

THE IMPORTANCE OF MAINTAINING SHALLOW-  
WATER HABITATS FOR THE MOVEMENT AND  
SURVIVAL OF STREAM FISHES

By

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**ABSTRACT:** Environmental flows are important for the conservation of stream biota. Although a range of flows are necessary for the persistence of aquatic species, minimum flow standards are often the most basic component. The objectives of this study were to determine the effects of low flows on habitat availability, habitat connectivity, and water temperature and assess the consequences to stream fish. Stream drying disproportionately affects shallow-water habitat availability in streams. Using a wetted-area approach, I found backwaters, riffles, and runs experienced the greatest loss of area in this study with decreasing discharge. Fish assemblage relationships with channel units were quantified using ordinations. Shallow-water channel units (i.e., riffles, runs, vegetated edgewaters) structured much of the fish assemblage in Barren Fork Creek, particularly benthic fishes. Additional fluvial specialists (e.g., cardinal shiner) were found in the Illinois and Flint Creek and related to higher-velocity habitats. Diel shifts in habitat use were observed in all streams suggesting connectivity between channel units to be important for fish. Continuous recaptures over about 50 days of four PIT-tagged species in Flint Creek were analyzed using a multistate model in MARK. I found survival probabilities of cardinal shiner and orangethroat darter were related to daily discharge. More importantly, transition probabilities were related to daily discharge for three species (i.e., cardinal shiner, orangethroat darter, and slender madtom). Transition probabilities were near zero at approximately  $0.43 - 0.57 \text{ m}^3/\text{s}$  suggesting reduced connectivity. Maximum mean daily water temperatures were:  $31.63 \text{ }^\circ\text{C}$  and  $29.55 \text{ }^\circ\text{C}$  for the Illinois River and Flint Creek, respectively. I used SSTEMP to model a 50% reduction in discharge that resulted in a  $0.32 \text{ }^\circ\text{C}$  and  $0.13 \text{ }^\circ\text{C}$  decrease in maximum water temperature in each of the two streams. Temperature modeling of Barren Fork Creek was difficult to interpret because of extreme low flows. Increasing discharge in the Illinois River and Flint Creek showed only minimal reductions in risk of exceeding critical thermal maximum (CTM) for fishes. Flint Creek, however, appeared to offer thermal refugia for many species. My results indicated a discharge of  $0.57 \text{ m}^3/\text{s}$  was a critical threshold for functional connectivity in Flint Creek. Based on channel morphology, I would expect restricted movements to occur above this threshold in the Barren Fork Creek and Illinois River.

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## CHAPTER I

### BACKGROUND

Societies have exploited the benefits of lotic ecosystems for millennia (Naiman et al. 2002) which have global consequences for stream biota (e.g., Australia, Gippel & Stewardson 1995; Asia, Dudgeon 2000; North America, Marchetti & Moyle 2001; Europe, Hughes and Rood 2003; Central America, Anderson et al. 2006; South America, Barletta et al. 2010; and Africa, White et al. 2012). Rivers and streams have been dammed, pumped, leveed, and channelized to meet industrial, agricultural, municipal, and recreational needs. Impoundments block fish migrations (e.g., Dauble & Geist 2000), water abstractions eliminate spawning habitat (e.g., Falke et al. 2010), and loss of flow variability reduces fish biodiversity (Palmer et al. 2008). Approximately 20% of all freshwater fishes are now threatened or endangered due to alteration and destruction of lotic systems (Moyle & Leidy 1992; Naiman et al. 2002).

Future demands on freshwater resources will increase with growing populations, industrialization, and changing climate (Carpenter et al. 1992, Vörösmarty et al. 2000, Gleick 2003, Mohseni et al. 2003, Hulme 2005, Bates et al. 2008, Palmer et al. 2008, Nelson et al. 2009, Vaughan et al. 2009). Global population growth and industrialization exhibit a “Mauna Loa” like curve (Vörösmarty et al. 2000). As countries expand, they increase water abstractions to serve the needs of their burgeoning populations (Jackson et al. 2001). Another consequence of growing societies is more water will be necessary to dilute increased discharge of municipal and industrial

wastewater (Postel et al 1996; Englert et al. 2013). Climate change will also contribute to more frequent and intense periods of drought in subtropical regions (Christensen et al. 2007; Min et al. 2011). Due to these compounding issues, many countries will face water scarcity by 2050 (Petts 2009).

In the United States, freshwater required for municipal, industrial, and agricultural uses has doubled since 1940 and is likely to double again by 2015 (Naiman et al. 1995). Population growth continues to increase, more irrigated lands are currently in cultivation (Balmford 2005), and recent advances in technology (e.g., hydrologic fracturing) have increased industrial water use (Entrekin et al. 2011). Water stress is particularly important to states of the High Plains Aquifer, where extensive abstraction will be compounded by decreased summer flows due to climate change (Döll & Zhang 2010). For example, Oklahoma is expected to see a 33% increase in water use by 2060 which will exceed the available water supply in 67% (55 of 82) of basins (Oklahoma Comprehensive Water Plan, OWRB 2012).

Increased freshwater demands alter the natural flow regime –the “master variable” controlling the biological integrity of lotic ecosystems (Power et al. 1995). Each river has a unique flow regime characterized by magnitude, frequency, duration, seasonal timing, and rate of change (Poff et al. 1997). Lotic biota have evolved with the natural flow regime and the ability of biota to respond to an altered flow regime is limited (Palmer et al. 2008). Shifts outside the natural range of flow variability change the biotic composition of lotic ecosystems (Anderson et al. 2006, Poff & Zimmerman 2010). For example, stream abstractions can produce unseasonal low-flow conditions more often, for longer periods, and with more extreme variability (Smakhtin 2001). These extreme periods of low flow lead to water-quality degradation, loss of habitat and connectivity, prevent completion of life cycles, and facilitate invasion by exotic species (Meyer et al. 1999, Bunn & Arthington 2002).

Environmental flows are legal protections for the natural flow regime that “maintain the components, functions, processes, and resilience of aquatic ecosystems” (Hirji et al. 2009). Environmental flows set target values for one or more aspects of flow regime to conserve key lotic ecosystem services (e.g., propagation of fish and wildlife). Minimum flow values are environmental flows that set a baseline for flow magnitude to maintain aquatic habitat for stream biota, riparian vegetation, human recreation, or aesthetics. During low-flow periods, minimum flows protect from over abstraction where loss of instream habitat is likely. Environmental flows establish the ecosystem as a legitimate water user, which is a critical legal hurdle in states with prior-appropriation water laws (e.g., Oklahoma).

Currently, Oklahoma has no legally-recognized protections for streamflows. Lotic systems lacking flow standards are at risk to degradation without a proposed management plan. This does not necessarily mean the rivers are in immediate peril; nevertheless, a management plan that includes flow recommendations would be highly useful. Unfortunately, minimum flow protections only refer to the amount of water protected from withdrawal (Annear et al. 2002). There is no guarantee that the quantity of water protected is the appropriate amount for aquatic organisms in the system. In many cases, minimum flows are much lower than the average natural flows.

Appropriate environmental flows needed to support fishes and their essential habitats require biological information documenting these needs to support the goal of enhanced streamflows (Reiser et al. 1989). Information particularly useful in supporting environmental flows is related to the fitness consequences of the aquatic community as a result of flow alterations. Habitat requirement is defined as “environmental features necessary for the persistence of individuals or populations” (Rosenfeld 2003). An understanding of the interactions between flow and habitat and the fitness of aquatic organisms provides an important foundation for management of water resources. Without this information, we can only proceed with overly conservative strategies

with negative social and economic consequences (Rosenfeld 2003), or no strategy, which may result in the loss of aquatic biota (Dudgeon et al. 2006) and important recreational opportunities (Daubert & Young 1981, Loehman & Loomis 2008).

The goal of this study was to examine how loss of stream discharge affects shallow-water habitats important for the movement and survival of stream fishes. To attain this goal, I established three objectives: 1) determine the discharge needed to maintain shallow-water habitats and their associated fish assemblage; 2) determine functional connectivity between channel units; and 3) examine the effects reduced flows on summer stream temperatures and the subsequent consequences to fishes. The first objective identified the fish assemblage and life stages (i.e., juvenile or adult) most susceptible to habitat loss with declining discharge. The second objective moved beyond loss of abiotic habitat to determine if biotic movements were affected by changing discharge. Lastly, the third chapter examined the temperature changes related to multiple discharge scenarios to determine which species were most susceptible to stream warming and if maintaining a higher discharge would provide thermal buffering at a level to prevent the critical thermal maxima of fishes from being exceeded.

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## CHAPTER II

### QUANTIFYING CHANGING HABITAT AVAILABILITY OF CHANNEL UNITS IMPORTANT TO OZARK FISH ASSEMBLAGES

#### INTRODUCTION

Environmental flows are important for conserving stream biota against current and future threats of water withdrawals. Rivers and streams are among the most endangered ecosystems on the planet due to anthropogenic demands for freshwater resources and the uncertainty presented by global climate change (Ricciardi & Rasmussen 1999). Current agricultural, industrial, commercial, and municipal needs already push the limits of available freshwater (Petts 1996; Jackson et al. 2001) and climate change will exacerbate anthropogenic variations in the natural flow regime (Vörösmarty et al. 2000). A river's flow regime is the "master variable" controlling the biological integrity of lotic ecosystems (Power et al. 1995) and is characterized by the frequency, timing, duration, magnitude, and rate of change of flows (Poff et al. 1997). Without legal protections, flow alterations will continue to diminish available habitat (Bunn & Arthington 2002). However, managers can protect important aspects of the natural flow regime for native biota by implementing ecologically-meaningful environmental flows (Acreman & Dunbar 2004).

Environmental flows are legally-recognized goals for streamflows that attempt to maintain or mimic some portion of the natural flow regime (Tharme 2003). For example, water releases below dams are controlled to produce natural flood pulses for migrating fish (Savannah River,

Georgia and South Carolina, Richter & Thomas 2007), prevent fish stranding (Columbia and Kootenai rivers, British Columbia; Irvine et al. 2009), and protect riparian habitat (Roanoke River, North Carolina; Richter & Thomas 2007). Although all aspects of the natural flow regime are ecologically important, low-flow magnitude is the most common flow consideration (e.g., Tallapoosa River, Alabama, Travnicek et al. 1994; multiple tributaries to the Tennessee River, Georgia and Tennessee, Bednarek et al. 2005; Oldman River, Alberta, Canada, Rood et al. 2005; Richter 2010). Minimum flows provide limits on water abstractions seasonally or during periods of drought and protect against the reduction of aquatic habitat and, in extreme cases, total stream drying (Jowett 1997).

Habitat is a driving element of biological integrity (Karr & Dudley 1981), but only those minimum flows that protect essential habitats will effectively protect fish and other aquatic biota. Essential habitats are “environmental features necessary for the persistence of individuals or populations” (Rosenfeld 2003). In lotic ecosystems, shallow waters are essential habitat for the production of macroinvertebrate prey items and the completion of the life histories of many stream fishes (Lobb III & Orth 1991, Aadland 1993, Brewer et al. 2006). For example, invertebrate-prey production in riffles supplements entire downstream reaches (Rabeni & Minshall 1977). Riffles are habitats with clean substrates and high dissolved oxygen essential for spawning and egg survival (Berkman & Rabeni 1987). Other shallow-water habitats (e.g., edgewaters and backwaters) act as important predator refugia for juvenile fishes (Schlosser 1987). Unfortunately, stream drying disproportionately affects the availability of shallow-water habitats which makes them the most vulnerable to abstractions.

Biological information is the key to determine essential habitats and to define the threshold for fitness consequences caused by low flows. Stream drying disproportionately affects the availability of shallow-water habitats which increases competition and displaces or isolates species into less favorable habitats (Magoulick & Kobza 2003). However, specific information

related to the habitat use of the aquatic community as a result of flow alterations is needed to assess minimum flow requirements. We need to know which species and age groups are using shallow-water habitats, both day and night, to accurately determine essential habitats. For example, juvenile fishes use shallow-water habitats (Schlosser 1987), so we can anticipate lost discharge may disproportionately affect recruitment. However, the threshold where shallow-water habitats become unsuitable for fish (and the point where fitness consequences are likely) varies geographically by stream and biologically by species and age class. Minimum flows without this supporting biological information risk not meeting the needs of aquatic biota.

The goal of this chapter was to provide biologically-relevant information showing the relationship among habitat, biota, and discharge for three Ozark Highland streams in northeast Oklahoma. To achieve this goal, I had two objectives: 1) determine the relationship between discharge and shallow-water habitats; and 2) identify the fish assemblage using shallow-water habitats both day and night.

## **METHODS**

### *Study Area*

The study streams are located in the Ozark Highlands ecoregion of northeastern Oklahoma, USA. All Ozark Highlands streams drain into the Arkansas River, the second largest tributary to the Mississippi River. Mean annual precipitation in the catchment is 100 cm and mean annual temperature is 13°C (Adamski et al. 1995). Lithology of the Ozark Highlands is mostly limestone and dolostone with interbedded chert (Fenneman 1938). Natural vegetation consists primarily of oak-hickory forest and grasslands, but is increasingly replaced by agriculture and residential areas (Adamski et al. 1995, Splinter et al. 2010). Streams in this area are typical of the Ozark Highlands with relatively clear flowing waters and cobble-gravel substrates.

Study sites were located on the upper Illinois River and its two major tributaries, Flint Creek and Barren Fork Creek (Figure 1). The State of Oklahoma has designated these waterways as “Scenic Rivers” which afford them special protection under the Oklahoma Scenic Rivers Act of 1970 (82 O.S. 1451-1471). The upper Illinois River is a 6<sup>th</sup> order stream with a drainage area of 2484 km<sup>2</sup> (959 mi<sup>2</sup>) and mean annual discharge of 26.8 m<sup>3</sup>/s (946 f<sup>3</sup>/s). Flint Creek is a 3<sup>rd</sup> order tributary with a drainage area of 285 km<sup>2</sup> (110 mi<sup>2</sup>) and mean annual discharge of 3.3 m<sup>3</sup>/s (118 f<sup>3</sup>/s). Barren Fork Creek is a 4<sup>th</sup> order stream with a drainage area of 936 km<sup>2</sup> (307 mi<sup>2</sup>) with a mean annual discharge of 9.3 m<sup>3</sup>/s (329 f<sup>3</sup>/s).

### Habitat characteristics

Channel units (discrete morphological features formed by interactions between the stream and surrounding landscape at high flows; Leopold et al. 1995) represent one of several spatial scales in the hierarchical organization of a stream system (Frissell et al. 1986). Channel units represent a scale at which many management activities in streams take place (e.g., Rabeni & Sowa 1996). I classified channel units, relevant to stream-fish populations in Ozark streams (Peterson & Rabeni 2001c), following descriptions of Rabeni & Jacobson (1993). My approach condensed channel units into five major categories (i.e., riffles, runs, pools, backwaters, and vegetated edgewater) because of their greater availability within the study area. Channel units were selected haphazardly from each study reach.

Microhabitat characteristics (i.e., depth, velocity, substrate composition) were measured haphazardly in each channel unit to verify appropriate classification. I measured depth (0.01 m) with a top-set wading rod and average water-column velocity (60% depth from the surface) using an electromagnetic-flow meter (0.1 m/s; Marsh-McBirney, Frederick, MD). Substrate composition was quantified following modified methods of Bain et al. (1985). Briefly, a 1-m<sup>2</sup> polyvinyl chloride (PVC) frame was used to delineate a specific area in each channel unit and I

visually estimated the percent coverage of each of the following substrate-size classes (mm diameter): silt (0 - 0.4), sand (0.5 – 1.9), gravel (2 - 19), cobble (20 - 200), boulder (> 200), bedrock, and organic material (e.g., aquatic plants, algae, leaves, and woody debris).

I measured discharge at each site and on each sampling occasion using the velocity-area method (Gordon 2004). To ensure accuracy of my measurements, measured discharges were compared to data from the nearest stream gage in Flint Creek (USGS gage #11110103), Barren Fork Creek (USGS gage #11110103), and the upper Illinois River (USGS gage #07196500; Figure 2-1).

#### Wetted-area data

I used a modification of the wetted-perimeter method to relate the surface area of individual channel units to discharge on Flint Creek, Barren Fork Creek, and the Illinois River. The wetted-perimeter method is a technique for defining minimum flows for the base-flow period of a stream (Annear et al. 2004). It uses a graphical representation of wetted perimeter versus discharge as a surrogate for physical habitat. Breakpoints (where there is a rapid change in perimeter) can then be calculated to suggest a minimum flow for each stream. Maintaining flow above the breakpoint is thought to protect a stream's food-producing capacity (HDR 2007) and carrying capacity (Leathe & Nelson 1986). I used surface area instead of wetted perimeter to get more complete information on available habitat of individual channel units.

Sample discharges needed for wetted-area analyses were estimated from annual flow-duration curves created with Indicators of Hydrologic Alteration (IHA) software (Nature Conservancy; Richter et al. 1996). Flow-duration curves represent discharge as a probability of exceeding zero based on historical mean daily discharge and were created for each study site. Samples for wetted-area analyses were selected from exceedance probabilities at 10% increments (e.g., 90%, 80%, 70%, etc.). I only considered flows  $\leq$  20% exceedance probability in the Illinois

River because the river is large and flows > 20% were unsafe for wading. Further, the focus of the project was to identify habitat thresholds under lower-flow conditions. At selected exceedance probabilities, a range of  $\pm 10\%$  discharge was created to allow for flexibility in field sampling.

I used global positioning system (GPS) technology to assess the changes in available channel unit area over multiple discharge scenarios. Channel units were mapped as polygons by walking the perimeter of each channel unit with a differential GPS (Trimble model Juno SB; accuracy  $\pm 1$  m). This allowed channel units to be repeatedly measured quickly and accurately through time (Dauwalter 2006; Reinfelds et al. 2004). Polygon data were differentially corrected in Pathfinder Office (Trimble software) using base station data located in Fayetteville, Arkansas (distance of 71 km). Wetted surface areas of each channel unit were calculated from corrected polygon data in ArcGIS 10.0 (ESRI, Redlands, CA).

Sampling events ( $n = 50$ ) occurred from May 2012 to May 2013 based on discharge (Table 2-1). Some channel units ( $n = 3$ ) were omitted from the analyses because of obvious discrepancies in data points (e.g., increased area at lower flows) caused by occasional low GPS accuracy. On 30 April 2013, preliminary USGS gage data from Barren Fork Creek at Eldon Springs was incorrect, so discharge measured at the site was used for the analyses. All other discharge measurements used USGS gage data.

### Habitat use by fishes

Diel habitat use by fishes in Flint Creek, Barren Fork Creek, and the Illinois River was evaluated by sampling shallow-water channel units during low-flow conditions in summer 2011 and 2012. Most of the fish sampling in 2011 was completed by the Oklahoma Water Resources Board and all 2012 data were collected by Oklahoma State University. Fishes were sampled using a 1-m<sup>2</sup> quadrat sampler (Peterson & Rabeni 2001a). The efficiency of this gear is greater than 50% in



channel units less than 0.5-m deep in Ozark streams (Peterson & Rabeni 2001a, Rettig 2003). This gear is most efficient for capturing Cyprinidae (84%) and Cottidae (80%), but less efficient for Percidae (54%) and Ictaluridae (31%; Peterson & Rabeni 2001a). Overall, the quadrat sampler is an effective gear for sampling many small-bodied fishes, is easy to employ, and produces quantifiable estimates of fish densities.

A sampling method similar to Peterson and Rabeni (2001a) was used to collect fishes with the quadrat sampler. Fish were trapped inside a 1-m<sup>2</sup> netted frame by quickly lowering and securing the quadrat sampler to the stream bottom. Trapped fish were “swept” downstream into an attached bag via natural and hand-induced flow while simultaneously disturbing the substrate to dislodge benthic fishes. Captured fishes were preserved in 10% formalin and brought back to the laboratory for identification. Each fish was identified to species and measured for total length (1.0 mm). Each species was separated into young-of-year and adult life stages because fish often exhibit ontogenetic shifts in habitat use (Schlosser 1987). Length-frequency histograms in combination with published length-at-age data were used to subset each species for analyses (Harvey 2008; Table 2-2).

Study sites were sampled for fishes both day and night. Day sampling occurred during daylight hours (approximately 0700 to 1800). Night sampling occurred from 2200 to 0400, but only at the three downstream study sites on each stream. Night sampling followed the same methods described above, but because sampling conditions were more difficult, I subsampled each channel unit only five times.

### Analyses

Available habitat thresholds (i.e., breakpoints) were determined using broken-line regression (package: segmented) of wetted area-discharge curves in R (Muggeo 2008, R Core Team 2012). This approach is continuous-piecewise linear where the response and explanatory variables are

represented by two straight lines connected at an unknown breakpoint (Muggeo 2008).

Segmented methods have been shown to outperform other breakpoint estimations (e.g., grid search, Julious, and Bayesian) when regression lines are continuous (Chen et al. 2011). I first created graphical representations of wetted surface area to discharge to visually estimate the probable breakpoints for each channel unit. I used initial visual breakpoint estimates as starting values to iteratively fit broken-line models until model convergence (Muggeo 2008).

Significance ( $\alpha \leq 0.05$ ) of each breakpoint was tested using Davie's test, which tests if slopes are significantly different on either side of the breakpoint (Muggeo 2008). If slopes were the same, the wetted area-discharge relationship was considered linear with no breakpoint. If slopes were different, estimates for the optimum breakpoint were provided with 90% confidence intervals. Mean breakpoints were then calculated for each channel unit type by study reach and stream because channel-unit geomorphology influences wetted-area analyses (Reinfelds et al. 2004). I excluded channel units with linear relationships between area and discharge (i.e., no breakpoint) and all pools when averaging across all channel units for stream breakpoints. Pools with significant breakpoints ( $n = 2$ ) were excluded because they lost little area compared to all other channel units. If the goal is to minimize habitat loss, the value for discharge at the breakpoint is the suggested minimum discharge (Robbins et al. 2006).

Fish abundances were analyzed using canonical correspondence analysis (CCA) in CANOCO 4.5 to relate channel unit to fish-community composition. CCAs were developed on each of the three streams for both day and night periods. Each species was separated into two groups (i.e., young-of-year and adult) based on published length-at-age data (Table 2-2). All species and age classes were included in the analyses but were square-root transformed to reduce the influence of the most abundant species (ter Braak 1986). Rare species (i.e., < 1% of total catch) were included in analyses to preserve any species interactions but were removed from the final CCA bi-plots to improve interpretation. A unimodal distribution for species-response curves (i.e., each species

has one optimal habitat) is assumed in CCA analyses because of zeroes in species-catch data (Palmer 1993). Scaling focused on inter-species distances. Significance ( $\alpha \leq 0.05$ ) of canonical axes was tested using an unrestricted Monte-Carlo test with 9,999 permutations.

## RESULTS

### Habitat characteristics

Sampled microhabitats were characteristic of channel units defined by Peterson and Rabeni (2001b; Figure 2-2). Runs and pools were relatively deep channel units, and riffles, backwaters, and vegetated edgewater were shallow. Water velocities were greatest in riffles and runs compared to other channel units. All channel units had predominately cobble-gravel substrates except for backwaters that were mostly smaller substrates and vegetated edgewater that were dominated by organic material (e.g., aquatic macrophytes; Table 2-3). For each stream, backwaters lost the most surface area over the range of measured discharges followed by riffles, runs, and then pools (Table 2-4).

### Wetted-area breakpoints

Flint Creek, the smallest of the three streams, had a mean breakpoint ( $\pm 90\%$  CL) at  $2.12 \text{ m}^3/\text{s} \pm 0.74$  based on riffle, run, and backwater channel units (Table 2-5). Riffles began to experience a significant decrease in available habitat at the highest discharges ( $2.66 \text{ m}^3/\text{s} \pm 0.83$ ) and lost 71.5% of their total area over the sampled discharges. Runs had a significant loss of available area beginning at lower discharges than riffles ( $1.88 \text{ m}^3/\text{s} \pm 0.76$ ) and maintained a majority of their available habitat (decreasing by only 35.8%) at the lowest discharge. However, backwaters had the lowest estimated breakpoint ( $1.23 \text{ m}^3/\text{s} \pm 0.38$ ), but lost the most available habitat (92.7%) over the sampling period. Pools had no significant declines in area over the measured

discharges. Available pool habitat was best described by mean area. Flint Creek's overall mean breakpoint had a flow-exceedance probability of 39% (range: 29 – 56%)—meaning daily discharges were above the breakpoint 39% of the time (Table 2-6). Compared to flow records from August (the month with the lowest discharge), daily discharges were above the mean breakpoint only 9% of the time and ranged from 5% to 23%.

Barren Fork Creek had a mean breakpoint ( $\pm 90\%$  CL) at  $7.39 \text{ m}^3/\text{s} \pm 4.12$  based on upstream ( $8.42 \text{ m}^3/\text{s} \pm 5.13$ ) and downstream ( $6.06 \text{ m}^3/\text{s} \pm 2.82$ ) study reaches (Table 2-7). Backwaters were the most sensitive to habitat loss (breakpoint at  $9.87 \text{ m}^3/\text{s} \pm 5.87$ ) and had the greatest loss of total area (93.1%) over the sampled discharges. In one backwater, the relationship of wetted area-discharge was linear ( $r^2 = 0.98$ ) and it lost 44% of its maximum measured habitat at the mean breakpoint for this habitat type. Runs ( $6.94 \text{ m}^3/\text{s} \pm 3.48$ ) and riffles ( $6.71 \text{ m}^3/\text{s} \pm 3.92$ ) had breakpoints considerably lower ( $\sim 3 \text{ m}^3/\text{s}$ ) than backwaters. Run and riffle breakpoints were similar, but riffles lost 20% more of their total habitat than runs at the lowest discharge (71.2% versus 51.2%, respectively). Three pools showed linear ( $r^2 = 0.90$ ;  $r^2 = 0.77$ ,  $r^2 = 0.66$ ) relationships between area and discharge; however, no more than 25% of maximum wetted area was lost over the measured discharges. Two additional pools had no significant relationships between area and discharge and were best described by their means (Table 2-7). Barren Fork Creek's overall mean breakpoint was exceeded by 30% of the historic daily flows and ranged from 19% to 52% exceedance (Table 2-6). For August, daily flows exceeded the breakpoint only 4% of the time (range: 2% - 13%).

