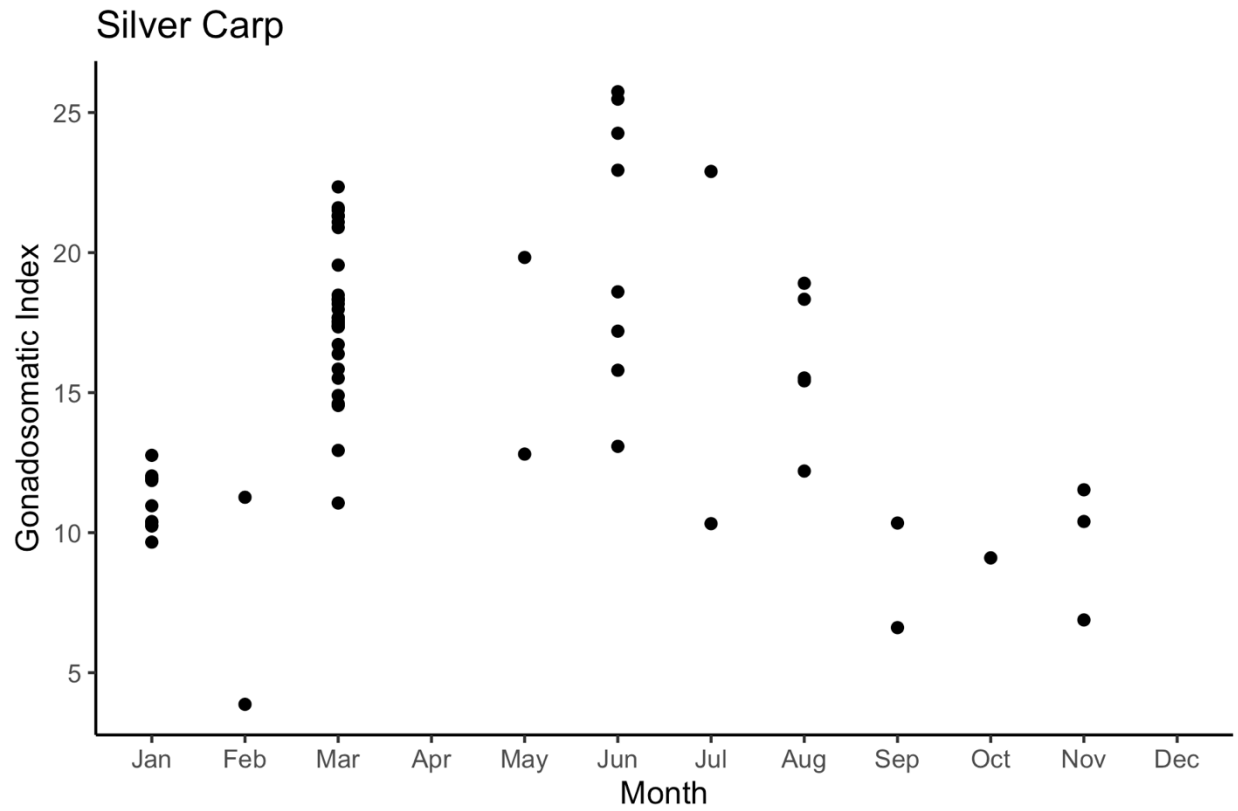
Appendix A, Figure 20. Age frequency histogram for Silver Carp (black bars) and Bighead Carp (gray bars) sampled from the lower Red River catchment in 2021-2023.



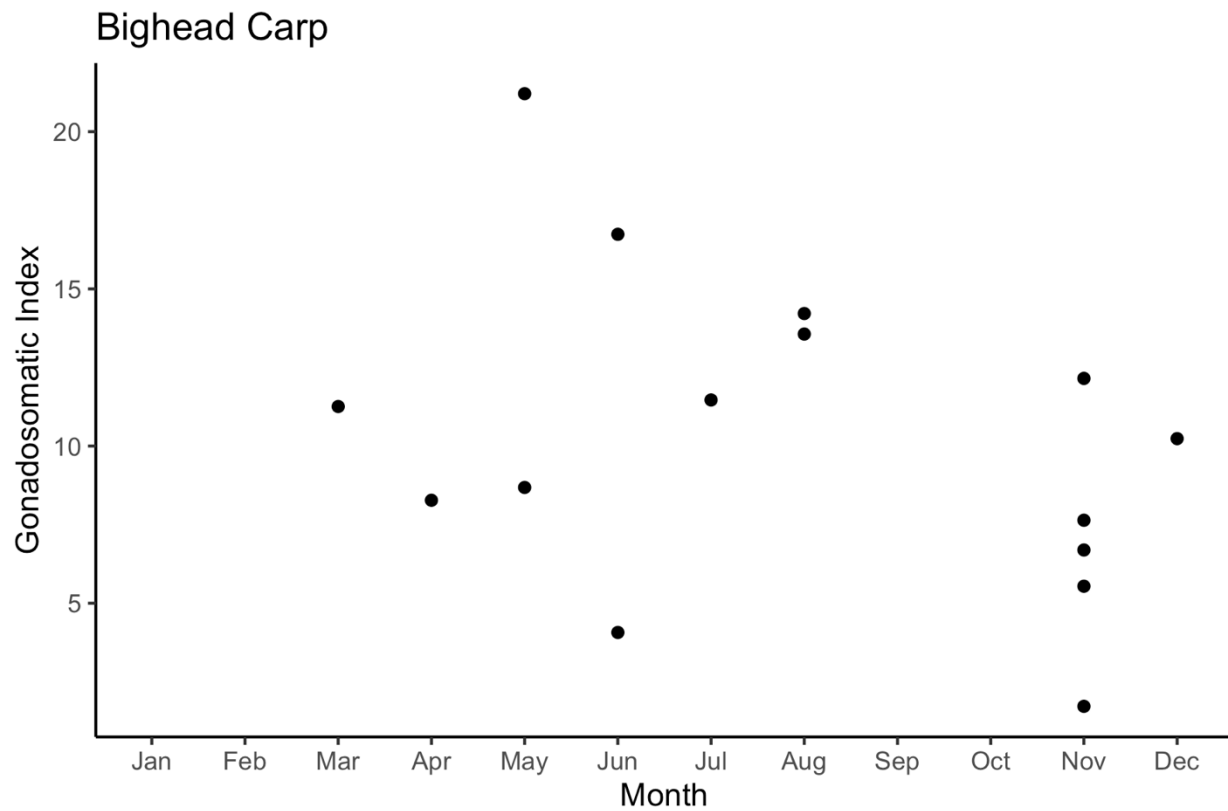
Appendix A, Figure 21. A von Bertalanffy growth curve fit to the mean back-calculated length-at-age for Silver Carp (left) and Bighead Carp (right) in the lower Red River catchment using data collected in 2021-2023.



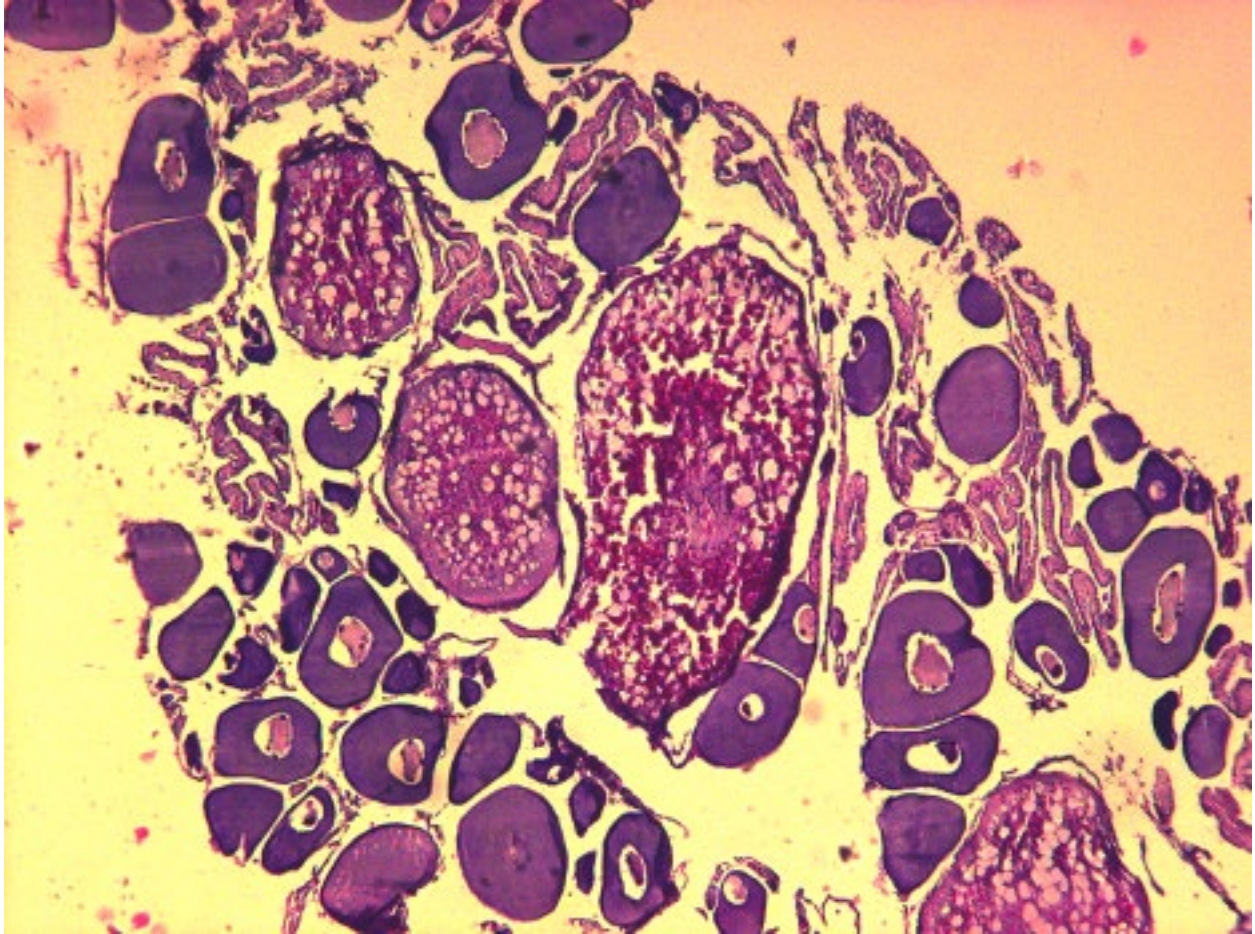
Appendix A, Figure 22. Chapman-Robson peak-plus catch-curve assessing mortality and recruitment variability of Silver Carp in the lower Red River catchment in 2021-2023. Z is the estimated total instantaneous mortality rate, and S is the estimated annual survival rate. The black dots indicate age classes that were included in the analysis (i.e., ages 6-14).



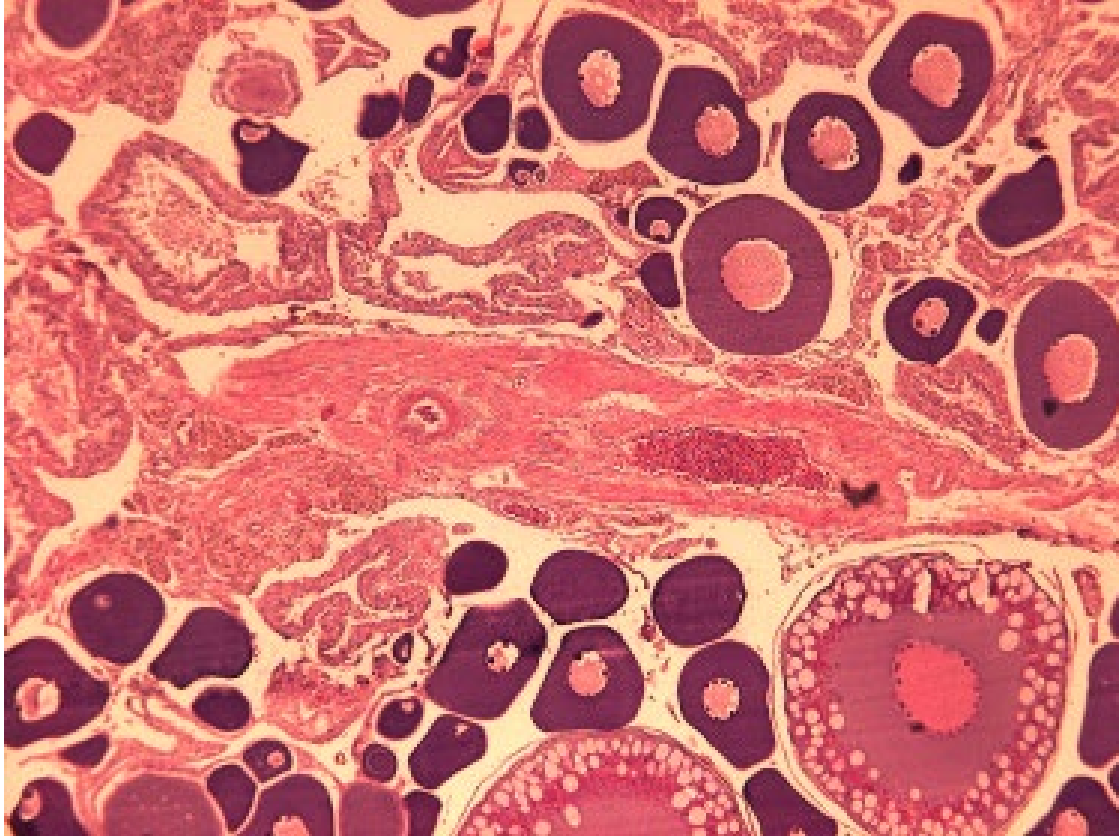
Appendix A, Figure 23. Gonadosomatic index by month for Silver Carp collected in the lower Red River catchment in 2021-2023.



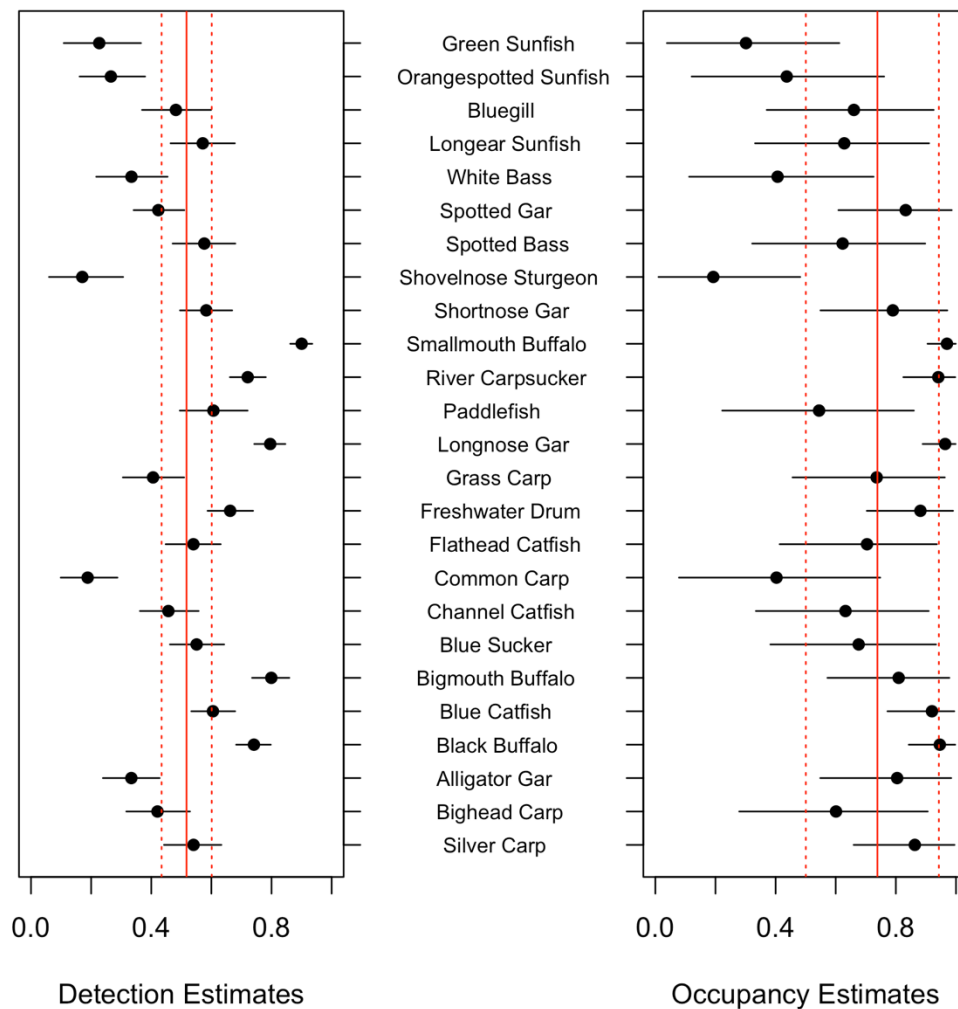
Appendix A, Figure 247. Gonadosomatic index by month for Bighead Carp collected in the lower Red River catchment in 2021-2023.



Appendix A, Figure 25. Histological section of a female Silver Carp ovary collected May 1, 2024, in the lower Red River catchment where post-ovulatory follicles are present resulting from a spawning event.



Appendix A, Figure 26. Histological section of a female Bighead Carp ovary collected May 1, 2024, in the lower Red River catchment where post-ovulatory follicles are present resulting from a spawning event.



Appendix A, Figure 27. Adult large-bodied fish species detection and occupancy estimates from the final occupancy model for the lower Red River basin (2021-2023). The black points represent the median (most likely) values from the posterior distribution for each species. The black bars represent the 90% credible intervals for those species. The solid red line shows the group mean (all species) for both the detection and occupancy estimates and the dotted red lines show the 90% credible intervals for those estimates.

Bluegill	-	-	-	-	-	+	-	+
Longear Sunfish	-	-	-	-	-	+	-	+
Green Sunfish	+	-	-	-	-	+	-	+
Orangespotted Sunfish	+	-	-	-	-	-	-	+
Spotted Bass	-	-	-	-	-	+	-	-
White Bass	-	-	-	-	-	+	-	+
	Slack	Drain	Elv	Sin	Salt	W:D	Dam	Q

Appendix A, Figure 28. Occupancy relationships of larger-bodied (i.e., adult and subadult) Centrarchidae and Moronidae in the Red River basin (2021-2023). Positive relationships are indicated with a red plus sign (+). Negative relationships are indicated with a black negative sign (-). Slack is greater than 1% of backwater, Drain is the drainage area, Elv is elevation, Sin is the segment sinuosity, Salt is the salinity, W:D is width-to-depth ratio, Dam is the distance from nearest upstream dam, and Q is the median discharge value.

Alligator Gar	-	-	-	-	-	-	-	-
Blue Catfish	-	-	-	-	-	+	-	-
Channel Catfish	-	-	-	-	-	+	-	+
Flathead Catfish	-	-	-	-	-	+	-	+
Longnose Gar	-	-	-	-	-	+	-	+
Shortnose Gar	-	-	-	-	-	+	-	+
Spotted Gar	-	-	-	-	-	-	-	+
	Slack	Drain	Elv	Sin	Salt	W:D	Dam	Q

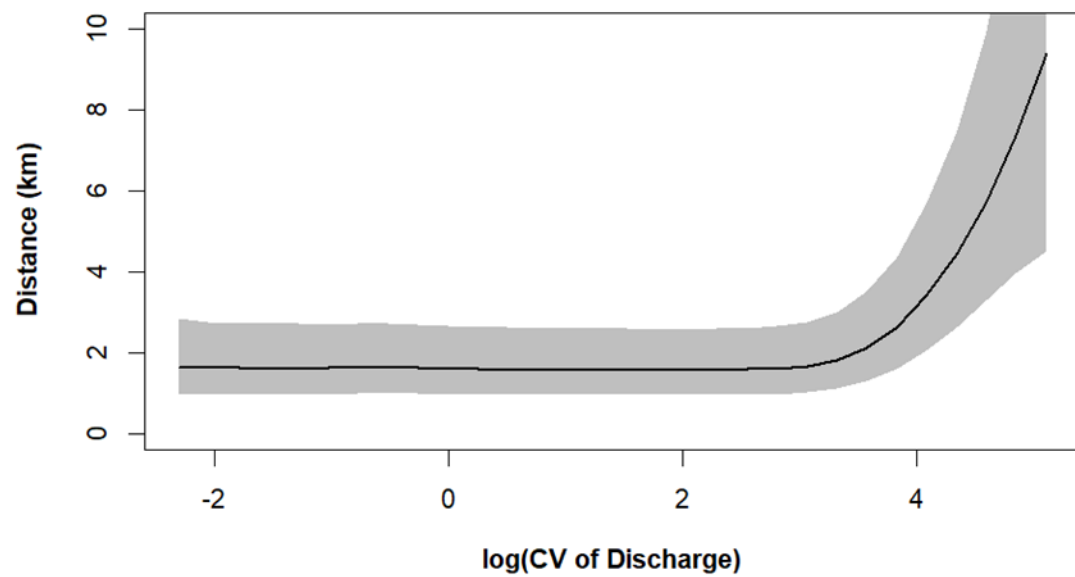
Appendix A, Figure 29. Occupancy relationships of larger-bodied Lepisosteidae and Ictaluridae species in the Red River basin (2021-2023). Positive relationships are indicated with a red plus sign (+). Negative relationships are indicated with a black negative sign (-). Slack is greater than 1% of backwater, Drain is the drainage area, Elv is elevation, Sin is the segment sinuosity, Salt is the salinity, W:D is width-to-depth ratio, Dam is the distance from nearest upstream dam, and Q is the median discharge value.

Bigmouth Buffalo	+	-	-	-	-	-	-	-
Black Buffalo	+	-	-	-	-	+	-	-
Blue Sucker	-	-	-	-	-	+	-	+
Paddlefish	+	-	+	-	-	-	-	-
River Carpsucker	+	-	-	-	-	+	-	+
Shovelnose Sturgeon	+	-	-	-	-	+	-	+
Smallmouth Buffalo	+	-	-	-	-	-	-	-
	Slack	Drain	Elv	Sin	Salt	W:D	Dam	Q

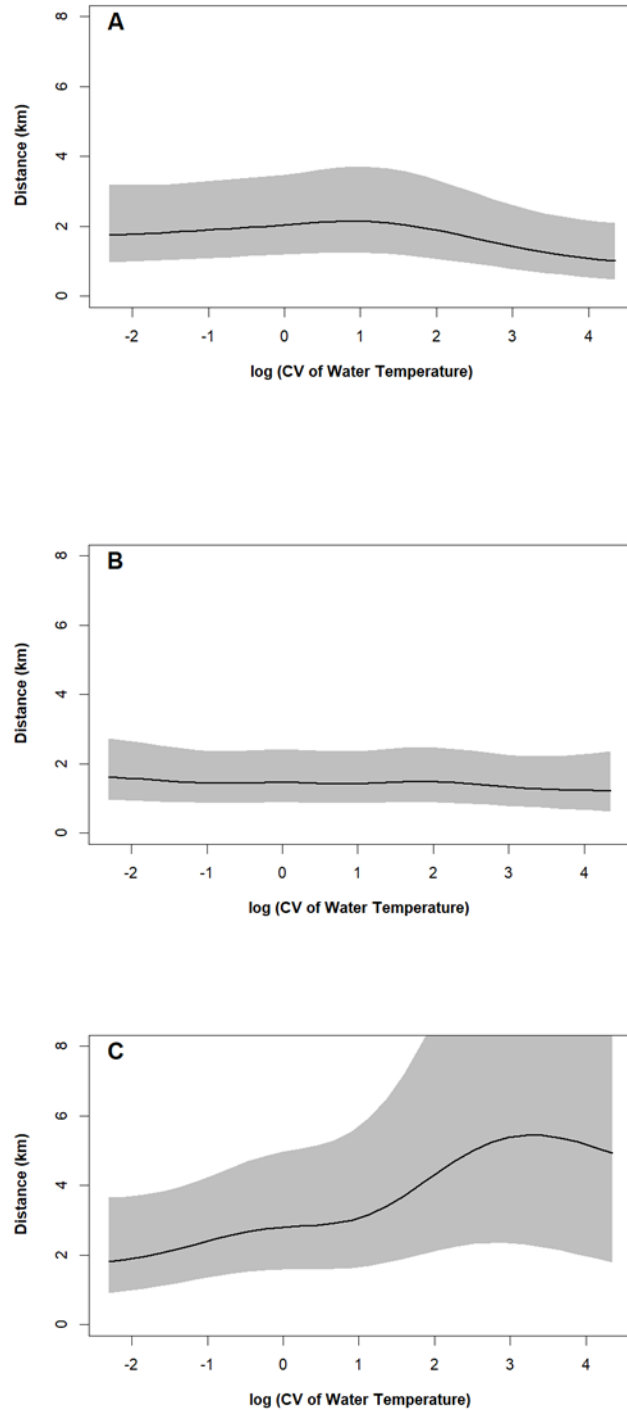
Appendix A, Figure 30. Occupancy relationships of larger-bodied Catostomidae species, Paddlefish, and Shovelnose Sturgeon in the Red River basin (2021-2023). Positive relationships are indicated with a red plus sign (+). Negative relationships are indicated with a black negative sign (-). Slack is greater than 1% of backwater, Drain is the drainage area, Elv is elevation, Sin is the segment sinuosity, Salt is the salinity, W:D is width-to-depth ratio, Dam is the distance from nearest upstream dam, and Q is the median discharge value.

Bighead Carp	+	-	+	-	-	-	-	-
Common Carp	+	-	-	-	-	-	-	+
Grass Carp	+	-	-	-	-	-	-	+
Silver Carp	+	-	-	-	+	-	-	-
Freswater Drum	-	-	-	-	-	+	-	+
	Slack	Drain	Elv	Sin	Salt	W:D	Dam	Q

Appendix A, Figure 31. Occupancy relationships of larger-bodied Sciaenidae (native species) and four carp species (all non-native species) in the Red River basin (2021-2023). Positive relationships are indicated with a red plus sign (+). Negative relationships are indicated with a black negative sign (-). Slack is greater than 1% of backwater, Drain is the drainage area, Elv is elevation, Sin is the segment sinuosity, Salt is the salinity, W:D is width-to-depth ratio, Dam is the distance from nearest upstream dam, and Q is the median discharge value.

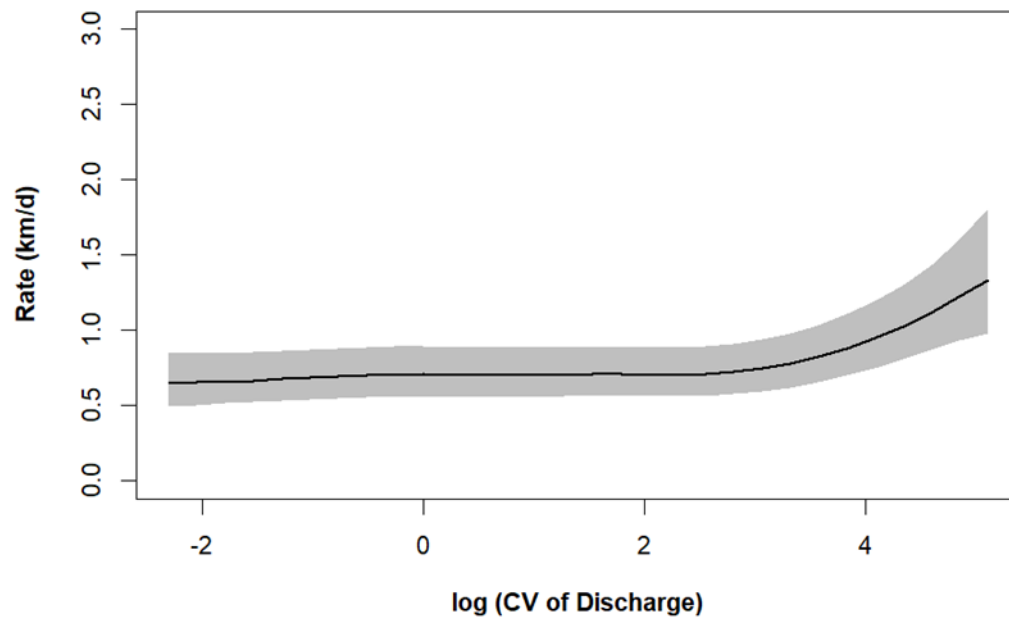


Appendix A, Figure 32. Relationship between CV of discharge and movement distance. Shaded areas around the line indicate the 95% confidence intervals. The y-axis shows the model-predicted distance traveled in river-kilometers between consecutive detections by tagged individuals. The x-axis shows natural-logged-transformed coefficient of variation of discharge.

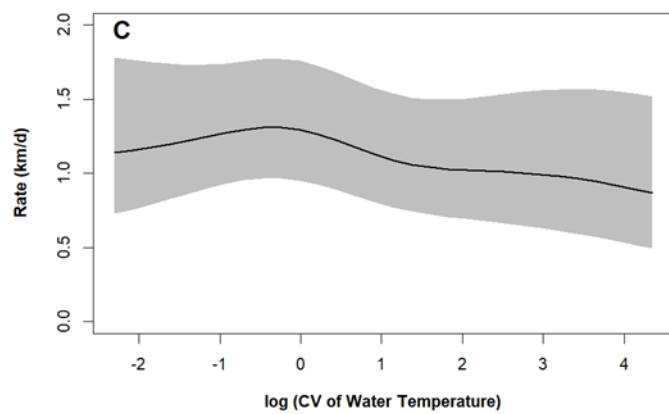
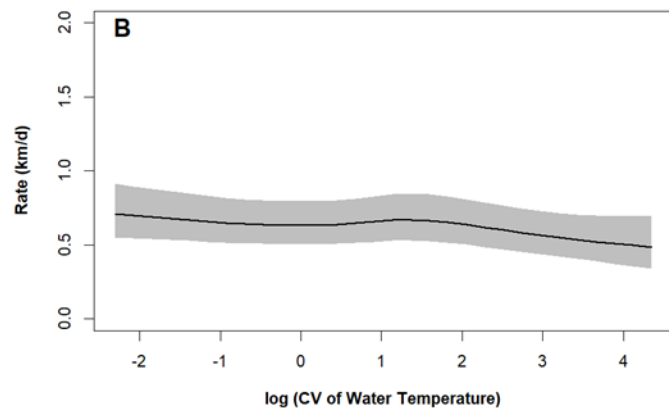
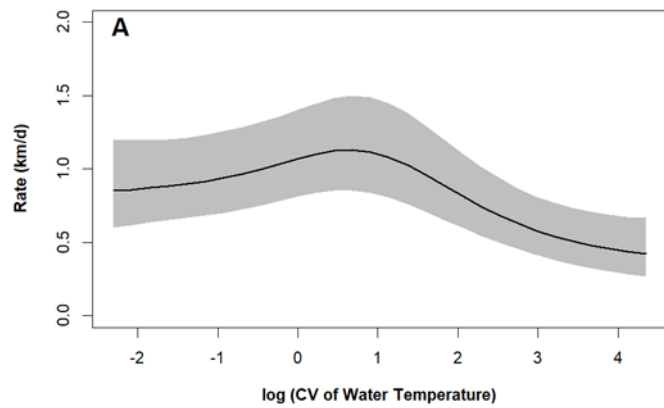


Appendix A, Figure 33. The relationship between CV of water temperature and movement distance at three levels of discharge. Shaded areas around the line indicate the 95 % confidence intervals. Panel A shows the relationship while holding discharge two standard deviations below

the mean. Panel B shows the relationship while holding discharge at the mean value. Panel C shows the relationship while holding discharge two standard deviations above the mean. The y-axis shows the model-predicted river-kilometers traveled between consecutive detections by tagged individuals. The x-axis shows the natural-logged-transformed coefficient of variation of water temperature.



