

MOVEMENT, HABITAT USE, AND POPULATION
DYNAMICS OF BLUE SUCKER IN THE SOUTHERN
GREAT PLAINS

By

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Abstract: Freshwater migratory fishes are among the most imperiled taxa in the world. However, mobile life histories make the study and conservation of migratory fishes challenging. The undefined scope of movement and habitat use often limit conservation efforts for the species. Fishes occupying large rivers are inherently difficult to capture and problematic to evaluate. Blue Sucker *Cyprinella elongatus* is an emblematic large-river migratory fish with a broad spatial distribution and considered a species of conservation concern in North America. I evaluated the movement patterns, habitat use, population dynamics, and the effect of variable flow and temperature on the recruitment and somatic growth of Blue Suckers. I determined that both resident and migratory populations exist within my study area. Blue Suckers were more likely to return to spawn during wet years and displayed spawning-site fidelity more frequently than they strayed. Blue Suckers that strayed, most often selected the only undammed tributary in the system. Additionally, I determined that Blue Suckers were most likely to spawn in conjunction with high flows that occurred in the late winter and early spring. I found that Blue Suckers were most likely to occur near riffles in the tributaries during the spawning season. My assessment of population dynamics of Blue Suckers in two flow-regulated streams indicated that the population was stable at one location and declining at the other location. In my assessment of the effects of flow and temperature on the recruitment and somatic growth of Blue Sucker, I estimated that growing seasons with high flows followed by declining flows resulted in strong recruitment the following spring. This indicates that summer conditions in the Red River were important for gonadal growth, although I did not detect a relationship between environmental conditions and somatic growth. Finally, I found a relationship between conditions in the undammed tributary and recruitment in the Red River. This indicates that preserving the natural flow of that tributary and its connectivity with the Red River is important to the conservation of Blue Sucker.

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CHAPTER I

INTRODUCTION

Despite the importance of rivers to human civilization, human activities have negatively affected most of the world's rivers. In much of Europe, Asia, and the U.S. and Mexico the threat to human water security and biodiversity is indexed at > 75%, indicating a high incidence of known stressors to freshwater resources (Vörösmarty *et al.*, 2010). The high incidence of threat to rivers is often a result of watershed disturbances, like agriculture and urbanization, working in unison with nonpoint source pollution (Vörösmarty *et al.*, 2010). Agriculture and urbanization in watersheds can result in excessive nutrient loads, sedimentation, organic matter, and pesticides (Karr & Schlosser, 1978; Cooper, 1993; Pusey & Arthington, 2003). An increase in impervious surface area is also associated with urbanization (Vörösmarty *et al.*, 2010). The inability of water to infiltrate the soils results in larger flow pulses and reduced base flows, thereby altering instream habitat (Barbec *et al.*, 2002). Water resource development such as dams, and others that result in river fragmentation, are perhaps the greatest stressor biodiversity (Vörösmarty *et al.*, 2010). In addition to fragmentation of riverine ecosystems (Stanford *et al.*, 1988), dams affect instream habitat by degrading water quality (Olden & Naimen, 2010) and disrupting natural flow (Poff *et al.*, 1997), thermal (Olden & Naimen, 2010), and sediment regimes (Wohl *et al.*, 2015). Given growing demand for freshwater and the relationship between water security and biodiversity, it is important that scientists work to understand and mitigate the effects of human civilization on the world's rivers.

Freshwater ecosystems account for a disproportionately large amount of the Earth's biodiversity, and migratory fishes in rivers account for a significant portion of threatened biodiversity. Despite covering < 1% of the Earth's surface, freshwater ecosystems are home to 25% of all vertebrate species and 48% of all fishes (Eschmeyer & Fong, 2013). Habitat loss and degradation is one of five major threats to freshwater fishes (Dudgeon *et al.*, 2006; Arthington *et al.*, 2016), but basic understanding of the habitat attributes required for riverine fishes is often lacking (Cooke *et al.*, 2012). The flow regime in rivers is a critical part of fish habitat and mitigating the effects of flow alteration has become a focus of freshwater fisheries biologists (Poff *et al.*, 1997; Jager & Smith, 2008; Acreman *et al.*, 2014). Because migratory fishes require a large expanse of habitat and a predictable flow regime to fulfill their life histories, dams and flow alterations are a major threat to migratory fish conservation in freshwater ecosystems (Lucas & Baras, 2001). Nearly 10% of the fishes of conservation concern in North America are widespread migratory fishes (Jelks *et al.*, 2008). The loss of migratory species in river networks can negatively affect narrow-ranging endemic fishes that account for \approx 80% of freshwater species of concern (Pringle, 1997; Jelks *et al.*, 2008). Migratory fishes can provide critical links within local food webs that sustain the more sedentary upstream ecosystems (Winemiller & Jepsen, 1998).

Migratory fishes provide ecosystem subsidies and services that are important to the function of communities, and in North America, catostomids are most likely to fill that ecological niche (Flecker *et al.*, 2010). Nutrient subsidies are most apparent in the Pacific Northwest where Pacific Salmon *Oncorhynchus* spp. transport marine derived nutrients inland (Flecker *et al.*, 2010). In coastal streams, the deposition of marine-derived nutrients, in the form of semelparous salmonid carcasses, significantly increases the growth of resident fish species and the next generation of Pacific salmonids (Wipfli *et al.*, 2003). These subsidies occur in iteroparous freshwater fishes as well (*e.g.*, *Semaprochilodus* spp., Winemiller & Jepsen, 2004; Flecker *et al.*, 2010). In the temperate regions of North America, catostomids have the greatest potential for providing nutrient subsidies (Cooke *et al.*, 2005; Flecker *et al.*, 2010). Most catostomids undergo upstream spawning migrations into tributaries during the spring (Page & Johnston, 1990). In addition to deposition of carcasses and young-of-year, catostomids are a likely host for freshwater mussel glochidia and may serve to aid upstream recolonization by mussels (O'Dee & Watters, 2000). Despite their ecological importance, catostomids have historically been labeled a "trash fish" and little effort has been devoted to understanding their life histories (Cooke *et al.*, 2005).

Many riverine fishes are migratory and travel to different locations to complete their life history, creating difficulties in identifying manageable stocks (Lucas & Baras, 2001). A fisheries stock or population includes a group of fish that share a discrete gene pool that is distinguishable from other members of the same species (MacLean & Evans, 1981). Migratory fishes complicate the delineation of stocks by spawning with individuals outside of their typical home range, potentially creating a widespread, ambiguous gene pool. Identifying the spawning locations of migratory fishes is useful to define stocks or discrete populations if the species displays a high rate of homing (MacLean & Evans, 1981). However, spawning site fidelity rates are unknown for many species, and the scale of migration patterns is often underestimated or unknown, further complicating the conservation of riverine fishes (Cooke *et al.*, 2012).

Understanding basic life history and movement patterns of riverine fishes allows for proper population assessments, and predictions of the effects of river changes on populations. Habitat and instream flows are intertwined factors, critical to the persistence of fishes in lotic environments (Arthington *et al.*, 2016). Fish migrate to access spawning, foraging, and refuge habitats, and instream flows provide migration cues and dictate the suitability of habitats at critical life stages (Poff *et al.*, 1997; Lucas & Baras, 2001). Because of the importance of those factors, successful conservation efforts require managers to understand the relationship of native species with habitat and instream flows (Cooke *et al.*, 2012). Habitat and instream flow information paired with knowledge of migration patterns allows managers to protect and restore habitat, and in streams with regulated flow, create a suitable flow regime to allow the fulfillment of life-history requirements (Acreman *et al.*, 2014; Brewer *et al.*, 2016). Further, understanding these relationships allows managers to more accurately evaluate the potential effects of proposed development projects (Cooke *et al.*, 2012).

Blue Sucker *Cyprinostomus elongatus* is a large river catostomid that occupies much of North America, but the life history of the species has only recently been investigated. Blue Sucker is native to the Mississippi River basin and considered vulnerable to extirpation in North America (Jelks *et al.*, 2008). Blue Sucker was a common food source in the central U.S. during the 19th century, but was nearly fished to extinction by the early 20th century (Coker, 1930). Throughout the 20th century, Blue Sucker was considered imperiled and by the end of the century Blue Sucker was being considered for listing on the threatened and endangered species list (Burr & Mayden, 1999). Burr and Mayden (1999) suggested that the perceived decline Blue Sucker was due to misunderstanding seasonal migration patterns

and life history of the species. In the 21st century, knowledge of Blue Sucker life history began to grow, and the species is now recognized as a potamodromous, large-river obligate. Blue Sucker inhabits large rivers the majority of the year and migrates to tributaries to spawn in the late winter and spring (Neely *et al.*, 2009). Annual Blue Sucker migrations may exceed 300 km (Neely *et al.*, 2009), or remain in a 3-km reach (Oliver *et al.*, 2017). Evidence of homing has recently been reported for Blue Suckers, but the rates of homing and straying to spawning locations remain unknown (Lyons *et al.*, 2016; Adams *et al.*, 2017). Habitat and environmental conditions associated with spawning Blue Sucker were described in the 1980's (Rupprecht & Jahn, 1980; Moss *et al.*, 1983), but additional studies have often been opportunistic and have increased uncertainty around Blue Sucker life history.

Although investigations of Blue Sucker have recently become more common, conservation of the species remains limited by a paucity of information. The conservation status of Blue Sucker is variable throughout their geographic distribution, but the species is of greatest concern along the periphery of the Mississippi River catchment (NatureServe, 2018). Stock assessments provide a benchmark for the conservation of fisheries stocks (Allen & Hightower, 2010), but I am only aware of one thorough stock assessment of Blue Sucker. The assessment was conducted in the Wabash River, IN, and led to IN being the only state where Blue Sucker is considered stable (Bacula *et al.*, 2009; NatureServe, 2018). Additional studies on Blue Sucker are warranted, because life history attributes are variable across the distribution (Burr & Mayden, 1999; Bacula *et al.*, 2009), and a solid understanding of basic life history attributes is among the major limitations of riverine fish conservation (Cooke *et al.*, 2012).

The goal of my dissertation was to improve the knowledge of Blue Sucker ecology, thereby providing information on how to improve monitoring and management efforts for the species.

I created four objectives to address five of the ten major factors limiting riverine fish conservation outlined by Cooke *et al.* (2012).

My four objectives were:

1. Determine the coarse-scale movement patterns and spawning site fidelity of adult Blue Sucker.
2. Determine fine-scale habitat use of Blue Sucker and timing of migration in major tributaries during the spawning season.

3. Determine population dynamics and estimate the growth trajectory of the Blue Sucker population.
4. Investigate the influence of environmental conditions on the recruitment and annual growth of individual Blue Suckers.

The first objective addressed movement at the riverscape scale to determine the temporal and spatial scope of Blue Sucker habitat use and aid in the identification of manageable populations. The second objective addressed Blue Sucker movement and habitat use at a finer temporal and spatial scale by focusing on the spawning season. I focused on the spawning season because the loss of spawning habitat or inability to access spawning habitat is commonly attributed to failed recruitment and the decline of migratory fishes. The third objective built upon the spatial dynamics of Blue Sucker populations observed in the first objective. I determined that Blue Suckers in the Red River potentially belong to separate populations, and evaluated vital statistics and population trajectory of two populations. The fourth objective investigates the past effects of environmental variability on the current age structure of the two populations evaluated in objective 3. Additionally, I generated hypotheses about spawning and recruitment from observations made in the previous objectives, and tested in objective 4. A chapter is devoted to each objective, and a final chapter is included to summarize my findings and the management implications of my research.

CHAPTER II

SEASONAL MOVEMENTS AND FIDELITY

INTRODUCTION

The North American Great Plains is home to several large river migratory fishes; however, many of these fishes are at an increased risk of extinction due to human landscape modifications (Fausch *et al.*, 2002). In particular, large river fishes have adapted their reproductive strategies to coincide with aspects of the natural flow regime; however, dams and water diversions have extensively altered these flow patterns (Poff *et al.*, 1997; Lucas & Baras, 2001; Poff *et al.*, 2007; Carlisle *et al.*, 2011). Seasonal flow pulses often serve as cues for both local (*e.g.*, deep pool to gravel shoal) and extensive (*e.g.*, estuary to river) spawning migrations (Lucas & Baras, 2001). Dams result in altered flow pulses rather than natural pulses driven by the climate (Poff *et al.*, 2007). The timing of unnatural flow pulses may not coincide with the timing of fish reproduction and can result in missed spawning cues and failed reproduction (Young *et al.*, 2011). High flows promote habitat connectivity and create suitable habitat for egg and larvae ontogeny (Lucas & Baras, 2001; Young *et al.*, 2011). Flow suppression contributes to the loss of suitable habitat for native species (Probst & Gido, 2004) and allows the encroachment of invasive woody vegetation (Stromberg *et al.*, 2007). Further, many species display natal site fidelity and the loss of their annual cues or suitable natal habitat can result in population declines (Pringle, 2001; Jansson *et al.*, 2007; Kiffney *et al.*, 2009).

The reproductive strategy of long-lived species generally puts them at a greater risk of extinction in altered environments. Large-river migratory fishes typically display a periodic life-history strategy and are large, highly fecund, long lived, and depend on predictable environmental conditions (*i.e.*, seasonal flooding) that are favorable for reproduction (Winemiller & Rose, 1992). Whenever favorable environmental conditions do not occur, long-lived periodic strategist may wait several years for favorable spawning conditions (*e.g.*, Cui-*ui Chasmistes cujus*, Scopettone *et al.*, 2000; Alligator Gar *Atractosteus spatula*, Buckmeier *et al.*, 2017). The periodic reproductive strategy is beneficial during drought conditions; however, in an altered environment, the frequency of favorable environmental conditions may be hindered (Lucas & Baras, 2001; Poff *et al.*, 2007). Conversely, opportunistic strategists are relatively short-lived and must be adaptable to highly variable environmental conditions making them generally more tolerant of human landscape alteration (*e.g.*, Red Shiner *Cyprinella lutrensis*, Vives, 1993; Brassy Minnow *Hybognathus hankinsoni*, Falke *et al.*, 2010). Periodic and opportunistic reproductive strategies represent extreme ends of a continuum rather than discrete strategies, and the majority of fishes display varying levels of multiple spawning strategies (Winemiller & Rose, 1992).

Making predictable use of both habitats and environmental conditions where an individual has previously spawned successfully can be beneficial to a population; however, displaying plasticity in life-history strategies is also advantageous in altered environments. Homing is an extreme example of site fidelity that has likely evolved because it increases the chances of survival and reproduction in migratory fishes (Gross, 1988; Thurow, 2016). However, migratory fishes that are rigid in their life-history strategies have limited adaptability in altered environments. For example, the loss of suitable habitat for semelparous Pacific salmonids is a well-documented population stressor (Keefer & Caudill, 2014). Plastic migratory fishes are not only able to persist in altered environments, but are also good colonizers. The migratory life-history combined with opportunistic reproductive strategy of Bighead Carp *Hypophthalmichthys nobilis*, is attributed to the rapid colonization of streams throughout North America (Coulter *et al.*, 2013). Populations of migratory species typically comprise individuals that may home, stray, or are sedentary during the spawning season (Rodriguez, 2002). Biologists often focus on members of the population who home to familiar reproductive habitat, because these individuals reveal specific areas that are important to the population (Lucas & Baras, 2001). However, those that stray or do not migrate, but are opportunistic in their

reproductive strategy can support the population when conditions are less than favorable, be a source of gene flow among populations, and allow the population to adapt to alterations in the environment (Lucas & Baras, 2001).

The objective of my study was to determine coarse-scale movement patterns of Blue Sucker *Cycleptus elongatus* Lesueur, 1817 during the spawning and non-spawning seasons. At the riverscape scale, I was specifically interested in movement among 1) different major tributaries of the Red River, 2) a 20-km tailwater-influenced reach of the Red River, and 3) the downstream portion of the Red River (*i.e.*, my unobservable state, see methods). Although I assumed Blue Sucker are able to move freely across the extensive free-flowing section of the mainstem Red River, I could not logistically track this area on regular tracking events. Rather, I quantified the probability of acoustically-tagged Blue Suckers returning to major tributaries to spawn each spring, and assessed spawning-site fidelity over three consecutive spawning seasons using a multistate mark-recapture analysis. I selected the tailwater-influenced reach of the Red River and three major tributaries because I hypothesized the areas had suitable spawning habitat for Blue Suckers and were primary migration pathways based on point sampling conducted by agencies and angler anecdotal observations. The goal of this study was to provide a riverscape-scale perspective on Blue Sucker migrations so that the population could be better evaluated (*i.e.*, where to sample and when) and provide key information on where and when fish migrated to spawning locations to ensure appropriate and spatially-explicit conservation and management actions could be developed for this elusive species.

METHODS

STUDY AREA

I conducted my study over four extensive reaches in the humid (114 -140 cm of rain annually, Woods, 2005) lower Red River catchment of Oklahoma (Fig. 1.1). The lower Red River begins downstream of Denison Dam, a hydropower dam that impounds Lake Texoma, and flows 333 km along the Oklahoma-Texas border before entering Arkansas. The mainstem Red River below Denison Dam remains free flowing until it reaches a series of locks and dams in Louisiana. My four tracking reaches were: the lower extent of the three major tributaries and a section of the mainstem ~20 km downriver of Denison Dam (hereafter referred to as the tailwater reach, Fig. 1.1). Tagged Blue Sucker were able to move above the upper extent of my tracking reaches

in the Blue and Muddy Boggy rivers, but I was only concerned about their entering or leaving the tributary. Approximately 45 km above my tracking extent, Blue Sucker movement was obstructed by a low head dam on the Blue River, whereas the mainstem Muddy Boggy River is free flowing to the headwaters with exception of a small dam on one of the minor tributaries. The lithology of the tributaries below the dams and the mainstem river is primarily alluvium (Woods, 2005), and consequently, the channel substrate is typically sand with few exceptions where bedrock is exposed or patches of gravelly riffles downriver of Dennison Dam.

Discharge data were available at four U.S. Geological Survey gaging stations (07331600, 07335500, 07332500, and 07335300, Fig. 1.1) and from the U.S. Army Corps of Engineers (<http://www.swt-wc.usace.army.mil/HUGO.lakepage.html>), and water temperature data were available at two of the USGS stations (07335500; 0733160). The lower Red River basin experienced severe drought (2011-2014) followed by a wet period in 2015 (climate.ok.gov; Fig. 1.2). Discharge at Arthur City was low at the time of tagging (mean 134 ± 172 SD m^3/s , February – April 2015), but was very high the following summer (peak = $6315 \text{ m}^3/\text{s}$ May 31, 2015). In 2016, a discharge pulse in late February ($>283 \text{ m}^3/\text{s}$, Arthur City gage) followed by a flood in mid-March ($> 1130 \text{ m}^3/\text{s}$) delayed my tracking efforts by two weeks. In early April, discharge dropped to $113 \text{ m}^3/\text{s}$ before increasing to $1130 \text{ m}^3/\text{s}$ and discharge remained high for the rest of the early season 2016 (Fig. 1.2). Conversely, in early season 2017, a small discharge pulse in March ($283 \text{ m}^3/\text{s}$) was followed by low discharge ($\approx 85 \text{ m}^3/\text{s}$) through early April. Water temperature at Denison and Arthur City were very similar in rates of change, but temperature at Denison was much cooler than at Arthur City during the springs and summers, but warmer during the winter (Fig. 1.3).

EXPERIMENTAL DESIGN

My study design followed a multistate capture-recapture Cormack-Jolly-Seber framework to estimate transition probabilities among major tributaries and the mainstem Red River (Cormack, 1964; Jolly, 1965; Seber, 1965). Briefly, the Cormack-Jolly-Seber framework is a capture-recapture approach that occurs simultaneously at multiple locations and does not assume a closed population. The multiple locations within the framework are termed states, and the capture-recapture data are used to estimate the probability of individuals moving among states between discrete time intervals, given the

probabilities of survival between time intervals and detection within time intervals (Brownie *et al.*, 1993; Schwarz *et al.*, 1996).

My tracking locations and seasons provided the framework for my states, where I considered three states relative to each individual fish. State A represented the reach where any individual was tagged, state B corresponded to the remaining three tracking reaches (*i.e.*, states A and B varied by individual), and state C was the unobservable downstream extent of the Red River (*i.e.*, the river segment that was not tracked). I hypothesized that states A and B would correspond to spawning habitat whereas state C would not. Two seasons were considered in my study: an 'early season' (Feb – May) that generally corresponded to the perceived spawning time, and a 'late season' (Jun – Jan) that represented post spawn and over-winter periods (Moss *et al.*, 1983; Vokoun *et al.*, 2003; Neely *et al.*, 2009). Together, the spatial and temporal delineations allowed me to observe movement in the context of anticipated spawning. I anticipated a high probability of Blue Suckers moving to state C during the late season. Therefore, observation of individuals in state A during the early season would reflect spawning site fidelity, whereas early season observations in state B would reflect straying. However, if the probability of Blue Suckers moving to state C during the late season was low, I assumed the Blue Suckers did not leave the spawning tributary.

I designed my study to meet the assumptions of a multistate Cormack-Jolly-Seber model. I used acoustic tags that allowed individual identification. My study was conducted within the expected life span of the tags (28 months of a 36-month lifespan). Lastly, I internally implanted the acoustic tags anticipating high tag retention (Welch *et al.*, 2007; Adams *et al.*, 2012; Carrera-Garcia *et al.*, 2017). Because subsequent detections were dependent on acoustic signals and allowed me to observe the fish without capturing and handling, I was not concerned with heterogeneity of individual behavior influencing detection (*e.g.*, trap-happy or trap-shy individuals). The assumption that all individuals would transition between states at the beginning and end of the early season required that detection in multiple states within a season was not possible. This assumption was easily met if fish transitioned to unobservable state C. However, if a fish moved into multiple tributaries during the same season, it would violate my assumption. The latter only occurred on one occasion (see results).

TAGGING AND TRACKING

In Feb-April 2015, I captured and tagged 119 Blue Suckers > 500-mm total length (TL) at four locations within the study area (Fig. 1.1). My capture and tagging protocols were approved by the Animal Care and Use Committee at Oklahoma State University (AG-14-21, Feb. 6, 2015). Sampling and tagging were conducted when Blue Suckers were expected to move into the tributaries to spawn (Moss, 1983; Vokoun *et al.*, 2003; Neely *et al.*, 2009; ODWC, Unpublished data) and thus, were more susceptible to capture. I used boat electrofishing (5.0 GPP Smith-Root, Vancouver, WA) to capture Blue Suckers in the Blue, Muddy Boggy, Kiamichi, and Red rivers. I used pulsed, direct current and adjusted the power output according to water conductivity based on recommendations of Miranda (2009). Water conductivities ranged 50 – 2000 μS , and I typically used 60hz frequency except in the Kiamichi River where low conductivities (< 75 μS) occasionally required 120hz to reach the target power of 2500 – 4000 watts. Blue Suckers were typically captured by drifting through swift-water habitats while electrofishing. Upon capture, Blue Suckers were placed in a 250-L tub on the stream bank and I changed the water approximately every 15 – 20 min to reduce holding stress. I anesthetized one Blue Sucker at a time with a 22 – 26 mg/L solution of Aquí-S 20E (New Zealand LTE. Lower Hutt, New Zealand). Once the fish lost equilibrium, I measured the total length (TL, 1 mm), weight (0.01 kg), and placed the fish ventral side up on a V-shaped cradle. Freshwater from the river was piped over the fish's gills during the tagging process. A 3 – 5-cm incision was made to the right of the midline of the ventral side of the fish, posterior to the left pelvic fin and anterior to the anal vent. An acoustic transmitter (CT-05-36-I, Sonotronic Inc. Tucson, AZ) was inserted into the abdominal cavity and 2 – 5 interrupted absorbable sutures were used to close the incision (2-0 PDO, 3/8 reverse cutting needle; Unify, AD Surgical, Sunnyvale, CA, USA). Acoustic tag weight was 10 g in water, and the smallest fish tagged was 700 g making the tag burden $\leq 1.4\%$ of body weight. I determined the sex of each fish based on macroscopic observation of the gonads. Following surgery and morphometric measurements, I placed fish in a freshwater recovery tank. I released each fish back to the collection reach after it resumed normal swimming (approximately 30 min). In addition to frequent water changes, I processed fish as rapidly as possible and avoided exposing holding tanks to direct sunlight (mean time in surgery = $6:56 \pm 1:29$ min SD). I minimized stress on the fish because accumulated stress in an individual can lead to increased susceptibility to infection and mortality (Adams *et al.*, 2012).

Passive receivers were placed at four locations within the study area to record fish moving into and out of my tracking reaches. I anchored passive submersible ultrasonic receivers (SUR; SUR-03, Sonotronic Inc. Tucson, AZ) in each of the three major tributaries near the confluence with the Red River, and in the Red River approximately 20 km downriver of Denison Dam (two at each location). Two SURs were placed at each location to indicate the direction the fish was traveling. However, detection by a single SUR was imperfect and including a second SUR at each location increased the probability of detecting a tagged fish (see Appendix A). I lost all but one SUR (Kiamichi River) in a 100-yr flood event in May 2015. In winter 2015, I replaced all of the SURs in the tributaries and in the tailwater reach and I maintained receivers at three of the four locations until spring 2017. The SURs placed in the Blue River were not functioning upon final retrieval.

To supplement my passive tracking, I actively tracked by boat and canoe in 2016 and 2017 to determine fish movement into the major tributaries and the tailwater reach. The tailwater reach and the major tributaries were tracked approximately once weekly in Feb – May 2016 and 2017 when the majority of spawning activity was historically observed by state agency personnel. Because spawning movements were of interest to my study, active tracking effort was greater in the early season than in the late season. Low discharge during the late season reduced the navigability of the streams, resulting in a greater dependence on SURs. However, I tracked each reach one time in July, September and October 2016. Active tracking comprised towing a hydrophone (TH-2, Sonotronics Inc.) behind my watercraft at 7 – 9 km/h (*i.e.*, slightly faster than the current) while I scanned acoustic frequencies (USR-08, Sonotronic Inc.). Upon identification of a tagged Blue Sucker, I recorded a GPS location and the date.

MOVEMENT ANALYSIS

I conducted my analysis using Program Mark (Cooch & White, 2016) to test three specific hypotheses regarding transition probabilities (Table 2.1). First, I hypothesized that male and female Blue Suckers had transition probabilities that reflected differences in energy investment into reproduction by the two sexes (hereafter referred to as ‘sex hypothesis’). Specifically, I expected females to display higher fidelity to spawning reaches because females often invest more energy into reproduction and fidelity is attributed to greater fitness compared to straying (Moyle & Cech, 1996; Lucas & Baras, 2001). Second, I hypothesized that fish tagged in reaches influenced by a dam release

were more likely to return (hereafter referred to as 'dam-release' hypothesis). The dam releases in the Red River tailwater and Kiamichi River would create multiple flood pulses and sustained discharge, providing a possible cue for spawning migration (Lucas & Baras, 2001); whereas, the other tributaries would be subjected to natural weather patterns and less predictable over finer temporal scales. Finally, I hypothesized that Blue Suckers tagged below Denison Dam displayed different movement patterns when compared to the major tributaries of the Red River (hereafter referred to as 'tributary hypothesis'). Fish typically migrate to spawn because adult foraging and refuge habitats do not match that of the young-of-year life stages (Lucas & Baras, 2001). However, the tailwater reach was not a separate tributary from the Red River and possessed the clean, coarse substrate required for spawning and ample sandy habitat for Blue Sucker forage items (Moss *et al.*, 1983), hence my hypothesis was that Blue Suckers might behave differently in the tailwater reach compared to the major tributaries.

I constructed four models to reflect each of my three hypotheses and a null hypothesis. I split the telemetered individuals into eight groups that could be combined to create my hypothesized models: males or females tagged in each of four sites (the tailwater reach, Blue, Muddy Boggy, or Kiamichi rivers, *i.e.*, two groups from each tributary) (Table 2.2). I first created the null model by combining all eight groups into a single model to obtain transition probabilities. Next, to compare the sex hypothesis, I combined my groups into male or female groups allowing comparison of transition probabilities between the two sexes. Similarly, the dam-release hypothesis was investigated by combining groups from reaches with dam releases (Red or Kiamichi rivers) or groups from reaches where flow is more weather-dependent (Blue and Muddy Boggy rivers). Finally, I tested the tributary hypothesis by combining groups from the tailwater reach or the major tributaries. I did not include sex and either of the location groups in the same model because my data were not adequate to estimate the 50+ parameters required to model four groups. Differences in transition probabilities between location groups could confound the effect of differences between sexes within location groups. Therefore, I excluded individuals from one of the location groups, allowing me to test for differences between the sexes within the location group of interest.

Because I was interested in movement, I focused my model on transition probabilities. I estimated survival as constant over time ($\phi_{.}$) (where the "." subscript denotes constant over time) and detection probability as dependent on time (P_t) (where the "t" subscript indicates time dependence) to prevent confounding parameter estimates

(Leberon *et al.*, 1992). Detection probability was inherently time dependent because my tracking effort varied among seasons. I conducted weekly-active tracking efforts in the early season of both years, whereas, active tracking effort was bimonthly during the late seasons. The loss of all but one SUR and only tracking the tailwater reach during the late-2015 season meant that tracking was extremely limited during the non-spawning season. Additionally, because state A or B was relative to each individual and not discrete physical locations, I did not estimate P_t separately for the two states. So, $P_t^{A \cup B}$ (where superscripts “A” and “B” indicate the states attributed to the parameter) represented the mean detection probabilities for all tracking reaches within each season. I fixed detection probability of the unobserved state C as $P_t^C = 0$. Transition probabilities ($\psi_t^{S:R}$) (where superscript “S” is the occupied state, and the fish is transitioning into state “R”) were estimated for each season and pair of states, excluding transitions directly between A and B ($\psi_t^{A:C}$, $\psi_t^{B:C}$, $\psi_t^{C:A}$, $\psi_t^{C:B}$). I assumed that Blue Suckers would spend a season in state C before moving between states A and B so, $\psi_t^{A:B} = \psi_t^{B:A} = 0$. This assumption was based on my data, where only one fish transitioned between A and B making the estimation of these parameters impossible (see results). Finally, because all Blue Suckers began in state A, transitions from states B or C in the first to the second season were not possible ($\psi_1^{B:C} = \psi_1^{C:A} = \psi_1^{C:B} = 0$), and because $\psi_t^{A:B} = \psi_t^{B:A} = 0$, a Blue Sucker could not occur in state B in season 2, so $\psi_2^{B:C} = 0$. I estimated parameters with binomial outcomes with a logit-link function (e.g., ϕ , survived or died; P_t , detected or not; $\psi_t^{A:C}$, moved or stayed); whereas, parameters with a polynomial outcome were estimated with a multinomial logit-link function (e.g., $\psi_t^{C:A}$ or $\psi_t^{C:B}$ or $\psi_t^{C:C}$, moved to A or B or remained in C).

Because estimated survival (ϕ) was confounded with $\psi_t^{C:C}$, I treated ϕ as a nuisance parameter and focused my hypothesis testing on transitions. The estimation of ϕ is dependent on an individual going undetected for multiple seasons, provided a reasonable chance for that individual to be detected each season. The longer an individual was undetected, the lower the probability the individual was still alive. In my study, individuals could move to and remain in state C where $P_t^C = 0$. It was very likely for a fish to remain alive and go undetected for multiple seasons. Although necessary for my model, my estimate of ϕ was not reliable; however, this estimate was not important to my hypothesis testing. Instead, I focused on the fish that were observed transitioning

back to states A or B ($\psi_t^{C:A}$ and $\psi_t^{C:B}$) as these parameters were needed to test my hypotheses.

I checked that model fit was adequate and parameter estimates were realistic before ranking the hypothesized models using AIC_c . Using the built-in median \hat{c} goodness-of-fit test, I verified that median $\hat{c} \leq 3$ and that the parameter estimates reflected what I generally observed in the field (Haddon, 2001; Cooch & White, 2013). For example, I expected lower detection probabilities during the late 2015 season because these detections were based on a single SUR station and only the tailwater reach was actively tracked. I also anticipated higher detection estimates for each of the early seasons due to the higher sampling effort in that season. I compared the models using AIC_c because the method is better suited to compare non-nested models and models with unequal parameters than the traditional likelihood-ratio test (Leberon *et al.*, 1992; Burnham & Anderson, 2002; Johnson & Omland, 2004). Using AIC_c provides a relative measure of model fit (*i.e.*, compared to all other models considered) with consideration for parsimony and a small sample size (Burnham & Anderson, 2002; Cooch & White, 2013; Aho *et al.*, 2014). After constructing the models, I ranked them from low to high using AIC_c values. Models and hypotheses with lower AIC_c values were better supported by the data than those with higher AIC_c values; however, models with a ≤ 2 difference in AIC_c were considered equally supported (Burnham & Anderson, 2002; Aho *et al.*, 2014).

For the top ranked model(s), I performed Markov-chain Monte-Carlo simulations (MCMC) to obtain robust parameter estimates and 95% credibility intervals (Buckland & Grathwaite, 1991; White *et al.*, 2009; Cooch & White, 2017). I used the mode and 95% credibility interval of these distributions to evaluate parameter estimates and the uncertainty around those estimates. The mode was used rather than the mean because it was comparable to the maximum likelihood estimates of the initial model, and the 95% credibility interval was comparable to the familiar 95% confidence interval in frequentist statistics (Kruschke, 2011). I did not specify informative prior distributions, because I did not have prior information to contribute; however, I did provide beta values estimated with logit or multinomial-logit link functions as recommended by Cooch & White (2017). I included 55,000 iterations with 4,000 tuning and 1,000 burn-in iterations to accommodate the large number of parameter estimates. Additionally, I replicated each MCMC 10 times to obtain Gelman - Rubin \hat{R} convergence statistics and verified that

parameter estimates had $\hat{R} < 1.1$ (Gelman & Rubin, 1996). I plotted the resulting mode and 95% credibility intervals for parameters of interest for comparison.

RESULTS

TAGGING AND TRACKING

I implanted acoustic tags in 119 Blue Suckers in spring 2015 and recaptured 66 individuals at least once during the study. I tagged 55 male and 64 female Blue Suckers (see appendix B). The mean TL of my tagged fish was 572 mm (43 mm SD) and the mean weight was 1470 g (407 mm SD). Males were covered in pronounced tubercles and gametes were expressed with abdominal pressure. Females displayed tubercles on their head and around their fins. Despite being gravid (*i.e.*, the presence of hydrated oocytes), abdominal pressure did not result in gamete expression. The M:F ratio of recaptured Blue Suckers approximated the M:F ratio of tagged Blue Suckers (M:F 0.81 and 0.87, respectively).

The effective sample size (*i.e.*, data available to provide reasonable parameter estimates) for my analysis was 200 observations after omitting data from one tagged female. The effective sample size is the number of fish tagged plus the number recaptured in all but the last season (Cooch & White, 2013). I recaptured 22 fish in ≥ 2 seasons. Only one female was recaptured in every season because she never left the tailwater reach, but was observed moving up and downstream throughout the study (Table 2.2). Only one female was excluded from the analysis because she transitioned between states A and B without spending a season in state C, violating an assumption of my design. The excluded Blue Sucker moved to Muddy Boggy and Kiamichi rivers during early season 2016 and to Muddy Boggy River in the early season 2017, but she used the tailwater reach outside of spawning.

MOVEMENT ANALYSIS

Each of the hypothesized models met the goodness-of-fit criteria; however, the model reflecting the tributary hypothesis had more support than the other hypotheses. The 95% upper bound for the median \hat{c} values were < 3.0 in all instances and the parameter estimates for survival and detection were similar among models. The tributary-hypothesis model had more support than all other models ($AIC_c = 420$) indicating that movement patterns of fish tagged in the tailwater were different than those tagged in the tributaries. The model associated with the dam hypothesis had no

more support than the null model (H_{dam} : $AIC_c = 428$ v. H_0 : $AIC_c = 428$), and the sex hypothesis model had less support than the null model (H_{sex} : $AIC_c = 458$ v. H_0 : $AIC_c = 428$). Omitting the fish sampled in the tailwater reach from the analysis did not improve the performance of the sex hypothesis model compared to the null model (H_{sex} : $AIC_c = 316$ v. H_0 : $AIC_c = 288$) indicating that males and females in my study had similar movement patterns.