The Illinois River, the largest of the three streams, had the highest mean breakpoint ( $\pm 90\%$  CL) of  $10.36 \text{ m}^3/\text{s} \pm 5.09$  based on the average of upstream ( $11.05 \text{ m}^3/\text{s} \pm 4.09$ ) and downstream ( $7.67 \text{ m}^3/\text{s} \pm 2.27$ ) reaches (Table 2-8). Backwaters had the highest breakpoint ( $11.59 \text{ m}^3/\text{s} \pm 5.40$ ) and lost the greatest amount of total area (76.6%). Two of these habitats effectively lost their entire available habitat (99.3% and 99.9%). Two additional backwaters lost habitat linearly

( $r^2 = 0.96$ ,  $r^2 = 0.96$ ) and decreased by 60% and 49% at the mean breakpoint for this habitat type. Riffles experienced a significant loss of area at  $9.99 \text{ m}^3/\text{s} \pm 2.80 \text{ m}^3/\text{s}$  and lost 76.4% of their total available habitat. A single riffle had a linear relationship between area and discharge ( $r^2 = 0.96$ ), but never lost more than 27% of its total available habitat at the lowest discharge. This particular channel unit lost only 21% of the total area at the mean breakpoint for riffles. The persistence of this single riffle appeared to be an exception because two other riffles dried completely and a third lost 99.6% of its area. Only one run had a significant breakpoint ( $8.53 \text{ m}^3/\text{s} \pm 1.42$ ) and it lost only 18.2% of its total area at the lowest sampled discharge. All runs combined lost a little over half of their total area (51.2%). Four runs had linear relationships between area and discharge ( $r^2 = 0.60$ ;  $r^2 = 0.93$ ;  $r^2 = 0.92$ ;  $r^2 = 0.89$ ) and lost 20%, 26%, 38%, and 43% of available habitat, respectively, at the mean breakpoint for this habitat type. Interestingly, the Illinois River had two pools with significant breakpoints (mean of  $5.72 \text{ m}^3/\text{s} \pm 2.14$ ). One additional pool had a linear relationship ( $r^2 = 0.78$ ) and lost 27% of its total available habitat and another pool was best described by its mean area. Daily flow records in the Illinois River exceeded the mean breakpoint more than half of the time (56%) and ranged from 42% at the upper limit to 80% at the lower limit (Table 2-6). In August, flow exceedance at the breakpoint dropped to 22% at the mean (range: 10% - 58%).

#### Habitat use by fishes

A total of 6,474 fish representing 34 species was collected from 162 channel units (1,278 subsamples) during summer 2011 and 2012 (Table 2-9). The most abundant species (89% of catch) were: slender madtom *Noturus exilis* (1,258), Ozark minnow *Notropis nubilus* (914), western mosquitofish *Gambusia affinis* (900), central stoneroller *Compostoma anomalum* (894), orangethroat darter *Etheostoma spectabile* (634), cardinal shiner *Luxilus cardinalis* (601), and banded darter *Etheostoma zonale* (560). The other 27 species comprised the remaining 11% of the total catch (Table 2-9). A majority of the catch was adult fishes (79%).

Fish assemblages and densities differed by stream and sampling period. Species diversity was greatest in the Illinois River (29 species) compared to Barren Fork Creek (23 species) and Flint Creek (18 species); however, mean fish densities were greater in Barren Fork Creek (7.3 fish/m<sup>2</sup>) than the Illinois River (6.5 fish/m<sup>2</sup>) and Flint Creek (4.1 fish/m<sup>2</sup>; Table 2-10). The most abundant groups in shallow-water habitats of the Illinois River were minnows (e.g., central stoneroller, Ozark minnow, and cardinal shiner), darters (e.g., banded darter and orangethroat darter), and topminnows (Table 2-10). Samples from Barren Fork Creek had the only occurrences of suckermouth minnow *Phenacobius mirabilis* and yellow bullhead *Ameiurus natalis*; however, overall densities were dominated by slender madtom (3.0 fish/m<sup>2</sup>). The less diverse Flint Creek had the only occurrences of northern studfish *Fundulus catenatus* and southern redbelly dace *Phoxinus erthrogaster*. The most abundant species in Flint Creek were western mosquitofish, slender madtom, and orangethroat darter (Table 2-10). Day sampling resulted in higher diversity (32 species) than night sampling (23 species); however, overall fish densities were similar (6.1 fish/m<sup>2</sup> for day and 5.8 fish/m<sup>2</sup> for night). A few species varied markedly in overall densities between diel periods. Ozark minnows had the greatest change in densities between periods from a day density of 1.1 fish/m<sup>2</sup> to 0.3 fish/m<sup>2</sup> at night. Slender madtom densities increased at night by 0.8 fish/m<sup>2</sup>. Central stoneroller, sunburst darter *Etheostoma mihileze*, and 10 others also had slightly higher densities at night.

Ordination results for the CCA based on daytime collections (hereafter referred to as 'day CCA') in Flint Creek were significant for the first and all canonical axes, whereas results from night CCA indicated all axes combined were significant, but the first axis was not (Table 2-11). All axes combined explained 23% of the variation for day samples and 39% for nighttime samples. The first two CCA axes explained the most variation in species dispersion and represented the strongest environmental gradients. The first axis of the daytime CCA was positively associated with higher-velocity channel units (i.e. riffles and runs; Table 2-12).

Conversely, the same axis was negatively associated with slackwater habitats. The second day axis appeared to be a gradient from exposed habitats (i.e., pools, runs, backwaters) to habitats with more available cover (i.e., vegetated edgewaters and riffles). I used the night CCA to interpret overall patterns of species and channel-units associations, but I did not interpret individual axes because of lack of significance of the first axis.

Species in Flint Creek were distributed along these environmental gradients and associated with particular channel units (Figure 2-3). The day CCA for Flint Creek indicated benthic species (e.g., adult banded darter, adult banded sculpin, and both ages of slender madtom) and fluvial minnows (e.g., adult cardinal shiner, and young-of-year central stoneroller) were positively associated with riffles and runs along the first environmental axis (Figure 2-3). Adult sunburst darter and all ages of western mosquitofish were negatively associated with the first day axis. Channel-unit associations remained the same at night except for a shift to backwaters by young-of-year central stoneroller and young-of-year slender madtom. Distributed along the second day axis were benthic-fluvial species (e.g., banded darter and banded sculpin *Cottus carolinae*) and young-of-year cardinal shiner. Some species appeared to use all habitats during the day (e.g., adult Ozark minnow and adult orangethroat darter) but were more habitat specific at night. Diel shifts were observed for adult sunburst darters from backwater habitats to pools.

Ordination results for both periods in Barren Fork Creek were significant for the first and all canonical axes (Table 2-13). All axes combined for the night period explained nearly twice the variation (48%) as the day period (24%). The day CCA indicated the first environmental axis was positively related to the most shallow channel units: riffles, vegetated edgewaters and runs (Table 2-14). The same axis was negatively related to slackwater channel units: pools and backwaters. Conversely, the second axis appeared to be primarily related to the dominance of macrophyte cover in vegetated edgewaters. Unlike the day CCA, the first environmental axis of the CCA based on nighttime collections appeared to be associated with water velocity (i.e.,

positively related to slackwater channel units) whereas the second axis appeared to be related to water depth (i.e., positively related to pools).

In Barren Fork Creek, the first environmental axis of the day CCA was positively associated with benthic species (e.g., adult banded sculpin and adult banded darter), but negatively associated with slackwater species (e.g., all ages of western mosquitofish) and minnows (e.g., adult Ozark minnows and adult carmine shiners *Notropis percobromus*; Figure 2-3). At night, these species had similar distributions along the first environmental axis and maintained the same channel unit associations. The second axis of the day CCA was strongly associated with young-of-year fishes (e.g., Ozark minnow and cardinal shiner) and adult longear sunfish *Lepomis megalotis*; however, these species were not collected at night. The second axis of the night CCA was instead associated with adult Ozark minnow and adult sunburst darter. Adult sunburst darter shifted from using backwaters during the day to pools at night. ‘No fish’ samples were strongly associated with pools during the day and backwaters at night.

The Illinois River results for the first and all axes were significant for both periods (Table 2-15). All axes combined explained 37% of the total variation at night—more than twice that of day (15%). The environmental gradient along the first axis in the day CCA appeared to be related to water velocity and was positively correlated with riffles and runs (Table 2-16). The same axis was negatively correlated with slackwater habitats. Similarly, the night CCA indicated the first axis was positively related to higher water velocities. The second axis in the day CCA was positively correlated with more open habitats. However, the same axis in the night CCA was related to water depth.

Species groups from daytime samples in the Illinois River had strong habitat associations along the first axis including: benthic species, minnows, sunfish, and top minnows (Figure 2-3). As expected, many benthic species (e.g., adult banded darter and both age classes of slender



madtom) were associated with riffles and runs. Conversely, minnows (e.g., both age classes of Ozark minnow), adult longear sunfish, and both age classes of western mosquitofish were associated with backwater habitats. At night, these species maintained their distributions along the first environmental axis and their associated channel units; however, adult longear sunfish shifted from pools and backwaters during the day to faster moving water at night (e.g., runs). The only species and life stage strongly distributed along the second day axis was young-of-year carmine shiner and it was not observed at night. The second night axis revealed shifts in habitat use for adult cardinal shiner toward relatively deeper habitats (e.g., pools and runs). Additionally, adult orangethroat darter appeared to move out of swifter waters and into backwaters at night. During the day, 'no fish' samples were more associated with relatively deeper habitats, perhaps an indication of my gear bias to shallower water.

## **DISCUSSION**

My results suggest backwater and riffle habitat availability were the most sensitive to changing discharge conditions. Backwaters are shallow, depositional habitats that are often disconnected from the main flow at baseflows (Rabeni & Jacobson 1993). Peterson and Rabeni (2001b) found backwaters to be only 2 – 3% of total surface area in Ozark streams. These relatively small, off-channel habitats were more susceptible to desiccation than main-channel habitats even when depths were similar; so it is not surprising that their breakpoint were higher than other channel units (e.g., riffles). Flint Creek backwaters were the only exception (likely due to a low sample size). Riffles also lost considerable habitat and even dried completely at low flows, but tended to persist longer than backwaters. Expectedly, runs and pools maintained the most available habitat and were the least sensitive to low flows. Only those pools with extensive shallow edgewater (e.g., Illinois River) had significant breakpoints, but habitat loss remained small relative to total

area. Vegetated edgewater had shallow mean depths and long sloping margins suggesting high rate of habitat a loss. My findings were similar to others studies that found riffles to be sensitive habitats to lost discharge (Aadland 1993); however, backwaters were not one of the habitats considered using the wetted-perimeter approach. Common techniques to estimate minimum flows use field observations only at riffles to determine the relationship of available habitat to discharge (e.g., Annear 1984; Gippel & Stewardson 1998; Reinfelds 2004). My results suggest minimum flow estimates would benefit from consideration of multiple channel units, particularly those known to be important for macroinvertebrate food production (e.g., riffles, Whitledge & Rabeni 2000), juvenile rearing (this study), or spawning habitat (e.g., riffles, Brewer et al. 2006).

Despite extensive habitat loss at low flows, shallow-water channel units were important summer habitats for stream fishes. Backwaters were previously not considered essential habitats because they were 'off channel' and did not affect overall stream connectivity (Gippel & Stewardson 1998). However, I found backwaters were used by young-of-year of multiple species (e.g., young of year longear sunfish, orangethroat darter, and wedgespot shiner *Notropis greeni*). Backwaters provide rearing habitats for young-of-year fishes avoiding high flows (Moore & Gregory 1988) and escaping predation (Schlosser 1987). Vegetated edgewater also provide rearing habitat for young-of-year species (Lobb & Orth 1991; Rabeni & Jacobson 1993; Peterson & Rabeni 2001c). Macrophytes add habitat complexity that protects juvenile fishes from predation and provides more surface area for forage production (Persson & Eklov 1995). My data showed vegetated edgewater were used by young-of-year rock bass *Ambloplites rupestris*—an important sport fish. As anticipated, riffles were associated with benthic species including: multiple darters, slender madtom, and banded sculpin. High-velocity flows and larger substrates provided plenty of cover for these benthic fishes (Rabeni & Jacobson 1993). Interstitial spaces also provided abundant surface area for growth of periphyton and macroinvertebrates, making riffle important sites for primary and secondary production (Allan & Castillo 1995; Whitledge &

Rabeni 2000; Brewer et al. 2009). Runs and pools were deeper-water habitats that never dried completely. However, loss of shallow edgewaters around runs and pools could affect the young-of-year of species that occupy these habitats. For example, loss of shallow edgewaters forced fish into deeper habitats and increased mortality in age-0 salmonids, presumably because of increased individual susceptibility to predation (Riley et al. 2009). By understanding species habitat use, we can better predict the response of the fish assemblage to stream drying.

Documenting fish shifts among channel units also suggests transitioning between habitats is important to the ecology of these species. Stream fishes use specific habitats that vary with season and diel period (Matheney & Rabeni 1995; Brewer & Rabeni 2008). Diel shifts in habitat use are likely related to predator-prey interactions (Kwak et al. 1992) that affect species activity levels (e.g., drift, Brewer & Rabeni 2008). The most interesting diel shifts were associated with movements between backwaters and other channel units (e.g., adult sunburst darter and adult orangethroat darter). Adult fish likely transitioned between habitats to access better foraging habitats at night (Worischka et al. 2012). For example, invertebrate prey activity peaked in shallow-water habitats at night particularly when prey abundances were high (Culp & Scrimgeour 1993; Railsback et al. 2005). Increased nocturnal drift also increased the accumulation of macroinvertebrates in depositional habitats (e.g., pools, Schram et al. 1998) that were then exploited by resident species (David et al. 2007). Predation also influences fish habitat use (e.g., Schlosser 1988; Kadye & Booth 2014). At night, predation risk from terrestrial (Harvey & Nakomoto 2013) and aquatic (Johnson & Dropkin 1993) piscivores is reduced which may open previously restricted habitats to prey fishes. Juvenile species also moved to backwater habitats at night (e.g., central stoneroller and slender madtom). Juvenile fish reduced activity at night and moved into resting habitats to conserve energy (Bonneau & Scarnecchia 1998; Railsback et al. 2005). Fish activity levels may be related to other observed patterns. For example, densities of slender madtom increased during night sampling and may be an artifact of increased

susceptibility to my gear as they actively forage (Brewer & Rabeni 2008). Interestingly, variations in fish abundances were better explained by channel units at night than during the day. This suggests the overall fish assemblage exhibited more continuous movements between habitats during the day and more static habitat use at night, potentially when more diurnal species are resting (Reeb 2002). Regardless of the reasons, documenting fish movements between different channel units highlights the need to maintain habitat connectivity during summer low flows for all life stages.

Breakpoint estimates, in many cases, were higher than would be expected for a minimal flow designation because of some bias in the wetted-area approach. This is especially true during low-flow periods (e.g., August) where daily flows rarely approached the estimated breakpoint in each stream. Some bias is caused by variations in stream geomorphology, which resulted in higher breakpoint estimates for small streams and lower estimates for large streams (Annear & Conder 1984; Jowett 1997). Breakpoint estimates for Flint Creek, Barren Fork Creek, and the Illinois River increased predictably with stream order and upstream reaches had higher breakpoint estimates than downstream reaches. According to the river continuum concept, a stream becomes wider and deeper as it continues downstream (Vannote et al. 1980). Therefore, downstream reaches have greater storage capacity and are less sensitive to changes in discharge. Conversely, upstream reaches require a greater proportion of mean flow to maintain habitat and are more sensitive to changes in discharge (Jowett 1997). Channel braiding (e.g., Barren Fork Creek) can also introduce an upward bias because flow is dissipated laterally (Jowett 1997). Most likely, the upward bias was due to the breakpoint definition. The definition states: habitat-discharge breakpoints are the minimum flow needed to keep habitats full to the *base* of their banks (Jowett 1997). This level of optimal habitat is not naturally occurring during periods of low flow. Habitat losses are part of these systems' natural flow regimes and are a condition to which native species have evolved (Ward & Stanford 1983; Resh et al. 1988; Lake 2003). My breakpoint

estimates are best viewed as the ideal situation and minimum flow expectations would gain most by focusing between the lower confidence bands of these estimates and the discharges where available habitat is drastically reduced. Further, a follow-up analysis that examines the frequency of these occurrences, by season, would also be insightful.

Determining breakpoints by channel unit using the methods used in this study is an informative approach to minimum flow estimation because suggested flows are relevant to fish, can target particular habitats, can be seasonal adjusted, and outcomes can be continuously monitored. Traditional wetted-perimeter approaches (e.g., Gippel & Stewardson 1998; Annear & Conder 1984; Reinfelds et al. 2004) assume riffles are the most sensitive to discharge and minimum flows based on riffles are satisfactory for the entire system. Results from this study suggest riffle breakpoints do not adequately protect all habitats and estimated breakpoints from multiple habitats would be beneficial. Understanding habitat loss by channel unit allows seasonal adjustment of minimum flow values to better match each systems natural flow regime and account for ontogenetic shifts in habitat use (Poff et al. 1997). For example, higher flows are needed to maximize riffle availability for spawning catostomids (e.g., redhorse species, northern hogsucker, and white sucker) in spring (Curry & Spacie 1984) and to maintain adequate connectivity for downstream migrations in late fall (Grabowski & Isely 2006). Continuous sampling protocols can be implemented to monitor target species and improve minimum flow recommendations through time (Poff & Zimmerman 2010; Arthington 2012)—a necessary step when attempting to balance stakeholder and ecosystem needs in a changing climate (Vörösmarty et al. 2000; Arthington 2012). However, additional research to quantify the fitness consequences associated with habitat loss and to determine the threshold for functional connectivity between habitats would improve the effectiveness of any flow recommendations.

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TABLES AND FIGURES

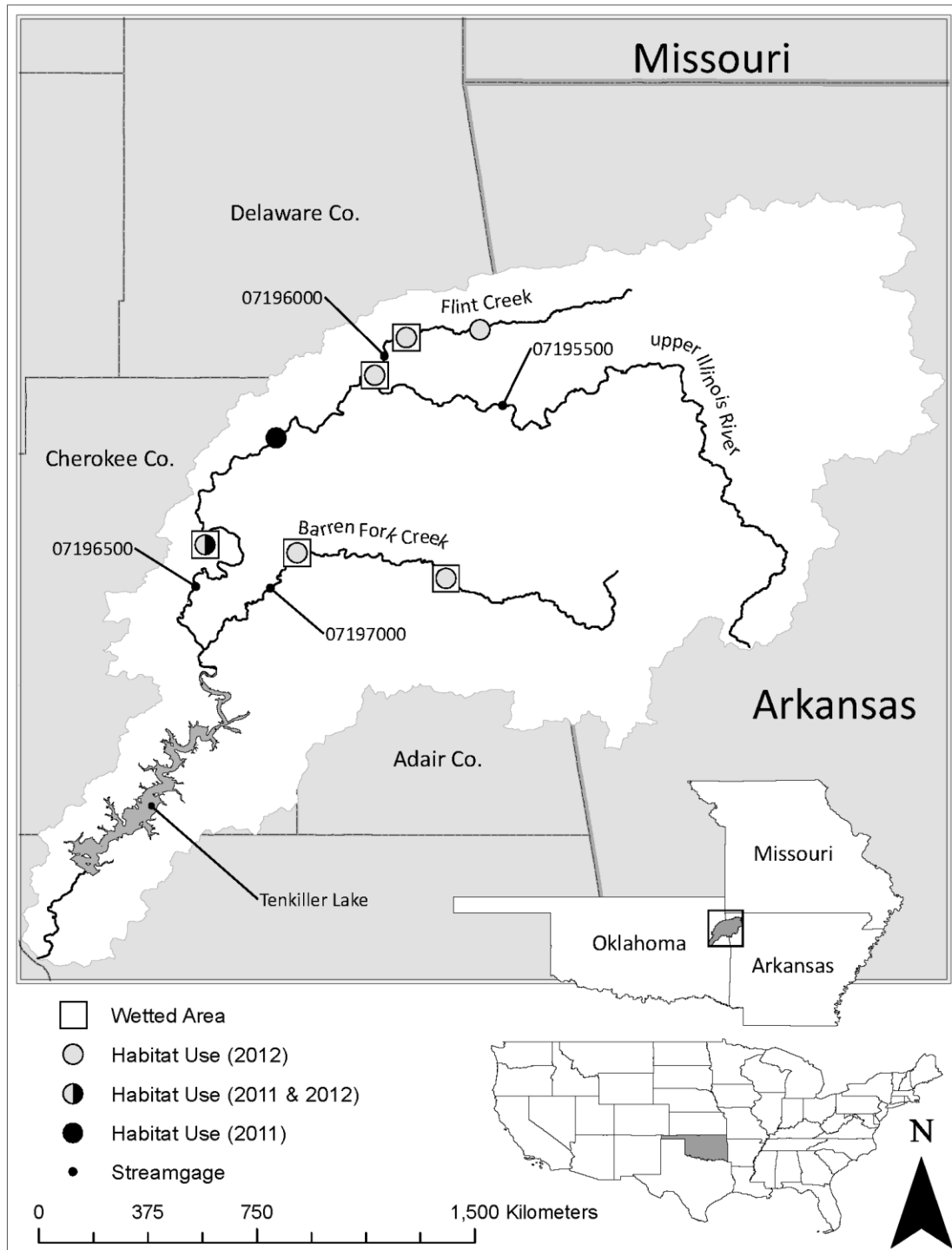


Figure 2-1.—Study sites in the Illinois River catchment for wetted-area and habitat-use sampling.



Table 2-1.—Number of wetted area-discharge samples taken over the range of flow-exceedance probabilities established from historical annual flow data using Indicators of Hydrologic Alteration.

Stream	95%	90%	80%	70%	60%	50%	40%	30%	20%	10%	Totals
Flint Creek	1	1	2	4	-	-	1	1	1	1	12
Barren Fork Creek	3	-	2	1	7	2	-	-	2	5	22
Illinois River	1	2	4	2	4	1	1	-	1	-	16
											50

Table 2-2.—Total lengths (TL) for young-of-year (YOY) fishes based on length-frequency histograms from catch data. Length-frequency histograms were compared to published length-at-age data to justify YOY-size thresholds. Length-at-age data of sister species were used where species-specific data was lacking. Sources of published data are also provided.