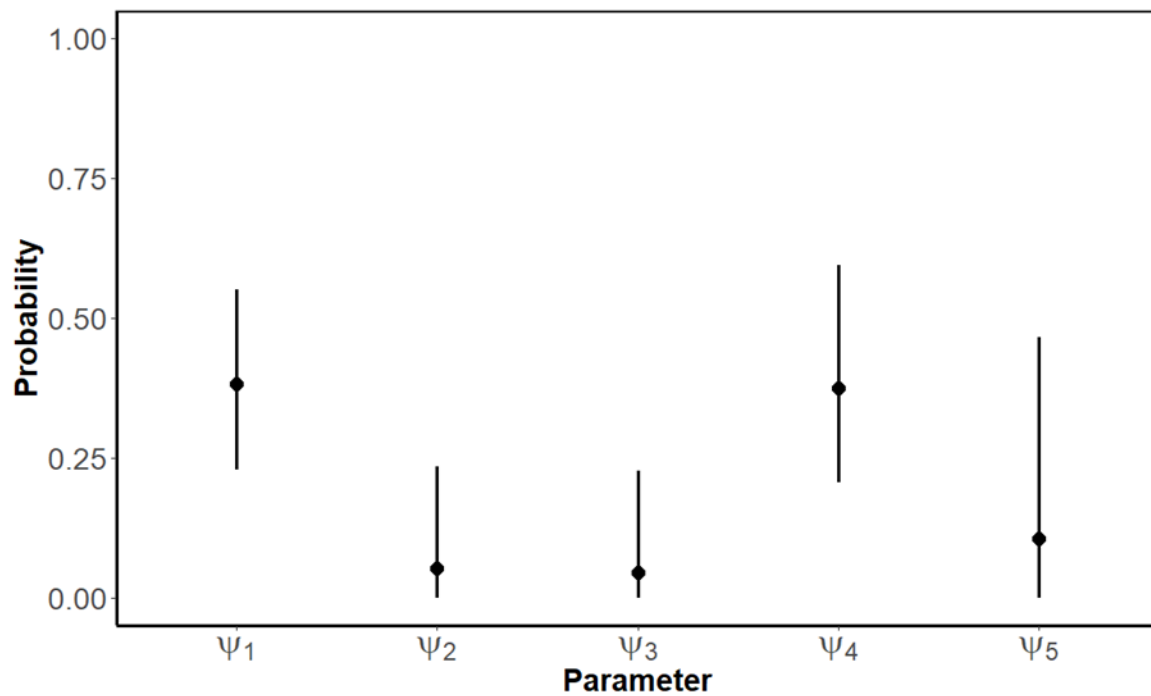
Appendix A, Figure 34. Relationship between CV of discharge and movement rate. Shaded areas around the line indicate the 95% confidence intervals. The y-axis shows the model-predicted river-kilometers traveled per day between consecutive detections by tagged individuals. The x-axis shows the natural-logged-transformed coefficient of variation of discharge.



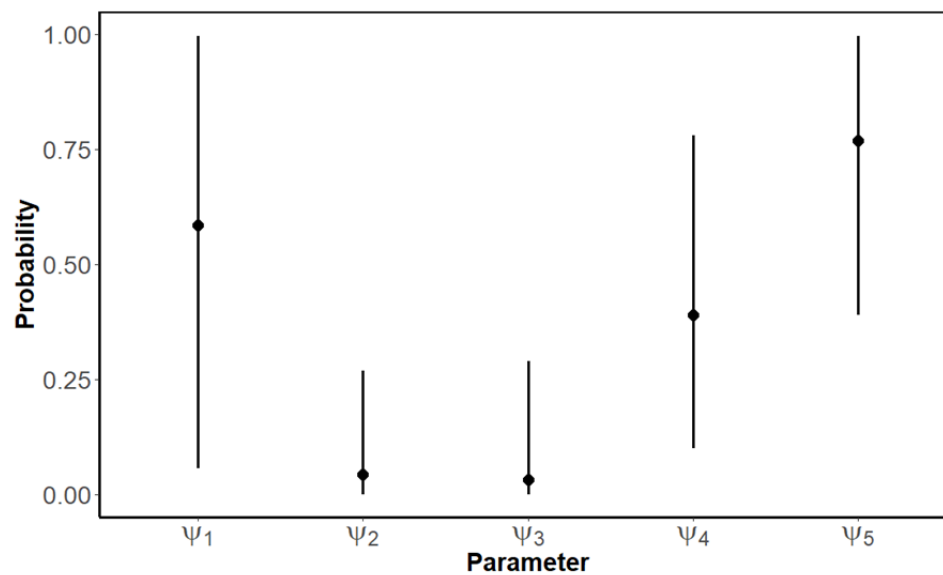
Appendix A, Figure 35. The relationship between CV of water temperature and movement rate at three levels of discharge. Shaded areas around the line indicate the 95 % confidence intervals.

Panel A shows the relationship while holding discharge two standard deviations below the mean.

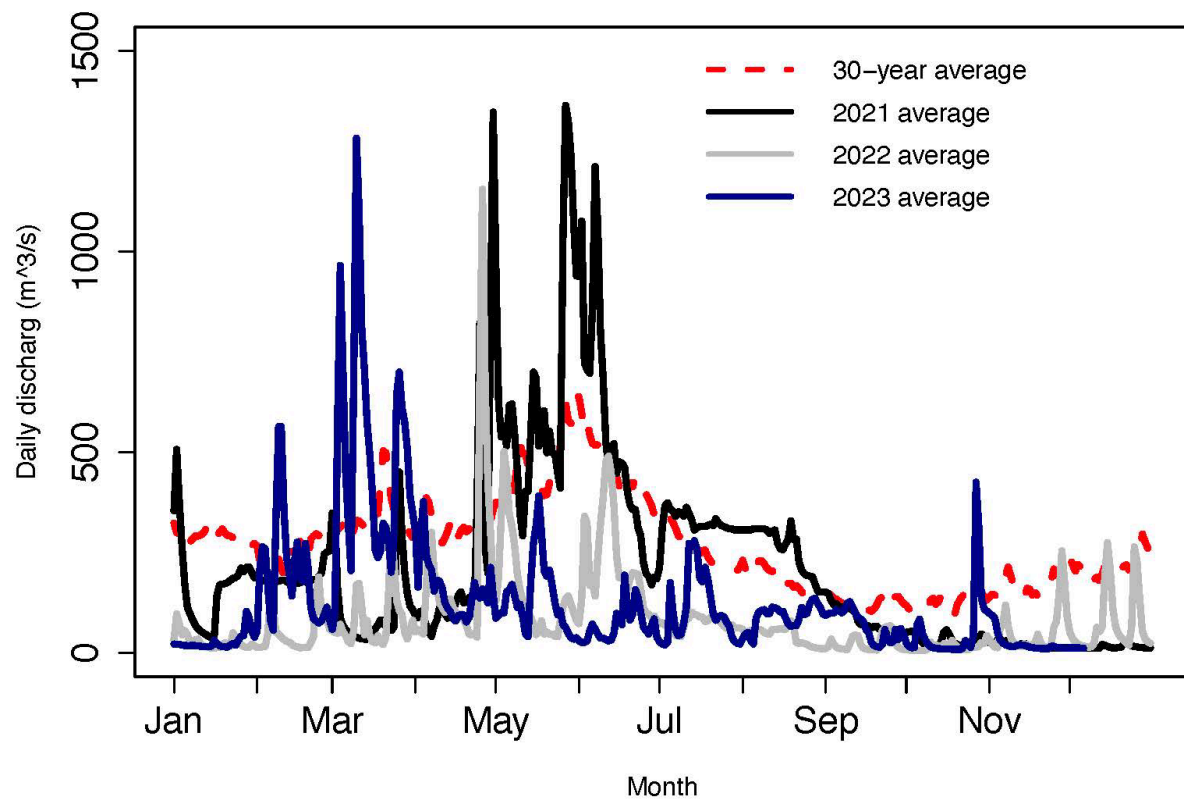
Panel B shows the relationship while holding discharge at the mean value. Panel C shows the relationship while holding discharge two standard deviations above the mean. The y-axis shows the model-predicted river-kilometers traveled per day between consecutive detections by tagged individuals. The x-axis shows the natural-logged-transformed coefficient of variation of water temperature.



Appendix A, Figure 36. Transition probability estimates from state A (tributary/backwater) to state M (mainstem river) from the MCMC simulation for the two-state model. The y-axis shows the probability of transitioning. The x-axis shows transition notation (ψ) between each time period. The black dots indicate the mode estimate, and the black lines indicate the 95% credibility intervals.



Appendix A, Figure 37. Transition probability estimates from state M to state A from the MCMC simulation for the two-state model. The y-axis shows the probability of transitioning. The x-axis shows transition notation (ψ) between each time period. The black dots indicate the mode estimate, and the black lines indicate the 95% credibility intervals.



Appendix A, Figure 38. Average daily discharge during the 2021-2023 sampling period. The red dotted line indicates the 30-year average discharge conditions. Data are from U.S. Geological Survey, stream gage 07335500 at Arthur City, TX.

Appendix B – Sulphur River Figures and Tables

Appendix B, Table 1. Summary of sampling dimensions and targeted life stages of bigheaded carp. See text for electrofishing settings.

Approach	Gear	Length	Height	Mesh size	Target stage
Passive	Gillnet	80'	6'	3.5", 4", 4.5"	Adult
	Gillnet	100'	6'	4"	Adult
	Hoop net	16'	4'	3"	Adult
	Mini-fyke net	4'	3'	1/8"	Age-0
Active	DC-Electrofishing				Age-0/Adult

Appendix B, Table 2. Environmental variables collected during field sampling.

Scale	Metric	Unit	Gear
Reach	Temperature	°C	YSI
Reach	Turbidity	m	Secchi Disc
Reach	Conductivity	µs/cm	YSI Pro10
Reach	Avg. transect depth	m	Humminbird Helix 9
Segment	River gauge height	ft	USGS Stream Gauge

Appendix B, Table 3. Summary (average $\pm$ SD) of environmental variables collected at each waterbody during the 2021 - 2023 sampling periods.

Year	Location	Temp. (°C)	Secchi depth (m)	Conductivity ($\mu\text{S}/\text{cm}$)
2021	Red River	24.7 ± 5.8	0.25 ± 0.0	639.6 ± 344.5
2021	Sulphur River	25.1 ± 5.0	0.36 ± 0.15	222.7 ± 141.6
2021	Wright Patman	26.3 ± 6.9	0.49 ± 0.10	144.5 ± 16.3
2022	Red River	25.5 ± 7.7	0.26 ± 0.05	616.4 ± 239.1
2022	Sulphur River	24.2 ± 7.7	0.30 ± 0.08	341.8 ± 170.3
2022	Wright Patman	28.1 ± 5.2	0.41 ± 0.10	216.2 ± 22.3
2023	Red River	24.9 ± 8.3	0.25 ± 0.06	631.1 ± 221.1
2023	Sulphur River	25.1 ± 6.3	0.34 ± 0.03	289.8 ± 220.1
2023	Wright Patman	27.1 ± 4.5	0.44 ± 0.31	206.2 ± 29.9

Appendix B, Table 4. Summary of Silver Carp captured from 2021 through 2023. No Silver Carp were captured during 2021. No Bighead carp were captured throughout the study. EF = pulsed-DC electrofishing; SIC = Silver Carp

Waterbody	Species	Date	Gear	Length (mm)	Mass (kg)	Age	Coordinates
Sulphur R.	SIC	6/25/2022	EF	928	9.16	10	N 33°06.281', W 93°51.970'
Red R.	SIC	9/17/2022	EF	815	6.74	8	N 33°04.632', W 93°50.712'
Red R.	SIC	4/16/2023	EF	1050	11.5	14	N 33°04.632', W 93°50.712'
Red R.	SIC	5/17/2023	EF	940	10.29	12	N 33°04.632', W 93°50.712'
Red R.	SIC	6/8/2023	EF	922	8.75	12	N 33°04.632', W 93°50.712'
Red R.	SIC	7/21/2023	EF				N 33°04.632', W 93°50.712'
Red R.	SIC	8/6/2023	EF	892	7.44	10	N 33°04.632', W 93°50.712'
Red R.	SIC	8/6/2023	EF	845	7.2	9	N 33°04.632', W 93°50.712'

Appendix B, Table 5. Species codes and common names of fishes collected during the project. *

Species code	Common name	Species code	Common name
BBH	Black bullhead	ISS	Inland silverside
BCF	Blue catfish	LEP*	Lepomis spp.
BLB	Black buffalo	LMB	Largemouth bass
BLC	Black crappie	LNG	Longnose gar
BLG	Bluegill	LSF	Longear sunfish
BLS	Blue sucker	OSS	Orangespotted sunfish
	Blackstripe		
BLT	topminnow	PDF	Paddlefish
BMB	Bigmouth buffalo	PIP	Pirate perch
BOW	Bowfin	RBS	Redbreast sunfish
BTS	Blacktail shiner	RCS	River carpsucker
CCF	Channel catfish	RDS	Red shiner
CLP	Common logperch	RES	Redear sunfish
CRP	Common carp	RIS	Ribbon shiner
DSF	Dollar sunfish	RSS	Redspotted sunfish
EMS	Emerald shiner	SHS	Shovelnose sturgeon
FCF	Flathead catfish	SIC	Silver carp
FHM	Fathead minnow	SMB	Smallmouth buffalo
FLI	Flier	SNG	Shortnose gar
FRD	Freshwater drum	SPB	Spotted bass
GAM	Mosquitofish	SPG	Spotted gar
GOR	Golden Redhorse	STB	Striped bass
GOS	Golden shiner	TFS	Threadfin shad
GOT	Golden topminnow	WHB	White bass
GRC	Grass carp	WHC	White crappie
GRP	Grass pickerel	WRM	Warmouth
GSF	Green sunfish	YLB	Yellow bass
GZS	Gizzard shad		

Appendix B, Table 6. Summary of average body length (total lengths) of fish species collected during the 2021 and 2023 sampling seasons. EF = electrofishing; GN = gillnet; EF/GN = herding into gillnets; HN = hoop net. See Appendix B, Table 5 for species key.

Waterbody	Species	EF Avg	SD	GN Avg	SD	EF/GN Avg	SD	HN Avg	SD
Sulphur River	BCF	488.1	152.2	431.0	--	410.1	36.5	393.5	44.5
Sulphur River	BLB	426.9	98.3	422.0	--	401.2	71.0		
Sulphur River	BLC	230.3	48.1			241.8	20.0	218.7	24.9
Sulphur River	BLG	97.6	39.2			139.0	--	155.0	--
Sulphur River	BLS	601.8	66.1						
Sulphur River	BLT	55.0	--						
Sulphur River	BMB	568.5	120.7	513.0	24.8	451.0	12.7		
Sulphur River	BOW	534.5	24.4			541.7	7.6		
Sulphur River	BTS	57.1	9.9						
Sulphur River	CCF	245.4	102.9	427.5	9.2	253.6	61.3	427.0	38.2
Sulphur River	CRP	590.6	77.1			534.0	164	740.0	--
Sulphur River	EMS	79.7	16.3						
Sulphur River	FCF	587.9	264.4					406.0	--
Sulphur River	FHM	66.0	17.1						
Sulphur River	FLI	69.7	11.2						
Sulphur River	FRD	223.2	128.3	513.7	47.4	341.5	27.6	365.0	--
Sulphur River	GOR	70.5	17.7						
Sulphur River	GOS	47.0	--						
Sulphur River	GRC	1003	120.1						

Sulphur River	GRP	90.0	--						
Sulphur River	GSF	66.5	18.6						
Sulphur River	GZS	167.2	82.7	411.5	3.5	375.2	140		
Sulphur River	ISS	66.3	13.6						
Sulphur River	LMB	191.3	109.9						
Sulphur River	LNG	910.1	217.1	1105	28.3	995.8	248	917.6	92.0
Sulphur River	LSF	92.9	19.9						
Sulphur River	OSS	57.9	8.8						
Sulphur River	PDF	1025	417.2						
Sulphur River	PIP	85.5	10.0						
Sulphur River	RCS	272.4	89.9	382.0	--	327.7	16.0	341.3	94.9
Sulphur River	RDS	56.8	7.1						
Sulphur River	RIS	48.4	5.0						
Sulphur River	RSS	82.0	--						
Sulphur River	SIC	928.0	--						
Sulphur River	SMB	453.4	106.9	469.3	77.0	410.9	59.5	430.0	123.0
Sulphur River	SNG	619.1	63.5			659.0	58.5	640.1	12.5
Sulphur River	SPB	167.4	72.7						
Sulphur River	SPG	582.1	78.4	749.0	--	639.7	85.6	591.2	47.3
Sulphur River	STB	342.5	36.1						
Sulphur River	TFS	66.9	17.3						
Sulphur River	WHB	197.8	83.8			226.8	13.9		
Sulphur River	WHC	152.6	70.0			241.0	--		

Sulphur River	WRM	84.0	30.6			
Sulphur River	YLB	130.8	40.4		210.0	14.1
Wright Patman	BBH	149.7	26.8			
Wright Patman	BCF	885.0	--			
Wright Patman	BLC	251.0	--			
Wright Patman	BLG	102.6	28.3			
Wright Patman	CCF	297.8	84.5			
Wright Patman	CLP	106.0	--			
Wright Patman	CRP	578.6	61.7			
Wright Patman	EMS	70.0	--			
Wright Patman	FCF	490.0	--			
Wright Patman	FRD	261.4	134.7			
Wright Patman	GOS	73.0	--			
Wright Patman	GSF	84.1	14.1			
Wright Patman	GZS	194.1	51.8			
Wright Patman	ISS	81.3	14.4			
Wright Patman	LMB	207.6	117.3			
Wright Patman	LSF	105.5	21.4			
Wright Patman	OSS	65.8	10.4			
Wright Patman	RDS	65.0	7.1			
Wright Patman	RIS	72.0	--			
Wright Patman	SMB	473.2	66.9			
Wright Patman	SPB	53.0	--			

Wright Patman	SPG	606.0	31.5					
Wright Patman	TFS	72.5	12.1					
Wright Patman	WHB	135.2	83.8					
Wright Patman	WHC	300.2	48.0					
Wright Patman	WRM	119.3	25.8					
Wright Patman	YLB	122.6	37.1					
Red River	BCF	598.1	198.9	530.0	21.2			
Red River	BLC	253.2	45.4					
Red River	BLG	83.8	39.7			121.0	--	
Red River	BLS	551.5	83.5			401.0	--	
Red River	BMB	619.9	66.8	609.0	--			
Red River	BTS	53.2	9.1					
Red River	CCF	156.6	91.7			202.0	--	
Red River	CRP			585.0	--			
Red River	EMS	81.0	--					
Red River	FCF	229.4	75.3					
Red River	FHM	74.3	8.0					
Red River	FLI	74.5	23.3					
Red River	FRD	256.3	116.1			246.4	69.5	
Red River	GRC	941.0	83.4	865.0	--			
Red River	GSF	77.5	20.8					
Red River	GZS	202.9	104.5	403.0	--	310.9	138	
Red River	LMB	178.5	55.5					
Red River	LNG	665.9	313.1					965.0 --
Red River	LSF	107.6	15.9					
Red River	OSS	64.6	17.4					
Red River	RCS	304.8	84.8	357.7	20.5	308.0	18.3	
Red River	RDS	48.7	10.7					
Red River	RSS	101.7	33.7					
Red River	SHS	585.0	--			460.0	--	
Red River	SIC	815.0	--					
Red River	SMB	519.5	144.4	412.0	50.5			
Red River	SNG	657.8	40.0	596.5	9.2	668.6	81.2	
Red River	SPB	158.6	67.2					
Red River	SPG	556.3	88.1			673.5	84.1	
Red River	TFS	62.1	20.3					

Red River	WHB	168.0	--			
Red River	WHC	233.9	71.5		233.5	54.4
Red River	WRM	106.0	35.4			

Appendix B, Table 7. Summary of the number of individuals collected in 2021 by waterbody and gear type. *Percent total was calculated separately for each waterbody. EF = electrofishing; GN = gillnet; EF/GN = herding into gillnets; HN = hoop net. See Appendix B, Table 5 for species key.

Waterbody	Species	EF	GN	EF/GN	HN	MFN	Total	% Total*
Sulphur River	BCF	85	2	23			110	4.5
Sulphur River	BLB	36					36	1.5
Sulphur River	BLC	52	2	1	7		62	2.5
Sulphur River	BLG	124		1	1	51	177	7.3
Sulphur River	BMB	77		1			78	3.2
Sulphur River	BOW	16		1		4	21	0.9
Sulphur River	BTS	36					36	1.5
Sulphur River	CCF	29	8	5	3	62	107	4.4
Sulphur River	CLP	4				45	49	2.0
Sulphur River	CRP	29		1		1	31	1.3
Sulphur River	DSF	1					1	0.0
Sulphur River	FCF	22				4	26	1.1
Sulphur River	FHM	5					5	0.2
Sulphur River	FLI	5				2	7	0.3
Sulphur River	FRD	152	2				154	6.3
Sulphur River	GOR	4				1	5	0.2
Sulphur River	GOS	1					1	0.0
Sulphur River	GSF	12					12	0.5
Sulphur River	GZS	475	1	9			485	19.9
Sulphur River	ISS	29					29	1.2
Sulphur River	LMB	42			1	2	45	1.8
Sulphur River	LNG	14		3	2		19	0.8
Sulphur River	LSF	91				45	136	5.6
Sulphur River	OSS	25				1	26	1.1
Sulphur River	PDF	1					1	0.0
Sulphur River	PIP	2				1	3	0.1
Sulphur River	RBS	2				2	4	0.2
Sulphur River	RCS	57	2	1			60	2.5
Sulphur River	RDS	17					17	0.7
Sulphur River	RES	3					3	0.1
Sulphur River	RIS	27				1	28	1.1
Sulphur River	SMB	201	7	1		44	253	10.4
Sulphur River	SNG	54	2				56	2.3

Sulphur River	SPB	76					76	3.1
Sulphur River	SPG	133	11	8	1	1	154	6.3
Sulphur River	TFS	15				1	16	0.7
Sulphur River	WHB	29	3			1	33	1.4
Sulphur River	WHC	35		2	1	3	41	1.7
Sulphur River	WRM	18					18	0.7
Sulphur River	YLB	16	1	1			18	0.7
Wright								
Patman	BCF	1					1	0.1
Wright								
Patman	BLG	46					46	5.8
Wright								
Patman	BMB	6					6	0.8
Wright								
Patman	CCF	13					13	1.6
Wright								
Patman	CLP	2					2	0.3
Wright								
Patman	CRP	4					4	0.5
Wright								
Patman	FLI	1					1	0.1
Wright								
Patman	FRD	4					4	0.5
Wright								
Patman	GOR	1					1	0.1
Wright								
Patman	GOS	1					1	0.1
Wright								
Patman	GZS	437					437	55.0
Wright								
Patman	ISS	110					110	13.8
Wright								
Patman	LMB	75					75	9.4
Wright								
Patman	LSF	20					20	2.5
Wright								
Patman	OSS	9					9	1.1
Wright								
Patman	RIS	17					17	2.1
Wright								
Patman	SMB	21					21	2.6
Wright								
Patman	SPB	8					8	1.0
Wright								
Patman	SPG	2					2	0.3

Wright Patman	TFS	2			2	0.3
Wright Patman	WHB	2			2	0.3
Wright Patman	WHC	2			2	0.3
Wright Patman	WRM	2			2	0.3
Wright Patman	YLB	9			9	1.1
Red River	BCF	11			11	1.6
Red River	BLB	3			3	0.4
Red River	BLG	8			8	1.2
Red River	BLS	7		6	13	1.9
Red River	BMB	1		1	2	0.3
Red River	BTS	3			3	0.4
Red River	CCF	8			8	1.2
Red River	EMS	2		153	155	22.4
Red River	FCF	49			49	7.1
Red River	FLI	2			2	0.3
Red River	FRD	17	1		18	2.6
Red River	GSF	1			1	0.1
Red River	GZS	26	5	1	32	4.6
Red River	ISS	2			2	0.3
Red River	LMB	8			8	1.2
Red River	LNG	9	1	2	12	1.7
Red River	LSF	26			26	3.8
Red River	RCS	27		16	43	6.2
Red River	RDS	3			3	0.4
Red River	RIS	6			6	0.9
Red River	RSS	1		7	8	1.2
Red River	SMB	10	1		11	1.6
Red River	SNG	2	1		3	0.4
Red River	SPB	13		80	93	13.5
Red River	SPG	11			11	1.6
Red River	TFS	2			2	0.3
Red River	WHB	2		155	157	22.7
Red River	WHC		1		1	0.1

Appendix B, Table 8. Summary of the number of individuals collected in 2022 by waterbody and gear type. *Percent total was calculated separately for each waterbody. EF = electrofishing; GN = gillnet; EF/GN = herding into gillnets; HN = hoop net; MFN = mini-fyke net. See Appendix B, Table 5 for species key.

Waterbody	Species	EF	GN	EF/GN	HN	MFN	Total	% Total*
Sulphur River	BCF	119	1	8	2	1	131	3.1
Sulphur River	BLB	87	1	9		3	100	2.4
Sulphur River	BLC	49		4	3		56	1.3
Sulphur River	BLG	159		1	1	11	172	4.1
Sulphur River	BLS	8					8	0.2
Sulphur River	BLT	1				6	7	0.2
Sulphur River	BMB	166	6	2			174	4.2
Sulphur River	BOW	4		3			7	0.2
Sulphur River	BTS	8				66	74	1.8
Sulphur River	CCF	49	2	5	2	1	59	1.4
Sulphur River	CRP	45		2	1		48	1.2
Sulphur River	EMS	3				5	8	0.2
Sulphur River	FCF	31			1		32	0.8
Sulphur River	FHM	8				1	9	0.2
Sulphur River	FLI	3					3	0.1
Sulphur River	FRD	232	3	2	1	1	239	5.7

Sulphur River	GOR	2					2	0.0
Sulphur River	GOS	1					1	0.0
Sulphur River	GRC	3					3	0.1
Sulphur River	GRP	1					1	0.0
Sulphur River	GSF	96				3	99	2.4
Sulphur River	GZS	184	2	19			205	4.9
Sulphur River	ISS	6				71	77	1.9
Sulphur River	LMB	43				1	44	1.1
Sulphur River	LNG	37	2	13	28		80	1.9
Sulphur River	LSF	77				2	79	1.9
Sulphur River	OSS	33				2	35	0.8
Sulphur River	PDF	2					2	0.0
Sulphur River	PIP	4					4	0.1
Sulphur River	RCS	148	1	9	3		161	3.9
Sulphur River	RDS	24				1	25	0.6
Sulphur River	RIS	9				87	96	2.3
Sulphur River	RSS	1					1	0.0
Sulphur River	SIC	1					1	0.0
Sulphur River	SMB	424	6	15	2	1	448	10.8
Sulphur River	SNG	122		7	9	1	139	3.3
Sulphur River	SPB	40				1	41	1.0
Sulphur River	SPG	424	1	34	6	3	468	11.3
Sulphur River	STB	2					2	0.0
Sulphur River	TFS	291				233	524	12.6
Sulphur River	WHB	35		4		1	40	1.0
Sulphur River	WHC	35		1		3	39	0.9
Sulphur River	WRM	13				1	14	0.3
Sulphur River	YLB	10		2			12	0.3
Sulphur River	GOT					2	2	0.0
Sulphur River	GAM					180	180	4.3
Sulphur River	LEP					208	208	5.0
Wright Patman	BBH	3					3	0.4
Wright Patman	BCF	1					1	0.1
Wright Patman	BLC	1					1	0.1
Wright Patman	BLG	74					74	8.7
Wright Patman	CCF	66					66	7.7
Wright Patman	CLP	1					1	0.1
Wright Patman	CRP	5					5	0.6
Wright Patman	EMS	1					1	0.1
Wright Patman	FCF	1					1	0.1
Wright Patman	FRD	8					8	0.9

Wright Patman	GOS	1				1	0.1
Wright Patman	GSF	18				18	2.1
Wright Patman	GZS	193				193	22.6
Wright Patman	ISS	43				43	5.0
Wright Patman	LMB	76				76	8.9
Wright Patman	LSF	26				26	3.0
Wright Patman	OSS	33				33	3.9
Wright Patman	RDS	2				2	0.2
Wright Patman	RIS	1				1	0.1
Wright Patman	SMB	16				16	1.9
Wright Patman	SPB	1				1	0.1
Wright Patman	SPG	5				5	0.6
Wright Patman	TFS	194				194	22.7
Wright Patman	WHB	47				47	5.5
Wright Patman	WHC	6				6	0.7
Wright Patman	WRM	3				3	0.4
Wright Patman	YLB	29				29	3.4
Red River	BCF	20	2			22	1.6
Red River	BLC	5				5	0.4
Red River	BLG	32		1	6	39	2.8
Red River	BLS	23		1		24	1.7
Red River	BLT				1	1	0.1
Red River	BMB	37	1			38	2.7
Red River	BTS	5			153	158	11.4
Red River	CCF	8		1		9	0.6
Red River	CRP		1			1	0.1
Red River	EMS	1				1	0.1
Red River	FCF	31				31	2.2
Red River	FHM	6				6	0.4
Red River	FLI	2				2	0.1
Red River	FRD	69		5	1	75	5.4
Red River	GRC	2	1			3	0.2
Red River	GSF	18			2	20	1.4
Red River	GZS	18	1	7		26	1.9
Red River	ISS				16	16	1.1
Red River	LMB	13				13	0.9
Red River	LNG	7			1	8	0.6
Red River	LSF	37				37	2.7
Red River	OSS	13			7	20	1.4
Red River	RCS	39	3	4		46	3.3
Red River	RDS	47			80	127	9.1

Red River	RIS				155	155	11.1
Red River	RSS	3				3	0.2
Red River	SHS	1		1		2	0.1
Red River	SIC	1				1	0.1
Red River	SMB	37	5			42	3.0
Red River	SNG	20	2	5		27	1.9
Red River	SPB	38				38	2.7
Red River	SPG	10		2		12	0.9
Red River	TFS	70			51	121	8.7
Red River	WHB	1				1	0.1
Red River	WHC	30		2	4	36	2.6
Red River	WRM	2				2	0.1
Red River	GAM				62	62	4.5
Red River	LEP				162	162	11.6

Appendix B, Table 9. Summary of the number of individuals collected in 2023 by waterbody and gear type. *Percent total was calculated separately for each waterbody. EF = electrofishing; GN = gillnet; EF/GN = herding into gillnets; MFN = mini-fyke net. Note: Hoop nets not used in 2023 because they were determined to be ineffective. See Appendix B, Table 5 for species key.