The MCMC simulation of the tributary-hypothesis model converged and had adequate explanatory power though the survival estimate was still confounded. The \hat{R} values associated with the ten replicated simulations indicated that all parameter estimates converged ($0.99 < \hat{R} < 1.01$). I did not recapture 45% (53 of 119) of my tagged fish so I did not know their fate relative to survival. However, because some of my tagged fish did return after missing for more than 18 months, the survival estimate was still high ($\phi = 0.96$, 0.95 – 0.98 95% CI). The survival estimate was confounded by the probability of fish remaining in state C (*i.e.*, fish could either die or survive in state C and I would not be able to distinguish between the conditions). Probabilities of Blue Suckers remaining in state C after being tagged were $\psi_2^{C:C} = 0.92$, $\psi_3^{C:C} = \psi_4^{C:C} = 1$ for fish in the tailwater, and $\psi_2^{C:C} = 0.17$, $\psi_3^{C:C} = 0.89$, $\psi_4^{C:C} = 0.81$ for fish tagged in the tributaries (Fig. 1.4).

The detection probabilities from the MCMC simulation of the tributary-hypothesis model were realistic representation of my tracking efforts. Compared to late 2015, detection probabilities were much higher in late 2016 when SURs were functioning throughout the season and I were able to track each reach on multiple occasions ($P_1^{AUB} = 0.08$ and $P_3^{AUB} = 0.96$, respectively). As expected, my limited tracking effort in the late seasons was reflected in broad credibility intervals (0.01 – 0.53 CI, 2015 and 0.61 – 1 CI, 2016). Additionally, detection probabilities from the early seasons were higher and met with less uncertainty ($P_2^{AUB} = 0.96$, 0.80 – 1 CI, 2016 and $P_4^{AUB} = 0.96$, 0.68 – 1 CI, 2017; Fig. 1.4).

Inspection of the MCMC simulation of the tributary-hypothesis model revealed differences between the movement patterns of tailwater and tributary fish movements, and movements of tributary fish differed between years. The greatest difference between tailwater and tributary fish was in the probability of tagged individuals leaving state A (Fig. 1.4). Blue Suckers tagged in the Red River were unlikely to leave the tailwater reach from the early to the late seasons ($\psi_1^{A:C} = 0.13$; 0.03 – 0.87 CI, $\psi_3^{A:C} = 0.03$; 0 –

0.27 CI); whereas, tributary fish were likely to leave the tributaries following the early seasons ($\psi_1^{A:C} = 0.98$; 0.46 – 1 CI, $\psi_3^{A:C} = 0.88$; 0.64 – 0.97 CI). Additionally, large credibility intervals accompanied transition estimates from state C to A or B, for fish tagged in the Red River, indicating a high level of uncertainty (Fig. 1.5). During the early 2016 season when discharge was typically high, Blue Suckers tagged in the tributaries displayed a tendency toward site fidelity ($\psi_2^{C:A} = 0.65$; 0.31 – 0.85 CI); however, some Blue Suckers strayed ($\psi_2^{C:B} = 0.18$; 0.06 – 0.41 CI; Fig. 1.5). In early season 2017 when discharge was low, the probability of fish returning the tributaries was low and there was not a clear preference between state A or B ($\psi_4^{C:A} = 0.10$; 0.01 – 0.31 CI, and $\psi_4^{C:B} = 0.09$; 0.01 – 0.30 CI) demonstrated by the few fish that did return (Fig. 1.5). Although most fish detected in 2016 displayed fidelity to each of the tagging tributaries, straying fish showed a tendency to use the free-flowing Muddy Boggy River (Fig. 1.6).

DISCUSSION

MOVEMENT PATTERNS

Like many other big river fishes, my results suggest Blue Sucker has migratory and non-migratory individuals within their populations. Tributary-tagged Blue Suckers migrated into the tributaries in the early season and typically left during the late season. Similar movement patterns were observed in the Grand River, Missouri (Vokoun *et al.*, 2003), middle Missouri River, Nebraska (Neely *et al.*, 2009), and Wisconsin River, Wisconsin (Lyons *et al.*, 2016). Conversely, I estimated a low probability of Blue Suckers in the tailwater reach leaving during the late season, potentially suggesting either mainstem movements or a non-migratory portion of the population. Resident individuals within a migratory population are not abnormal (Rodriguez *et al.*, 2002), in fact, freshwater salmonid and catostomid populations often comprise both migratory and sedentary members (*e.g.*, Brown Trout *Salmo trutta*, Lucas & Baras 2001; Sonora Sucker *Catostomus insignis* & Desert Sucker *Catostomus clarkia*, Booth *et al.*, 2014; Robust Redhorse *Moxostoma robustum*, Fisk *et al.*, 2015). As indicated in my hypothesis, the tailwater reach provides suitable habitat for multiple life history requirements of Blue Sucker. Blue Sucker display positive rheotaxis (Moss *et al.*, 1983), and the year round water flows, associated with hydropower releases, may serve as an attractive habitat feature. The fish I recaptured in the tailwater reach may also be using the reach as non-spawning habitat and migrate elsewhere during the early season. The fish I excluded from the analysis appeared to be spawning in Muddy Boggy River and

spending the rest of her time in the tailwater reach. Individuals that may have spawned in an unobservable region of the study area and returned to the tailwater reach within the early season would have falsely appeared as non-migrants in my study. Unfortunately, I was unable to distinguish the absence of an individual from a missed detection at a fine temporal resolution to provide better information on this possibility.

Fish tagged in the tributaries were more likely to home to the tributary where they were tagged rather than straying, and the probability of fish returning appeared to correspond to flow patterns. The estimated probability of Blue Suckers from tributaries displaying strong site fidelity in 2016 (78% of those returning) was similar to other iteroparous fishes. Male Smallmouth Bass *Micropterus dolomieu* returned within 200-m of previously documented spawning sites 81% of the time (Ridgeway *et al.*, 1991), and adult Weakfish *Cynoscion regalis* returned to their natal habitat to spawn 60-81% of the time (Thorrold *et al.*, 2001). Although my three-year study was not adequate to quantify the relationship between flow and migration probability, Blue Suckers appeared to be more likely to return to the tributary where they were tagged when flows were higher. In the wetter 2016 period, the probability of Blue Suckers returning to any tributary was 0.83 compared to 0.19 in 2017 when flows were much lower. Dam releases can initiate spawning migrations (Cushman *et al.*, 1985; Lucas & Baras, 2001), but the influence of dams was not detected with my dam-release hypothesis model. Overall in 2017, I detected very few tagged Blue Suckers returning to the tributaries (n = 5) compared to the abundance observed in the tailwater (n = 15). The relatively small differences observed among tributaries lent little explanatory power for the dam-release hypothesis (Fig. 6). However, flows associated with dam releases are known attractants for migratory fishes (Bunt *et al.* 2001; Aarestrup *et al.* 2003), and managers should not discount the potential influence of dam releases during low-flow years.

I did not detect a difference between male and female movement patterns as I hypothesized suggesting both sexes invest considerable effort into migrating for reproduction. In many animals, it is assumed that the females invest more energy into reproduction than males, because oocytes are more energy dense than sperm (Leonard & Lukowiak, 1984; Berglund *et al.*, 1986). However, males that grow tubercles and guard territory expend more energy than is required for sperm production alone and thus, compensate for the energy difference (Moyle & Cech, 2004). Energy expensive male territoriality has been documented for several catostomids, including Creek Chubsuckers *Erimyzon oblongus* (Page & Johnston, 1990), Robust Redhorse (Grabowski & Isely,

2006), and Flannelmouth Suckers *Catostomus luttipinnis* (Weiss *et al.*, 1998). When sampling during the early season 2017, I observed male Blue Suckers displaying energetically expensive spawning behavior. Blue Suckers arrived earlier and remained in the area longer than females, and had pronounced tubercles covering their body (Dyer personal observation). Similarly, Lyons *et al.* (2016) observed ripe male Blue Suckers arriving at the spawning area and remaining later than females in the Wisconsin River. Although males returned more frequently than females (1 yr interval verses 2 yr) in the Wisconsin River, recaptures were low and variation in return rates was similar for each sex (Lyons *et al.*, 2016). Due to the spatial and temporal resolution of my study (*i.e.*, weekly tracking events for each reach), I were not able to document the residence time of individuals at specific habitats. Future efforts to determine whether spawning Blue Suckers display territoriality would be beneficial to improve understanding of these patterns.

Nearly 50% of the fish I tagged were never detected again post tagging, and I can only speculate their fate. Detection in the tracking reaches of my study area was reasonably high (except during major floods) because the reaches were a relatively quiet acoustic environment and were tracked frequently with both passive and active methods. However, I had to consider a large part of the study area unobservable. When flows were high (850 m³/s), I tracked the 313 km of the Red River that comprised the Oklahoma portion of state C during the late season 2015. I detected four Blue Suckers near the Arkansas border, but for the majority of this area the river was not navigable at average discharge (20-yr average discharge 150 m³/s; Arthur City gage, July - November). Unfortunately, under high flows, environmental noise likely prevented acoustic tag detection in this reach. It is also possible that Blue Suckers suffered mortality or tag loss. Tag retention and survival in captive Razorback Suckers *Xyrauchen texanus* was 100% after six months (Karam *et al.*, 2008). Karam *et al.* (2008) reported high mortality (84%) in tagged Razorback Suckers in the wild, but suggested that the mortality was due to predation rather than tagging. Rechisky & Welch (2010) found tag retention as high as 95% after 24 weeks, and tag related mortality was \leq 85% in surgically tagged Chinook Salmon smolt *Oncorhynchus tshawytscha*. A tag loss rate of 16% was observed for Steelhead smolt *Oncorhynchus mykiss* with lengths > 140 mm (Welch *et al.*, 2007). I could not assess tag loss directly, but I examined the incisions of several Blue Suckers recaptured during electrofishing and the incision healed as expected. Rechisky & Welch (2010) speculated that unabsorbed sutures leading to open

wounds were the primary cause for tag-related mortalities. I recognize the possibility of tag loss, tag malfunction or mortality as possible reasons for not detecting more tagged fish. However, Blue Suckers had the potential to emigrate, or may not have made another spawning migration within the duration of my study.

Emigration and non-constant spawning frequencies are two possible explanations for low recapture of Blue Suckers. In 2015, an abnormally wet spring followed by Hurricane Bill, created two major flood crests (5,635 m³/s on June 2, and 3,284 m³/s June 25; Arthur City gage). Extreme flooding can temporarily displace fish, but both displacement and the ability to recolonize is species specific (Dolloff *et al.*, 1994; Fritz *et al.*, 2002). Blue Sucker morphology is adapted to high flows and Blue Suckers are typically good at maintaining their position in swift habitats (Moss *et al.*, 1983). However, the flood in 2015 was exceptional and Blue Suckers could have moved substantial distances downriver. Additionally, tagged Blue Suckers may have been part of a larger population or meta-population. Layher (2007) documented spawning Blue Suckers in a bend in the Red River, Arkansas. I detected four Blue Suckers I tagged near Arkansas in late 2015. The four fish traveled 120 – 240 km from their tagging location and were approximately 170 km from those described by Layher (2007). In the Missouri River, Neely *et al.* (2009) documented Blue Suckers traveling an average 249 km and a maximum of 334 km during spring spawning migrations. I have no reason to assume that the population tagged in my study does not have a much larger range than defined by my tracking extent. It is also possible that Blue Suckers may not return to spawn every year. For many iteroparous females, environmental conditions or food availability can determine spawning frequencies (McBride *et al.*, 2015). For example, Cui-ui in Lake Pyramid, Nevada, only reproduce in wet years when lake tributaries are flowing and year classes are reported as much as 17-yrs apart (Scoppettone *et al.*, 2000). Lyons *et al.* (2016) noted the spawning return intervals of PIT tagged Blue Suckers for both males and females were as long as six years, but this was potentially confounded with low detection rates. Similarly, two individuals in my study went undetected from the time they were tagged in 2015 until 2017, but I cannot know whether these two individuals chose not to spawn or avoided detection in 2016. Unfortunately, a study duration of approximately 10-yrs would be needed to detect the possible trends in highly migratory populations with irregular migration patterns. The current available acoustic tag life limits these opportunities without the use of other approaches (*e.g.*, PIT tags) and annual sampling.

Of the four reaches, Muddy Boggy River was the most frequently used tributary by straying Blue Suckers. Muddy Boggy River was the only stream in my study without a dam located on the main channel. Blue River has a low-head dam, but the flow patterns still generally followed natural patterns. However, Blue River has lower discharge (median discharge 2 m³/s vs. 7 m³/s, respectively) and a smaller drainage area when compared to Muddy Boggy River. Further, Blue River has several bedrock shoals near the confluence with the Red River that likely require elevated discharge to provide connectivity for large-bodied fish like Blue Sucker. I suspect that the natural flow regime of the Muddy Boggy River, combined with larger river size, and good connectivity, make it an important tributary for straying fish. I also assumed that the location where an individual was tagged was not a result of straying; however, when I tagged fish in early 2015, discharge was low and dam releases from Denison Dam or Hugo Reservoir may have drawn Blue Suckers into the tailwater reach or Kiamichi River (as I observed in 2017). Thus, it is possible that many of the fishes observed straying to Muddy Boggy River were actually returning to natal habitat that was unavailable in other years due to low discharge.

MANAGEMENT IMPLICATIONS

Management of dams on large rivers that mimic the natural flow regime can provide suitable spawning habitat and potentially mitigate the effects of extreme drought periods. Blue suckers displayed some plasticity in spawning location choice, as many presumably strayed to Muddy Boggy River when discharge was high, but moved into dammed tributaries when discharge was low. As the larger of the two undammed streams in my study, Muddy Boggy River may serve as the preferred location for migrating Blue Suckers in wet years; thus, protecting spawning habitat quality and connectivity within Muddy Boggy and Red rivers would be beneficial to the persistence of these populations. The plasticity in spawning-site selection by some Blue Suckers indicates that dam releases could be used as a management tool to provide spawning cues and habitat during drought conditions (e.g., Jager & Smith, 2008). For example, King *et al.* (1998) successfully manipulated the timing, magnitude and duration of dam releases to facilitate spawning activities by Clanwilliam Yellowfish *Barbus capensis*. However, efforts to examine both the extent of spawning habitat across the basin and the effects on recruitment below dams would be beneficial to understanding the perceived benefits of discharge-related management options. First, extreme drought in a

portion of the basin may not be problematic if there are other areas lower in the basin that provide adequate straying possibilities. Second, spawning below a dam can be detrimental to the population if post-spawning conditions are inadequate for recruitment (King *et al.*, 1998). Below dams, recruitment can suffer due to egg desiccation (Grabowski & Isely, 2007), poor water quality (Müller *et al.*, 2008; Olden & Naimen, 2010), or stunted juvenile growth (Weyers *et al.*, 2003). Although I did not examine the quality of juvenile habitat below the dam, I observed water levels dropping as much as 2 m immediately following spawning activity in the Kiamichi River due to dam operations. Managing discharge with major fluctuations during the spawning period may be detrimental to these populations. Rather, some consideration of ramping discharge in both directions could generate the energy required and facilitate an ecological benefit.

Conservation plans for inland fisheries are often restricted to geopolitical boundaries; however, many large river migratory fishes such as Blue Sucker would benefit from interstate collaborative efforts. I show that many individual Blue Sucker are highly migratory annually and the abundance of missing tagged fish suggests I likely underestimate these migration distances. Within states collaboration among agencies, universities and the federal government are a common component to State Wildlife Action Plans (Lauber *et al.*, 2011); however, collaboration among state agencies in large rivers is rare for fishes not listed as threatened or endangered. The U.S. Fish and Wildlife Service has jurisdiction over threatened and endangered species and collaborates with many agencies (Ballweber & Schramm, 2010). For example, the Great Plains Fish and Wildlife Conservation Office collaborates with several federal and state agencies to manage Pallid Sturgeon and Paddlefish (James, 2018). In North America, the Migratory Bird Treaty has resulted in the successful management of migratory waterfowl due to collaboration among the Canadian Wildlife Service, U.S. Fish and Wildlife Service and various state agencies throughout the U.S. (Anderson *et al.*, 2018). Globally, marine fisheries are collaboratively managed among countries (Brown, 2017). The large river migratory fishes, such as those found in the Red River, would benefit from similar collaborative efforts among agencies. This is particularly important for species that are difficult to study, and do not benefit from federal listing designations.

Table 2.1 Candidate models used in my analysis of Blue Sucker *Cycoreptus elongatus* movement. Hypotheses names refer to: the tributary hypothesis – movement patterns of fish tagged in the tributaries differ from fish tagged in the tailwater reach, null hypothesis – there is no difference in movement patterns among groups of fish, dam-release hypothesis – movement patterns of fish tagged in reaches with large dams differ from fish tagged in undammed tributaries, and sex hypothesis – movement patterns of male fish differ from female fish. The model notation references the survival parameter (φ) was estimated as a constant (\cdot), detection probability (P) was estimated as a function of time ($n = 4$ seasons) and location, where location was either detectable (states A & B) or undetectable (state C, $n = 2$ locations), and transitions between states (ψ) was estimated as a function of time, state (A, B, or C, $n = 3$ states), and either sex (male or female, $n = 2$), dam (reach was dammed or not dammed, $n = 2$), or tributary (reach was Red River tailwater or tributary, $n = 2$). K indicated the number of parameters estimated in each model, and the AIC scores are provided along with the difference in AIC score between each model and the top model.

Hypothesis	Model	K	AIC _c	Δ AIC _c
Tributary	$\varphi \sim \cdot, P \sim \text{time}, \psi \sim \text{time} + \text{state} + \text{tributary}$	29	420	0
Null	$\varphi \sim \cdot, P \sim \text{time}, \psi \sim \text{time} + \text{state}$	17	428	8
Dam-release	$\varphi \sim \cdot, P \sim \text{time}, \psi \sim \text{time} + \text{state} + \text{dam}$	29	428	8
Sex	$\varphi \sim \cdot, P \sim \text{time}, \psi \sim \text{time} + \text{state} + \text{sex}$	29	458	38

Table 2.2 Capture histories of acoustic telemetered Blue Suckers used in Multistate Cormack-Jolly-Seber model. Capture histories in the first column consist of a string of five characters, where each character position represents a time interval, beginning with tagging in spring 2015 and ending with spring 2017. Each character represents the state that a fish was detected in: “A” state where the fish was tagged, or “B” any other detectible state. Values of zero indicate the individual was not detected. The columns to the right indicate the number of fish from each group (sex X location) displaying a given capture history.

Capture History	Tailwater		Blue		Muddy Boggy		Kiamichi	
	F	M	F	M	F	M	F	M
A0000	4	3	7	9	8	11	8	3
A000B	0	0	1	0	0	1	0	0
A00AA	0	1	0	0	0	0	0	0
A00B0	0	0	1	0	0	0	0	0
A0A00	1	4	2	5	8	0	6	5
A0A0A	0	0	1	0	0	0	1	1
A0AA0	0	0	0	1	0	1	0	0
A0AAA	8	3	0	0	0	0	0	1
A0B00	1	1	3	0	0	0	1	3
A0B0B	0	0	0	0	0	0	0	1
A0BB0	0	0	0	0	0	1	0	0
AAAA0	1	0	0	0	0	0	0	0
AAAAA	1	0	0	0	0	0	0	0

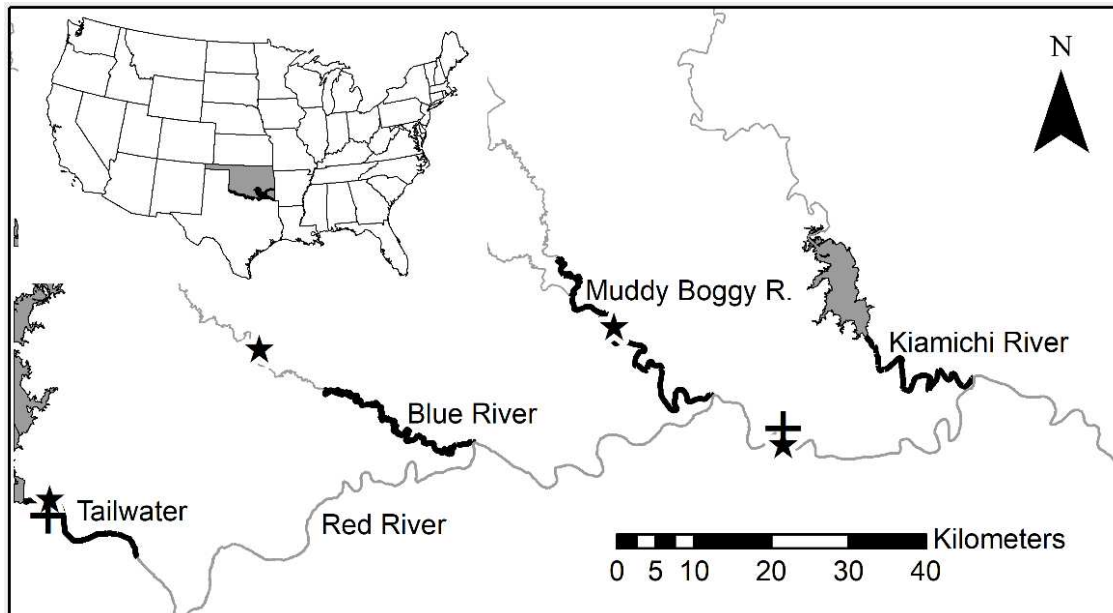


Fig. 2.1 I conducted a multistate mark-recapture study of Blue Sucker in the lower Red River, Oklahoma, USA. Stream reaches in bold reference active tracking and tagging locations, and represented state A or B in individual capture histories. The Red River in non-bold font connects A and B states and is referred to as state C in the multistate framework. Stars indicate the location of the U.S. Geological Survey stream gages, and crosses represent the location of temperature loggers. The SURs were located in each of the active tracking reaches within 2 km of the lower extent.

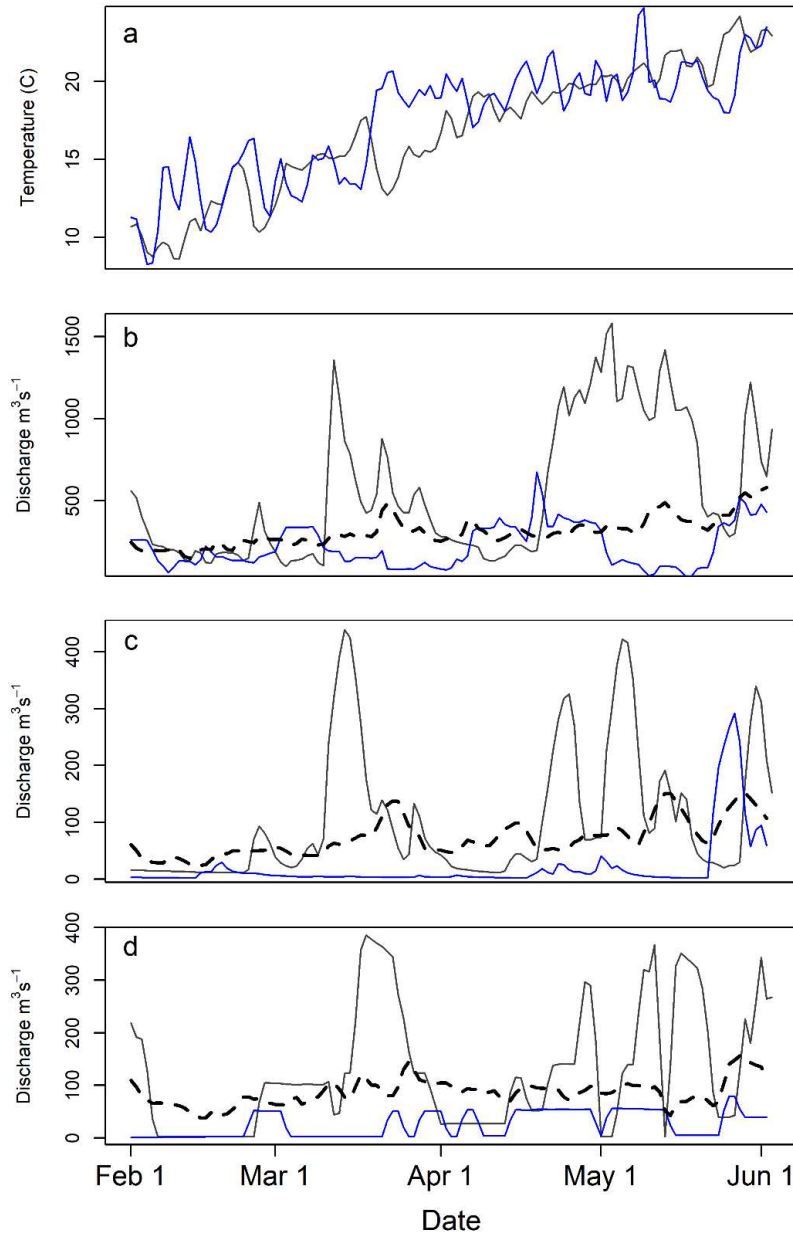


Fig. 2.2 Temperature in the Red River (Arthur City) (a), and hydrographs representing stream discharge in cubic meters per second (m^3s^{-1}) in the Red (b), Muddy Boggy (c), and Kiamichi (d) rivers during the early seasons 2016 (gray), 2017 (blue), and the 20-year average (1998 – 2017, dashed)

(https://waterdata.usgs.gov/ok/nwis/uv?site_no=07335500, accessed 2/27/18).

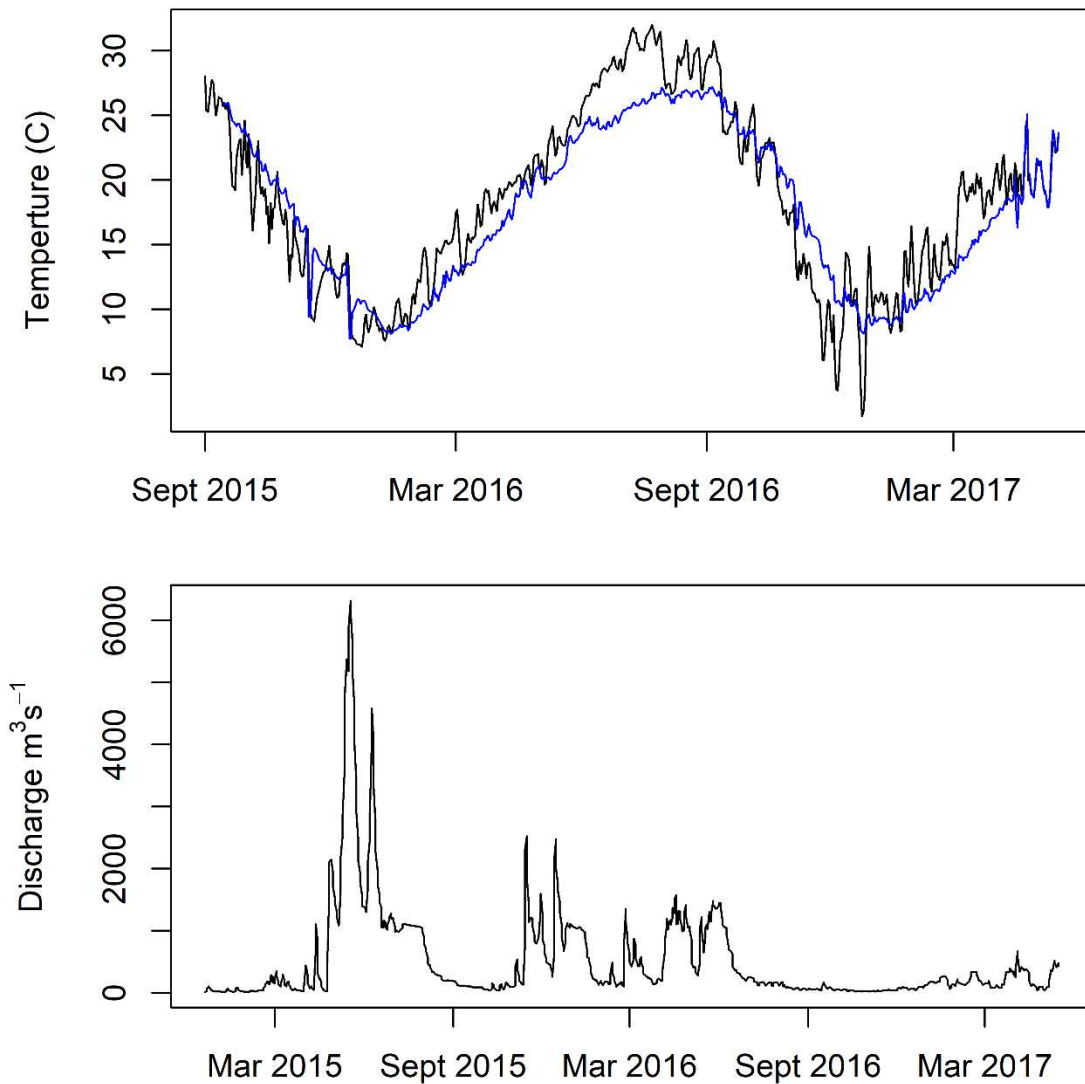


Fig. 2.3 Water temperatures (top) and stream discharge (bottom) recorded during my Blue Sucker movement study. Water temperatures were recorded in the Red River at Arthur City (black, USGS gage 07335500) and Denison (blue, USGS gage 07331600), and stream discharge data are from Arthur City (https://waterdata.usgs.gov/ok/nwis/uv?site_no=07335500, accessed 2/27/18).

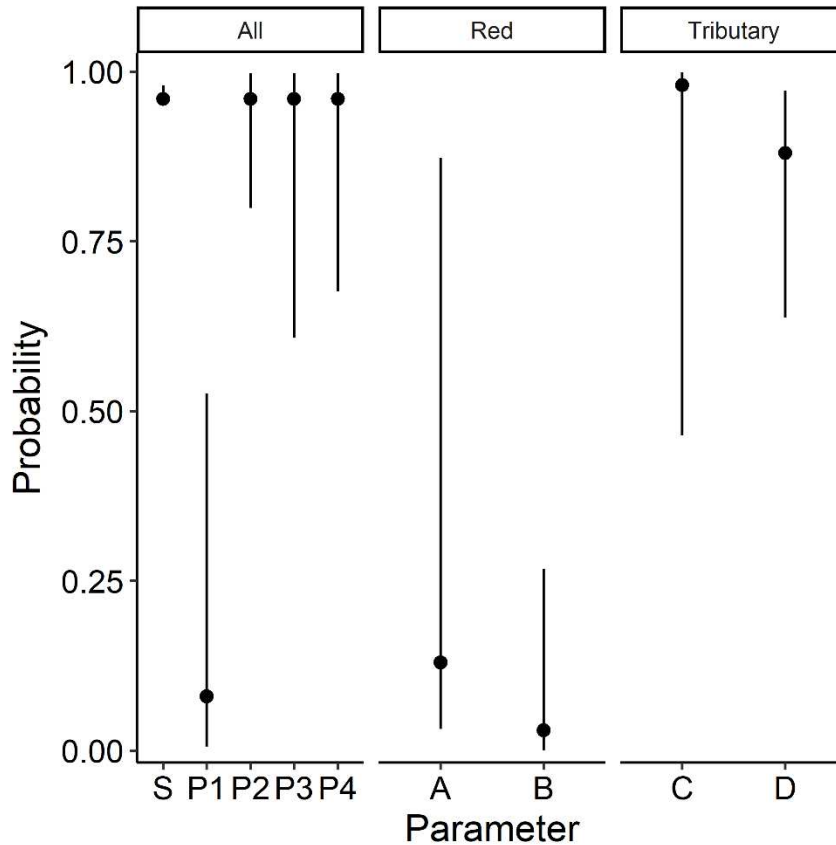


Fig. 2.4 Parameter estimates from Markov-Chain Monte-Carlo simulations of a multistate Cormack-Jolly-Seber model of coarse-scale movement patterns by Blue Suckers in the lower Red River, OK. From left to right, parameter estimates on the x-axis are: S.) probability of survival (ϕ), P1.) detection probability in late-season 2015 (P_1^{AUB}), P2.) P in early-season 2016 (P_2^{AUB}), P3.) P in late-season 2016 (P_3^{AUB}), P4.) P in early-season 2017 (P_4^{AUB}), and transition probabilities of fish leaving the state they were tagged in, from early to late seasons 2015 ($\psi_1^{A:C}$) and 2016 ($\psi_3^{A:C}$) for fish in the Red River tailwater (A & B), and fish tagged in tributaries (C & D). Dots indicate the mode and vertical bars represent 95% credibility intervals from posterior distributions.

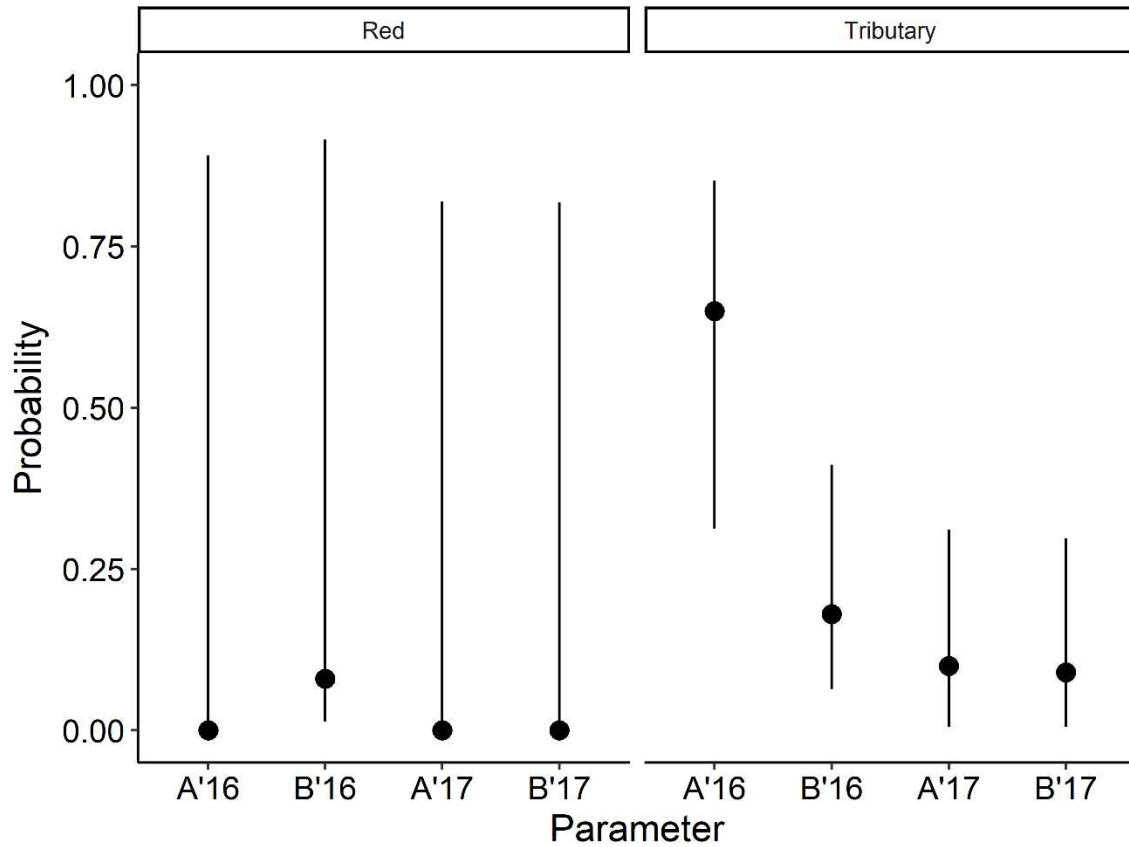


Fig. 2.5 Parameter estimates from a Markov-Chain Monte-Carlo simulation of a multistate Cormack-Jolly-Seber model of coarse-scale blue sucker movement in the lower Red River, OK. Parameters on the x-axis are the probability of an individual from either the Red River tailwater reach (Red) or a tributary returning to the state it was tagged (A) or a different observable state (B) in early-season 2016 ('16) or 2017 ('17). Dots indicate the mode and vertical bars represent 95% credibility intervals from posterior distributions.