Species	Length freq. (TL)	Published (TL)	Sister species	Publication	
<i>Etheostoma zonale</i>	banded darter	30	30.5	-	Pflieger et al. (1975)
<i>Cottus carolinae</i>	banded sculpin	30	27.9	mottled sculpin	Pflieger et al. (1975)
<i>Notropis amblops</i>	bigeye chub	50	50.8	-	Pflieger et al. (1975)
<i>Notropis boops</i>	bigeye shiner	30	30.5	-	Pflieger et al. (1975)
<i>Moxostoma duquesni</i>	black redhorse	90	88.9	-	Pflieger et al. (1975)
<i>Fundulus olivaceus</i>	blackspotted topminnow	30	30.5	plains killifish	Pflieger et al. (1975)
<i>Lepomis macrochirus</i>	bluegill	35	38.1	-	Pflieger et al. (1975)
<i>Labidesthes sicculus</i>	brook silverside	60	63.5	-	Pflieger et al. (1975)
<i>Luxilus cardinalis</i>	cardinal shiner	40	43.2	duskystripe shiner	Pflieger et al. (1975)
<i>Notropis percobromus</i>	carmine shiner	35	40.6	roseymace shiner	Reed (1957)
<i>Campostoma anomalum</i>	central stoneroller	60	58.4	-	Pflieger et al. (1975)
<i>Ictalurus punctatus</i>	channel catfish	66	66	-	Pflieger et al. (1975)
<i>Semotilus atromaculatus</i>	creek chub	65	63.5	-	Pflieger et al. (1975)
<i>Erimystax x-punctatus</i>	gravel chub	60	63.5	-	Pflieger et al. (1975)
<i>Lepomis cyanellus</i>	green sunfish	40	43.2	-	Pflieger et al. (1975)
<i>Etheostoma blenniodes</i>	greenside darter	60	63.5	-	Pflieger et al. (1975)
<i>Percina caprodes</i>	logperch	70	73.7	-	Pflieger et al. (1975)
<i>Lepomis megalotis</i>	longear sunfish	35	33	-	Pflieger et al. (1975)
<i>Gambusia affinis</i>	western mosquitofish	15	variable	-	Pyke (2005)
<i>Hypentilium nigricans</i>	northern hogsucker	100	86.4	-	Pflieger et al. (1975)
<i>Fundulus catenatus</i>	northern studfish	40	42.3	-	Fisher (1981)
<i>Etheostoma spectabile</i>	orangethroat darter	30	27.9	-	Pflieger et al. (1975)
<i>Notropis nubilus</i>	Ozark minnow	35	30.5	-	Pflieger et al. (1975)

<i>Nocomis asper</i>	redspot chub	50	48.3	hornyhead chub	Pflieger et al. (1975)
<i>Ambloplites rupestris</i>	rock bass	60	40.6	-	Pflieger et al. (1975)
<i>Noturus exilis</i>	slender madtom	40	48.3	-	Pflieger et al. (1975)
<i>Micropterus dolomieu</i>	smallmouth bass	90	88.9	-	Pflieger et al. (1975)
<i>Phoxinus erthrogaster</i>	southern-redbelly dace	30	27.9	-	Pflieger et al. (1975)
<i>Phenacobius mirabilis</i>	suckermouth minnow	40	36	-	Etnier (1993)
<i>Etheostoma mihileze</i>	sunburst darter	35	35	stippled darter	Hotalling & Taber (1987)
<i>Lepomis gulosus</i>	warmouth	60	40.6	-	Pflieger et al. (1975)
<i>Notropis greeni</i>	wedgespot shiner	40	45.7	-	Pflieger et al. (1975)
<i>Catostomus commersonii</i>	white sucker	100	96.5	-	Pflieger et al. (1975)
<i>Amerius natalis</i>	yellow bullhead	175	177.8	-	Pflieger et al. (1975)

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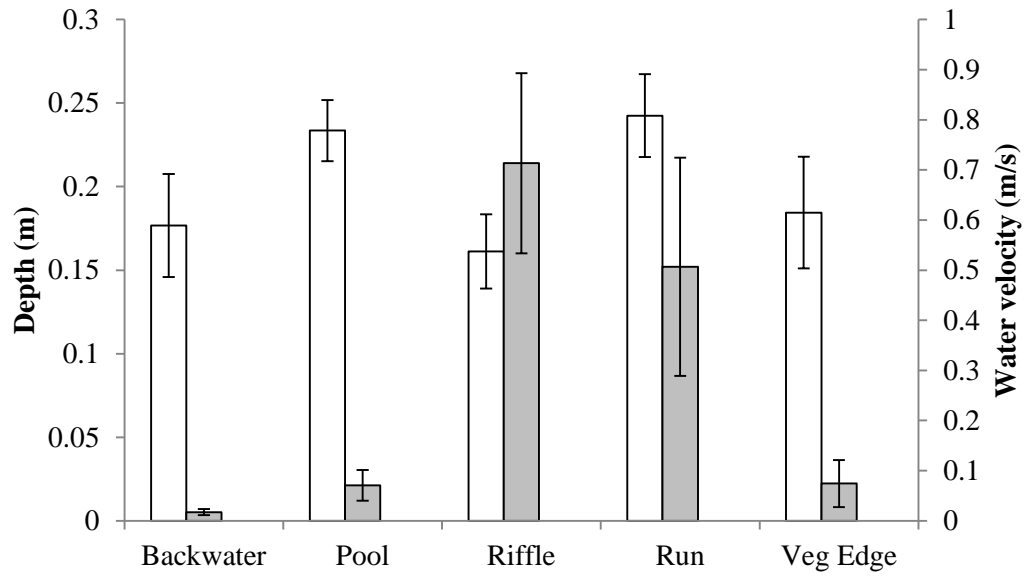


Figure 2-2.—Mean ( $\pm$  95% confidence limits) depth (m) and velocity (m/s) by channel-unit for all habitat-use samples. Hollow bars represent depth and grey bars represent velocity.

Table 2-3.—Mean substrate composition (percent coverage/m<sup>2</sup> ± 95% C.L.) by channel unit for habitat-use samples taken in summer 2012. Substrate-size classes (mm diameter) are: silt (0 - 0.4), sand (0.5 – 1.9), gravel (2 - 19), cobble (20 - 200), boulder (> 200), bedrock, and organic material (e.g., aquatic plants, algae, leaves, and woody debris).

Substrate	Backwater		Pool		Riffle		Run		Vegetated Edgewater	
	%	±	%	±	%	±	%	±	%	±
Organic	9.9	6.1	3.9	2.6	0.2	0.4	0.7	0.9	47.4	8.4
Silt	14.3	4.4	8.3	4.4	0.1	0.1	2.4	2.0	8.4	4.0
Sand	13.9	6.9	8.2	3.8	2.7	1.9	10.5	4.9	4.0	1.7
Gravel	45.7	6.8	52.2	7.0	50.7	8.8	53.9	9.0	30.1	8.2
Cobble	15.8	5.9	24.2	5.9	44.9	8.5	30.8	9.2	9.2	4.5
Boulder	0.4	0.5	2.2	3.0	1.4	1.6	1.0	1.2	0.7	1.2
Bedrock	0.0	0.0	1.0	1.8	0.0	0.0	0.7	1.3	0.1	0.2

Table 2-4.—Percent habitat lost (minimum area/maximum area) over the course of the study (2012 – 2013) by channel unit and stream.

Stream	Backwater	Riffle	Run	Pool
Flint Creek	92.7%	70.9%	33.5%	0.0%
Barren Fork Creek	89.6%	78.4%	50.2%	14.4%
Illinois River	84.4%	79.5%	35.1%	16.9%

Table 2-5.—Mean breakpoint estimates (90% C.L.) for wetted area-discharge relationships by channel unit. Breakpoints are discharges presented as m<sup>3</sup>/s and ft<sup>3</sup>/s. Significance of each breakpoint was tested with a Davie’s Test. The R<sup>2</sup> was provided where relationships were linear.

Flint Creek						
Channel Unit	Breakpoint ± 90% C.I.				Davie's Test	R <sup>2</sup>
	m <sup>3</sup> /s	±	ft <sup>3</sup> /s	±	<i>p</i> -value	
Backwater	1.23	0.38	43.44	13.42	<0.001	-
Pool	-	-	-	-	0.410	<i>mean</i>
Pool	-	-	-	-	0.640	<i>mean</i>
Riffle	3.35	1.23	118.30	43.44	<0.001	-
Riffle	1.25	0.31	44.14	10.95	<0.001	-
Riffle	3.39	0.96	119.72	33.90	<0.001	-
Run	2.42	1.11	85.46	39.20	<0.001	-
Run	2.06	0.79	72.75	27.90	<0.001	-
Run	1.15	0.39	40.61	13.77	<0.001	-
Stream Mean	2.12	0.74	74.92	26.08		

Table 2-6.—Mean breakpoints ( $\pm$  90% C.L.) by stream excluding pool channel units. Also included are annual flow-exceedance probabilities for the mean breakpoint and the upper and lower confidence intervals for each stream.

Stream	Breakpoint $\pm$ 90% C.L.				Exceedance Probability		
	m <sup>3</sup> /s	$\pm$	ft <sup>3</sup> /s	$\pm$	mean	upper	lower
Flint Creek	2.12	0.74	74.92	26.08	39%	19%	52%
Barren Fork Creek	7.39	4.12	260.89	145.52	30%	29%	56%
Illinois River	10.36	5.09	365.90	179.71	56%	42%	80%



Table 2-7.—Mean breakpoint estimates (90% C.L.) for wetted area-discharge relationships by channel unit. Breakpoints are discharges presented as m<sup>3</sup>/s and ft<sup>3</sup>/s. Significance of each breakpoint was tested with a Davie’s Test. The R<sup>2</sup> was provided where relationships were linear.

Barren Fork Creek								
	Channel Unit	Breakpoint ± 90% C.I.				Davie's Test	R <sup>2</sup>	
		m <sup>3</sup> /s	±	ft <sup>3</sup> /s	±	p-value		
Lower reach	Backwater	-	-	-	-	0.002	0.98	
	Backwater	6.12	1.40	216.13	49.44	<0.001	-	
	Pool	-	-	-	-	<0.001	0.90	
	Pool	-	-	-	-	0.001	0.77	
	Pool	-	-	-	-	0.143	<i>mean</i>	
	Riffle	8.61	2.64	304.06	93.23	<0.001	-	
	Riffle	7.67	4.25	270.86	150.09	<0.001	-	
	Riffle	2.75	1.85	97.12	65.33	0.037	-	
	Run	4.73	4.63	167.04	163.51	0.067	-	
	Run	7.68	3.07	271.22	108.42	<0.001	-	
	Run	4.83	1.89	170.57	66.75	<0.001	-	
		<i>Reach Mean</i>	<i>6.06</i>	<i>2.82</i>	<i>213.86</i>	<i>99.54</i>		
	Upper reach	Backwater	14.22	10.01	502.18	353.50	<0.001	-
Backwater		9.27	6.20	327.37	218.95	<0.001	-	
Pool		-	-	-	-	0.195	<i>mean</i>	
Pool		-	-	-	-	0.181	0.66	
Riffle		7.60	2.57	268.39	90.76	<0.001	-	
Riffle		2.89	1.71	102.06	60.39	0.007	-	
Riffle		4.66	1.75	164.57	61.80	<0.001	-	
Riffle		12.8	12.70	451.68	448.50	<0.001	-	
Run		5.06	1.49	178.69	52.62	<0.001	-	
Run		10.26	2.36	362.33	83.34	<0.001	-	
Run		9.06	7.41	319.95	261.68	0.003	-	
	<i>Reach Mean</i>	<i>8.42</i>	<i>5.13</i>	<i>297.47</i>	<i>181.28</i>			
	Stream Mean	7.39	4.12	260.89	145.52			

Table 2-8.—Mean breakpoint estimates (90% C.L.) for wetted area-discharge relationships by channel unit. Breakpoints are discharges presented as m<sup>3</sup>/s and ft<sup>3</sup>/s. Significance of each breakpoint was tested with a Davie’s Test. The R<sup>2</sup> was provided where relationships were linear.

Illinois River							
	Channel Unit	Breakpoint ± 90% C.I.				Davie's Test	R <sup>2</sup>
		m <sup>3</sup> /s	±	ft <sup>3</sup> /s	±	p-value	
Lower reach	Backwater	-	-	-	-	0.388	0.96
	Backwater	5.61	4.00	198.12	141.26	<0.001	-
	Pool	5.57	3.52	196.70	124.31	0.015	-
	Pool	-	-	-	-	0.862	<i>mean</i>
	Riffle	10.59	7.05	373.98	248.97	<0.001	-
	Riffle	11.49	7.98	405.77	281.81	<0.001	-
	Riffle	5.11	4.47	180.46	157.86	<0.001	-
	Run	-	-	-	-	0.419	0.92
	Run	-	-	-	-	0.001	0.89
	<i>Reach Mean</i>	<i>7.67</i>	<i>5.40</i>	<i>271.01</i>	<i>190.84</i>		
Upper reach	Backwater	-	-	-	-	0.686	0.96
	Backwater	17.40	7.21	614.48	254.62	0.001	-
	Backwater	11.76	7.37	415.30	260.27	0.006	-
	Pool	5.87	2.23	207.30	78.75	0.004	-
	Pool	-	-	-	-	0.403	0.78
	Riffle	8.91	2.87	314.66	101.35	<0.001	-
	Riffle	-	-	-	-	0.451	0.96
	Riffle	13.85	3.43	489.11	121.13	<0.001	-
	Run	8.53	1.42	301.23	50.15	<0.001	-
	Run	-	-	-	-	0.445	0.60
	Run	-	-	-	-	0.062	0.93
	<i>Reach Mean</i>	<i>11.05</i>	<i>4.09</i>	<i>390.35</i>	<i>144.38</i>		
	Stream Mean	10.36	5.09	365.90	179.71		

Table 2-9.—Total abundances of fishes sampled from Flint Creek and Barren Fork Creek (summer 2012) and the upper-Illinois River (summer 2011 and 2012). Species codes used for CCAs are the three-letter abbreviations.

Species	Species Code	Flint Creek	Barren Fork	Illinois River	Total
slender madtom	sld	249	746	263	1,258
Ozark minnow	ozk	55	221	638	914
western mosquitofish	mqr	319	211	370	900
central stoneroller	cst	107	226	561	894
orangethroat darter	otd	164	167	303	634
cardinal shiner	cds	87	152	362	601
banded darter	bnd	17	33	510	560
carmine shiner	cms	-	7	148	155
longear sunfish	lgs	20	2	74	96
banded sculpin	bsc	49	37	9	95
sunburst darter	sbd	26	47	16	89
channel catfish	cct	-	-	49	49
greenside darter	gsd	5	4	34	43
redspot chub	rdc	19	3	3	25
rock bass	rkb	14	4	6	24
bluegill	blg	5	2	12	19
wedgespot shiner	wdg	-	-	18	18
gravel chub	gvc	-	-	17	17
northern hogsucker	nhg	-	5	11	16
bigeye chub	bgc	-	1	11	12
northern studfish	nsf	12	-	-	12
blackspotted topminnow	btp	3	5	-	8
green sunfish	gsf	4	2	1	7
bigeye shiner	bgs	-	4	2	6
creek chub	crk	-	-	6	6
black redhorse	brh	-	-	3	3
smallmouth bass	smb	-	1	2	3
southern redbelly dace	srd	3	-	-	3
brook silverside	bks	-	-	2	2
logperch	lgp	-	-	1	1
suckermouth minnow	skm	-	1	-	1
warmouth	war	-	-	1	1
white sucker	wts	-	-	1	1
yellow bullhead	ybh	-	1	-	1
Total	34	1,158	1,882	3,434	6,474

Table 2-10.—Combined day and night total abundances (n) and means of fish densities (fish/m<sup>2</sup>) by stream and channel unit based on 2011 to 2012 habitat-use data. Empty (no fish) samples are not included. Abbreviated channel units are: BW=backwater and Veg=vegetated edgewater.

Species	n	Densities							
		Flint Creek	Barren Fork	Illinois River	BW	Pool	Riffle	Run	Veg
slender madtom	1,258	0.837	3.017	0.444	1.08	0.251	2.638	1.122	1.144
Ozark minnow	914	0.163	0.701	1.415	0.709	1.53	0.142	0.427	1.664
western mosquitofish	900	1.219	0.883	0.802	1.503	0.873	0.012	0.184	2.337
central stoneroller	894	0.395	0.807	1.185	0.905	0.708	0.916	0.721	1.124
orangethroat darter	634	0.551	0.737	0.503	0.317	0.535	0.465	0.337	1.278
cardinal shiner	601	0.277	0.526	0.634	0.561	0.447	0.442	0.665	0.41
banded darter	560	0.054	0.141	0.724	0.011	0.028	1.194	0.402	0.226
carmine shiner	155	-	0.026	0.224	0.048	0.29	0.015	0.126	0.055
longear sunfish	96	0.091	0.008	0.134	0.062	0.148	0.011	0.059	0.165
banded sculpin	95	0.165	0.136	0.014	0.016	0.022	0.285	0.042	0.072
sunburst darter	89	0.093	0.167	0.022	0.019	0.167	0.011	0.076	0.127
channel catfish	49	-	-	0.11	-	0.012	0.031	0.009	0.206
greenside darter	43	0.018	0.017	0.054	-	0.006	0.115	0.025	0.023
redspot chub	25	0.089	0.014	0.014	-	0.006	-	0.004	0.167
rock bass	24	0.056	0.021	0.016	0.007	0.003	0.005	0.01	0.122
bluegill	19	0.013	0.015	0.034	0.007	0.015	-	-	0.098
wedgespot shiner	18	-	-	0.036	0.013	0.027	0.008	-	0.039
gravel chub	17	-	-	0.022	-	0.009	0.023	0.015	-
northern hogsucker	16	-	0.022	0.023	0.014	0.016	0.009	0.034	0.006
bigeye chub	12	-	0.003	0.016	0.009	0.031	-	-	-
northern studfish	12	0.04	-	-	0.006	0.012	-	0.028	0.006
blackspotted topminnow	8	0.014	0.022	-	0.014	0.006	-	0.006	0.025

green sunfish	7	0.018	0.011	0.003	0.005	0.012	-	-	0.03
bigeye shiner	6	-	0.018	0.004	0.007	0.022	-	-	0.006
creek chub	6	-	-	0.015	-	-	-	0.002	0.032
black redhorse	3	-	-	0.005	-	0.008	0.004	-	-
smallmouth bass	3	-	0.005	0.005	-	0.006	0.006	-	0.006
southern redbelly dace	3	0.014	-	-	0.021	-	-	-	-
brook silverside	2	-	-	0.003	-	0.007	-	-	-
logperch	1	-	-	0.003	-	-	0.006	-	-
suckermouth minnow	1	-	0.005	-	-	0.006	-	-	-
warmouth	1	-	-	0.003	0.007	-	-	-	-
white sucker	1	-	-	0.001	-	0.003	-	-	-
yellow bullhead	1	-	0.008	-	0.011	-	-	-	-
<b>Total</b>	<b>6,474</b>	<b>4.106</b>	<b>7.311</b>	<b>6.47</b>	<b>5.351</b>	<b>5.207</b>	<b>6.337</b>	<b>4.294</b>	<b>9.368</b>

Table 2-11.—Flint Creek "Day" and "Night" CCA with channel units as explanatory variables. Eigenvalue for each ordination axis and the sum of all eigenvalues are provided along with total explained variance. "Day CCA" was significant for the first (F-ratio = 3.734,  $P < 0.05$ ) and all canonical axes (F-ratio = 1.906,  $P < 0.05$ ). "Night CCA" was not significant for the first axis (F-ratio = 2.164,  $P = 0.056$ ), but was significant for all canonical axes (F-ratio = 1.438,  $P < 0.05$ ).

Axes	Day				Night			
	1	2	3	4	1	2	3	4
Eigenvalues:	0.330	0.156	0.064	0.044	0.359	0.163	0.111	0.089
Species-environment correlations:	0.912	0.836	0.559	0.571	0.907	0.933	0.697	0.902
% variance of species data:	13.0	19.1	21.7	23.4	19.4	28.2	34.2	39.0
% variance of species-environment relation:	55.6	81.8	92.7	100	49.7	72.3	87.7	100
Sum of all eigenvalues:				2.539				1.853
Sum of all canonical eigenvalues:				0.593				0.723

Table 2-12.—Flint Creek correlation coefficients between channel units and CCA axes (i.e., environmental gradients) both day and night.

Environmental Variables	Day		Night	
	Axis 1	Axis 2	Axis 1	Axis 2
Channel Units:				
Riffle	0.7964	-0.5077	0.8516	0.0177
Run	0.2597	0.3252	0.2545	-0.3580
Pool	-0.1388	0.4362	-0.5878	-0.6336
Vegetated Edgewater	-0.7548	-0.6409	-0.3312	0.8852
Backwater	-0.1186	0.5463	-0.1310	0.0488

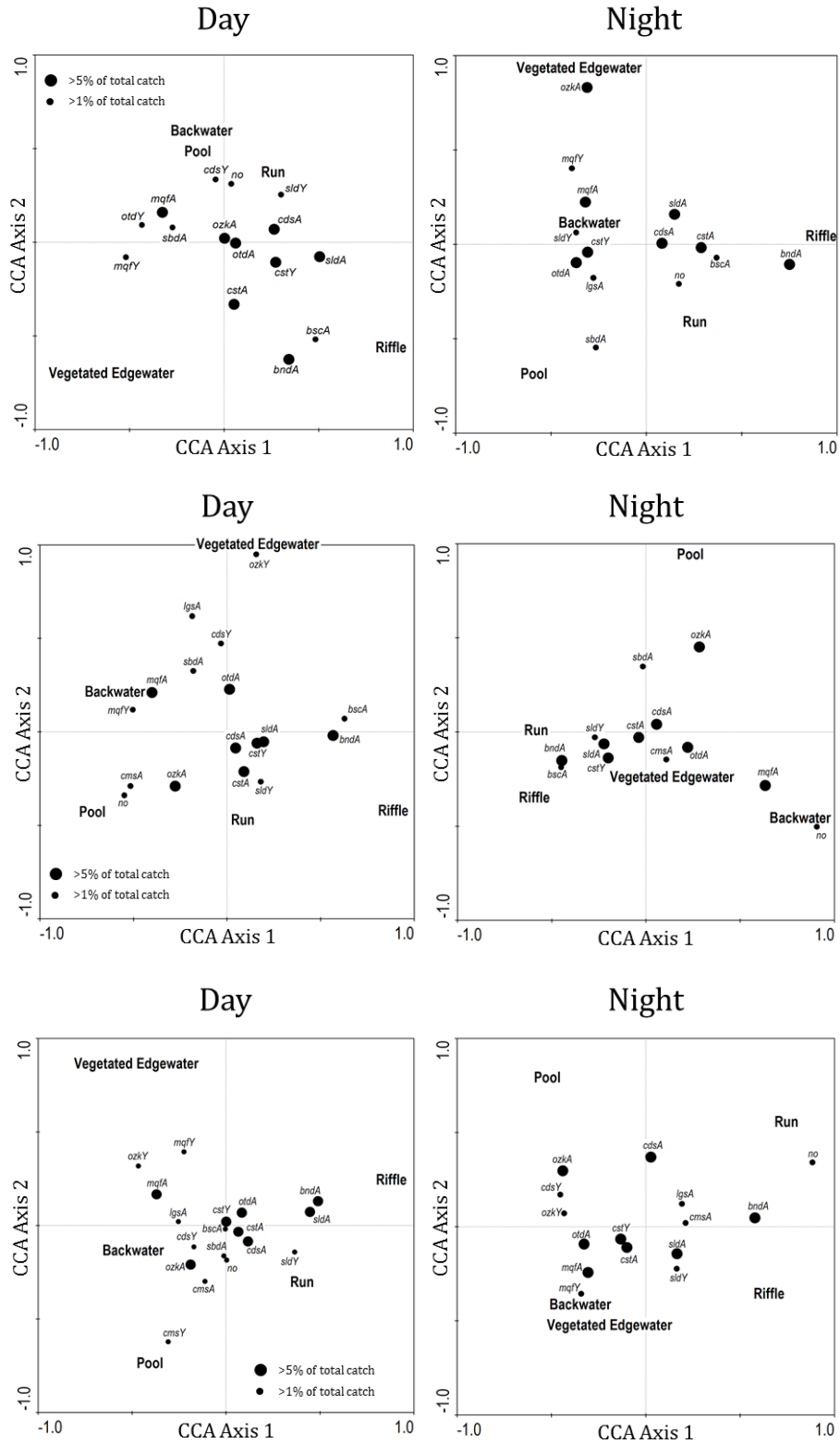


Figure 2-3.—Canonical Correspondence Analysis (CCA) plots for samples taken in the day and night periods from Flint Creek (upper panel), Barren Fork Creek (middle panel) and the Illinois River (lower panel). Species codes are listed in Table 2.9.



Table 2-13.—Barren Fork Creek "Day" and "Night" CCA with channel units as explanatory variables. Eigenvalue for each ordination axis and the sum of all eigenvalues are provided along with total explained variance. "Day CCA" was significant for the first (F-ratio = 2.634,  $P = < 0.05$ ) and all canonical axes (F-ratio = 1.914,  $P = < 0.05$ ). "Night CCA" was significant for the first (F-ratio = 3.287,  $P = < 0.05$ ) and all canonical axes (F-ratio = 2.267,  $P = < 0.05$ ).

Axes	Day				Night			
	1	2	3	4	1	2	3	4
Eigenvalues:	0.221	0.175	0.107	0.037	0.416	0.207	0.133	0.044
Species-environment correlations:	0.842	0.856	0.797	0.654	0.965	0.923	0.898	0.879
% variance of species data:	9.9	17.7	22.5	24.2	24.7	37.1	44.9	47.6
% variance of species-environment relation:	40.9	73.3	93.1	100	52.0	77.9	94.5	100
Sum of all eigenvalues:				2.234				1.683
Sum of all canonical eigenvalues:				0.540				0.800

Table 2-14.—Barren Fork Creek correlation coefficients between channel units and CCA axes (i.e., environmental gradients) both day and night.

Environmental Variables	Day		Night	
	Axis 1	Axis 2	Axis 1	Axis 2
Channel Units:				
Riffle	0.8025	-0.3678	-0.5061	-0.2962
Run	0.0270	-0.4177	-0.5139	-0.0178
Pool	-0.6439	-0.3747	0.2400	0.9463
Vegetated Edgewater	0.1483	0.9051	0.1371	-0.1867
Backwater	-0.4317	0.1662	0.7843	-0.4356

Table 2-15.—Illinois River "Day" and "Night" CCA with channel units as explanatory variables. Eigenvalue for each ordination axis and the sum of all eigenvalues are provided along with total explained variance. "Day CCA" was significant for the first (F-ratio = 4.244,  $P < 0.05$ ) and all canonical axes (F-ratio = 2.359,  $P < 0.05$ ). "Night CCA" was significant for the first (F-ratio = 2.148,  $P < 0.05$ ) and all canonical axes (F-ratio = 1.450,  $P < 0.05$ ).

Axes	Day				Night			
	1	2	3	4	1	2	3	4
Eigenvalues:	0.265	0.167	0.065	0.044	0.360	0.194	0.126	0.068
Species-environment correlations:	0.822	0.821	0.640	0.616	0.958	0.833	0.863	0.726
% variance of species data:	7.3	11.9	13.7	14.9	17.7	27.2	33.4	36.7
% variance of species-environment relation:	49.0	79.8	91.8	100	48.2	74.0	90.9	100
Sum of all eigenvalues:				3.635				2.038
Sum of all canonical eigenvalues:				0.541				0.750

Table 2-16.—Illinois River correlation coefficients between channel units and CCA axes (i.e., environmental gradients) both day and night.

Environmental variables	Day		Night	
	Axis 1	Axis 2	Axis 1	Axis 2
Channel Units:				
Riffle	0.8352	0.1661	0.5670	-0.3016
Run	0.3952	-0.1903	0.6764	0.5062
Pool	-0.4034	-0.6822	-0.5201	0.7424
Vegetated Edgewater	-0.4766	0.8465	-0.1919	-0.4689
Backwater	-0.2851	-0.1460	-0.3462	-0.3597

## CHAPTER III

### DEFINING FUNCTIONAL CONNECTIVITY BETWEEN CHANNEL UNITS OF AN OZARK STREAM

#### INTRODUCTION

Anthropogenic alterations to natural flow regimes threaten key ecosystem services (e.g., propagation of fish and wildlife) provided by functioning lotic ecosystems. Fish and other aquatic biota have evolved to a system's unique flow regime and are limited in their response to unnatural changes (Poff et al. 1997). For example, changes to natural flow regimes lead to reduces species fitness, invasion by exotic species, and extirpation of native species (Bunn & Arthington 2002; Poff & Zimmerman 2010). Unfortunately, flow alterations— a consequence of dams, channelization, water withdrawals, and levees to leverage freshwater resources— are a global problem (Postel et al. 2003). Increasing demands on these limited water resources coupled with future climate change put streams at risk of increased flow alteration (Vörösmarty et al. 2000; Poff et al. 2003). Lotic ecosystems are now considered one of the most threatened ecosystems on the planet in need of effective protections (Ricciardi & Rasmussen 1999; Dudgeon et al. 2006).