Waterbody	Species	EF	GN	EF/GN	MFN	Grand Total	% Total
Red River	BCF	20	1	1		22	0.9
Red River	BLB	2				2	<0.1
Red River	BLC	2				2	<0.1
Red River	BLG	16			4	20	0.7
Red River	BLS	2				2	0.1
Red River	BMB	12				12	0.5
Red River	BTS	1			55	56	2.0
Red River	CCF	4				4	0.2
Red River	EMS	19				19	0.8
Red River	FCF	30				30	1.2
Red River	FHM	2				2	0.1
Red River	FRD	4	10	1		15	0.6
Red River	GSF	3				3	0.1
Red River	GZS	26	1	3		30	1.2

Red River	LMB	4				4	0.2
Red River	LNG	7	5	1		13	0.5
Red River	LSF	7				7	0.3
Red River	OSS	2				2	0.1
Red River	RCS	9	9			18	0.7
Red River	RDS	82				82	3.4
Red River	RES	1				1	<0.1
Red River	RIS	1				1	<0.1
Red River	RSS	1			63	64	2.1
Red River	SIC	5				5	0.2
Red River	SMB	8	1			9	0.4
Red River	SNG	9	4	5		18	0.7
Red River	SPB	8				8	0.3
Red River	SPG	9	1	2		12	0.5
Red River	TFS	32				32	1.3
Red River	WHB	2				2	0.1
Red River	WHC	18				18	0.7
Red River	WRM	1				1	<0.1
Red River	BHM	14				14	0.6
Sulphur River	BBH	1				1	<0.1
Sulphur River	BCF	47	13	2		62	2.6
Sulphur River	BLB	30	5			35	1.4
Sulphur River	BLC	17	1	3		21	0.9
Sulphur River	BLG	60	4		6	70	2.9
Sulphur River	BLT	1				1	<0.1
Sulphur River	BMB	53	3	1		57	2.3
Sulphur River	BOW	3	3			6	0.2
Sulphur River	BTS	3				3	0.1
Sulphur River	CCF	15	41	7		63	2.6
Sulphur River	CRP	13	1			14	0.6
Sulphur River	EMS	7				7	0.3
Sulphur River	FCF	18				18	0.7
Sulphur River	FHM	4				4	0.2
Sulphur River	FRD	63	6	8		77	3.2
Sulphur River	GAM				250	250	10.3
Sulphur River	GRC	2				2	0.1
Sulphur River	GRP	1				1	0.0
Sulphur River	GSF	39				39	1.6
Sulphur River	GZS	204	24	43		271	11.1
Sulphur River	ISS	11				11	0.5
Sulphur River	LEP				121	121	5.0
Sulphur River	LMB	8	1			9	0.4
Sulphur River	LNG	23	2	2		27	1.1

Sulphur River	LSF	31			31	1.3
Sulphur River	OSS	22			22	0.9
Sulphur River	PDF	1			1	<0.1
Sulphur River	RCS	130	5	2	137	5.6
Sulphur River	RDS	182			182	7.5
Sulphur River	RES	2			2	0.1
Sulphur River	RSS	3			3	0.1
Sulphur River	SMB	158	19	15	192	7.9
Sulphur River	SNG	15	6	5	26	1.1
Sulphur River	SPB	16	7		23	0.9
Sulphur River	SPG	99	9	12	120	4.9
Sulphur River	TFS	239			239	9.8
Sulphur River	WHB	57	12	2	71	2.9
Sulphur River	WHC	60	5	16	81	3.3
Sulphur River	WRM	10			10	0.4
Sulphur River	YLB	27	10	23	60	2.5
Sulphur River	GAM	2			2	0.1
Sulphur River	BHM	28			28	1.2

Appendix B, Table 10. Summary of average catch rates (catch per unit effort) of fish species by waterbody, gear, and year. EF = electrofishing; GN = gillnet; EF/GN = herding into gillnets; HN = hoop net; MFN = mini-fyke net. See Appendix B, Table 5 for species key.

Year	Gear	Species	Red River	Sulphur River	Wright Patman
2021	EF	BCF	0.52 (0.47)	0.81 (1.46)	0.04 (0.09)
2021	EF	BLB	0.14 (0.26)	0.41 (0.76)	
2021	EF	BLC		0.61 (1.29)	
2021	EF	BLG	0.38 (0.45)	1.53 (2.64)	1.75 (2.59)
2021	EF	BLS	0.33 (0.75)		
2021	EF	BMB	0.05 (0.13)	1.04 (1.62)	0.43 (1.13)
2021	EF	BOW		0.19 (0.62)	
2021	EF	BTS	0.14 (0.26)	0.35 (0.68)	
2021	EF	CCF	0.40 (0.87)	0.33 (0.56)	0.57 (0.37)
2021	EF	CLP		0.04 (0.11)	0.07 (0.12)
2021	EF	CRP		0.34 (0.57)	0.25 (0.35)
2021	EF	DSF		0.01 (0.05)	
2021	EF	EMS	0.10 (0.25)		
2021	EF	FCF	2.33 (5.03)	0.21 (0.28)	
2021	EF	FHM		0.05 (0.20)	
2021	EF	FLI	0.10 (0.16)	0.04 (0.13)	0.14 (0.38)
2021	EF	FRD	0.81 (1.29)	1.54 (1.93)	0.36 (0.75)
2021	EF	GOR		0.04 (0.14)	0.14 (0.38)
2021	EF	GOS		0.01 (0.04)	0.04 (0.09)

2021	EF	GSF	0.05 (0.13)	0.13 (0.33)	
2021	EF	GZS	1.26 (2.01)	5.02 (5.77)	17.79 (18.04)
2021	EF	ISS	0.10 (0.25)	0.35 (0.88)	5.21 (8.68)
2021	EF	LMB	0.38 (1.01)	0.50 (0.75)	3.00 (2.15)
2021	EF	LNG	0.50 (0.59)	0.15 (0.25)	
2021	EF	LSF	1.24 (1.50)	0.86 (0.99)	0.82 (0.99)
2021	EF	OSS		0.25 (0.61)	0.32 (0.55)
2021	EF	PDF		0.01 (0.07)	
2021	EF	PIP		0.02 (0.07)	
2021	EF	RBS		0.02 (0.08)	
2021	EF	RCS	1.33 (1.94)	0.63 (0.65)	
2021	EF	RDS	0.14 (0.38)	0.16 (0.34)	
2021	EF	RES		0.03 (0.10)	
2021	EF	RIS	0.29 (0.76)	0.26 (0.65)	1.57 (3.31)
2021	EF	RSS	0.05 (0.13)		
2021	EF	SMB	0.48 (0.63)	2.17 (2.82)	0.75 (0.95)
2021	EF	SNG	0.10 (0.25)	0.59 (0.58)	
2021	EF	SPB	0.62 (0.45)	0.73 (0.84)	0.29 (0.65)
2021	EF	SPG	0.55 (0.74)	1.37 (2.04)	0.11 (0.20)
2021	EF	TFS	0.10 (0.25)	0.18 (0.58)	0.07 (0.19)
2021	EF	WHB	0.10 (0.25)	0.36 (0.75)	0.07 (0.19)
2021	EF	WHC		0.41 (1.01)	0.07 (0.19)
2021	EF	WRM		0.29 (1.07)	0.07 (0.12)
2021	EF	YLB		0.20 (0.42)	0.32 (0.55)
2021	GN	BCF		0.50 (0.71)	
2021	GN	BLC		0.50 (0.71)	
2021	GN	CCF		2.00 (0.71)	
2021	GN	FRD		0.50 (0.00)	
2021	GN	GZS		0.25 (0.35)	
2021	GN	RCS		0.50 (0.71)	
2021	GN	SMB		1.75 (2.47)	
2021	GN	SNG		0.50 (0.00)	
2021	GN	SPG		2.75 (3.18)	
2021	GN	WHB		0.75 (1.06)	
2021	GN	YLB		0.25 (0.35)	
2021	EF/GN	BCF		1.93 (2.77)	
2021	EF/GN	BLC		0.07 (0.15)	
2021	EF/GN	BLG		0.07 (0.15)	
2021	EF/GN	BMB		0.07 (0.15)	
2021	EF/GN	BOW		0.20 (0.45)	
2021	EF/GN	CCF		0.73 (1.30)	
2021	EF/GN	CRP		0.20 (0.45)	
2021	EF/GN	FRD	0.06 (0.14)		

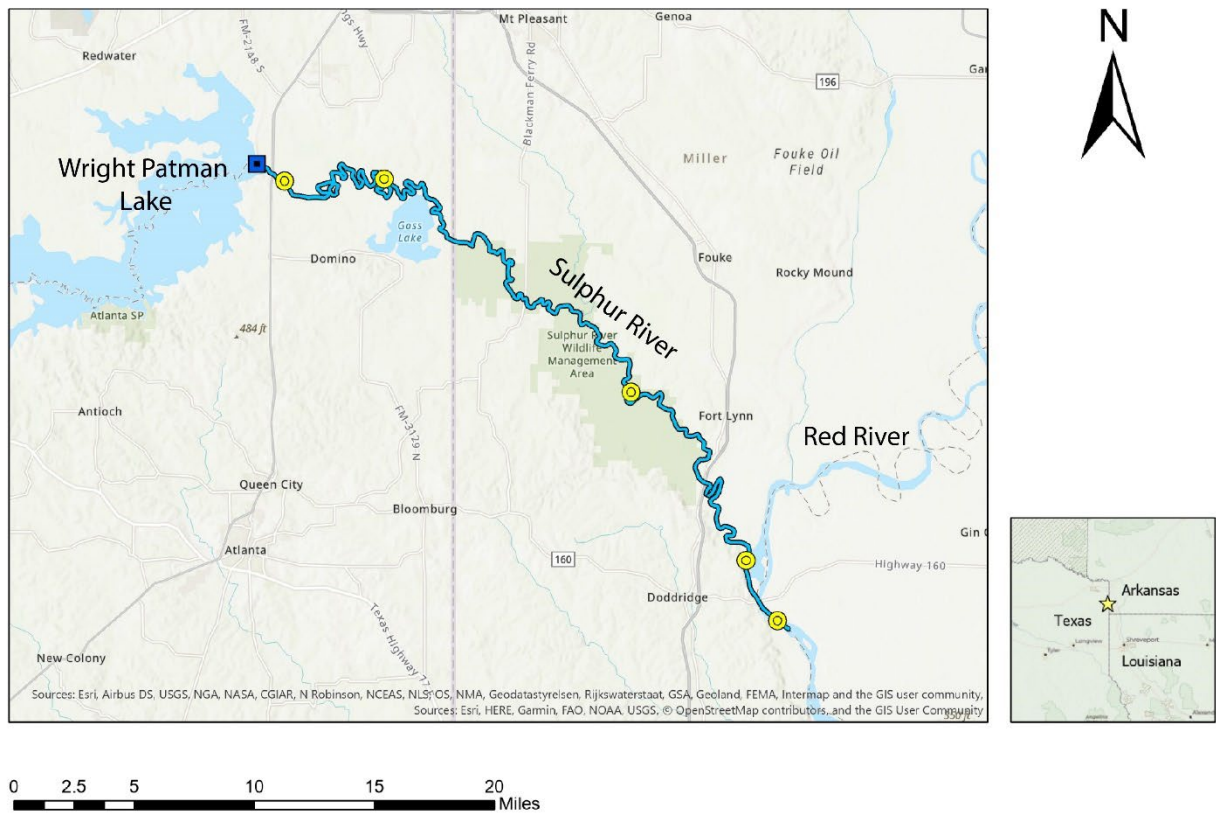
2021	EF/GN	GZS	0.28 (0.44)	0.87 (0.96)	
2021	EF/GN	LNG	0.06 (0.14)	0.33 (0.47)	
2021	EF/GN	RCS		0.07 (0.15)	
2021	EF/GN	SMB	0.06 (0.14)	0.07 (0.15)	
2021	EF/GN	SNG	0.06 (0.14)		
2021	EF/GN	SPG		1.33 (2.09)	
2021	EF/GN	WHC	0.06 (0.14)	0.13 (0.30)	
2021	EF/GN	YLB		0.07 (0.15)	
2021	HN	BLC		0.13 (0.42)	
2021	HN	BLG		0.02 (0.07)	
2021	HN	CCF		0.06 (0.15)	
2021	HN	LMB		0.02 (0.07)	
2021	HN	LNG		0.04 (0.14)	
2021	HN	SPG		0.01 (0.05)	
2021	HN	WHC		0.02 (0.07)	
2022	EF	BBH			0.11 (0.28)
2022	EF	BCF	0.83 (0.85)	1.18 (1.21)	0.04 (0.09)
2022	EF	BLB		0.89 (1.14)	
2022	EF	BLC	0.21 (0.17)	0.49 (0.72)	0.04 (0.09)
2022	EF	BLG	1.33 (1.49)	1.57 (1.27)	2.71 (2.70)
2022	EF	BLS	0.96 (1.53)	0.08 (0.34)	
2022	EF	BLT		0.01 (0.04)	
2022	EF	BMB	1.54 (3.43)	1.51 (1.90)	
2022	EF	BOW		0.04 (0.14)	
2022	EF	BTS	0.21 (0.25)	0.07 (0.15)	
2022	EF	CCF	0.33 (0.47)	0.48 (0.34)	2.42 (1.31)
2022	EF	CLP			0.04 (0.09)
2022	EF	CRP		0.46 (0.82)	0.18 (0.19)
2022	EF	EMS	0.04 (0.12)	0.03 (0.09)	0.04 (0.09)
2022	EF	FCF	1.29 (1.52)	0.28 (0.36)	0.04 (0.09)
2022	EF	FHM	0.25 (0.58)	0.08 (0.22)	
2022	EF	FLI	0.08 (0.15)	0.03 (0.08)	
2022	EF	FRD	2.88 (2.87)	2.21 (3.64)	0.29 (0.22)
2022	EF	GOR		0.02 (0.06)	
2022	EF	GOS		0.01 (0.05)	0.04 (0.09)
2022	EF	GRC	0.08 (0.15)	0.03 (0.07)	
2022	EF	GRP		0.01 (0.05)	
2022	EF	GSF	0.75 (0.87)	0.89 (1.62)	0.80 (1.59)
2022	EF	GZS	0.75 (0.94)	1.87 (2.72)	6.93 (2.93)
2022	EF	ISS		0.06 (0.16)	1.63 (2.13)
2022	EF	LMB	0.54 (0.31)	0.42 (0.47)	2.79 (0.96)
2022	EF	LNG	0.29 (0.33)	0.35 (0.34)	
2022	EF	LSF	1.54 (1.40)	0.77 (1.02)	0.96 (0.83)

2022	EF	OSS	0.54 (1.40)	0.31 (0.60)	1.32 (1.29)
2022	EF	PDF		0.02 (0.05)	
2022	EF	PIP		0.04 (0.15)	
2022	EF	RCS	1.63 (1.33)	1.50 (1.50)	
2022	EF	RDS	1.96 (2.12)	0.23 (0.44)	0.07 (0.19)
2022	EF	RIS		0.09 (0.28)	0.04 (0.09)
2022	EF	RSS	0.13 (0.25)	0.01 (0.03)	
2022	EF	SHS	0.04 (0.12)		
2022	EF	SIC	0.04 (0.12)	0.01 (0.05)	
2022	EF	SMB	1.54 (1.48)	4.28 (3.42)	0.57 (0.69)
2022	EF	SNG	0.83 (1.38)	1.19 (1.27)	
2022	EF	SPB	1.58 (1.48)	0.37 (0.38)	0.04 (0.09)
2022	EF	SPG	0.42 (0.46)	4.22 (4.61)	0.18 (0.31)
2022	EF	STB		0.02 (0.10)	
2022	EF	TFS	2.92 (2.92)	2.66 (3.04)	7.01 (6.98)
2022	EF	WHB	0.04 (0.12)	0.37 (0.73)	1.99 (3.16)
2022	EF	WHC	1.25 (1.02)	0.38 (0.66)	0.21 (0.30)
2022	EF	WRM	0.08 (0.15)	0.12 (0.28)	0.11 (0.20)
2022	EF	YLB		0.11 (0.18)	1.07 (1.27)
2022	GN	BCF	0.29 (0.49)	0.14 (0.38)	
2022	GN	BLB		0.14 (0.38)	
2022	GN	BMB	0.14 (0.38)	0.86 (1.57)	
2022	GN	CCF		0.29 (0.49)	
2022	GN	CRP	0.14 (0.38)		
2022	GN	FRD		0.43 (0.79)	
2022	GN	GRC	0.14 (0.38)		
2022	GN	GZS	0.14 (0.38)	0.14 (0.38)	
2022	GN	LNG		0.29 (0.49)	
2022	GN	RCS	0.43 (0.53)	0.14 (0.38)	
2022	GN	SMB	0.71 (0.95)	0.50 (0.96)	
2022	GN	SNG	0.29 (0.76)		
2022	GN	SPG		0.14 (0.38)	
2022	EF/GN	BCF		0.33 (0.71)	
2022	EF/GN	BLB		0.38 (0.63)	
2022	EF/GN	BLC		0.17 (0.25)	
2022	EF/GN	BLG		0.08 (0.15)	
2022	EF/GN	BLS	0.04 (0.12)		
2022	EF/GN	BMB		0.08 (0.15)	
2022	EF/GN	BOW		0.13 (0.25)	
2022	EF/GN	CCF	0.04 (0.12)	0.21 (0.35)	
2022	EF/GN	CRP		0.08 (0.15)	
2022	EF/GN	FRD	0.21 (0.31)	0.08 (0.24)	
2022	EF/GN	GZS	0.17 (0.25)	0.92 (1.35)	

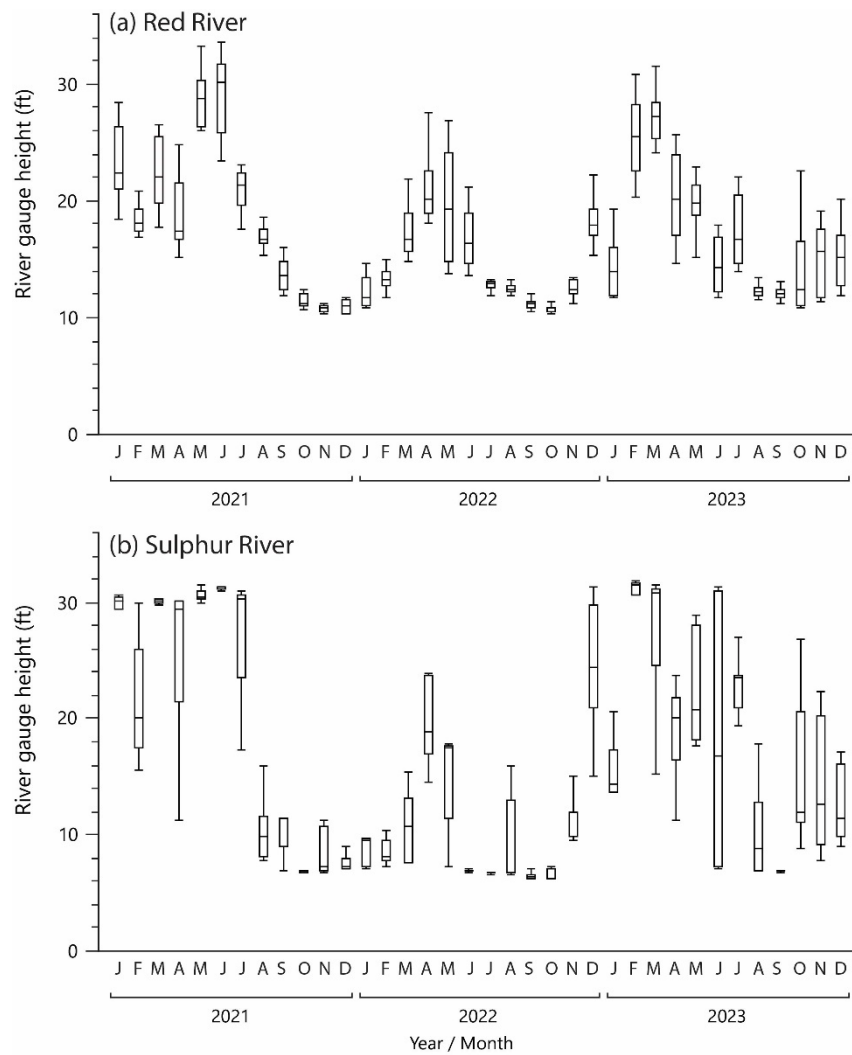
2022	EF/GN	LNG		0.54 (0.59)
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2022	EF/GN	SHS	0.04 (0.12)	
2022	EF/GN	SMB		0.63 (0.81)
2022	EF/GN	SNG	0.13 (0.25)	0.38 (0.38)
2022	EF/GN	SPG	0.08 (0.15)	1.42 (0.71)
2022	EF/GN	WHB		0.17 (0.47)
2022	EF/GN	WHC	0.08 (0.24)	0.04 (0.12)
2022	EF/GN	YLB		0.08 (0.15)
2022	HN	BCF		0.01 (0.04)
2022	HN	BLC		0.02 (0.05)
2022	HN	BLG		0.01 (0.03)
2022	HN	CCF		0.01 (0.04)
2022	HN	CRP		0.01 (0.03)
2022	HN	FCF		0.01 (0.03)
2022	HN	FRD		0.01 (0.03)
2022	HN	LNG	0.02 (0.06)	0.15 (0.77)
2022	HN	RCS		0.02 (0.07)
2022	HN	SMB		0.01 (0.04)
2022	HN	SNG		0.05 (0.21)
2022	HN	SPG		0.03 (0.09)
2022	MFN	BCF		0.06 (0.17)
2022	MFN	BLB		0.20 (0.50)
2022	MFN	BLG	1.42 (1.38)	1.71 (2.02)
2022	MFN	BLT	0.33 (0.58)	0.67 (1.32)
2022	MFN	BTS	33.92 (29.42)	9.59 (15.71)
2022	MFN	CCF		0.04 (0.11)
2022	MFN	EMS		0.28 (0.83)
2022	MFN	FHM		0.13 (0.26)
2022	MFN	FRD	0.33 (0.58)	0.11 (0.22)
2022	MFN	GAM	7.83 (7.01)	17.91 (27.35)
2022	MFN	GOT		0.11 (0.33)
2022	MFN	GSF	0.67 (1.15)	0.17 (0.50)
2022	MFN	ISS	2.56 (3.60)	4.33 (11.72)
2022	MFN	LEP	31.33 (46.29)	9.76 (22.74)
2022	MFN	LMB		0.07 (0.22)
2022	MFN	LSF		0.15 (0.44)
2022	MFN	OSS	1.00 (1.00)	0.67 (1.30)
2022	MFN	RDS	1.67 (2.89)	6.41 (18.85)
2022	MFN	RIS	6.33 (7.77)	11.56 (22.84)
2022	MFN	SMB		0.06 (0.17)
2022	MFN	SNG		0.11 (0.22)
2022	MFN	SPB		0.06 (0.17)

2022	MFN	SPG		0.20 (0.45)
2022	MFN	TFS	2.33 (3.21)	13.28 (27.53)
2022	MFN	WHB		0.06 (0.17)
2022	MFN	WHC	1.11 (1.17)	0.39 (0.65)
2022	MFN	WRM		0.06 (0.17)
2023	EF	BBH		
2023	EF	BCF	1.33 (0.53)	1.09 (1.11)
2023	EF	BLB	0.33 (0.00)	0.91 (0.48)
2023	EF	BLC	0.33 (0.00)	0.63 (0.45)
2023	EF	BLG	1.78 (1.92)	1.05 (1.13)
2023	EF	BLS		
2023	EF	BLT		
2023	EF	BMB	1.33 (1.20)	1.01 (1.29)
2023	EF	BOW		0.31 (0.05)
2023	EF	BTS		0.38 (0.18)
2023	EF	CCF	0.67 (0.47)	0.46 (0.28)
2023	EF	CRP		0.51 (0.28)
2023	EF	EMS		0.46 (0.22)
2023	EF	FCF	2.00 (0.67)	0.54 (0.38)
2023	EF	FHM		0.33 (0.14)
2023	EF	FRD	0.67 (0.47)	1.48 (2.37)
2023	EF	GAM		0.25 (0.00)
2023	EF	GRC		0.25 (0.00)
2023	EF	GRP		
2023	EF	GSF	0.33 (0.00)	0.98 (0.80)
2023	EF	GZS	2.17 (1.55)	3.15 (5.25)
2023	EF	ISS		0.67 (0.75)
2023	EF	LMB	0.44 (0.19)	0.50 (0.33)
2023	EF	LNG	1.17 (0.71)	0.58 (0.50)
2023	EF	LSF	1.17 (1.18)	0.66 (0.31)
2023	EF	OSS		0.55 (0.52)
2023	EF	PDF		
2023	EF	RCS	0.75 (0.42)	1.95 (2.60)
2023	EF	RDS	5.47 (10.19)	2.86 (3.75)
2023	EF	RES		0.25 (0.00)
2023	EF	RIS		
2023	EF	RSS		0.38 (0.18)
2023	EF	SIC	0.42 (0.17)	
2023	EF	SMB	0.67 (0.47)	2.36 (1.54)
2023	EF	SNG	0.60 (0.43)	0.38 (0.23)
2023	EF	SPB	0.89 (0.69)	0.43 (0.26)
2023	EF	SPG	0.75 (0.42)	1.55 (1.45)
2023	EF	TFS	2.67 (2.24)	3.88 (6.85)

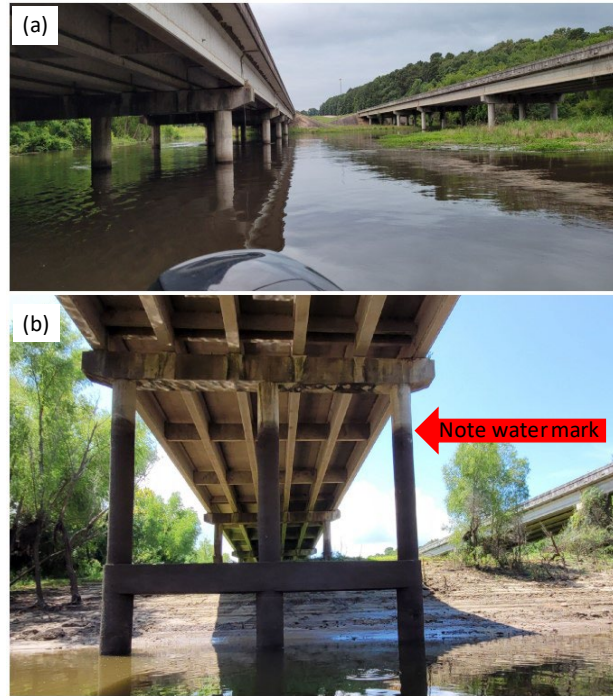
2023	EF	WHB		1.15 (1.68)
2023	EF	WHC	1.50 (1.69)	1.58 (1.73)
2023	EF	WRM		1.25 (1.41)
2023	EF	YLB		1.20 (1.52)
2023	EF	BHM	2.33 (1.89)	1.17 (0.68)
2023	GN	BCF		
2023	GN	BLB		1.25 (0.35)
2023	GN	BLC		
2023	GN	BLG		1.00 (0.71)
2023	GN	BMB		
2023	GN	BOW		0.75 (0.35)
2023	GN	CCF		6.83 (10.10)
2023	GN	CRP		
2023	GN	FRD	2.50 (2.83)	1.50 (0.71)
2023	GN	GZS		4.00 (5.22)
2023	GN	LMB		
2023	GN	LNG	0.83 (0.58)	
2023	GN	RCS	1.50 (1.73)	1.25 (1.06)
2023	GN	SMB		2.38 (2.78)
2023	GN	SNG	1.00 (0.71)	1.00 (0.87)
2023	GN	SPB		
2023	GN	SPG		1.13 (0.95)
2023	GN	WHB		
2023	GN	WHC		0.83 (0.58)
2023	GN	YLB		
2023	EF/GN	BCF		
2023	EF/GN	BLC		0.50 (0.24)
2023	EF/GN	BMB		
2023	EF/GN	CCF		1.17 (0.24)
2023	EF/GN	FRD		0.89 (0.96)
2023	EF/GN	GZS	0.33 (0.00)	4.78 (7.41)
2023	EF/GN	LNG		0.33 (0.00)
2023	EF/GN	RCS		
2023	EF/GN	SMB		1.25 (0.83)
2023	EF/GN	SNG	0.56 (0.38)	0.83 (0.71)
2023	EF/GN	SPG		1.33 (0.33)
2023	EF/GN	WHB		
2023	EF/GN	WHC		1.33 (0.67)
2023	EF/GN	YLB		2.56 (1.84)



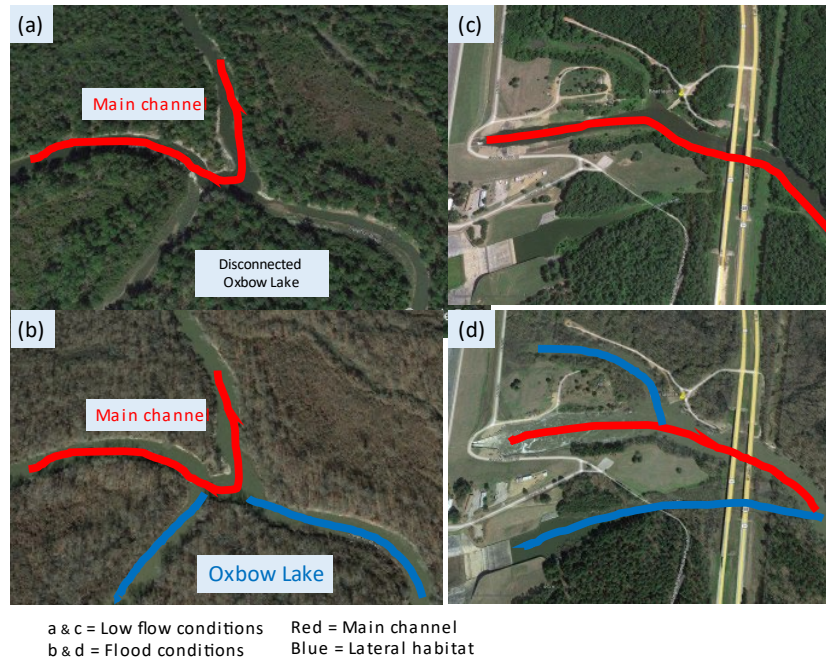
Appendix B, Figure 1. Map of the 72-km segment of the lower Sulphur River that runs below Wright Patman Lake (blue square = dam) to the Red River. Yellow circles represent river sampling sites.