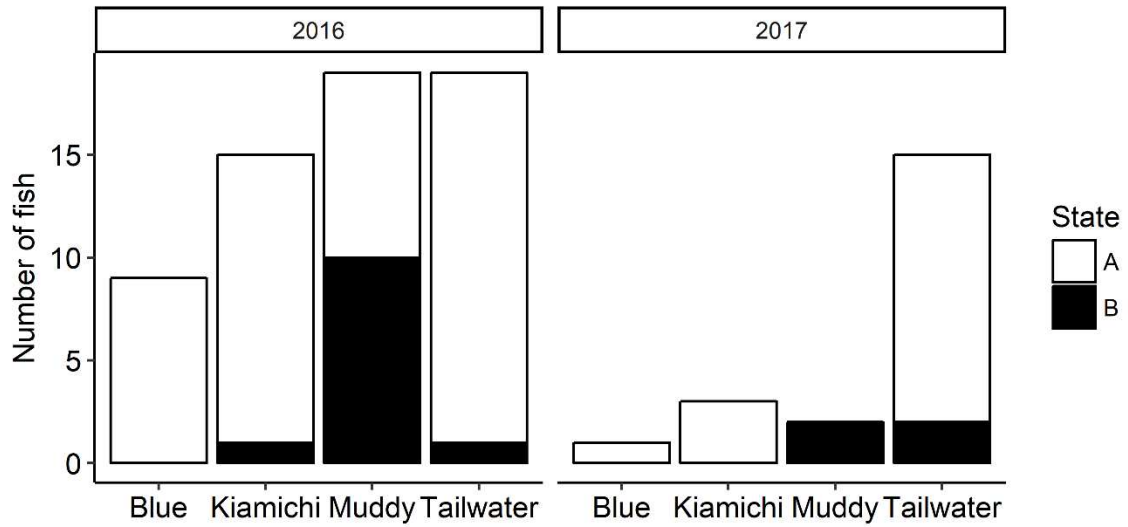


Fig. 2.6 Number of telemetered Blue Suckers detected in tracking locations in early seasons 2016 and 2017. Tracking locations are the three major tributary rivers (Blue, Kiamichi, and Muddy Boggy rivers) and the Red River just downstream of Denison Dam, OK, USA (Tailwater). Hollow bars indicate the number of fish returning to the location they were tagged (state A) and the number of fish straying (state B) are shown with black bars.

CHAPTER III

TIMING AND SPAWNING HABITAT USE

INTRODUCTION

Many large-river migratory fishes of the Mississippi River catchment are species of conservation concern (Jelks *et al.*, 2008) including catostomids that facilitate energy transfer across this large region (Cooke *et al.*, 2005; Flecker *et al.*, 2010). Migratory fishes are disproportionately affected by human alteration of river environments (Hinch *et al.*, 2005; Cooke *et al.*, 2012). Changes to river connectivity (*i.e.*, longitudinal, Ward & Stanford, 1983; lateral, Junk *et al.*, 1989) and natural flow patterns (Poff *et al.*, 1997) are due to human modifications that disrupt completion of the life histories of river fishes (Fausch *et al.*, 2002). In fact, approximately 80% of freshwater fishes of conservation concern in North America inhabit localized ecoregions in the upper extents of catchments (Jelks *et al.*, 2008), and the loss of connectivity with the downstream ecosystem may contribute to the decline of those populations (Pringle, 1997). Catostomidae comprises many of the affected species; however, their decline has not been the focus of conservation or recovery efforts due to the perceived lack of importance to humans (Cooke *et al.*, 2005). However, when compared to other migratory fishes, catostomids are more likely to provide energy subsidies to isolated communities and facilitate ecological processes that require upstream movement (*i.e.*, glochidia, Flecker *et al.*, 2010).

Blue Sucker *Cyprinostomus elongatus* is a species of concern with a complex life history. Blue Sucker is broadly distributed across the Mississippi River catchment, and

was thought to be declining throughout the 20th century (Burr & Mayden, 1999). However, Burr & Mayden (1999) suggested that the perceived decline of Blue Sucker was due to misunderstanding the migratory life history of the species. Blue Sucker life history is better understood today, but is considered vulnerable to extirpation in North America (Jelks *et al.*, 2008; NatureServe, 2018). Blue Sucker occupy the mainstem of large rivers for the majority of the year (Neely *et al.*, 2010), and may migrate up to 300 km in the spring (Metee *et al.*, 2015; Neely *et al.*, 2009) to spawn in swift, deep water with coarse substrates (Moss *et al.*, 1983, Vokoun *et al.*, 2003a; Zymonas & Probst, 2007). A rise in temperature in conjunction with rising discharge is thought to cue Blue Sucker to undergo spawning migrations (Vokoun *et al.*, 2003), although the water temperatures and the timing of movements vary regionally. Blue Sucker in the Red River, AR, initiated spawning in mid-March when water temperatures reached 15 – 20 °C (Mettee *et al.*, 2015; Layher, 2007). However, Blue Sucker in the Grand River, MO, and the Missouri River, NE, spawned in late April to May when water temperatures were as low as 10 °C (Vokoun *et al.*, 2003, Neely *et al.*, 2009). Like many migratory fishes, migratory cues and habitat selection by Blue Sucker are likely dependent on many environmental factors (Lucas & Baras, 2001). Reports of Blue Sucker spawning habitat are reasonably consistent, but often based on opportunistic observations (*e.g.*, Vokoun *et al.* 2003a; Layher, 1998). Targeted efforts to understand Blue Sucker spawning cues have often suffered from the difficulty of observing Blue Suckers during the spawning season and resulted in more uncertainty regarding spawning behavior (*e.g.*, Zymonas & Probst, 2007; Lyons *et al.*, 2016). Understanding the relationship between life history and habitat and discharge remains a limiting component of fish conservation (Cooke *et al.*, 2012).

Large rivers in Oklahoma harbor Blue Sucker at the southwestern extent of the Mississippi River catchment and offer a broad range of physicochemical conditions for studying drivers of spawning migrations and habitat use by the species. There is limited information about the distribution and habitat use by these populations and most existing information has been derived from occasional collections over time (*e.g.*, 1940-1988, Burr & Mayden) rather than targeted sampling for the species. However, spawning was documented in the Red River, AR and the population is suspected to be declining based on changes in catch-per-unit-effort sampling (Layher, 1998; Layher, 2007). Additionally, sampling efforts by the Oklahoma Department of Wildlife Conservation (ODWC) have documented congregations of Blue Sucker beneath a dam on a large tributary of the

Red River (C. Tacket, ODWC, personal communication). Only one other recent collection in the Oklahoma portion of the basin has been reported (*i.e.*, Muddy Boggy River, S. Brewer, Unpublished data). I began conducting targeted Blue Sucker sampling in the Red River catchment of Oklahoma to improve the understanding of habitat use and spawning migrations. My study objectives were to determine: 1) the relationship between environmental conditions and Blue Sucker movements into the Red River tributaries, and 2) spring-time habitat selection by Blue Suckers in the Red River tributaries. Recognizing the factors driving movement and habitat use during the critical spawning season would be beneficial to developing both long-term monitoring strategies and identifying management options for improving population abundances if needed or desired by the management agencies.

METHODS

STUDY AREA

I conducted my study at four sites within the lower Red River basin of Oklahoma. Lower Red River basin is located in the wet South Central Plains ecoregion where oak-hickory-pine forests are interlaced with prairies (Woods, 2005). The mainstem lower Red River begins as a tailwater downstream of Denison Dam, a hydropower dam constructed in 1944 that impounds Lake Texoma (Fig. 3.1). Impoundment via Denison Dam disrupted the sediment regime resulting in channel down-cutting and a 20-km long series of coarse substrate regions that resemble riffles (tailwater reach). From Denison Dam, the Red River flows 333-km and generally comprises sand and alluvium substrates and a braided channel. Three major tributaries contribute significant discharge to the mainstem Red River of OK (listed from west to east, average annual flow in parentheses): Blue River (8.7 m³/s), Muddy Boggy River (46.3 m³/s), and the Kiamichi River (60.8 m³/s). Blue River is a 4th order (Strahler, 1957) stream and has a low-head dam ≈ 90-km upstream of the Red River confluence that acts as a fish-passage barrier. Muddy Boggy River is a 5th order, free-flowing river. The Kiamichi River is impounded by Hugo Dam 28-km upstream of the mainstem confluence. All three tributaries differ from the mainstem in having Cretaceous age gravel and clay substrates, heavy quantities of instream cover, and regular bedrock outcrops. Dams marked the upper extents of my study reaches in the tailwater reach and the Kiamichi River. I designated bridge crossings 40 and 50 km upstream of Blue and Muddy Boggy river confluences with the

Red River, respectively, as the upper extent of the reaches, because the reaches could be tracked within a single day.

MOVEMENT AND HABITAT SELECTION

In spring 2015, I captured and tagged both male and female Blue Suckers from my four study sites. I captured Blue Suckers via boat electrofishing (5.0 GPP, Smith-Root, Vancouver, WA) and surgically implanted acoustic tags (CT-05-36-I, Sonotronic Inc. Tucson, AZ) in 120 individuals. I measured total length (1 mm TL), wet weight (0.01 kg WW) and documented sex based on the macroscopic observation of the gonads. I selected 30 individuals \geq 500-mm TL from each site for tagging. I anesthetized individual Blue Suckers in a 22 – 26 mg/L bath of Aqui-S 20E (New Zealand LTE. Lower Hutt, New Zealand), and began surgery once the individual lost equilibrium. Individuals were placed dorsal side up in a V-shaped cradle with freshwater supplied to the gills. I removed scales between the left pelvic ray and the anal vent and applied povidone iodine to the area. I inserted an acoustic tag through a 3 – 5 cm incision and closed the wound with 2 – 5 simple-interrupted sutures (2-0 PDO, 3/8 reverse cutting needle; Unify, AD Surgical, Sunnyvale, CA, USA). Acoustic tag weight (10 g in water) was 1.4% of the weight of the smallest tagged Blue Sucker (0.7 kg WW). Following surgery, I placed individuals inside a dark 250-L tank of frequently refreshed river water at a shaded location during recovery. Once a fish regained equilibrium and actively avoided human contact, I released it back into the stream.

I monitored Blue Sucker movement and habitat use during spring 2016 and 2017. I actively tracked each reach approximately weekly by traveling downstream via canoe while towing a hydrophone (TH-2 Sonotronic Inc. Tucson, AZ) at a speed slightly faster than the water current (7 – 9 kph). I scanned each of the 15 acoustic channels (69 – 83 kHz) with an acoustic receiver (USR-08, Sonotronic Inc. Tucson, AZ) for 3 sec and listened for tagged fish. Upon detection of an acoustic tag, I maneuvered into a position where I obtained the unique aural sequence, and marked the location via GPS (Garmin 76CSx, Garmin International, INC, Olathe, KS). I used a directional hydrophone (DH-4, Sonotronic Inc. Tucson, AZ) to identify the aural code when I was unable to isolate the tag signal from environmental noise or other tags. Additionally, I monitored Blue Sucker movement between portions of the mainstem Red River and my study sites using passive submersible ultrasonic receivers (SUR-03, Sonotronic Inc. Tucson, AZ). I suspended SURs \approx 0.5 m below the water surface from buoys that I anchored within 2-

km of the downstream extent of each study reach. I maintained 2 SURs at each location from winter 2015 to spring 2017, except Blue River. SURs placed in Blue River were not functioning upon retrieval.

I conducted sonar surveys in the field to obtain data used to map habitat features that I hypothesized would influence habitat selection of spawning Blue Suckers. I used side-imaging sonar (Lowrance HDS Gen 3 with LSS2 side-scan transducer; Tulsa, OK) to create sonar images and record depths of the streambed throughout my study reaches. I conducted my surveys by mounting the sonar to the bow of a canoe and traveling the thalweg of each reach downstream at 7 – 9 kph at moderate to high flows (*i.e.*, 60th – 80th percentile). I transferred sonar data to a computer for processing using Sonar TRX Pro software (version 16.1.6300.32523, Leraand Engineering Inc., Honolulu, HI). I created a gray-scale mosaic of the streambed from side-image sonar channels, and extracted spatially referenced depths from the down-scan sonar channel.

I used QGIS (version 2.18, QGIS Development Team, 2014) to organize spatial data and create habitat variables for habitat selection analysis. In QGIS, I delineated stream banks and straight or meander river sections within each study reach to serve as habitat units (referred to as river sections hereafter). I defined meanders as areas where the direction of the thalweg changed by $\geq 45^\circ$ and straight river sections comprised the area between meanders. I limited straight river sections to 400 m because tag-detection trials indicated that 200 m, up or downstream, was the upper range for detecting an acoustic tag in large, straight river sections. I numbered the river sections (downstream to the upstream extent) of each study reach to use as a surrogate for distance upstream of the Red River confluence. Using a combination of aerial photos and field observations, I documented the presence of riffles in the study reaches. I created a binomial variable where river sections possessing a riffle or adjacent to a riffle were given a value of 1, and other river sections were given the value 0. I visually identified woody debris and coarse substrate (> 64 mm) on the side-scan sonar images (Kaesler & Litts, 2010), and created polygons around patches of each habitat variable. I validated the presence coarse substrate and woody debris using field data in Red and Kiamichi rivers, and used aerial photos during low flows in Blue and Muddy Boggy rivers. I calculated the proportional area covered by either woody debris or coarse substrate within each river section to account for instream cover and potential spawning habitat (Moss *et al.*, 1983). I used spatially referenced depth data to calculate mean depth of

each river section. Side-imaging sonar surveys in each study reach were collected within a single day, so water depths of river sections were proportional within a study reach.

DATA ANALYSIS

I excluded individuals in the tailwater reach from analysis because they did not display season movement patterns. I previously determined that Blue Suckers tagged in tributaries displayed seasonal migration patterns, but Blue Suckers in the tailwater reach were unlikely to leave and instead behaved as residents (Chapter 2). Because fish in the tailwater reach rarely left the reach, I could not observe Blue Suckers returning during the spawning season, or confidently attribute habitat selection to spawning behavior. For those reasons, I excluded the tailwater reach from further analysis. However, three individuals tagged in the tailwater reach were included in my analysis because they used Muddy Boggy River during at least one spawning season.

Timing – I created predictor variables from temperature and discharge data to assess the relationship between the timing of Blue Sucker spawning migration and environmental cues. I obtained hydrology data from each stream recorded by USGS stream gages (Blue River 07332500, and Muddy Boggy 07335300; <https://waterdata.usgs.gov/ok/nwis/rt>) and USACE dam release data (<http://www.swt-wc.usace.army.mil/HUGO.lakepage.html>). I calculated mean magnitude of stream flow at each location during each week using R Statistical Software (vers. 3.4.5, R Core Team, 2018). Additionally, I obtained temperature data from a temperature logger (HOBO onset Pro v2, Onset Computer Corporation, Bourne, MA) that I placed in the Red River at the Highway 271 Bridge crossing, as water temperature data were not available in the tributaries (Fig. 3.1). I calculated the minimum temperature recorded each week and the degree days between the start of the spawning season and the end of each week. I used minimum temperature rather than mean temperature because the minimum temperature reflected the residual heat retained by the environment (Mathez, 2009).

I created generalized linear mixed models (glmm) to assess the effect of environmental variables on the timing of Blue Sucker spawning migrations into the tributaries. My response variable was the number of Blue Suckers detected for the first time in the season, each tracking week, during the spring season. I expected correlation between temperature variables, so I used Pearson's correlation to determine the

temperature variable with the strongest linear relationship with the number of Blue Suckers moving into tributaries. Using the 'lme4' package (Bates *et al.*, 2018), I fit a glmm with a Poisson distribution (Lawless, 1987; Bolker *et al.*, 2009). First, I created a null model with only random effects for location, year, and week, where week was an observation-level effect in my models to accommodate overdispersion (Bolker *et al.*, 2009). Next, I created two hypothesized models to test: 1.) the relationship between number of Blue Suckers and temperature, and 2.) the relationship between number of Blue Suckers and magnitude of discharge. Additionally, because migratory fishes often respond to both temperature and hydrology (Lucas & Baras, 2001), I considered an interactive effect of temperature and magnitude of discharge (Table 3.1). I included the location, year, and week as random effects in each model. I ranked models using AIC_c and considered the model with the lowest score the best supported by my data, provided that the score was > 2 points different (Burnham & Anderson, 2002; Johnson & Omland, 2004). I assessed the effect size of my models with the coefficient of determination (R²).

Habitat selection – I assessed habitat selection by Blue Suckers using two general approaches. First, I considered a type I habitat selection analysis, where use and availability were measured at the population level (Manly *et al.*, 2007). With the type I approach, I did not distinguish individuals within the population and I considered all measured habitat available to the Blue Sucker population. The disadvantage to this approach is that I ignored the potential correlations among multiple observations of an individual. I did not observe individuals often enough to create a temporal or spatial correlation (*i.e.*, > 20), and therefore did not violate assumptions of independence (Cryer & Kung-Sik, 2008). However, I completed a second analysis of habitat selection by individuals observed on ≥ 4 occasions using a type II approach where habitat use was measured independently for each individual and availability was measured at the population level. I observed individuals using multiple tributaries in my study, and consequently considered all measured habitat available to individuals in the population. The disadvantage to the type II approach is that it requires many observations of each individual. The two approaches complemented one another and provided more information regarding my data.

I assessed type I Blue Sucker spawning habitat selection using a forward model selection approach. I used the count of Blue Sucker observations in each river section as my response variable. To assess habitat selection I used generalized linear model

with a Poisson distribution and a forward selection approach guided by AIC_c (Bolker *et al.*, 2009). I calculated the AIC_c scores with the 'AICcmodavg' package (Mazzerolle, 2016). The initial model included only an intercept and a fixed effect for study reach. I individually added the habitat variables riffle, woody debris, coarse substrate, mean depth, and upstream distance to the initial model. The added variable that produced the lowest AIC_c score was retained and the process repeated until the addition of variables no longer reduced the AIC_c score (Bolker *et al.*, 2009). I included interactive effects and polynomial effects of retained continuous variables in subsequent iterations of the model selection process. I retained additional parameters if their inclusion resulted in a reduction of AIC_c score by ≥ 2 points (Burnham & Anderson, 2002; Johnson & Omland, 2004). I assesses the final model for satisfaction of linear model assumptions, overdispersion, and effect size using R².

I use a multivariate outlying mean index (OMI) analysis to determine type II habitat selection by 24 individuals. With OMI, the distance between mean habitat use by an individual and the mean habitat available to the population (marginality) was used to determine habitat selection by individuals (Dolédéc *et al.*, 2000; Calenge, 2011). I conducted the OMI analysis using the 'adehabitatHS' package (Calenge, 2011). In the OMI analysis, two matrices were constructed: Z, to represent the available habitat with n river sections (rows) and p habitat variables (columns), and Y, to represent used habitat with n river sections (rows) and t individuals (columns). A principal components analysis (PCA) was used to determine the correlations among and the relative availability of habitat variables in matrix Z. With the PCA analysis, the river sections with the mean habitat conditions make up the center of the multivariate space, and each variable represents a dimension within that space. Next, river sections in matrix Y were weighted for each individual based on the proportional use of the river sections by a given individual. From this, the mean habitat use by each individual was determined, and the marginality of each individual was calculated. An eigen-analysis was used to create an x-axis and y-axis (*i.e.*, principle components) that explain the most and the second most marginality in habitat conditions among river sections. The dimensions associated with habitat variables were plotted with their origin at the intersection of x and y axes, and individuals were plotted based on their mean habitat use within multivariate space. This allowed a visual assessment of habitat selection by individuals relative to availability (Calenge, 2011).

RESULTS

I successfully implanted acoustic tags into 119 Blue Suckers, and redetected 37% of those individuals in a tributary at least once during my study (see Appendix B). Of the 120 Blue Suckers tagged, one Blue Sucker in the Red River tailwater perished before being released. I suspect that the mortality was due to a gill injury during weighing and not due to my surgical methods. Females displayed light tuberculation around their head and fins, and were apparently gravid (*i.e.*, protruding abdomen), but did not display gametes when pressure was applied to the abdomen. Males had heavy tuberculation and readily expressed gametes with light abdominal pressure. I tagged ten juvenile Blue Suckers that I expected were female because I observed several mature males of the same size, but gravid females tended to be larger (> 530 mm TL). In the tributaries, I observed 80% of redetections in 2016, and I suspected that low flows in 2017 resulted in the paucity of redetections in the tributaries (Fig. 3.2).

Timing – I determined that the combination of cool temperatures and high flows resulted in the greatest expected abundances of Blue Suckers moving into the tributaries. The two temperature variables had a weak relationship with the abundances of Blue Suckers in the tributaries. However, the relationship with weekly mean, minimum temperature was slightly stronger than the relationship with degree days ($r = -0.09$ vs. 0.04 , respectively). Weekly mean streamflow magnitude was moderately correlated with Blue Sucker abundance ($r = 0.42$). The model with temperature and magnitude interaction scored 7.2 AIC_c points lower than the second ranked model and explained a moderate amount of the variation in my data (conditional $R^2 = 0.42$; Table 3.1). The model indicated that high flows occurring at low temperatures, or earlier in the season, resulted in a greater expected abundance of Blue Suckers entering the tributaries (Fig. 3.3). Conversely, high flows occurring at moderate or warmer temperatures had either little effect or a negative effect on Blue Suckers moving into the tributaries.

Habitat selection – My type I habitat selection analysis indicated that Blue Suckers in the tributaries positively associated with riffles and woody debris. The final model had $R^2 = 0.17$, and included the fixed effects for riffle presence, woody debris, and study reach (Table 3.2). I estimated a higher abundance of Blue Suckers in the Kiamichi River ($\beta_{\text{Kiamichi}} = 2.19 \pm 0.32$ SE) than in either Blue or Muddy Boggy rivers ($\beta_{\text{intercept}} = -2.44 \pm 0.22$ SE & $\beta_{\text{Muddy}} = -0.25 \pm 0.30$ SE, respectively). Blue Suckers were more likely to occur

in a river section that possessed a riffle or was adjacent to a riffle ($\beta_{\text{riffle}} = 0.90 \pm 0.22 \text{ SE}$) and were positively associated with woody debris ($\beta_{\text{wood}} = 0.38 \pm 0.14 \text{ SE}$; Fig. 4). However, the amount of woody debris in my study was typically 0 – 20% and the 95% confidence intervals reflected the increasing uncertainty in river sections with > 30% woody debris (Fig. 3.4).

I included 10 individual Blue Suckers with ≥ 4 redetections in my type II habitat selection analysis. Combined I included 56 observations, with 55% in the Kiamichi River, 30% in the Blue River, and 15% in the Muddy Boggy River. The relatively limited number of detections in the tributaries was due to low residency time in the study reaches (≈ 2 weeks). Additionally, the low flows in 2017 (Fig. 3.2) likely contributed to the detection of only 1 individual in Blue River, 2 individuals in Muddy Boggy River and 3 individuals in the Kiamichi River .

Differences in habitat availability among the different study reaches was apparent in the OMI analysis. The x-axis accounted for 70% of the marginality and was negatively associated with coarse substrate and mean depth and positively associated with distance upstream and woody debris (Fig. 3.5). Fish on the left side of the plot (negative loadings) selected river sections within 50 river sections (25 – 30 rkm) from the confluence, and with < 10% woody debris and, mean depths > 2 m, whereas fish on the right side of the plot (positive loadings) selected more woody debris, shallower depths, and moved further upstream. The five individuals with the strongest negative loading were from the Kiamichi River where available upstream distances were < 60. Coarse substrate was typically sparse (< 10%), and had little effect on habitat selection. The y-axis explained an additional 18% of the marginality, and was positively associated with riffle presence, and negatively associated with depth. The upper point cloud reflected riffle presence and the lower point cloud illustrated riffle absence. Most of the fish spent time in river sections with and without riffles, and the position of an individual between the point clouds reflected the proportional amount of time that individual spent in either habitat. Fish E was the only individual that was not observed in or near a riffle, and was the only fish with unknown sex at the time of tagging (Table 3.3). The three individuals most strongly associated with riffles were males (C, W, D), whereas females spent proportionally more time in river sections without riffles.

DISCUSSION

I found Blue Sucker were more likely to migrate to tributaries during the spring when flow pulses occurred in conjunction with cool water temperatures. Blue Suckers were less likely to initiate migrations as water temperatures warmed, contradicting the 18° C spawning temperature previously reported for Blue Sucker in Red River, Arkansas, (Layher, 2007). Other Blue Sucker populations initiated spawning movements when temperatures were as low as 10 °C (e.g., Missouri, Vokoun *et al.*, 2003; Nebraska, Neely *et al.* 2009). It was possible that Blue Suckers in my study were ready to spawn once water temperatures exceeded 10 °C, but flow conditions were not appropriate at that time. In the absence of a flow pulse, Blue Suckers may have chosen to spawn under low flow conditions, or resorbed their oocytes and spawned the next year. Similarly, delayed spawning migrations of Atlantic Salmon *Salmo salar* have been attributed to hydrology (Thorstad *et al.*, 2008). Intrinsic factors (e.g., maturation stage and hormones) were thought to motivate fish movements near the end of the spawning season (Thorstad *et al.*, 2008). In the Blue and Muddy Boggy rivers, the timing of flow pulses were dependent on precipitation; however, dams in the Red and Kiamichi rivers may alter the timing of natural flow pulses (Young *et al.*, 2011). Dam releases that do not consider the effects of timing and magnitude of the flow pulse can be detrimental to spawning fish (Young *et al.*, 2011). Although my observations are unique and occurred over three spawning seasons, my data are based on a limited sample size and should be interpreted with caution (*i.e.*, untagged fish representing a large portion of the population are unobservable).

Although my sample size was limited, I have multiple lines of evidence suggesting riffles are important to spawning Blue Suckers and it is supported by findings from other studies. First, I captured most of my fish for tagging in riffles. Although it was possible that my electrofishing equipment was more efficient in the shallow water and rocky substrate associated with riffles (Reynolds, 1996), I sampled other shallow areas of the stream and did not capture large numbers of fish. Second, river sections with riffles or adjacent to riffles accounted for 47% of Blue Sucker use observations compared to only 26% of the available river sections being associated with riffles. The result is likely more complex as suggested by my type II analysis where individual Blue Suckers spent time in river sections with and without riffles. However, my limited data suggest future efforts might examine differences among sex as several of the male fish used riffles or areas adjacent to riffles more frequently than females. The males of

several catostomid species are territorial during the spawning season (e.g., Creek Chubsucker *Erimyzon oblongus*, Page & Johnston, 1990; Greater Redhorse *Moxostoma valenciennesi*, Cooke & Bunt, 1999).

I detected a positive association of Blue Suckers with woody debris, but this relationship may not be limited to spawning activity. Woody debris is not typically associated with Blue Sucker spawning habitat (e.g., Moss *et al.*, 1983; Vokoun *et al.*, 2003). However, woody debris was important in predicting the presence of Southeastern Blue Sucker *Cycleptus meridionalis* in the Pearl River, Louisiana, at any time of year (i.e., spawning or non-spawning, Oliver *et al.*, 2017). Woody debris can provide refuge and forage opportunities to fish (Pusey & Arthington, 2003). Blue Suckers likely seek refuge to conserve energy during spawning migrations (Lucas & Baras, 2001), but foraging behavior during the spawning season is unknown for the species. I frequently captured Blue Suckers below brush piles in swift water when electrofishing in the Kiamichi River for a separate study (chapters 3 and 4) and it is possible that Blue Suckers were using the habitat as refuge from the strong current.

Coarse substrate was commonly associated with Blue Sucker spawning habitat in previous studies (Vokoun *et al.*, 2003, Lyons *et al.*, 2016, Oliver *et al.*, 2017), but I did not detect a relationship with coarse substrate in my analysis. Coarse substrate was common in the Kiamichi River (median coverage 29%), but was rare in Blue and Muddy Boggy rivers (median coverage 0 & 3%, respectively), and I more commonly observed gravel substrate in riffles in those reaches. Gravel substrate was difficult to distinguish from finer substrate in side-scan sonar images (Hamill *et al.*, 2018), and because I was unable to identify gravel with any degree of certainty, I did not include it in my analysis.

I used acoustic tags in my study due to high water conductivities (800-2000 μS) in the Red River, although the technology has some limitations in lotic environments. Acoustic tags are difficult to detect in swift water environments that Blue Suckers were hypothesized to inhabit; however, water conductivities $> 800 \mu\text{S}$ cause the rapid attenuation of radio waves making the use of radio tags impractical in my study (Adams *et al.*, 2012). Alternatively, the environmental noise associated with turbulent water and moving substrate in riffles make acoustic tags difficult to detect (Adams *et al.*, 2012). In the tributaries, riffles were small and I was often able to detect fish from the bank using the directional hydrophone. On one occasion, I detected three Blue Suckers in a single riffle in Blue River by standing on the bank with the directional hydrophone. It is likely that I missed individuals due to the complexity of some riffles. I cannot quantify the

presence of fish that were undetected, and a potential consequence was that fish inhabiting noisy environments were underrepresented in my data.

It is apparent that Blue Suckers moved throughout the lower Red River, and used the tributaries during the spawning season. Although the number of Blue Suckers observed using the tributaries was small, the unregulated Muddy Boggy and Blue Rivers appear to possess suitable spawning habitat and natural flows. The water conductivities in these tributaries were lower than in the Red River ($< 500 \mu\text{S}$), and further research into habitat selection in the tributaries should implement radio tags and a finer temporal resolution (*i.e.*, daily). Additionally, dam management may be important in low-flow years, when migratory fishes are attracted to dam releases (Bunt, 2001). My results indicate that Blue Suckers were more likely to move into tributaries in conjunction with large flow pulses occurring early in the spawning season (Feb – March) compared to flow pulses occurring later in the spring (April – May). Large flow pulses created by dam-releases are typically due to heavy precipitation, leaving managers with little choice in the timing of such dam-releases. However, managers should consider how the rate of change in discharge affects fish that may be attracted to the tailwater to spawn (Young *et al.*, 2011).

Table 3.1 Candidate models considered for my analysis of Blue Sucker migration timing. Each model is provided, Y_{ijk} is the expected abundance of Blue Suckers entering location j during week i of year k , β_0 is the estimated model intercept, β_1 is the estimated coefficient associated with either the mean, minimum temperature during week i (temperature) or the mean stream discharge (magnitude) during week i , β_2 and β_3 are the estimated coefficients associated with magnitude and the interactive effect of temperature and magnitude, respectively. τ_1 is the random effect of tributary j (location), τ_2 is the random effect of year k , and τ_3 is the random effect of week i . All random effects were distributed $\sim N(0, \sigma^2)$, and the observation level effect of week served as the over-dispersion parameter associated with the negative binomial distribution. Additionally, I provide the AIC_c scores associated with each model, the difference in AIC_c score between a given model and the top ranked model (ΔAIC_c), and the conditional R^2 value.

Name	Model	AIC_c	ΔAIC_c	R^2
interaction	$Y_{ijk} = \beta_0 + \beta_1 temperature_i + \beta_2 magnitude_i + \beta_3 temperature_i * magnutude_i + \tau_1 location_j + \tau_2 year_k + \tau_3 week_i$	183.2	0	0.42
temperature	$Y_{ijk} = \beta_0 + \beta_1 temperature_i + \tau_1 location_j + \tau_2 year_k + \tau_3 week_i$	190.4	7.2	0.49
magnitude	$Y_{ijk} = \beta_0 + \beta_1 magnitude_i + \tau_1 location_j + \tau_2 year_k + \tau_3 week_i$	192.0	8.8	0.31
Null	$Y_{ijk} = \beta_0 + \tau_1 location_j + \tau_2 year_k + \tau_3 week_i$	194.3	11.1	0.38

Table 3.2 Forward model selection process for type I habitat selection analysis. The best supported models from each step is provided, Y_{ij} is the expected abundance of Blue Suckers in river section i of location j , β_0 is the estimated model intercept, and the subsequent β_x 's are the coefficients associated with a tributary (location), riffle presence (riffle), or woody debris (%) (wood). The final model is shown in bold.

Name	Model	AIC _c	ΔAIC _c
Step 0	$Y_{ij} = \beta_0 + \beta_1 location_{ij}$	474.0	0
Step 1	$Y_{ij} = \beta_0 + \beta_1 riffle_i + \beta_2 location_j$	461.3	12.7
Step 2	$Y_{ij} = \beta_0 + \beta_1 riffle_i + \beta_2 wood_i + \beta_3 location_j$	455.9	18.1
Step 3	$Y_{ij} = \beta_0 + \beta_1 riffle_i + \beta_2 wood_i + \beta_3 riffle * wood_i + \beta_4 location_j$	456.5	17.5

Table 3.3 Biological data of Blue Suckers included in my Type II habitat selection analysis. The fish code corresponds to the letter assigned to individual fish in Fig. 5. Demographics are coded as Sex (M = male, F = female, or U = unknown); TL (total length); and WW (wet weight). Tag Location refers to the study reach in which an individual was initially captured and tagged.

Fish Code	Sex	TL (mm)	WW (g)	Tag Location
C	M	511	1050	Blue
D	M	515	1200	Kiamichi
E	U	549	1130	Muddy Boggy
G	F	610	2260	Kiamichi
I	M	518	900	Kiamichi
M	F	660	2570	Muddy Boggy
S	F	519	1150	Blue
T	F	625	2350	Kiamichi
V	M	563	1350	Blue
W	M	533	1340	Kiamichi

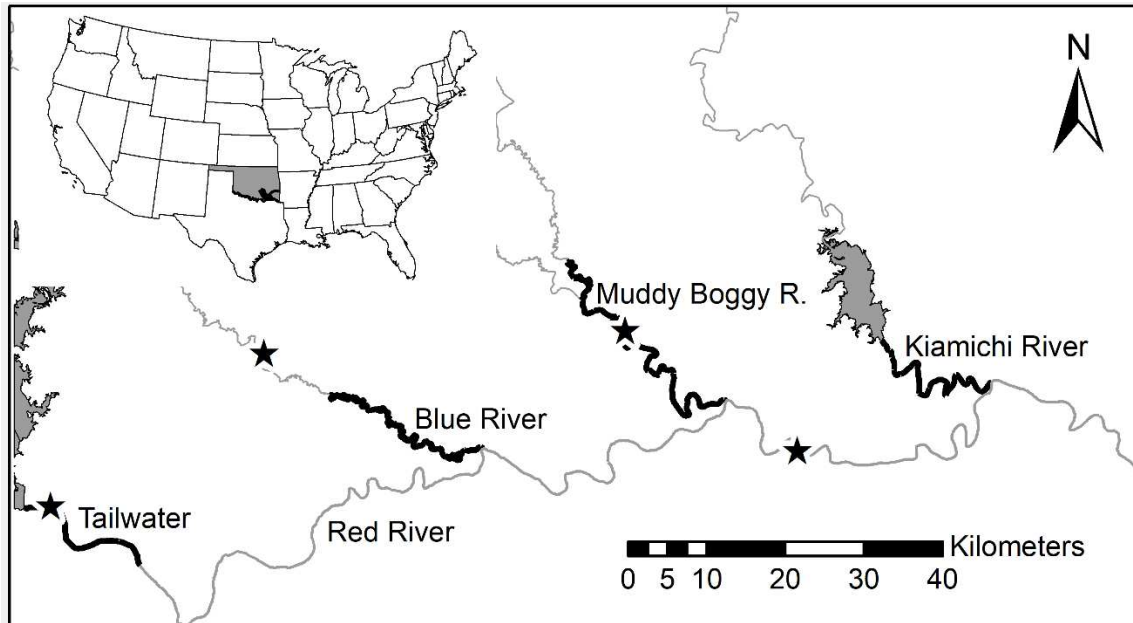


Fig. 3.1 The location of my movement timing and spawning habitat selection study. The reaches where Blue Suckers were tagged and tracked are in bold. Stars indicate the location of the U.S. Geological Survey stream gages. The temperature logger was located at the star in Red River between Muddy Boggy and Kiamichi rivers.