Environmental flow is a term used to describe the amount of water needed to sustain lotic

ecosystems and associated human benefits (Brisbane Declaration 2007) and is often established to protect or improve stream function from past, present, and future flow threats (Arthington 2012). One of the more commonly introduced environmental-flow protections is a minimum flow value (Richter 2010). Minimum flows maintain a desired level of discharge by limiting human abstractions and water diversions during periods of baseflow. The goal is to maintain stream discharge above a threshold where instream uses begin to decline (e.g., where fish experience reduced fitness; Annear & Conder 1984). Minimum flow values are an important consideration when protecting lotic ecosystems, but are only one aspect of a stream's natural flow regime. Other aspects of the flow regime (i.e., timing, duration, and the rate of change of flow) need to be addressed for full protections (Poff et al. 1997).

Minimum flow theory assumes a relationship between stream discharge, habitat availability, and fish fitness (Annear & Conder 1984; Jowett 1997). Discharge structures physical habitat in aquatic ecosystems (Jowett 1997; Bunn and Arthington 2002) and habitat is a major determinant of biotic composition (Schlosser 1982), so often the relationship between flow and habitat is used as a surrogate for biological response (Jowett 1997). Techniques to estimate minimum flows (e.g., hydraulic methods) use field observations to determine the relationship between discharge and stream geometry (Annear & Conder 1984; Gippel et al. 1998). For example, the wetted-area approach measures the relationship between stream discharge and channel unit area (see Chapter II). Hydraulic methods are commonly used to estimate instream-flow needs (Jowett 1997). Unfortunately, fish fitness and abundance do not have a linear response to available habitat (Jowett 1997). Supplemental approaches that estimate biological responses to flow would complement the existing frameworks.

Other weaknesses of the wetted-area method lie in assessing habitat connectivity (i.e., functional connectivity) and usable habitat for stream fishes (Annear et al. 2004). Wetted-area methods determine the discharge needed to maintain water across channel units but, depending on

the character of the stream, this level of discharge may be unsuitable for some species. For example, the discharge where flow across shallow-water riffles is maintained may be too shallow to allow passage of large-bodied species (e.g., adult smallmouth bass) effectively isolating them to a single habitat. This level of discharge may even be too shallow for riffle-dwelling fishes (e.g., darters) thereby forcing these small-bodied fishes out of optimal habitats. Reduced habitat connectivity and habitat loss confer a disadvantage to overall fish fitness (Fullerton et al. 2010). However, when and to what degree they negatively affect fish fitness are often unknown. To retain suitable habitat with minimum flows, it is beneficial to consider the necessity of movement within and between channel units to fish fitness.

Supplementing minimum flow estimates with direct measures of fish fitness would help make recommended flows biologically relevant and likely improve the success of stream conservation efforts. In particular, linking estimates of fish survival to hydraulic-based methods such as wetted area add biological support to flow recommendations (see Chapter II). Survival is a straight forward way to measure fish fitness, but connectivity and habitat use are also significant factors controlling species persistence through time (Fullerton et al. 2010). For example, isolated stream-fish communities of the Great Plains have reduced biodiversity (Perkin & Gido 2012) and are more susceptible to species loss (Fagan et al. 2002). Eventually, highly fragmented and dewatered Great Plains stream segments are dominated by benthic fishes (Perkin et al. 2014). To better protect fish from water withdrawals and altered flow dynamics for the long-term, it is important to incorporate analyses about stream connectivity. Only minimum flows that are biologically relevant, defensible, and capable of quantifying trade-offs can effectively protect fisheries (Annear & Conder 1984).

New technologies allow managers to supplement available-habitat measures with biologically-relevant and defensible data. Historically, obtaining real-time information on fish movement and survival was impossible because the technology simply was not available. Recent

advances in tag technology (e.g., radio-frequency identification RFID) now allow researchers to obtain these data. Study designs can incorporate mark-recapture experiments to supplement environmental-flow estimations even with small-bodied fishes (e.g., darters, madtoms, minnows, etc.). Assumptions of a relationship between available habitat and fish fitness can now be empirically supported for the stream-fish assemblage. The objective of this study was to use mark recapture to quantify the effect of low flows on the transition probability and daily survival of the following stream fishes: cardinal shiner *Luxilus cardinalis*, central stoneroller *Campostoma anomalum*, orangethroat darter *Etheostoma spectabile*, slender madtom *Noturus exilis*, and juvenile smallmouth bass *Micropterus dolomieu*.

## METHODS

### Study area

My mark-recapture experiments were conducted in Flint Creek, a 3<sup>rd</sup> order tributary of the Illinois River in northeastern Oklahoma, USA (Figure 3-1). The study catchment was located in the Ozark Highlands ecoregion and all streams in the catchment drain to the Arkansas River, the second largest tributary to the Mississippi River. Mean annual precipitation is 100 cm and mean annual temperature is 13 °C (Adamski et al. 1995). Flint Creek drains 285 km<sup>2</sup> (110 mi<sup>2</sup>) of oak-hickory forest, grasslands, and agriculture and residential areas (Adamski et al. 1995, Splinter et al. 2010). Lithology is mostly limestone, dolostone, sandstone, and shale (Fenneman 1938). Karst topography contributes to a number of spring upwellings (Whitledge et al. 2006). Flint Creek is characterized by low suspended sediment loads during baseflow conditions, cobble-gravel substrates, and has a mean annual discharge of 3.3 m<sup>3</sup>/s (118 f<sup>3</sup>/s). A 1-km section of stream containing a complex of riffle, run, pool, and backwater channel units was selected as my study reach (Figure 3-2).



### Study design

Mark-recapture methods were used to evaluate fish movements and survival in Flint Creek during low-flow conditions in 2011 (August – October) and 2012 (June – November). Fishes were implanted with passive-integrated-transponder (PIT) tags and ‘recaptured’ using stationary long-range receivers. The benefits of PIT tags are infinite life span, passive identification, small size, and unique codes (Prentice et al. 1990a, Gibbons & Andrews 2004, Cucherousset et al. 2005) that allow continuous track tagging through space and time. Similar mark-recapture approaches have successfully used PIT-tag technology to estimate population parameters for stream fishes (e.g., Smithson & Johnston 1999; Teixeira & Cortes 2007; Booth et al. 2013).

### Laboratory PIT-tag retention

Tag loss and the effects of tagging on fish growth and survival violate mark-recapture assumptions and bias parameter estimates that can handicap attempts to effectively manage fish population (Burnham et al. 1987). Bolland et al. (2009) recommended PIT tags be evaluated for each species prior to conducting field studies to avoid violating mark-recapture assumptions. This is an especially important consideration for juveniles and other small-bodied fishes where tagging may result in increased mortality or decreased growth (Prentice et al. 1990b).

To address these potential issues, I conducted laboratory experiments to assess the fitness consequences of intraperitoneal-placed PIT tags in six fish species (cardinal shiner, central stoneroller, greenside darter *Etheostoma blennioides*, orangethroat darter, slender madtom, and juvenile smallmouth bass; Table 3-1). Each fish ( $\geq 55$  mm TL) was injected with a 12-mm PIT tag into the peritoneum and held in a 38-L aquarium with a control fish of the same species for 38 to 45 days. Control fish were subject to the same steps as treatment fish, but were not tagged. Because of their larger size, smallmouth bass (mean TL  $>125$  mm) were injected with a 23-mm PIT tag and held in 2,400-L tanks in groups of ten (i.e., 5 treatment and 5 control fish) for 35

days. Growth rates (weight final - weight initial), survival (% alive), and tag retention (% retained) were calculated at the conclusion of the experiment. Significant differences ( $\alpha = 0.05$ ) in mean growth for treatment versus control fish by species were analyzed with a Welch's t-test in R (R Core Team 2012). Welch's t-test does not make the assumption of homogeneity of variance and uses a correction to adjust degrees of freedom (Field et al. 2012). Normality of growth data by species and treatment was tested with a Shapiro-Wilk test ( $\alpha = 0.05$ ). Any significant effect of PIT-tag placement found in this study was applied to mark-recapture models to make empirical estimates more accurate. If there were statistically significant differences, I also provide Pearson's correlation coefficient ( $r$ ) as an estimate of effect size (Field et al. 2012).

### *Fish sampling*

Mark-recapture efforts were focused on stream fishes that used shallow-water channel units. Total fish tagged was 438 (136 from year one and 302 from year two; Table 3-2). Fish were captured using multiple pass seining techniques. Though seining is not expected to have high efficiencies when sampling small fish (Pierce et al. 1990), it is generally associated with lower mortality than other standard gear types such as electrofishing (Dolan & Miranda 2004, Bonar et al. 2009). Sampling with this low-mortality gear reduced extraneous factors that may have influenced survival of tagged fishes. Seine methods varied depending on channel-unit characteristics (e.g., deep-slow velocity waters with a typical seine haul and swift waters by kick seine; Bonnar et al. 2009). Each channel unit was sampled until no more than 20 fish were captured (which is the number of fish it took 30 minutes to tag), so the maximum holding time of untagged fish was less than 1 hour. Additional steps were taken to reduce fish stress during sampling by limiting scale loss due to abrasion and exposure to air and direct sunlight. Each seine net was dipped in a slime-coat protectant (Vidalife: Western Chemical, Ferndale, WA) and captured fishes were immediately transferred to a flow-through cage placed in a shaded area of the stream. A minimum of 25 fish per species needed to be tagged in each year to qualify for

analysis. Orangethroat darter and slender madtom could only be analyzed for year two (i.e., 2012) because not enough individuals were captured in the first year to qualify for analysis (Table 3-2).

### Fish tagging

Fishes were PIT tagged using techniques that minimized stress during tagging. Fish were transferred from holding cages to an anesthetic bath (2.5 mL MS-222 stock solution per liter H<sub>2</sub>O) for five minutes or until fish began to lose equilibrium (Hauer and Lamberti 2006). Required concentration of anesthetic varied depending on water temperature due to metabolic rate increases with water temperature that sped absorption of MS-222. Fish were removed from the MS-222, measured for total length (1 mm), and tagged. All fish were handled with wet gauze dipped in a slime-coat protectant to limit loss of slime coat and scales. I also avoided double-handling fish because stress in fish is cumulative (Barton et al. 1986).

I used a 12 gauge needle to inject 12-mm HDX tags (Oregon RFID, Portland, OR) for fish less than 120 mm. The angle of the needle was 45° above the belly of the fish and inserted anteriorly into the peritoneum following methods from Prentice et al. (1990a). The 12-mm tags used in this study weighed approximately 0.1 g and have been used to successfully tag fish as small as 60 mm (Cucherousset et al. 2005). Fish larger than 120 mm were implanted with 23-mm HDX tags using a scalpel and finger pressure to gently place the tag in the peritoneum. Tagged fish were placed in a flow-through cage to allow recovery while being protected from predators. Cages were placed in a low-velocity area of the stream to prevent fish from being impinged in the downstream end. Fish were allowed to recover for a minimum of 24 hours (Hauer & Lamberti 2006). Fish were then checked for mortalities, normal respiration, and normal swimming prior to being release back into the channel unit of capture. Release involved lowering one side of the cage into the water column to allow fish to swim out voluntarily.

### Recaptures

I used a series of long-range receivers with multiple antennas as remote-monitoring stations to passively detect fish movements among channel units. Antennas were constructed with 10 to 12 gauge braided copper wire fashioned into loops within protective PVC conduit following the methods of Zydlewski et al. (2001). Antennas spanned the width of the stream at transition zones between different channel units and were wired to nearby battery-powered receivers in groups of three or four. Additionally, one receiver and single-antenna station was located at the downstream extent of the study reach for a total of eight antennas throughout the study reach (Figure 3-2). As PIT tags passed through the antenna loop, radio frequencies transmitted the unique identification code to the receiver where it was recorded along with a date and time stamp. Read range varied by the size of the antenna, the size of the PIT tag, and the angle of the fish passing through the antenna, but was between 7 to 15 cm for 12-mm tags and 12 to 24 cm for 23-mm tags (personal observation). Data from the receivers were downloaded on a weekly basis and batteries were changed out as needed in an effort to maintain continuous recaptures. Unfortunately, stream shade and overcast conditions interfered with solar recharge and reduced battery power to receivers for 11 days in August and 5 days in September 2011. Days with missing data were addressed in the analysis.

### Analysis

Fish capture histories were analyzed using the multistate model in Program MARK (White & Burnham 1999) through the R package RMark (Laake 2013; R Core Team 2012). Multistate models are an extension of the Cormack-Jolly-Seber CJS model (Brownie et al. 1993) that allow fish to transition between different states (Hodges & Magoulick 2011). Like the CJS model, multistate models produce estimates for apparent survival ( $S$ ) and recapture probability ( $p$ ), but add a third estimate for transition probability ( $\Psi$ ). Apparent survival ( $S$ ) is true survival times

the probability of an individual remaining in the study area, recapture probability ( $p$ ) is the likelihood of recapturing an individual given that it is alive, and transition probability ( $\Psi$ ) is the likelihood of an individual transitioning between states given that it is alive and in that state (Cooch & White 2006). Analysis using the RMark interface had the benefits of automated design matrices that input directly into Program MARK, easy model building, and concise model outputs (Cooch & White 2006).

Structure of multistate-capture histories is dependent on the time scale of interest (i.e., time steps and period length), the number of states, and number of tagged individuals. My study design collected continuous recaptures, which were compressed to daily time steps to match the required Program MARK input while still maintaining a fine temporal scale. Additionally, time periods for analyses (i.e., number of days) were reduced to 50 days in 2012 to match data from 2011. Days where antennas were not collecting recaptures were corrected for by varying the sampling intervals in RMark from single day to multiple day time steps (Laake 2013). Two states, the channel unit of original capture (i.e., A) and all other channel units (i.e., B), were designated in my multistate models to estimate movements between habitats. All analyses were split by species and year to restrict model size and accommodate changes to the study design (i.e., an increase in sampling effort, increase in number of tagged individuals, and longer recapture duration in year two). These steps reduced the overall model complexity (i.e., the number of estimable parameters).

Candidate models for apparent survival ( $S$ ), recapture probability ( $p$ ), and transition probability ( $\Psi$ ) were built based on the variables of time, discharge, effort, or as a constant. I hypothesized daily changes in stream discharge were a significant factor affecting stream-fish movements and survival; therefore, daily discharge was added as an occasion covariate to candidate models. Because the number of channel units sampled during marking or the number of antennas actively collecting recapture data influences the recapture probability, I also included

a model covariate for ‘effort’ in my candidate models. Effort was a simple count of the number of antennas collecting data on a given day or the number of channel units sampled during marking. An effort covariate also compensates for non-reading antennas due to low battery. Effort can only affect recapture probability and was not included for estimates of survival and transition probability. All candidate models ( $n = 27$ ) were compared to an invariable standard model (e.g., constant survival through time) and a time-varying model for survival, recapture, and transition probability (Table 3-3).

An information-theoretic approach was used for model selection from a set of candidate models using an adjusted Akaike’s information criterion QAICc (for small sample size). First, model fit was assessed for the most saturated (i.e., time dependent) model using the variance inflation factor (i.e.,  $\hat{c}$ ) goodness-of-fit approach in U-CARE (Choquet et al. 2009). This test measured for and corrected model overdispersion or ‘noise’ present in the data. Typically, overdispersion is caused by too many model parameters (Cooch & White 2006); however, my models had few covariables and any overdispersion was most likely caused by violating model assumptions (e.g., unequal recapture probability). If the most general model adequately fit the data, reasonable values for  $\hat{c}$  (e.g.,  $\leq 3.0$ ) were used to adjust model AICc scores for each model set. I used a model averaging approach to eliminate ambiguity when selecting the best model based on AIC ranks (Burnham and Anderson 2002). Weighted models (i.e.,  $\Delta\text{QAICc} < 10$ ) were averaged to obtain parameter estimates and 95% confidence intervals. Averaged models were then used to graphically display the relationship of fish survival and transition probability to discharge. Additionally, the overall importance of covariables was determined by summing support over all weighted models (Burnham & Anderson 2002).

There can be ambiguity in selecting the best model, especially when there is nearly equal support between top models; therefore, averaging across all models by weight was used to eliminate some of the uncertainty using AIC model selection. According to Anderson &

Burnham (2002), summing support over models is regarded as superior to making inferences concerning the relative importance of variables based only on the best model. To follow this approach there must be balance in the number of models that contain each variable of interest (Cooch & White 2006).

## RESULTS

### Laboratory PIT tag retention

Tag retention and survival of tagged fish was high for most species. Tag retention after 60 days was 100% for all species except orangethroat darter. A single tag was lost in the first 30 days and dropped retention to 88% for this species. Survival 24 hours after tagging was 100% for all species and remained high for the duration of the study (Table 3-4). After 30 days, treatment survival was 100% for cardinal shiner, central stoneroller, greenside darter, slender madtom, 96% for smallmouth bass, and 89% for orangethroat darter. After 60 days, treatment survival dropped to 88% for central stoneroller and to 56% for orangethroat darter, but remained the same for all other species (Table 3-4). Low survival by orangethroat darter may be due to improper diet because negative growth was seen for both treatment and control fishes (Figure 3-3). All mortalities were excluded from growth and retention analyses.

All species growth data fit the assumption of normality via the Shapiro-Wilk test ( $P < 0.05$ ). Results from the Welch's t-test indicated differences in growth between trials for orangethroat darter were not significant  $t_{(13,418)} = 1.41$ ,  $P = 0.18$ , so I combined the trials into one analysis.

Mean growth was not significantly different between treatment and control groups for all experimental fishes: cardinal shiners  $t_{(18,601)} = -0.35$ ,  $P = 0.73$ ; central stoneroller  $t_{(9,964)} = 0.29$ ,  $P = 0.78$ ; greenside darter  $t_{(2,331)} = 0.18$ ,  $P = 0.87$ ; orangethroat darter  $t_{(13,42)} = 1.41$ ,  $P = 0.18$ ;

slender madtom  $t_{(21.87)} = 1.21$ ,  $P = 0.24$ ; and smallmouth bass  $t_{(4.26)} = -0.41$ ,  $P = 0.71$  (Figure 3-3). My results suggest PIT tagging these fishes does not negatively affect growth and is therefore an appropriate approach for short-term, field-based studies.

#### Transition probability (movement) and survival

Significant violations in goodness-of-fit were observed in the global model for all species except year one cardinal shiner (Table 3-5). All violations were the result of the M.ITEC Test which checks for variations in recapture probability (the assumption is the probability is equal). However, estimates for  $\hat{c}$  overdispersion factors were all reasonable (e.g.,  $< 2.0$ ) and well below acceptable thresholds. Therefore, all model results ranked by QAICc scores were deemed acceptable.

Recapture probabilities were adequate for 2011 and 2012 for most of the species included in the analyses. All species in 2012 had mean recapture probabilities  $\geq 0.20$ . In 2011, all species except smallmouth bass met the threshold for meaningful inference of 0.20 (Hewitt et al. 2010). Mean recapture probability for smallmouth bass in 2011 was 0.16.

#### Cardinal shiner

Model results for cardinal shiner indicated discharge was related to apparent survival and transition probability in both years. For 2011 data, the best fit model suggested survival was influenced by discharge, recapture probability by effort, and transition probability was constant through time (Table 3-6). However, this model had only 39% of the Akaike weight and seven other models were also possible (though with less weight). Summed weights over model parameters indicated daily discharge had the most support (0.60) as an important predictor of survival (Table 3-7). Model-averaged results estimated high survival over daily time steps and was typically above 80% in 2011 and above 90% in 2012; however, much wider confidence bands are observed as discharge increases above  $1.7 \text{ m}^3/\text{s}$  in 2011 (Figure 3-4). A constant



transition probability had more support (0.75) than one influenced by discharge (0.25) in 2011. In 2012, only two models were weighted, each having support for discharge in predicting survival and transition probability (Table 3-6). Summed weights showed daily discharge (1.0) as the better predictor of both survival and transition probability (Table 3-7). Recapture probability had more support as a constant in my models (0.74). Transition probabilities in 2011 were consistently near 0.1, but transition probabilities in 2012 were variable and increased from 0.1 to approximately 0.4 above  $0.57 \text{ m}^3/\text{s}$  (Figure 3-5). Transition probabilities were near zero below  $0.43 \text{ m}^3/\text{s}$  suggesting movement became extremely limited at this flow.

### Central stoneroller

Model results for central stoneroller showed relatively constant survival and transition probabilities in both years. Each year had eight weighted models and no model had overwhelming support. In 2011, the best fit model suggested survival was constant, recapture probability was influence by effort, and transition probability was constant through time (Table 3-8). However, this model had only 37% of the Akaike weight. Mean probability of survival was approximately 0.7 and relatively stable with wider confidence limits as discharge increased (Figure 3-6). Mean probability of survival was higher in 2012 approximately 0.9. Summed weights over model parameters indicated discharge was more strongly related to survival, whereas transition probabilities for central stonerollers were relatively constant in 2011 (Table 3-9). Mean transition probability was relatively low in both sample years ( $\sim 0.25$  to  $0.30$ ) with much greater variation in confidence limits in 2011 (Figure 3-7). Recapture probability appeared to be relatively constant. Again in 2012, no model showed substantial support as being the best model (Table 3-8). Across all models, survival and transition probability were relatively constant over time (Table 3-9). There was a slight positive trend in transition probability above  $0.71 \text{ m}^3/\text{s}$ . Recapture probability was influenced by effort (i.e., number of antennas operating at any one time).

### Orangethroat darter

Seven models were weighted for orangethroat darter, but the best-fit model had over twice the support (58% Akaike weight) as the next model (20%, Table 3-10). The best-fit model for 2012 data supported discharge as an important predictor of survival and transition probability with constant recapture probability. Summed weights over model parameters indicated discharge (0.82) was the best predictor of survival (Table 3-11). Survival in 2012 was relatively high throughout the study (> 80%, Figure 3-8). Discharge was strongly related to transition probability of the darter (0.95). Transition probability increased above 0.57 m<sup>3</sup>/s suggesting more movement by the species when discharge was above this level (Figure 3-9). Recapture probability was relatively constant through 2012 (> 0.30).

### Slender madtom

Model results suggest discharge may be related to apparent survival, but there was full support for the relationship between discharge and transition probability (Table 3-12). Only two models were weighted for slender madtom. The best-fit model (61% Akaike weight) showed constant survival, recapture variable with effort, and transition probabilities variable by discharge. Similarly, the alternative weighted model showed the same relationship with recapture variability and transition probability but, did not support discharge as a predictor of survival (Table 3-12). Summed weights over model parameters support constant survival (0.61, Table 3-13). Survival in 2012 was consistently high across measured discharges (> 90%, Figure 3-8). Discharge, however, was fully supported as an important predictor of transition probability (Table 3-13). Transition probability and associated variation increased with increasing discharge, particularly at discharges above 0.57 m<sup>3</sup>/s (Figure 3-9). As expected, effort was the best predictor of recapture probability with decreasing recaptures when there were fewer active antennas.

### Smallmouth bass

Model results for smallmouth bass showed little relationship between daily discharge and survival or transition probability in both years (Table 3-14). Each year had eight weighted models and the best-fit model in each year suggested model parameters were constant. Constant parameter estimates had roughly 70% to 75% of the model weight. Mean survival was high in both sample years though, in 2011, parameter estimates were more variable when discharge exceeded approximately  $1.7 \text{ m}^3/\text{s}$  (Figure 3-10). Discharge influenced transition probability slightly more in 2011 (0.31) than in 2012 (0.26; Table 3-15), but the range of discharges encountered by fish during the 2011 study period was much greater. In 2012, transition probability appeared stable over time ( $\sim 0.3$ ; Figure 3-11), whereas there was a very slight linear increase in 2011 and much wider confidence limits above  $1.7 \text{ m}^3/\text{s}$  (Figure 3-11). Recapture probabilities were under 0.2 in 2011, but near 0.3 in 2012. Smallmouth bass recapture probabilities in 2011 were the lowest of all species and were below ideal thresholds for making valid inferences.

## DISCUSSION

My results suggest the summer transition probabilities of warmwater stream fish were influenced by discharge and may relate somewhat to the home range of these species. Variability in transition probabilities among species was probably related to both biotic (e.g., general species ecology and home range) and abiotic (e.g., habitat availability) factors. Fish habitat use and movements are driven by species traits such as feeding strategy and predator avoidance. Benthic insectivores (e.g., orangethroat darter and slender madtom) feed on macroinvertebrates in riffles (Phillips & Kilambi 1996; Gillette 2012) where densities of macroinvertebrates are often highest (Brown & Brussock 1991). Consequently, many darters appear to obtain resources from small areas and have estimated home ranges (i.e., average linear distance) of less than 50 m (typically not transitioning between adjacent channel units; Schwalb et al. 2011). Little work has been done

on home ranges of madtoms, but Fuselier (2012) found few transitions among riffles by Neosho madtom *Noturus placidus* suggesting smaller home ranges. Cardinal shiner also use riffles where they feed on downstream drift (Alexander & Perkin 2013) and take advantage of the shallow water to avoid larger-bodied predators (Matthews et al. 1994). Low transition probabilities observed in the first year (the higher flow year with no loss of connectivity) may suggest a smaller home range for this species.