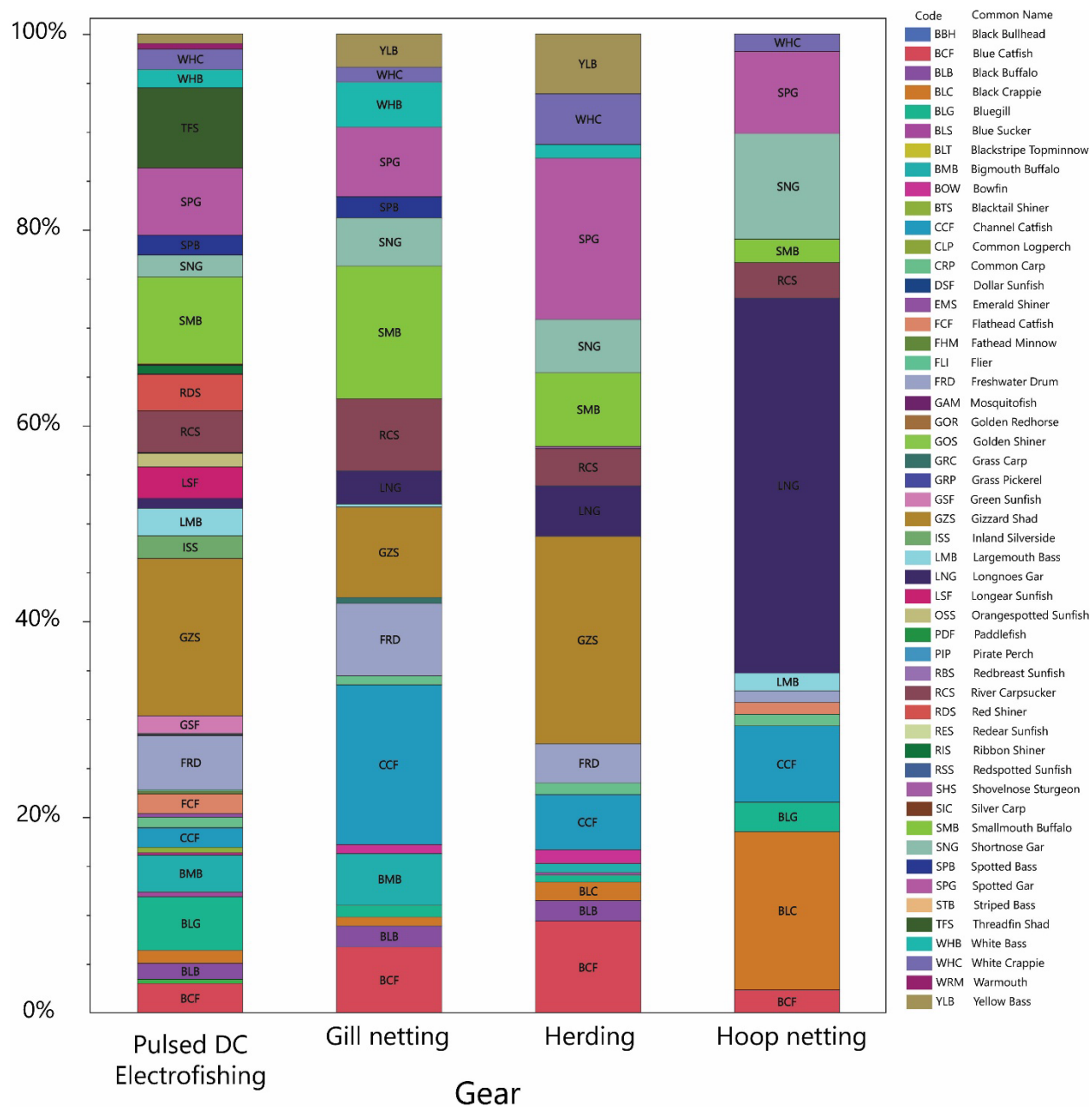
Appendix B, Figure 2. Hydrograph of river gauge height (ft) for the (A) Red River (USGS station: 07344370) and (B) Sulphur River (USGS station: 07344210) from Dec-31-2020 through Dec-31-2023. Boxplots reflect daily gauge heights for each month and year.



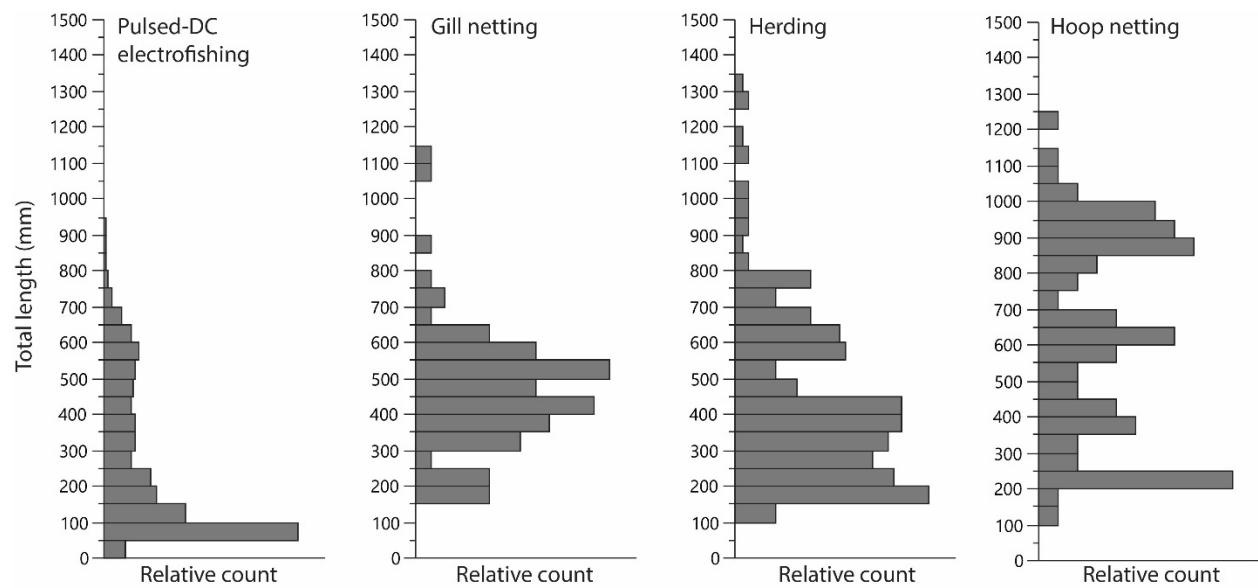
Appendix B, Figure 3. Photograph of the Sulphur River during (a) high and (b) low flow conditions. Note the water mark stain on the same pillar in panel b. Photos were taken approximately one month apart during 2021.



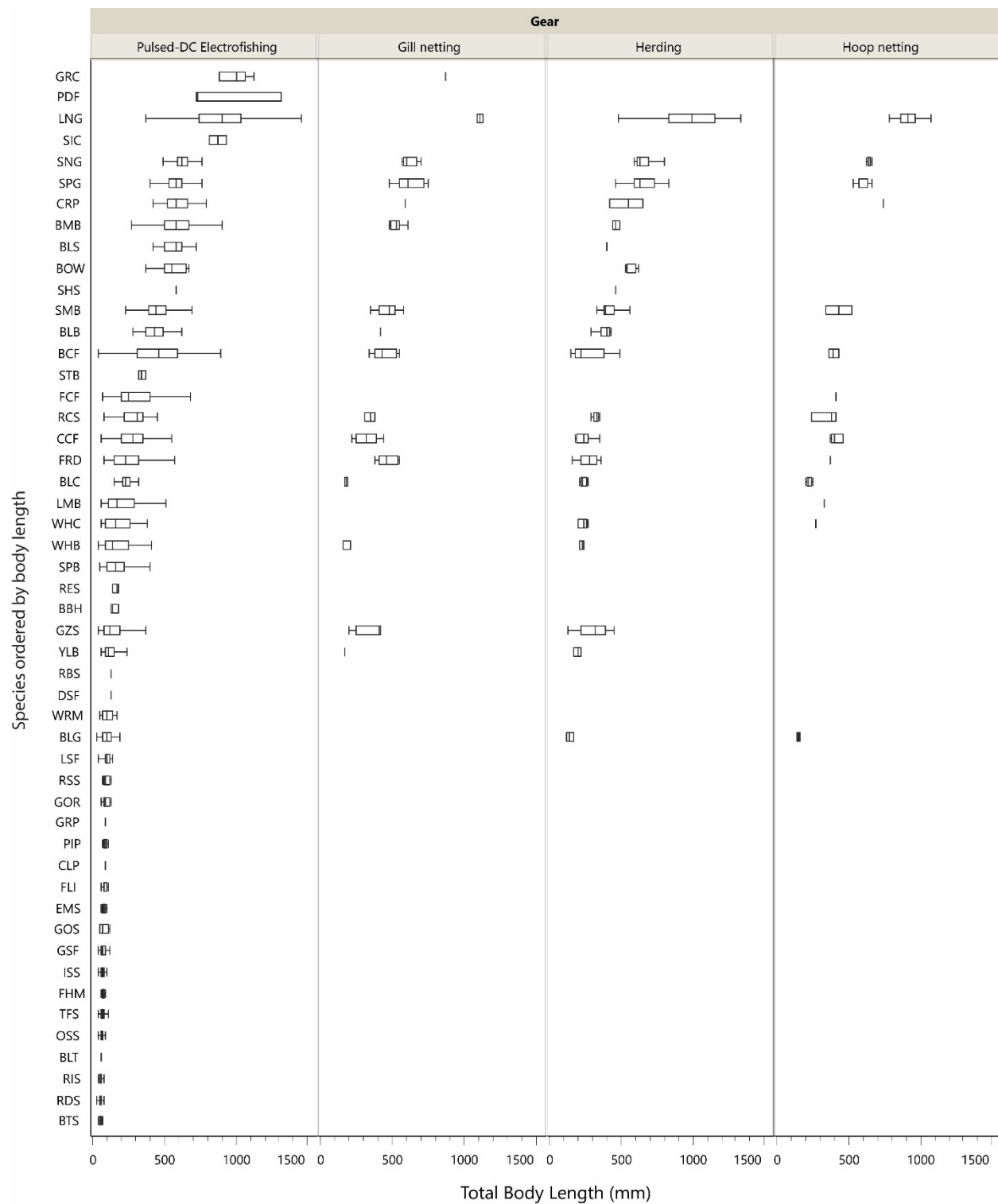
Appendix B, Figure 4. Aerial imagery of lateral river habitats including oxbow lakes (a, b) and side channels (c, d) in the Sulphur River. During base flow conditions, these habitats are disconnected from the main channel and are inaccessible for sampling (a, c). In contrast, during flooding, these habitats are accessible to sampling via boat.



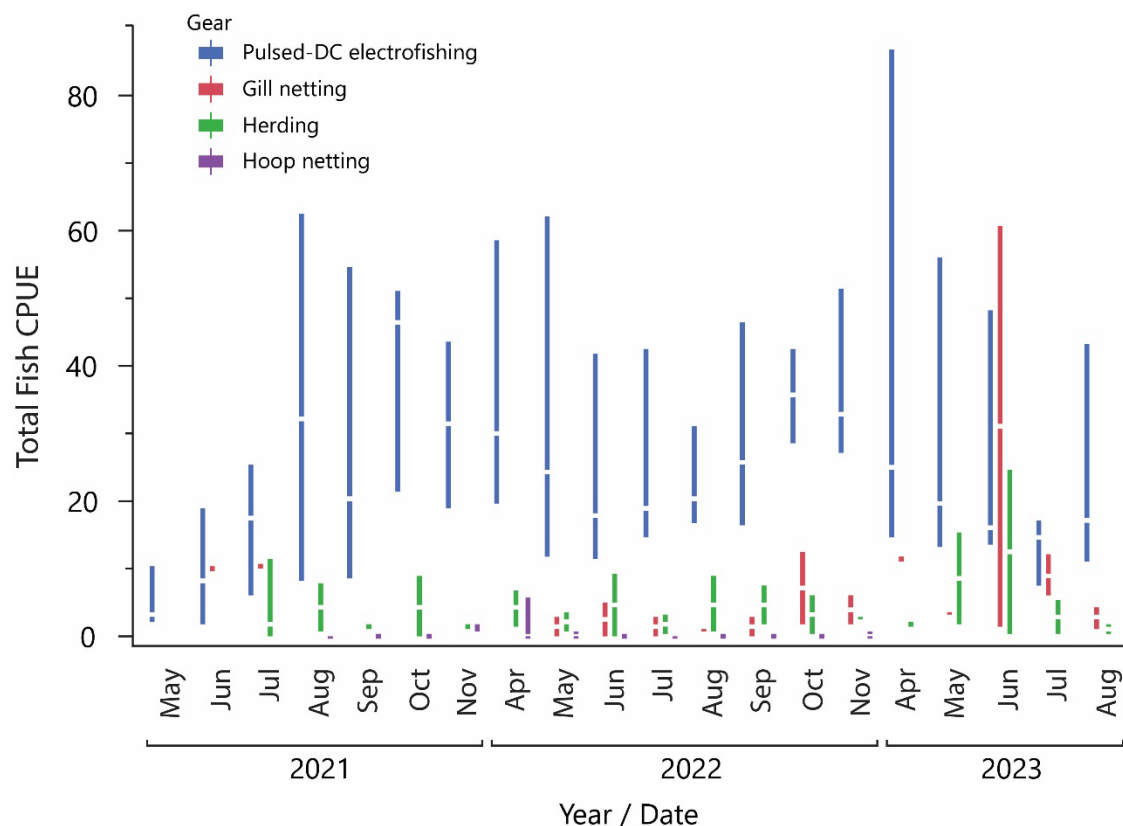
Appendix B, Figure 5. Composition of fishes captured by pulsed-DC electrofishing, gill netting, herding, and hoop netting during the 2021-2023 field seasons. Data from the Red River and Sulphur River were combined. Refer to species codes and corresponding common names.



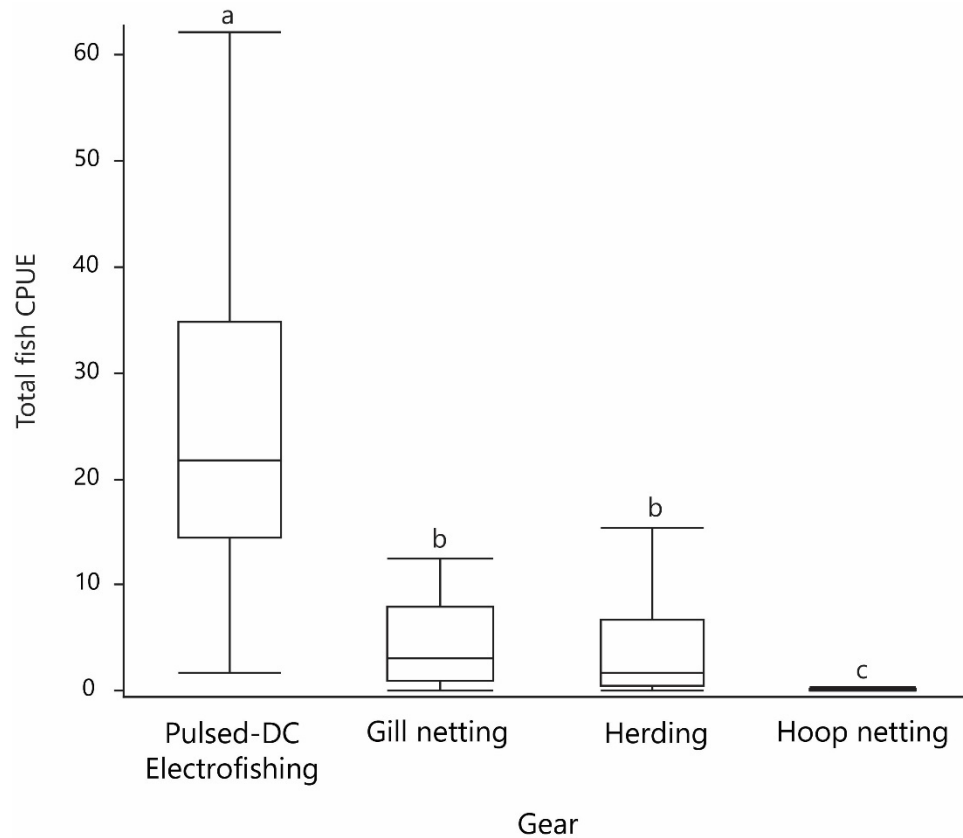
Appendix B, Figure 6. Length-frequency distribution of all fishes captured by pulsed-DC electrofishing, gill netting, herding, and hoop netting in the Sulphur River from 2021-2023. Counts are relative to each gear type.



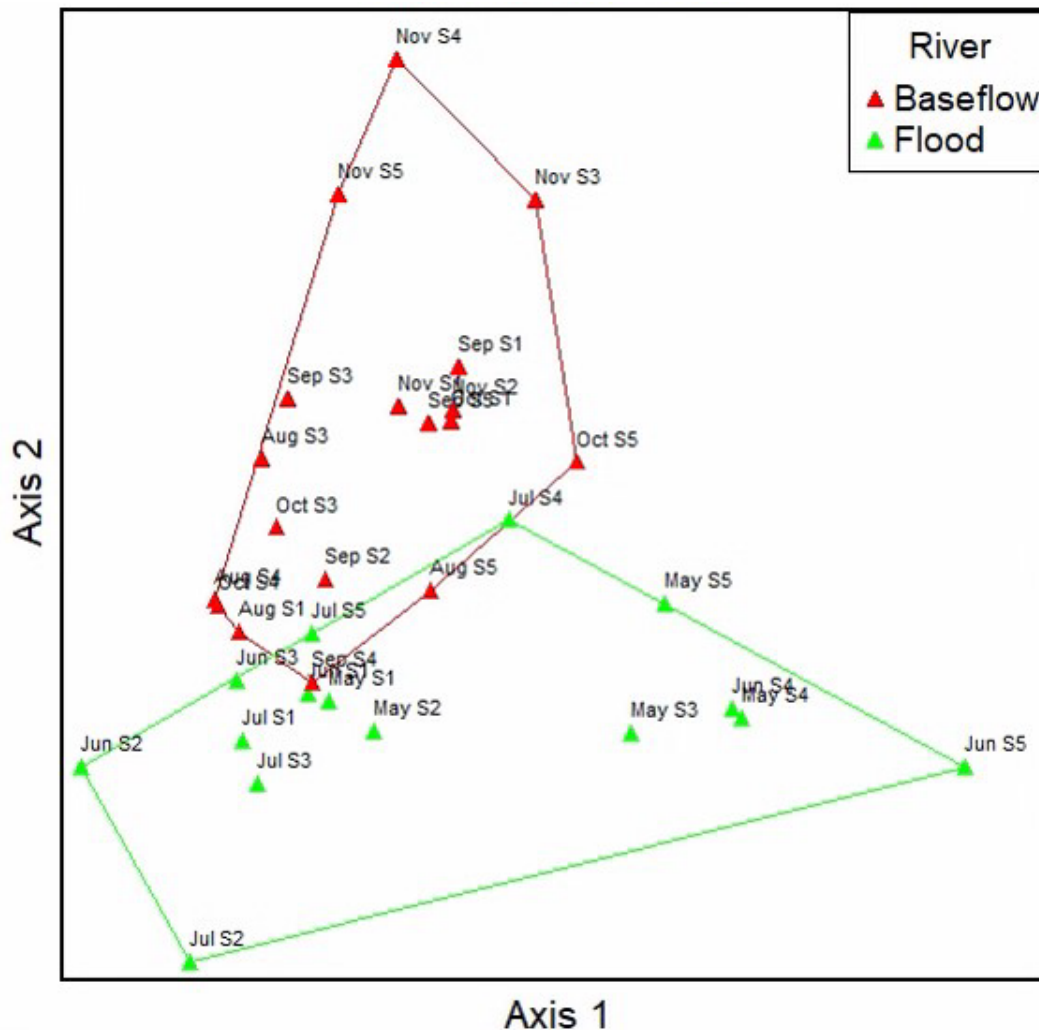
Appendix B, Figure 7. Boxplots of fish species captured via pulsed-DC electrofishing, gill netting, herding, and hoop netting in the Sulphur River and Red River during the 2021-2023 field seasons. Refer to Appendix B, Table 5 for species codes.



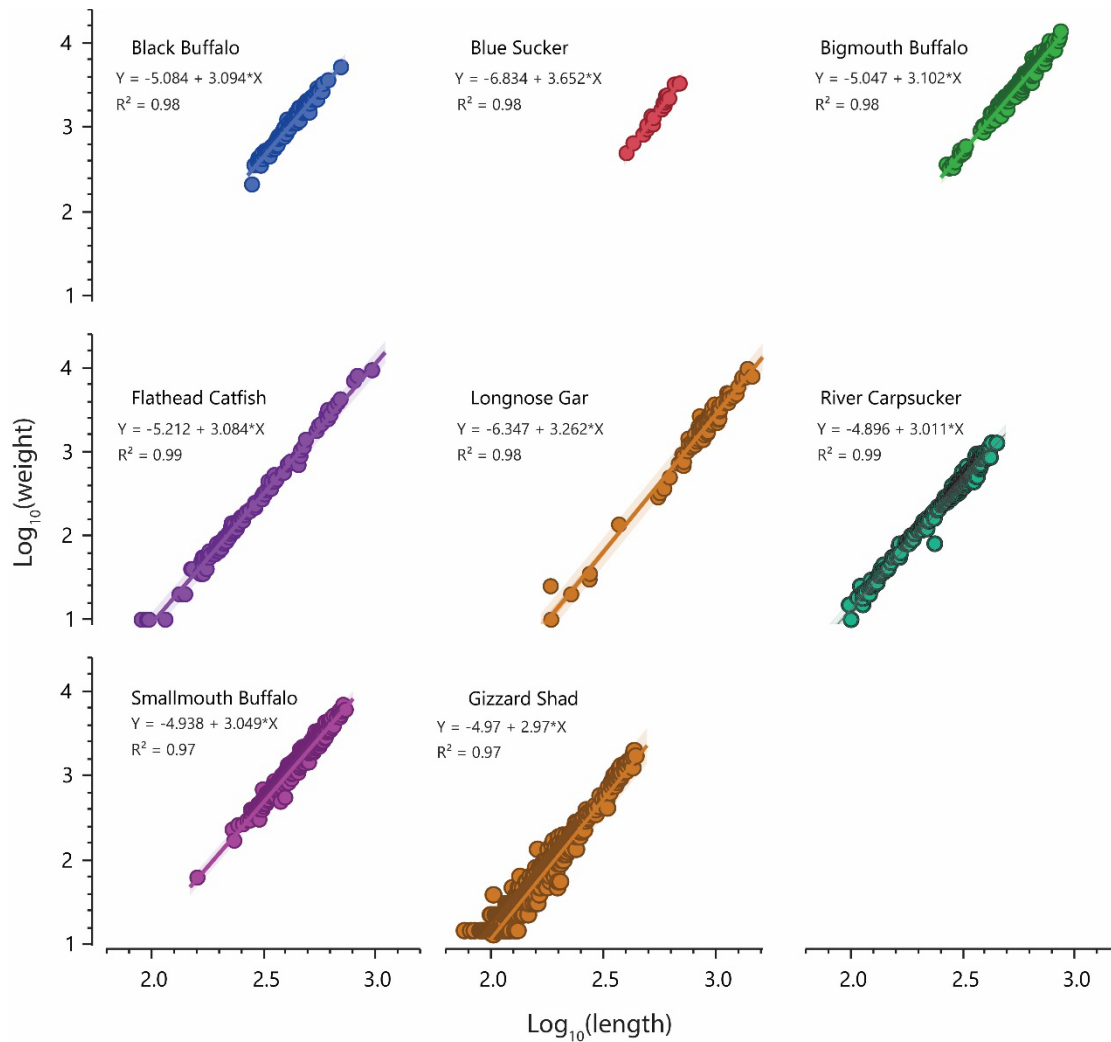
Appendix B, Figure 8. Boxplots summarizing monthly total catch per unit effort of fish using pulsed-DC electrofishing (active approach; blue), gill netting (passive approach; red), herding (active approach; green), and hoop netting (passive approach; purple) in the Sulphur River from 2021-2023. Note: sampling effort differs between active and passive sampling approaches. Values are provided for basic comparison of catches among sampling approaches.



Appendix B, Figure 9. Boxplots of total fish catch per unit effort (CPUE) in the Sulphur River (2021-2023). Letters (a,b,c) denote differences among gears following a post hoc pairwise comparisons (Wilcoxon method).



Appendix B, Figure 10. Changes in the Sulphur River fish assemblage from May through November 2021 in relation to the altered flow regime. Composition and structure of the fish assemblage differed between the spring flood (green triangles, lines) and summer baseflow (red triangle, line) conditions (analysis: non-metric multidimensional scaling (NMDS); PC-ORD 7).



Appendix B, Figure 11. Length-weight relationships of Bigmouth Buffalo *Ictiobus cyprinellus*, Smallmouth Buffalo *I. bubalus*, Black Buffalo *I. niger*, Longnose Gar *Lepisosteus osseus*, Blue Sucker *Cycleptus elongatus*, Flathead Catfish *Pylodictis olivaris*, River Carpsucker *Carpionodes carpio*, and Gizzard Shad *Dorosoma cepedianum* captured along the Sulphur River from 2021-2023. Linear models were used to establish baseline conditions for each native species to track potential effects of bigheaded carp.



Appendix B, Figure 12. Photograph of woody debris spanning the width of the Sulphur River at low-flow conditions. Such habitat features may block or slow down the movements of Bigheaded carp and other fishes during certain seasons. Photo taken in late summer, 2021.



Appendix B, Figure 13. Photograph of a constrained stream channel connecting a shallow pool on the Sulphur River. Photo taken in late summer, 2021.

Attachments

The following attachments are peer-reviewed publications generated by this project attached here and referenced in the report to avoid duplicate publication.

Attachment A: Birdsall, B, Dattilo J, Fuqua Z, Brewer SK. 2024. Processing time and precision of aging structures for Bighead Carp and Silver Carp in the lower Red River catchment in the southern Great Plains. *North American Journal of Fisheries Management*, 44: 973-986

Attachment B: Vilchez M, Dattilo J, Brewer SK. 2024. Length-weight relationships of native and non-native fishes in the lower Red River catchment, USA. *Journal of Applied Ichthyology*, 5 pages.

Attachment C: Ramsey PQ, Dattilo J, DeVries DR, Brewer SK. 2023. Evaluating the context dependency of large river nursery habitats for fishes in the lower Red River catchment. *River Research Applications*, 40: 761-779.

ARTICLE

Processing time and precision of aging structures for Bighead Carp and Silver Carp in the lower Red River catchment in the southern Great Plains

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Funding information

Arkansas Game and Fish Commission, Grant/Award Number: F20Ap1153; United States Fish and Wildlife Service, Grant/Award Number: FWS-FAC2021001740; Oklahoma Department of Wildlife Conservation, Grant/Award Number: F21AF00118; Texas Parks and Wildlife Department, Grant/Award Number: CA-0001012

Abstract

Objective: Population demographics of invasive species are commonly evaluated to better develop management actions that are useful for reducing their abundance or controlling the population. Bighead Carp *Hypophthalmichthys nobilis* and Silver Carp *H. molitrix* are emblematic invaders in the United States, where they continue to expand their range. There is currently no consensus about which hard structure from these species is best for age estimation. Our study objective was to compare the processing time and precision of five hard structures used for age estimation of both species.

Methods: We sampled fish in the lower Red River catchment of Oklahoma, Arkansas, and Texas during summer and autumn 2021–2022 and removed the lapillus otolith, left primary pectoral fin ray, postcleithrum, urohyal bone, and anterior-most pterygiophore of the dorsal fin from both Bighead and Silver carp. The structures ($n = 1204$) were either embedded in epoxy or thin-sectioned and mounted on slides. Two readers estimated the age of the fish by using each structure and came to a consensus. Processing time was recorded from the onset of laboratory processing to the termination of polishing the cross sections for age estimation.

Result: Processing of otoliths was comparable to or faster than processing of the other structures and resulted in the highest between-reader agreement. The lowest coefficients of variation in age estimation were represented using lapillus otoliths for Bighead Carp and postcleithra for Silver Carp. Our age bias plots indicated that all other structures underestimated age relative to the lapillus otoliths.

Conclusion: Our results indicated that using lapillus otoliths for age estimation of these species would have the highest between-reader agreement and would incur no additional laboratory processing time. However, validation is needed to assess whether lapillus otoliths correctly age these species. From a management perspective, use of this structure would facilitate improved population comparisons.

KEYWORDS

aging, demographics, invasive species, Red River

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INTRODUCTION

The usefulness of population and demographic models is related to the accuracy of the data that are used to construct them. Inaccurate age data—due to either overestimation or underestimation—lead to discrepancies between conclusions and actual population demographics, resulting in the possibility of erroneous management decisions (Bradford 1991; Reeves 2003; Koenigs et al. 2013). For example, maximizing recruitment and maximizing growth are often goals when managing sport fish populations, and managers can develop models from age data that relate annual fish production to various density-dependent or density-independent factors, such as forage availability and environmental parameters, respectively (Maceina 1997; Weisberg et al. 2010). Guided by model results, managers identify the conditions that lead to high fish production and attempt to mimic those conditions in subsequent years (Sammons and Bettoli 2000; Sammons et al. 2002; Sakaris 2013). The same concept can be applied to invasive species management, albeit typically with the goal of population reduction. In these cases, assigning appropriate ages is important to ensure that year-classes and the associated variables of interest are matched accordingly. Failure to assign appropriate ages could increase the chances of type I or type II errors and could result in management decisions that do not meet the desired goal. Understanding the sources of aging errors and minimizing those errors where possible are important aspects of developing a reliable age estimation protocol with the lack of a validated aging structure.

The two main concerns related to estimating fish ages are the accuracy of the estimate and the repeatability or precision of the age estimate. Age accuracy refers to how closely the estimated age for a given structure approximates the true age of the fish. An important step in minimizing accuracy error is to use a validated age estimation method. Age estimation methods can be validated for a species by confirming the frequency of the formation of growth increments on a structure and across the entire estimated ages by using known-age or marked fish, but this can be difficult for application to invasive species (Beamish and McFarlane 1983; Rugg et al. 2014; Anderson et al. 2023). For example, validation through oxytetracycline marking may be a viable method to validate a structure, but this requires the release of marked individuals, which is illegal for invasive Bighead Carp *Hypophthalmichthys nobilis* and Silver Carp *H. molitrix* in many states, and they would be difficult to recapture. Due to the rigorous methods required to carry out a validated age estimation study, exact ages are seldom known and the assigned ages are often termed “age estimates” rather than true ages. Precision estimates are used as a surrogate to assess the best structure

Impact Statement

We showed that among the five structures compared for age estimation in Bighead and Silver carp, the lapillus otolith had the highest between-reader agreement and is an efficient structure to use when estimating age for these species. Use of lapillus otoliths will improve the comparability of population demographics, but accuracy still needs to be assessed.

for estimating the age of fish when no structure has been validated; precision describes the reproducibility of age estimates for a given structure (Campana 2001). Common precision metrics include between-reader agreement (BRA) and the mean coefficient of variation (CV), with the highest BRA and lowest mean CV indicating the highest precision (Seibert and Phelps 2013). The precision of age estimation varies between fish species, individual structures, and ages (Maceina and Sammons 2006; Phelps et al. 2007; Edwards et al. 2011). Comparing precision between aging structures for a species is important for developing an aging protocol to be used in population assessments.

Since their detection in the 1970s, Bighead and Silver carp have proliferated and expanded their invasion throughout the central and southeastern United States. Bighead and Silver carp were first imported into the United States in the early 1970s by a commercial fish producer but were all eventually transferred to various government agencies and research institutions by the mid-1970s for production and water quality control studies (Freeze and Henderson 1982; Kelly et al. 2011). Bighead and Silver carp likely escaped from state hatchery facilities; by the 1990s, populations were established in the Mississippi River basin (Kolar et al. 2007; Kelly et al. 2011). Their range has now expanded to many other major river basins, and they have been detected in 23 states (Kolar et al. 2005). With the spread of Bighead and Silver carp, management agencies have transitioned to efforts to contain and reduce population numbers (Conover et al. 2007; Irons et al. 2011; Chick et al. 2020).