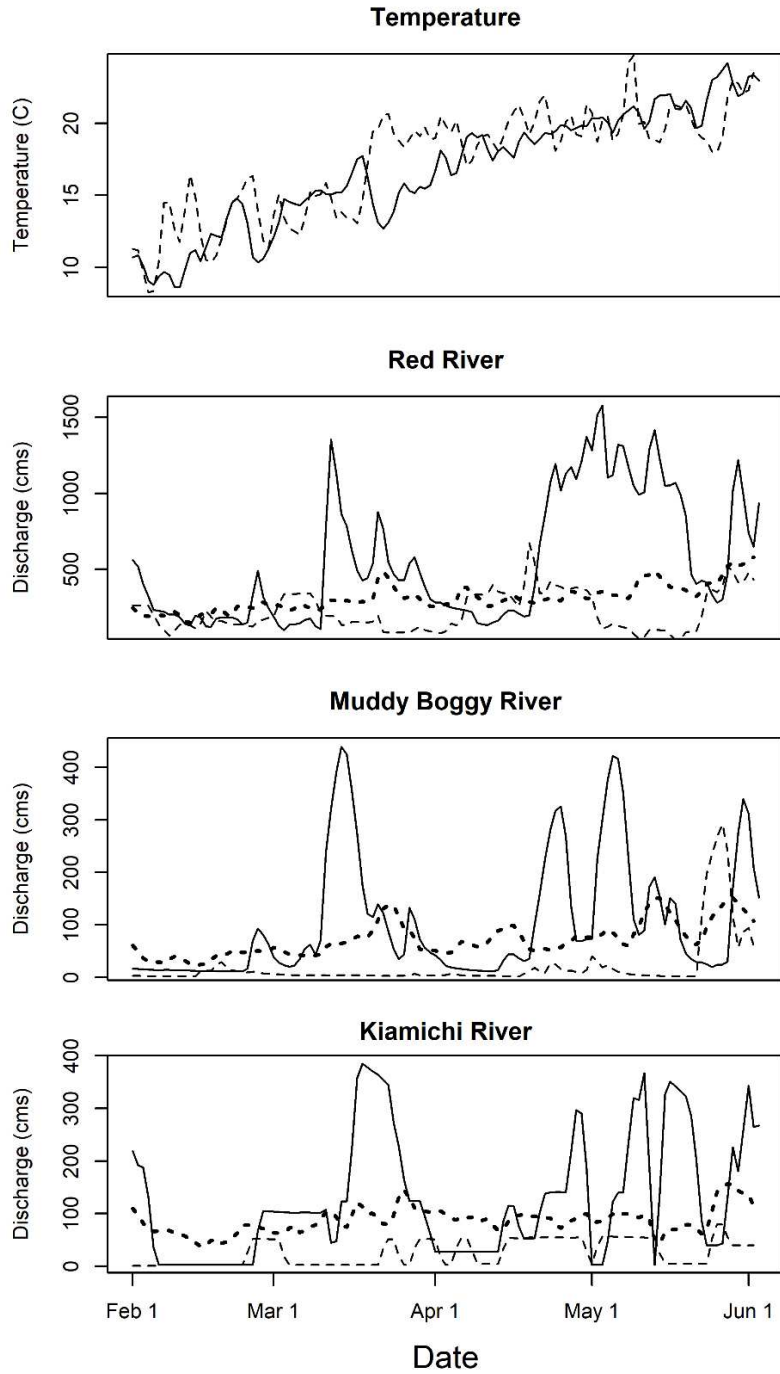


Fig. 3.2 Water temperature at the Red River (Hwy 271 bridge, collected using a HOBO, Onset), and hydrographs representing stream discharge (cms) in the Red, Muddy Boggy, and Kiamichi rivers during the early-seasons 2016 (solid), 2017 (dashed), and the 20-year average (1998 – 2017, dotted) (https://waterdata.usgs.gov/ok/nwis/uv?site_no=07335500, accessed 2/27/18).

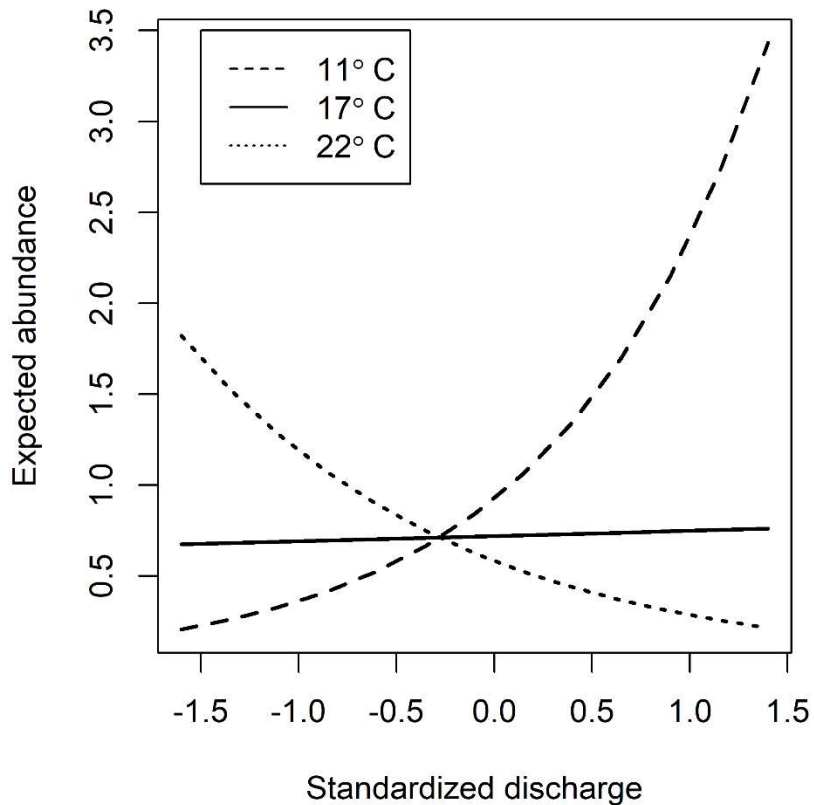


Fig. 3.3 The relationship between environmental conditions and the timing of Blue Suckers entering the tributaries during the spawning season. Because the tributaries were different sizes, and the magnitude of stream discharge was not equal among tributaries, I standardized the streamflow magnitude data on the x-axis. Therefore, values on the x-axis represent the mean (0) and standard deviations from the mean. To display the interactive effect of temperature and magnitude, I included a line to represent temperature of 11° C (dashed), 17° C (solid), and 22° C (dotted). The expected abundance of Blue Suckers moving into the tributaries in a given week is indicated on the y-axis.

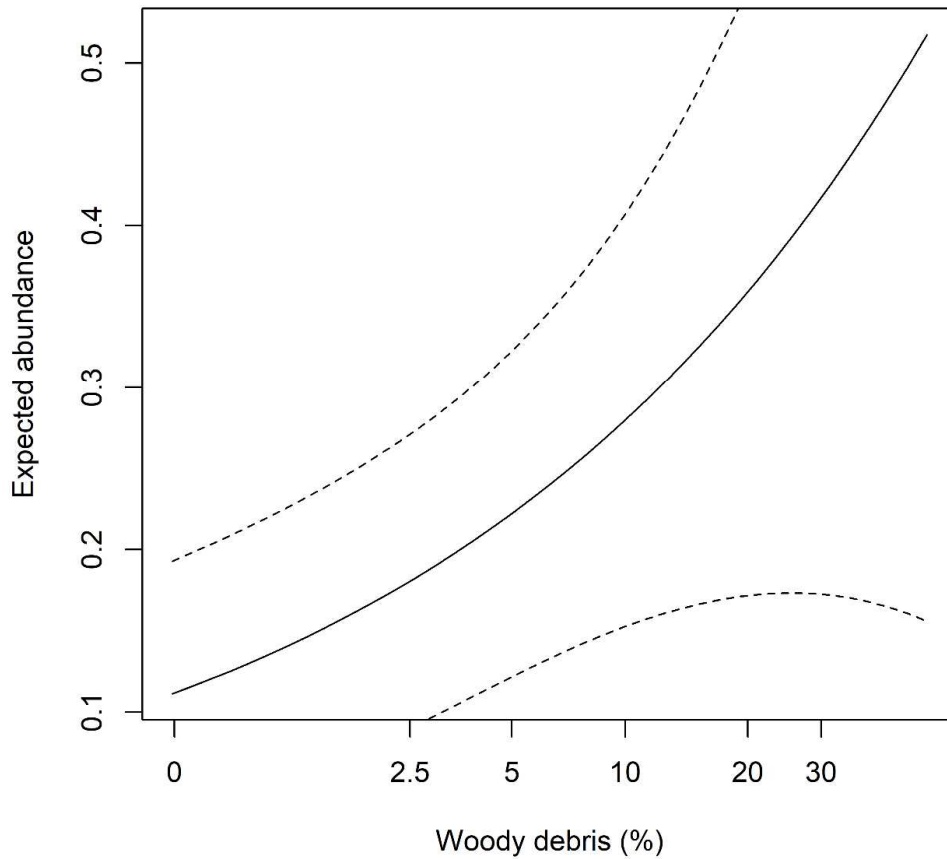


Fig. 3.4 The relationship between woody debris (natural-log scale) and the expected abundance of Blue Suckers in a river section in Blue River when a riffle was present. The solid line indicates the mean estimate and the dashed lines represent 95% confidence intervals. The relationship was similar in each study reach, but more fish were expected to occur in the Kiamichi river than in the Blue or Muddy Boggy rivers. Fewer Blue Sucker were expected to occur in river sections without riffles.

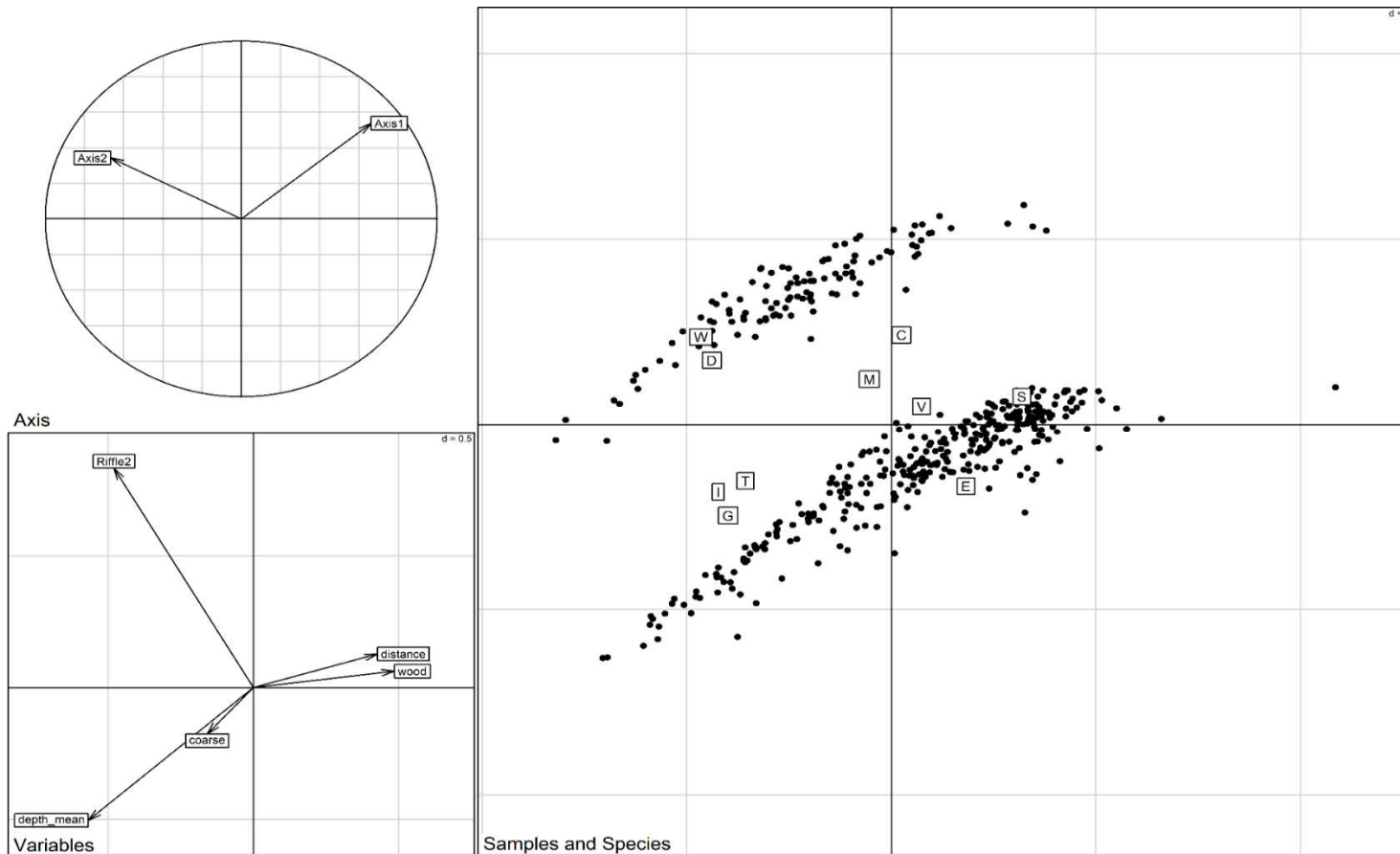


Figure 3.5 Results from the outlying mean index, type II habitat selection analysis. The circular panel on the top left shows the relationship of the x and y axes in multivariate space, the relationship of the environmental variables is shown on the bottom left, and the large panel on the right shows the relationship of habitat selected by Blue Suckers compared to available habitat. The dots indicate river sections available to the population, and letters indicate the mean habitat selection by individuals. See Table 2 for information on individual Blue Suckers.

CHAPTER IV

POPULATION DYNAMICS

INTRODUCTION

Blue Sucker *Cycleptus elongatus* is a broadly distributed large-river obligate with a cryptic life history (Burr & Mayden, 1999). The species is considered potamodromous as individuals spend the summer and winter in large rivers and can migrate > 300 km to tributaries to spawn in the spring (Morey & Berry, 2003; Neely *et al.*, 2009). In the tributaries, Blue Suckers seek out swift water and coarse substrate where they aggregate and spawn (Moss *et al.*, 1983; Vokoun *et al.*, 2003; Lyons *et al.*, 2016). Blue Suckers may be locally abundant at spawning locations; however, the swift and deep water they occupy make sampling and population evaluations difficult (Moss *et al.*, 1983; Morey & Berry, 2003; Layher, 2007). Blue Sucker eggs have not been observed in the wild (Adams *et al.*, 2006), but observations in a hatchery indicate that larval Blue Suckers hatch and absorb the yolk sac within 10 days of fertilization in 20° C water temperature (Semmens, 1985). The presence of larval Blue Sucker is rarely documented, but this life stage is hypothesized to drift to slack-water environments where they quickly grow (Yeager & Semmens, 1987; Fisher & Willis, 2000; Adams *et al.*, 2006; Bednarski & Scarnecchia, 2006). Juvenile Blue Suckers are also rare in fish samples; thus, their habitat use and movement patterns are unknown (Hand & Jackson, 2003; Morey & Berry, 2003; Bacula *et al.*, 2009). Blue Sucker growth is most rapid before reaching adulthood at 3 or 5 years for males and females, respectively (Rupprecht & Jahn, 1980). Blue Sucker may live > 30 years in some locations and can grow > 800 mm in total length (TL, Hand & Jackson, 2003; Bednarski & Scarnecchia, 2006).

Blue Sucker is considered vulnerable to extinction (Jelks *et al.*, 2008), though the status of the species is highly variable across the geographic range. Bacula *et al.* (2009) concluded the Blue Sucker population in the Wabash River, IN was stable after conducting a thorough assessment. Successfully reproducing populations have also been documented in tributaries of the Missouri and Mississippi rivers (*e.g.*, Montana, Bednarski & Scarnecchia, 2006; Wisconsin, Lyons *et al.*, 2016; and Missouri, Vokoun *et al.*, 2003). Alternatively, populations in Pennsylvania are extinct, and Blue Suckers are considered critically imperiled in New Mexico, Ohio, West Virginia and Nebraska (Burr & Mayden, 1999; NatureServe, 2018). Lastly, population status of Blue Sucker in major tributaries of the Southern Great Plains draining to the southern extent of the Mississippi River is unknown.

Flow alteration and fragmentation are two intertwined impediments thought to disrupt the successful completion of Blue Sucker life history. Extensive damming across the Blue Sucker distribution is associated with flow suppression, encroachment by woody vegetation, and excessive deposition of fine sediment thereby reducing suitable Blue Sucker spawning habitat (Collier *et al.*, 1996). In addition to the physicochemical changes caused by dams (see overview by Olden & Naimen, 2010), flow alterations may disrupt spawning cues for migratory fishes (Young *et al.*, 2011) and reduce fish recruitment (*e.g.*, Clanwilliam Yellowfish *Barbus capensis*, King *et al.*, 1998; Robust Redhorse *Moxostoma robustum*, Grabowski & Isely, 2007; Golden Perch *Macquaria ambigua*, Rolls *et al.*, 2013). Dams are also responsible for fragmentation of river ecosystems, and fragmentation can inhibit gene flow among migratory populations, as seen in Blue Sucker (Bessert & Orti, 2008), and Mountain Sucker, *Pantosteus jordani* (Bertrand *et al.*, 2016). Given the abundance of altered riverscapes across the globe (Vörösmarty *et al.*, 2010), documenting life histories and monitoring the effects of stream alterations on Blue Suckers and sympatric species is critical to the conservation of native fishes (Cooke *et al.*, 2012; Arthington *et al.*, 2016).

Population dynamics and basic life history data are critical to management and conservation of populations, but data are limited for many riverine fishes (Cooke *et al.*, 2012). Because fish populations are difficult to observe, managers use vital statistics for insight into the dynamics of populations (Allen & Hightower, 2010). Vital statistics are estimates from demographic data to describe mortality, growth, and recruitment rates in a population (Allen & Hightower, 2010). Although the observation of population dynamics may take > 20 years of data (Ricker, 1975), population models provide a link between

vital statistics and the trajectory of a population (Caswell, 2001). In the current study, I determine the vital statistics of two Blue Sucker populations and evaluate the vulnerable conservation ranking of Blue Sucker in Red River, Oklahoma. In Oklahoma, knowledge of Blue Suckers in the Red River is sparse, consisting only of occasionally documented occurrences from the 1940's – 1990's (Burr & Mayden, 1999; MARIS database, <https://www.sciencebase.gov/catalog/item/51c45ef1e4b03c77dce65a84>, 10/26/18), and an opportunistic capture in a tributary to the Red River in 2014. The goal of my study was to establish basic population information to use as a foundation for future monitoring efforts in the Red River, and to inform the conservation ranking of Blue Suckers in North America. The objectives were to: 1.) estimate the necessary vital statistics from empirical data to model the growth of two spawning populations, 2.) use age-based population models to project the growth trajectories of each population, and 3.) explore the modeled relationship between observed variation in recruitment and variable annual spawning activity.

METHODS

STUDY AREA

My study was focused on two populations of Blue Sucker located in the lower Red River basin. The lower Red River begins as a tailwater downriver of Lake Texoma, a 36,000-ha hydropower reservoir. I designated the upper 20-rkm of the lower Red River as the tailwater reach and sampled the first population from that location (Fig. 4.1). In the tailwater reach, the dam release of sediment-starved waters has resulted in \approx 20 rkm of degraded stream bed and coarse substrates are now abundant (in an otherwise sandbed river). The remaining downriver channel of Red River is narrower when compared to historic morphometry of a southern Great Plains River (see Brewer *et al.*, 2016). The Kiamichi River is one of three major tributaries (\geq 4th order; Strahler, 1957) draining to the lower Red River within Oklahoma. The Kiamichi River is impounded by Hugo Dam 28-km upriver of the Red River confluence, and I sampled my second population from the Kiamichi River within 2 km of Hugo Dam.

FIELD SAMPLING

I sampled Blue Suckers during their spawning migrations, when they were locally abundant, to conduct my population assessments. In 2016, I sampled the Red River for one week each in early March and late May. In 2017, I increased my sampling effort and

sampled the Red and Kiamichi rivers (Fig. 4.1) once weekly from February 17 – March 30, and twice weekly from April 1 – May 2. I increased sampling effort in April, because Blue Suckers were more abundant during that time. My population assessments of Blue Sucker were primarily based on fish captured in the Red and Kiamichi rivers in 2017; however, length-at-age data from the Red River tailwater collected in 2016 were used in the growth analysis.

At each location, I sampled Blue Suckers using boat electrofishing to collect demographic data for my population assessments. I sampled swift-water habitats with a single anode, 5.0 GPP, boat-mounted electrofishing unit (Smith-Root, Vancouver, Washington). I shocked while drifting the boat with the current, and a chase boat was used to collect Blue Suckers that surfaced behind the shock boat (Moss *et al.*, 1983, Layher, 1998). I adjusted the electrofisher amperage output according to the water conductivity at each site (Miranda, 2009). I typically used 60 kHz frequency, although low conductivities in the Kiamichi River (<75 μS) occasionally required 120 kHz to achieve the target output (2.5 – 4 amps, 1000 v/high setting; Miranda, 2009).

Upon capturing Blue Suckers, I recorded data determine the vital statistics of the populations. I measured the total length (1 mm, TL) and weight (0.01 kg WW) of each Blue Sucker, and determined sex based on gamete expression or secondary sex characteristics (*e.g.*, pronounced tubercles coving the body for males, or tubercles only around head and fins and protruding abdomen of females; Lyons *et al.*, 2016). The leading pectoral ray was removed for age estimation. I collected pectoral rays because the method is non-lethal and is as precise as otoliths for aging Blue Suckers (Rupprecht & Jahn, 1980; Labay *et al.*, 2011; Acre *et al.*, 2017). In 2016, I removed the leading pectoral fin ray of every other Blue Sucker captured in each 25-mm length bin to create a length-age key. However, my data from 2016 indicated no length-age relationship for adult Blue Sucker, so I began collecting pectoral rays from all Blue Suckers in 2017. Pectoral rays were stored in coin envelopes (#1 Brown Kraft Coin Envelopes, Staples, Farmingham, MA) and allowed to air dry until I returned to the lab. In 2017, I also collected ovaries from the first three females captured in each 25-mm length bin at each location. I preserved the ovaries in 10% formalin.

LABORATORY PROCESSING

I sectioned and aged pectoral fin rays to determine the age structure of the two populations. Pectoral fin rays were dried in a dehydrator for ≥ 1 week before I set the

proximal 7 – 9 mm portion of the ray in epoxy resin. Once the resin cured, I sectioned each ray (0.7 – 1.1 mm) from the proximal end, using a Bueler low-speed Isomet® saw (Buellers Isomet 1000, Lake Bluff, IL), and then mounted three sections to a microscope slide using a nitrocellulose-ethyl acetate solution (*i.e.*, clear fingernail polish; Bednarski & Scarnechia, 2006). For each fish, two observers independently enumerated the annuli (opaque rings) with 40x magnification and transmitted light. When readers disagreed on the age of an individual, a consensus age was obtained in collaboration with one another (Casselmann, 1983; Panfili *et al.*, 2002).

Absolute fecundity (total quantity of oocytes possessed by an individual during the current spawning cycle) was estimated for each fish by subsampling and weighing oocytes from the preserved ovaries. For each fish, I collected subsamples of 100 oocytes from the anterior, middle, and posterior portion of each ovary (Daugherty *et al.*, 2008). I weighed individual subsamples using an electronic balance (0.001 g, Mettler AT250, American Instrument Exchange, Inc., Haverhill, MA), calculated the average subsample weight, and divided by 100 to determine the average weight of each oocyte. I then estimated the absolute fecundity for an individual by dividing the total weight of an ovary by the average oocyte weight for an individual (Daugherty *et al.* 2008).

It is common for individual fish to possess oocytes in multiple developmental stages (Corriero *et al.*, 2003), and the inclusion of multiple sizes of oocytes in the ovary can affect the accuracy of my gravimetric estimation methods. Therefore, I did not distinguish developmental stages of oocytes in my subsamples, with the exception of perinuclear and atretic stages (terminology from Corriero *et al.*, 2003). I did not enumerate perinuclear stage oocytes because they did not represent the viable oocytes for the present spawning cycle, and I could not differentiate atretic oocytes for enumeration. I did not remove the weight associated with perinuclear oocytes, because I assumed the weight was negligible. Additionally, atretic oocytes represented potential oocytes from the current spawning cycle and although I was unable to enumerate atretic oocytes, their mass was included in my absolute fecundity estimates.

VITAL STATISTICS

I estimated vital statistics from empirical data to create the necessary parameters for age-based population models. The vital statistics for populations typically include mortality, growth and recruitment rates because monitoring these parameters can indicate problems in the population stability (Allen & Hightower, 2010). I estimated four

parameters related to population vital statistics: the probability of an oocyte recruiting to age-1 (S_1), annual probability of surviving post-recruitment (S_2), age-specific probability of reaching maturity ($p(a)$), and age-specific fecundity ($f(a)$; Caswell, 2001). The estimated parameters dictate the portion of each age cohort that survives, matures, and contributes to the next generation.

I constructed catch curves to estimate the annual probability of survival for recruited fish (S_2) and variability in recruitment for the two Blue Sucker populations. I only considered data from 2017 in the catch-curve models, because age data collected in 2016 were subsampled and did not represent the relative abundances of age classes in the population. Using the 'FSA' package (Ogle, 2017) in Program R (version 3.4.4, R Core Team 2018), I developed catch curves by regressing the \log_e -transformed abundance of fish in each age class against age. I considered the age corresponding to the peak of the age-frequency plot (modal age class) the earliest age that recruited to my gear (Miranda & Bettoli, 2007). Because the modal age class could be due to a strong age class, rather than the youngest recruited age class, I constructed additional catch-curve models that included the age class prior to the modal age class (Miranda & Bettoli, 2007). I accepted the model with the largest coefficient of determination (R^2) as the best model (Zar, 2010). Although age classes > 20 years were rare in my sample, I did not exclude them from the analysis because they supported the declining trend of the younger age classes. I used the slope of the models to provide estimates of the instantaneous mortality rate (Z) and calculate the annual survival rate (S_2 ; Ricker, 1975), where

$$S_2 = e^{-Z}.$$

The R^2 value from the resulting models provided the recruitment coefficient of determination (RCD), a measure of recruitment variability (Isermann *et al.*, 2002).

To determine the age-specific fecundity rates ($f(a)$), I developed multiple hypotheses regarding the relationship of absolute fecundity with either size or age. Because females from the Red River tailwater and the Kiamichi River may have represented different populations, I included a random effect for site in each hypothesized model. I developed four generalized linear mixed models: 1) fecundity predicted by age, 2) fecundity predicted by TL, 3), second-order polynomial relationship of fecundity with age and 4) second-order polynomial relationship between fecundity and TL. I ranked all models using AIC_c and considered the lowest AIC_c score my top model (Burnham & Anderson 2002; Johnson & Omland 2004). If AIC_c scores of top ranked

models differed by < 2 points, I retained the simpler model in favor of parsimony (Burnham & Anderson, 2002; Johnson & Omland, 2004). I checked that the top model met the assumptions of a generalized linear model (Zar, 2010).

I modeled the relationship between TL and age because I was creating an age-based model and fecundity was potentially linked to TL rather than age. Although it was possible to exclude TL in my model by modeling the relationship between fecundity and age, incorporating TL provided a better understanding of population dynamics and uncertainty in my age-based model. I created von Bertalanffy growth curves for each population using the FSA package in R (version 0.8.20; Ogle, 2018), but high variability in length at age in Red River and lack of fish aged < 6 in Kiamichi River resulted in erroneous models (e.g., estimated L^∞ was not observed in my data; Fig. 4.2). Therefore, I used a multiple hypothesis approach to model the relationship between TL and age to compare adult growth rates between sexes and populations. I included Blue Sucker age data from both 2016 and 2017 in my growth analyses, because they were based on individual length-at-ages and not relative abundances within age classes. I considered four linear models: 1.) TL of adult Blue Suckers, predicted by the interaction of age and location, 2.) TL predicted by the interaction of age and sex, 3.) TL predicted by sex and the interaction of age and location, and 4.) TL predicted by location and the interaction of age and sex. I ranked the four models using AIC_c and the best model was considered to have the lowest AIC_c score. If AIC_c scores were similar (< 2 point difference) I selected the simpler model, provided that the assumptions of a generalized linear model were met (Burnham & Anderson, 2002; Johnson & Omland, 2004; Zar, 2010).

To incorporate TL into the fecundity parameter, I developed a function in R to reflect the possibility that fecundity ($f(a)$) was a function of size rather than age. I used my top model of adult Blue Sucker growth to estimate the TL of an individual based on its age,

$$\widehat{TL}_i = \beta_0 + \beta_1 gender_i + \beta_2 Age_i + \beta_3 Location_i + \beta_4 Age_i * Location_i$$

where \widehat{TL}_i is the expected TL of fish i , β_0 is the model intercept, and $\beta_1 - \beta_4$ are the estimated coefficients associated with the age, sex, capture location and the interaction between age and capture location of individual i . The \widehat{TL}_i estimate was then used to predict the fecundity of the individual,

$$\widehat{f(a)}_i = \beta_0 + \beta_1 \widehat{TL}_i + \beta_2 \widehat{TL}_i^2 + \tau Location_j$$

where $\widehat{f(a)}_i$ is the expected number of eggs (absolute fecundity) given the TL of individual i and capture location j . The β parameters are the estimated coefficients and τ is the random effect associated with capture location.

I used logistic regression to determine the age-specific probability of a juvenile Blue Sucker reaching maturity ($p(a)$) for both locations. I based my function on the Red River population because I observed adult and juvenile Blue Suckers in equal proportions at that location. To create the $p(a)$ function, I regressed the binomial maturity status of individuals against age,

$$\widehat{p(a)}_i = \beta_0 + \beta_1 \text{age}_i,$$

Where β_0 is the model intercept, and β_1 is the estimated coefficient associated with the age of fish i . Similar to Rupprecht & Jahn (1980), I found the youngest mature male was age 3 and the youngest mature female was age 5. Because I was interested in the probability of a female being mature given her age, I excluded the three-year old male from the analysis. Because of the similarity of the two populations in adult age distribution, I used the age-at-maturity function for both populations (Fig. 4.3). In the Red River, I observed all Blue Suckers were mature by age 12, but younger age classes still included juveniles. In the Kiamichi River, I only captured adults and determined that Blue Suckers did not fully recruit to the gear until age 13.

I was unable to estimate the probability of age-1 recruitment (S_1) directly, so I used my other estimates to approximate the probability. I defined S_1 as the probability of an oocyte becoming an age-1 Blue Sucker, and obtained the parameter as,

$$S_1 = N_1^{J(1)} / N_1^E,$$

or the mean expected abundance of age-1 juveniles ($N_1^{J(1)}$) divided by the mean expected abundance of oocytes (N_1^E) in the population (Vaughn & Saila, 1976). The line associated with the catch-curve models provided an estimate of the mean expected abundance in each age class. I estimated, $N_1^{J(1)}$ by back-transforming the expected abundance when age = 1. To estimate N_1^E I used the catch-curve model to provide mean abundance estimates for all ages. I multiplied the estimated abundance of each age class by the respective probability of reaching maturity ($p(a)$) to separate the estimated adult abundance from the juveniles. Next, I multiplied the age-specific abundance of adults by the age-specific fecundity rates ($f(a)$), and I obtained N_1^E from the sum of the resulting product.

POPULATION MODEL

With the necessary parameters estimated, I created a female-based Leslie matrix model the growth of each population. I only modeled the female population because population growth is dependent upon egg production, and I assumed the abundance of males did not contribute to egg survival (Gotelli, 2008). I constructed a matrix with dimensions 25 X 25, corresponding to the average age of the oldest Blue Suckers observed in each population (*i.e.*, 23 and 28). The matrix took the form,

$$\begin{bmatrix} F_1 & F_2 & \cdot & \cdot & \cdot & F_{25} \\ S_1 & 0 & & & & 0 \\ 0 & S_2 & & & & 0 \\ \cdot & & \cdot & & & \cdot \\ \cdot & & & \cdot & & \cdot \\ 0 & 0 & \dots & 0 & S_2 & 0 \end{bmatrix}$$

where, the columns correspond to age, the survival probabilities were listed diagonally, age-specific fecundities were listed across the top row, and all other values were zero. The fecundity rates were based on $p(a)$ and $f(a)$, because maturity was a prerequisite for a Blue Sucker to contribute to the next generation. Therefore,

$$F_a = f(a) * M:F * p(a),$$

where, F_a is the realized fecundity for fish of age a , $f(a)$ was the age-specific mean fecundity (described previously), M:F was 0.5, to remove the males, and $p(a)$ is the age-specific probability of being mature (described previously).

I used Markov-chain Monte Carlo (MCMC) simulations to incorporate demographic stochasticity and the uncertainty around my parameter estimate into my population growth models. I incorporated demographic stochasticity into my models by treating the transitions to the next age class (S_2), and from immaturity to maturity ($p(a)$), of each individual as random, rather than deterministic events (Caswell, 2001). However, my model resulted in the presence of millions of oocytes in the population following each spawning season, and computing efficiency required that the probability of age-1 recruitment (S_1) be treated as deterministic rather than random. Because my fecundity functions provided mean estimates of TL and absolute fecundity given the age of an individual, I allowed the β -estimates within the function to vary randomly for each individual. For each β -estimate I created a random normal distribution with the β -

estimate as the mean, and the standard error (SE) of the estimate as the standard deviation parameter in the distribution. For each individual, each β -estimate in the function was selected at random and the absolute fecundity of that individual was determined from the resulting function. Similarly, I incorporated uncertainty in my annual survival probabilities (S_2) by randomly selecting the S_2 for each simulated year from a uniform distribution with the upper and lower bounds set by the 95% confidence intervals around my S_2 estimates.

I conducted 1000 iterations of my MCMC simulations to estimate discrete population growth (λ) over 50 years. In each iteration, I estimated λ as the geometric mean of N_t/N_{t-1} , where N is population abundance at time step t (Caswell, 2001). The λ estimate for each iteration was used to calculate the mean λ and 95% credibility intervals for λ . When $\lambda > 1$, the population was growing exponentially, $\lambda \approx 1$, the population was stable, and $\lambda < 1$ indicated exponential decay of the population (Caswell 2001).

I used my population models to explore the relationship between variable annual spawning frequencies and the recruitment variability observed in the two populations. The previously described model assumed that each female contributed oocytes to the population each year, but individual female Blue Suckers may not return to spawn every year (Lyons *et al.*, 2016, Chapter 2). My estimate of age-1 recruitment (S_1) represented the average recruitment probability in any year, and mitigated the variability in individual spawning behavior. However, intermittent spawning by females does increase uncertainty in my estimates of population growth. I simulated variable spawning probabilities and evaluated the effect of spawning variability on the recruitment variability observed in my data. In my MCMC simulations, I randomly selected the annual spawning probability for each year from a beta distribution with a user-defined mean. The contribution of oocytes by a female to the next generation was treated as a random event guided by the spawning probability for that year. Because my estimate of S_1 included the variability associated with intermittent spawning, the parameter was adjusted upward to maintain the growth rate (λ) estimated in the previous version of the model. I used the RCD value estimated from my catch-curve models as the metric for recruitment variability. At the end of each simulation, I estimated the RCD by fitting a catch-curve model to the last simulated year and extracting the R^2 value. I manipulated the mean spawning probability in my beta distribution and the S_1 parameter so that the 95% credibility intervals of the simulated RCD and λ contained the observed RCD value from my catch-curve models and the λ estimate from the previous model. I recognize

that recruitment variability may be influenced by non-constant age-1 recruitment; however, incorporating non-constant spawning provided a representation of the uncertainty in my projections.

RESULTS

VITAL STATISTICS

I included demographic data from 176 Blue Suckers from the Red River and 123 from the Kiamichi River in the population vital statistic assessments. In the Red River, TL ranged 315 – 640 mm and ages were estimated 3 – 28 years, and in the Kiamichi River, TL ranged 516 – 714 mm and ages were estimated 6 – 23 years (Fig. 4.2). In 2017, the Red River M:F = 0.26, and M:F = 2.94 in Kiamichi River. I did not estimate the M:F ratio from my 2016 sampling because many of the individuals captured appeared to be post spawn and I was unable to determine the sex based on gamete expression. Additionally, I collected and analyzed ovaries of 13 females from the Red River and 18 females from the Kiamichi River.

Blue Suckers in the Red River tailwater had a lower annual mortality rate, but greater variability in recruitment than those in the Kiamichi River. The modal age in the Red River population was 6 years; however, including 5-year old fish produced an overall better model suggesting that Blue Suckers recruited to sampling methods at age 5 ($R^2 = 0.46$ v. 0.52 , respectively). The instantaneous mortality rate ($z \pm 95\%$ confidence interval (CI)) of Blue Sucker in the Red River tailwater was 0.11 ± 0.05 and annual survival rate ($S \pm 95\%$ CI) was $89.5 \pm 1.05\%$ (Fig. 4.4). The modal age of the Kiamichi River population was 13 years, and including 12-year old fish did not improve the model ($R^2 = 0.79$ v. 0.77 , respectively). For that reason, I considered fish younger than 13 underrepresented in my sample, and determined that $z = 0.27 \pm 0.09$ and $S = 76 \pm 1.09\%$ in the Kiamichi River. In the Red River tailwater, the RCD was 0.52, compared to 0.79 in the Kiamichi River, indicating greater variability in recruitment in the Red River tailwater.