Stream drying may also be a factor that drives transition probabilities of these species. At low flows, shallow-water channel units lose available habitat rapidly (see Chapter II). Species common in shallow-water channel units (e.g., cardinal shiner, orangethroat darter, and slender madtom) are more likely to move into adjacent habitats as available habitat shrinks. Other species commonly move out of riffles and into deeper, refuge habitats during drought conditions (e.g., Atlantic salmon parr *Salmo salar*, Armstrong et al. 1998; darters *Etheostoma spp.*, Roberts & Angermeier 2007; bigeye shiner *Notropis boops*, Hodges & Magoulick 2011). Small home ranges and increased predation risk probably limited these species dispersal beyond adjacent refuge habitat at low flows. Highland stoneroller *Camptostoma spadiceum* also move from riffles to pools during stream drying (Hodges & Magoulick 2011). However, stonerollers quickly reduce standing crop of algae (Power et al. 1985; Gelwick & Matthews 1992) and will continue foraging across channel units, even in the presence of predators (Harvey et al. 1988). Adult smallmouth bass use deeper channel units, have larger home ranges (~150 m), and are most active in summer (Todd & Rabeni 1989; Dauwalter & Fisher 2008; Brewer 2013). Smallmouth bass of the size tagged in this study (i.e., < 125 mm TL) move into riffles presumably to feed on increased densities of invertebrate prey (Fore et al. 2007). Runs and pools retain more area at low flows (see Chapter II), which, in combination with behavioral traits, likely contributed to consistent transition probabilities observed for these species in both years. Other species found in

deeper habitats during low flows maintain movements to monitor adjacent habitat conditions and find optimal foraging habitat as conditions change throughout summer (Gowan & Fausch 2002).

Daily fish survival was also related to discharge for many species, but cardinal shiner and orangethroat darter showed the strongest relationships. Increased discharge related to decreased cardinal shiner survival at a daily time step. Similar increases in mortality with discharge have been observed for stream fish and are hypothesized to be because of vulnerability to high-flow events during particular seasons (Grossman et al. 1998). Short-term increases in flow magnitude may have displaced fish downstream (Wesner 2011), increased exposure to predators (Bain et al. 1988; Magoulick & Kobza 2003), or caused physical stress from physical or chemical changes to the environment (Schlosser 1990; Pujolar et al. 2011). This may be why some of the species showed increased variability in daily survival estimates at higher flows; however, there are other factors related to high-flow events that are more likely.

The relationship between daily survival and discharge was most likely related to three factors: the temporal scale of the analyses, tag detection at higher flows, and emigration from the study area. Marsh-Matthews and Matthews (2010) also found that fine daily time steps were too short to detect the long-term implications of reduced flows on fish fitness. I anticipate that survival would actually increase with some increase in discharge (when within a 'normal' range of occurrences) when viewed over more coarse time steps (e.g., months or seasonally). For example, seasonal dewatering of stream reaches has led to mass mortality of Ouachita madtom *Noturus lachneri* (Gagen et al. 1998), loss of drying riffle habitats lowered survival of bigeye shiner (Hodges & Magoulick 2011), and stream drying appeared to reduce survival of smallmouth bass over the summer (Hafs et al. 2010). High flows influenced tag detections, which were apparent in the range of variability of parameter estimates at higher discharges (e.g., central stoneroller survival). High discharge can raise water levels above antenna detection range and cause environmental disturbance (e.g., vibrations) that reduce antenna efficiency. Discharge

can also cause permanent emigration from the study area (David & Cross 2002). Both of these factors cause issue with calculations using Cormack-Jolly-Seber models and can bias population estimates (Lebreton et al. 1992; Horton & Letcher 2008). Variable recapture probability has a marginal bias in survival estimates (Abadi et al. 2013), but permanent emigration is indistinguishable from death. High emigration rates likely biased survival estimates downward and potentially confounded my results of decreased survival with discharge. This appears to be the case for central stoneroller in 2011 that had had the lowest probability of survival (~70%), compared to all other species that had daily survival around 90%.

Survival is a straightforward way to measure fish fitness, but connectivity between habitats is also a significant factor controlling species persistence through time (Fullerton et al. 2010). My results suggest habitat connectivity is lost at approximately 0.43 – 0.57 m<sup>3</sup>/s for several small-bodied fishes in Flint Creek. Loss of connectivity due to stream drying reduces downstream drift, decreases water quality, and eliminates movements into and across shallow-water habitats (Matthews & Marsh-Matthews 2003; Arthington 2012). Lack of connectivity may affect the growth of drift-feeding fishes (e.g., smallmouth bass, Paragamian & Wiley 1987). Shallow, isolated habitats are also more susceptible to atmospheric conditions (e.g., extreme temperatures) that raise water temperatures, drop dissolved oxygen, and increase fish stress (Lake 2003). Fish that forage or shelter from predators in shallow waters get pushed into deeper habitats and become more concentrated as available habitats continue to shrink, increasing competition and predation (Magoulick & Kobza 2003). Though biota can quickly recolonize rewetted habitats (Lonzarich et al. 1998), the long-term effects to populations (e.g., fecundity, body condition, life expectancy) can vary by species (Marsh-Matthews & Matthews 2010) and may result in a decline in fish diversity (Perkin et al. 2014). The importance of connected habitats can also be inferred from both short (e.g., diel) and long-term (e.g., migrations) habitat shifts by fishes. As short foraging trips between channel units become impossible, so do long distance migrations between

tributaries necessary to escape entire dewatered reaches or for important life-history events (Jaeger et al. 2014). Isolating stream-fish communities can reduce fish biodiversity (e.g., Great Plains, Perkin & Gido 2012) and makes species more susceptible to extinction (Fagan et al. 2002).

The power of my approach comes from identifying transition probabilities so managers know at what discharge fish may become restricted to certain habitats when establishing environmental-flow protections. I found many of the species in this study had decreased transitions at approximately  $0.57 \text{ m}^3/\text{s}$  ( $20 \text{ ft}^3/\text{s}$ ). A weakness of the wetted-area method in determining minimum flow values is that it does not relate habitat connectivity (i.e., functional connectivity) to usable habitat for stream fishes (Annear et al. 2004). Supplementing minimum streamflow estimates with measures of connectivity help make recommended flows biologically relevant and likely improves the success of stream conservation efforts. Additionally, consideration of stream connectivity at multiple spatial scales (e.g., channel unit and tributary to tributary) would be beneficial for the long-term protection of fishes from flow alterations and climate change. Further, incorporating bioenergetics approaches into future studies during extreme low flows would provide insight into the effects of habitat isolation on fish growth and persistence. Linking a population model to flow scenarios is an alternative approach that would be beneficial to improving our understanding of the effects of flow alteration on fishes.

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TABLES AND FIGURES

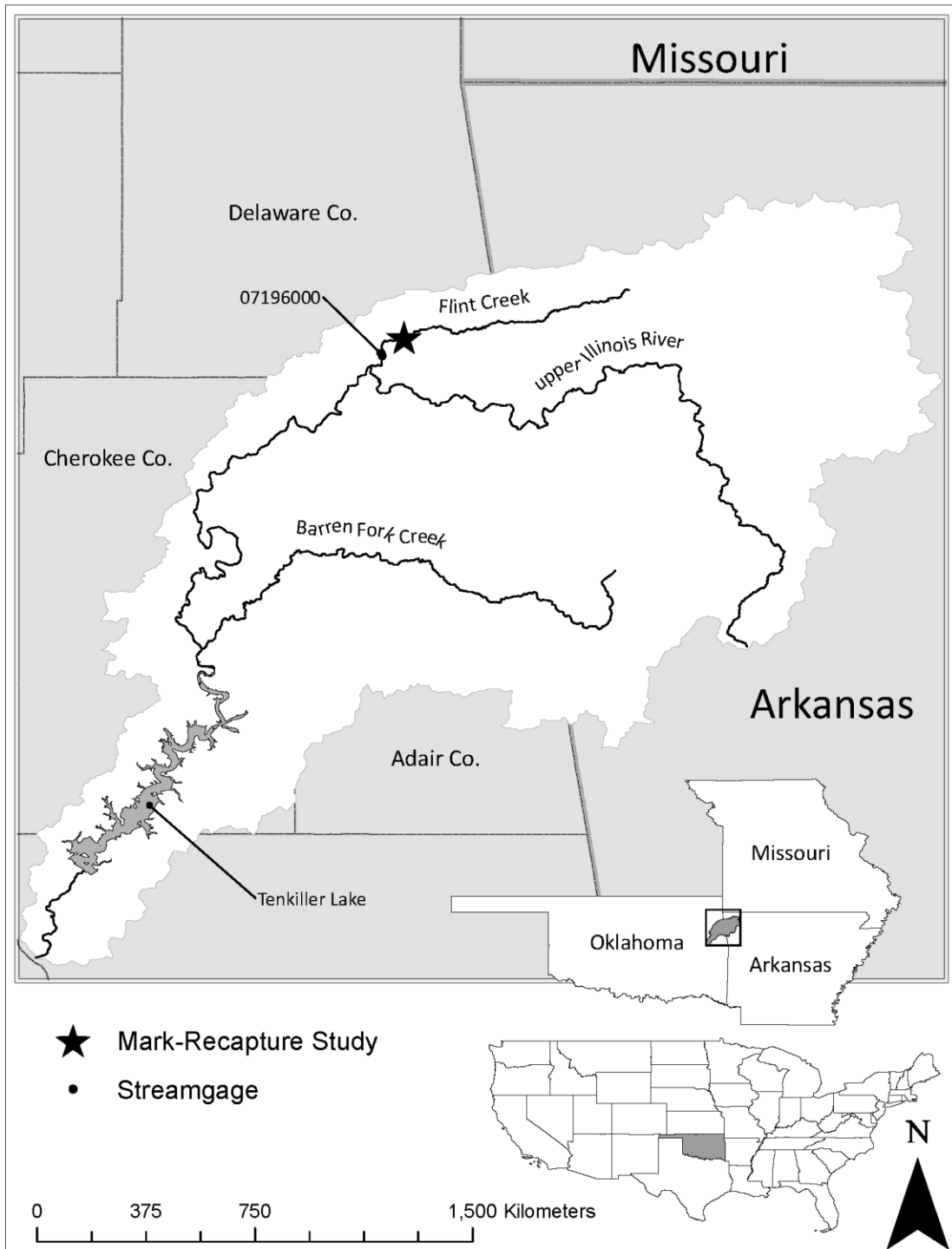


Figure 3-1.—Study site in the Illinois River catchment for mark-recapture experiments.

Table 3-1.—Mean total length (TL; 1.0 mm) and total weight (WT; 1.0 g) of fish PIT tagged in tag retention trials. Standard deviations (SD) of means are provided. PIT tag length and diameter (1.0 mm) for each species group are also indicated.

Species		n	TL ± SD	WT ± SD	Tag size
<i>Luxilus cardinalis</i>	cardinal shiner	24	91.1 ± 9.6	6.8 ± 2.3	12.0 x 2.12
<i>Campostoma anomalum</i>	central stoneroller	16	94.3 ± 18.2	7.3 ± 4.5	12.0 x 2.12
<i>Noturus exilis</i>	slender madtom	26	76.8 ± 10.0	3.4 ± 1.4	12.0 x 2.12
<i>Etheostoma spectabile</i>	orangethroat darter	14	60.2 ± 5.7	2.1 ± 0.7	12.0 x 2.12
<i>Etheostoma blennioides</i>	greenside darter	6	94.8 ± 9.3	6.5 ± 1.7	12.0 x 2.12
<i>Micropterus dolomieu</i>	smallmouth bass	50	148.0 ± 10.8	31.0 ± 7.2	23.0 x 3.65

Table 3-2.—Mean total length (1 mm) of species PIT tagged in 2011 and 2012 for my multi-state model analyses. The minimum (Min), maximum (Max) and standard deviation (SD) of total lengths are also provided.

Year	Species	n	Mean	Min	Max	SD
2011	cardinal shiner	27	102.7	78	112	9.7
	central stoneroller	46	102.8	81	133	11.4
	smallmouth bass	63	72.1	59	143	13.8
2012	cardinal shiner	112	92.2	68	150	13.8
	central stoneroller	58	98.6	72	137	14.4
	orangethroat darter	48	62.3	55	71	4.0
	slender madtom	44	74.4	62	91	8.0
	smallmouth bass	40	73.5	53	132	24.4
Total		438				

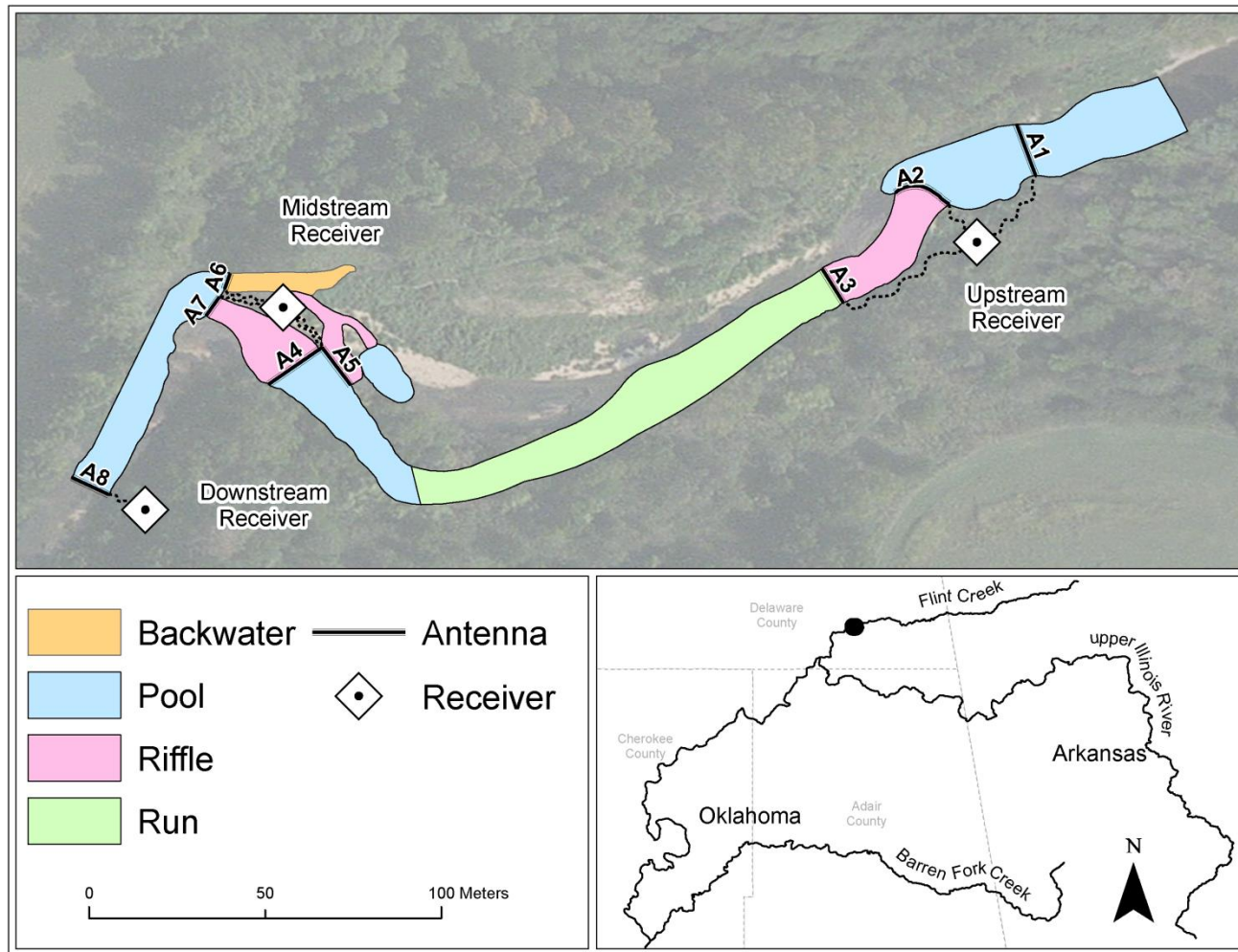


Figure 3-2.—PIT-tag antenna locations across multiple channel units in Flint Creek.

Table 3-3.—Models and parameter descriptions used in multistate mark-recapture analyses.

#	Model	Parameter Descriptions:		
		Survival (S)	Recapture (p)	Transition (Psi)
1	S(~1)p(~1)Psi(~1)		<b>All Constant</b>	
2	S(~1)p(~1)Psi(~Q)	Constant.	Constant.	Variable with discharge.
3	S(~1)p(~1)Psi(~time)	Constant.	Constant.	Variable by day.
4	S(~1)p(~effort)Psi(~1)	Constant.	Variable by number of active antennas.	Constant.
5	S(~1)p(~effort)Psi(~Q)	Constant.	Variable by number of active antennas.	Variable with discharge.
6	S(~1)p(~effort)Psi(~time)	Constant.	Variable by number of active antennas.	Variable by day.
7	S(~1)p(~time)Psi(~1)	Constant.	Variable by day.	Constant.
8	S(~1)p(~time)Psi(~Q)	Constant.	Variable by day.	Variable with discharge.
9	S(~1)p(~time)Psi(~time)	Constant.	Variable by day.	Variable by day.
10	S(~Q)p(~1)Psi(~1)	Variable with discharge.	Constant.	Constant.
11	S(~Q)p(~1)Psi(~Q)	Variable with discharge.	Constant.	Variable with discharge.
12	S(~Q)p(~1)Psi(~time)	Variable with discharge.	Constant.	Variable by day.
13	S(~Q)p(~effort)Psi(~1)	Variable with discharge.	Variable by number of active antennas.	Constant.
14	S(~Q)p(~effort)Psi(~Q)	Variable with discharge.	Variable by number of active antennas.	Variable with discharge.
15	S(~Q)p(~effort)Psi(~time)	Variable with discharge.	Variable by number of active antennas.	Variable by day.
16	S(~Q)p(~time)Psi(~1)	Variable with discharge.	Variable by day.	Constant.
17	S(~Q)p(~time)Psi(~Q)	Variable with discharge.	Variable by day.	Variable with discharge.
18	S(~Q)p(~time)Psi(~time)	Variable with discharge.	Variable by day.	Variable by day.
19	S(~time)p(~1)Psi(~1)	Variable by day.	Constant.	Constant.
20	S(~time)p(~1)Psi(~Q)	Variable by day.	Constant.	Variable with discharge.
21	S(~time)p(~1)Psi(~time)	Variable by day.	Constant.	Variable by day.
22	S(~time)p(~effort)Psi(~1)	Variable by day.	Variable by number of active antennas.	Constant.
23	S(~time)p(~effort)Psi(~Q)	Variable by day.	Variable by number of active antennas.	Variable with discharge.
24	S(~time)p(~effort)Psi(~time)	Variable by day.	Variable by number of active antennas.	Variable by day.
25	S(~time)p(~time)Psi(~1)	Variable by day.	Variable by day.	Constant.
26	S(~time)p(~time)Psi(~Q)	Variable by day.	Variable by day.	Variable with discharge.
27	S(~time)p(~time)Psi(~time)		<b>Global Model</b>	

Table 3-4.—Survival (S) and PIT-tag retention (R) (both expressed as percentages) for treatment fishes after 30, 60, and 90 days. Growth (G = weight<sub>final</sub> – weight<sub>initial</sub>) is represented as the mean weight gain (+/-) after 35 - 43 days.

Species	n	30 days		60 days		90 days		G	
		S	R	S	R	S	R		
<i>L. cardinalis</i>	cardinal shiner	12	100	100	100	100	100	+	
<i>C. anomalum</i>	central stoneroller	8	100	100	88	100	88	+	
<i>N. exilis</i>	slender madtom	13	100	100	100	100	n/a	n/a	-
<i>E. spectabile</i>	orangethroat darter	9	89	88	56	88	n/a	n/a	-
<i>E. blennioides</i>	greenside darter	3	100	100	100	100	n/a	n/a	-
<i>M. dolomieu</i>	smallmouth bass	25	96	100	96	100	n/a	n/a	+

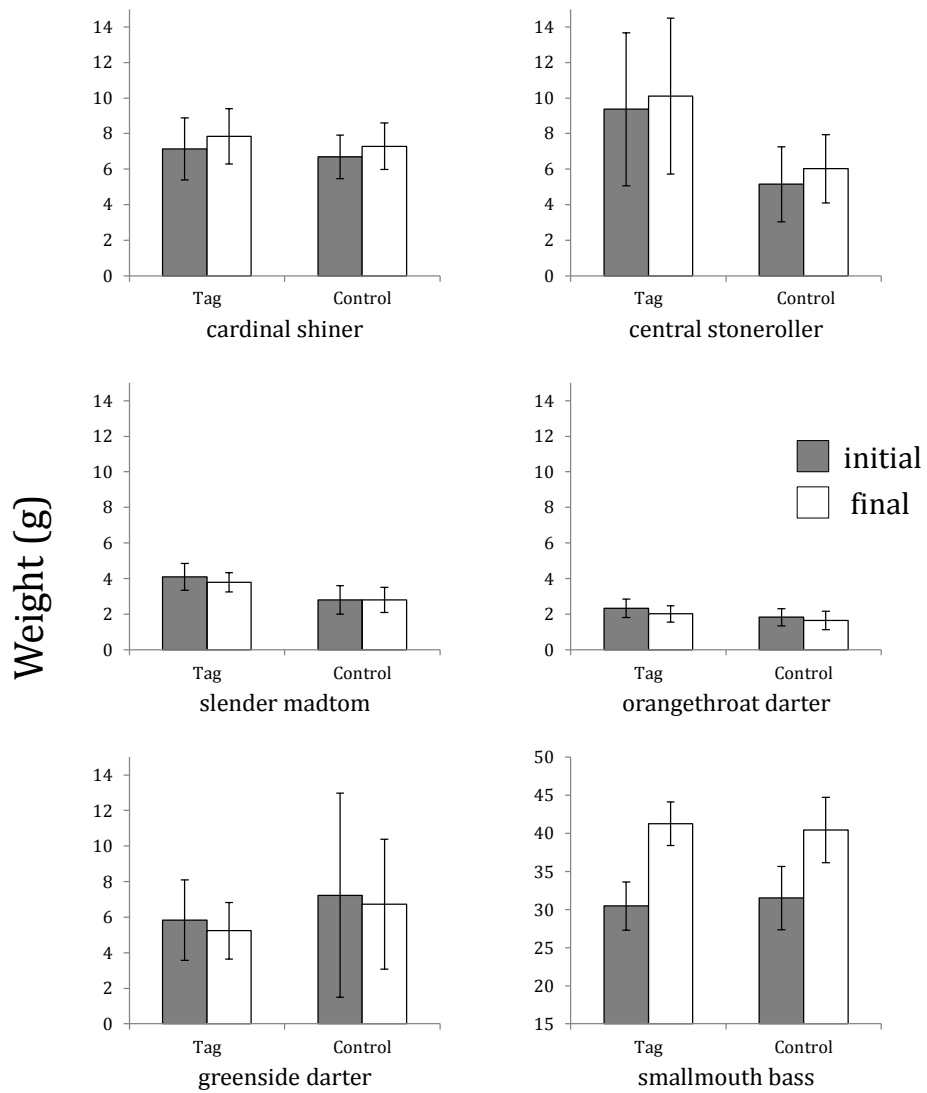


Figure 3-3.—Mean growth (mean weight  $\pm$  95% confidence limits) of treatment and control fishes during PIT-tag retention experiments. Growth was not significantly different between PIT-tag treatment and control groups for any species.



Table 3-5. —Goodness-of-fit testing and estimate of the variance inflation factor (i.e.,  $\hat{c}$ ) for each global model (i.e., time varying model) used in multistate mark-recapture analyses.

Year	Species	$X^2$	df	Estimate of $\hat{c}$	U-CARE significant violations
2011	cardinal shiner	14.806	9	1.65	-
2011	central stoneroller	12.754	10	1.28	TEST M.ITEC
2011	smallmouth bass	42.367	22	1.93	TEST M.ITEC
2012	cardinal shiner	215.115	111	1.94	TEST M.ITEC
2012	central stoneroller	156.98	79	1.99	TEST M.ITEC
2012	orangethroat darter	71.186	41	1.74	TEST M.ITEC
2012	slender madtom	33.896	33	1.03	TEST M.ITEC
2012	smallmouth bass	95.769	64	1.5	TEST M.ITEC

Note: TEST M.ITEC tests the  $H_0$ : there is no difference in the probabilities of being re-encountered in the different states at  $i+1$  between the animals in the same state at occasion  $i$  whether encountered or not encountered at these data, conditional on presence at both occasions (Choquet et al. 2005).

Table 3-6.—Top model results for 2011 and 2012 data on cardinal shiner data. Only those models with Delta QAICc < 10 are weighted and included in the total candidate models.

Cardinal Shiner								
Year	#	Model	npar	QAICc	DeltaQAICc	Weight	QDeviance	$\hat{c}$
2011	13	S(~Q)p(~effort)Psi(~1)	5	251.79	0	0.39	187.44	1.65
	4	S(~1)p(~effort)Psi(~1)	4	253.53	1.74	0.16	191.52	1.65
	1	S(~1)p(~1)Psi(~1)	3	253.95	2.16	0.13	194.2	1.65
	14	S(~Q)p(~effort)Psi(~Q)	6	254.01	2.22	0.13	187.25	1.65
	10	S(~Q)p(~1)Psi(~1)	4	255.44	3.65	0.06	193.42	1.65
	5	S(~1)p(~effort)Psi(~Q)	5	255.68	3.89	0.06	191.33	1.65
	2	S(~1)p(~1)Psi(~Q)	4	256.03	4.23	0.05	194.01	1.65
	11	S(~Q)p(~1)Psi(~Q)	5	257.59	5.8	0.02	193.24	1.65
2012	11	S(~Q)p(~1)Psi(~Q)	5	1053.4	0	0.74	837.64	1.94
	14	S(~Q)p(~effort)Psi(~Q)	6	1055.5	2.05	0.26	837.64	1.94

Table 3-7.—Summed weights of parameter covariates across all weighted models for cardinal shiner.