As Bighead and Silver carp continue to expand their range within the United States, the estimation of ages for sampled fish has become increasingly common. However, no standard structure has been used for these estimates, likely due to the difficulty in estimating age for these species. Several structures have been used for estimating the ages of Bighead and Silver carp, but precision estimates or structure comparisons have received limited evaluations or have not been reported (Johal et al. 2001; Hayer

et al. 2014; Kamilov 2014; Kurbanov and Kamilov 2015). To our knowledge, for each species there have only been two instances in which aging precision or structural comparisons were evaluated and reported. Nuevo et al. (2004) indicated that sagittal otoliths and cleithrum bones from Bighead Carp did not contain discernible annuli, and they reported low reader agreement on age estimates produced via scales (BRA = 19%; $n = 57$) and pectoral fin rays (BRA = 23%; $n = 98$). Scales are commonly used to age Bighead and Silver carp in their native range (Fange et al. 2022); however, age estimates that were produced using scales tended to be lower than fin ray age estimates when fin ray estimates were high, but they tended to be higher than fin ray age estimates when fin ray estimates were low (estimated ages were 1–5 years). Schrank and Guy (2002) reported that scales and dorsal fin rays were more precise than pectoral fin rays for age estimation of Bighead Carp, although precision estimates and methods were not indicated. The age estimates resulting from enumerating annuli on scales were similar to estimates obtained via reading dorsal fin rays for 3–5-year-old fish (estimated age = 3–7 years; $n = 151$). Williamson and Garvey (2005) used pectoral fin rays to estimate the ages of 145 Silver Carp and reported a BRA value of 60% and a between-reader CV of 13% (estimated age = 0–5 years). In the most comprehensive structure comparison analyses, Seibert and Phelps (2013) evaluated the scales, opercles, pectoral fin rays, vertebrae, postcleithra, asteriscus otoliths, and lapillus otoliths of Silver Carp. They found that the lapillus otoliths resulted in the highest estimated age (2–10 year; $n = 120$) and had the highest between-reader precision, with a BRA of 76% and a CV of 4.2%, followed by postcleithra (BRA = 58%; CV = 9.7%) and pectoral fin rays (BRA = 57%; CV = 10.2%). Seibert and Phelps (2013) also compared age estimates from lapillus otoliths with estimates produced from pectoral fin rays, vertebrae, and postcleithra and reported 78, 53, and 49% agreement, respectively, between structures.

Effective management of Bighead and Silver carp is important and would benefit from an understanding of the current population status to predict future trends. In its simplest form, a fish population can be described by estimated abundance or density. To better manage a population, having information on the rate functions (i.e., mortality, recruitment, and growth) that most influence fish abundance is important (Ricker 1975; Allen and Hightower 2010). Age data are key to describing population-level metrics and developing models that help to inform management decisions (Campana 2001). Estimates of demographic rates are useful for population monitoring and for assessing whether a population is growing or declining (Quist and Isermann 2017) and would be a fundamental component in understanding

the current and future status of Bighead and Silver Carp populations.

Because of the efforts to both understand newly invasive Bighead and Silver Carp populations and reduce their abundance, our objective was to evaluate the precision of several structures to guide future age estimation protocols for these species. We did not have access to known-age fish, so we focused on precision rather than accuracy. We built on the study of Seibert and Phelps (2013) by including Bighead Carp in our analyses and by incorporating an additional structure (i.e., pterygiophore) in our evaluation. The anterior-most pterygiophore of the dorsal fin was included because several agency biologists have mentioned considering the structure for use in aging. Bighead and Silver carp were first documented in the lower Red River, Oklahoma, in 2012; since their detection, little research has been conducted regarding population demographics, habitat use, or expansion through the catchment (Patton and Tacket 2012). Furthermore, the fish that we examined were from this relatively recent invasion, representing some of the southernmost populations of Bighead and Silver carp, for which a structural comparison has not been completed. Additionally, the physicochemical conditions of the lower Red River catchment are quite different from those in the Mississippi River basin, where many similar studies have been conducted.

METHODS

Study area

Our study was conducted in the lower Red River catchment of Oklahoma, Arkansas, and Texas, where Bighead and Silver carp have recently expanded (Figure 1). The Red River catchment is primarily located in the southern Great Plains and is emblematic of relatively extreme physicochemical conditions that are rather different from those of other major catchments where Bighead and Silver carp are established. The catchment is characterized by relatively high salinity due to upper basin salt springs, salt seeps, and brine from oil fields (Laughlin and Lacewell 1981; Hargrave and Taylor 2010). It is a low-gradient river characterized by a large floodplain, with oxbow lakes and some backwater connections (Benke et al. 2005). Water temperatures can reach 39°C due to high air temperatures; the lack of riparian cover, which is caused by meandering and braided alluvial channels (Benke et al. 2005); and limited groundwater influence due to droughts and groundwater pumping (Krueger et al. 2017; Smith et al. 2021). The upper river was impounded by Dennison Dam in 1944. Our study occurred on the lower main-stem Red River from Dennison Dam

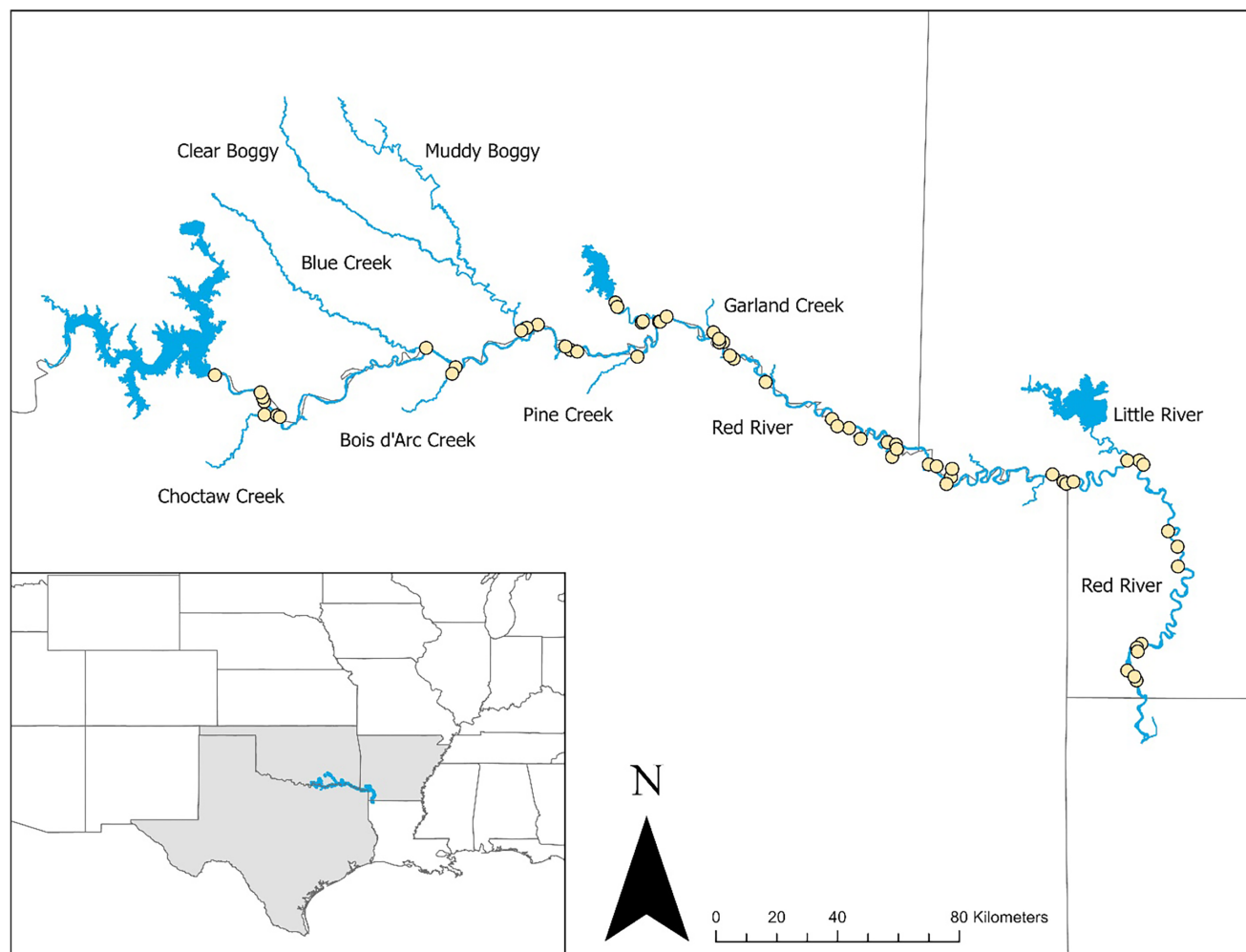


FIGURE 1 Sites (yellow circles) that were sampled for Bighead and Silver carp in the lower Red River catchment of Oklahoma, Texas, and Arkansas from May 2021 through October 2022.

downriver to the Arkansas–Louisiana border (~620 rkm), along with several major tributaries in Oklahoma and Texas. Bighead and Silver carp were first documented in the lower Red River in 2012. Limited research has been conducted since their detection, including information on population demographics, habitat use, and distribution within the catchment (Patton and Tacket 2012).

Fish sampling

We collected Bighead and Silver carp during the summer and autumn of 2021–2022 by using a variety of sampling gears. Fish were sampled at sites located throughout the Red River catchment downriver of Lake Texoma, Oklahoma, to the Arkansas–Louisiana border. We sampled fish by using a combination of gill nets (54.8 m long [main-stem Red River] or 30.5 m long [tributaries] $\times$ 3.65 m tall, with 8.90-, 10.16-, and 10.80-cm bar mesh panels) and standardized boat electrofishing. Boat electrofishing was

conducted using an 80-A shocking unit (pulsed DC) with expandable anode arrays (Midwest Lakes Electrofishing Systems). We used standard American Fisheries Society electrofishing settings based on conductivity, adjusting the settings to reach a target power (Miranda 2009). Water conductivity in the tributaries was much lower than that in the main-stem Red River. We used a high-conductivity Infinity HC-80 shock box (Midwest Lakes Electrofishing Systems) with voltage set to high range (pulsed-DC current, >300 V, 60 Hz) for the tributaries and low range (pulsed-DC current, <300 V, 60 Hz) for the main-stem Red River. Beginning at the upstream end of the site, we slowly motored the boat downstream in a cloverleaf pattern with electrical current applied for 10 s with 5-s “off-peddle” intervals to increase the effectiveness of capturing Bighead and Silver carp and to drive fish into the nets and shoreline (Bouska et al. 2017). We removed the lapillus otoliths, the left postcleithrum, the urohyal bone, the left primary pectoral fin ray, and the anterior-most pterygiophore of the dorsal fin from a subset of captured Bighead and Silver carp.

Structure removal and processing

Lapillus otoliths, located at the posterior of the skull, were removed, cleaned, mounted to slides, and polished (see Appendix Figures A.1–A.5). We used a hacksaw to cut through the top of the skull in line with the juncture of the pre-opercle and the opercle. Both lapillus otoliths were removed, cleaned of all tissue, and placed into coin envelopes marked with the individual fish identification number. In the laboratory, one otolith from each fish was immersed in epoxy resin (Wes System) and allowed to harden for 24 h. We sectioned each otolith with a low-speed saw (IsoMet low-speed precision cutter; Buehler), and two 0.35–0.50-mm cross sections were removed from the center of the otolith, ensuring the inclusion of the nucleus (Figures A.1–A.5). We then polished each cross section with 3- μ m diamond lapping film (20.32-cm diameter, plain backing; Electron Microscopy Sciences) and mounted it onto a slide using thermoplastic cement (Hugh Courtright and Co).

The other four structures were removed in the field and frozen at the end of every sampling day. The left postcleithrum, situated near the cleithrum, was removed from the pectoral girdle by making an incision posterior to the pectoral fin ray (Figures A.1–A.5; Johal et al. 2000a). The urohyal bone, located in the lower jaw, was removed by making an incision at the anterior juncture with the ventral hypohyal bones and the dorsal juncture with the first basibranchial bone (Figures A.1–A.5; Johal et al. 2000b). The left primary pectoral fin ray was removed in most instances; however, if the left pectoral fin ray was damaged or missing, then we removed the right pectoral fin ray. The anterior-most pterygiophore of the dorsal fin, located between the primary dorsal fin ray and vertebrae, was removed using a hacksaw. We made two vertical cuts: one cut behind the first dorsal fin ray and another about 8–10 cm anterior to the first cut. We made an incision through the left side of the fish to the vertebrae between the two vertical cuts and removed the pterygiophore (Figures A.1–A.5). The urohyal, postcleithrum, fin ray, and pterygiophore were placed in a 3.79-L (1-gal) bag labeled with the individual fish identification number and were frozen.

In the laboratory, we cleaned, sectioned, and mounted the left primary pectoral fin ray, postcleithrum, urohyal bone, and anterior-most pterygiophore of the dorsal fin. The fin rays were allowed to dry for 24–48 h, and any residual flesh was removed. We removed one 0.5–0.6-mm cross section at the juncture of the knuckle and the ray. The postcleithrum, urohyal bone, and pterygiophore were immersed in 60–70°C water for about 3–5 min and then were cleaned and allowed to air-dry for 24 h (Johal et al. 2000b). We took two 0.5–0.6-mm cross sections from each postcleithrum and urohyal bone according to the

methods described by Johal et al. (2000a, 2000b), and a single cross section of the pterygiophore at the widest portion of the x-axis was obtained by using a low-speed saw (IsoMet). All cross sections were polished using 3- μ m diamond lapping paper and mounted on a slide using thermoplastic cement.

Processing time was calculated as the sum of all labor conducted for each individual structure throughout laboratory processing. Time was recorded for otoliths during the immersion in epoxy resin, sectioning, and polishing of the cross sections. For fin rays, time was recorded during the removal of any excess tissue, sectioning, and polishing of the cross section. For all other structures, the time was recorded for immersion in the water bath, removal of all flesh, sectioning, and polishing of the cross sections. Processing of all structures was completed by experienced laboratory members, and each person processed the same quantity of each structure to reduce bias between individuals.

Age estimation

Two readers separately enumerated the annuli of the sectioned structures by using a consensus approach if necessary. Readers used a compound microscope with transmitted light to enumerate annuli. An annulus was defined as a pair of translucent and opaque bands that continued uninterrupted around the nucleus (Dzul et al. 2012). The edge was counted as an annulus for fish captured prior to April 1 because an annulus may be created during the spawning season (Minard and Dye 1998; Ericksen 1999). To avoid reader bias, the readers were given no prior knowledge of the fish's length, weight, or age. Readers compared ages for discrepancies. If there was disagreement on the estimated age, then the readers discussed how they derived the age and reached a consensus.

Data analyses

We compared structure processing times, BRA values (i.e., compared to the exact year of age), and CVs associated with the ages estimated for Bighead and Silver carp. We also assessed age bias of each structure compared to ages derived from the lapillus otolith because Seibert and Phelps (2013) found that other structures underestimated age compared to otoliths. Moreover, otoliths are commonly used as the standard for comparison (e.g., Campana 2001; Maceina et al. 2007; Phelps et al. 2007; Seibert and Phelps 2013). We quantified the processing time for each structure and tested for significant differences using structures from Bighead and Silver carp

combined. We conducted a one-way analysis of variance ($\alpha=0.05$). Tukey's honestly significant difference (HSD) post hoc test was used to determine where significant differences between the processing times of structures occurred (Damon and Harvey 1987). We calculated the BRA and mean CV for each structure to determine agreement and precision. We assumed that a higher BRA value indicated that annuli were easily discernible and resulted in more consistent age estimates between readers (Seibert and Phelps 2013). The CV describes the variation between age estimates. A low CV indicated that in instances when readers did not agree on the age, the differences in age estimates were relatively low (when compared to a higher CV). Finally, we constructed age bias plots to compare the relative bias of each structure in comparison with lapillus otoliths (Campana et al. 1995). This relative bias is informative because structures may have a high BRA or CV but may greatly underestimate age compared to other structures with similar BRA or CV values.

RESULTS

We compared the processing time of the five structures used for aging both Bighead and Silver carp combined ($n=1204$ structures). There was a significant difference in processing time between the structures ($F_{4, 1199}=33.87$, $p<0.01$; Figure 2). Results of Tukey's HSD test for

multiple comparisons indicated that the processing time was significantly lower for fin rays compared to pterygiophores ($p<0.01$; 95% confidence interval [CI] = 319.83–649.27 s), lower for postcleithra versus fin rays ($p=0.02$; 95% CI=17.83–334.59 s), lower for otoliths compared to pterygiophores ($p<0.01$; 95% CI=374.61–779.60 s), lower for postcleithra than for pterygiophores ($p<0.01$; 95% CI=496.04–825.48 s), and lower for urohyal bones versus pterygiophores ($p=0.025$; 95% CI=22.69–339.21 s). There was no significant difference in processing time between otoliths and postcleithra ($p=0.78$), otoliths and fin rays ($p=0.71$), otoliths and urohyal bones ($p=0.66$), or urohyal bones and fin rays ($p=0.99$).

We compared age estimates, BRA values, and CVs associated with Bighead Carp structures ($n=235$). We estimated that Bighead Carp were 3–17 years old based on examination of lapillus otoliths, but fish were estimated as younger when the other structures were used (maximum age=9–11 years; Table 1). For Bighead Carp, lapillus otoliths had the highest BRA (70%), whereas other structures had BRA values of 52–68% (Table 1). The mean CVs associated with age estimates of Bighead Carp were lowest for lapillus otoliths and highest for pterygiophores (Table 1).

Age estimates, BRA values, and CVs associated with Silver Carp structures ($n=959$) were also compared. We estimated that Silver Carp were 3–14 years old by using lapillus otoliths, but the use of other structures indicated

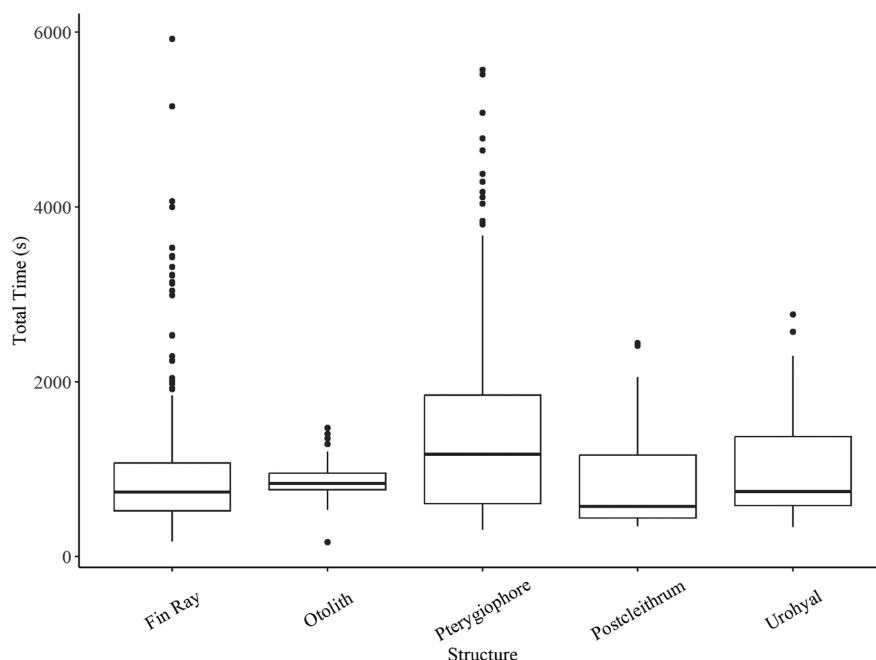


FIGURE 2 Processing time (s) for five aging structures (fin rays [$n=278$], otoliths [$n=132$], pterygiophores [$n=239$], postcleithra [$n=278$], and urohyal bones [$n=267$]) for Bighead and Silver carp collected in the lower Red River catchment from May 2021 through October 2022. The line within the box plot denotes the median, the upper and lower ends of the box plots are the upper and lower quartiles, the ends of the whiskers are the minimum and maximum values, and the black circles are the outliers.

TABLE 1 Between-reader agreement (BRA), mean coefficient of variation (CV), and the average (mean), range (minimum [min] and maximum [max]), and variation (standard deviation [SD]) of the consensus estimated age (years) for Bighead and Silver carp collected from May 2021 through October 2022 in the lower Red River catchment of Oklahoma, Texas, and Arkansas. The sample size for aging (*N*) is provided. A subset of these structures was used to estimate processing time (see Figure 2).

Structure	<i>N</i>	BRA (%)	Mean CV (%)	Mean	Min	Max	SD
Bighead Carp							
Otolith	82	70	4.7	10.05	3	17	2.96
Postcleithrum	84	60	5.95	5.89	3	11	1.62
Fin ray	84	68	6.43	6.24	3	10	1.79
Urohyal bone	85	62	6.27	5.62	3	11	1.79
Pterygiophore	60	52	7.31	5.03	2	9	1.57
Silver Carp							
Otolith	254	77	3.6	6.67	3	14	2.54
Postcleithrum	251	76	3.31	4.6	3	10	1.35
Fin ray	256	73	3.88	4.54	3	12	1.79
Urohyal bone	252	68	5.98	4.55	3	11	1.73
Pterygiophore	194	63	5.89	3.38	2	7	0.99

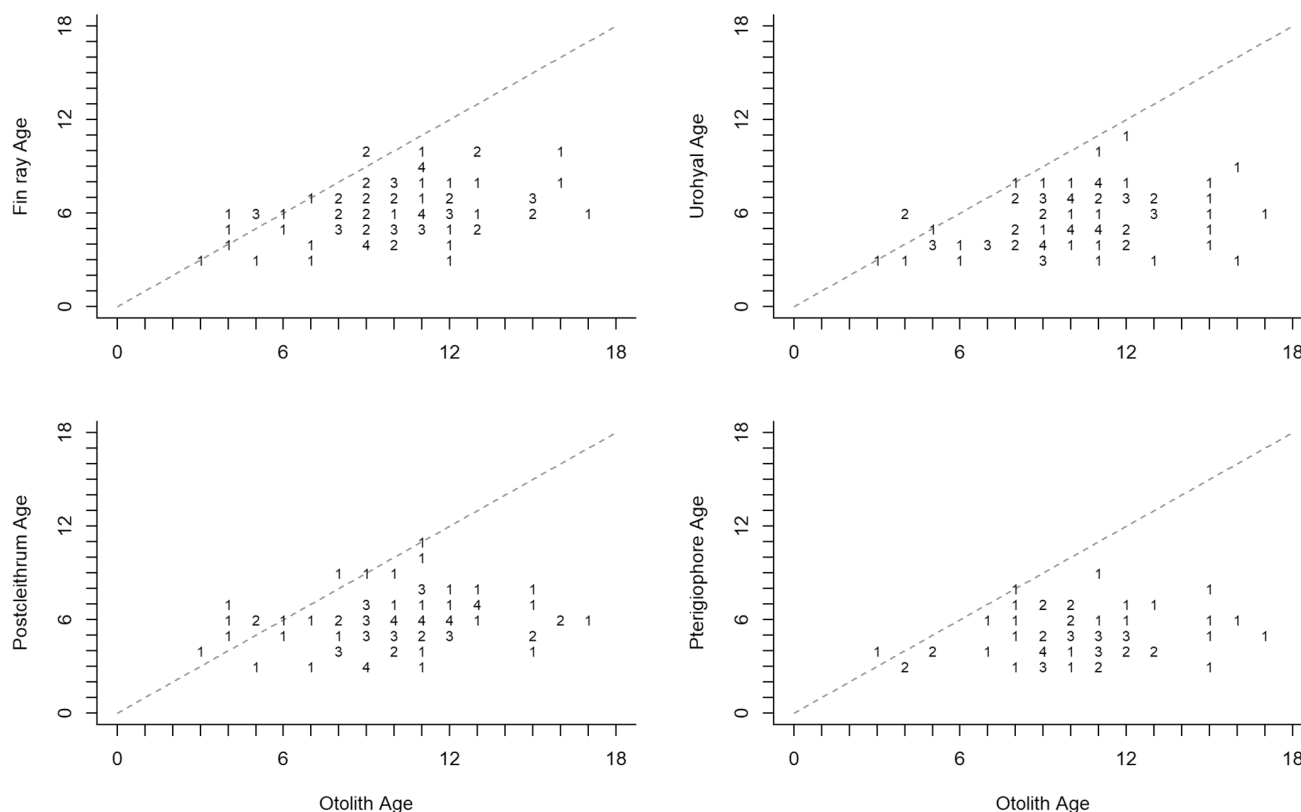


FIGURE 3 Age bias plots for Bighead Carp, comparing age estimates from fin rays, urohyal bones, postcleithra, or pterygiophores (y-axis) to age estimates from lapillus otoliths (x-axis). The dashed line represents perfect agreement between structures. The quantity of age estimates is indicated by the numbers. Numbers above the dashed line indicate that the structure overestimated age compared to lapillus otoliths, and numbers below the dashed line indicate that the structure underestimated age compared to lapillus otoliths.

that Silver Carp were younger. Silver Carp ages were estimated to be 3–10 years based on examination of postcleithra, 3–11 years based on urohyal bones, 3–12 years based on fin rays, and 2–7 years based on pterygiophores

(Table 1). Lapillus otoliths resulted in the highest BRA (77%) for estimating the age of Silver Carp. Between-reader agreement for other structures ranged from 63% to 76% (Table 1). The highest mean CV for Silver Carp age

estimation was associated with the urohyal bone, and the lowest mean CV was associated with the postcleithrum (Table 1).

Our age bias plots indicated either that (1) the fin ray, postcleithrum, urohyal bone, and pterygiophore underestimated carp age or (2) the otoliths overestimated age. Age bias plots indicated that all structures underestimated age relative to lapillus otoliths for most age-classes of both species (Figures 3 and 4). When lapillus otoliths were used to estimate age, Silver and Bighead carp older than age 4 were consistently 2–5 years older relative to estimates made using the other structures. These differences became exaggerated as the estimated age increased.

DISCUSSION

Our results indicated that the use of lapillus otoliths for estimating age and monitoring populations of Bighead Carp would be beneficial because they had the highest BRA value and lowest mean CV, reflected older ages compared to the other structures, and had a processing time that was comparable to or faster than those of the other

structures. We obtained similar precision estimates (i.e., BRA and CV) for the otoliths and postcleithra of Silver Carp. However, the use of lapillus otoliths for estimating age and monitoring populations of Silver Carp would be beneficial given the bias to younger age estimates for the other structures and given the similar or more efficient processing time for lapillus otoliths. This recommendation could change if a structure is verified for age estimation of these species. Values of BRA for lapillus otoliths (Silver Carp: BRA=77%; Bighead Carp: BRA=70%) were relatively low compared to those reported for other species, such as Walleye *Sander vitreus* (BRA=98%), Largemouth Bass *Micropterus nigricans* (BRA=91%), Smallmouth Bass *M. dolomieu* (BRA=94%), Yellow Perch *Perca flavescens* (BRA=98%), and Brown Bullhead *Ameiurus nebulosus* (BRA=92%; Isermann et al. 2003; Maceina and Sammons 2006). These results highlight the difficulty in estimating age for these species. Longer-lived fishes are inherently more difficult to age compared to fishes with shorter lifespans due to crowding of annuli, especially for fish in warmwater systems when growth is more consistent (Quist and Isermann 2017). For example, Dunton et al. (2016) found that the BRA for long-lived, coolwater Atlantic Sturgeon *Acipenser oxyrinchus* was 63% when

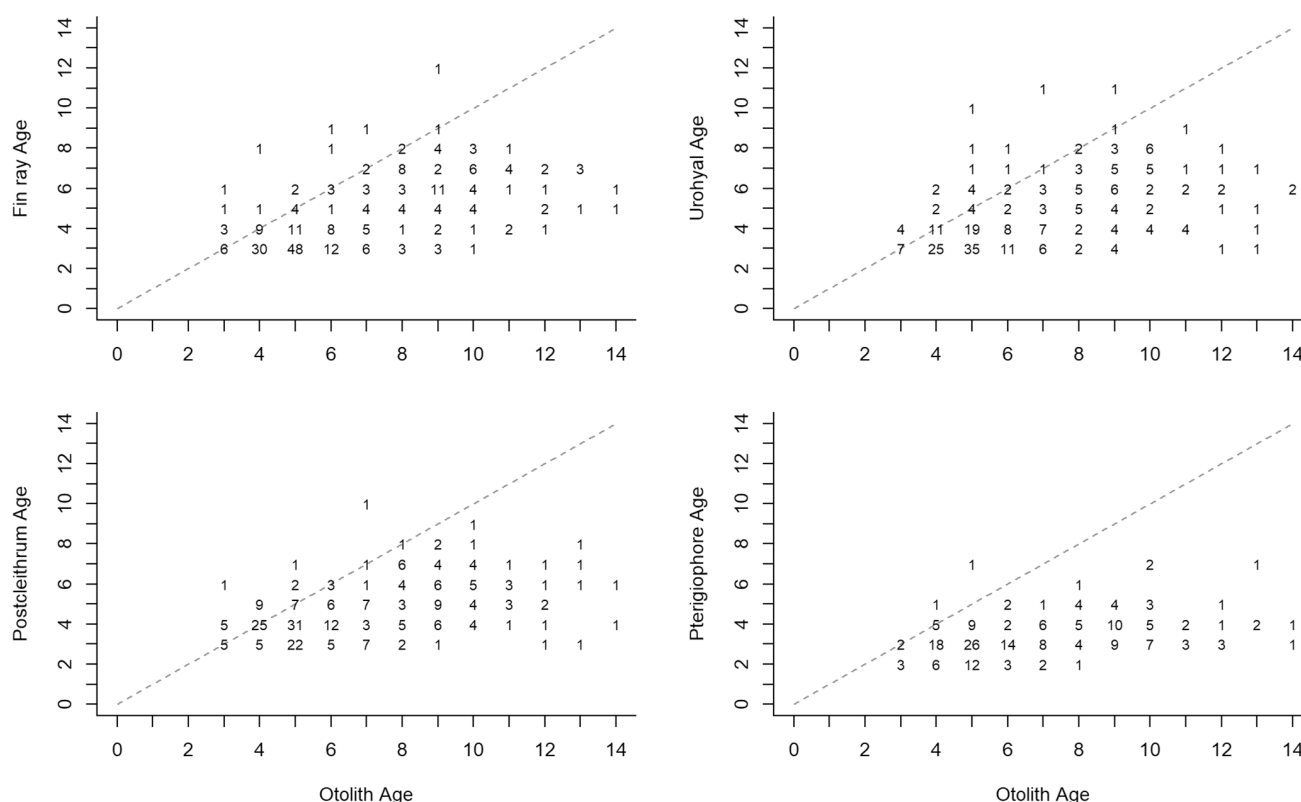


FIGURE 4 Age bias plots for Silver Carp, comparing age estimates from fin rays, urohyal bones, postcleithra, or pterygiophores (y-axis) to age estimates from lapillus otoliths (x-axis). The dashed line represents perfect agreement between structures. The quantity of age estimates is indicated by the numbers. Numbers above the dashed line indicate that the structure overestimated age compared to lapillus otoliths, and numbers below the dashed line indicate that the structure underestimated age compared to lapillus otoliths.

using fin rays, and Labay et al. (2011) found that the BRA for long-lived, warmwater Blue Sucker *Cycleptus elongatus* was 50% for fin rays.

Our results support the notion that similar aging structures may not be the most appropriate for taxonomically similar species. Our results contrast with those of Seibert and Phelps (2013), as we found that using lapillus otoliths for age estimation of Silver Carp resulted in precision similar to that obtained with postcleithra. However, the patterns observed when estimating the age of Bighead Carp were similar to those described for Silver Carp by Seibert and Phelps (2013). Results on aging structure precision for a given species do not always apply to a similar species. For example, the asteriscus and lapillus otoliths have been validated for age estimation of Bigmouth Buffalo *Ictiobus cyprinellus* (Lackmann et al. 2021), whereas only lapillus otoliths have been used to age Smallmouth Buffalo *I. bubalus* and Black Buffalo *I. niger* (Paukert and Long 1999; Love et al. 2019). Although it may be easier to use other structures (Schrank and Guy 2002) to estimate the age of Bighead Carp, the resulting age estimates would be underestimated compared to those obtained from otoliths (i.e., not to be confused with accurate). Similar results have been found with other species, including saugeye (Sauger *S. canadensis* × Walleye), catostomids, and cyprinids (Quist et al. 2007; Koch et al. 2018). Agencies using lapillus otoliths instead of other commonly used structures (i.e., fin rays) or structures with similar precision (i.e., postcleithra) would not incur additional laboratory time.