My model of a polynomial relationship between TL and fecundity had the most support (Table 1). On average, Blue Suckers in the Kiamichi River were more fecund than those in the Red River (mean difference = 7,440 eggs; Fig. 4.5). I estimated the range of absolute fecundity for the smallest Blue Suckers at 50,000 – 60,000 oocytes and the largest Blue Suckers exceeded 100,000 oocytes.

My top-ranked linear model of adult growth rates indicated that growth rates varied by sex and the interaction between age and location (Table 4.2). Adult Blue

Suckers in the Red River grew an average of 3.7 mm TL per year (0.7 SE, β_2), but in the Kiamichi River grew an additional 2.7 mm TL (1.0 SE, β_4) more per year (Fig. 4.6). Although growth rates differed between locations, Blue Suckers were generally similar in size at the two locations (20.06 mm TL, 13.38 SE, β_3). Additionally, female Blue Suckers averaged 37-mm TL (4.64 SE, β_1) larger than males. My top model explained a moderate amount of the variation ($R^2 = 0.37$), but growth was highly variable among individuals.

I determined that the probability of female Blue Suckers reaching maturity increased from the age of 5 to the age of 12 years. The model explained a moderate amount of the variation in my observations ($R^2 = 0.64$), and the probability of reaching maturity increased from near 0 at age 4 to near 1 at age 12. At age 9, 50% of Blue Sucker were expected to have reached maturity. Because the age structure of adults was similar at both locations, I considered the function representative of the Kiamichi River population as well (Fig. 4.3).

POPULATION MODEL

My population models indicated a stable population in the Red River tailwater and a declining Kiamichi River population, although the mean annual spawning probability was similar for the two populations. I estimated $\lambda = 1.006$ (1.004 – 1.008 95% credibility intervals) in the Red River tailwater population. When the mean annual spawning probability was 0.18 (Fig. 4.7), the observed RCD was replicated and the λ credibility intervals increased to 0.993 – 1.011. In the Kiamichi River, $\lambda = 0.939$ (0.937 – 0.942), and RCD was replicated when mean annual spawning probability was 0.17. The credibility intervals around λ increased to 0.936 – 0.958 when the variable spawning probabilities were incorporated in the Kiamichi River population model.

DISCUSSION

My survival and recruitment estimates for Blue Sucker were comparable to other catostomid populations; however, my fecundity estimates were relatively low compared to other Blue Sucker populations. In a study of seven catostomid species from four streams in Iowa, Quist and Spiegel (2012) reported annual survival rates and RCD values ranging from 38 – 82% (S) and 0.49 – 0.95 (RCD). My observed annual survival rate in Kiamichi River was similar to that of Blue Suckers observed in the Wabash River, IN (*i.e.*, $S \approx 76\%$; Bacula *et al.*, 2009), but the annual survival rate in the Red River was

high ($S = 89.5\%$) compared to other catostomid populations. In a separate study, I estimated that Blue Suckers in the Red River were less likely to undergo seasonal migration than those in the Kiamichi River, potentially contributing to the lower mortality rate (Chapter 2). Blue Suckers in my study were smaller and less fecund than those in the Wabash River, IN (Daugherty *et al.*, 2008; Bacula *et al.*, 2009). The largest Blue Suckers in my study had TL ≈ 50 mm less than the largest Blue Suckers captured in the in the Wabash River, IN (Daugherty *et al.*, 2008). In addition to being larger, Blue Suckers from the Wabash River, IN were more fecund. I estimated Blue Sucker with TL ≈ 675 mm to have 100,000 – 120,000 oocytes per individual, compared to a mean of 150,000 oocytes per individual in Blue Suckers of the same size in the Wabash River (Daugherty *et al.*, 2008). Further, Daugherty *et al.* (2008) estimated mean absolute fecundities $> 200,000$ oocytes per individual for fish with TL > 700 mm. The difference in fecundity was likely due to regional variation in size and growth within the species. For example, smaller body sizes of American Shad *Alosa sapidissima* were attributed to lower latitudes along the North American Atlantic coast (Leggett & Carscadden, 1978), and a similar trend in body sizes along the American Pacific Coast manifested in lower fecundities at lower latitudes in Pacific salmonids *Oncorhynchus spp.* (Beacham, 1982).

My simulated spawning probability best matched 2017, but intraspecies variability in iteroparus fishes can be high. In Chapter 2, I estimated the probability of Blue Suckers returning to the tributaries to spawn in spring 2016 was 0.83, and 0.19 in 2017. My theoretical spawning distributions used to replicate the observed RCD values (Fig. 6) indicate that my estimates during 2017 were common, and my estimate for the exceptionally wet 2016 was a rare occurrence. Many iteroparous migratory fishes undergo nonconsecutive spawning migrations. In the Wisconsin River, female Blue Suckers typically returned to spawn every other year, whereas many of the males returned each year (Lyons *et al.*, 2016). Female American Shad along the east coast were less likely to return to spawn than males in any population and the probability of repeat spawning declined at lower latitudes, ranging from complete post-spawn mortality in Florida to 65% probability of returning in New Brunswick (Leggett & Carscadden, 1978). The similarity in simulated spawning behavior in my study populations indicates the differences in population growth may be due to post spawn variation in recruitment rather than spawning behavior.

Examining the relationships between water releases and recruitment of Blue Sucker would be beneficial to understanding the effects of dam operations on this

species. Although I was not able to empirically evaluate age-1 recruitment, it is possible that dam release practices in the system contributed to recruitment variability. On several occasions, I observed Blue Suckers actively spawning over gravel bars and riprap during dam releases only to have their spawning habitat dewatered an hour later when water releases ceased. Dam release practices have been implicated in stranding the eggs of the endangered Robust Redhorse *Moxostoma robustum* (Grabowski and Isely, 2007), stunting the growth of juvenile Robust Redhorse and V-lip Redhorse *M. collapsum* (Weyers *et al.*, 2003), and reducing the availability of suitable age-0 habitat for Clanwilliam Yellowfish *Barbus capensis* (King *et al.*, 1998). In addition to reducing available habitat, dam releases caused young-of-year mortality by rapidly changing the temperature, dissolved oxygen, and hydraulic conditions of the habitat in the Olifants River, South Africa (King *et al.*, 1998).

I treated Blue Sucker occupying the two river systems as separate populations, but conservation efforts would benefit from consideration of metapopulation dynamics and should be explored further. Treating subpopulations as individual populations can result in reduced stock biomass and a high probability of exploitation, whereas, ignoring the spatial structure of the population could result in the loss of subpopulations (Ying *et al.*, 2011). Immigration into the Red River tailwater population may explain the high survival probability and low RCD estimated for the population. Catch curve models and the RCD assume a closed population. Adults immigrating into the population would violate the closure assumption, and adding adults to the population after recruitment would result in a slower declining slope and a higher survival probability. Additionally, the recruitment variability of the immigrants and immigration rates would be reflected in a lower RCD estimate. Therefore, my results should be interpreted with caution until population closure or metapopulation dynamics are evaluated.

Although my study was based on a single year of data, it indicated that population stability may be problematic and serves as a starting point to improve monitoring and management efforts. At a minimum, the Blue Sucker population should be monitored over several years to capture interannual population trends, a variety of hydrologic and temperature cycles, and improved representation across the basin. A monitoring program would reduce uncertainty in my vital statistic estimates, provide a more reliable population trajectory, and provide a way to evaluate the effects of dam management on Blue Sucker.

Table 4.1 Result from my analysis of Blue Sucker fecundity. The name of the model represents the hypothesized relationship with absolute fecundity (number of eggs per individual). I provide the scores used to rank the models (AIC_c), the difference in score of each model compared to the top model (ΔAIC_c), and the conditional R² value associated with each model.

Name	Model	K	AIC _c	ΔAIC _c	R ²
TL ²	$\widehat{Fecundity}_i = \beta_0 + \beta_1 \widehat{TL}_i + \beta_2 \widehat{TL}_i^2 + \tau * Location_j$	3	655.9	0	0.44
Age ²	$\widehat{Fecundity}_i = \beta_0 + \beta_1 \widehat{age}_i + \beta_2 \widehat{age}_i^2 + \tau * Location_j$	3	671.5	15.6	0.48
TL	$\widehat{Fecundity}_i = \beta_0 + \beta_1 \widehat{TL}_i + \tau * Location_j$	2	672.2	16.3	0.43
Age	$\widehat{Fecundity}_i = \beta_0 + \beta_1 \widehat{age}_i + \tau * Location_j$	2	707.2	51.3	0.40

Table 4.2 Result from growth of adult Blue Suckers. The name of the model indicates the hypothesized relationship with somatic growth. I provide the scores used to rank the models (AIC_c), the difference in score of each model compared to the top model (ΔAIC_c), and the R² value associated with each model.

Name	Model	K	AIC _c	ΔAIC _c	R ²
Sex + Age * Location	$\widehat{TL}_i = \beta_0 + \beta_1 gender_i + \beta_2 Age_i + \beta_3 Location_i + \beta_4 Age_i * Location_i$	5	1910.5	0	0.37
Location + Age * Sex	$\widehat{TL}_i = \beta_0 + \beta_1 Location_i + \beta_2 Age_i + \beta_3 gender_i + \beta_4 Age_i * gender_i$	5	1914.9	4.4	0.35
Age * Sex	$\widehat{TL}_i = \beta_0 + \beta_1 gender_i + \beta_2 Age_i + \beta_3 Age_i * gender_i$	4	1921.2	10.7	0.33
Age * Location	$\widehat{TL}_i = \beta_0 + \beta_1 Age_i + \beta_2 Location_i + \beta_3 Age_i * Location_i$	4	1965.4	55.1	0.16

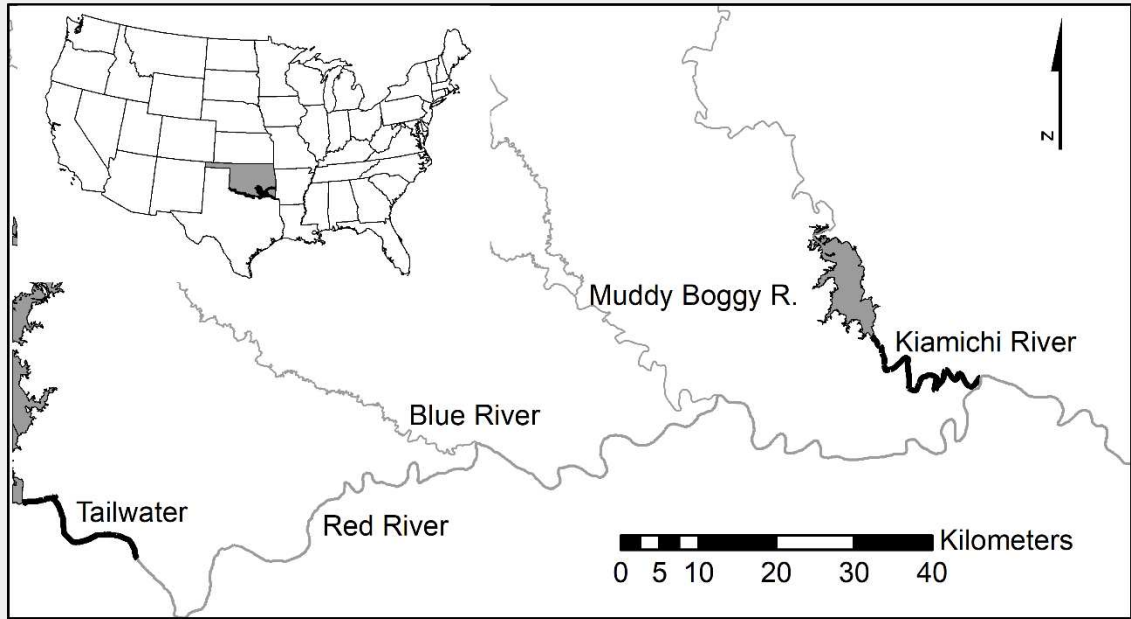


Fig. 4.1 The lower Red River, Oklahoma and major tributaries. Fish were collected in the Tailwater and Kiamichi River (shown in bold).

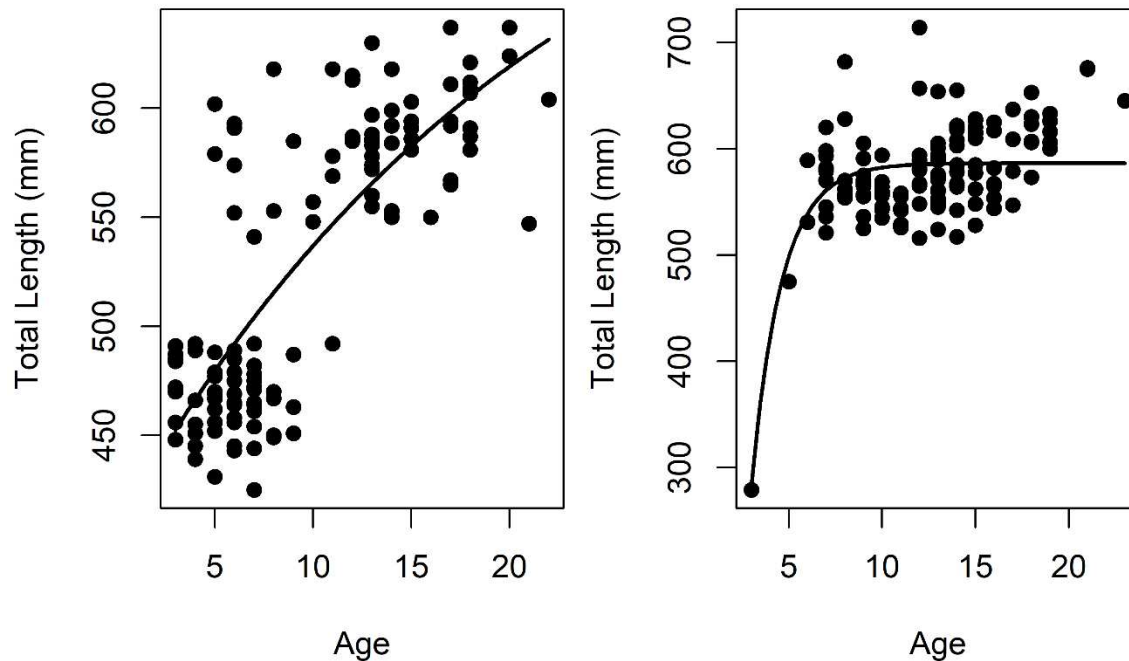


Fig. 4.2 Von Bertalanffy growth curves for Blue Suckers in the Red (left) and Kiamichi (right) rivers. The von Bertalanffy estimates for the Red River population were: $L^{\infty} = 762$ mm, $K = 0.05$, $t_0 = -16$ mm. Parameter estimates in Kiamichi River were: $L^{\infty} = 586$ mm, $K = 0.62$, $t_0 = 2$ mm. Because juveniles were missing from the Kiamichi River population, I included two juveniles from Muddy Boggy River to fit the curve (juvenile total lengths = 279 & 475 mm, aged 3 & 5 years, respectively).

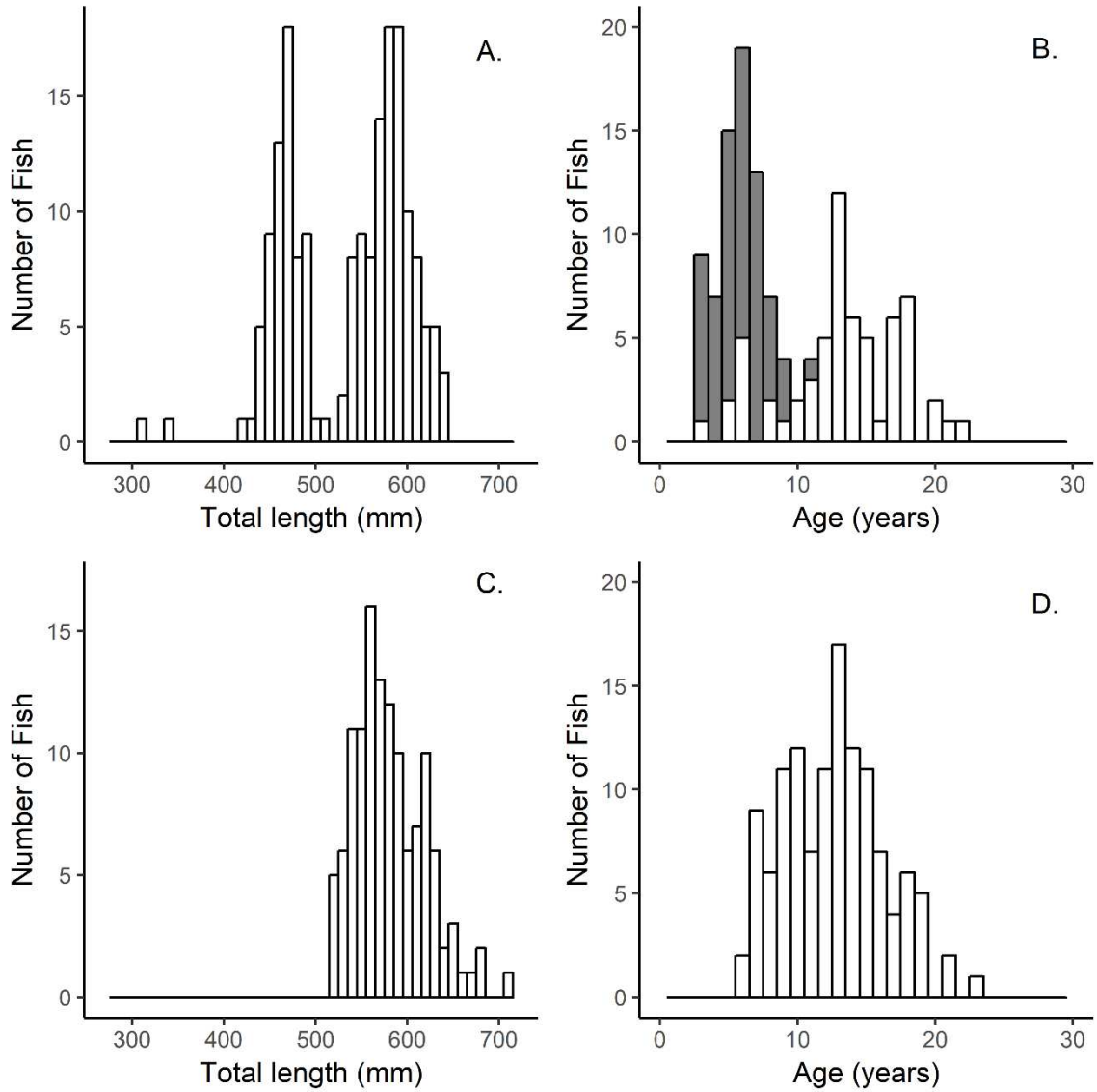


Fig. 4.3 Length-frequency and age-frequency histograms of Blue Suckers caught in Red (A. and B.) and Kiamichi (C. and D.) rivers in 2017. The age-frequency plot for the Red River (B.) is stratified by mature (white) and juvenile (gray) fish. I did not capture juveniles in the Kiamichi River.

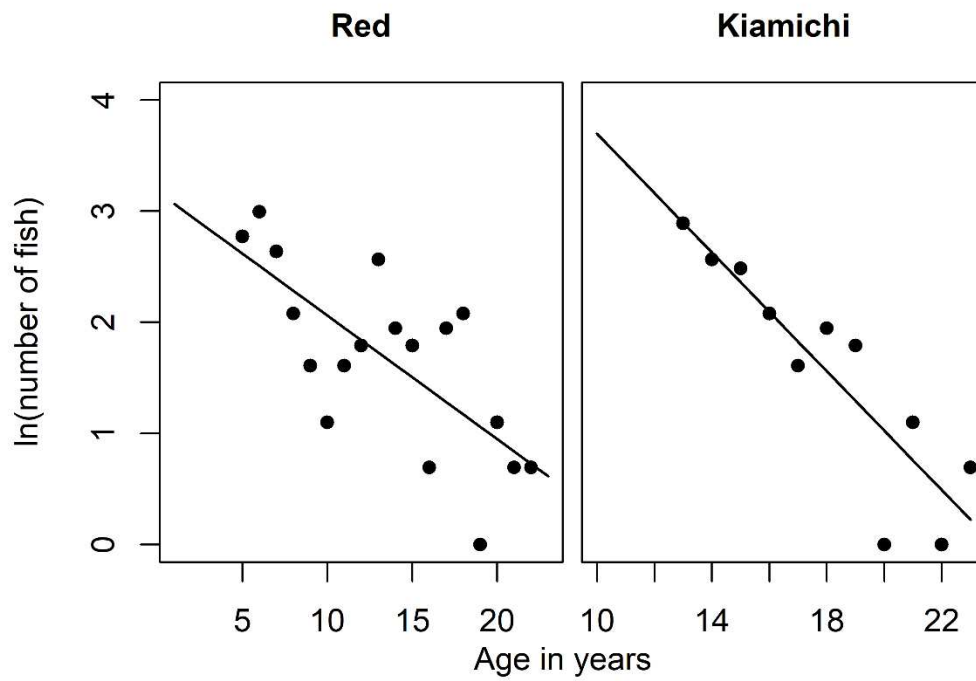


Fig. 4.4 Catch-curve models for Blue Suckers captured in the Red River tailwater (left) and the Kiamichi River (right).

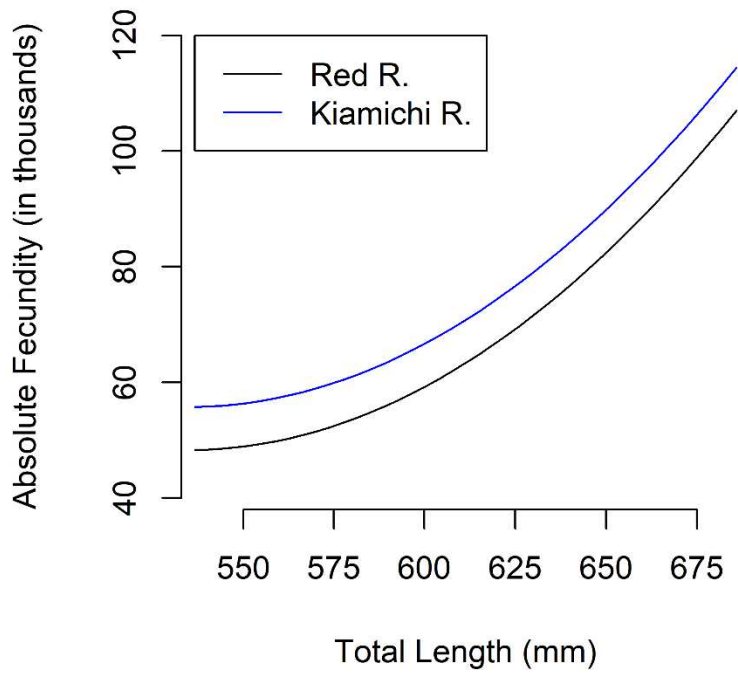


Fig. 4.5 Relationship between absolute fecundity (total number of eggs per individual) and Blue Sucker size (total length) in the Red River tailwater and Kiamichi River.

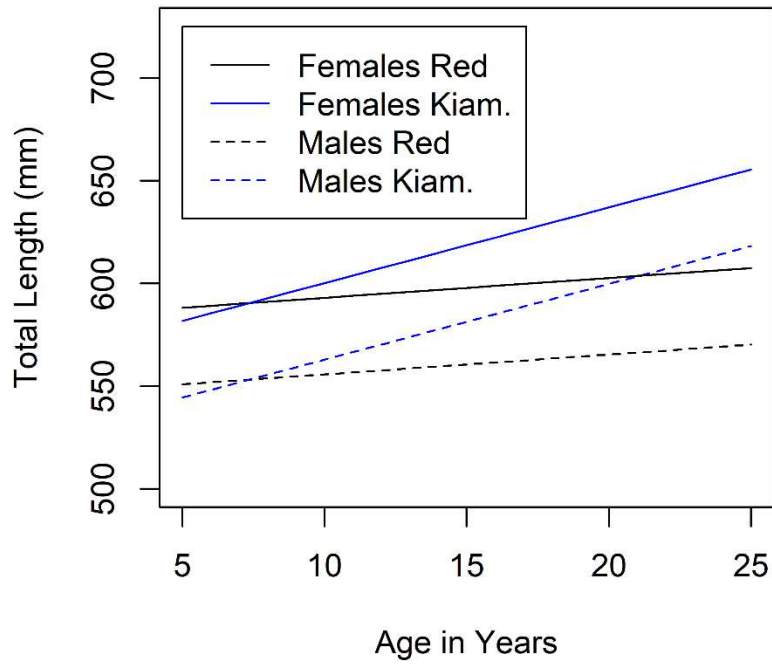


Fig. 4.6 Comparison of male and female adult Blue Sucker growth rates from Red and Kiamichi rivers.

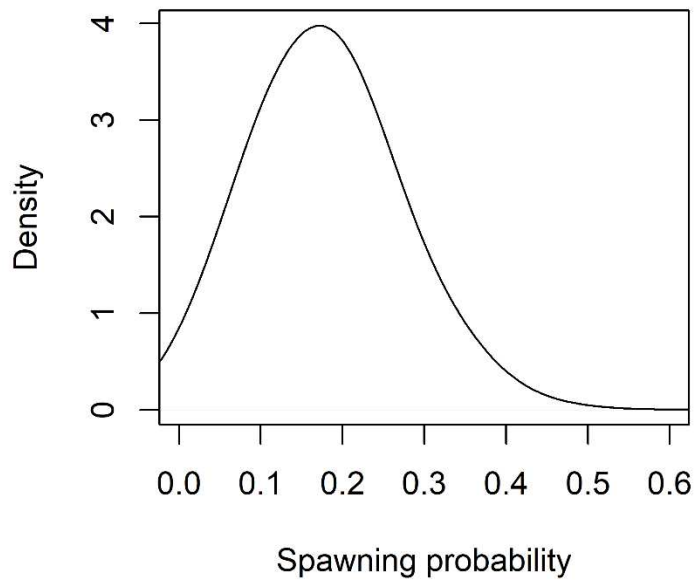


Fig. 4.7 Density plot representing the theoretical annual spawning probability used in my simulation of variable Blue Sucker spawning activity. I determined the probability of Blue Suckers spawning by randomly drawing a value from the distribution at the beginning of each simulated year.

CHAPTER V

ENVIRONMENTAL INFLUENCE ON POPULATION DYNAMICS

INTRODUCTION

Water availability provided via dams is a necessary resource for the developing world, but science recognizes negative effects of dams on riverine ecosystem services. Historically, rivers supported human civilization by providing clean water, fertile soils, and food, but modern societies use dams to retain water, control floods, and provide electricity (Vörösmarty *et al.*, 2010; Poff & Schmidt, 2015). Dams fragment riverine ecosystems (Fausch *et al.*, 2002), and alter the natural flow (Poff *et al.*, 1997), thermal (Olden & Naimen, 2010), and sediment regimes (Wohl *et al.*, 2015). Disruption of riverine processes impairs pollution abatement, limits soil fertility soil on the floodplain, and limits food supplies in some regions (Cushman, 1985; Jager & Smith, 2008). In particular, migratory fishes were once a reliable food source for humans, but the prevalence of dams in the world's developed nations has resulted in widespread decline of migratory fishes (Lucas & Baras, 2001).

Recruitment, the number of fish that survive from hatch to a size or life stage of management importance, is a key factor that determines population structure and is influenced by both temperature and flow conditions. Recruitment influences the size structure and abundance of populations. In particular, recruitment is considered one of the most important rate functions in fish populations (Gulland, 1982; Allen & Hightower, 2010). Temperature affects spawning condition of adults as well as survival and development of eggs and young-of-year (Lucas & Baras, 2001; Moyle & Cech, 2004). Hydrological conditions can influence recruitment success (Bonvechio & Allen, 2005; Dutterer *et al.*, 2013; Rolls *et al.*, 2013); though, these effects are not well understood for

many species, especially migratory populations (Cooke *et al.*, 2012). Few studies have described hydrology-recruitment relationships for non-game species in particular (Cooke *et al.*, 2005) despite the important ecological role they play in many river systems.

River discharge and temperature are among several factors that affect fish growth, another key management target. Select flow conditions can increase benthic invertebrate abundances, thereby increasing growth rates in fishes via bottom-up trophic effects (Weisberg & Burton 1993; Rutherford *et al.*, 1995). Additionally, temperature affects foraging behavior and metabolic efficiency of fishes, and extreme temperature can be a limiting factor on growth rates. Like recruitment, growth rates are important to fish populations because size dictates survival, maturity and fecundity of individuals in the population (Allen & Hightower, 2010).

Migratory fishes are ecologically important to aquatic communities, but dams often limit successful reproduction, recruitment, and even growth. Migratory fishes facilitate upstream nutrient flow and other processes that are critical to persistence of local, endemic communities (Flecker *et al.*, 2010). These endemic fishes comprise \approx 80% of North American freshwater fishes of conservation concern (Jelks *et al.*, 2008), and species loss may be exacerbated via limited access by migratory fishes to these habitats (Pringle, 2001). Reproductive failure by migratory individuals may occur due to disruption of environmental cues that are hypothesized to trigger both aggregation and spawning (Lucas & Baras, 2001). When cues are appropriate, flow suppression by dams can physically prevent access to spawning habitat or create unsuitable habitat (Junk *et al.*, 1989; Poff *et al.*, 1997). Instability of habitat below dams can negatively affect recruitment of young-of-year fishes (King *et al.*, 1998). Additionally, suppressed flow and unstable habitat can restrict growth by limiting access to foraging habitat (Junk *et al.*, 1989) and reducing secondary productivity (Cushman, 1985; Jowett, 2003). Failed reproduction and recruitment are commonly referenced as a primary stressor on migratory fishes and restricted growth may exacerbate the problem; however, fishery managers lack the information needed to adequately mitigate the perceived consequences of dams on migratory populations (Lucas & Baras, 2001).

Blue Sucker *Cyprinella elongatus* is an emblematic, migratory, large-river catostomid with a declining spatial distribution. Blue Suckers are capable of annual spawning migrations in excess of 300 km (Neely *et al.*, 2009). Given the migratory life history and broad distribution, Blue Suckers likely provide nutrient and process subsidies to many aquatic communities (Flecker *et al.*, 2010). However, their “non-game” status

coupled with their inconspicuous life history has resulted in a paucity of research on the species' life history (Burr & Mayden, 1999; Cooke *et al.*, 2005). Blue Sucker are a species of conservation concern in 21 of the 22 states that they occupy and are locally extinct in Pennsylvania (NatureServe, 2018). With the prevalence of dams and regulated flow throughout Blue Sucker distribution (Pracheil *et al.*, 2013), conservation of Blue Sucker depends on understanding the effects of flow and temperature on recruitment and growth of populations (Cooke *et al.*, 2012).

I investigated the influence of environmental conditions on the recruitment and somatic growth of Blue Suckers sampled from two flow-regulated rivers of the Southern Great Plains. My goal was to provide managers with information useful to improving flow conditions below dams to benefit Blue Sucker recruitment and growth. Because young-of-year Blue Sucker are inherently difficult to sample and recruitment relationships require extensive temporal sampling, I used information obtained from adult Blue Sucker to examine the effects of stream flow and temperature on recruitment and somatic growth. My general approach was to rank several hypotheses regarding the temporal and spatial influence of discharge and temperature conditions on the age structure of two populations. I back-calculated annual growth by Blue Sucker to assess relationships with flow and temperature conditions.

METHODS

STUDY AREA

I sampled Blue Suckers from reaches of the lower Red River catchment downstream of Denison (10-km downstream) and Hugo dams (2-km downstream), Oklahoma. The Red River originates in the Texas Panhandle and flows east forming the Oklahoma-Texas border (Fig. 5.1). The Red River is impounded for hydropower generation and flood control via Dennison Dam, creating a delineation between what agencies consider the lower and upper Red River. Although the upper Red River is ecologically significant for several threatened pelagophilic fishes (Worthington *et al.*, 2018), the lower Red River is home to many large river fishes. There are three relatively large tributaries within the lower Red River of Oklahoma, but the Kiamichi River is of interest because Blue Suckers congregate below the dam during the spawning period. The Kiamichi River is impounded by Hugo Dam 28-km upstream of the confluence with the Red River. Hugo Lake is managed for recreational fishing, flood control, and water storage. Both rivers have

reaches where water releases could be managed to improve down river conditions for fishes.

FIELD SAMPLING AND PROCESSING

In March – May 2017, I sampled Blue Suckers using electrofishing to collect pectoral rays across a range of fish sizes. Blue Suckers were captured using a 5.0 GPP electrofishing unit (Smith-Root, Inc. Vancouver, WA) mounted to a 14' Jon boat (Alumacraft Boat Co. Arkadelphia, AR). Water conductivity ranged 50 – 2000 μS (Kiamichi and Red rivers, respectively) and I adjusted power output according to Miranda (2009). I typically used 60 Hz frequency direct current, although low conductivities in the Kiamichi River occasionally required 120 Hz to achieve the target output (2500 – 4000 watts). I used a chase boat whenever possible because Blue Suckers often surfaced behind the electrofishing boat. Captured Blue Suckers were measured (1 mm total length, TL), and a 2-cm section of the leading left pectoral ray was removed using diagonal cutters to clip the ray as near the articulation point as possible (Bednarski & Scarnecchia, 2006). I collected pectoral rays because they were considered the most reliable, non-lethal age structure in catostomids (Beamish & McFarlane, 1969; Labay *et al.*, 2011; Acre *et al.*, 2017). I stored pectoral rays in individual #1 manila coin envelopes, and allowed the rays to air-dry until I transported them to the laboratory.

I processed pectoral rays in the laboratory for age and growth analysis. I further dried pectoral rays in a dehydrator for at least one week. Next, I cleaned excess flesh from each specimen, clipped the proximal end to < 1 cm, and set the clipped segment in epoxy (DeVries & Frie, 1996). Once the epoxy cured, I cut \approx 1-mm sections with a low-speed saw (Bueller Isomet 1000, Lake Bluff, IL) and mounted those sections to a microscope slide using a nitrocellulose-acetate solution (clear fingernail polish; Bednarski & Scarnecchia, 2006). I viewed the sectioned pectoral rays under 40x magnification with transmitted light. I defined annuli as concentric, nested rings of opaque bone tissue, and two readers independently enumerated annuli in each pectoral ray (Casselman, 1983; Panfili *et al.*, 2002; Labay *et al.*, 2011). When readers disagreed, I simultaneously reviewed the specimen in question and readers reached a consensus age. I photographed the clearest section from each pectoral ray using a camera mounted to the microscope (Leica DFC295, Leica Camera, Inc. Allendale, NJ). For each fish, I used ImageJ software with the OpenJ plugin (version 1, Schneider *et al.*, 2012)

to measure the distance between the inner-most annuli and the outer edge of each annuli along the apex of the ray in the photograph (Ricker, 1975; Panfili *et al.*, 2002; Quist & Spiegel, 2012). The distances between consecutive annuli were proportional to somatic growth and served as a record of annual growth of individuals (DeVries & Frie, 1996).

DATA PROCESSING

Recruitment – I created catch-curve models and used the model residuals to determine year-class strength. Using R statistical software (version 3.4.4, R Core Development Team, 2018), I created catch-curve models for each population by regressing fish abundance in each year class against age (*i.e.*, years). The slope of the regression line described the mean instantaneous mortality rate of the population, and the line represented the mean expected abundance in each year class given the mean mortality rate (Ricker, 1975; Miranda & Betolli, 2007). The residuals were the difference between the observed abundance and the expected abundance for each year class, and served as a measure of recruitment variability (Guy & Willis, 1995; Isermann *et al.*, 2002). Positive residuals indicated the observed abundance was greater than the expected reflecting a strong year class, whereas negative residuals indicated a weak year class (Maceina, 1997). I created catch-curve models for each population, calculated the residuals, and rescaled them on the student's t scale allowing comparison of the residuals from each catch-curve model (Maceina, 1997). Because I collected Blue Suckers in spring 2017, the outer-most annuli represented recent growth leading up to winter 2016/2017 (Peterson *et al.*, 1999). Therefore, the year class an individual belonged to was determined as 2017 minus the age of the individual.