Cardinal Shiner				
	covariates	S	p	Psi
2011	~1	0.40	0.26	0.75
	time	0	0	0
	Q	0.60	-	0.25
	effort	-	0.74	-
	covariates	S	p	Psi
2012	~1	0	0.74	0
	time	0	0	0
	Q	1.0	-	1.0
	effort	-	0.26	-

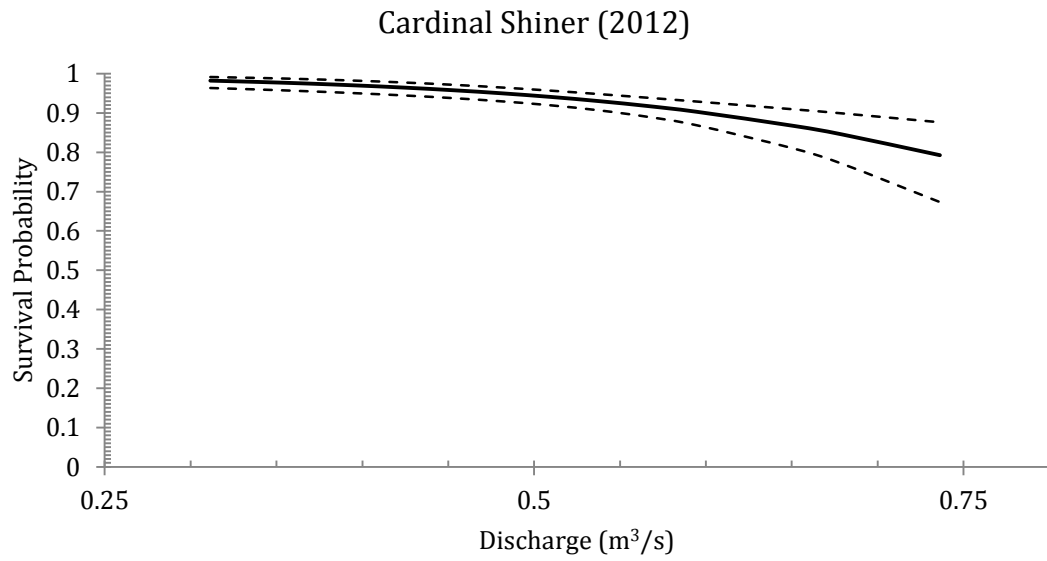
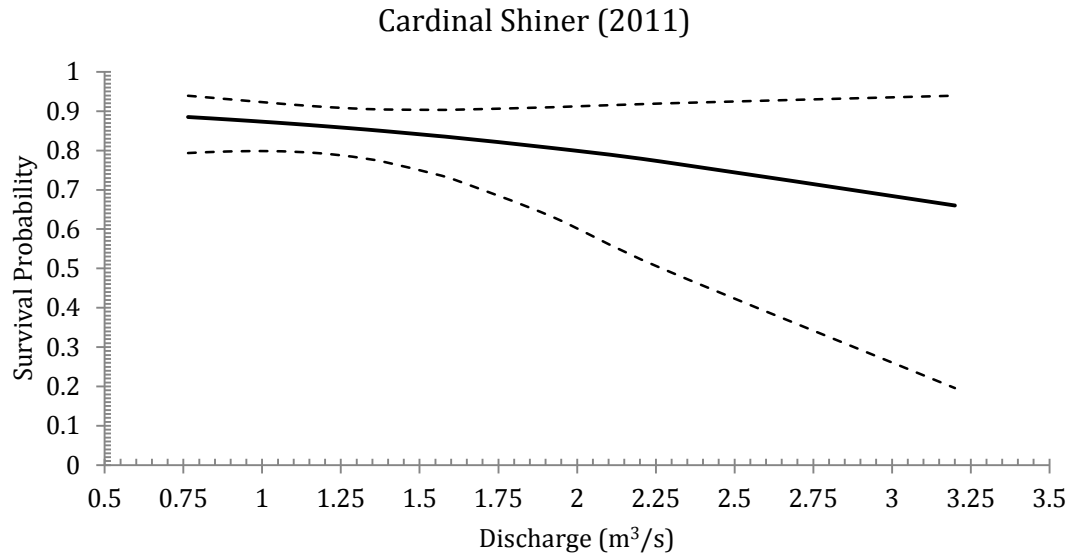
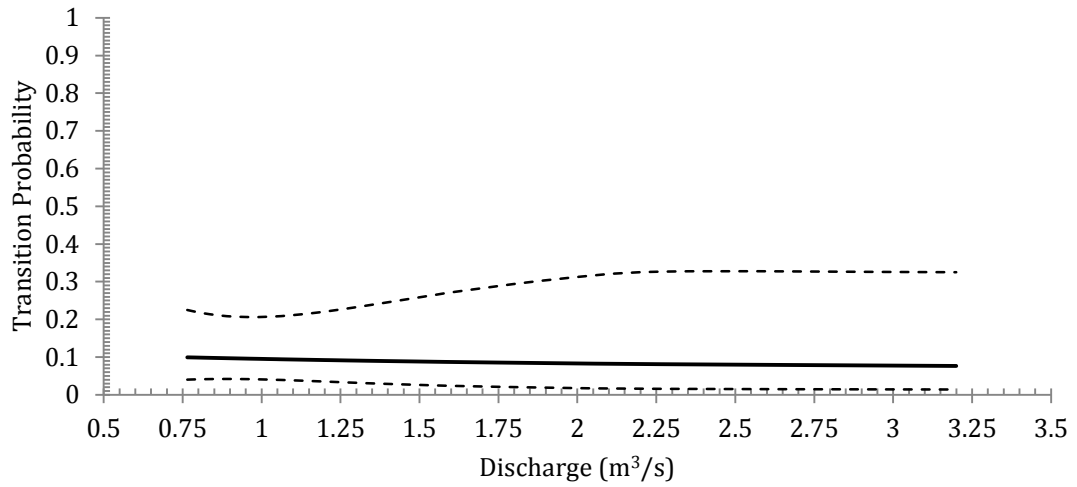


Figure 3-4.—Survival probability of cardinal shiner related to discharge during 2011 (upper panel) and 2012 (lower panel).

### Cardinal Shiner (2011)



### Cardinal Shiner (2012)

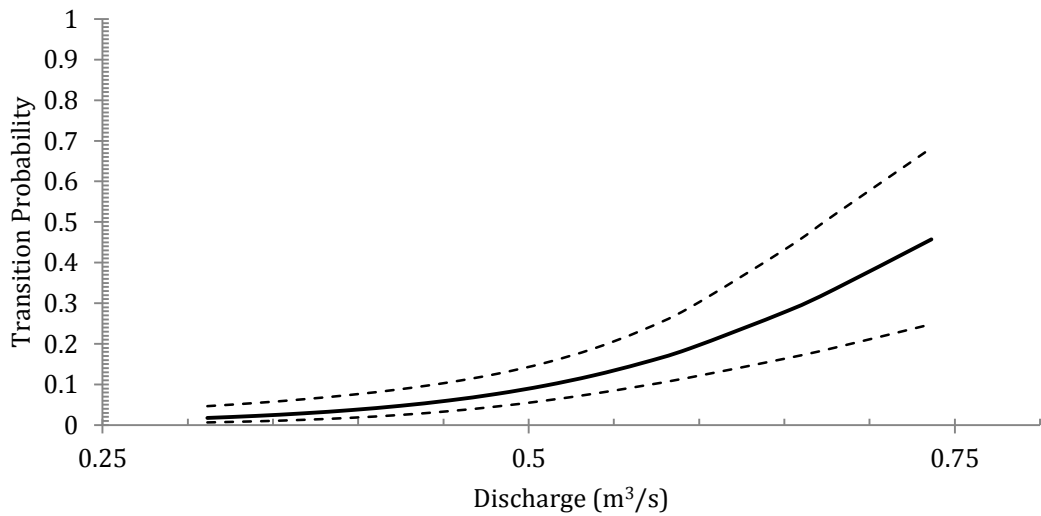


Figure 3-5.—Probability of cardinal shiner transitioning to adjacent channel units with changing discharge during 2011 (upper panel) and 2012 (lower panel).

Table 3-8—Top model results for 2011 and 2012 central stoneroller data. Only those models weighted (i.e., Delta QAICc < 10) are included of the 27 total candidate models.

Central Stoneroller								
Year	#	Model	npar	QAICc	DeltaQAICc	Weight	QDeviance	$\hat{c}$
2011	4	S(~1)p(~effort)Psi(~1)	4	179.98	0	0.37	104.31	1.28
	1	S(~1)p(~1)Psi(~1)	3	181.54	1.56	0.17	108.09	1.28
	13	S(~Q)p(~effort)Psi(~1)	5	181.69	1.71	0.16	103.74	1.28
	5	S(~1)p(~effort)Psi(~Q)	5	182.24	2.26	0.12	104.28	1.28
	10	S(~Q)p(~1)Psi(~1)	4	183.65	3.67	0.06	107.98	1.28
	2	S(~1)p(~1)Psi(~Q)	4	183.74	3.76	0.06	108.07	1.28
	14	S(~Q)p(~effort)Psi(~Q)	6	184.02	4.04	0.05	103.71	1.28
	11	S(~Q)p(~1)Psi(~Q)	5	185.92	5.93	0.02	107.96	1.28
2012	10	S(~Q)p(~1)Psi(~1)	4	646.24	0	0.23	542.24	1.99
	11	S(~Q)p(~1)Psi(~Q)	5	646.47	0.23	0.20	540.40	1.99
	1	S(~1)p(~1)Psi(~1)	3	647.04	0.80	0.15	545.10	1.99
	2	S(~1)p(~1)Psi(~Q)	4	647.26	1.01	0.14	543.26	1.99
	13	S(~Q)p(~effort)Psi(~1)	5	648.18	1.94	0.09	542.11	1.99
	14	S(~Q)p(~effort)Psi(~Q)	6	648.43	2.18	0.08	540.27	1.99
	4	S(~1)p(~effort)Psi(~1)	4	648.88	2.64	0.06	544.89	1.99
	5	S(~1)p(~effort)Psi(~Q)	5	649.12	2.87	0.05	543.05	1.99

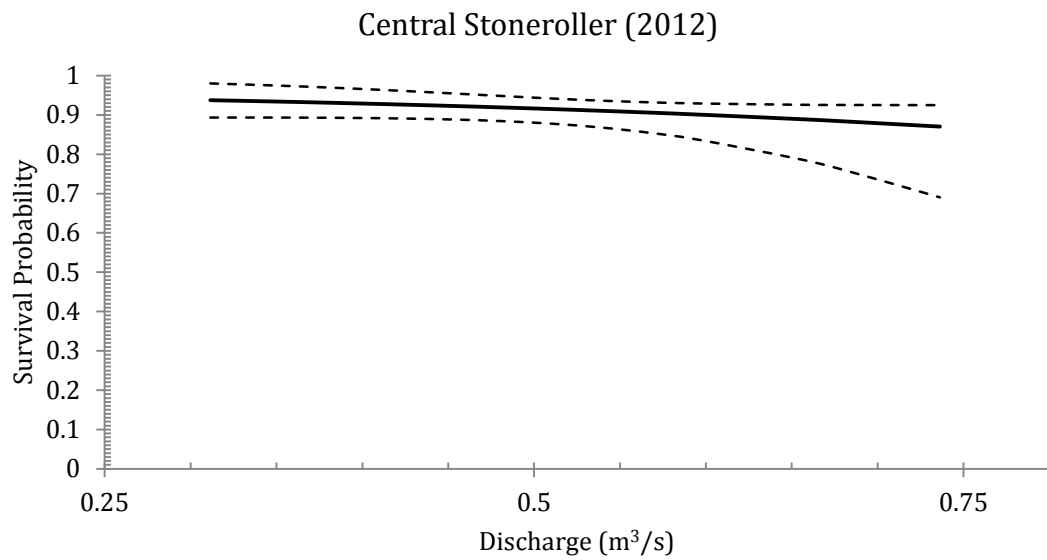
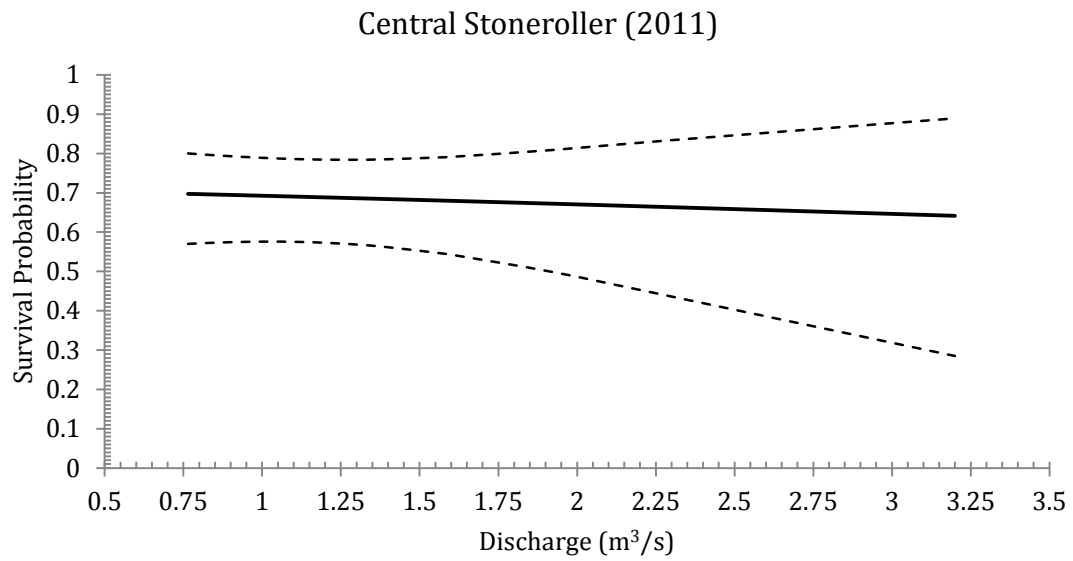


Figure 3-6.—Survival probability of central stoneroller related to discharge during 2011 (upper panel) and 2012 (lower panel).

Table 3-9. —Summed weights of parameter covariates across all weighted models for central stoneroller.

Central Stoneroller				
	covariates	S	p	Psi
2011	~1	0.41	0.72	0.53
	time	0	0	0
	Q	0.59	-	0.47
	effort	-	0.28	-
	covariates	S	p	Psi
2012	~1	0.72	0.30	0.76
	time	0	0	0
	Q	0.28	-	0.24
	effort	-	0.70	-



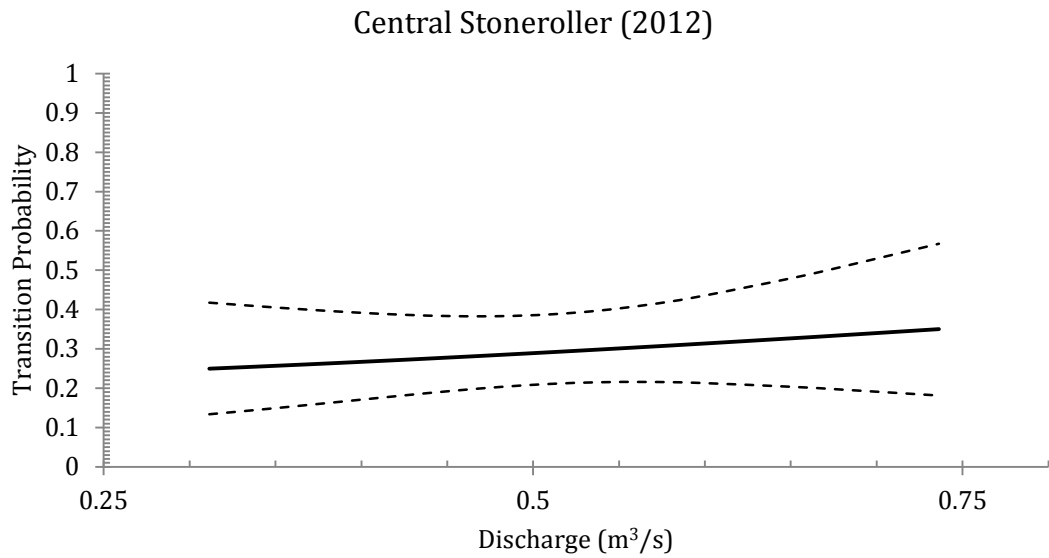
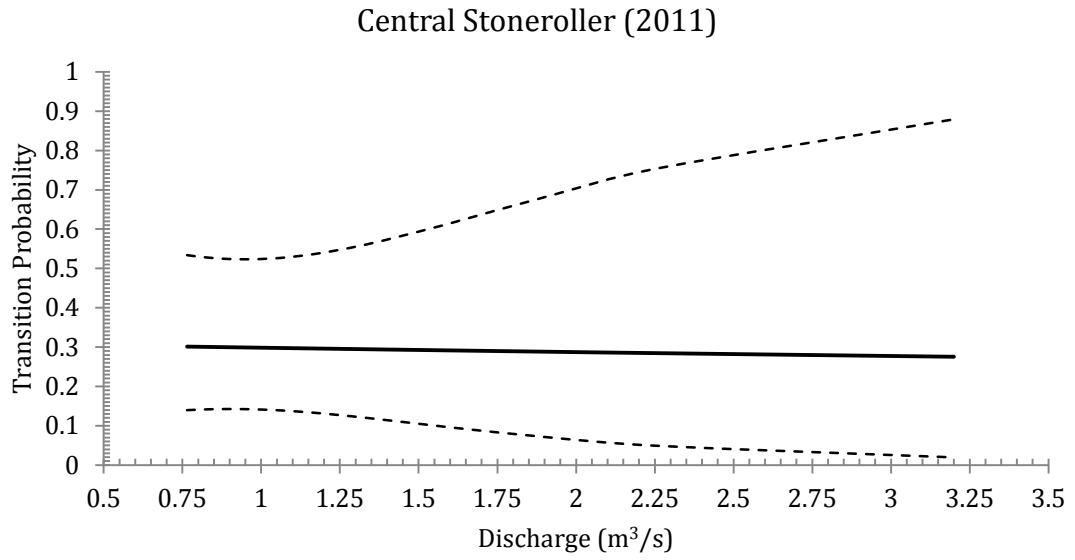


Figure 3-7.—Probability of central stoneroller transitioning to adjacent channel units with changing discharge during 2011 (upper panel) and 2012 (lower panel).

Table 3-10.—Top model results for 2012 orangethroat darter data. Only those models weighted (i.e., Delta QAICc < 10) are included of the 27 total candidate models.

Orangethroat Darter								
Year	#	Model	npar	QAICc	DeltaQAICc	Weight	QDeviance	$\hat{c}$
2012	11	S(~Q)p(~1)Psi(~Q)	5	408.65	0	0.58	303.37	1.74
	14	S(~Q)p(~effort)Psi(~Q)	6	410.76	2.10	0.20	303.34	1.74
	2	S(~1)p(~1)Psi(~Q)	4	411.76	3.10	0.12	308.58	1.74
	5	S(~1)p(~effort)Psi(~Q)	5	413.78	5.12	0.04	308.49	1.74
	10	S(~Q)p(~1)Psi(~1)	4	414.36	5.70	0.03	311.18	1.74
	13	S(~Q)p(~effort)Psi(~1)	5	416.44	7.78	0.01	311.15	1.74
	1	S(~1)p(~1)Psi(~1)	3	417.49	8.83	0.01	316.40	1.74

Table 3-11. —Summed weights of parameter covariates across all weighted models.

Orangethroat Darter				
	covariates	S	p	Psi
	~1	0.18	0.74	0.05
2012	time	0	0	0
	Q	0.82	-	0.95
	effort	-	0.26	-

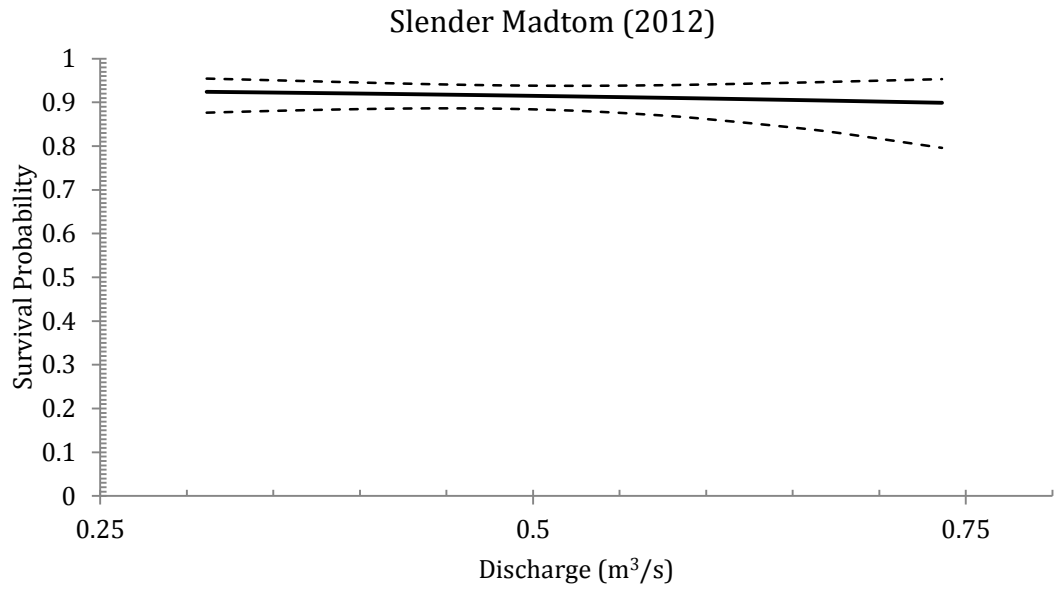
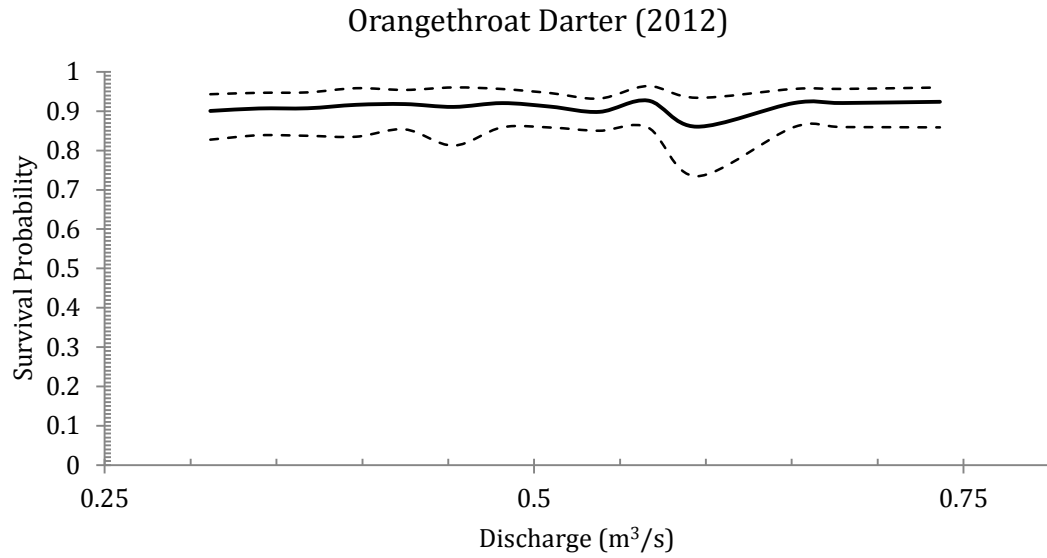
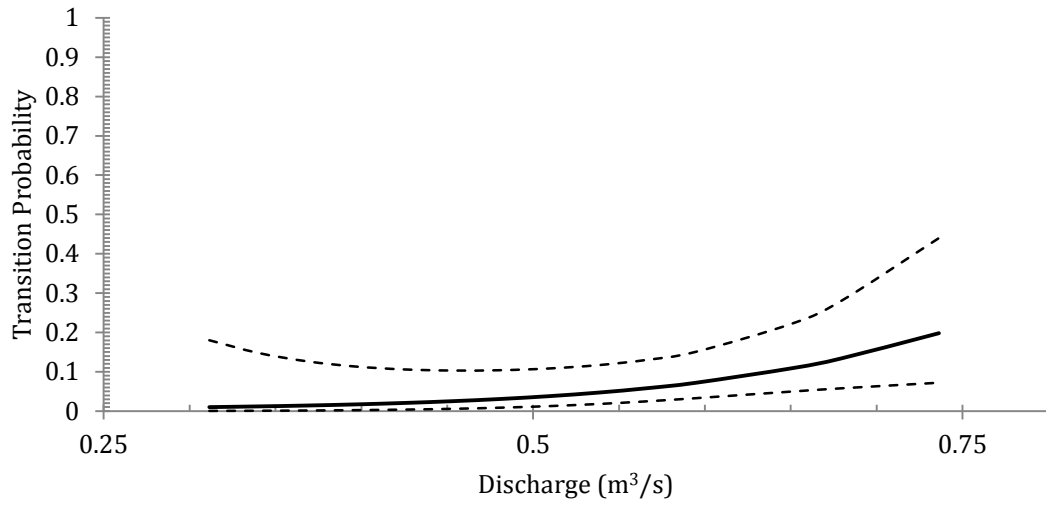


Figure 3-8.—Survival probability of orangethroat darter (upper panel) and slender madtom (lower panel) related to discharge during the 2012 field season.