Ages of Bighead and Silver carp have been estimated using a variety of structures (i.e., fin rays, scales, postcleithra, urohyal bones, and otoliths), and the use of each structure to estimate age has trade-offs. Although fin rays are easily collected, they may underestimate age for Silver Carp due to the erosion of the central lumen (Seibert and Phelps 2013). Scales result in underestimated ages of Bighead and Silver carp due to crowding of the annuli and nondistinct annuli (Sikstrom 1983; Johal et al. 2000b; Seibert and Phelps 2013). Scales are highly variable in size and shape, and environmental stressors can cause scales to be reabsorbed to account for fluctuations in fish mass. This reabsorption causes annuli to be laid down and removed at variable rates, resulting in biased age estimates (Khan et al. 2011). The postcleithra and urohyal bones can be sectioned to determine age in Silver Carp (Johal et al. 2000b). However, living structures such as the postcleithra and urohyal bones have been observed to behave irregularly with deficiencies in phosphorus. Typically acquired through diet, the lack of sufficient phosphorus can lead to bone resorption and decreases in bone formation and mineralization (Chinkichi and Hiroshi 1976; Lall and Lewis-McCrea 2007). When compared to other structures, using these two bones has resulted in lower BRA

than lapillus otoliths (Seibert and Phelps 2013). Although various structures have been used to estimate the age of Bighead and Silver carp, a consensus should be reached regarding the use of a single structure to enhance data sharing efforts and comparability. The use of different structures can potentially change estimates of various population demographics and hinder management efforts.

When comparing population estimates for Bighead and Silver carp, it is important to standardize aging structures to avoid biased conclusions. Although different calcified structures have been used to produce viable age estimates for Bighead and Silver carp (Williamson and Garvey 2005), keeping aging structures consistent across all populations and studies will minimize bias. Comparisons between a population for which age was estimated with fin rays versus a population for which age was estimated with otoliths can be difficult because the differences in estimates may be a function of the difference between structures and not the underlying population demographics. For example, comparing two studies of Bighead and Silver carp collected from the Mississippi River, Williamson and Garvey (2005) found that pectoral fin rays had a BRA value of 60% whereas Seibert and Phelps (2013) found lapillus otoliths to have a BRA of 76%. An attempt to compare these studies would produce biased results, as the variance of reader agreement would be difficult to reconcile.

Using structures that often underestimate the age of fish (i.e., fin rays, scales, postcleithra, and urohyal bones) is an important consideration when comparing demographics (i.e., mortality, growth, and recruitment) with those of populations that have been aged using otoliths. For example, an underestimate of fish age can increase the slope of a catch curve due to the resulting increase in the proportion of younger fish, leading to a higher perceived mortality rate (Prull et al. 2023). Additionally, if fish are assigned younger ages, then there can be a shift in the inflection point of growth functions, leading to an increased population growth coefficient (i.e., k) or theoretical maximum length (i.e., L_{∞} ; Prince and Pulos 1983). Furthermore, underestimating the age of fish may decrease the probability of missing year-classes in a given sample. This can lead to the illusion of more stable recruitment when using a method such as the recruitment variability index (Isermann et al. 2002). By using a standardized method for aging Bighead and Silver carp, more accurate analyses can be conducted across populations—barring major concerns related to accuracy.

As Silver and Bighead carp continue to expand their invasion front, identifying a standard aging structure is pertinent to ensure comparable age estimation and assessment of these species in different physicochemical conditions. We emphasize the need for a study to validate the

accuracy of otoliths for Bighead and Silver carp. However, we acknowledge that this would be challenging given the risk of additional introductions and the difficulty in recapturing fish or holding them for long periods of time. In the meantime, use of lapillus otoliths to produce consistent results and allow for comparability among Bighead and Silver Carp populations would be prudent.

ACKNOWLEDGMENTS

This research is a contribution of the Alabama Cooperative Fish and Wildlife Research Unit (U.S. Geological Survey, Alabama Department of Conservation and Natural Resources, Auburn University, and Wildlife Management Institute). The Oklahoma Department of Wildlife Conservation, Arkansas Game and Fish Commission, and Texas Parks and Wildlife provided project funding. We thank Trevor Bannister, Tyler Murphy, Kyle Rempe, Eli Wilson, Jordan Ramey, Paul Ramsey, and Aiden Maddux for technical assistance. Monica McGarity, Matt Horton, and Jim Burroughs provided helpful feedback during the project. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government. This study was performed under the auspices of Auburn University Protocol 2020-380C.

CONFLICT OF INTEREST STATEMENT

The authors have no conflicts of interest to declare.

DATA AVAILABILITY STATEMENT

Data may be requested from Jim Burroughs (jim.burroughs@odwc.ok.gov).

ETHICS STATEMENT

This research meets the ethical and legal requirements of sampling within Texas, Arkansas, and Oklahoma. Sampling was performed with state collection permits from the Arkansas Game and Fish Commission (031620231) and Texas Parks and Wildlife (SPR-1020-146) and with a letter of authorization to sample fishes from the Oklahoma Department of Wildlife Conservation. This study was performed under the auspices of Auburn University's Animal Care and Use Protocol 2020-380C.

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APPENDIX A

AGE ESTIMATION STRUCTURES

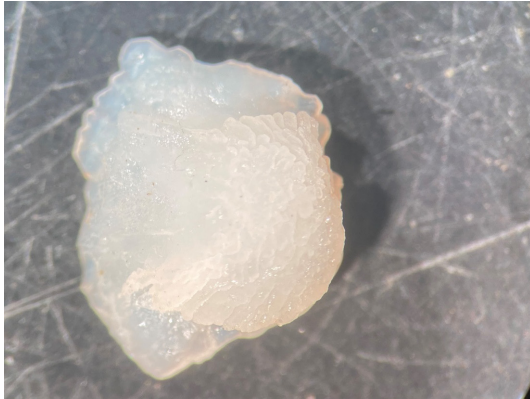


FIGURE A.1 A lapillus otolith from a Silver Carp that was sampled in the lower Red River catchment of Oklahoma, Texas, and Arkansas in November 2021.

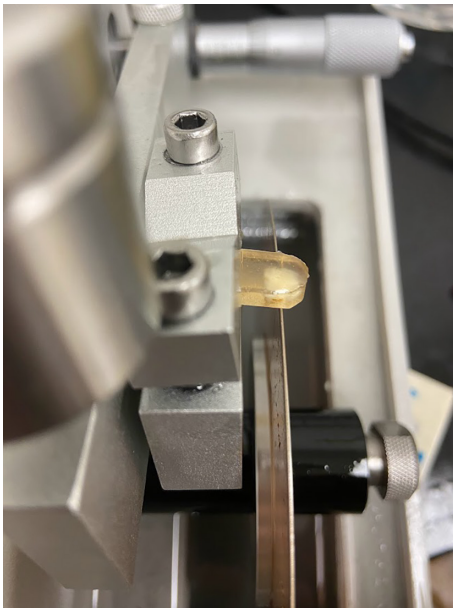


FIGURE A.2 A lapillus otolith from a Silver Carp that was sampled in the lower Red River catchment in November 2021. The otolith was embedded in epoxy just prior to sectioning.



FIGURE A.3 The left postcleithrum from a Silver Carp sampled in the lower Red River catchment in November 2021.

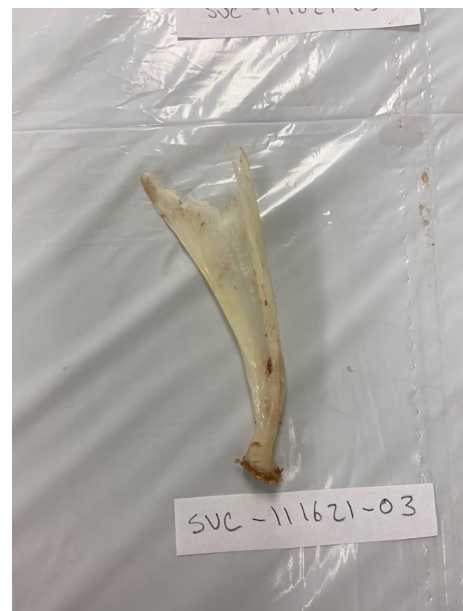


FIGURE A.4 The urohyal bone from a Silver Carp sampled in the lower Red River catchment in November 2021.



FIGURE A.5 Pterygiophores from a Silver Carp (left) and a Bighead Carp (right) sampled in the lower Red River catchment in November 2021.

Research Article

Length-Weight Relationships of Native and Non-Native Fishes in the Lower Red River Catchment, USA

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Received 28 May 2024; Accepted 11 September 2024

Academic Editor: Pronob Das

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Length-weight relationships are useful for stock assessments and modeling alternative conservation and management strategies for both native and non-native fishes. We developed length-weight relationships for 18 native and non-native riverine fishes in the lower Red River catchment. Fishes were sampled in the summer and autumn seasons between May 2021 and March 2024 via electrofishing and gill nets. Measurements for each specimen consisted of total length (mm) and weight (g). We provide L-W relationships for 14 native fishes consisting of 5 families (Lepisosteidae, Catostomidae, Ictaluridae, Sciaenidae, and Polyodontidae) and 4 non-native species belonging to the family Cyprinidae. We collected 6,845 individuals ranging from 67 alligator gar to 1,848 smallmouth buffalo. All the L-W relationships were significant ($p < 0.05$), and the majority (72% of species) of relationships between length and weight had r^2 values > 0.70 . Our findings provide insight into the L-W relationships of riverine fishes and can be useful for modeling alternatives targeted at native fishes of recreational value and the removal efforts of non-native fishes.

1. Introduction

Length-weight (L-W) relationships are important for fisheries management including their use in stock assessments, production modeling, and biomass estimations [1]. L-W relationships are useful for a variety of conservation and management applications ranging from calculating indices related to fish condition to assessing growth patterns in relation to environmental factors [2]. L-W relationships are insightful for fish-population dynamics across spatial and temporal scales. In this study, we modeled L-W relationships for 18 native and non-native riverine fishes of the lower Red River catchment, USA, to describe the biometric relationships among varying large-bodied fishes of the Red River catchment located in the Great Plains ecoregion. The Red River basin is characterized by extended periods of floods and droughts [3]. These highly dynamic shifts in flow regime are relatively unique to the lower Red River catchment and variation of natural conditions that native assemblages in the

region have evolved to cope with but may become more difficult with the introduction and presence of non-native species such as carp. Although much is known about the effects of both Bighead and Silver Carp where they have been introduced and become established over time (i.e., competition of planktivorous resources with native riverine fishes), little information is available in the Red River, where carp are not yet well established [4, 5]. By investigating the L-W relationships among both native and non-native riverine fishes, insight into the population demographics of the lower Red River catchment assemblage can be further assessed to produce effective management strategies that target the removal or control of non-native species and the management of native species.

2. Materials and Methods

We sampled 6,845 fishes comprising 14 native and four non-native species in reaches (1.5–2 rkm) of the lower Red River

catchment in Arkansas (above the AR-LA state line), Oklahoma, and Texas. We sampled using a combination of electrofishing and gillnet from May 2021 until March 2024. We set three experimental sinking gillnets (54.8 m long in the mainstem Red River and 30.5 m long in the tributaries with 8.9, 10.16, and 10.8 cm bar-length mesh panels) and allowed them to soak for approximately 6 h. After net placement, we electrofished using standard settings (American Fisheries Society) with an 80 amp high-conductivity electrofishing unit (Midwest Lakes Electrofishing Systems, Polo, Missouri). Beginning at the upstream end of the site, we electrofished downstream in a cloverleaf pattern throughout each reach. Total length (mm, ± 1 mm) and weight (g, ± 10 g) were recorded for fishes sampled from both gears.

Total length and weight were log (base 10)-transformed, and linear regression was used to determine length-weight (L-W) relationships using the following equation:

$$\log_{10} W = \log_{10} a + b(\log_{10} TL), \quad (1)$$

where W is the weight (g), TL is the total length (mm), a is the intercept, and b is the slope of the regression. All regressions were developed using Program R (version 4.2.2). We also examined the L-W relationships between female and male Bighead Carp as that was the only species where we were confident in sex determination (i.e., related to an ongoing tagging and age and growth study).

3. Results

We sampled native and non-native riverine fishes from the Red River. Our sample size varied based on relative abundance, ranging from 67 alligator gar to 1,848 smallmouth buffalo (Table 1). Total length and weight varied but were limited to the size of fishes that were recruited to our sampling gears. We report the 95% confidence interval for a and b . Our estimated values of b reflect an acceptable range between 2.5 and 3.5, indicating adequate sample sizes and distribution of species' proportional growth [2]. However, paddlefish (2.89), shortnose gar (3.88), and grass carp (2.80) a and b values did not fall within this acceptable range. All the L-W relationships were significant ($p < 0.05$), and most of the relationships (72% of species) between length and weight had coefficients of determinations > 0.70 (Table 1). The species with the strongest L-W relationships ($r^2 > 0.90$) were blue catfish, channel catfish, and flathead catfish, whereas the species with the weakest L-W relationships ($r^2 < 0.60$) were paddlefish, shortnose gar, and bighead carp.

We examined the L-W relationships between female and male bighead carp ($r^2 = 0.58$) and silver carp ($r^2 = 0.79$) to investigate the variability in our estimates of L-W relationships. We found that although young female bighead carp were initially smaller than their male counterparts, as they grew older, they became much bigger than male bighead carp (Figure 1). However, female silver carp recruited to our sample gears were always bigger than their male counterparts (Figure 2).

4. Discussion

Our results indicate that the L-W relationships of native and non-native fishes varied considerably. This variation might be attributed to smaller sample sizes or error in measurements for species with narrow length ranges or associated sex classes [6]. Variation in b may relate to biological, environmental, and sampling effects (e.g., maturity, seasonality, and size ranges of species captured [7–9]). Most of our L-W relationship equations were similar to those previously documented by Hall and Jenkins [10], Bur [11], Brown and Murphy [12], Bister et al. [13], Neely et al. [14], Stewart et al. [15], Wanner and Klumb [16], Snow et al. [17]; however, the species L-W relationships included in Bister et al. [13] are only those with an $r^2 > 0.90$. The L-W relationships we report for black buffalo and shortnose gar, however, were different than previous reports. Our findings showed that L-W relationships between taxonomically related species in the family Lepisosteidae consisting of alligator gar, longnose gar, shortnose gar, and spotted gar had varied a values despite moderate differences in sampling sizes and b values which may be related to variation in both body shape and condition [2, 18]. Though this trend was not noted in other taxonomically related species (e.g., Ictaluridae and Cyprinidae), this variation might be worth exploring in understudied fishes such as shortnose gar.

Though our findings among bighead and silver carp L-W relationships were similar to those within their native and introduced range [19, 20], we did observe variability in our L-W relationships for both species. Therefore, we further investigated the differences between female and male carp species. Generally, we found that both female bighead and silver carp in the lower Red River catchment were larger compared to male carp. The differences in L-W relationships between female and male carp species may be due to our sampling season occurring during the spawning period [21, 22] and/or to catchment-specific conditions [23]. DeGrandchamp et al. [24] examined female and male carp species in the Illinois River catchment and found that female bighead carp were larger than males, whereas no sex class differences were observed in female and male Silver Carp. However, Papoulas et al. [25] found that silver carp females from the Missouri River were larger than males, whereas no sex differences were observed in bighead carp. Given that both these studies occurred in different rivers and regions, there is evidence to suggest that L-W relationships among both bighead and silver Carp may be site-specific and allotted to local environmental conditions.

Our findings not only provide greater insight into the L-W relationships of both native and non-native fishes of the Great Plains ecoregion which are scarce but also broaden our understanding of basic relationships among native and non-native fishes that are likely to be abundant and wide-ranging throughout riverine environments [26].

TABLE 1: Length-weight relationships for 18 fish species (14 native and 4 non-native) sampled from the lower Red River catchment of Arkansas, Texas, and Oklahoma.

Species	N	TL range (mm)	(mean ± SE)	Weight range (g) (mean ± SE)	L-W regression	b	Intercept 95% CI	Slope 95% CI	r ²
Alligator gar (<i>Atractosteus spatula</i>)	67	801.0–2012.0	(1347.5 ± 25.1)	2250.0–50000.0 (14596 ± 969.8)	log ₁₀ weight = -6.25 + 3.31 (log ₁₀ length)	3.31	-7.13 -5.37	3.03 3.60	0.89
Bigmouth buffalo (<i>Ictiobus cyprinellus</i>)	974	343.0–1021.0	(610.3 ± 2.5)	425.0–22000.0 (4085.9 ± 66.0)	log ₁₀ weight = -5.65 + 3.31 (log ₁₀ length)	3.31	-5.94 -5.36	3.21 3.42	0.80
Black buffalo (<i>I. niger</i>)	682	289.0–990.0	(661.8 ± 3.7)	125.0–22000.0 (4974.1 ± 110.5)	log ₁₀ weight = -6.36 + 3.55 (log ₁₀ length)	3.55	-6.72 -6.00	3.42 3.67	0.82
Blue catfish (<i>Ictalurus furcatus</i>)	199	128.0–1118.0	(654.1 ± 11.7)	30.0–21000.0 (3647.9 ± 237.6)	log ₁₀ weight = -6.17 + 3.41 (log ₁₀ length)	3.41	-6.58 -5.76	3.27 3.56	0.92
Blue sucker (<i>Cycleptus elongatus</i>)	431	247.0–731.0	(530.0 ± 3.7)	75.0–4700.0 (1236.7 ± 33.6)	log ₁₀ weight = -6.58 + 3.53 (log ₁₀ length)	3.53	-6.96 -6.21	3.39 3.67	0.86
Channel catfish (<i>I. punctatus</i>)	57	195.0–1052.0	(567.1 ± 20.3)	100.0–18600.0 (2541.2 ± 387.3)	log ₁₀ weight = -6.38 + 3.50 (log ₁₀ length)	3.50	-7.16 -5.60	3.21 3.78	0.92
Flathead catfish (<i>Pylodictis olivaris</i>)	113	154.0–1146.0	(418.9 ± 23.7)	40.0–25250.0 (2314.8 ± 439.8)	log ₁₀ weight = -5.36 + 3.14 (log ₁₀ length)	3.14	-5.65 -5.06	3.03 3.26	0.96
Freshwater drum (<i>Aplodinotus grunniens</i>)	198	182.0–749.0	(407.0 ± 7.7)	50.0–5600.0 (1017.3 ± 72.6)	log ₁₀ weight = -5.30 + 3.12 (log ₁₀ length)	3.12	-5.77 -4.77	2.93 3.31	0.84
Longnose gar (<i>Lepisosteus osseus</i>)	594	433.0–1512.0	(886.9 ± 8.4)	50.0–11900.0 (2149.6 ± 85.1)	log ₁₀ weight = -7.37 + 3.59 (log ₁₀ length)	3.59	-7.70 -7.03	3.47 3.70	0.87
Paddlefish (<i>Polyodon spathula</i>)	205	950.0–1690.0	(1296.7 ± 9.8)	3500.0–39000.0 (9178.1 ± 275.5)	log ₁₀ weight = -2.89 + 2.19 (log ₁₀ length)	2.19	-3.98 -1.79	1.84 2.54	0.42
River carpsucker (<i>Carpionodes carpio</i>)	621	151.0–689.0	(333.5 ± 2.8)	25.0–3700.0 (459.0 ± 15.2)	log ₁₀ weight = -4.10 + 2.65 (log ₁₀ length)	2.65	-4.45 -3.76	2.51 2.79	0.70
Shortnose gar (<i>L. platostomus</i>)	167	481.0–810.0	(622.3 ± 5.8)	250.0–2000.0 (792.1 ± 26.1)	log ₁₀ weight = -3.88 + 2.42 (log ₁₀ length)	2.42	-4.91 -2.86	2.05 2.78	0.50
Smallmouth buffalo (<i>Ictiobus bubalus</i>)	1848	182.0–920.0	(528.5 ± 1.9)	50.0–15700.0 (2507.4 ± 32.5)	log ₁₀ weight = -5.37 + 3.20 (log ₁₀ length)	3.20	-5.59 -5.15	3.12 3.28	0.76
Spotted gar (<i>L. oculatus</i>)	115	353.0–853.0	(549.9 ± 9.2)	95.0–2250.0 (613.4 ± 35.2)	log ₁₀ weight = -5.76 + 3.10 (log ₁₀ length)	3.10	-6.51 -5.01	2.82 3.37	0.81
*Bighead carp (<i>Hypophthal. nobilis</i>)	119	868.0–1382.0	(1110.8 ± 9.8)	8250.0 37500.0 (17177.5 ± 567.8)	log ₁₀ weight = -4.23 + 2.77 (log ₁₀ length)	2.77	-5.52 -2.93	2.35 3.20	0.58
*Common carp (<i>Cyprinus carpio</i>)	30	494.0–986.0	(699.0 ± 21.9)	750.0 13600.0 (5094.0 ± 567.6)	log ₁₀ weight = -6.61 + 3.61 (log ₁₀ length)	3.61	-8.25 -4.98	3.03 4.18	0.85
*Grass carp (<i>Ctenopharyn. idella</i>)	110	614.0–1560.0	(945.3 ± 11.2)	5000.0 21750.0 (9941.6 ± 355.0)	log ₁₀ weight = -2.80 + 2.28 (log ₁₀ length)	2.28	-3.78 -1.82	1.95 2.60	0.63
*Silver carp (<i>Hypophthal. molitrix</i>)	315	532.0–1100.0	(877.8 ± 4.5)	1750.0 16900.0 (8497.0 ± 145.0)	log ₁₀ weight = -5.21 + 3.10 (log ₁₀ length)	3.10	-5.72 -4.69	2.92 3.27	0.79

*Fishes that are non-native to the Red River catchment. Sampling was completed from May 2021 to March 2024 using electrofishing and gill nets. Sample size (N) varied by species. The range of the species total length (TL) and weight are provided. The regression equation (L-W regression), b values, and 95% confidence intervals (CI) for the intercept and slopes and the coefficient of determination (r²) are indicated.

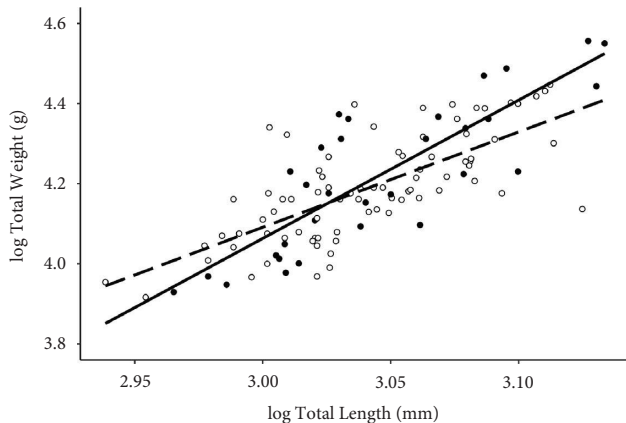


FIGURE 1: Relationship between total length (mm) and body weight (g) (log-transformed) for both sexes of bighead carp in the lower Red River catchment of Arkansas, Texas, and Oklahoma. The black dots and the solid line represent female bighead carp, whereas white dots and the dashed line represent male bighead carp.

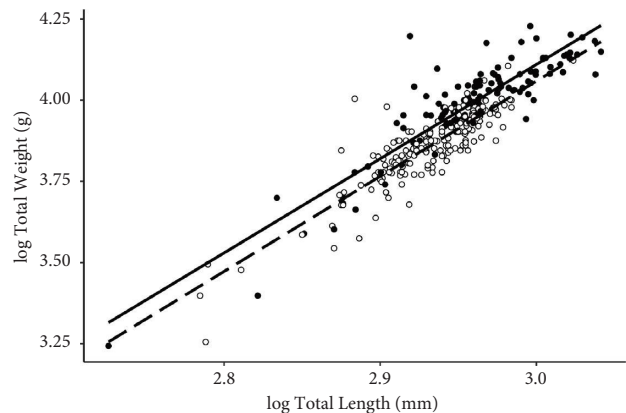


FIGURE 2: Relationship between total length (mm) and body weight (g) (log-transformed) for both sexes of silver carp in the lower Red River catchment of Arkansas, Texas, and Oklahoma. The black dots and the solid line represent female silver carp, whereas white dots and the dashed line represent male silver carp.

Data Availability

Data used in this study to support our findings are available upon request from Jim Burroughs (jim.burroughs@odwc.ok.gov).

Disclosure

Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

Conflicts of Interest

The authors declare that they have no conflicts of interest.

Acknowledgments

This research was supported by Oklahoma Department of Wildlife Conservation (10.13039/100014728) (F21AF00118), Arkansas Game and Fish Commission (F20AP11538), Texas Parks and Wildlife Department (10.13039/100004930) (CA-0001012), and the U.S. Fish and Wildlife Service (10.13039/100000202) (FAC2021001740). This research is a contribution of the Alabama Cooperative Fish and Wildlife Research Units (United States Geological Survey, Alabama Department of Conservation and Natural Resources, Auburn University, and Wildlife Management Institute cooperating). Funding was provided by the Oklahoma Department of Wildlife Conservation (Jim Burroughs, jim.burroughs@odwc.ok.gov), Arkansas Game and Fish Commission, Texas Parks and Wildlife, and U.S. Fish and Wildlife Service. We thank Paul Ramsey, Ben Birdsall, Aiden Maddux, Daniel Bryant, and many technicians for field assistance. We thank Monica McGarrity for assistance with the grants. Steve Rider provided helpful comments on an earlier version of our manuscript. This study was performed under the auspice of Auburn University Animal Care and Use protocol # 2020-3800.

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RESEARCH ARTICLE

WILEY

Evaluating the context dependency of large river nursery habitats for fishes in the lower Red River catchment

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Funding information

Texas Parks and Wildlife Department; Arkansas Game and Fish Commission; Oklahoma Department of Wildlife Conservation; U.S. Fish and Wildlife Service

Abstract

Both fine- and coarse-scale physicochemical conditions affect the quantity and quality of nursery habitats within riverine ecosystems. Nursery habitats in large, braided, and sandbed streams such as the lower Red River of Oklahoma, Texas, and Arkansas are not well described and likely vary among species. Identification of nursery habitats is important for developing proper conservation and management actions. We used an occupancy model framework to determine how hierarchical habitat factors related to the occupancy of 38 juvenile fish species. Our findings indicate that large river nursery habitats can generally be defined by reaches with off-channel slack-water habitat, having deep pools but shallow thalweg depths, typically located further away from dams, and with low percentages of limestone lithology. Species within the same genera often exhibited variable relationships with river slope, amount of large woody debris, channel shape, discharge, and position of reaches within the stream network. Our results indicate important species-specific relationships that define nursery habitats, indicating an important context dependency of nursery habitats even within fishes that are taxonomically similar. If the goal is to improve recruitment by native fishes, then consideration of the important species-specific differences would be beneficial if improvements are made to nursery habitats. Moreover, careful consideration of the effects of dam operations will help maintain proper connectivity to off-channel habitats important in downriver portions of the river network.

KEYWORDS

connectivity, dams, fish distribution, multiscale analysis, nursery habitat, occupancy model, species composition

1 | INTRODUCTION

Freshwater ecosystems are among the most diverse ecosystems on Earth; however, they may also be the most endangered (Reid et al., 2019). Despite covering only 2.3% of the Earth's surface, freshwater ecosystems account for 9.5% (126,000 species) of the world's described animal species (Balian et al., 2008). Dudgeon et al. (2006) lists over-exploitation, flow modification, water pollution, habitat-degradation, and invasive species as the five major threats to freshwater biodiversity. Many of these threats work interactively and can

include cumulative effects on ecosystems (Richter et al., 1997). Additional human activities further intensify the threats to freshwater biodiversity by affecting many of the natural biological and ecological processes associated with lotic organisms (Allan & Flecker, 1993; Palmer & Ruhí, 2019; Richter et al., 1997). Impoundments, in particular, have been related to declines in native fish populations, particularly as related to reproductive success (Agostinho et al., 2008; Bergstedt & Bergersen, 1997; Fenkes et al., 2016; Larinier, 2001).

Dams are now integrated into many fragmented riverscapes and can affect many natural processes of riverine ecosystems. Dams are a

significant threat to riverine fishes as approximately 65% of the world's large rivers are impounded (Grill et al., 2019). Dam construction alters flow and temperature regimes and has reduced both habitat heterogeneity and riverscape connectivity (Brewer et al., 2016; Magilligan & Nislow, 2005; Olden & Naiman, 2010). Dams disrupt natural flow patterns, including the magnitude, duration, frequency, timing, and rate of change (Poff et al., 1997; Poff & Zimmerman, 2010), and similar effects are associated with altered thermal regimes (Caissie, 2006). The remaining fragmented landscape may be problematic for migrating fishes that are downstream but require habitat that is upstream of the dam (e.g., migrating salmon, Hilborn, 2013; and Paddlefish, Zigler et al., 2004) or fishes that rely on downstream drift dynamics to position their offspring in suitable nursery habitats (Worthington et al., 2014). Impoundments also alter the downstream sediment regime and stream power which affects the creation and maintenance of nursery habitats (Grant et al., 2003). For example, the suspended sediment load in the lower Missouri River has been reduced by 83% since the closure of the most downstream dam (Jacobson et al., 2009) and the fish nursery habitat has largely been diminished due to the associated flow regulation and channelization (Tracy-Smith et al., 2012). Regardless of alterations to the riverscape, the quality and quantity of nursery habitats are key for maintaining successful reproduction and recruitment by fishes (Rochette et al., 2010).

Understanding the drivers of fish distributions is important for both our ecological understanding and management planning; however, the integration of scale in our examination of distributions is often lacking (e.g., local vs. ecosystem). Distributions, particularly those of riverine species, can be used to discern species assemblages, metapopulations, movement of species, and critical habitat (Chandler et al., 2015; Dorazio et al., 2006; Falke et al., 2010). Rosenfeld and Hatfield (2006) describe how critical habitat designation continues to be one of the most convoluted problems of species management because of the multifaceted aspects of biology, ecology, locality, and landscape constraints. The designation of critical habitats is essential for management planning. For example, Esselman and Allan (2011) developed a reserve stream network for native fishes in several rivers draining to the coast of Belize in an effort to conserve critical management zones. Despite the importance of spatial scale to distributions, many studies do not account for both the fine and coarse-scale habitat constraints placed on species' distributions. Yet, coarse-scale landscape constraints (i.e., climate and geology) are considered important higher-level determinants of fish distributions (Frissell et al., 1986; Hynes, 1975). Although advancements have been made in our understanding of riverine fish distributions, few studies have included landscape constraints to delineate the nursery habitats of juvenile fishes, particularly complete assemblages (Devarajan et al., 2020).

Riverine nursery habitats are important for the success of juvenile fishes and are often threatened habitats due to human landscape modifications. Riverine nursery habitats typically provide warm water temperatures (King, 2004), shelter from high water velocities and predators (Rempel et al., 1999), and also tend to have higher concentrations of food (Fuiman & Werner, 2002). Refugia from high water velocity is particularly important in large river systems (Freeman

et al., 2001; Rempel et al., 1999) as it can better allow young fish to acquire the resources necessary for survival without being displaced (Hart & Finelli, 1999). Nursery habitats facilitate higher growth rates in young fishes, so they are better prepared to move to alternative habitats and survive their first winter (Humphries et al., 1999). For example, Hoxmeier and DeVries (1997) found that American Paddlefish *Polyodon spathula* in the lower Alabama River, Alabama, used oxbow lakes as nursery areas, suggesting a growth benefit prior to reaching reproductive size. Flow modification due to damming, climate change, habitat loss, and invasive species has the potential to decrease both the quality and quantity of nursery habitats (Rochette et al., 2010). The loss of nursery habitat can result in a recruitment bottleneck because fishes are the most environmentally sensitive during early ontogeny (Fuiman & Werner, 2002).