Somatic growth – To estimate individual annual growth, I measured the distance between the outer edges of consecutive annuli excluding age 0-1. The distance between two consecutive annuli in a pectoral ray was proportional to the somatic growth of that individual during a given year (Ricker, 1975; DeVries & Frie, 1996). Pectoral rays did not contain a well-defined centrum due to a blood vessel in the center of the ray (Panfili *et al.*, 2002; Bacula *et al.*, 2009). Therefore, I was not able to measure growth between hatching and age 1, and instead began my measurements at the age 1 – 2 increment. Because the apices of annuli rarely followed a straight line, I took my measurements between the outer edge of the age-1 annulus apex and the outer edge of each

consecutive annulus apex (Fig. 5.2). I calculated annual growth as the difference in length between the consecutive annuli. Measuring annuli in this manner mitigated the effect of nonlinear apices in individual pectoral rays.

Environmental conditions – I obtained discharge and temperature data from remote monitoring stations in my study area. Discharge data were available from the U.S. Geological Survey (USGS) gages in the tailwater below Denison Dam (07331600), in the Red River, between the tailwater and Kiamichi River confluence near Arthur City, TX, and in the Muddy Boggy River (gages 07331600, 07335500, and 07335300, respectively; <https://waterdata.usgs.gov/ok/nwis/rt>, accessed: 1/8/2018). I obtained dam release data from Hugo Dam (USACE; <http://www.swt-wc.usace.army.mil/HUGO.lakepage.html>, accessed: 1/8/2018) given this represents the bulk of flow in the Kiamichi River below Hugo reservoir (*i.e.*, there is no stream gage). Temperature data were unavailable so I used air temperature recorded by a weather station in Hugo, OK as surrogate (http://www.mesonet.org/index.php/weather/daily_data_retrieval, accessed: 6/19/2018). Although it was possible to estimate water temperature from the recorded air temperature, models of the relationship are imperfect and introduce error into my analysis (*e.g.*, Morrill *et al.*, 2005). I was primarily interested in accounting for winter or growing seasons that were cooler or warmer relative to the other winters and growing seasons included in my analysis, and air temperature provided an adequate measure (*e.g.*, Quist & Spiegel, 2012).

I defined seasons that corresponded to life-history events of Blue Sucker that could be used to develop my hypotheses. I defined three seasons that related to Blue Sucker movements and growth: 1.) spawning, February 1 – May 31; 2.) growing, June 1 – September 30; and 3.) winter, October 1 – January 31. These seasons corresponded to observed movement patterns and temperature changes (Peterson *et al.*, 1999; Neely *et al.*, 2009; Chapter 2). I used R statistical software to calculate stream flow metrics related to timing, frequency, duration, and variability of seasonal discharge at each location.

I calculated flow patterns and temperature metrics that I hypothesized to influence the growth, foraging, spawning success, and recruitment of Blue Suckers. I chose metrics described by Olden and Poff (2003) to describe the magnitude, timing, variability, duration, rate of change, or frequency of seasonal flow pulses (Table 5.1).

The magnitude and timing of high-flow pulses during the spawning season provide necessary cues for Blue Sucker spawning migration (Moss *et al.*, 1983; Vokoun *et al.*, 2003; Chapter 3), and are potentially related to the spawning success of potamodromous fishes (Lucas & Baras, 2001; Thurow, 2016). High or low flow magnitude, variability, and duration of high or low flows, during the growing and winter seasons, affect growth and survival of riverine fishes by altering habitat availability (King *et al.*, 1998; Young *et al.*, 2011) and secondary productivity (Bednarek & Hart, 2005, Rolls *et al.*, 2013). The rate of change and variability of flow pulses can result in the stranding of nests (Grabowski & Isley, 2007) and stunted growth of young-of-year (Weyers *et al.*, 2003). Additionally, I calculated the maximum temperature over the growing season (*i.e.*, see seasons above), minimum temperature during the winter season, and the average temperature from both the growing and winter seasons. Extreme temperatures can reduce foraging efficiency and growth of fishes (Moyle & Cech, 2004).

I used hydrology data from multiple locations, because Blue Suckers were mobile. I previously documented Blue Suckers moving throughout the lower Red River to fulfill their life-history requirements (Chapter 2). In this study, I was attempting to match the conditions Blue Suckers experienced during the growing, winter, and spawning seasons to year-class strength or individual growth. I could not assume that Blue Suckers occupied the reach where captured throughout their lives. Blue Suckers display some degree of fidelity to spawning sites (Lyons *et al.*, 2016; Chapter 2), and foraging habitat (Adams *et al.*, 2017), however, variability exists within the population and it is unclear whether homing to spawning habitats is natal. Further, the behavior of age-0 Blue Sucker is unknown (Adams *et al.*, 2006), and Blue Suckers may remain in their natal tributary or drift to the mainstem after hatching. I addressed the possibility of the spatial influence on year-class strength using hypothesis testing.

DATA ANALYSIS

Hypotheses – I created eight candidate models that each represented different hypotheses regarding the relationship between year-class strength and environmental conditions. First, I created a model to serve as 1.) *null hypothesis* that included an intercept and the sample locations as a fixed effect. Next, I included three hypotheses to reflect the temporal dynamics of Blue Sucker life history in the mainstem Red River (Table 5.2): 2.) *adult hypothesis*, reflected growing and winter season conditions in the

mainstem, leading up to the spawning season, 3.) *main spawning hypothesis*, reflected conditions in the mainstem river during the spawning season, and 4.) *main recruitment hypothesis*, that represented growing and winter season conditions in the mainstem following the spawning season. I included four additional hypotheses to reflect the spatial dynamics of the Blue Sucker population. Two hypotheses were based on strong spawning-site fidelity: 5.) *local spawning hypothesis*, described conditions during the spawning season, at the sampling locations, and 6.) *local recruitment hypothesis*, reflected conditions following the spawning season, at the sampling locations. The last two hypotheses were based on the observation that straying Blue Suckers frequently selected Muddy Boggy River over other tributaries (Chapter 2): 7.) *Muddy Boggy spawning hypothesis*, described flow conditions during the spawning season in Muddy Boggy River, and 8.) *Muddy Boggy recruitment hypothesis*, reflected conditions post-spawning season in Muddy Boggy River.

Recruitment – I evaluated the statistical support for each of my eight hypotheses by using a model-selection framework and hypothesis-testing framework. I included several variables to represent each hypothesis based on ecological theory, but statistical theory favors parsimony and several of my variables were likely redundant (Akaike, 1973). Therefore, I used a forward-selection process to reduce my variable set representing each hypothesis to create the best-supported candidate model of my data (Akaike, 1973; Hobbs & Hillborn, 2006). I then used AIC_c to rank my selected candidate models and determine the relative support of my eight hypotheses (Johnson & Omland, 2004).

First, I created a generalized linear model (candidate model) to represent each of the aforementioned eight hypotheses. I used a forward selection approach guided by AIC_c to create the best candidate model of my data to represent each hypothesis (Akaike, 1973; Hobbs & Hillborn, 2006; Bolker *et al.*, 2009). I calculated the AIC_c scores with the 'AICcmodavg' package (Mazzerolle, 2016). The null model included an intercept and a fixed effect for location. To create additional candidate models, I individually added the habitat variables associated with each hypothesis to the null model (Table 1). I retained the added variable that produced the lowest AIC_c score and repeated the process until the addition of variables no longer reduced the AIC_c score by ≥ 2 points (Bolker *et al.*, 2009). I included an interactive effect of the retained variables with location in subsequent iterations of the model selection process. Because I only had a single year of data to construct catch curve models, my data were limited to the age-at-

recruitment and the longevity of individuals in the population at each location (15 and 7 years for Red and Kiamichi rivers, respectively). Fitting a generalized linear model to a small data set risks overfitting, however using AIC_c as a selection criterion prevents overfitting by penalizing the addition of excessive parameters (Akaike, 1973; Johnson & Omland, 2004). I reviewed residual plots to ensure that each candidate model satisfied the assumptions of generalized linear models (*i.e.*, normality, heteroscedasticity, independence) and calculated R² to determine how well each model explained the variation in my data (Zar, 2001).

Next, I determined the best-supported recruitment hypotheses by ranking the candidate models using AIC_c. I considered the model with the lowest AIC_c score to represent the hypothesis best supported by my data (Johnson & Omland, 2004; Hobbs & Hillborn, 2006). I considered models with a difference < 2 AIC_c points of the top model equally supported by my data. Additionally, I calculated the Akaike weights of each candidate model as a measure of relative support for the hypotheses (Hobbs & Hillborn, 2006).

Somatic growth – I used a mixed effects model to describe the annual somatic growth of individual Blue Suckers. I used the lme4 package in R (Bates *et al.*, 2018) to model incremental length data from Blue Sucker pectoral rays following the methods described by Weisberg *et al.*, (2010). My null model was:

$$Y_{cka} = t_a + h_{c+a-1} + f_{ck} + l_k + \varepsilon_{cka}$$

Where: Y_{cka} was the incremental measurement at age a , from fish k of year class c ,

t_a was a fixed effect corresponding to the age of a fish,

h_{c+a-1} was a random effect corresponding to the year a fish was age a with $\sim N(0, \sigma_h^2)$,

f_{ck} was a random effect associated with an individual fish k from year class c , with $\sim N(0, \sigma_f^2)$,

l_k was a random effect associated with the capture location of fish k , with $\sim N(0, \sigma_l^2)$, and

ε_{cka} represented the unexplained variation, or residuals in my model and was $\sim N(0, \sigma^2)$.

I used a forward stepwise approach to select seasonal flow or temperature variables (Table 5.1), and retained variables that best explained ε_{cka} . I used AIC_c to evaluate the

effect of additional variables and favored parsimony when additional variables failed to lower the AIC_c scores ≥ 2 points of the simpler model (Burnham & Anderson, 2002; Johnson & Omland, 2004). I visually inspected residual plots to ensure that my final model satisfied the assumptions of generalized linear models (*i.e.*, normality, heteroscedasticity, independence). I calculated the conditional R² to determine the portion of variability explained by the fixed and random effects in the final model (Nakagawa & Schielzeth, 2013).

RESULTS

FISH SAMPLING

I captured and aged 126 Blue Suckers downstream of Dennison Dam and 123 from the Kiamichi River, but truncated year-class data between the age-at-recruitment and the 20-year class. I determined that Blue Suckers fully recruited to my sampling gear at age 5 and age 13 in the Red and Kiamichi rivers, respectively. Blue Sucker with ages > 20 were poorly represented in my data (2 & 3 individuals in Red & Kiamichi rivers, respectively), and were excluded from analysis. I used 110 fish from Red River and 65 fish from Kiamichi River in my analysis of year class strength.

Similarly, I truncated my growth data to include only length increments formed within the past 20 years in my growth analysis. Although Blue Sucker reached 23 years of age, I excluded increments with age > 20 from analysis due to incomplete environmental data from those years. I was not limited by the age at recruitment for this objective, and included 109 and 110 individuals from the Red and Kiamichi rivers, respectively, resulting in 2,180 growth increments for my analysis.

ANALYSIS

Environmental data – I observed considerable variability in discharge and temperature during my study period. Mean magnitude of discharge during the growing seasons ranged 33 – 975 m³/s in the mainstem Red River, with the highest flows occurring in 2007 and lowest flows occurring 2012. Flows were most variable in 2011 with 1.74 coefficient of variation, and most stable in 2005 with 0.28 coefficient of variation. Mean magnitude of discharge ranged 21 - 429 m³/s during the winter. Mean average temperature during the growing season ranged 23.8 – 28.0 °C and 8.3 – 12.8 °C during the winter.

Recruitment – I successfully created candidate models for 5 of 8 hypotheses, but I did not retain any additional variables for 3 of the hypotheses. Each of my candidate models (excluding the *null* hypothesis) retained an interactive effect with location (Table 5.2). A negative relationship with mean change in flow direction was retained in the *Adult condition* and *Red spawning* hypotheses. Additionally, a positive relationship with mean magnitude was retained in the *Red spawning* hypothesis. The rate of decline in discharge was negatively related to year-class strength in the *local spawning* hypothesis. The three hypotheses indicated that Blue Sucker recruitment was strongest when seasons began with high flow and discharge declined over time. Conversely, recruitment was poor when high flows were sustained during the season or when flows increased more frequently than they declined during the season. A positive relationship with duration of low flows (< 25th percentile) was retained in my candidate model of the *Muddy Boggy recruitment* hypothesis. My data did not support the inclusion of environmental variables in the *Red recruitment*, *local recruitment*, or *Muddy Boggy spawning* hypotheses.

The *adult condition* and *Muddy Boggy recruitment* hypotheses accounted for 76% of the total Akaike weight and were equally supported by my data. The *adult condition* hypothesis included an interactive effect between location and mean change in flow during the growing season (Table 5.2). The *adult condition* model indicated that recruitment was higher when flows began high and decreased throughout the growing season (slope = -1.03 ± 0.36 SE, Fig. 5.3). I did not detect a relationship between year-class strength in the Kiamichi River and mean changes in flow in the *adult condition* model (slope = 0.21 ± 0.26 SE). The model associated with the *Muddy Boggy recruitment* hypothesis indicated winters in Muddy Boggy River with more days where stream flow was < 25th percentile resulted in higher Blue Sucker recruitment in the Red River (slope = 0.83 ± 0.38 SE). However, I did not detect a relationship between recruitment in the Kiamichi River and winter flows in Muddy Boggy River (slope = -0.10 ± 0.32 SE, Fig. 5.3).

Somatic growth – The null growth model was best supported by my growth data, and explained a moderate amount of variation in growth (conditional $R^2 = 0.31$). My results indicate that individual growth varied randomly among individuals and years, and the fixed effect of age best explained annual growth. As expected, Blue Sucker annual growth decreased as Blue Suckers aged (slope = -0.04 ± 0.004 SE).

DISCUSSION

My *adult condition* hypothesis was the most supported and indicated that Red River summer flow conditions were important for gonadal growth; however, the Weisberg model did not suggest summer conditions (*i.e.*, or anything else) related to somatic growth. The results of my *adult condition* model indicated that a wet spring followed by dry summer resulted in higher recruitment the following year. These flow patterns can flush allochthonous nutrients into the river early in the growing season and provide relatively stable conditions for macroinvertebrate production (Dodds, 2002; Jowett, 2003). I did not detect a similar relationship between stream flow and somatic growth, but mature Blue Suckers in my study area showed an average annual growth of only 2 – 4 mm (Chapter 4) suggesting that Blue Suckers invested relatively little energy into somatic growth after reaching maturity (Bednarski & Scarnecchia, 2006; Bacula *et al.*, 2009).

I detected a statistical link between flows in the Muddy Boggy River and recruitment in the Red River possibly indicating metapopulation dynamics thereby suggesting that connectivity in the lower Red River is important. The failure to recognize metapopulation dynamics is a major limitation to the management of potamodromous fishes (Thurow, 2016). Understanding how Blue Suckers spawning in the Muddy Boggy River contribute to the Red River tailwater population will be important for the management of the population (Ying *et al.*, 2011; Spurgeon *et al.*, 2018). My results indicated that low-flow winters in Muddy Boggy River result in better recruitment in the Red River, but understanding the mechanisms behind this relationship will require further research. Blue Sucker may leave Muddy Boggy River in response to unsuitable conditions (Lucas & Baras, 2001), or stable conditions associated with low flows may produce an abundance of recruits that disperse due to high densities (Humphries *et al.*, 1999; Mallen-Cooper & Stuart, 2003). Despite uncertainty in the mechanisms of the Muddy Boggy – Red River relationship, it is apparent that connectivity between the locations is important to the population.

The small sample size from the Kiamichi River resulted in a large degree of uncertainty in my recruitment analysis of Kiamichi River fish. The number of parameters in my final models were high compared to the observations from the Kiamichi River resulting in little explanatory power for that population (Babyak, 2004). Given the

uncertainty around my estimates for the Kiamichi population, concluding that the Kiamichi River population was unaffected by environmental conditions is inappropriate. The selection of an interactive effect with location in each of my models suggested that the response of the Kiamichi River population to environmental conditions may be different than that of the Red River population. However, I observed less variability in recruitment in the Kiamichi River compared to the Red River ($R^2 = 0.76$ v. 0.52 , respectively, Chapter 4). Considering the uncertainty in my estimates, it was possible that Blue Suckers in the Kiamichi River had a similar, but weaker response to stream flow. Future work is need to understand the relationships of the Kiamichi River population with environmental conditions.

I used flow variables calculated from daily discharge data in my analyses, but evaluating the effects of flow alterations at a sub-daily resolution would benefit managers. I was unable to quantify the sub-daily flow variation associated with 4 to 12-hr hydropower releases that occurred during the spring season. Consistent sub-daily hydropower releases would appear as stable flows at the daily resolution, without recording the daily fluctuation of $283 \text{ m}^3/\text{s}$ that I observed when sampling. Areas with drastic sub-daily fluctuations in flow can strand eggs and larvae on gravel bars, leaving the nests to desiccate (Grabowski & Isley, 2007, Young *et al.*, 2011). Further, such flows can reduce survivability by stressing and stunting the growth of young-of-year fishes (Weyers *et al.*, 2003), reducing the availability and quality of prey (Bednarek & Hart, 2005; Rolls *et al.*, 2013), and reducing the availability of suitable habitat for young-of-year fishes (King *et al.*, 1998, Olden & Naiman 2010). In dry years, fish may be artificially stimulated to spawn below dams (Bunt, 2001), and avoiding extreme sub-daily flow alterations may produce stable populations within the system, rather than sinks in difficult years (Nagrodski *et al.*, 2012).

The results of my study provide the potential management options of designer flows and sanctuary rivers. Recreating the natural flow regime is often suggested as dam management strategy; however, this strategy is often impractical and may not be necessary (Jager & Smith, 2008; Acreman *et al.*, 2014). Instead, understanding how fish populations have persisted in an altered environment can allow managers to determine and create designer flows (*i.e.*, flow regimes created for specific ecological objectives) that benefit native fishes (Acreman, *et al.*, 2014; Chen & Olden, 2017). An example of a designer flow based on or results would be a large flow pulse early in the growing season (May – June) and then a return to a typical or moderate hydropower operation

for the rest of the season (Acreman *et al.*, 2014). Additionally, if Muddy Boggy River provides a source population for Blue Sucker (*i.e.*, *Muddy Boggy recruitment hypothesis*), it could be treated as a sanctuary river (Moyle & Mount, 2007). Marine sanctuaries are proven to provide source population to exploited saltwater fisheries (Halpern, 2003), and similar to my study, the availability of unregulated tributaries positively influenced the recruitment of Paddelfish *Polyodon spathula* (Prachiel *et al.*, 2009).

The relationship between environmental conditions and fish recruitment and growth are complex, but managers would benefit from a better understanding of the effects of flow alteration on native fishes. My results suggest that seasonal flows are important to the different life-history requirements of Blue Suckers, and that connectivity within the river network is likely important to the conservation of the species. Blue Sucker life history may not be identical to other sympatric large-river fishes, but flow management that benefits Blue Suckers would likely benefit other native species that are adapted to the natural conditions of the Southern Great Plains. Properly-timed designer flows that protect spawning fishes and their offspring and promote individual growth and health may provide a balance between human needs and biodiversity conservation (Acreman *et al.*, 2014). The human demand for water is likely to increase in the future (Vörösmarty *et al.*, 2010), and it is important that I work mitigate the effects of flow regulation on the freshwater fisheries population dynamics.

Table 5.1 Variables included in year-class strength and annual growth analyses.

Variables were adapted from Oden and Poff (2003). Each variable was calculated from mean daily flow data within a season. Variables associated with the adult condition hypothesis were calculated from the year prior to each year class, whereas all others were from the same year as each year class. The growing season corresponded to warm months (June – September), winter corresponded to cooler months (October – January) and spawning season was primarily spring months (February – May).

Hypothesis	Metric	Variable	Description	Gage	Season
Adult Condition	Magnitude	M _A 1	mean discharge	Arthur City	Growing
	Variability	M _A 3	coefficient of variation	Arthur City	Growing
	Variability	M _A 5	Skewness	Arthur City	Growing
	Variability	R _A 6-7	change of flow	Arthur City	Growing
	Frequency	F _L 1	days with Q <25 th percentile	Arthur City	Growing
	temperature	T _{MAX}	maximum temperature	Arthur City	Growing
	Magnitude	M _A 1	mean discharge	Arthur City	Winter
	Variability	M _A 3	coefficient of variation	Arthur City	Winter
	Variability	M _A 5	Skewness	Arthur City	Winter
	Variability	R _A 6-7	change of flow	Arthur City	Winter
	Frequency	F _L 1	days with Q <25 th percentile	Arthur City	Winter
	temperature	T _{MIN}	minimum temperature	Arthur City	Winter
	Red	Magnitude	M _A 1	mean discharge	Arthur

Spawning				City	
	Variability	M _A 3	coefficient of variation	Arthur City	Spawning
	Variability	M _A 5	Skewness	Arthur City	Spawning
	Variability	R _A 6-7	change of flow	Arthur City	Spawning
	Timing	T _H 1	date of maximum flow	Arthur City	Spawning
	Timing	T _H 50	Earliest date Q > median	Arthur City	Spawning
	rate of change	R _A 1	Mean rate of positive changes in flow	Arthur City	Spawning
	rate of change	R _A 3	Mean rate of negative changes	Arthur City	Spawning
	rate of change	R _A 8	Number of flow reversals	Arthur City	Spawning
	Duration	D _H 50	Consecutive days where Q > median	Arthur City	Spawning
Red Recruitment	Magnitude	M _A 1	mean discharge	Arthur City	Growing
	Variability	M _A 3	coefficient of variation	Arthur City	Growing
	Variability	M _A 5	Skewness	Arthur City	Growing
	Variability	R _A 6-7	change of flow	Arthur City	Growing
	Frequency	F _L 1	days with Q <25 th percentile	Arthur City	Growing
	Temperature	T _{MAX}	maximum temperature	Arthur City	Growing
	Magnitude	M _A 1	mean discharge	Arthur City	Winter

	Variability	M _A 3	coefficient of variation	Arthur City	Winter
	Variability	M _A 5	Skewness	Arthur City	Winter
	Variability	R _A 6-7	change of flow	Arthur City	Winter
	Frequency	F _L 1	days with Q <25 th percentile	Arthur City	Winter
	temperature	T _{MIN}	minimum temperature	Arthur City	Winter
Local Spawning	Magnitude	M _A 1	mean discharge	Denison/Hugo	Spawning
	Variability	M _A 3	coefficient of variation	Denison/Hugo	Spawning
	Variability	M _A 5	Skewness	Denison/Hugo	Spawning
	Variability	R _A 6-7	change of flow	Denison/Hugo	Spawning
	Timing	T _H 1	date of maximum flow	Denison/Hugo	Spawning
	Timing	T _H 50	Earliest date Q > median	Denison/Hugo	Spawning
	rate of change	R _A 1	Mean rate of positive changes in flow	Denison/Hugo	Spawning
	rate of change	R _A 3	Mean rate of negative changes	Denison/Hugo	Spawning
	rate of change	R _A 8	Number of flow reversals	Denison/Hugo	Spawning
	Duration	D _H 50	Consecutive days where Q > median	Denison/Hugo	Spawning
Muddy Boggy	Magnitude	M _A 1	mean discharge	Muddy	Spawning

Spawning				Boggy	
	Variability	M _A 3	coefficient of variation	Muddy Boggy	Spawning
	Variability	M _A 5	Skewness	Muddy Boggy	Spawning
	Variability	R _A 6-7	change of flow	Muddy Boggy	Spawning
	Timing	T _H 1	date of maximum flow	Muddy Boggy	Spawning
	Timing	T _H 50	Earliest date Q > median	Muddy Boggy	Spawning
	rate of change	R _A 1	Mean rate of positive changes in flow	Muddy Boggy	Spawning
	rate of change	R _A 3	Mean rate of negative changes	Muddy Boggy	Spawning
	rate of change	R _A 8	Number of flow reversals	Muddy Boggy	Spawning
	Duration	D _H 50	Consecutive days where Q > median	Muddy Boggy	Spawning
Local recruitment	Magnitude	M _A 1	mean discharge	Denison/ Hugo	Growing
	Variability	M _A 3	coefficient of variation	Denison/ Hugo	Growing
	Variability	M _A 5	Skewness	Denison/ Hugo	Growing
	Variability	R _A 6-7	change of flow	Denison/ Hugo	Growing
	Frequency	F _L 1	days with Q < 25 th percentile	Denison/ Hugo	Growing
	Temperature	T _{MAX}	maximum temperature	Denison/ Hugo	Growing
	Magnitude	M _A 1	mean discharge	Denison/ Hugo	Winter

	Variability	M _A 3	coefficient of variation	Hugo Denison/ Hugo	Winter
	Variability	M _A 5	Skewness	Denison/ Hugo	Winter
	Variability	R _A 6-7	change of flow	Denison/ Hugo	Winter
	Frequency	F _L 1	days with Q <25 th percentile	Denison/ Hugo	Winter
	Frequency	F _L 10	days with Q <10 th percentile	Denison/ Hugo	Winter
	temperature	T _{MIN}	minimum temperature	Denison/ Hugo	Winter
Muddy Boggy recruitment	Magnitude	M _A 1	mean discharge	Muddy Boggy	Growing
	Variability	M _A 3	coefficient of variation	Muddy Boggy	Growing
	Variability	M _A 5	Skewness	Muddy Boggy	Growing
	Variability	R _A 6-7	change of flow	Muddy Boggy	Growing
	Frequency	F _L 1	days with Q <25 th percentile	Muddy Boggy	Growing
	temperature	T _{MAX}	maximum temperature	Muddy Boggy	Growing
	Magnitude	M _A 1	mean discharge	Muddy Boggy	Winter
	Variability	M _A 3	coefficient of variation	Muddy Boggy	Winter
	Variability	M _A 5	Skewness	Muddy Boggy	Winter
	Variability	R _A 6-7	change of flow	Muddy Boggy	Winter

Growth model	Frequency	$F_L 1$	days with Q <25 th percentile	Muddy Boggy	Winter
	Temperature	T_{MIN}	minimum temperature	Muddy Boggy	Winter
	Magnitude	$M_A 1$	mean discharge	Arthur City	Growing
	Variability	$M_A 3$	coefficient of variation	Arthur City	Growing
	Variability	$M_A 5$	Skewness	Arthur City	Growing
	Frequency	$F_L 1$	days with Q <25 th percentile	Arthur City	Growing
	Frequency	$F_H 1$	days with Q >75 th percentile	Arthur City	Growing
	Temperature	T_{MEAN}	mean temperature	Arthur City	Growing
	Magnitude	$M_A 1$	mean discharge	Arthur City	Winter
	Variability	$M_A 3$	coefficient of variation	Arthur City	Winter
	Variability	$M_A 5$	Skewness	Arthur City	Winter
	Frequency	$F_L 1$	days with Q <25 th percentile	Arthur City	Winter
	Frequency	$F_H 1$	days with Q >75 th percentile	Arthur City	Winter
	Temperature	T_{MEAN}	mean temperature	Arthur City	Winter

Table 5.2 Candidate models for each year-class strength hypothesis, AIC_c scores, difference in AIC_c (Δ AIC_c), Akaike weights (ω), and conditional R². In the models, β_0 refers to the intercept, β_x coefficients are associated with the selected environmental variables from Table 1, and location j (l_j).

Hypothesis	Model	AIC _c	Δ AIC _c	Ω	R ²
Adult conditioning	$Y_{ij} = \beta_0 + \beta_1 l_j + \beta_2 R_{A67ij} + \beta_3 l_j * R_{A67ij} + \varepsilon_{ij}$	63.7	0	0.41	0.42
Muddy Boggy recruitment	$Y_{ij} = \beta_0 + \beta_1 l_j + F_L 1_{ij} + \beta_2 l_j * F_L 1_{ij} + \varepsilon_{ij}$	64.0	0.3	0.35	0.41
Local spawning	$Y_{ij} = \beta_0 + \beta_1 l_j + \beta_2 R_{A3ij} + \beta_3 l_j * R_{A3ij} + \varepsilon_{ij}$	66.1	2.4	0.12	0.35
Red spawning	$Y_{ij} = \beta_0 + \beta_1 l_j + \beta_2 M_A 1_{ij} + \beta_3 R_{A67ij} + \beta_4 l_j * M_A 1_{ij} + \varepsilon_{ij}$	66.7	3.0	0.09	0.44
Null	$Y_{ij} = \beta_0 + \beta_1 l_j + \varepsilon_{ij}$	68.8	5.1	0.03	0.01

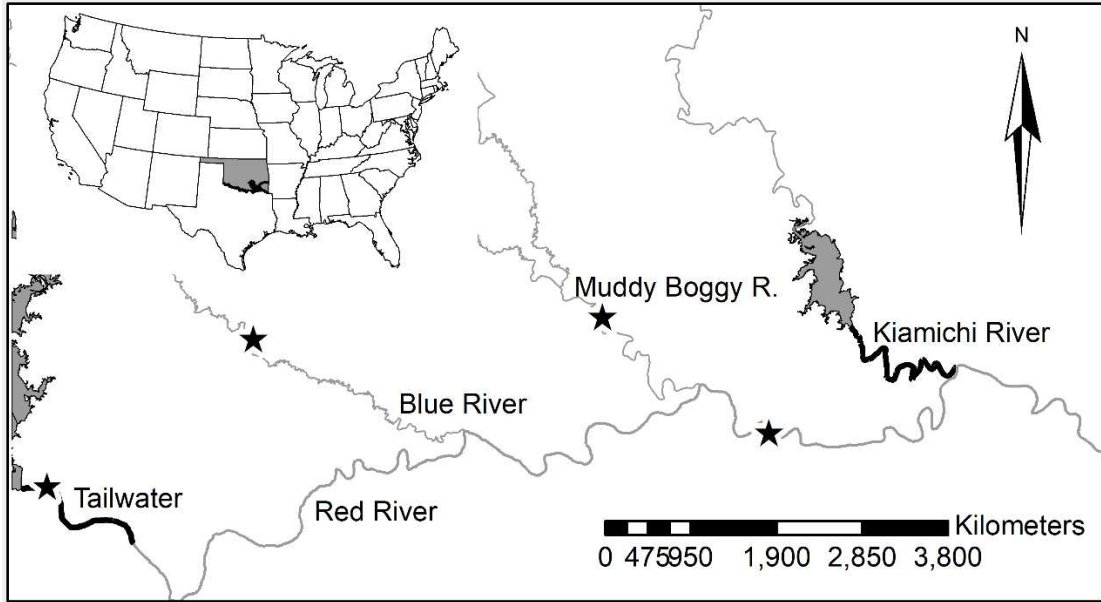


Fig. 5.1 The lower Red River, Oklahoma and major tributaries. Fish were collected in the Tailwater and Kiamichi River. Stars represent streamflow gages and crosses represent temperature loggers used in the study.

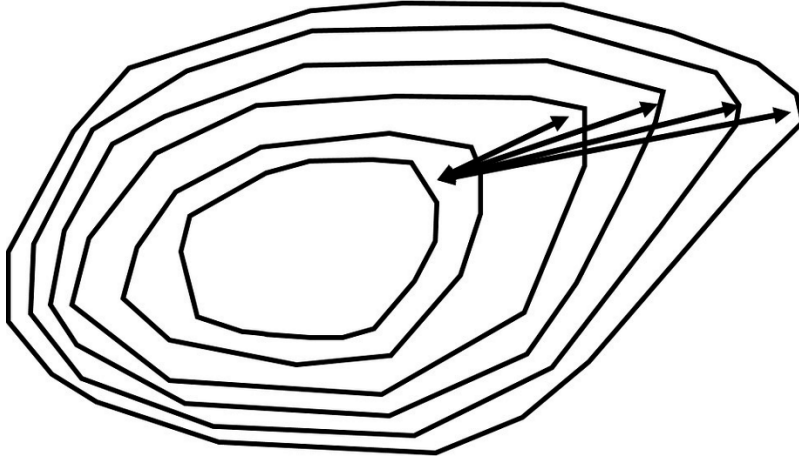


Fig. 5.2 Diagram of pectoral ray used in my growth study. Concentric lines represent annuli and arrows represent incremental measurements between age-1 annulus and the apex of each annulus.

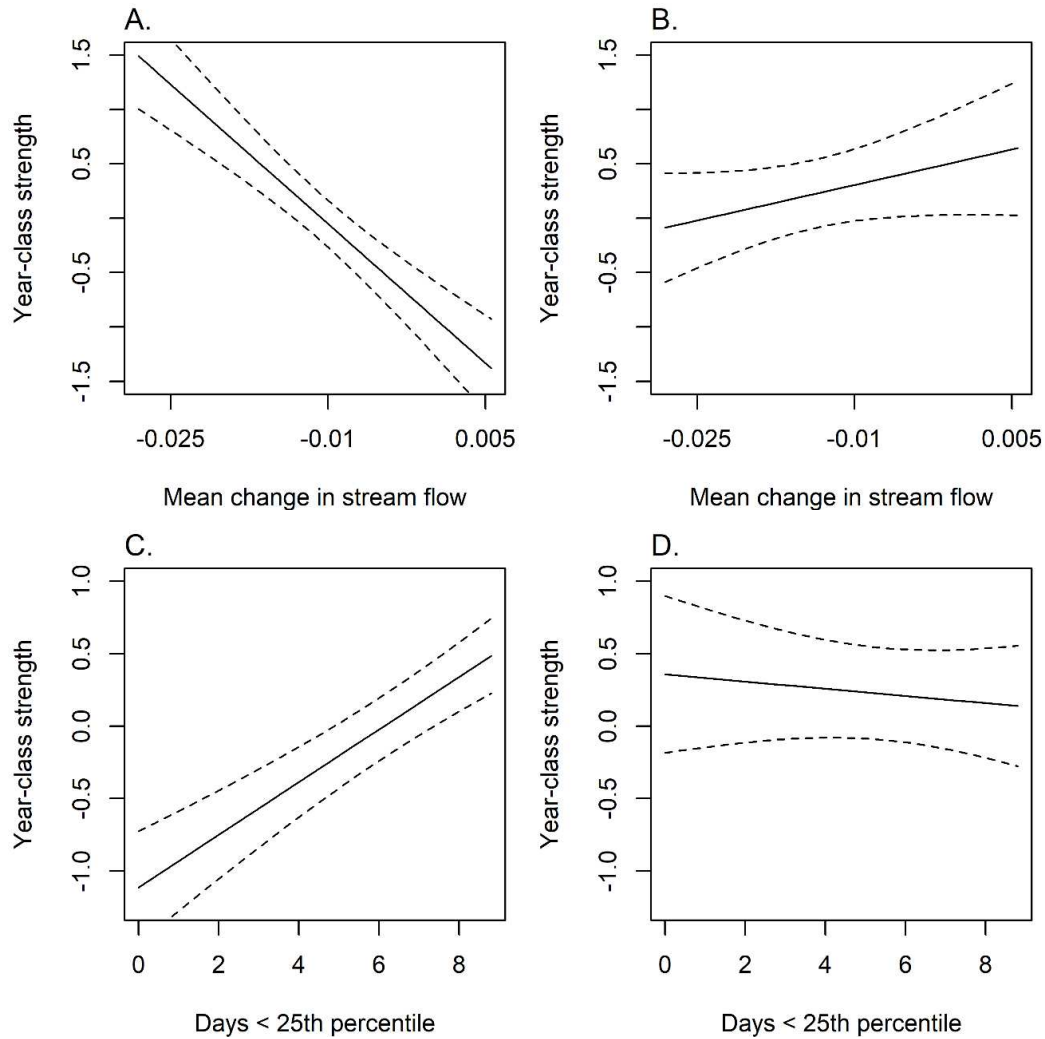


Fig. 5.3 Relationship of Blue Sucker recruitment (year-class strength) with environmental conditions. The top panels represent the *Adult Condition* hypothesis and the relationship of year-class strength with the mean change in stream flow during the growing season prior to spawning for the Red (A.) and Kiamichi (B.) river populations. The bottom panels represent the *Muddy Boggy recruitment* hypothesis and the relationship of year-class strength with the number of days having stream flow < 25th percentile in the Muddy Boggy River during the winter following spawning season for the Red (C.) and Kiamichi (D.) river populations.