### Orangethroat Darter (2012)



### Slender Madtom (2012)

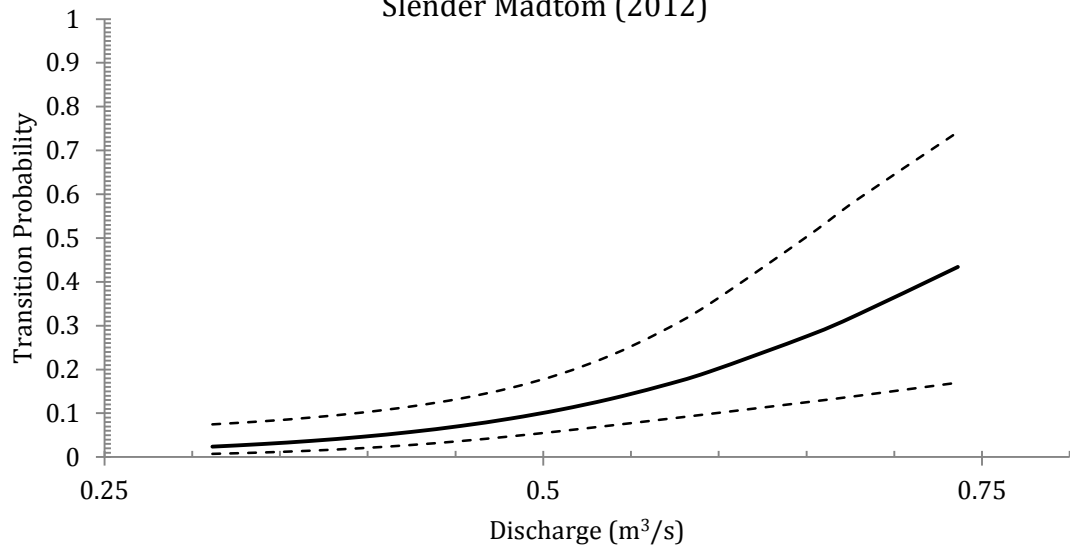


Figure 3-9.—Probability of orangethroat darter (upper panel) and slender madtom (lower panel) transitioning to adjacent channel units with changing discharge during 2012.

Table 3-12.—Top model results for 2012 slender madtom data. Only those models weighted (i.e., Delta QAICc < 10) are included of the 27 total candidate models.

Slender Madtom								
Year	#	Model	npar	QAICc	DeltaQAICc	Weight	QDeviance	$\hat{c}$
2012	5	S(~1)p(~effort)Psi(~Q)	5	700.86	0	0.61	575.52	1.03
	14	S(~Q)p(~effort)Psi(~Q)	6	701.78	0.92	0.39	574.30	1.03

Table 3-13. —Summed weights of parameter covariates across all weighted models for slender madtom.

Slender Madtom				
	covariates	S	p	Psi
2012	~1	0.61	0	0
	time	0	0	0
	Q	0.39	-	1.0
	effort	-	1.0	-

Table 3-14.—Top model results for 2011 and 2012 smallmouth bass data. Only those models weighted (i.e., Delta QAICc < 10) are included of the 27 total candidate models.

Smallmouth Bass								
Year	#	Model	npar	QAICc	DeltaQAICc	Weight	QDeviance	$\hat{c}$
2011	1	S(~1)p(~1)Psi(~1)	3	313.67	0	0.38	251.7	1.93
	2	S(~1)p(~1)Psi(~Q)	4	315.25	1.57	0.17	251.15	1.93
	4	S(~1)p(~effort)Psi(~1)	4	315.78	2.10	0.13	251.68	1.93
	10	S(~Q)p(~1)Psi(~1)	4	315.78	2.11	0.13	251.69	1.93
	5	S(~1)p(~effort)Psi(~Q)	5	317.39	3.71	0.06	251.14	1.93
	11	S(~Q)p(~1)Psi(~Q)	5	317.39	3.72	0.06	251.15	1.93
	13	S(~Q)p(~effort)Psi(~1)	5	317.93	4.26	0.05	251.68	1.93
	14	S(~Q)p(~effort)Psi(~Q)	6	319.57	5.90	0.02	251.14	1.93
2012	1	S(~1)p(~1)Psi(~1)	3	693.84	0	0.40	608.59	1.5
	4	S(~1)p(~effort)Psi(~1)	4	695.82	1.98	0.15	608.49	1.5
	2	S(~1)p(~1)Psi(~Q)	4	695.87	2.03	0.14	608.54	1.5
	10	S(~Q)p(~1)Psi(~1)	4	695.92	2.07	0.14	608.59	1.5
	5	S(~1)p(~effort)Psi(~Q)	5	697.87	4.03	0.05	608.44	1.5
	13	S(~Q)p(~effort)Psi(~1)	5	697.92	4.07	0.05	608.49	1.5
	11	S(~Q)p(~1)Psi(~Q)	5	697.97	4.13	0.05	608.54	1.5
	14	S(~Q)p(~effort)Psi(~Q)	6	699.99	6.15	0.02	608.44	1.5



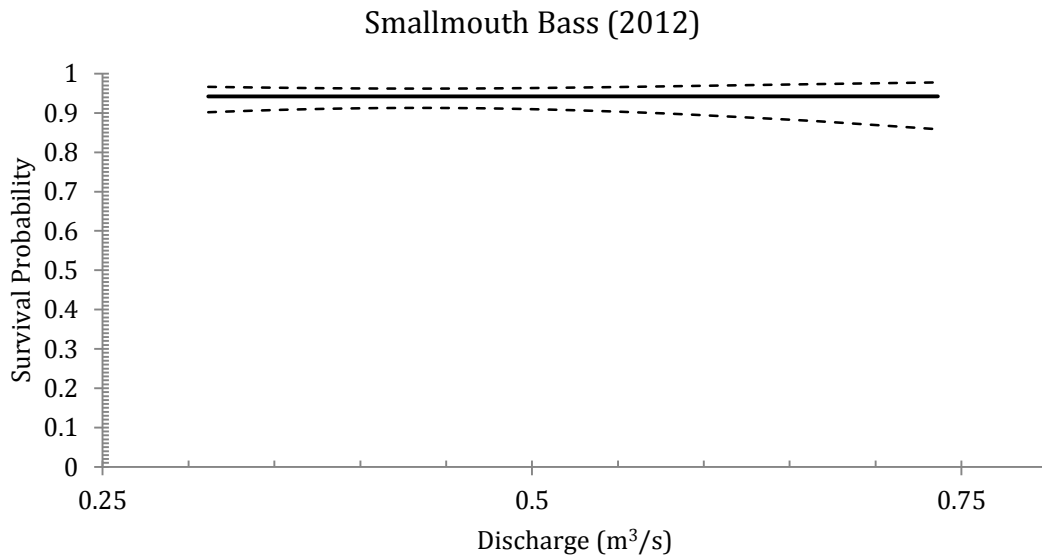
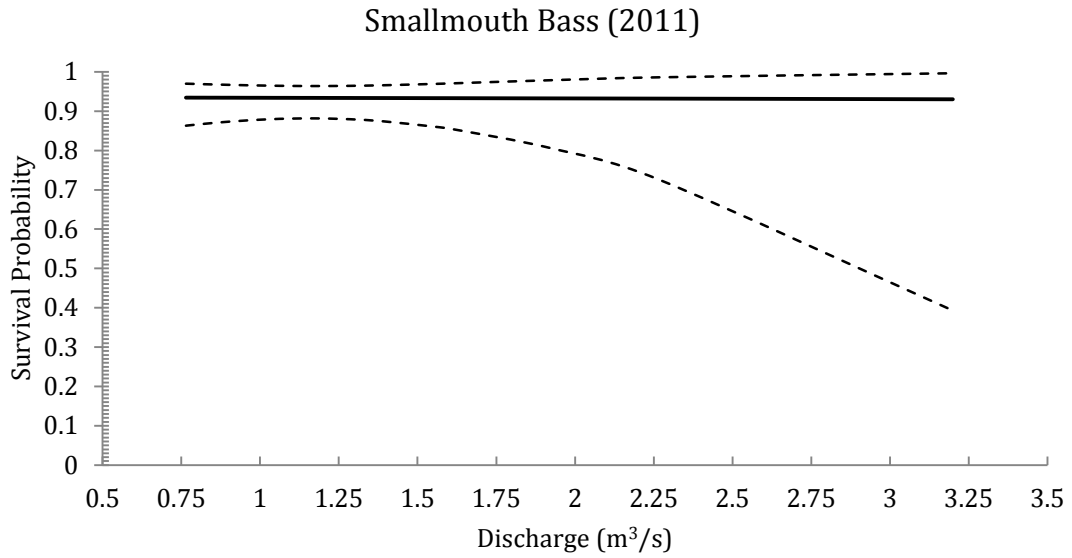


Figure 3-10.—Survival probability of smallmouth bass related to discharge during 2011 (upper panel) and 2012 (lower panel).

Table 3-15. —Summed weights of parameter covariates across all weighted models for smallmouth bass.

Smallmouth Bass				
	covariates	S	p	Psi
2011	~1	0.74	0.74	0.69
	time	0	0	0
	Q	0.26	-	0.31
	effort	-	0.26	-
	covariates	S	p	Psi
2012	~1	0.74	0.73	0.74
	time	0	0	0
	Q	0.26	-	0.26
	effort	-	0.27	-

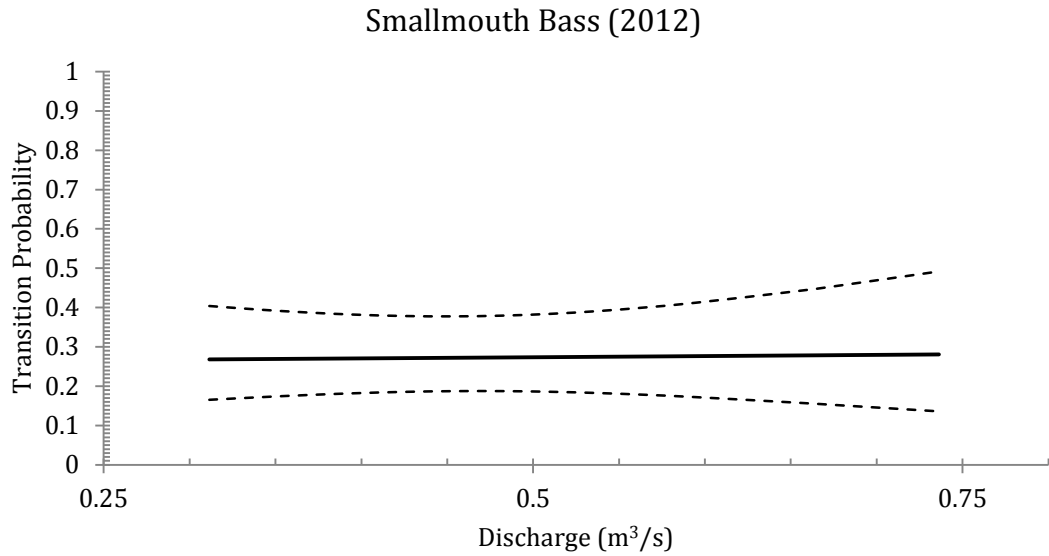
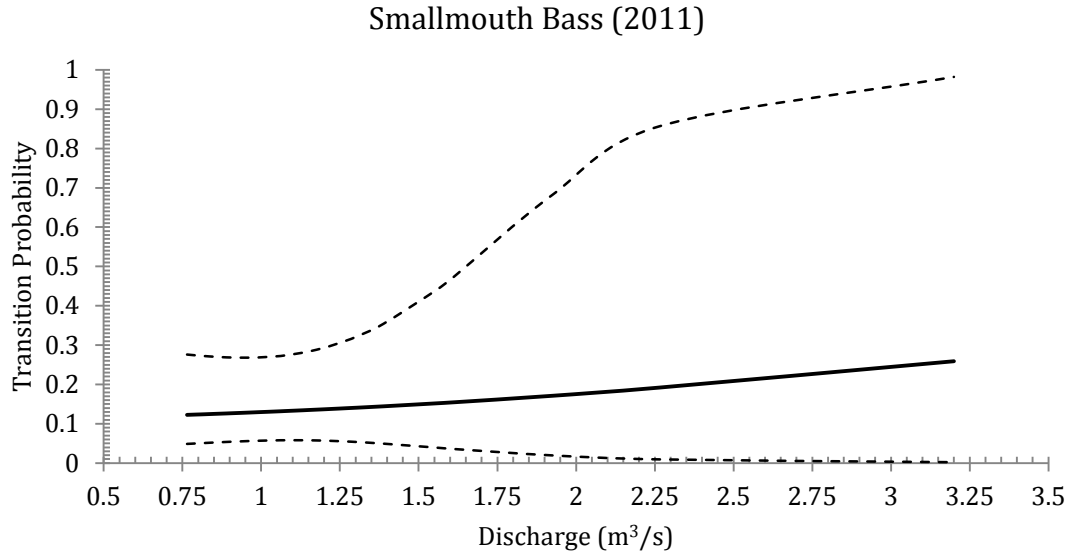


Figure 3-11.—Probability of smallmouth bass transitioning to adjacent channel units with changing discharge during 2011 (upper panel) and 2012 (lower panel).

## CHAPTER IV

### TEMPERATURE SUITABILITY FOR THE PERSISTENCE OF OZARK STREAM FISH: DOES INCREASING BASEFLOW IMPROVE THERMAL CONDITIONS?

#### INTRODUCTION

Environmental-flow recommendations would benefit from consideration of flow quantity and flow quality to improve the success of long-term stream conservation (Caissie 2006; Olden and Naiman 2010). Water quality (e.g., temperature) governs the water available for human consumption, recreation, and controls overall stream function and biotic integrity (Karr and Dudley 1981). Functioning lotic ecosystems provide key ecosystem services (e.g., potable water, reduction of aquatic pathogens and pests, propagation of fish and wildlife, and quality of life) upon which society ultimately depend (Karr and Dudley 1981; Postel et al. 2005; Loomis et al. 2000). Environmental flows aim to protect normal stream function and aquatic-ecosystem services by maintaining aspects of the natural flow regime (Poff et al. 1997; Arthington et al. 2006; Richter et al. 2010). However, without explicit consideration of water quality, flow prescriptions may not have the desired result.

Temperature is the most pervasive water-quality parameter because it controls physical and chemical processes that shape aquatic ecosystems (Wootton 1990; Brown et al. 2004; Delpha et

al. 2009). Introduced energy (i.e., heat) increases water solubility which makes nutrients (e.g., carbon, nitrogen, and phosphorus) more biologically available and increases molecular activity which speeds chemical reactions (Cummins 1974; Allan 1995; Stumm & Morgan 2012). For example, stream metabolism is essentially a series of biochemical reactions that increases exponentially with water temperature (Brown et al. 2004). Stream metabolism regulates most biological processes, so ecosystem dynamics are largely a consequence of water temperature (Brown et al. 2004). High metabolism and dissolved organic carbon fuels primary (e.g., phytoplankton, algae, and cyanobacteria) and secondary (e.g., diatom, protozoa, and hyphomycete fungi) production (Stumm and Morgan 2012). Increased water temperatures also speed leaf litter leaching and the breakdown of lignin which better conditions detritus for stream invertebrates (Cummins et al. 1989; Allan 1995). Greater microbial and invertebrate activity speed decomposition rates and tighten nutrient spiraling (Cummins 1974; Irons 1994; Ferreira & Chauvet 2011).

Water temperature shapes the behavior (e.g., Taniguchi et al. 1998; Reese & Harvey 2002; Armstrong et al. 2013), condition (e.g., Legler et al. 2010; Johnston et al. 2011; Kuparinen et al. 2011), and distribution of higher aquatic organisms (e.g., Brewer 2013; Dyer et al. 2013). Temperatures signal fish to begin different life-history behaviors including seasonal migrations and spawning (Wootton 1990). Fish growth, development, and life spans are all temperature dependent (Coutant 1976; Magnuson et al. 1979; Brown et al. 2004). Smallmouth bass have been found to select temperatures near their bioenergetics optima under field conditions (Brewer 2013). Selecting optimal temperature is important because temperatures experienced at early life stages influenced adult size, fecundity, and age of maturity of stream fishes (Legler 2010). This is due in part to a rise in heterotrophic respiration with temperature that creates more energy for foraging, growth, and reproduction (Brown et al. 2004). Temperature, because of its effects on metabolism, also determines rates of competition, predation, resistance to parasites and disease,

and eventual death (Elliott 1981; Brown et al. 2004; Roberts et al. 2012). For example, creek chub *Semotilus atromaculatus* became competitively dominant over brook trout *Salvelinus fontinalis* and brown trout *Salmo trutta* at higher temperatures optimal for foraging and growth of this species (Taniguchi et al. 1998). Optimal temperatures improve fish condition and reduce contraction rates of parasites and diseases (Le Morvan et al. 1998; Macnab & Barber 2012). Survival is also greatest for fishes under optimal temperatures (Smale & Rabeni 1995b; Beitinger et al. 2000). Optimal temperatures vary by species and the climate to which fish have evolved (Allan & Flecker 1993); hence, water temperatures are used to characterize different stream ecosystems (e.g., cold, cool, warm) and their distinct biota.

Fish function shows plasticity over a range of temperatures, but there are thermal thresholds where fish will experience reduced fitness. Depending on the magnitude and rate of thermal changes there may be minor readjustments of metabolic rates, or major changes in the distribution of species that can lead to extirpation or extinction (Coutant 1976). Fish vary metabolism to maintain homeostasis in the face of changing temperatures (Wootton 1990), but this either increases resource requirements or lowers available energy. Eventually homeostasis becomes unsustainable because of low energy, low oxygen, or food limitations and fish suspend growth, reproduction, reduce stored energy (e.g., fat, muscle, eggs), and enter into a state of reduced movement or torpor (Allan & Flecker 1993). For example, fishes exposed to suboptimal temperatures had reduced body condition (e.g., lower growth and higher mortality; Dickerson 1999), poorer fecundity (e.g., oocyte mutation and hermaphroditism; Lukšienė et al. 2000), and limited swimming ability (e.g., increased predation and displacement; Ward et al. 2003). Heterotrophic respiration is theorized to play a role in aging, so chronic exposure to even slight temperature increases reduces life expectancies (Brown et al. 2004). At extreme temperatures, the enzymes that catalyze metabolic reactions lose shape and become ineffective (Molles 1999). Fish exposed to high temperatures lose nerve function and express uncoordinated movements,

spasms, and eventual death (Baldwin & Hochachka 1970). Both elevated and lowered temperatures have been implicated in fish kills (Durham et al. 2006; Donaldson et al. 2008) and extirpation of native species (e.g., below tailraces; Lessard & Hayes 2003; Haxton & Findlay 2008). Consideration of thermal requirements would benefit future conservation and management actions (Olden and Naiman 2010).

Climate change and water abstractions have contributed to significant warming in streams (Webb and Nobilis 2007; Kaushal et al. 2010) and as human continue to change the thermal environment, the condition and composition of unprotected streams is also anticipated to change. Projections estimate a 14.2% loss of temperature-suitable habitat for warmwater fishes and a 50% loss for cool and coldwater fishes of the United States (Eaton and Scheller 1996). Over half of the variation in stream temperatures has been accounted for by discharge during summer extremes (Isaak et al. 2012), so it is unlikely that traditional management strategies aimed only at improving the riparian corridor will do much to protect thermal regimes except in very small streams (Whitledge et al. 2006). The goal of this chapter was to examine the role discharge plays in dictating stream temperature changes at the reach scale. My specific objectives were: 1) predict changes in summer stream temperatures under flow alterations; 2) compare predicted temperature changes to critical thermal maximum in stream fishes as a means of assessing the influence of temperature on Ozark fish assemblages.

## **METHODS**

### *Study sites*

The general study catchment is described in Chapter II. I modeled summer-stream temperatures at hypothetically lowered discharges for three 1-km reaches on Flint Creek, Barren Fork Creek, and the Illinois River (Figure 4-1). Flint Creek is a 3<sup>rd</sup> order tributary with a drainage area of 285

km<sup>2</sup> (110 mi<sup>2</sup>) with mean annual discharge of 3.3 m<sup>3</sup>/s (118 f<sup>3</sup>/s). Barren Fork Creek is a 4<sup>th</sup> order stream with a drainage area of 936 km<sup>2</sup> (307 mi<sup>2</sup>) with a mean annual discharge of 9.3 m<sup>3</sup>/s (329 f<sup>3</sup>/s). The upper Illinois River is a 6<sup>th</sup> order stream with a drainage area of 2,484 km<sup>2</sup> (959 mi<sup>2</sup>) with mean annual discharge of 26.8 m<sup>3</sup>/s (946 f<sup>3</sup>/s). Study sites were chosen haphazardly based on representativeness of the reach to the entire system.

Oklahoma experienced a drought in summer 2012 (Karl et al. 2012). Over the 62 days in July to August 2012, air temperatures (mean ± S.D.) were 27.91 ± 3.25 °C, maximum air temperatures were 35.20 ± 3.77 °C (Table 4-1), and only 84.84 mm of rain fell (1.37 ± 0.64 mm/day) in the study area. In the Illinois River, drought conditions resulted in a mean daily discharge of 3.09 ± 1.75 m<sup>3</sup>/s (65% below historic means) and measured water temperatures as high as 32.09 °C (Table 4-1). Similarly, Barren Fork Creek had an extreme low discharge of 0.40 ± 0.11 m<sup>3</sup>/s (88% below historic means) and water temperatures up to 31.47 °C. Flint Creek's mean daily discharge was 0.46 ± 0.16 m<sup>3</sup>/s (71% below historic means), but water temperatures never exceeded 30.30 °C.

#### Stream-temperature modeling

Summer water temperatures were modeled at hypothetically lowered discharges using the U.S. Geological Survey Stream Segment Temperature Model (SSTEMP) Version 2.0 (Bartholow 1999). The SSTEMP model calculates the heat gained or lost from a stream segment to predict mean daily stream temperatures and estimate daily minimum and maximum temperatures (Bartholow 1999).

The steps completed for SSTEMP modeling were: 1) collection of stream and weather data, 2) model calibration, and 3) stream temperature prediction under theoretical conditions (Bartholow 1989). Stream temperatures were recorded hourly using temperature loggers (HOBOLogger Onset, Bourne, MA). All meteorological data were collected from Mesonet



stations (Brock et al. 1995) near Jay, OK (station No. 51, JAYX) and Westville, OK (station No. 104, WEST). Stream geometry (i.e., elevations, distances, and stream widths) was measured using a rangefinder and GPS (Bartholow 1989). Stream shading was estimated from measured stream characteristics (i.e., mean width, azimuth, latitude, and topographic altitude) and riparian vegetation characteristics (i.e., distance from stream, crown diameter, crown height, and vegetation density; Bartholow 1999). Stream discharge was measured using the velocity-area method (Gordon 2004) with an electromagnetic-flow meter (Marsh-McBirney, Frederick, MD) and wading rod (0.6 depth or 0.2 and 0.8. depth depending on water depth; Gordon 2004). Manning's  $n$  was set to 0.035 and the default temperature gradient of  $1.650 \text{ j/m}^2/\text{s}/\text{C}$  was used for my scenarios. Maximum stream temperatures were estimated from models based on measured stream conditions in summer 2012 (a drought year). Model fit was assessed using root-mean-squared error (RMSE) and coefficient of determination ( $r^2$ ).

After goodness-of-fit testing, built models were used to predict stream temperatures under low-flow conditions during the hottest and driest period of summer 2012. The mean daily discharge for July – August 2012 was collected from the nearest USGS stream gage (Watts, OK #07195500; Eldon, OK #11110103; Kansas, OK #11110103) and used as the model base flow (i.e., 100%). Discharge was then hypothetically lowered (i.e., 90%, 75%, and 50%) to mimic extreme-low flows (Table 4-2). I also modeled scenarios of increased discharge (e.g., 200% or higher) to assess the feasibility of lowering stream temperatures via flow management (Table 4-2). The overall mean breakpoint estimates that were provided in Chapter II were also included as a scenario. I created exceedance probabilities using Monte Carlo resampling (3000 trials with 100 samples/trial) that randomly selected estimates for input parameters based on prior distributions (Bartholow 1999). I used standard deviations associated with mean daily data as parameter distributions during resampling. To isolate the effects of low flows on summer stream temperatures, all meteorology variables and stream inflow temperatures were given prior

distributions. All other stream hydrology, geometry, and shading variables had a distribution of zero and remained static at their respective means. Percent possible sun was given a distribution of  $90 \pm 10\%$  and time of year (i.e., day length) was set to August 1<sup>st</sup>.

Model results were compared to published critical thermal maximum (CTM) data for warmwater fishes (literature compiled in 2011; Table 4-3) to determine if lethal thermal thresholds would be exceeded by the discharge scenarios that I used. CTM is the accepted method for measuring temperature tolerance in fishes (Lutterschmidt & Hutchison 1997). During CTM studies, the water temperature increases at a rate fast enough ( $1^{\circ}\text{C}$  per min to  $1^{\circ}\text{C}$  per h; Becker & Genoway 1979) to prevent acclimation and continues to increase until the fish reaches loss of equilibrium, operculum spasms, or death (Lutterschmidt & Hutchinson 1997). Given time to acclimate to rising temperatures, stream fishes may tolerate higher temperature than many CTM studies suggest (Becker & Genoway 1979). Therefore, studies using acclimation temperatures below  $20.0^{\circ}\text{C}$  were excluded from the meta-analysis, as these stream temperatures are considerably lower than expected summer values in my study systems. For those species with multiple published CTMs, the lowest CTM with the highest acclimation temperature was used for my comparison.