Nursery habitats used by a myriad of native species occupying large river systems are not well understood, despite the importance of the information for conservation and management. Riverine nursery habitat has traditionally been defined by both relatively shallow water and low-velocity habitats (Love et al., 2017). Additionally, tributaries (Pracheil et al., 2009), pools (de Ávila-Simas et al., 2014), sandbars (Tracy-Smith et al., 2012), side channels (Pease et al., 2006), and floodplains (King, 2004) are all considered important nursery habitats in various mainstem rivers. However, in large, braided rivers some of these channel features can be relatively rare, particularly due to human modifications (Blodgett & Stanley, 1980; Mollenhauer et al., 2022). Moreover, the spatial context for defining nursery habitats for different assemblage members is often lacking. Therefore, the traditional definition of nursery habitat is too general to effectively aid in developing meaningful conservation and management actions. Our objective was to quantify nursery habitats for the lower Red River fish assemblage, including considerations of reach-scale factors that are often overlooked. The findings of our study describe species-specific relationships that can inform habitat conservation and restoration in similarly regulated riverscapes.

2 | METHODS

2.1 | Study area

This study took place in the lower Red River catchment of Oklahoma, Texas, and Arkansas, USA (Figure 1). The Red River catchment drains more than 241,00 km² from east-central New Mexico to Louisiana and is affected by flow regulation. The upstream portion of the catchment is located in the semi-arid regions west of the Cross Timbers (26–29 cm of annual rainfall, Woods et al., 2005) and becomes more humid in the eastern and southern portions of the catchment (127–152 cm of annual rainfall, McCorkle et al., 2016). The Red River terminates near the “three rivers confluence” where the Red River, Old River, and the Atchafalaya River combine with the tributary outflow of the Mississippi River into the Atchafalaya basin. A mix of savanna, woodland, and grasslands comprise the Cross Timbers region where flow variability and stream drying are common (Mollenhauer

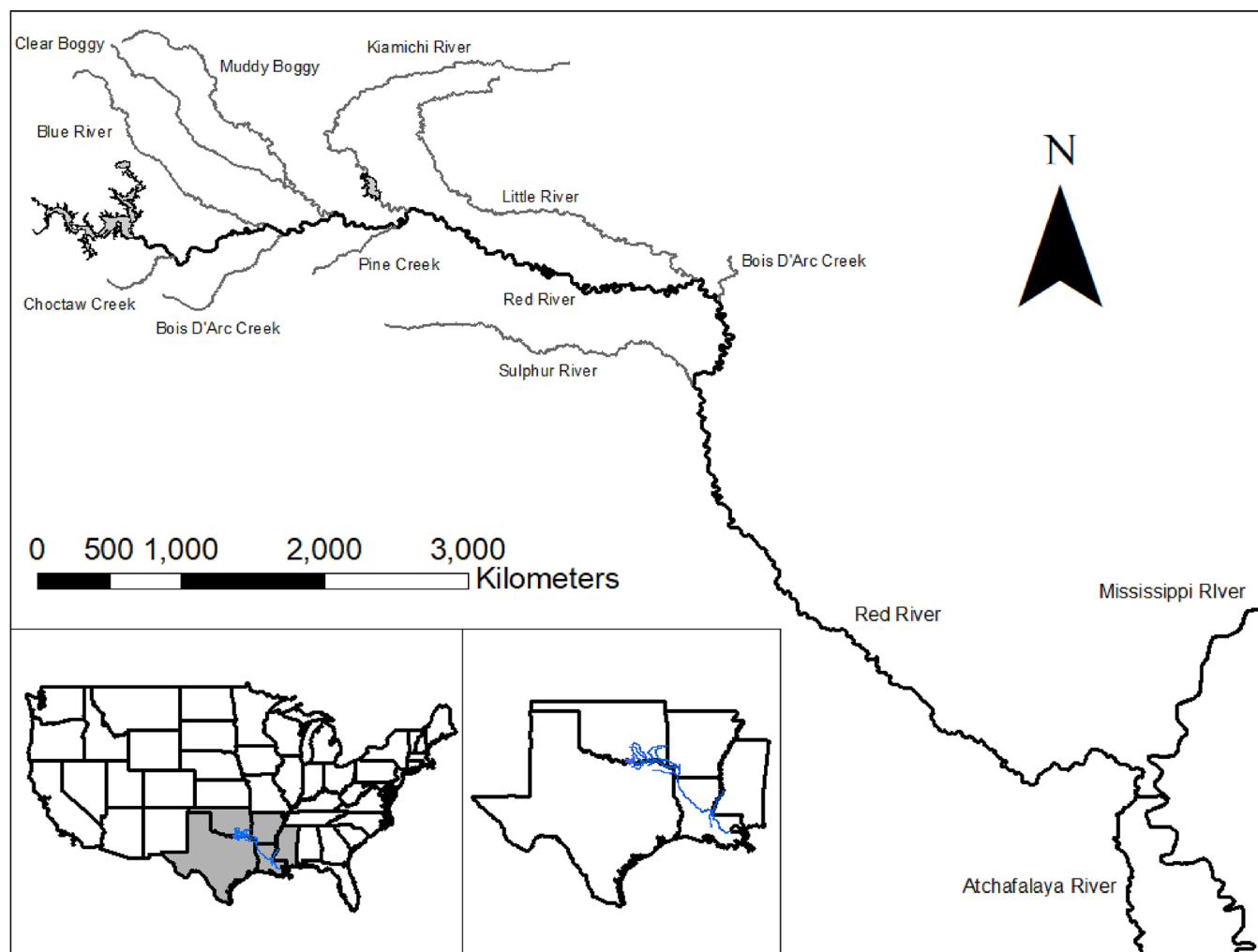


FIGURE 1 The lower Red River catchment from Lake Texoma to the “three rivers confluence” where the Red River, Old River, and the Atchafalaya River meet and combine the distributary outflow of the Mississippi River with the Atchafalaya catchment. Many of the major tributaries are impounded including Bois D'Arc Creek which began holding water in 2021 but is currently closed to recreation. [Color figure can be viewed at [wileyonlinelibrary.com](https://onlinelibrary.wiley.com/terms-and-conditions)]

et al., 2022). The mainstem Red River is a relatively wide, sand bed river that forms the Oklahoma–Texas border. The upper and lower portions of the Red River catchment are divided by Lake Texoma (impounded by Denison Dam in 1944, Riggs & Bonn, 1959). Lake Texoma was impounded for recreation and hydropower and there are currently no minimum flow releases or ramping rates. Discharge releases represent either flood flows or flows associated with power generation, resulting in discharge fluctuations (average variation of 243 m³/s since 1996) during the spring and summer of most years.

Our study area was in the lower portion of the catchment where the Red River is influenced by several major tributaries: the Kiamichi River, Muddy Boggy Creek, Bois D'Arc Creek, Choctaw Creek, and the Blue River. Below Denison Dam, the mainstem Red River is unimpounded until it reaches a sequence of locks and dams in Louisiana. The mainstem river of the lower catchment shifts to an alluvial valley and sand-bed river except for areas immediately downstream of Lake Texoma where downcutting has led to exposed bedrock (Woods et al., 2005). The river is braided in many sections and large woody

debris is common in the channel. The major tributaries traverse several upland ecoregions, but the lower portion of the tributaries are separated by impoundments in most cases. The Muddy Boggy River is an exception and is free-flowing except for its smaller tributary (Clear Boggy Creek) which is impounded by Pine Creek Reservoir. The upland tributaries typically comprise cobble and gravel streambeds that transition to silt and sand at their lower extents. Unlike the Red River in Oklahoma, the mainstem river in Arkansas has been heavily dredged and contains several wing dike structures that direct flow to create deeper water. As the river progresses downstream, it transitions to a deeper alluvial channel until it reaches the first lock and dam in Louisiana.

2.2 | Site selection

We sampled 67 reaches (hereafter sites, ≈275-m long) in the lower Red River catchment between Denison Dam, Oklahoma, and the Arkansas–Louisiana state line from May through October of 2021 and

2022. Each site consisted of several channel unit complexes including pools, sandbars, back dunes, backwaters, side channels, and runs. The sites were haphazardly chosen in areas across the catchment with reasonable river access and with permissions from private landowners where needed. Sites were a minimum of 250 m apart to prevent disturbing fish from adjacent reaches while sampling. Each site was surveyed multiple times over one season. The sampling season was May through October because this is when juvenile fish tend to occupy nursery habitat before encountering their first winter (Scheidegger & Bain, 1995), and it was reasonable to consider our sites closed at the species level (e.g., new species would not immigrate and emigrate from the site). We sampled each site (i.e., hereafter survey) approximately three times during the sampling season to account for incomplete sampling detection (see Section 2.5).

2.3 | Fish sampling

We sampled juvenile fishes using two different gear types during daylight hours. Using a combination of gears diminishes some of the sampling bias associated with a single-gear approach (Clark et al., 2007). Each site was sampled with mini-fyke nets and beach seines. First, we set 3 mini-fyke nets (0.6 m × 4.5 m lead, 0.6 × 1.2 m trap, 3-mm mesh) in <2 m of water at locations adjacent to the shoreline to target small-bodied fishes (Eggleton et al., 2010). Mini-fyke nets are commonly used to sample juvenile fishes (Eggleton et al., 2010) and sometimes capture higher numbers of fishes when compared to other gears (Fago, 1998; Pope et al., 2009; Snow et al., 2017). Next, we used a beach seine (1.8 m × 4.6 m, 3-mm mesh and 1.8 m × 9.2 m, 3-mm mesh) to sample wadeable habitat across the reach using a modified (i.e., smaller) version of the encirclement technique (Bayley & Herendeen, 2000). Transects were established throughout wadeable habitat at each site and seine hauls were completed across each transect. Individual seine hauls were limited to 25 m to maintain gear efficiency (Bayley & Herendeen, 2000; Lombardi et al., 2014).

2.4 | Environmental conditions

We quantified select physicochemical factors across multiple spatial scales (i.e., reach, segment, and catchment) to account for the hierarchical arrangement of river ecosystems. Stream habitat is inherently hierarchical where finer levels of organization are nested within coarser landscape constraints (Frissell et al., 1986; Imhof et al., 1996). For example, finer-scale channel unit conditions (i.e., substrate) used by fish are often influenced by coarse factors (i.e., drainage area and geology) of the surrounding catchment (Wiens, 2002). Our sites consisted of a 200- to 300-m river section containing a variety of channel unit features and reflected the finest scale of observation. Stream segments represent the physicochemical conditions between two 5th-order tributary confluences, where multiple reaches may occur. Catchments reflect the total upstream area draining to the downstream end of each site. Coarser-scale (e.g., segment and catchment)

habitat factors are conditions that apply to multiple sites sampled within the same stream segment or catchment (i.e., nested).

We quantified six covariates that we hypothesized would relate to the detection of juvenile fishes. We measured sampling effort (i.e., area sampled) because greater effort may result in higher species' detection (Simonson & Lyons, 1995). We quantified water temperature (1.0°C) because as fish grow, they become more active and more susceptible to sample gears (Goffaux et al., 2005; Guy et al., 2009). We included calendar day because abundance also changes during the first summer of life via spawning events and high early life mortality and can affect detection (Wedgeworth, 2021). Water clarity (1.0 cm) was quantified using a Secchi disk (Preisendorfer, 1986) because clearer water allows fishes to evade gears more easily (Mollenhauer et al., 2018). We also measured dissolved oxygen (1.00 mg/L) because decreased oxygen can inhibit fish detection (Coutant, 1985; Tyler & Targett, 2007). Lastly, discharge is a primary factor that affects fish detection because it influences our ability to adequately or effectively sample different habitats (Lyon et al., 2014).

The six detection covariates were quantified at three locations of each site during each survey. Calendar day and sampling effort were recorded for each survey. We measured the water temperature (1.0°C) and dissolved oxygen (DO, 1.00 mg/L) at 0.5 m below the water's surface using a handheld, multi-parameter, water-quality meter (YSI ProDSS). These measurements were quantified at three locations site locations during each survey and then averaged to represent physicochemical conditions. Alternatively, a single daily discharge was obtained from the nearest U.S. Geological Survey (USGS) stream gages (Red River, 07331600, Denison, TX; Red River, 07335500, Arthur City, TX; Red River, 07336820, De Kalb, TX; Bois D'Arc Creek, 07332622, Honey Grove, TX; Red River, 07337000, Index, AR; Red River, 07341500, Fulton, AR; Red River, 07344370, Spring Bank, AR; Blue River, 07332500, Blue, OK; Muddy Boggy Creek, 07335300, Unger, OK; Kiamichi River, Army Corps HGL02, Hugo, OK) and applied to each survey.

We quantified six occupancy covariates during each survey that were hypothesized to relate to occupancy of nursery habitat by riverine fishes. We expect juvenile fishes to expend less energy and therefore experience higher growth rates within slackwater habitats of larger rivers (Johnson et al., 2006; Korman & Campana, 2009; Nislow et al., 2015); thus, we quantified the percentage of total slackwater habitats (e.g., forewater, backwater, and side-channels, Galat et al., 2004) using a tape measure (1.0 m) and range finder (Simmons Volt 600 Laser Rangefinder, 1.0 m) to calculate the percent of the total site area. The length of deeper pools at each site was quantified using sidescan sonar (Humminbird Helix 12) where a pool was defined as the length of the area within a reach that contained a steep (>2 m) change in depth. Large woody debris (hereafter LWD) and other complex habitat features are used by juvenile fishes to avoid predation (Everett & Ruiz, 1993; Thompson et al., 2018) and increase food consumption (e.g., higher biofilm and invertebrate densities; Allouche, 2002). Thus, we quantified the percent of LWD (diameter > 10 cm and length > 1.5 m, Kaeser & Litts, 2008) at each site and width-to-thalweg depth ratios (W:D) that describe channel shape

(i.e., wide and shallow or narrow and deep). We also quantified salinity (1.0 ppt) during each survey because conditions can be quite variable in this catchment (Hargrave & Taylor, 2010), and salinity can influence the osmoregulatory ability of fishes with a wide range of tolerances in freshwater fishes (Matthews et al., 1977).

We quantified zooplankton density and species composition (i.e., clade) because of their role in providing food to juvenile fishes (Fernando, 1994). We quantified zooplankton at each site using a vertical planktonic tow net (12.7 cm diameter, 40 cm length, and 80- μ m mesh). The net was lowered 1.2 m into the water column and pulled back to the surface. Three subsamples were taken across each site. We rinsed the samples and preserved them in 70% ethanol for later laboratory processing. In the laboratory, samples were filtered through 300, 200, 100, and 80 μ m sieves, respectively, to remove sediment and concentrate samples. The filtered samples were then stored in 70% ethanol. A 1-mL subsample was pipetted (VWR Ergonomic High-Performance Single-Channel 100–1000 μ L pipettor) from each sample, and all organisms in the subsample were enumerated on a 1-mm² gridded Segewick-Rafter cell (Wildlife Supply Company, Model# 1801-G20). Organisms were grouped into Cladocerans, Copepods, and Rotifers and enumerated.

We used USGS stream gage data to quantify the general flow patterns to relate to the occupancy of nursery habitat at the segment scale. Similar to detection, we obtained the gage data from the nearest USGS gages (see above). However, for occupancy, we used the daily discharge values during the season (May–October) to calculate median discharge. All discharge values were scaled (i.e., divided) by drainage area thereby making them comparable across sites (e.g., high discharge in a tributary was comparable to high discharge in the mainstem).

We used existing geospatial data and GIS tools (ESRI, 2022) to calculate the remaining occupancy covariates: distance to the nearest upstream dam, river sinuosity, slope, drainage area, limestone lithology, and a landscape disturbance index. At each site, we quantified the distance of the downstream sample reached to the nearest upstream dam (1.0 km) using the National Hydrology Dataset (NHDplus; U.S. Geological Survey, 2017) flowlines and ArcGIS Pro spatial analyst. The Great Plains are home to many pelagic broadcast spawning fishes that require discharge events to successfully reproduce and develop in the downstream drift (Perkin et al., 2015; Worthington et al., 2018). Many of these pelagophils are thought to spawn downriver of dams (Mollenhauer et al., 2021), thus, only certain portions of the river may have nursery habitats available depending on the drift requirements (i.e., may be hundreds of kilometers downriver depending on flow, Worthington et al., 2014).

At the segment scale, we measured the sinuosity and slope using the NHDplus flowlines and GIS spatial analysis. The sinuosity (0.1 index) of a stream describes channel meander and was calculated by dividing the thalweg length by the straight-line distance of the segment. The choice of spawning location is associated with channel sinuosity (Fukushima, 2001); therefore, nursery habitats may also be influenced by sinuosity. Higher stream gradients (i.e., slopes) can increase water velocity (Camana et al., 2016), which can negatively

affect nursery habitat occupancy. Therefore, we calculated segment slope (1.0%) as the change in elevation between the upstream and downstream extent of each stream segment and divided by the thalweg length (i.e., channel distance measured down the middle of the channel; Bain & Stevenson, 1999).

We used the NHDplus flow lines, the National Land Cover Dataset, and National Geologic Map Database to calculate the remaining catchment-scale occupancy covariates (i.e., drainage area, landscape disturbance, and limestone lithology). First, we measured the area draining to the downstream extent of each reach (hereafter drainage area, 1.0 km²), using the watershed tool in ArcGIS Pro. Drainage area describes the size and relative position of a reach within a stream network and can affect juvenile fish assemblage composition (Schlosser, 1995). We also quantified landscape disturbance (hereafter LDI) following Brown and Vivas (2005) using the 2021 National Land Cover Dataset (further NLCD; Dewitz, 2021) and a modification of Mouser et al. (2019) (see below). Human land-use modifications can disproportionately affect the quality and quantity of riverine nursery habitat (Britton & Pegg, 2011; Rochette et al., 2010; Schlosser, 1995). However, land-cover types tend to be multicollinear because they sum to 100%; thus, combining land cover into a single index is helpful when analyzing data using multiple regression scenarios (Genovese et al., 2001). We characterized the level of LDI following Mouser et al. (2019). Because catchment limestone compositions control local pH and water hardness conditions (Frissell et al., 1986; Stevenson, 1997) that can affect egg survival (Parker & McKeown, 1987; Swain et al., 2020), we quantified the percentage contained within the catchment draining to each site. We gathered State Geologic Map Compilation (SGMC) geology data from the USGS National Geologic Map Database (Horton et al., 2017), and applied it to the previously calculated drainage areas and calculated the percentage of each geologic category within the drainage areas.

2.5 | Statistical analyses

We built a multispecies single-season occupancy model for juvenile native fishes to quantify nursery habitat use (MacKenzie et al., 2002). An occupancy model allows for the estimation of a probability of occurrence while accounting for incomplete detection by the sampling gears. Variation in both detection and occupancy is explained by collected environmental covariates (Mackenzie, 2006). We chose an occupancy model framework because imperfect detection of species by sampling gears is typical within aquatic ecosystems (Benoit et al., 2018; Wedderburn, 2018). Accounting for false absences (i.e., the failure to detect a species when it is present) is important for developing meaningful occupancy relationships (i.e., not just modeling gear inefficiencies).

We built an occupancy model (OM) using temporally replicated surveys from May through October 2021–2022. Repeated surveys allow for the model to create estimates of both a detection probability (p_i) and an occupancy probability (ψ_i) (Kéry & Royle, 2015). We established our sample season during the spawning season of many native

fishes of the catchment (e.g., after the water has reached $>18^{\circ}\text{C}$). The season ended while juvenile fishes were still using nursery habitats but before water temperatures declined appreciably during late autumn. We quantified both detection and occupancy covariates to explain variations in detection or occupancy probabilities (MacKenzie et al., 2002). We spaced our sites at least 250 m apart so surveying one site did not influence detection at an adjacent site. Lastly, we included grouping factors to account for the nested nature of river systems to account for pseudoreplication in these data.

We transformed our data where needed, checked correlations, and standardized these data prior to model development. Our detection covariates of dissolved oxygen, visibility, seining effort, and discharge were log-transformed due to their right-skewedness. Next, we checked detection covariates to ensure they were not multicollinear ($|r| > 0.50$; Roever et al., 2014) using Pearson's correlation coefficient (Table S1). All detection variables had $|r| \leq 0.35$ and were therefore retained for the model building process. We completed the same process for occupancy covariates (Table S2). We retained W:D, median discharge, and slope for model development. The percent of limestone lithology, slope, LWD, thalweg depth, W:D, and zooplankton counts were all log-transformed due to skewed distributions. Additionally, drainage area, percent of deep pools, and percent of slackwater were transformed into categorical variables based on natural breaks in the data (i.e., bimodal). Categorical transformation of drainage area represented either high ($>50,000\text{ km}^2$) or low ($<50,000\text{ km}^2$) drainage areas, whereas deep pools and slackwater represented either the presence or absence. Categorical covariates were tested for independence by evaluating the frequency at which they occurred together at each site. Lastly, all continuous covariates were standardized to a mean of zero and a standard deviation of one to improve model convergence and interpretation (MacKenzie et al., 2017; Mackenzie & Royle, 2005).

We built the occupancy model using covariates to inform the variation in both detection and occupancy. We built the detection component of the model by choosing two covariates that were hypothesized to have shared relationships among juvenile fishes and gear detection (i.e., not species specific) so more emphasis could be placed on the occupancy portion of the model. Therefore, variation between species was not included in the detection model. To determine which detection covariates should be retained, we fit a global detection model and assessed the effect sizes of the covariates. Discharge and water temperature had the greatest effect sizes and are commonly used to explain detection (Carpenter-Bundhoo et al., 2023; Maire et al., 2019); therefore, we fit the detection model (i.e., all species) with these two parameters to avoid overfitting the model.

The detection component of the model is expressed as:

$$\text{logit}(p_{ijk}) = a_i + \beta_1 X_{1[ij]} + \beta_2 X_{2[ij]},$$

$$\text{for } i = 1, 2, \dots, i, \text{ for } j = 1, 2, \dots, j, \text{ for } k = 1, 2, \dots, k,$$

$$a_i \sim t(\mu, \sigma^2, v),$$

$$\beta_m \sim t(\mu, \sigma^2, v),$$

where p_{ijk} is the detection probability during survey j at site i for species k , a_i is the detection intercept, β_1 & β_2 are the coefficients for the two covariates, and X_1 & X_2 are the detection covariates.

The occupancy portion of the model was built similarly to the detection portion, except we fit species-specific relationships using the covariates. The detection component was held constant as the occupancy component was fit. We fit the occupancy component with the presence of slackwater, the presence of deeper-water pools, high or low drainage area, and the continuous covariates of thalweg depth, W:D, LWD, distance to the nearest upstream dam, median discharge, slope, and percent limestone lithology. Each species was modeled around the group mean, hyperparameter μ . The interpretation is similar to a random-slopes model where individual species are treated as random intercepts rather than focusing on interspecies differences. The resulting occupancy probabilities are interpreted similarly to individual models but with the power of a single model (Kéry & Royle, 2015). We also included grouping factors for both segment and sample year to account for any unexplained variability within the model. The inclusion of grouping factors within the model also accounts for pseudoreplication and spatial correlation created by the nested site study design (Wagner et al., 2006).

The occupancy component of the model is expressed as:

$$\text{logit}(\psi_{ik}) = a_i + \beta_{1[ij]} X_{1[k]} + \beta_{2[ij]} X_{2[k]} \dots + \beta_{N[ij]} X_{N[j]} + \gamma_{k[s]} + \gamma_{k[y]},$$

$$\text{for } i = 1, 2, \dots, i, j = 1, 2, \dots, j,$$

$$a_i \sim t(\mu, \sigma^2, v),$$

$$\beta_{N[ij]} \sim t(\mu, \sigma^2, v),$$

$$\gamma_{k[s]} \sim t(\mu, \sigma^2, v), \text{ for } s = 1, 2, \dots, 3,$$

$$\gamma_{k[y]} \sim t(\mu, \sigma^2, v), \text{ for } y = 1, \dots, 2$$

where ψ_{ik} is the species probability of occurrence at site i for species k , a_i is the occurrence intercept, β_N are the coefficients for the occurrence covariates X_N for species k at site i , $\gamma_{k[s]}$ is the segment grouping factor, $\gamma_{k[y]}$ is the year grouping factor, μ is the mean hyperparameter of the assemblage, and σ is the variance around hyperparameter μ .

We used vague priors to calculate the posterior distributions. When informative prior information is not available, vague priors are used to give the model a starting point for estimating parameters with minimal effect on the model results (Kéry & Royle, 2015; Kruschke, 2014). Vague truncated normally distributed priors (i.e., t-distribution) were given to main effects, and vague gamma priors were applied to their standard deviations. The t-distribution adds a normality parameter v (see equation above) which accounts for heavy tails and can improve model fit (Kruschke, 2014). Lastly,

uniform priors were given to the detection and occurrence intercepts to aid in model convergence.

We assessed the posterior distribution of the model and covariates using Markov Chain Monte Carlo simulations (Marjoram et al., 2003). We ran 150,000 iterations on 3 chains with a burn-in of 10,000 and thinning of 5. The model was fitted using the package *jagsUI* (Kellner, 2015) and the program JAGS (Plummer, 2003) within the statistical computing software R (Version 4.2.2, R Core Team, 2022). The back-transformed logit parameter was used to calculate the detection and occurrence probabilities. Model convergence was evaluated using the Brooks–Gelman–Rubin statistic $\hat{R}$ (Gelman et al., 1992), where parameter estimations, $\hat{R} < 1.1$, indicate the appropriate mixing of chains. Lastly, we used an omnibus goodness-of-fit test (i.e., evaluating chi-squared discrepancies; MacKenzie & Bailey, 2004), where $\hat{c}$ values within 1.00 to 1.02 are considered to have adequate dispersion (Kéry & Royle, 2015). Additionally, the Bayesian p-value also provides a posterior predictive check, where values near 0.5 (i.e., values that are not close to 0 or 1) are considered to fit the observed data (Conn et al., 2018; Kéry & Royle, 2015; Kruschke, 2014).

3 | RESULTS

3.1 | Fish sampling

We completed 242 surveys across 104 sites (37 repeated across both years) in the lower Red River catchment during 2021 and 2022. During 2021 and 2022, 54 sites and 97 surveys, and 50 sites and 145 surveys were completed, respectively. Of the 67 sites, 39% ($n = 26$) were in the Oklahoma portion of the Red River (hereafter OK Red), 36% ($n = 24$) in the Arkansas portion of the Red River (hereafter AR Red), and 25% ($n = 17$) were in the major tributaries of Texas, Oklahoma, and Arkansas (hereafter tributaries).

We sampled 34,243 juvenile fishes during both sample seasons, including 42 species (Table S3) and 28 genera within the families Atherinopsidae, Catostomidae, Centrarchidae, Cyprinidae, Clupeidae, Fundulidae, Ictaluridae, Lepisosteidae, Moronidae, Percidae, and Sciaenidae. The most abundant species was Red Shiner *Cyprinella lutrensis* ($n = 4885$), followed by Bullhead Minnow *Pimephales vigilax* ($n = 4851$), and Mosquitofish *Gambusia affinis* ($n = 2345$), and the rarest species with juveniles collected was the Skipjack Herring *Alosa chrysochloris* ($n = 2$).

3.2 | Environmental conditions

Physicochemical conditions varied among sites and by stream location (i.e., OK Red, AR Red, and tributaries) particularly between the tributaries and the mainstem Red River (both AR and OK sections). As expected, drainage areas increased with sites downriver from the tributaries (3280 km²) through the OK Red (115,170 km²) to the AR Red (128,723 km²). The OK Red had a shallower thalweg depth, on

average (2.07 m), when compared to both the AR Red (2.69 m) and tributaries (2.39 m). Tributaries had generally deeper and narrower river channels, whereas the mainstem Red River had wider and shallower channels. However, the AR Red had a deeper and narrower channel compared to the OK Red. Given the ecoregion differences, the tributary sites had higher limestone composition (9.46%) compared to both portions of the mainstem Red River. The major tributaries also had lower average salinity levels (0.27 ppt) when compared to the Red River (OK Red = 0.51 ppt and AR Red = 0.40 ppt). Lastly, sinuosity and LDI were, on average, more similar across sites than other conditions.

Average detection covariates varied across surveys; however, detection covariates were more similar between stream locations, on average, than occupancy covariates. Surveys completed earlier in the season had lower average water temperatures (27.4°C), and higher scaled discharges (0.003), on average, than later surveys (temperature = 28.3°C and discharge = 0.0008). Tributary reaches were surveyed, on average, earlier (calendar date = 73) than those in the mainstem Red River (OK Red = 84 and AR Red = 87) due to river access (i.e., some ramps are only useable during higher discharge conditions). Water clarity was higher during surveys of the OK Red (43.2 cm) than both the AR Red (28.6 cm) and tributaries (36.8 cm). Lastly, dissolved oxygen and seining effort were similar among surveys and stream locations.

3.3 | Occupancy modeling

Prior to model building, we omitted data from a few sites and species. We retained data from 99 of the 104 sites for analyses. We omitted 5 sites because some had single surveys and others were missing physicochemical covariate information. Additionally, we retained data on 38 species for modeling. We omitted 4 species from model development because they were either ubiquitous or non-native. Species with extremely high occupancy (e.g., Red Shiner, Bullhead Minnow, and Mosquitofish) were removed from the dataset to aid in model convergence. We also omitted the Common Carp because our focus was on the nursery habitat of native fishes.

The final model converged and had adequate fit (Table S4). All model parameters displayed appropriate chain mixing with $\hat{R} < 1.1$ (Kéry & Royle, 2015). The OM displayed adequate dispersion of posterior values ($\hat{c}$ of 1.003), and adequate goodness-of-fit with a Bayesian p-value of 0.505.