CHAPTER VI

MANAGEMENT IMPLICATIONS AND FUTURE RESEARCH

I documented Blue Suckers using the tailwaters of the two major dams in my study, and managers would benefit from considering the effect of dam management practices on the population. Blue Suckers were abundant for longer periods of time in the Hugo and Denison Dam tailwaters compared to the unregulated tributaries. My objectives did not address the link between Blue Sucker movement and attraction to dam release waters, but the phenomenon is established and environmental engineers use flow attraction in the design of fish ladders (Bunt, 2001). The rapid change in stream discharge that is often associated with dam management practices can be detrimental to the survival, growth and recruitment of riverine fishes (King *et al.*, 1998; Weyers *et al.*, 2003; Grabowski & Isley, 2007). Given my observations of tailwater use by Blue Suckers in spawning condition, water managers should consider the effects of the timing, magnitude, and rate of change of controlled discharge on the migratory fishes that may be attracted to artificial flow pulses (Jager & Smith, 2008; Young *et al.*, 2011).

The seasonal movement patterns of Blue Sucker serve as a useful framework for the future study and management of the species. Limited understanding of the scope and seasonality of movement patterns is a major limitation to the identification of manageable populations of riverine fishes (Cooke *et al.*, 2012; Cooke *et al.*, 2016). Blue Suckers were capable of moving throughout the lower Red River, and displayed spawning-site fidelity. Spawning-site fidelity is often used in the identification of manageable populations of migratory fishes (MacLean & Evans, 1981). I documented strong evidence of spawning site fidelity in only one of two years, and uncertainty still exists in the movement patterns of Blue Suckers over their \approx 20-year lifespan. Genetic analysis and otolith microchemistry may provide further insight into the connectivity of

Blue Sucker populations within the river network (e.g., Bertrand *et al.*, 2016; Spurgeon *et al.*, 2018). At a minimum, regular population monitoring would be beneficial to evaluating the complexity of Blue Sucker spawning behavior and movement patterns. Successful management of the species will require an understanding of the rate and spatial dynamics of gene flow throughout the river catchment (Ying *et al.*, 2007; Cooke *et al.*, 2016).

In three of my four objectives, I observed evidence of metapopulation dynamics in the Red River and further investigation is warranted for successful management. Blue Suckers that did not display reach fidelity (*i.e.*, tagged location) typically used Muddy Boggy River (chapter 2), and I observed a statistical link between recruitment in the Red River tailwater and environmental conditions in Muddy Boggy River (chapter 5). Additionally, I observed individuals using the Red River tailwater during the non-spawning season, and moving to Muddy Boggy River during each spawning season (chapter 2). Further, I estimated high survival rate and large recruitment variability in the Red River tailwater population, possibly indicating the presence of immigrants into the population (chapter 4). It is possible that the Red River tailwater represents a sink population that is periodically restored by Muddy Boggy River immigrants, although the evidence of this is tangential. Treating potential sub-populations as separate populations is a safer approach, from a management perspective, than assuming a single large population (Begg *et al.*, 1999), but ignoring the connectivity among populations may result in the failure to recognize population declines (Ying *et al.*, 2011).

My study indicates that protecting connectivity in the lower Red River, and designation of Muddy Boggy River as a sanctuary stream would likely benefit Blue Sucker. Recognition of metapopulation dynamics and source populations provides the management option of designating sanctuary streams. Marine sanctuaries have proven successful in the conservation and management of saltwater populations (Halpern, 2003), and designating unregulated tributaries, like Muddy Boggy River, was recently proposed as a method to mitigate the effects of flow regulation (Moyle & Mount, 2007). Similar to my findings in Chapter 5, environmental flows in unregulated tributaries have a stronger influence on recruitment of Paddelfish *Polyodon spathula* than conditions in the mainstem Missouri River, NE (Prachiel *et al.*, 2009). Preventing flow regulation in Muddy Boggy River and protecting existing connectivity in the lower Red River will be important to the conservation of large-river fishes in the lower Red River.

I provided empirical evidence of the spatial scope of the Blue Sucker population and interstate collaboration would benefit research and management of Blue Suckers and other migratory fishes. My results indicate that residents may exist below Denison Dam, but Blue Suckers using the major tributaries may periodically leave the state boundaries (Chapter 2). I documented four individuals moving between the tributaries and the Oklahoma-Arkansas border. Collaborative efforts with Arkansas, Texas, and Louisiana would benefit Blue Suckers and other migratory large-river fishes. Collaborative management plans that span several geopolitical boundaries have been successful for migratory birds (Anderson *et al.*, 2018) and marine fisheries (Brown, 2017), and similar plans are needed for freshwater fishes.

REFERENCES

- Aarestrup, K., Lucas, M. C., & Hansen, J. A. (2003). Efficiency of a nature-like bypass channel for sea trout (*Salmo trutta*) ascending a small Danish stream studied by PIT telemetry. *Ecology of Freshwater Fish*, *12*, 160-168.
- Acre, M. R., Alejandro, C., East, J., Massure, W. A., Miyazono, S., Pease, J. E., . . . Grabowski, T. B. (2017). Comparison of the precision of age estimates generated from fin rays, scales, and otoliths of Blue Sucker. *Southeastern Naturalist*, *16*(2), 215-224.
- Acreman, M., Arthington, A. H., Colloff, M. J., Couch, C., Crossman, N. D., Dyer, F., . . . Young, W. (2014). Environmental flows for natural, hybrid, and novel riverine ecosystems in a changing world. *Frontiers in Ecology and the Environment*, *12*(8), 466-473.
- Adams, J. D., Bergthold, C. L., Haas, J. D., Pegg, M. A., & Mestl, G. E. (2017). Blue Sucker summer utilization distributions and inter-annual fidelity to summering habitats. *Transactions of the Nebraska Academy of Sciences*, *37*, 18-27.
- Adams, N. S., Beeman, J. W., & Eiler, J. H. (2012). *Telemetry techniques: a user guide for fisheries research*. Bethesda, MD: American Fisheries Society.
- Adams, S. R., Flinn, M. B., Burr, B. M., Whiles, M. R., & Garvey, J. E. (2006). Ecology of larval blue sucker (*Cycleptus elongatus*) in the Mississippi River. *Ecology of Freshwater Fish*, *15*, 291-300.
- Aho, K., Derryberry, D., & Peterson, T. (2014). Model selection for ecologists: the worldviews of AIC and BIC. *Ecology*, *95*, 631-636.
- Akaike, H. (1973). Maximum likelihood identification of Gaussain autoregressive moving average models. *Biometrika*, *60*(2), 255-265.
- Allen, M. S., & Hightower, J. E. (2010). Fish population dynamics: mortality, growth, and recruitment. In W. A. Hubert & M. C. Quist (Eds.), *Inland Fisheries Management in North America, 3rd edition* (pp. 43-80). Bethesda, MD: American Fisheries Society.
- Anderson, M. G., Alisaukas, R. T., Batt, B. D. J., Blohm, R. J., Higgins, K. F., Perry, M. C., . . . Williams, C. K. (2018). The migratory bird treaty and a century of waterfowl conservation. *The Journal of Wildlife Management*, *82*, 247-259.
- Arthington, A. H., Dulvy, N. K., Gladstone, W., & Winefield, I. J. (2016). Fish conservation in freshwater and marine realms: status, threats and management. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *26*, 838-857.
- Babyak, M. A. (2004). What you see may not be what you get: a brief, nontechnical introduction to overfitting in regression-type models. *Psychosomatic Medicine*, *66*, 411-421.
- Bacula, T., Daugherty, D., Sutton, T., & Kennedy, A. (2009). Blue sucker stock characteristics in the Wabash River, Indiana-Illinois, USA. *Fisheries Management and Ecology*, *16*(1), 21-27.
- Ballweber, J., & Schramm, H. L. j. (2010). The legal process in fisheries management. In W. A. Hubert & M. C. Quist (Eds.), *Inland Fisheries Management in North America* (pp. 107-132). Bethesda, MD: American Fisheries Society.

- Barbec, E., Schulte, S., & Richards, P. L. (2002). Impervious surfaces and water quality: a review of current literature and its implications for watershed planning. *Journal of Planning Literature*, 16(4), 499-514.
- Bates, D., Maechler, M., Bolker, B., Walker, S., Christensen, R. H. B., Singman, H., . . . Green, G. (2018). Linear mixed-effects models using 'Eigen' and S4 (Version 1.1-18-1). <http://lme4.r-forge.r-project.org/>: CRAN.
- Beacham, T. D. (1982). Fecundity of coho salmon (*Oncorhynchus kisutch*) and chum salmon (*O. keta*) in the northeast Pacific Ocean. *Canadian Journal of Zoology*, 60, 1463-1469.
- Beamish, R. J., & Harvey, H. H. (1969). Age determination in the White Sucker. *Journal Fisheries Research Board of Canada*, 26(3), 633-638.
- Bednarek, A. T., & Hart, D. D. (2005). Modifying dam operations to restore rivers: ecological responses to Tennessee River Dam mitigation. *Ecological Applications*, 15(3), 997-1008.
- Bednarski, J., & Scarnecchia, D. L. (2006). Age structure and reproductive activity of the Blue Sucker in the Milk River, Missouri River drainage, Montana. *The Prairie Naturalist*, 38(3), 169-182.
- Begg, G. A., Friedland, K. D., & Pearce, J. B. (1999). Stock identification and its role in stock assessment and fisheries management: an overview. *Fisheries research*, 43, 1-8.
- Berglund, A., Rosenqvist, G., & Svensson, I. (1986). Reversed sex roles and parental energy investment in zygotes of two pipefish (Syngnathidae) species. *Marine Ecology Progress Series*, 29, 209-215.
- Bertrand, K. N., VanDeHey, J. A., Pilger, T. J., Felts, E. A., & Turner, T. F. (2016). Genetic structure of a disjunct peripheral population of mountain sucker *Pantosteus jordani* in the Black Hills, South Dakota, USA. *Conservation Genetics*, 17, 775-784.
- Bessert, M. L., & Orti, G. (2008). Genetic effects of habitat fragmentation on blue sucker population in the upper Missouri River (*Cycleptus elongatus* Lesueur, 1918). *Conservation Genetics*, 9, 821-832.
- Bolker, B. M., Brooks, M. E., Clark, C. J., Geange, S. W., Poulsen, J. R., Stevens, M. H. H., & White, J. S. S. (2009). Generalized linear mixed models: a practical guide for ecology and evolution. *Trends in Ecology and Evolution*, 24(3), 127-135.
- Bonvechio, T. F., & Allen, M. S. (2005). Relations between hydrological variables and year-class strength of sportfish in eight Florida waterbodies. *Hydrobiologia*, 532, 193-207.
- Booth, M. T., Flecker, A. S., & Hairston, N. G. J. (2014). Is mobility a fixed trait? Summer movement patterns of catostomids using PIT telemetry. *Transactions of the American Fisheries Society*, 143(4), 1098-1111.
- Brewer, S. K., McManamay, R. A., Miller, A. D., Mollenhauer, R., Worthington, T. A., & Arsuffi, T. (2016). Advancing environmental flow science: Developing frameworks for altered landscapes and integrating efforts across disciplines. *Environmental Management*, 58, 175-192.
- Brown, B. E. (2017). Large marine ecosystem fisheries management with particular reference to Latin America and the Caribbean Sea. *Environmental Development*, 22, 111-119.
- Brownie, C., Hines, J., Nichols, J., Pollock, K., & Hestbeck, J. (1993). Capture-recapture studies for multiple strata including non-Markovian transitions. *Biometrics*, 1173-1187.

- Buckland, S. T., & Garthwaite, P. H. (1991). Quantifying precision of mark-recapture estimates using the bootstrap and related methods. *International Biometric Society*, 47(1), 255-268.
- Buckmeier, D. L., Smith, N. G., Daugherty, D. J., & Bennett, D. L. (2017). Reproductive ecology of Alligator Gar: Identification of environmental drivers of recruitment success. *Journal of the Southeastern Association of Fish and Wildlife Agencies*, 4, 8-17.
- Bunt, C. M. (2001). Fishway entrance modifications enhance fish attraction. *Fisheries Management and Ecology*, 8, 95-105.
- Burnham, K. P., & Anderson, D. R. (2002). *Model Selection and Multimodel Inference: A Practical Information-Theoretic Approach* (2nd ed.). New York, NY: Springer-Verlag.
- Burr, B. M., & Mayden, R. L. (1999). A new species of *Cycleptus* (Cypriniformes: Catostomidae) from Gulf Slope drainages of Alabama, Mississippi, and Louisiana, with a review of the distribution, biology, and conservation status of the genus. *Alabama Museum of Natural History Bulletin*, 20, 19-57.
- Calenge, C. (Producer). (2011, 11/4/2018). Exploratory analysis of the habitat selection by the wildlife in R: the adehabitatHS package. Retrieved from <http://www.idg.pl/mirrors/CRAN/web/packages/adehabitatHS/vignettes/adehabitatHS.pdf>
- Carlisle, D. M., Wolock, D. M., & Meador, M. R. (2011). Alteration of streamflow magnitudes and potential ecological consequences: a multiregional assessment. *Frontiers in Ecology and the Environment*, 9(5), 264-270.
- Carrera-Garcia, E., Kordek, J., Gesset, C., & Jacobs, L. (2017). Tracking juvenile sturgeon in the wild: Miniature tag effects assessment in a laboratory study on Siberian sturgeon (*Acipenser baerii*). *Fisheries research*, 186, 337-344.
- Casselman, J. M. (1983). *Age and growth assessment of fish from their calcified structures - techniques and tools*. Retrieved from Miami, FL:
- Caswell, H. (2001). *Matrix Population Models: Construction, Analysis, and Interpretation*. Sunderland, MA: Sinauer Associates, Inc. Publishers.
- Chen, W., & Olden, J. D. (2017). Designing flows to resolve human and human environmental water needs in a dam-regulated river. *Nature Communications*, 8(2158), 1-10.
- Coker, R. E. (1930). Studies of common fishes of the Mississippi River. *Bulletin of the Bureau of Fisheries*, 141-225.
- Collier, M., Webb, R. H., & Schmidt, J. C. (1996). *Dams and Rivers: A Primer on the Downstream Effects of Dams*. Menlo Park, CA: U.S. Geological Survey.
- Cooch, E., & White, G. C. (2013). *Multi-state models...* Retrieved from <http://www.phidot.org/software/mark/docs/book/>
- Cooch, E., & White, G. C. (2017). *Markov Chain Monte Carlo (MCMC) estimation in MARK...* E. Cooch & G. C. White (Eds.), Retrieved from <http://www.phidot.org/software/mark/docs/book/>
- Cooke, S. J., & Bunt, C. M. (1999). Spawning and reproductive biology of the Greater Redhorse *Moxostoma valenciennesi*, in the Grand River, Ontario. *The Canadian Field Naturalist*, 113, 497-502.
- Cooke, S. J., Bunt, C. M., Hamilton, S. J., Jennings, C. A., Pearson, M. P., Cooperman, M. S., & Markle, D. F. (2005). Threats, conservation strategies, and prognosis for suckers (Catostomidae) in North America: insights from regional case studies of a diverse family of non-game fishes. *Biological Conservation*, 121, 317-331.

- Cooke, S. J., Martins, E. G., Struthers, D. P., Gutowsky, L. F. G., Power, M. E., Doka, S. E., . . . Krueger, C. C. (2016). A moving target-incorporating knowledge of the spatial ecology of fish into the assessment and management of freshwater fish populations. *Environmental Monitoring and Assessment*, 2016, 188-239.
- Cooke, S. J., Paukert, C., & Hogan, Z. (2012). Endangered river fish: factors hindering conservation and restoration. *Endangered Species Research*, 17, 179-191.
- Cooper, C. M. (1993). Biological effects of agriculturally derived surface water pollutants on aquatic systems - a review. *Journal of Environmental Quality*, 22(3), 402-408.
- Cormack, R. M. (1964). Estimates of survival from the sighting of marked animals. *Biometrika*, 51(3/4), 429-438.
- Corriero, A., Desantis, S., Deflorio, M., Acone, F., Bridges, C. R., De La Serna, J. M., . . . De Metrio, G. (2003). Histological investigation on the ovarian cycle of the bluefin tuna in the western and central Mediterranean. *Journal of Fish Biology*, 63, 108-119.
- Coulter, A. A., Keller, D., Amberg, J. J., Bailey, E. J., & Goforth, R. R. (2013). Phenotypic plasticity in the spawning traits of bigheaded carp (*Hypophthalmichthys* spp.) in novel ecosystems. *Freshwater Biology*, 58(5), 1029-1037.
- Cryer, J. D., & Kung-Sik, C. (2008). *Time Series Analysis: With Applications in R* (G. Casella, S. Fienberg, & I. Okin Eds. second ed.). New York, NY: Springer.
- Cushman, R. M. (1985). Review of ecological effects of rapidly varying flows downstream from hydroelectric facilities. *North American Journal of Fisheries Management*, 5, 330-339.
- Daugherty, D. J., Bacula, T. D., & Sutton, T. M. (2008). Reproductive biology of blue sucker in a large Midwestern river. *Journal of Applied Ichthyology*, 24(3), 297-302. doi:10.1111/j.1439-0426.2007.01042.x
- Davies, R. (2015). US president declares disaster for Red River Floods, Louisiana. Retrieved from FloodList.com website: <http://floodlist.com/america/usa/president-declares-disaster-for-red-river-floods-louisiana>
- DeVries, D. R., & Frie, R. V. (1996). Determination of Age and Growth. In B. R. Murphy & D. W. Willis (Eds.), *Fisheries Techniques, 2nd edition* (pp. 483-512). Bethesda, MD: American Fisheries Society.
- Dodds, W. K. (2002). *Freshwater Ecology: Concepts and Environment Applications* (A. P. Covich, J. A. Stanford, R. Stein, & R. G. Wetzel Eds.). San Diego, CA: Academic Press.
- Doledec, S., Chessel, D., & Gimaret-Carpentier, C. (2000). Niche separation in community analysis: a new method. *Ecology*, 81(1), 2914-2927.
- Dolloff, C. A., Flebbe, P. A., & Owen, M. D. (1994). Fish habitat and fish populations in a southern Appalachian watershed before and after Hurricane Hugo. *Transactions of the American Fisheries Society*, 123, 668-678.
- Dudgeon, D., Arthington, A. H., Gessner, M. O., Kawabata, Z. I., Knowler, D. J., Leveque, C., . . . Sullivan, C. A. (2006). Freshwater biodiversity: importance, threats, status and conservation challenges. *Biological Reviews*, 81, 163-182.
- Dutter, A. C., Messing, C., Cailteux, R., Allen, M. S., Pine, W. E., & Strickland, P. A. (2013). Fish recruitment is influenced by river flows and floodplain inundation at Apalachicola River, Florida. *River Research and Applications*, 29, 1110-1118.
- Falke, J. A., Bestgen, K. R., & Fausch, K. D. (2010). Streamflow reductions and habitat drying affect growth, survival, and recruitment of Brassy Minnow across a Great Plains riverscape. *Transactions of the American Fisheries Society*, 139(5), 1566-1583.

- Fausch, K. D., Torgersen, C. E., Baxter, C. V., & Li, H. W. (2002). Landscapes to riverscapes: Bridging the Gap between research and conservation of stream fishes. *BioScience*, *52*, 483-498.
- Fisher, S. J., & Willis, D. W. (2000). Observations of age-0 Blue Sucker, *Cycleptus elongatus* utilizing an upper Missouri River backwater. *Journal of Freshwater Ecology*, *15*(3), 425-427.
- Fisk, J. M. I., Kwak, T. J., & Heise, R. J. (2015). Effects of regulated river flows on habitat suitability for the Robust Redhorse. *Transactions of the American Fisheries Society*, *144*, 792-806.
- Flecker, A. S., McIntyre, P. B., Moore, J. W., Anderson, J. T., Taylor, B. W., & Hall, R. O. j. (2010). Migratory fishes as material and process subsidies in riverine ecosystems. *American Fisheries Society Symposium*, *73*, 559-592.
- Fritz, K. M., Tripe, J. A., & Guy, C. S. (2002). Recovery of three fish species to flood and seasonal drying in a tallgrass prairie stream. *Transactions of the American Fisheries Society*, *105*, 209-218.
- Gelman, A., & Rubin, D. B. (1996). Markov chain Monte Carlo methods in biostatistics. *Statistical Methods in Medical Research*, *5*, 339-355.
- Grabowski, T. B., & Isely, J. J. (2006). Seasonal and diel movements and habitat use of Robust Redhorses in the lower Savannah River, Georgia and South Carolina. *Transactions of the American Fisheries Society*, *135*(5), 1145-1155.
- Grabowski, T. B., & Isely, J. J. (2007). Spatial and temporal segregation of spawning habitat by catostomids in the Savannah River, Georgia and South Carolina, U.S.A. *Journal of Fish Biology*, *70*, 782-798.
- Gross, M. R., Coleman, R. M., & McDowall, R. M. (1988). Aquatic productivity and the evolution of diadromous fish migration. *Science*, *239*, 1291-1293.
- Gulland, J. A. (1982). Why do fish numbers vary? *Journal of Theoretical Biology*, *97*, 69-75.
- Haddon, M. (2001). *Modelling and quantitative methods in fisheries*. London, UK: Chapman & Hall/CRC.
- Halpern, B. S. (2003). The impact of marine reserves: Do reserves work and does reserve size matter? *Ecological Applications*, *13*(1), 117-137.
- Hamill, D., Buscombe, D., & Wheaton, J. M. (2018). Alluvial substrate mapping by automated texture segmentation of recreational-grade side scan sonar imagery *PLoS ONE*, *13*(3), 1-28.
- Hand, G. R., & Jackson, D. C. (2003). Blue sucker stock characteristics in the upper Yazoo River basin, Mississippi, USA. *Fisheries Management and Ecology*, *10*, 147-153.
- Hinch, S. J., Cooke, S. J., Healey, M. C., & Farrell, A. P. (2005). Behavioural physiology of fish migration: salmon as a model approach. In K. A. Sloman, R. W. Wilson, & S. Balshine (Eds.), *Behaviour and Physiology of Fish*. Cambridge, MA: Academic Press.
- Hobbs, N. T., & Hillborn, R. (2006). Alternatives to statistical hypothesis testing in ecology: a guide to self teaching. *Ecological Applications*, *16*(1), 5-19.
- Humphries, P., King, A. J., & Koehn, J. D. (1999). Fish, flows and flood plains: links between freshwater fishes and their environment in the Murray-Darling River system, Australia. *Environmental Biology of Fishes*, *56*, 129-151.
- Iserman, D. A., McKibbin, W. L., & Willis, D. W. (2002). An analysis of methods for quantifying crappie recruitment variability. *North American Journal of Fisheries Management*, *22*(4), 1124-1135.

- Jager, H. I., & Smith, B. T. (2008). Sustainable reservoir operation: can we generate hydropower and preserve ecosystem values? *River Research and Applications*, 24, 340-352.
- James, D. (2018). Great Plains Fish and Wildlife Conservation Office. Retrieved from <https://www.fws.gov/mountain-prairie/fisheries/greatPlainsFWCO.php>
- Jansson, R., Nilsson, C., & Malmqvist, B. (2007). Restoring freshwater ecosystems in riverine landscapes: the roles of connectivity and recovery processes. *Freshwater Biology*, 52, 589-596.
- Jelks, H. L., Walsh, S. J., Burkhead, N. M., Contreras-Balderas, S., Diaz-Pardo, E., Hendrickson, D. A., . . . Warren, M. L. j. (2008). Conservation status of imperiled North American freshwater and diadromous fishes. *Fisheries*, 33(8), 372-407.
- Johnson, J. B., & Omland, K. S. (2004). Model selection in ecology and evolution. *Trends in Ecology and Evolution*, 19(2), 101-108.
- Jolly, G. M. (1965). Explicit estimates from capture-recapture data with both death and immigration-stochastic model. *Biometrika*, 1/2(225-247).
- Jowett, I. G. (2003). Hydraulic constraints on habitat suitability for benthic invertebrates in gravel-bed rivers. *River Research and Applications*, 19, 495-507.
- Junk, W. J., Bayley, P. B., & Sparks, R. E. (1989). The flood pulse concept in river-floodplain systems. *Proceedings of the International Large River Symposium Canadian Special Publication of Fisheries and Aquatic Science*, 106, 110-127.
- Kaesler, A. J., & Litts, T. L. (2010). A novel technique for mapping habitat in navigable streams using low-cost side scan sonar. *Fisheries*, 35(4), 163-174.
- Karam, A. P., Kesner, B. R., & Marsh, P. C. (2008). Acoustic telemetry to assess post-stocking dispersal and mortality of razorback sucker *Xyrauchen texanus*. *Journal of Fish Biology*, 73, 719-727.
- Karr, J. R., & Schlosser, I. J. (1978). Water resources and the land-water interface *Science*, 201(4352), 229-234.
- Keefer, M. L., & Caudill, C. C. (2014). Homing and straying by anadromous salmonids: a review of mechanisms and rates. *Reviews in Fish Biology and Fisheries*, 24, 333-368.
- Kiffney, P. M., Pess, G. R., Anderson, J. H., Faulds, P., Burton, K., & Riley, S. C. (2009). Changes in fish communities following recolonization of the Cedar River, WA USA by Pacific Salmon after 103 years of local extirpation. *River Research and Applications*, 25, 438-452.
- King, J., Cambray, J. A., & Impson, N. D. (1998). Linked effects of dam-released floods and water temperature on spawning of the Clanwilliam yellowfish *Barbus capensis*. *Hydrobiologia*, 384, 245-265.
- Labay, S., Kral, J., & Stukel, S. (2011). Precision of age estimates derived from scales and pectoral fin rays of blue sucker. *Fisheries Management and Ecology*, 18(5), 424-430.
- Lauber, T. B., Stedman, R. C., Decker, D. J., & Knuth, B. A. (2011). Linking knowledge to action in collaborative conservation. *Conservation Biology*, 25(6), 1186-1194.
- Lawless, J. F. (1987). Negative binomial and mixed poisson regression. *The Canadian Journal of Statistics / La Revue Canadienne de Statistique*, 15(3), 209-225.
- Layher, W. G. (1998). *Status and distribution of pallid sturgeon, blue sucker, and other large river fishes in the Red River, Arkansas*. Retrieved from Pine Bluff, AR:
- Layher, W. G. (2007). *Life history of the Blue Sucker in the Red River, Arkansas*. Retrieved from Arkansas Game and Fish Commission:
- Leberon. (1992). Modeling survival.

- Leggett, W. C., & Carscadden, J. E. (1978). Latitudinal variation in reproductive characteristics of American Shad (*Alosa sapidissima*): evidence for population specific life history strategies in fish. *Journal of Fisheries Research Board of Canada*, 35, 1469-1478.
- Leonard, J. L., & Lukowiak, K. (1984). Male-female conflict in a simultaneous hermaphrodite resolved by sperm trading. *The American Midland Naturalist*, 124, 282-286.
- Lucas, M., & Baras, E. (2001). *Migration of Freshwater Fishes*. Ames, IA: Blackwell Science Ltd.
- Lyons, J., Walchak, D., Haglund, J., Kanehl, P., & Pracheil, B. (2016). Habitat use and population characteristics of potentially spawning shovelnose sturgeon *Scaphirhynchus platyrhynchus* (Rafinesque, 1820), blue sucker (*Cycleptus elongatus* Lesueur, 1817), and associated species in the lower Wisconsin River, USA. *Journal of Applied Ichthyology*, 2016, 1-13.
- Maceina, M. J. (1997). Simple application of using residuals from catch-curve regressions to assess year-class strength in fish. *Fisheries research*, 32, 115-121.
- MacLean, J. A., & Evans, D. O. (1981). The stock concept, discreteness of fish stocks, and fisheries management. *Canadian Journal of Fisheries and Aquatic Research*, 38, 1889-1898.
- Mallen-Cooper, M., & Stuart, I. G. (2003). Age, growth and non-flood recruitment of two potamodromous fishes in a large semi-arid/temperate river system. *River Research and Applications*, 19, 697-719.
- Man, A., Law, R., & Polunin, N. V. C. (1995). Role of marine reserves in recruitment to reef fisheries: A metapopulation model. *Biological Conservation*, 71(197-204).
- Manly, B. F. J., McDonald, L. L., Thomas, D. L., McDonald, T. L., & Erickson, W. (2002). Resource selection by animals: statistical design and analysis for field studies. *Journal of Animal Ecology*, 63(3), 1-215.
- Mathez, E. A. (2009). *Climate Change: The Science of Global Warming and Our Energy Future*. Chichester, NY: Columbia University Press.
- Mazerolle, M. J. (2017). Model selection and multimodel inference based on (Q) AIC(c) (Version 2.1-1). CRAN.
- McBride, R. S., Somarakis, S., Fitzhugh, G. R., Albert, A., Yaragina, N. A., Wuenschel, M. J., . . . Basilone, G. (2015). Energy acquisition and allocation to egg production in relation to fish reproductive strategies. *Fish and Fisheries*, 16, 23-57.
- Melnychuk, M. C. (2012). Detection efficiency in telemetry studies: definitions and evaluation methods. In N. S. Adams, J. W. Beeman, & J. H. Eiler (Eds.), *Telemetry Techniques: A Users Guide for Fisheries Research* (pp. 339-357). Bethesda, MD: American Fisheries Society.
- Mettee, M. F., Shepard, T. E., O'Neil, P. E., & McGregor, S. W. (2015). Biology, spawning, and movements of *Cycleptus meridionalis* in the lower Alabama River, Alabama. *Southeastern Naturalist*, 14(1), 147-172.
- Miranda, L. E., & Bettoli, P. W. (2007). Mortality. In C. S. Guy & M. L. Brown (Eds.), *Analysis and interpretation of freshwater fisheries data*. (pp. 229-278). Bethesda, MD: American Fisheries Society.
- Miranda, L. S. (2009). Standardizing electrofishing power for boat electrofishing. In S. A. Bonar, W. A. Hubert, & D. W. Willis (Eds.), *Standard Methods for Sampling North American Freshwater Fishes* (pp. 223-230). Bethesda, MD: American Fisheries Society.

- Morey, N. M., & Berry, C. R. j. (2003). Biological characteristics of the Blue Sucker in the James River and the Big Sioux River, South Dakota *Journal of Freshwater Ecology*, 18(1), 33-41.
- Morrill, J. C., Bales, R. C., & Conklin, M. H. (2005). Estimating stream temperature from air temperature: implications for future water quality *Journal of Environmental Engineering*, 131(1), 139-146.
- Moss, R. E., Scanlan, J. W., & Anderson, C. A. (1983). Observations in the natural history of the Blue Sucker (*Cycleptus elongatus*) in the Neosho River. *American Midland Naturalist*, 109(1), 15-22.
- Moyle, P. B., & Cech, J. J. J. (2004). *Fishes: An Introduction to Ichthyology* (5th ed.). Upper Saddle River, NJ: Prentice Hall.
- Moyle, P. B., & Mount, J. F. (2007). Homogenous rivers, homogenous faunas. *Proceeding of the National Academy of Science*, 104(14), 5711-5712.
- Muller, B., Berg, M., Yao, Z. P., Zhang, X. F., Wang, D., & Pfluger, A. (2008). How polluted is the Yangtze river? Water quality downstream from the Three Gorges Dam. *Science of the Total Environment*, 402, 232-247.
- Nagrodski, A., Raby, G. D., Hasler, C. T., Taylor, M. K., & Cooke, S. J. (2012). Fish stranding in freshwater systems: sources, consequences, and mitigation. *Journal of Environmental Management*, 103, 133-141.
- Nakagawa, S., & Schielzeth, H. (2013). A general and simple method for obtaining R^2 from generalized linear mixed-effects models. *Methods in Ecology and Evolution*, 4, 133-142.
- NatureServe. (2018). NatureServe Explorer: An online encyclopedia of life [web application] 7.1. Retrieved from <http://explorer.natureserve.org>
- Neely, B. C., Pegg, M. A., & Mestl, G. E. (2009). Seasonal use distributions and migrations of blue sucker in the Middle Missouri River. *Ecology of Freshwater Fish*, 18, 437-444.
- Neely, B. C., Pegg, M. A., & Mestl, G. E. (2010). Seasonal resource selection by blue suckers *Cycleptus elongatus*. *Journal of Fish Biology*, 76, 836-851.
- O'Dee, S. H., & Watters, G. T. (2000). New or confirmed host identifications for ten freshwater mussels. *Proceedings of the Conservation, Captive Care, and Propagation of Freshwater Mussels Symposium*, 1998, 77-82.
- Ogle, D. (2018). Simple Fisheries Stock Assessment Methods (Version 0.8.20). CRAN.
- Olden, J. D., & Naiman, R. J. (2010). Incorporating thermal regimes into environmental flows assessments: modifying dam operations to restore freshwater ecosystem integrity. *Freshwater Biology*, 55, 86-107.
- Olden, J. D., & Poff, N. L. (2003). Redundancy and the choice of hydrologic indices for characterizing streamflow regimes. *River Research and Applications*, 19, 101-121.
- Oliver, D. C., Kelso, W. E., & Kaller, M. D. (2017). Relative abundance, movements, and habitat use of Southeastern Blue Sucker *Cycleptus meridionalis* in the lower Pearl River, Louisiana. *American Midland Naturalist*, 177, 263-276.
- Page, L. M., & Johnston, C. E. (1990). Spawning in the creek chubsucker, *Erimyzon oblongus*, with a review of spawning behavior in suckers (Catostomidae). *Environmental Biology of Fishes*, 27, 265-272.
- Panfili, J., de Pontual, H., & Wright, P. J. (2002). *Manual of Fish Sclerochronology* (J. Panfli, H. de Pontual, & P. J. Wright Eds.). Brest, France: Ifremer-IRD.
- Peterson, M. S., Nicholson, L. C., Snyder, D. J., & Fulling, G. L. (1999). Growth, spawning, preparedness, and diet of *Cycleptus meridionalis* (Catostomidae). *Transactions of the American Fisheries Society*, 128(5), 900-908.

- Poff, N. L., Allan, D., Bain, M. B., Karr, J. R., Prestegard, K. L., Richter, B. D., . . . Stromberg, J. C. (1997). The natural flow regime. *BioScience*, 47(11), 769-784.
- Poff, N. L., Olden, J. D., Merritt, D. M., & Pepin, D. M. (2007). Homogenization of regional river dynamics by dams and global biodiversity implications. *Proceedings of the National Academy of Sciences*, 104(14), 5732-5737.
- Poff, N. L., & Schmidt, J. C. (2016). How dams can go with the flow: Small changes to water flow regimes from dams can help to restore river ecosystems. *Science*, 353, 1099-1100.
- Pracheil, B., McIntyre, P. B., & Lyons, J. D. (2013). Enhancing conservation of large-river biodiversity by accounting for tributaries. *Frontiers in Ecology and the Environment*, 11(3), 124-128.
- Pracheil, B. M., Pegg, M. A., & Mestl, G. E. (2009). Tributaries influence recruitment of fish in large rivers. *Ecology of Freshwater Fish*, 18, 603-609.
- Pringle, C. M. (1997). Exploring how disturbance is transmitted upstream: going against the flow. *Journal of the North American Benthological Society*, 16(2), 425-436.
- Pringle, C. M. (2001). Hydrologic connectivity and the management of biological reserves: A global perspective. *Ecological Application*, 11(4), 981-998.
- Probst, D. L., & Gido, K. B. (2004). Responses of native and nonnative fishes to natural flow regime mimicry in the San Juan River. *Transactions of the American Fisheries Society*, 133, 922-931.
- Pusey, B. J., & Arthington, A. H. (2003). Importance of the riparian zone to the conservation and management of freshwater fish: a review. *Marine and Freshwater Research*, 54, 1-16.
- Quist, M. C., & Spiegel, J. R. (2012). Population demographics of catostomids in large river ecosystems: effects of discharge and temperature on recruitment dynamics and growth. *River Research and Applications*, 28, 1567-1586.
- Rechisky, E. L., & Welch, D. W. (2010). Surgical implantation of acoustic tags: Influence of tag loss and tag-induced mortality on free-ranging and hatchery-held spring Chinook (*O. tshawytscha*) smolts. *PNAMP Special Publication: tagging, telemetry and marking neasures for monitoring fish populations-A compendium of new and recent science for use in informing technique and decision modalities: Pacific Northwest Aquatic Monitoring Partnership Special Publication*, 2, 71-96.
- Reynolds, J. B. (1996). Electrofishing. In B. R. Murphy & D. W. Willis (Eds.), *Fisheries Techniques, 2nd edition* (pp. 221-254). Bethesda, MD: American Fisheries Society.
- Ricker, W. E. (1975). Computation and interpretation of biological statistics of fish populations. *Bulletin of the Fisheries Research Board of Canada*, 191, 1-382.
- Ridgeway, M. S., MacLean, J. A., & MacLeod, J. C. (1991). Nest-site fidelity in a centrachid fish, the smallmouth bass (*Micropterus dolomieu*). *Canadian Journal of Zoology*, 69, 3103-3105.
- Rodriguez, M. A. (2002). Restricted movement in stream fish: the paradigm is incomplete, not lost. *Ecology*, 83(1), 1-13.
- Rolls, R. J., Grown, I. O., Khan, T. A., Wilson, G. G., Ellison, T. L., Prior, A., & Waring, C. C. (2013). Fish recruitment in rivers with modified discharge depends on the interacting effects of flow and thermal regimes. *Freshwater Biology*, 58, 1804-1819.
- Rupprecht, R. J., & Jahn, L. A. (1980). Biological notes on Blue Suckers in the Mississippi River. *Transactions of the American Fisheries Society*, 109(3), 323-326.
doi:10.1577/1548-8659(1980)109<323:bnobsi>2.0.co;2

- Rutherford, A., Kelso, W. E., Bryan, F., & Glenn, C. (1995). Influence of physicochemical characteristics on annual growth increments of four fishes from the lower Mississippi River. *Transactions of the American Fisheries Society*, 124, 687-697.
- S., W., & Burton, W. (1993). Enhancement of fish feeding and growth after an increase in minimum flow below the Conowingo Dam. *North American Journal of Fisheries Management*, 13, 103-109.
- Schneider, C. A., Rasband, W. S., & Eliceiri, K. W. (2012). NIH Image to ImageJ: 25 years of image analysis. *Nature Methods*, 9(7), 676-682.
- Schwarz, C. J., & Arnason, A. N. (1996). A general methodology for the analysis of capture-recapture experiments in open populations. *Biometrics*, 52(3), 860-873.
- Scoppetone, G. C., Rissler, P. H., & Buettner, M. E. (2000). Reproductive longevity and fecundity associated with nonannual spawning in Cui-ui. *Transactions of the American Fisheries Society*, 129, 658-669.
- Seber, G. A. F. (1965). A note on the multiple-recapture census. *Biometrika*, 52(1/2), 249-259.
- Semmens, K. (1985). Induced spawning of the Blue Sucker (*Cycleptus elongatus*). *The Progressive Fish Culturist*, 47, 119-120.
- Spurgeon, J. J., Pegg, M. A., & Halden, N. M. (2018). Mixed-origins of channel catfish in a large-river tributary. *Fisheries research*, 198, 195-202.
- Stanford, J. A., Hauer, F. R., & Ward, J. V. (1988). Serial discontinuity in a large river system. *Internationale Vereinigung fur theoretische und angewandte Limnologie: Verhandlungen*, 23(2), 1114-1118.
- Strahler, A. N. (1957). Quantitative analysis of watershed geomorphology. *Transactions of the American Geophysical Union*, 38(6), 913 - 920.
- Stromberg, J. C., Lite, S. J., Marler, R., Paradzick, C., Shafroth, P. B., Shorrock, D., . . . White, M. S. (2007). Alter stream-flow regimes and invasive plant species: the tamarix case. *Global Ecology and Biogeography*, 16(3), 381-393.
- Survey, O. C. (2018). Precipitation History - Annual, South Central. Retrieved from http://climate.ok.gov/index.php/climate/climate_trends/precipitation_history_annual_statewide/CD08/prcp/Annual
- Team, Q. D. (2014). QGIS Geographic Information System. (Version 2.18). <http://qgis.osgeo.org>: Open Source Geospatial Foundation Project.
- Team, R. C. (2018). R: A language and environment for statistical computing (Version 3.4.4). Vienna, Austria: R Foundation for Statistical Computing. Retrieved from <https://www.R-project.org/>
- Thorrold, S. R., Latkoczy, C., Swart, P. K., & Jones, C. M. (2001). Natal homing in a marine fish metapopulation. *Science*, 291, 297-299.
- Thorstad, E. B., Okland, F., Aarestrup, K., & Heggberget, T. G. (2008). Factors affecting the within-river spawning migration of Atlantic salmon, with emphasis on human impacts. *Reviews in Fish Biology and Fisheries*, 18, 345-371.
- Thurow, R. F. (2016). Life history of potamodromous fishes. In P. Morais & F. Daverat (Eds.), *An Introduction to Fish Migration* (pp. 29-54). Boca Raton, FL: CRC Press Taylor & Francis Group, LLC.
- Vaughan, D. S., & Salla, S. B. (1976). A method for determining mortality rates using the Leslie Matrix. *Transactions of the American Fisheries Society*, 105(3), 380-383.
- Vokoun, J. C., Guerrant, T. L., & Rabeni, C. F. (2003). Demographics and chronology of a spawning aggregation of Blue Sucker (*Cycleptus elongatus*) in the Grand River, Missouri, USA. *Journal of Freshwater Ecology*, 18(4), 567-575.
- Vorosmarty, C. J., McIntyre, P. B., Gessner, M. O., Dudgeon, D., Prusevich, A., Green, P., . . . Davies, P. M. (2010). Global threats to human water security and river biodiversity. *Nature*, 467(555-561).

- Ward, J. V., & Stanford, J. A. (1983). The serial discontinuity concept of lotic ecosystems. In T. D. Fontaine & S. M. Bartell (Eds.), *Dynamics of Lotic Ecosystems* (pp. 29-42). Ann Arbor, MI: Ann Arbor Science Publishers.
- Weisberg, S., Spangler, G., & Richmond, L. S. (2010). Mixed effects models for fish growth. *Canadian Journal of Fisheries and Aquatic Research*, *67*, 269-277.
- Weiss, S. J., Otis, E. O., & Maughan, O. E. (1998). Spawning ecology of the flannelmouth sucker, *Catostomus luttipinnis* (Catostomidae), in two small tributaries of the lower Colorado River. *Environmental Biology of Fishes*, *52*(4), 419-433.
- Welch, D. W., Batten, S. D., & Ward, B. R. (2007). Growth, survival, and tag retention of steelhead trout (*O. mykiss*) surgically implanted with dummy acoustic tags. *Hydrobiologia*, *582*, 289-299.
- Weyers, R. S., Jennings, C. A., & Freeman, M. C. (2003). Effects of pulsed, high-velocity water flow on larval robust redhorse and V-lip redhorse. *Transactions of the American Fisheries Society*, *132*(1), 84-91.
- White, G., & Cooch, E. (2016). Program MARK. Colorado State University. Retrieved from www.phidot.org/software/mark/background/index.html
- White, G. C., Burnham, K. P., & Barker, R. J. (2009). Modeling demographic processes in marked populations. *Environmental and Ecological Statistics*, *3*, 1119-1127.
- Winemiller, K. O., & Jepsen, D. B. (1998). Effects of seasonality and fish movement on tropical river food webs. *Journal of Fish Biology*, *53*, 267-296.
- Winemiller, K. O., & Jepsen, D. B. (2004). Migratory neotropical fish subsidize food webs of oligotrophic blackwater rivers. In A. Polis, M. E. Power, & G. R. Huxel (Eds.), *Food webs at the landscape level*. (pp. 115-132). Chicago, IL: University of Chicago Press.
- Winemiller, K. O., & Rose, K. A. (1992). Patterns of life-history diversifications in North American fishes; implications for population regulation. *Canadian Journal of Fisheries and Aquatic Research*, *49*(10), 2196-2218.
- Wipfli, M. S., Hudson, J., & Caouette, J. (1998). Influence of salmon carcasses on stream productivity: response of biofilm and benthic macroinvertebrates in southeastern Alaska U.S.A. *Canadian Journal of Fisheries and Aquatic Research*, *55*, 1503-1511.
- Wohl, E., Bledsoe, B. P., Jacobson, R. B., Poff, N. L., Rathburn, S. L., Walters, D. M., & Wilcox, A. C. (2015). The natural sediment regime in rivers: broadening the foundation for ecosystem management. *BioScience*, *65*(4), 358-371.
- Woods, A. J., Omernik, D. R., Butler, D. R., Ford, J. G., Henley, J. E., Hoagland, B. W., . . . Moran, B. C. (Producer). (2005, 8/23/2016). Ecoregions of Oklahoma (color poster with map, descriptive text, summary tables, and photographs). Retrieved from ftp://newftp.epa.gov/EPADDataCommons/ORD/Ecoregions/ok/ok_front.pdf
- Worthington, T. A., Echelle, A. S., Perkin, J. S., Mollenhauer, R., Farless, N., Dyer, J. J., . . . Brewer, S. K. (2018). The emblematic minnows of the North American Great Plains: A synthesis of threats and conservation opportunities. *Fish and Fisheries*, *19*(2), 271-307.
- Yeager, B. L., & Semmens, K. J. (1987). Early development of the Blue Sucker, *Cycleptus elongatus*. *Copeia*, *1987*(2), 312-316.
- Ying, Y., Chen, Y., L., L., & Gao, T. (2011). Risks of ignoring fish population spatial structure in fisheries management. *Canadian Journal of Fisheries and Aquatic Research*, *68*, 2101-2120.
- Young, P. S., Cech, J. J. J., & Thompson, L. C. (2011). Hydropower-related pulsed flow impacts on stream fishes: a brief review, conceptual model, knowledge gaps, and research needs. *Reviews in Fish Biology and Fisheries*, *21*, 713-731.

- Zar, J. H. (2010). *Biostatistical Analysis, 5th edition*. Upper Saddle River, NJ: Prentice Hall.
- Zymonas, N. D., & Probst, D. L. (2007). *Ecology of Blue Sucker and Gray Redhorse in the lower Pecos River, New Mexico 2000 - 2006*. Retrieved from Santa Fe, NM:

APPENDICIES

APPENDIX A: Detection of Acoustic Telemetry Tags

Electronic transmitters and telemetry studies can provide valuable information about movement and habitat use of fishes, but the methods are not without limitations. The use of electronic transmitters allows multiple observations of individuals at a fine temporal and spatial resolution and may be one of the most important technological advances in the study of fish movement and behavior (Lucas & Baras, 2001). Acoustic and radio transmitters are most commonly implemented in fisheries studies, and each have advantages and disadvantages depending on the situation (Adams *et al.*, 2012). Radio tags transmit a signal through freshwater with low conductivities (*i.e.*, < 800 μS) and air allowing easy detection. However, high water conductivities result in the rapid attenuation of radio waves, and the trailing antenna associated with radio transmitters results in a higher probability of tag loss (Adams *et al.*, 2012). Acoustic tags transmit a signal through water, regardless of water conductivity, but are effectively undetectable in air. Acoustic tags require a relatively uninterrupted underwater path between the transmitter and the receiver, and environmental noise can reduce the detectability of the signal (Adams *et al.*, 2012). I chose to use acoustic tags for my study, do to high conductivities in the mainstem Red River (1200 – 2000 μS) and high retention rates documented in other catostomids (Razorback Sucker *Xyrauchen texanus*; Karam *et al.*, 2008).

I experimented with the limitations of acoustic tags in my study, because riverine environments are less than ideal for acoustic telemetry (Melnychuk, 2012). With acoustic telemetry, there are three sources of variability in detection: 1. the tag must be in range of the receiver at the time the receiver is listening to the channel of the tag, 2. the tag must ping while the receiver is listening to the channel of the tag, and 3. environmental noise cannot interfere with signal transmission. Acoustic signals can be transmitted 1-km in an open environment, but the streambed or stream banks in riverine environments

can interrupt the signal. Therefore, the effective range of an acoustic transmitter is often limited to a single channel unit in the river. Acoustic transmitters are programmed to transmit at different frequencies to avoid signal collisions, requiring acoustic receivers to scan multiple channels (Adams *et al.*, 2012). The reduced range of the acoustic signal in the river reduces the probability of a transmitter pinging within range of the receiver at the time the receiver is scanning the frequency of the signal. The first two sources of variation are random, whereas, the third source is dependent upon environmental conditions. The most common source of environmental noise was associated with streamflow. Elevated stream flows create environmental noise by increasing turbulence in the water column and mobilizing sediment along the streambed (Adams *et al.*, 2012). Therefore, I tested the ability to detect transmitters with my passive telemetry gear and active telemetry protocol in a variety of conditions and locations.

TRANSMITTERS

I acquired 130 acoustic transmitters with variable frequencies and ping intervals for my study, and retained 10 acoustic transmitters to estimate detection error. The frequency of the acoustic transmitters varied between 69 – 83 kHz and ping intervals ranged 1100 – 1500 milliseconds (ms). The combination the frequency and ping interval allowed identification of individuals using the passive receivers (SUR). Ping intervals of tags on the same frequency were ≥ 50 ms apart to avoid mistaken identity. Additionally, each transmitter produced a unique aural sequence to allow identification with active telemetry. The aural sequence was created by 3 – 4 sequences of pings at the rate specified by the ping interval and sequences were separated with a pause 2x the ping interval (www.sonotronics.com, accessed: 11/28/2018).

ACTIVE TRACKING

Methods – I conducted an active detection experiment in each of my tracking reaches to estimate detection error associated with my protocol. I designed the experiment to accommodate the aforementioned sources of variation in transmitter detection. I randomly selected a channel unit for every 5-km of reach length to represent variability in size of channel units at each location. I conducted trials on multiple days to represent different stream flow conditions (2 – 3 days per location). Finally, I replicated each day and each location three times. Each replicate consisted of placing a transmitter near the streambed in a randomly selected channel unit. A tracking crew then carried out the

standard protocol by towing a hydrophone at 7-9 km/h while scanning 15 acoustic frequencies (≈ 3 seconds per channel). Tracking replicates began either at the head of the channel unit or 250-m upstream of the transmitter (large units) and ended at the mouth of the channel unit or 250-m below the transmitter. The receiver operator was unaware of the transmitter's frequency prior to the first detection. To test for bias due to operator awareness of a transmitter in the channel unit, I placed transmitters in the test locations in Red and Kiamichi rivers without the tracking crew's knowledge.

I analyzed the effect of stream flow on acoustic transmitter detection at each study reach. I averaged the binomial response of each replicate for each channel unit within each trial to determine the mean detection probability. I converted the discharge recorded by the USGS stream gage in the study reach at the time of the trial to a percentile that was relative to stream discharge recorded by the gage over the past 20 years. I used a linear regression model to explain the relationship between mean detection probability and the interactive effect of stream discharge and the study reach. I visually assessed residual plots to ensure the assumptions of linear models were satisfied.

I tested for possible bias associated with receiver operator's knowledge of transmitter locations. I used a two-way ANOVA to compare trials where the tracking crew placed the transmitter (known treatment) and trials where the tracking crew was unaware of transmitter presence (blind treatment) in the Kiamichi and Red rivers. I visually inspected residual plots to ensure satisfaction of linear regression assumptions. I assessed statistical significance at $\alpha = 0.05$.

Results – Detection probabilities were unaffected by stream flow except in Blue River where I detected a positive relationship. In each reach except Blue River, I detected a weak negative relationship with discharge (Fig. A.1), but with a considerable amount of uncertainty (slope \pm SE; Red = -0.22 ± 0.21 , Muddy Boggy = 0.28 ± 0.40 , Kiamichi = -0.41 ± 0.84). I detected a strong positive relationship between stream flow and detection probability in Blue River (Blue = 1.64 ± 0.53). Low detection rates in Blue River were associated with flows of $1.8 \text{ m}^3/\text{s}^{-1}$ and resulted in a greater chance of the streambed of woody debris disrupting the acoustic signal. Among the other three study reaches, detection probability was best in the slow moving, deep Kiamichi River (mean = 0.94 ± 0.24 SD), and worst in the often turbulent, wide, and shallow Red River (mean = 0.58 ± 0.41 SD). Although detection probabilities in the lower half of Muddy Boggy River

resembled those of the Kiamichi River, smaller channel units in the upper reaches of Muddy Boggy River were more similar to Blue River reducing the mean probability of detection for the study reach (mean = 0.76 ± 0.28 SD). My model would benefit from data collected at additional stream flows, as it explained 39% of variance (R^2).

I determined that knowledge of a transmitter's presence did not bias the probability of detection in my study. The results of my two-way ANOVA analysis indicated that there was not a significant difference between blind and known treatments ($F_{1,38} = 0.14$, $p = 0.71$). However, detection probabilities were significantly greater in the Kiamichi River than in the Red River ($F_{1,38} = 10.75$, $p < 0.01$).

PASSIVE TRACKING

Methods – I placed SURs within 2-km of each stream's confluence with the Red River, and monitored them throughout the study. I suspended each SUR from buoy that I tethered to an anchor in the stream. I weighted the receiver end of each SUR to ensure the receiver remained submerged. The buoy kept the receiver within 0.5-m of the water surface at low to moderate flows; however, the buoy became submerged when water depth exceed the length of the tether (*i.e.*, > 5 m). Backflow from high flows in the Red River resulted in a calm, quiet acoustic environment in Muddy Boggy and Kiamichi rivers regardless of flows. High flows resulted in turbulent water in Red and Blue rivers, and I was unable to maintain SURs in Blue River. I checked that SURs were functioning every 2 – 3 months, and downloaded data and replaced batteries every 6 – 7 months.

I used observational data from passive and active telemetry to assess the success rate of SUR detections. Initially, I designed an experiment that involved towing acoustic transmitters past an SUR and documented whether it recorded the transmitter. However, this experimental design require unfounded assumptions about Blue Sucker movement behavior and speed in the reach with the SUR. My experimental trials resulted in a single record of the transmitter when the SUR was successful. However, SURs recorded fish multiple times when they entered or left the reach and a single detection was indicative of a false identification. SURs frequently recorded environmental noise, and on one occasion, an SUR in Kiamichi River recorded a frequency and ping interval that matched a transmitter that I recorded in the Red River on the same day. I determined that my experimental trials were not representative of reality, and instead assessed SUR success by comparing the ratio of transmitters

recorded by SURs in each reach to the total number of transmitters recorded in the reach.

Results – The SURs performed best in Muddy Boggy River, but had variable success in Red and Kiamichi rivers. The SURs placed in Muddy Boggy River recorded 91% (20 of 22) of the individuals detected during the springs 2016 and 2017. Success was more variable in the Kiamichi River, where SURs recorded 90% (27 of 30), 59% (10 of 17), and 0% (0 of 3) in springs 2015, 2016, and 2017. I placed SURs 3 – 4 m underwater (50% of depth) in the Kiamichi River prior to spring 2017, in response to vandalism during the summer 2016. Setting the SURs deeper may have prevented further vandalism, but I observed one SUR laying on the streambed during the low flows of 2017. SURs recorded relatively few of the Blue Suckers detected in the tailwater reach, 0% (0 of 23) in 2016, and 31% (5 of 16) in 2017. Both SURs in the Red River tailwater were buried in sand for most of spring 2016 and were incapable of detecting acoustic tags. Further, Blue Suckers likely remained in the Red River tailwater year round (Chapter 2) and would not have passed the SURs.

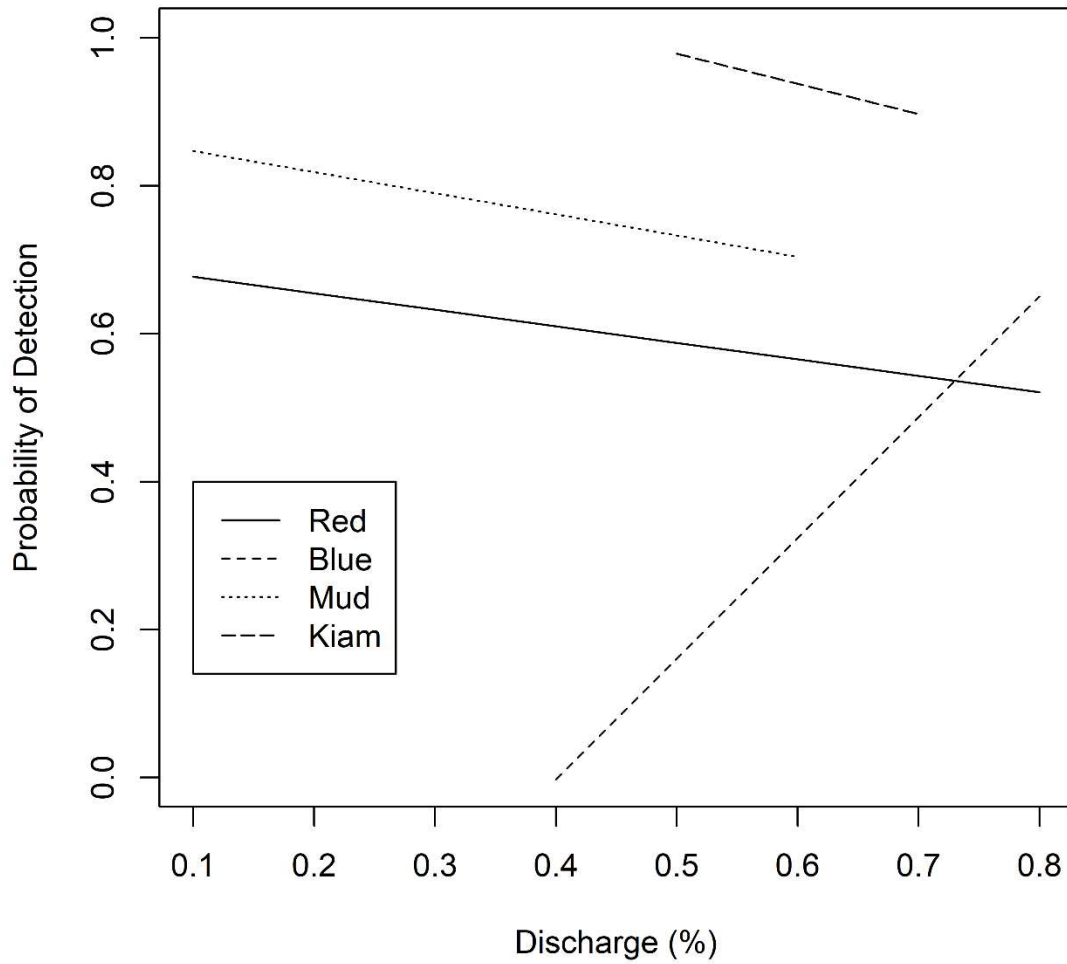


Fig. A.1 Results of active-telemetry, detection-error trials in the lower Red River. Stream discharge is shown as a percentile of the flows observed over the past 20 years in each study reach. I did not extrapolate the lines of any reach outside of my observations.

APPENDIX B: Demographics of Blue Suckers used in Telemetry Studies

I captured and surgically implanted acoustic transmitters into 119 Blue Suckers in the Red River and its major tributaries during spring 2015. I tagged 30 Blue Suckers in each study reach, but one individual perished before release in the Red River tailwater resulting in 29 tagged Blue Suckers at that location. I attributed my only mortality to a gill injury when weighing the individual, and began using a net cradle to weigh fish for the remainder of the study. Tagged male to female ratio was 1:1 ($n = 110$), with 10 fish of unknown gender. I suspect the unknown Blue Suckers to be females who have not yet reached sexual maturity, based on the observed sizes of mature fish (Fig. A.3). The mean total length and weight of tagged males was 564 mm (± 42 SD) and 1357 g (± 357 SD) and 587 mm (± 40 SD) and 1651 g (± 391 SD) for females (Fig. A.2).

I detected 68 individuals on 281 occasions during my study (Table A.1). The majority of the detections were made in 2016 (73%), and 48% and 89% of detections in 2016 and 2017, respectively, were made in Red River tailwater. Overall, I detected 24 individuals from Red River tailwater (166 detections), 14 from Blue River (39 detections), 10 from Muddy Boggy River (22 detections), and 19 from Kiamichi River (54 detections). The ratio of tags to detections was similar for males and females, but fish with unknown sex were detected less frequently (percent detected, F = 63%, M = 55%, U = 30%, Table A.2).

Table A.1 Demographics of Blue Suckers used in telemetry studies (Chapters 2 & 3). The number active tracking recaptures of each individual is provided for each year and location. I indicated individuals that were only detected by passive telemetry with 'SUR' rather than a numeral.

Aural Code	Sex	TL (mm)	WW (g)	Tag Location	Blue	Recaptures							Total	
						2016			2017					
						Kiam	Muddy	Red	Blue	Kiam	Muddy	Red		
3-3-3-4	M	565	1470	Muddy										0
3-3-4	F	621	1900	Red				9					4	13
3-3-4-4	M	561	1200	Red				8					5	13
3-3-4-5	F	555	1750	Blue										0
3-3-4-6	M	634	1950	Kiam			1							1
3-3-4-7	M	567	1460	Blue										0
3-3-5-4	F	567	1680	Blue										0
3-3-5-5	F	631	1990	Muddy										0
3-3-6-5	M	585	1730	Kiam				SUR						0
3-3-6-6	F	644	1960	Muddy										0
3-3-7-7	F	640	1940	Muddy										0
3-3-7-8	M	520	900	Red					12				8	20
3-3-7-8	F	580	1300	Red					12				8	20
3-3-8-4	F	610	1700	Red					6				1	7
3-3-8-6	F	632	1800	Muddy					SUR					0
3-3-8-7	F	610	1820	Blue										0
3-4-3-7	M	499	1120	Blue										0
3-4-3-8	U	519	900	Kiam										0
3-4-4	F	589	1300	Red					2				2	0
3-4-4-8	F	660	2570	Muddy			4							4
3-4-5	M	526	1070	Muddy										0
3-4-5-4	M	613	1570	Muddy										0

Aural Code	Sex	TL (mm)	WW (g)	Tag Location	Recaptures									
					Blue	2016			2017			Total		
						Kiam	Muddy	Red	Blue	Kiam	Muddy		Red	
3-4-5-5	M	504	800	Red					3					3
3-4-5-6	F	575	1300	Red					5			2		7
3-4-6-4	M	630	1720	Blue	2									2
3-4-7-4	F	645	1980	Blue										0
3-4-7-5	F	610	2260	Kiam		3				2				5
3-5-3-5	F	531	1410	Kiam		1								1
3-5-3-6	U	518	990	Blue										0
3-5-3-7	F	519	1150	Blue	8				1					9
3-5-3-8	M	563	1350	Blue	4									4
3-5-4-5	F	589	1570	Kiam					1					1
3-5-4-6	M	520	900	Red			SUR							0
3-5-5	M	574	1500	Red					1			6		7
3-5-5-7	F	534	1180	Blue			SUR							0
3-5-5-8	M	572	1300	Blue										0
3-5-6	M	549	1140	Muddy										0
3-5-6-8	F	625	2350	Kiam		6								6
3-5-7-4	U	533	1070	Muddy			1							1
3-5-7-5	F	608	1500	Red										0
3-5-7-6	M	622	1670	Blue										0
3-5-7-8	M	582	1100	Red										0
3-5-8-4	U	595	1000	Red										0
3-6-3-6	F	569	1460	Blue										0
3-6-3-7	U	524	900	Kiam										0
3-6-4-7	M	516	920	Muddy		SUR						SUR		0
3-6-4-8	F	600	1300	Red										0

Aural Code	Sex	TL (mm)	WW (g)	Tag Location	Blue	Recaptures							Total
						2016				2017			
						Kiam	Muddy	Red	Blue	Kiam	Muddy	Red	
3-6-5	M	568	1400	Kiam		1	1						2
3-6-5-6	M	556	1180	Blue									0
3-6-5-7	M	560	1230	Blue									0
3-6-6	F	562	1650	Muddy			1						0
3-6-6-6	M	614	1940	Kiam		1							1
3-6-6-7	M	524	1120	Blue									0
3-6-7-7	F	539	1540	Kiam									0
3-6-7-8	F	595	1970	Kiam									0
3-7-4-8	M	587	1300	Red				1					1
3-7-5-7	M	515	1200	Kiam		9				3			12
3-7-5-8	F	562	1200	Red				4					4
3-7-6	F	606	1600	Red				4				5	9
3-7-6-7	M	510	1040	Blue	2								2
3-7-7	F	648	2460	Muddy			3						3
3-7-7-4	F	548	1570	Blue			SUR						0
3-8-7	M	531	1340	Kiam									0
3-8-8	M	580	1500	Kiam									0
3-8-8-8	U	494	800	Kiam		SUR							0
4-4-4	M	557	1100	Red									0
4-4-5	F	555	1390	Kiam		2							2
4-4-5-6	M	600	1740	Blue									0
4-4-5-7	F	564	1760	Kiam		1							1
4-4-5-8	M	629	1900	Red				4				6	10
4-4-6-5	M	610	1720	Kiam		3							3
4-4-6-8	F	581	1950	Kiam		1							1

Aural Code	Sex	TL (mm)	WW (g)	Tag Location	Recaptures								Total	
					Blue	2016			2017			Red		
						Kiam	Muddy	Red	Blue	Kiam	Muddy			Red
4-4-7	F	542	1440	Blue			SUR							0
4-4-8	U	577	1350	Muddy										0
4-4-8-8	F	582	1720	Blue			SUR							0
4-5-4-5	M	529	1060	Muddy			3							3
4-5-5-6	M	532	1220	Muddy										0
4-5-5-8	F	576	1400	Red					6			5		11
4-5-6-5	M	512	850	Blue	5									5
4-5-6-8	M	662	2490	Muddy										0
4-5-7-7	F	530	1200	Kiam										0
4-6-4-8	M	565	1450	Muddy										0
4-6-5	F	710	2790	Muddy										0
4-6-5-5	U	509	1000	Kiam										0
4-6-6	F	606	1890	Blue										0
4-6-6-8	M	562	1480	Kiam			SUR							0
4-7-4-7	M	530	700	Red					2					2
4-7-4-8	F	594	1610	Muddy			SUR							0
4-7-7	M	571	1350	Muddy										0
4-7-8	F	514	900	Red					4			2		6
4-8-5	M	677	2160	Kiam			SUR							0
4-8-5-8	M	553	1430	Kiam			1	2						3
4-8-6	M	588	1400	Red					1					1
4-8-8	M	626	2050	Muddy										0
5-5-5	M	544	1290	Muddy								SUR		0
5-5-5-8	F	588	1640	Blue	2									2
5-5-6-6	F	549	1200	Red			2		2		1	1		6

Aural Code	Sex	TL (mm)	WW (g)	Tag Location	Blue	Recaptures							Total	
						2016			2017					
						Kiam	Muddy	Red	Blue	Kiam	Muddy	Red		
5-5-7-7	M	536	1290	Kiam										0
5-5-7-8	F	621	2070	Blue	2									2
5-6-5-7	M	518	900	Kiam		6				1				7
5-6-7	F	567	1430	Muddy										0
5-6-7-8	M	609	1570	Blue	1									1
5-6-8	F	594	1960	Muddy			SUR							0
5-6-8-8	F	572	1500	Red				4				4		8
5-7-6-7	M	511	1050	Blue	6									6
5-7-6-8	M	548	1240	Muddy										0
5-7-8-7	U	512	1050	Muddy										0
5-7-8-8	F	535	1400	Kiam										0
5-8-7	F	591	1500	Red				1						1
5-8-7-8	F	613	2120	Blue								2		2
5-8-8	M	550	1270	Muddy										0
5-8-8-8	F	599	1500	Red				1						1
6-6-6	F	608	1490	Muddy										0
6-6-7-8	U	549	1130	Muddy			4							4
6-7-7	F	565	1490	Kiam										0
6-7-8	F	587	1400	Red			SUR							0
6-8-6-8	M	577	1300	Red				5				5		10
6-8-8-7	M	539	1160	Blue										0
7-8-8	M	533	1340	Kiam		4								4

Table A.2 Summary of telemetry data used in telemetry studies (Chapters 2 & 3). I provide the number of individuals tagged and redetected, and total number of redetections made of fish tagged at each location and of each sex group.

Location where tagged	Female			Male			Unknown		
	Tagged	Detected	Total Detections	Tagged	Detected	Total Detections	Tagged	Detected	Total Detections
Blue	14	8	19	15	6	20	1	0	0
Kiamichi	12	7	17	14	11	36	4	1	1
Muddy	12	6	11	14	3	6	4	2	5
Red	16	14	98	12	10	68	1	0	0
Total	54	35	145	55	30	130	10	3	6

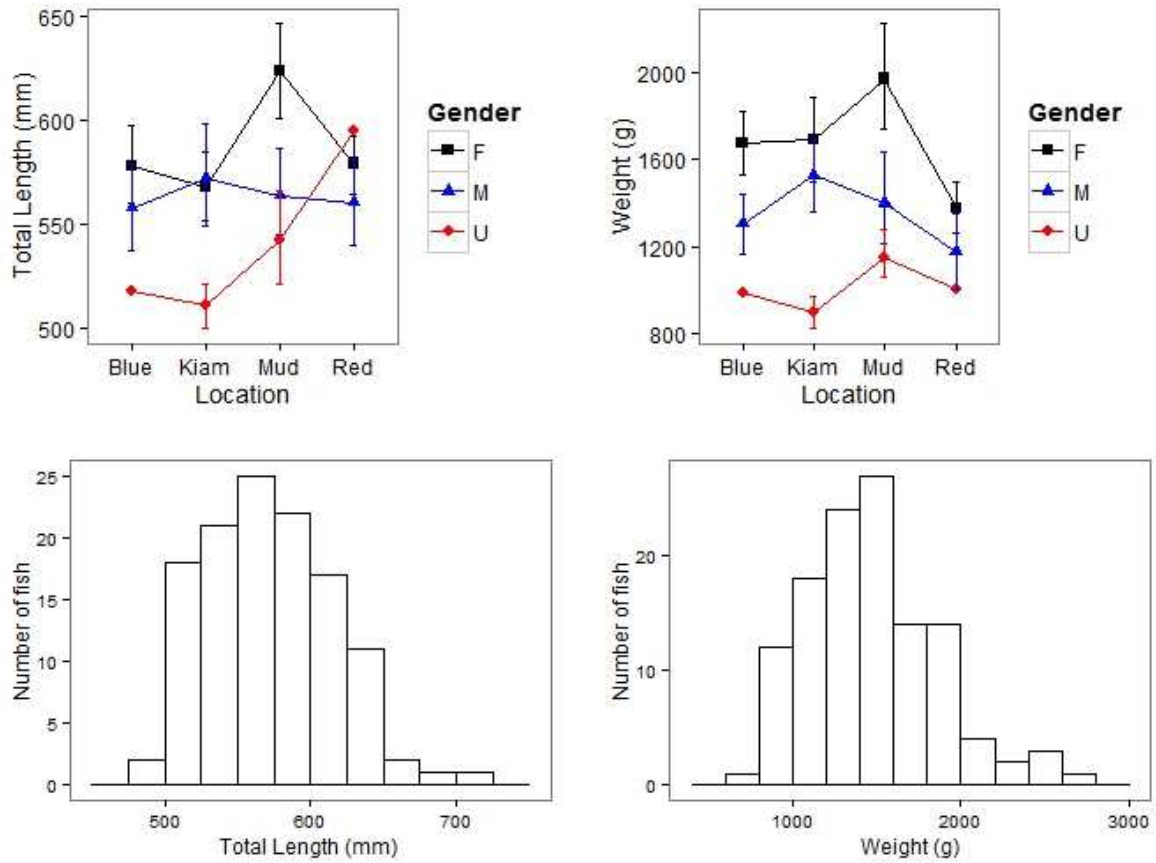


Fig. A.2 Top panel: size distribution of female (black squares), male (blue triangles) and unknown gender (red diamonds) Blue Suckers used in the telemetry studies. Blue Suckers were tagged in the Blue, Kiamichi (Kiam), Muddy Boggy (Mud) and Red rivers. Bottom panel: length and weight distributions of all tagged Blue Suckers.

VITA

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Doctor of Philosophy

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I determined seasonal movement patterns, spawning habitat use, population dynamics, and the relationship between recruitment and environment conditions for Blue Sucker in a Great Plains river system.

I established the spatial distribution and dry season habitat use for four Ouachita Mountain endemic crayfish species.

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