The probability of detection and occupancy varied by species (Figure 2); however, some relationships with covariates were shared though there were differences in effect sizes. The group mean detection probability was 0.19, with detection of individual species ranging from 0.04 to 0.70 (Table I). Species detection increased with increasing water temperatures, and discharge conditions (Table S5). Further, the group mean occupancy probability was 0.57 with the individual species ranging from 0.15 to 0.96 (Table I). Juveniles of all 38 fishes had positive occupancy relationships with reaches having deep pools and slackwater habitats present, and the distance from the nearest

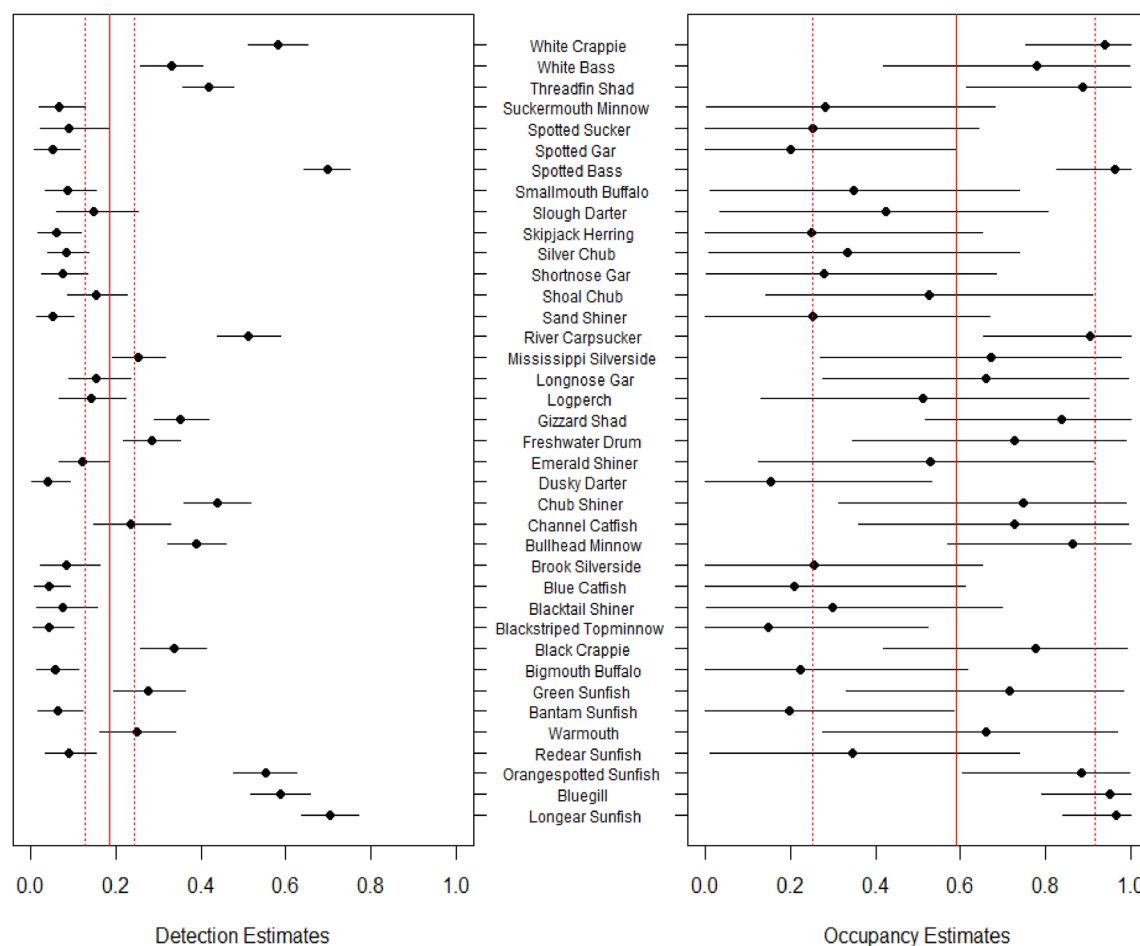


FIGURE 2 Species detection and occupancy estimates from the final occupancy model. The black points represent the median (i.e., most likely) values from the posterior distribution for each species. The black bars represent the 90% credible intervals for those species. The solid red line shows the group mean (all species) for both the detection and occupancy estimates and the dotted red lines show the 90% credible intervals for those estimates. [Color figure can be viewed at wileyonlinelibrary.com]

upstream dam (Figures 3–6). Lastly, all species had a negative occupancy relationship with deeper thalwegs and the percentage of limestone within the catchment (Figures 3–6). Although some species had the same relationship with thalweg depth, the effect size of these relationships differed. Some species (e.g., Longear Sunfish and Bantam Sunfish *Lepomis symmetricus*) had relatively weak negative relationships, whereas Warmouth *Lepomis gulosus* and Redear Sunfish *Lepomis microlophus* had stronger relationships.

Several nursery habitat relationships were species-specific (Table S6). The occupancy relationships with drainage area, segment slope, amount of LWD, W:D ratio, and seasonal median discharge were variable among species and taxonomic groups. Five species (Chub Shiner *Notropis potteri*, Gizzard Shad *Dormosa cepedianum*, Mississippi Silverside *Menidia audens*, Threadfin Shad *Dormosa petenense*, and White Bass *Morone chrysops*) were positively associated with larger drainage areas, whereas all other species were negatively related. Most juvenile species were negatively associated with LWD except for Channel Catfish *Ictalurus punctatus*, Longnose Gar *Lepisosteus osseus*, and Slough Darter *Etheostoma gracile*. Seasonal median discharge had a generally positive relationship with most juvenile

fishes; however, Longear Sunfish *Lepomis megalotis*, Orangespotted Sunfish *Lepomis humilis*, Logperch *Percina caprodes*, and Silver Chub *Macrhybopsis storeriana* had negative relationships with median discharge. The segment slope and W:D ratio were split between positive and negative relationships among all species. For example, Dusky Darter *Percina sciera* exhibited a strong negative relationship with slope, whereas Freshwater Drum *Aplodinotus grunniens* had a strong positive relationship. Moreover, Blacktail Shiner *Cyprinella venusta* had a strong negative relationship with the W:D ratio, whereas Shoal Chub *Macrhybopsis hyostoma* exhibited a strong positive relationship. Lastly, the grouping factors of segment and year accounted for the variance of 1.425 and 1.194, respectively.

4 | DISCUSSION

Our results indicate that nursery habitats in large rivers are largely context-dependent, even for closely related species. Nursery habitats in the lower Red River can generally be described as reaches containing off-channel slackwater habitat, having deep pools, with shallow

TABLE 1 Naïve estimates of occupancy in comparison to model estimated occupancy (ψ) and detection probability (ρ) for juvenile fishes in the lower Red River catchment of Oklahoma, Texas, and Arkansas, USA.

Species	Naïve ψ	ψ (LHDI–UHDI)	ρ (LHDI–UHDI)
Bantam Sunfish	0.030	0.197 (0.013–0.799)	0.064 (0.023–0.165)
Bigmouth Buffalo	0.040	0.224 (0.018–0.823)	0.058 (0.020–0.155)
Black Crappie	0.394	0.776 (0.250–0.974)	0.336 (0.249–0.436)
Blackstriped Topminnow	0.020	0.149 (0.008–0.770)	0.044 (0.011–0.151)
Blacktail Shiner	0.030	0.301 (0.025–0.848)	0.075 (0.022–0.225)
Blue Catfish	0.030	0.210 (0.015–0.815)	0.043 (0.013–0.134)
Bluegill	0.717	0.950 (0.606–0.996)	0.587 (0.501–0.668)
Brook Silverside	0.051	0.256 (0.021–0.832)	0.084 (0.029–0.215)
Bullhead Minnow	0.576	0.863 (0.349–0.987)	0.390 (0.315–0.476)
Channel Catfish	0.242	0.725 (0.186–0.970)	0.237 (0.144–0.359)
Chub Shiner	0.455	0.746 (0.147–0.973)	0.439 (0.348–0.534)
Dusky Darter	0.010	0.156 (0.009–0.766)	0.041 (0.010–0.145)
Emerald Shiner	0.172	0.528 (0.077–0.938)	0.122 (0.067–0.214)
Freshwater Drum	0.414	0.727 (0.174–0.967)	0.285 (0.211–0.374)
Gizzard Shad	0.535	0.837 (0.289–0.985)	0.351 (0.282–0.436)
Green Sunfish	0.313	0.714 (0.172–0.966)	0.276 (0.190–0.394)
Logperch	0.121	0.512 (0.083–0.929)	0.142 (0.071–0.263)
Longear Sunfish	0.646	0.964 (0.668–0.997)	0.703 (0.620–0.777)
Longnose Gar	0.232	0.661 (0.123–0.966)	0.154 (0.090–0.268)
Mississippi Silverside	0.414	0.671 (0.140–0.963)	0.254 (0.189–0.337)
Orangespotted Sunfish	0.586	0.883 (0.378–0.988)	0.553 (0.462–0.638)
Redear Sunfish	0.071	0.346 (0.038–0.880)	0.091 (0.040–0.193)
River Carpsucker	0.636	0.903 (0.418–0.991)	0.511 (0.426–0.601)
Sand Shiner	0.040	0.254 (0.020–0.849)	0.052 (0.018–0.136)
Shoal Chub	0.172	0.526 (0.084–0.925)	0.153 (0.088–0.259)
Shortnose Gar	0.071	0.278 (0.025–0.852)	0.075 (0.031–0.172)
Silver Chub	0.101	0.335 (0.032–0.867)	0.085 (0.041–0.164)
Skipjack Herring	0.020	0.251 (0.021–0.842)	0.061 (0.022–0.165)
Slough Darter	0.061	0.424 (0.051–0.908)	0.149 (0.064–0.306)
Smallmouth Buffalo	0.081	0.350 (0.035–0.875)	0.089 (0.040–0.196)
Spotted Bass	0.788	0.962 (0.645–0.997)	0.697 (0.629–0.758)
Spotted Gar	0.020	0.201 (0.013–0.801)	0.055 (0.014–0.159)
Spotted Sucker	0.040	0.254 (0.020–0.830)	0.092 (0.030–0.261)
Suckermouth Minnow	0.040	0.282 (0.024–0.849)	0.068 (0.024–0.167)
Threadfin Shad	0.646	0.886 (0.389–0.992)	0.418 (0.349–0.490)
Warmouth	0.222	0.658 (0.143–0.951)	0.251 (0.157–0.372)
White Bass	0.444	0.779 (0.235–0.977)	0.331 (0.253–0.424)
White Crappie	0.717	0.938 (0.545–0.995)	0.580 (0.498–0.663)

Note: Lower (LHDI) and upper (UHDI) 95% high-density intervals are provided.

average thalweg depths, and further away from dams with lower percentages of limestone geology. Although taxonomically similar species are often thought to use similar habitats, that is not always the case (Lowe-McConnell, 1987). For example, we found that Green Sunfish *Lepomis cyanellus* and Redear Sunfish *L. microlophus* were positively associated with wider, shallower channels, whereas Bantam Sunfish, Bluegill *Lepomis macrochirus*, Longear Sunfish, and Orangespotted Sunfish tended to occur in reaches with narrower and deeper

channels. Although these species are not of conservation concern, it demonstrates the flaws of assuming closely related species share habitat choices because they have other shared traits (e.g., body morphology and feeding strategies). Changes in channel slope also appeared to provide context dependency to nursery habitats where fishes in the genera *Ictalurus*, *Ictiobus*, *Pomoxis*, *Lepomis*, and *Dorosoma* all had species with opposing relationships with segment slope. Increased slope can lead to stronger water velocities (Gordon et al., 1992),

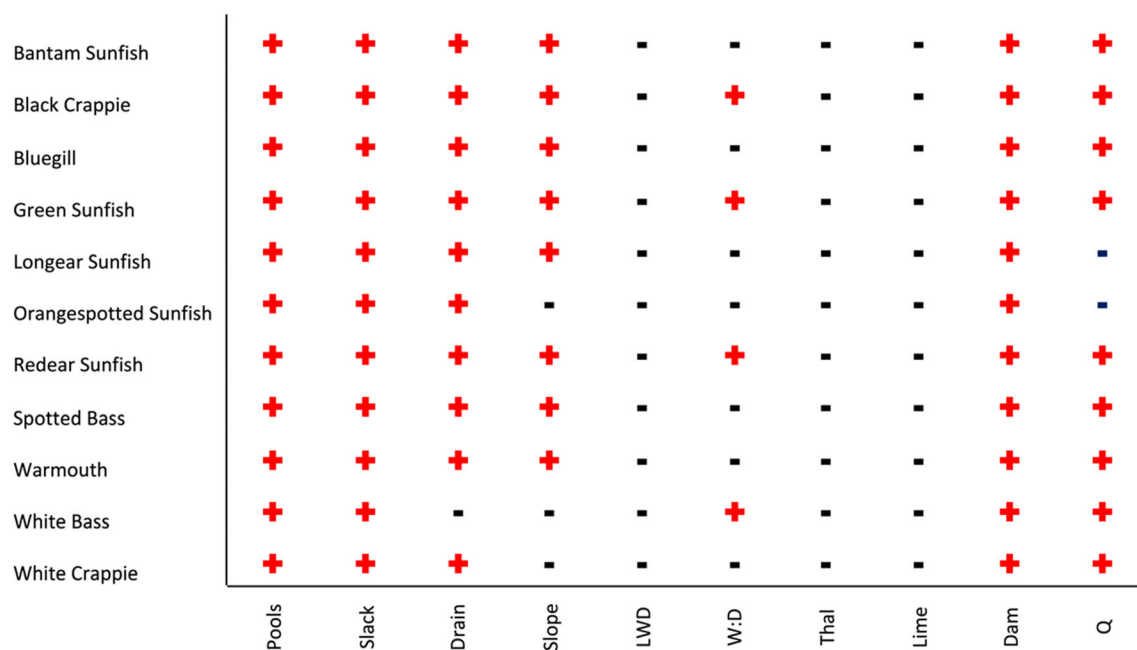


FIGURE 3 Occupancy relationships of juvenile Centrarchidae and Moronidae species. Positive relationships are indicated with a red plus sign (+). Negative relationships are indicated with a black negative sign (-). Slack is the presence of slackwater, Drain is the drainage area where low drainage area is the reference, LWD is large woody debris, W:D is the width-to-depth ratio, Thal is average thalweg depth, Lime is the percentage of limestone, Dam is the distance from the nearest upstream dam, and Q is the median discharge value. [Color figure can be viewed at [wileyonlinelibrary.com](https://onlinelibrary.wiley.com/doi/10.1002/jra.4259)]

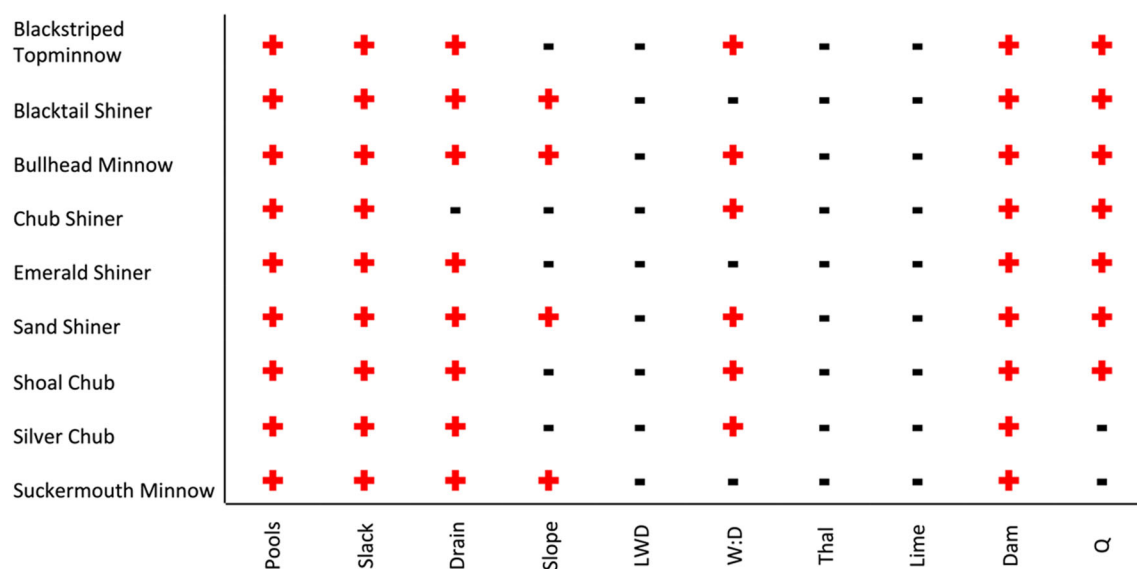


FIGURE 4 Occupancy relationships of juvenile Cyprinidae species. Positive relationships are indicated with a red plus sign (+). Negative relationships are indicated with a black negative sign (-). Slack is the presence of slackwater, Drain is the drainage area, LWD is large woody debris, W:D is the width-to-depth ratio, Thal is average thalweg depth, Lime is the percentage of limestone, Dam is the distance from the nearest upstream dam, and Q is the median discharge value. [Color figure can be viewed at [wileyonlinelibrary.com](https://onlinelibrary.wiley.com/doi/10.1002/jra.4259)]

create more heterogenous water depths (Troutman et al., 2007), and diversify the channel units within the river segment (Harvey & Bencala, 1993). Common species may be more tolerant of homogenous water depths with low water velocities (e.g., Spotted Bass and

Bluegill); however, rarer species (e.g., Skipjack Herring and Bigmouth Buffalo) may benefit from the higher water velocity that creates more diverse habitats (Marchetti & Moyle, 2001; Walters et al., 2003). Although the mechanisms for these associations are unknown, these

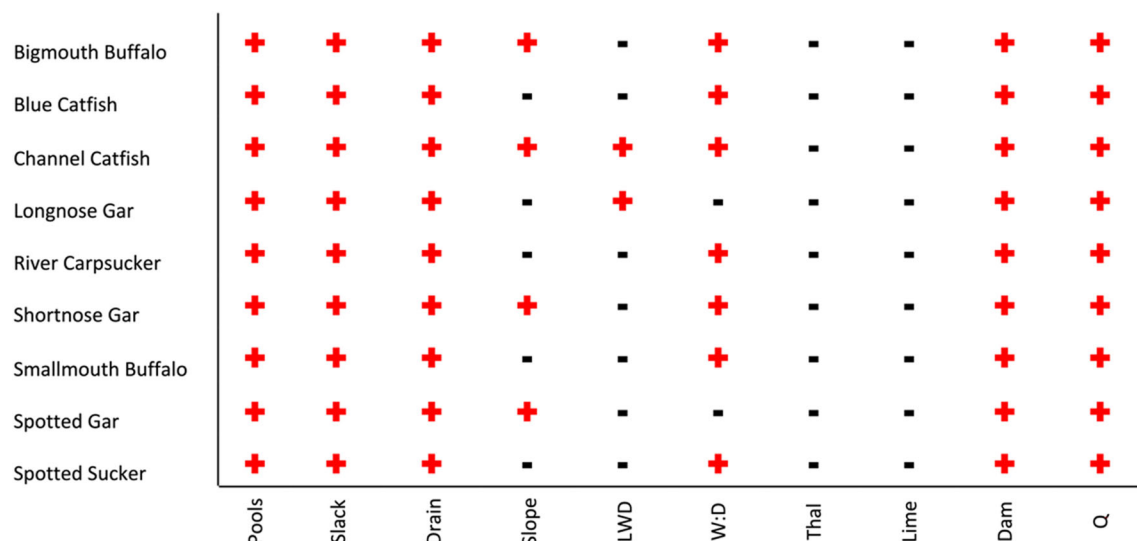


FIGURE 5 Occupancy relationships of common juvenile large river fish families Catostomidae, Ictaluridae, and Lepisosteidae species. Positive relationships are indicated with a red plus sign (+). Negative relationships are indicated with a black negative sign (-). Slack is the presence of slackwater, Drain is the drainage area, LWD is large woody debris, W:D is the width-to-depth ratio, Thal is average thalweg depth, Lime is the percentage of limestone, Dam is the distance from the nearest upstream dam, and Q is the median discharge value. [Color figure can be viewed at [wileyonlinelibrary.com](https://onlinelibrary.wiley.com/doi/10.1002/jra.4259)]

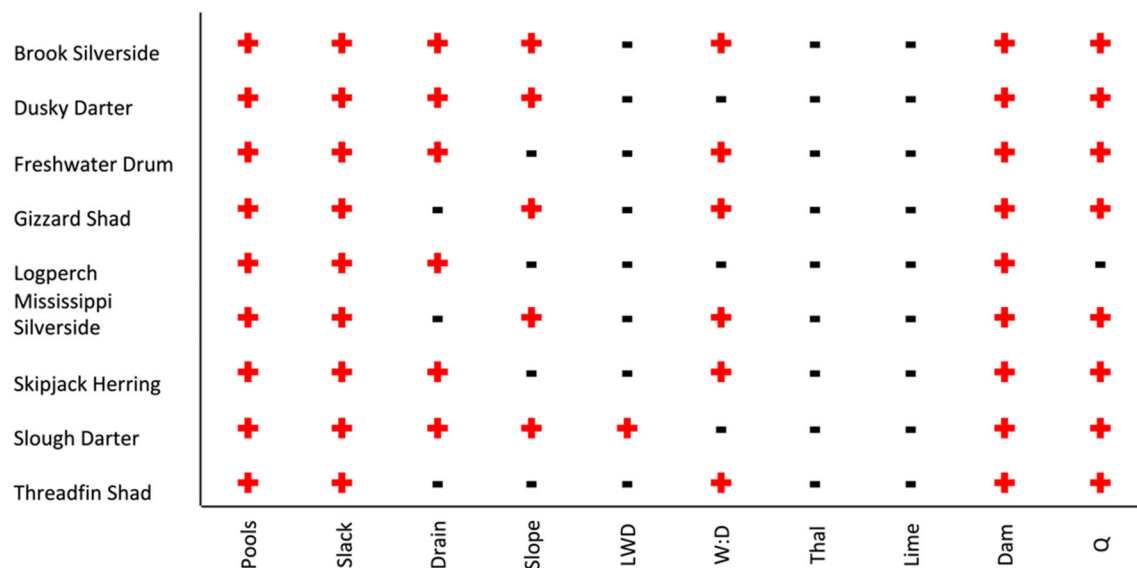


FIGURE 6 Occupancy relationships of remaining juvenile fish families Atherinidae, Clupidae, Percidae, and Sciaenidae. Positive relationships are indicated with a red plus sign (+). Negative relationships are indicated with a black negative sign (-). Slack is the presence of slackwater, Drain is the drainage area, LWD is large woody debris, W:D is the width-to-depth ratio, Thal is average thalweg depth, Lime is the percentage of limestone, Dam is the distance from the nearest upstream dam, and Q is the median discharge value. [Color figure can be viewed at [wileyonlinelibrary.com](https://onlinelibrary.wiley.com/doi/10.1002/jra.4259)]

varying relationships within closely related species indicate that river slope and W:D lead to different nursery habitats for assemblage members.

The nursery habitat in the mainstem Red River tended to be associated with species with certain reproductive traits, whereas the limited nursery habitats available in the tributaries were important to a variety of species. Fish species such as the Mississippi Silverside,

Gizzard Shad, Threadfin Shad, White Bass, and Chub Shiner were more strongly associated with the mainstem Red River, whereas all remaining sampled species (87%) were more strongly associated with the major tributaries. The five species primarily associated with the mainstem are all open substrate spawners (Balon, 1975) that scatter their eggs, although specific requirements (i.e., substrate type used and parental care) differ among the species (Frimpong &

Angermeier, 2010). Chub Shiner, for example, is likely a broadcast spawner that relies on moving water to maintain eggs and larvae in suspension during development (Worthington et al., 2018) and typically spawns over sand substrate (Lee et al., 1980). The tributaries in the lower catchment also contain some sandy substrate; however, there are more areas consisting of mud, fine gravel, silt, and rocky shoals when compared to the mainstem (Pigg, 1977), and their fragment length has been sufficiently shortened in many cases due to impoundments. Similarly, tributaries were positively associated with fish recruitment in the Missouri River of South Dakota (Pracheil et al., 2009). Tributaries tend to provide lower discharge conditions than mainstem rivers (Beckmann et al., 2005; Pracheil et al., 2009) and may function as important source populations for many assemblage members (Laub et al., 2018). The impoundments in the catchment may limit available nursery habitat in the tributaries, emphasizing the importance of the remaining free-flowing tributaries.

Although some species had similar relationships with physicochemical factors, the strength of the relationships varied among species, indicating environmental vulnerabilities for some species. Longear Sunfish, Bluegill, Orangespotted Sunfish, and Bantam Sunfish all had weak negative relationships with thalweg depth, whereas the effect size of this parameter was much stronger for Warmouth, Red-ear Sunfish, and Green Sunfish probability of occupancy. Juvenile fishes, in general, tend to be associated with shallower channels (Lobb & Orth, 1991); however, some species may be more vulnerable to channelization that would alter channel dimensions and associated habitat (Langler & Smith, 2001). Fishes that are considered stronger competitors (e.g., Spotted Bass, Longear Sunfish, and White Bass) may have weaker relationships with thalweg depth because of their ability to thrive in a variety of habitats (Edge et al., 2020; Guenther & Spacie, 2006; Travnichek et al., 1995). As another example, the occupancy of most juvenile fishes was negatively related to the percentage of LWD in the reach. Although some juvenile fishes use large woody debris for shelter in mainstem rivers (Cederholm et al., 1997; Clark et al., 2007; Roni & Quinn, 2001), LWD may also increase predation risk due to its association with piscivores (e.g., Catfishes and black basses; Daugherty & Sutton, 2005; Gocłowski et al., 2013). The variable relationships among species of the assemblage suggest that a more complex habitat can provide more benefits to diverse fish species.

Contradictory to our hypothesis, seasonal discharge was generally positively related to juvenile fish occupancy and may be related to food availability or proximity to spawning areas. During low discharge conditions, many backwaters, forewaters, and side channels were somewhat disconnected from the river or in some cases, completely dry. Higher discharge conditions facilitate enhanced connectivity between the main river channel and important slackwater habitats (Górski et al., 2013; Spurgeon et al., 2019) to provide access to important nursery grounds (Junk et al., 1989; Poff et al., 1997; Zeug & Winemiller, 2008). Higher discharge conditions can lead to lower zooplankton concentrations within river channels (Spaink et al., 1998; Thorp & Mantovani, 2005). However, zooplankton may be residing in off-channel slackwaters in higher concentrations (Vietz et al., 2013)

than in the main channel. The positive associations between juvenile fishes and both slackwater and higher discharge conditions emphasize the importance of the connectivity between the off-channel habitats and the main river channel. The ephemeral nature of off-channel nursery habitats can cause juvenile fishes to be cut off from the main river channel for a period of time (Lyon et al., 2010), presenting a unique transient definition of large river nursery habitats, one that temporally oscillates with discharge conditions. The importance of the connection between mainstem and off-channel habitats may also be due to the proximity of spawning habitat and rearing habitat (Richards et al., 1992). For example, Smallmouth Bass segregates spawning and rearing habitats where rearing habitat provides areas of lower predation risk and bioenergetic benefits (Miller & Brewer, 2020). A similar situation could be anticipated for large rivers where suitable nursery habitats are chosen due to proximity to suitable spawning areas. Lack of suitable spawning habitat may also be why distance to the nearest upstream dam is important in defining nursery habitats.

As expected, accounting for detection was important in modeling the occupancy of juvenile river fishes. Accounting for incomplete detection of riverine fishes has been indicated as important in a variety of studies (Mcmanamay et al., 2014; Reid & Haxton, 2017; Staton et al., 2022) including other large rivers (Pracheil et al., 2019; Pritt et al., 2014; Scheidegger & Bain, 1995). Warmwater fishes generally spawn during the spring and early summer in the Great Plains ecoregion (Fausch & Bestgen, 1997). As water temperatures warm, juvenile fishes grow faster (Coutant, 1976; McDowall, 1994; Robinson & Childs, 2001), become more active (Lyon et al., 2008), and achieve larger sizes (Schlosser, 1987) thereby increasing detection. In addition to temperature, we also found detection increased with increasing discharge. Although increased discharge conditions may negatively influence sampling gear efficacy in some cases (Casselman et al., 1990; Gwinn et al., 2016; Rabeni et al., 2009), these conditions also increase connectivity with important off-channel slackwater habitats (Junk et al., 1989; Nakamoto et al., 2020) and increase the availability of shallow-water habitats until some threshold is reached (Tracy-Smith et al., 2012). Moreover, increasing discharge may also prevent fishes from detecting the sampling gear (Lyon et al., 2014; Wedderburn, 2018). Our work emphasizes the need to account for incomplete detection in variable, large river environments if the goal is to adequately represent the complete assemblage.

Accounting for incomplete detection is particularly important to assess changes in distributions or occupancy over time, both of which are important for a variety of conservation and management considerations. Accounting for detection is important when surveying for smaller-bodied, rarer, and cryptic species within aquatic ecosystems (Albanese et al., 2011; Schloesser et al., 2012; Wedderburn, 2018) but may also help understand fish-habitat relationships of more common species (Guillera-Aroita, 2017; Sliwinski et al., 2016). In fact, some species are quite difficult to detect but are quite common across a catchment (Mollenhauer et al., 2022). The importance of accounting for detection when sampling for small, juvenile fishes in the lower Red River catchment is evident from the low detection rates of some species with relatively high occupancy. (e.g., Silver Chub *M. storeriana*,

Logperch *Percina carbonaria*, and Redear Sunfish *L. microlophus*). Without accounting for detection probability, occupancy estimates would have been much lower than the modeled outcome (Mackenzie et al., 2009) and relationships with the nursery habitat parameters would have been altered (see also Gerber et al., 2020). Having unbiased estimates of occupancy can be especially important with management concerns such as invasive species and climate change. With the introduction of invasive Bighead Carp *Hypophthalmichthys nobilis* and Silver Carp *Hypophthalmichthys molitrix*, concerns over changes in occupancy or condition of juvenile fishes (Schrack et al., 2003) or in assemblage structure collectively may be warranted (DeBoer et al., 2018). Having baseline data on juvenile assemblages and habitat associations are important for monitoring changes in these populations over time and evaluating management actions.

Increasing either the quality or quantity of nursery habitat is becoming a common goal of larger river management; thus, understanding species-specific needs and providing information on surrogate species options may be useful. Low-velocity habitats are used by a variety of riverine fishes (Scheidegger & Bain, 1995). Flows that are associated with the formation of complex littoral zones, connections with backwater habitat, and the formation of sandbars are considered as restoration options in large rivers. For example, building sandbars, reconnecting floodplains, and providing low-water nursery habitats were goals of consideration for flow alternatives on the lower Missouri River (U.S. Fish and Wildlife Service, 2000). If the goal of species management is to maintain or increase recruitment of the species within an assemblage, then understanding the differences between coarse-scale and species-specific nursery habitats would be beneficial for understanding how habitat benefitting one species may be detrimental to another target species. Agencies sometimes select surrogate species for management objectives when little is known about a species of interest (Caro & O'Doherty, 1999), often using species within the same genera as surrogates. However, species-specific relationships with environmental conditions and coarse-scale landscape constraints indicate that the selection of surrogate species will, in some cases, be misguided. There is a trade-off between the management of individual species of concern and indistinct practices that may apply to broader fish assemblages (Wiens et al., 2008). Traditionally defined nursery habitats can be used for assessing finer scale habitat (e.g., shallow reaches containing pools and slackwaters; Fremling et al., 1989; Humphries et al., 2006); however, proper evaluations for habitat management actions benefit from an understanding of the coarse-scale habitat conditions that shape the context dependency among species (Allan, 2004; Elith & Leathwick, 2009). Moreover, important landscape constraints (e.g., position within the stream network and gradients) may act as filters (Poff, 1997) for the nursery habitats available to a species and thus, worth consideration when defining nursery habitats. Having a broader view of the physicochemical conditions that define nursery habitats for species of interest is important when devising conservation and restoration practices and when selecting surrogate species for other management objectives.

ACKNOWLEDGMENTS

This research is a contribution of the Alabama Cooperative Fish and Wildlife Research Unit (United States Geological Survey, Alabama Department of Conservation and Natural Resources, Auburn University, and Wildlife Management Institute cooperating). Funding was provided by the Oklahoma Department of Wildlife Conservation, Arkansas Game and Fish Commission, Texas Parks and Wildlife, and the U.S. Fish and Wildlife Service. We thank Ben Birdsall, Jordan Ramey, Kyle Rempe, Trevor Bannister, Tyler Murray, Daniel Paulson, and Eli Wilson for technical and field assistance, and Steve Sammons for his comments on an early manuscript draft. Special thanks to Monica McGarity, Angie Rodgers, Jim Burroughs, and Matthew Horton for grant assistance and project feedback. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government. This study was performed under the auspices of Auburn University Animal Care and Use protocol #2020-380C. There is no conflict of interest declared in this article.

DATA AVAILABILITY STATEMENT

The R code used for analyses are included in the supplementary materials. All geospatial data were collected from publicly available sources. Fish data may be obtained via a request to Jim Burroughs (jim.burroughs@odwc.ok.gov).

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SUPPORTING INFORMATION

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How to cite this article: Ramsey, P. Q., Dattilo, J., DeVries, D. R., & Brewer, S. K. (2024). Evaluating the context dependency of large river nursery habitats for fishes in the lower Red River catchment. *River Research and Applications*, 40(5), 761–779. <https://doi.org/10.1002/rra.4259>