## RESULTS

Overall goodness-of-fit (i.e., RMSE and  $r^2$ ) for SSTEMP models was excellent for predicting mean daily water temperatures in Flint Creek, Barren Fork Creek, and the Illinois River (Table 4-4). Accuracy of maximum temperatures models was also excellent for the Illinois River (RMSE = 0.71 and  $r^2 = 0.94$ ). Maximum-temperature models for Flint Creek (RMSE = 2.36 and  $r^2 = 0.83$ ) and Barren Fork Creek (RMSE = 2.56 and  $r^2 = 0.97$ ) were consistent, but slightly overestimated maximum water temperatures.

Stream discharge influenced mean maximum water temperatures and the probability of exceeding temperatures in the Illinois River and Flint Creek. In the Illinois River, mean maximum water temperature was 31.63 at mean daily discharge (i.e., 100% or 3.09 m<sup>3</sup>/s). Reducing discharges by 10%, 25%, and 50% increased maximum water temperatures by 0.06 °C, 0.15 °C, and 0.32 °C (Figure 4-2). Likewise, reduced flows slightly increased the probability of higher water temperatures (Figure 4-3). For example, a 50% decrease in summer discharge increased the probability of exceeding 33.0 °C from 11% to 24%. Flint Creek followed a similar pattern as the Illinois River, except maximum water temperatures were much cooler. Maximum water temperatures were: 29.55 at mean daily discharge (i.e., 100% or 0.46 m<sup>3</sup>/s), and a 10%, 25%, and 50% reduction in discharge increased maximum temperatures by only 0.01 °C, 0.05 °C, and 0.13 °C (Figure 4-2). Most importantly, under all discharge scenarios, the probability of maximum temperatures exceeding 31 °C (i.e., the upper thermal limit for many fish species) was less than 10% (Figure 4-4). Overall variation in temperatures under multiple flow scenarios was small. For example, at a 10% exceedance probability, maximum temperatures ranged from 30.33 °C to 30.81 °C (a difference of only 0.48 °C).

Water temperature predictions of Barren Fork Creek were difficult to reconcile. Barren Fork Creek had a maximum water temperature of 31.95 ± 0.35 °C at the mean daily discharge for July and August (i.e., 0.40 m<sup>3</sup>/s). Unexpectedly, reductions to 90%, 75%, and 50% of discharge decreased maximum water temperatures by 0.02 °C, 0.07 °C, and 0.23 °C (Figure 4-2). This indicates lower discharges have the unrealistic potential for reducing maximum water temperatures (e.g., exceedance probabilities that intersected below 32 °C), most likely because of the breakdown in the width-discharge equation at extreme low flows (Bartholow 2000). However, the probabilities of exceeding higher temperatures followed similar patterns as the Illinois River and Flint Creek (i.e., negative correlation with discharge). For example, a 50% decrease in summer baseflows increased the probability of exceeding 34.0 °C from 13% to 18%.

Flows increased to 200% of the mean (i.e., 0.80 m<sup>3</sup>/s) had only a 0.01 °C decrease in maximum water temperature and a probability of exceeding 34.0 °C near 10%. However, at 3.27 m<sup>3</sup>/s (i.e., my extreme increase of 818% of the mean discharge and the lower limit of my breakpoint estimate from Chapter II), I predicted a decrease of 0.55 °C in maximum water temperature and a very small (< 5%) probability of exceeding 34.0 °C.

Decreased discharge in the Illinois River and Barren Fork Creek increased the probability of exceeding CTM thresholds whereas Flint Creek appears to be reasonably buffered against temperature fluctuations (Figures 4-4). For example, at 100% discharge in the Illinois River, there was a 39% chance of exceeding CTM for duskystripe shiner *Notropis pilsbyri* (i.e., sister species to cardinal shiner *Luxilus cardinalis*), a 25 – 31% probability of exceeding CTM for southern redbelly dace *Phoxinus erthrogaster*, banded sculpin *Cottus carolinae*, and redbfin shiner *Lythrurus umbratilis*, and an 11% probability of exceeding CTM for central stoneroller *Campostoma anomalum* and bluntnose minnow *Pimephales notatus*. A 50% decrease in discharge increased exceedance probabilities to 49% for duskystripe shiner; 36 – 41% for southern redbelly dace, banded sculpin, and redbfin shiner; and 25% for central stoneroller and bluntnose minnow. At this extreme low discharge, CTM for roseyface shiner *Notropis rubellus*, northern hogsucker *Hypentilium nigricans*, greenside darter *Etheostoma blennioides*, and banded sculpin were approached but not exceeded. Alternatively, a 100% increase in discharge lowered exceedance probabilities for several species and removed others from immediate risk. Barren Fork Creek had extreme low-flow conditions in 2012 that increased the potential for exceeding CTM for several species (Figure 4-4). Decreased flows increased the possibility of exceeding temperature thresholds only slightly. For example, the probability of exceeding CTM for greenside darter and banded sculpin ranged from 10% at full discharge to 15% at half discharge. Increased discharge reduced the chance of exceeding temperature thresholds, but a potential to exceed CTM remained at all discharges considered. The unlikely situation of increasing

discharge by 818% was the only scenario that effectively reduced the risk of exceeding CTM for all species in Barren Fork Creek. Unlike the other study systems, maximum water temperatures in Flint Creek had less than a 10% probability of exceeding CTM for any species even at the lowest flows.

## DISCUSSION

Fishes in the Illinois River and Barren Fork Creek were most susceptible to thermal stress and temperature stress seemed likely for multiple species in these systems. Abundant minnow species such as cardinal shiner and central stoneroller likely experienced temperatures at or above CTM during summer 2012. Maximum temperatures in Barren Fork Creek had the potential to stress more of the assemblage including: common benthic species (e.g., banded sculpin and greenside darter) and local game fish (e.g., northern hogsucker). Continued loss of discharge only exacerbated the risk of thermal stress to fishes in these systems. For example, as habitats shrank, shallow waters became more susceptible to warming and fish were likely more exposed to direct sunlight (i.e., less available riparian shade; Whitley & Rabeni 2006).

Thermal stress was likely in these systems even when CTM values were not exceeded because CTMs represent conservative thermal thresholds and are typically much higher than preferred temperatures (i.e., those optimal for growth and reproduction; Hasnain et al. 2013). This is due in part to survival temperatures being investigated typically in a controlled laboratory setting. The lab temperature at which fish are acclimated to at the beginning of these studies is an important aspect controlling the CTM of fish (Beitinger et al. 2000). Fish in our study were conditioned to high summer temperatures and potentially different CTM than the same species identified in the literature conditioned to lower lab temperatures. Additionally, investigations into fish CTM overwhelmingly use adult life stages in experiments. However, fish vary in their

temperature tolerances by life stage (McCullough 1999). For example, smallmouth bass *Micropterus dolomieu* are more vulnerable to heat death as fertilized eggs and fry than as adults (Shuter et al. 1980). Conversely, juvenile smallmouth bass have higher optimal growth temperatures than adults, but are more vulnerable to high temperatures because growth declines rapidly above thermal thresholds and smaller fish have less stored energy (Shuter & Post 1990). However, because the CTMs used in this study are conservative estimates based on experiments using adult fish, I can reasonably assume that thermal stress was likely affecting multiple life stages in Barren Fork Creek and Illinois River.

Overexposure of stream fishes to high temperatures may reduce the fitness (e.g., growth, survival, and fecundity) of populations and addition of other stressors (likely in a natural setting) may exacerbate fitness consequences. Fish can survive at temperatures near or above CTM for some period of time, but chronic exposure increases stress, limits growth and reproduction, changes behavior, and facilitate invasion by non-native species (Coutant 1976). Confounding biotic (e.g., age, parasites, and competition) and abiotic (e.g., dissolved oxygen and pollution) factors can also lower CTM of fishes (Smale & Rabeni 1995b). In fact, lethal-temperature thresholds are a function of temperature, exposure length, and other stressors. Wehrly et al. (2007) found significant reductions in fish thermal tolerances after chronic exposure to adverse temperatures. Exposure to high temperature combined with other stressors may result in increased mortality (Coutant 1976). Eventual shifts in community composition and abundance are likely due to thermal stress at multiple temporal scales (Mohseni et al. 2003).

Stream connectivity has implications for stream temperatures at multiple spatial scales. Continuous discharge links instream habitats (see Chapter III), so reduced discharge is correlated with reduced longitudinal (Vannote et al. 1980), horizontal (Junk et al. 1989), and vertical connectivity (Fox 2004). Longitudinal connectivity influences the water quality of receiving systems as discharge from cooler-water tributaries acts as buffers against extreme water

temperatures and continued downstream heating (Macdonald et al. 2012). For example, Flint Creek discharged maximum water temperatures 2.0 °C cooler into the Illinois River during summer 2012. Lost discharge reduced horizontal connectivity at the channel-unit scale (see Chapter II) and created isolated habitats that heated at different rates than main-channel habitats. The SSTEMP model assumed constant mixing and a mean temperature was predicted for the entire reach; however, backwaters are shallow off-channel habitats that often lose connectivity during periods of low flow. Fish can become isolated in these habitats when waters recede and consequently have no mode of escape if water temperatures become bioenergetically unfavorable. Under drought conditions of summer 2012, measured temperatures in an isolated backwater exceeded the highest predicted maximum water temperatures (e.g., 34.1 °C) in the Illinois River. Although some isolation or fragmentation of habitat is normal in streams, the increased frequency of these events due to anthropogenic disturbance is harmful for stream-fish communities (Perkin et al. 2014). Loss of vertical or groundwater connectivity would also negatively affect thermal regimes. Depleted groundwater storage has been related to reduced baseflows and higher water temperatures (Zektser et al. 2005). My results suggest Flint Creek provides a cooler and more stable thermal regime compared to the other rivers and could provide important refugia to fishes as atmospheric conditions warm and surface waters become reduced. Maintaining connectivity of Flint Creek to the Illinois River may be important for fishes that need to seek refugia from temperature-stressed habitats. There may be locations on Barren Fork Creek and the Illinois River that offer thermal refugia to fishes (e.g., springs or areas of significant hyporheic storage). Identifying the locations of thermal refugia may benefit the persistence of some fish populations that occur in the region and have lower CTM values.

Environmental flows have been used as a successful management strategy to maintain thermal regimes and protect fish, but the approach to achieving this protection varies depending on how streamflow is regulated. Use of minimum flows to reduce summer stream temperatures

and avoid exceeding thermal thresholds for fishes has been achieved below dams on the Flathead River in Montana (Stanford & Hauer 1992), River Haddeo in the United Kingdom (Webb & Walling 1997), and the Nechako River in Canada (Macdonald et al. 2012). Recommendations of modified dam operations to benefit stream temperatures for fish are not an unusual practice (e.g., Halleraker et al. 2007; Bartholow 2010). Many examples related to thermal management used hypolimnetic-dam releases to achieve the thermal goal of cooler water. Rivers and streams that are not regulated by large dams, however, do not have reservoir flow releases that may be used to alter temperature regimes. One available management strategy for these systems includes use of minimum flow values to cap stream abstractions (Poole & Berman 2001). These ‘cease-to-pump limits’ minimize human exacerbation during critical low flows. Zeigler et al. (2012) found climate change was altering thermal regimes for endemic fish and suggested reduced water abstractions as an effective management option. If thermal protection of streams is the goal, then groundwater inflows that establish baseflow levels and moderate stream temperatures in receiving streams (Brunke & Gonser 1997) should also be considered. Pumping wells for agricultural or municipal water can draw down the alluvial aquifer and reduce groundwater connectivity in streams, even if not directly appropriated from surface waters (Poole & Berman 2001; Fox 2004; Fox et al. 2011). This source of water withdrawal can lead to appreciable changes to baseflows in Oklahoma streams (Fox et al. 2011) even though current legislation does not recognize groundwater-surface water connections in most basins. Though my results show that minimum flows do little to prevent temperature stress in a single stream reach, at the larger stream scale, minimum flows would benefit longitudinal connectivity to potential thermal refugia (e.g., Flint Creek).



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TABLES AND FIGURES

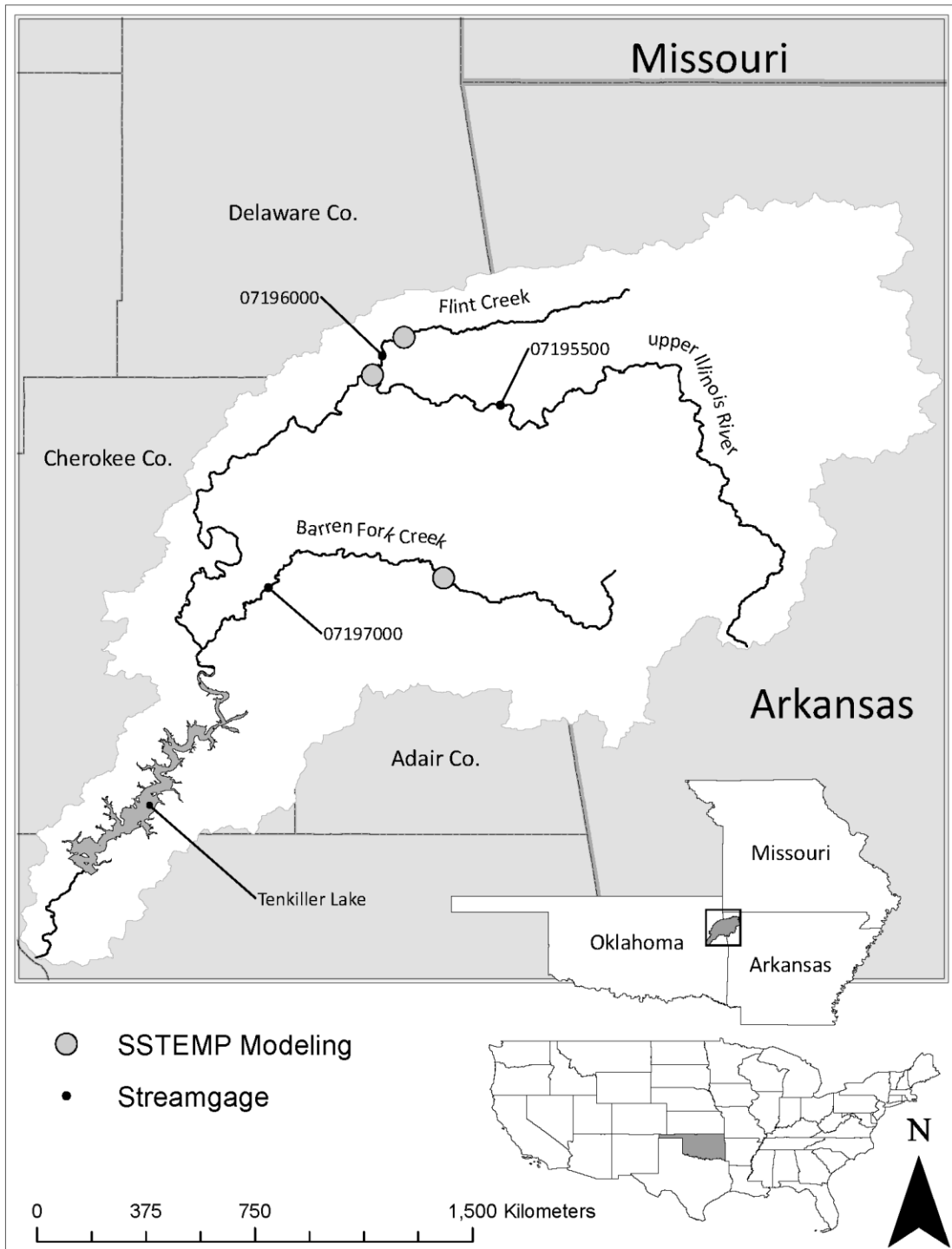


Figure 4-1.—Study sites for SSTEMP modeling in the Illinois River catchment.

Table 4-1.—Mean water temperature (H<sub>2</sub>O; °C), discharge (Q; m<sup>3</sup>/s), meteorological conditions (i.e., percent humidity, wind speed [m/s], solar radiation [J/sec\*m<sup>2</sup>]), and ground temperature [°C ] from July – August 2012 used in SSTEMP modeling for Flint Creek, Barren Fork Creek, and the Illinois River. Standard deviations (SD) of means are provided.

	Illinois River		Barren Fork Creek		Flint Creek		Air °C		Humidity %	Wind m/s	Solar radiation	Soil °C
	H <sub>2</sub> O	Q	H <sub>2</sub> O	Q	H <sub>2</sub> O	Q	Mean	Max				
Mean	28.37	3.09	28.51	0.4	27.09	0.46	27.91	35.2	53.68	2.73	248.24	29.49
SD	1.84	1.75	1.23	0.11	0.94	0.16	3.25	3.77	12.98	0.7	54.04	2.69

Table 4-2. —Range of discharges (m<sup>3</sup>/s) used for SSTEMP modeling for Flint Creek, Barren Fork Creek, and Illinois River in 2012. All discharges are presented as a percentage of the mean daily discharge observed in July-August 2012.

	Percent of Discharge (m <sup>3</sup> /s)						
	100%	90%	75%	50%	200%	300%	818%
Flint Creek	0.46	0.41	0.35	0.23	0.92	1.38	-
Barren Fork	0.40	0.36	0.30	0.20	0.80	-	3.27
Illinois River	3.09	2.78	2.32	1.55	6.23	-	-

Note: discharges were raised (e.g., 200%, 300%, and 818%) to levels within 90% C.L. for mean breakpoint estimates (mean of all habitats) obtained from objective one

Table 4-3.—Summary of known critical thermal maxima (CTM) for species found in the Illinois River watershed or related species.

Species	Mean (°C)	SD	Acclimation (°C)	Rate of increase	Publication	
<i>Notropis pilsbyri</i>	duskystripe shiner	32	-	22.5	2.0 °C day <sup>-1</sup>	Hickman & Dewey (1973)
<i>Phoxinus erthrogaster</i>	southern-redbelly dace	32.3	-	21.5	0.5 °C min <sup>-1</sup>	Scott (1987)
<i>Cottus carolinae</i>	banded sculpin	32.4	0.204	20	0.3 °C min <sup>-1</sup>	Walsh et al. (1997)
<i>Lythrurus umbratilis</i>	redfin shiner	32.5	-	-	1.0 °C min <sup>-1</sup>	Maness & Hutchison (1980)
<i>Cottus carolinae</i>	banded sculpin	32.8	0.16	20	0.3 °C min <sup>-1</sup>	Walsh et al. (1997)
<i>Campostoma anomalum</i>	central stoneroller	33	-	30	1.0 °C day <sup>-1</sup>	Cherry et al. (1977)
<i>Pimephales notatus</i>	bluntnose minnow	33	-	30	1.0 °C day <sup>-1</sup>	Cherry et al. (1977)
<i>Notropis rubellus</i>	roseface shiner	34	-	30	1.0 °C day <sup>-1</sup>	Cherry et al. (1977)
<i>Hypentilium nigricans</i>	northern hogsucker	34	-	33	1.0 °C day <sup>-1</sup>	Cherry et al. (1977)
<i>Etheostoma blenniodes</i>	greenside darter	34.2	0.4	20	1.0 °C min <sup>-1</sup>	Hlohowskyj & Wissing (1984)
<i>Cottus carolinae</i>	banded sculpin	34.2	0.1	25	0.3 °C min <sup>-1</sup>	Walsh et al. (1997)
<i>Cottus carolinae</i>	banded sculpin	34.3	0.1	25	0.3 °C min <sup>-1</sup>	Walsh et al. (1997)
<i>Etheostoma blenniodes</i>	greenside darter	34.9	0.7	20	1.0 °C min <sup>-1</sup>	Hlohowskyj & Wissing (1984)
<i>Micropterus dolomieu</i>	smallmouth bass	35	-	33	1.0 °C day <sup>-1</sup>	Cherry et al. (1977)
<i>Luxilus zonatus</i>	bleeding shiner	35.3	0.5	26	2.0 °C hour <sup>-1</sup>	Smale & Rabeni (1995a)
<i>Notropis rubellus</i>	rosyface shiner	35.3	0.23	26	2.0 °C hour <sup>-1</sup>	Smale & Rabeni (1995a)
<i>Micropterus salmoides</i>	largemouth bass	35.4	0.47	20	0.3 °C min <sup>-1</sup>	Currie et al. (1998)
<i>Etheostoma flabellare</i>	fantail darter	35.5	1.1	20	1.0 °C min <sup>-1</sup>	Hlohowskyj & Wissing (1984)
<i>Ictalurus punctatus</i>	channel catfish	35.5	0.38	20	1.0 °C min <sup>-1</sup>	Cheetham et al. (1976)
<i>Lepomis macrochirus</i>	bluegill	35.5	-	21.5	2.0 °C day <sup>-1</sup>	Hickman & Dewey (1973)
<i>Nocomis biguttatus</i>	hornyhead chub	35.6	0.55	26	2.0 °C hour <sup>-1</sup>	Smale & Rabeni (1995a)
<i>Semotilus atromaculatus</i>	creek chub	35.7	0.44	26	2.0 °C hour <sup>-1</sup>	Smale & Rabeni (1995a)
<i>Phoxinus erthrogaster</i>	southern-redbelly dace	35.9	0.42	26	2.0 °C hour <sup>-1</sup>	Smale & Rabeni (1995a)
<i>Etheostoma flabellare</i>	fantail darter	36	0.5	20	1.0 °C min <sup>-1</sup>	Hlohowskyj & Wissing (1984)
<i>Etheostoma flabellare</i>	fantail darter	36	0.54	26	2.0 °C hour <sup>-1</sup>	Smale & Rabeni (1995a)
<i>Labidesthes sicculus</i>	brook silversides	36	0.44	26	2.0 °C hour <sup>-1</sup>	Smale & Rabeni (1995a)
<i>Lythrurus umbratilis</i>	redfin shiner	36.2	0.54	26	2.0 °C hour <sup>-1</sup>	Smale & Rabeni (1995a)
<i>Notropis nubilus</i>	Ozark minnow	36.2	0.62	26	2.0 °C hour <sup>-1</sup>	Smale & Rabeni (1995a)
<i>Micropterus salmoides</i>	largemouth bass	36.3	0.6	26	2.0 °C hour <sup>-1</sup>	Smale & Rabeni (1995a)
<i>Ictalurus punctatus</i>	channel catfish	36.4	0.25	20	0.3 °C min <sup>-1</sup>	Currie et al. (1998)
<i>Micropterus salmoides</i>	largemouth bass	36.5	0.51	24	0.2 °C min <sup>-1</sup>	Fields et al. (1987)

<i>Etheostoma spectabile</i>	orangethroat darter	36.5	0.15	26	2.0 °C hour <sup>-1</sup>	Smale & Rabeni (1995a)
<i>Noturus exilis</i>	slender madtom	36.5	0.24	26	2.0 °C hour <sup>-1</sup>	Smale & Rabeni (1995a)
<i>Pimephales notatus</i>	bluntnose minnow	36.6	0.48	26	2.0 °C hour <sup>-1</sup>	Smale & Rabeni (1995a)
<i>Micropterus salmoides</i>	largemouth bass	36.7	0.76	20	1.0 °C min <sup>-1</sup>	Smith & Scott (1975)
<i>Micropterus salmoides</i>	largemouth bass	36.7	0.59	25	0.3 °C min <sup>-1</sup>	Currie et al. (1998)
<i>Notemigonus crysoleucas</i>	golden shiner	36.8	0.37	26	2.0 °C hour <sup>-1</sup>	Smale & Rabeni (1995a)
<i>Micropterus dolomieu</i>	smallmouth bass	36.9	0.31	26	2.0 °C hour <sup>-1</sup>	Smale & Rabeni (1995a)
<i>Fundulus sciadicus</i>	plains topminnow	37	0.3	26	2.0 °C hour <sup>-1</sup>	Smale & Rabeni (1995a)
<i>Ambloplites rupestris</i>	rock bass	37	-	36	1.0 °C day <sup>-1</sup>	Cherry et al. (1977)
<i>Campostoma anomalum</i>	central stoneroller	37.2	0.33	26	2.0 °C hour <sup>-1</sup>	Smale & Rabeni (1995a)
<i>Micropterus salmoides</i>	largemouth bass	37.3	0.6	32	1.0 °C day <sup>-1</sup>	Fields et al. (1987)
<i>Ictalurus punctatus</i>	channel catfish	37.5	0.52	24	1.0 °C min <sup>-1</sup>	Cheetham et al. (1976)
<i>Micropterus salmoides</i>	largemouth bass	37.5	0.64	24	0.2 °C min <sup>-1</sup>	Fields et al. (1987)
<i>Campostoma anomalum</i>	central stoneroller	37.7	0.5	26	0.5-0.8 °C min <sup>-1</sup>	Mundahl (1990)
<i>Etheostoma flabellare</i>	fantail darter	37.7	0.5	27	0.5-0.8 °C min <sup>-1</sup>	Mundahl (1990)
<i>Lepomis megalotis</i>	longear sunfish	37.8	0.84	26	2.0 °C hour <sup>-1</sup>	Smale & Rabeni (1995a)
<i>Pimephales notatus</i>	bluntnose minnow	37.9	0.5	25	0.5-0.8 °C min <sup>-1</sup>	Mundahl (1990)
<i>Amerius natalis</i>	yellow bullhead	37.9	0.44	26	2.0 °C hour <sup>-1</sup>	Smale & Rabeni (1995a)
<i>Lepomis cyanellus</i>	green sunfish	37.9	0.75	26	2.0 °C hour <sup>-1</sup>	Smale & Rabeni (1995a)
<i>Lepomis macrochirus</i>	bluegill	37.9	0.68	26	2.0 °C hour <sup>-1</sup>	Smale & Rabeni (1995a)
<i>Ictalurus punctatus</i>	channel catfish	38	0.39	20	0.3 °C min <sup>-1</sup>	Watenpaugh et al. (1995)
<i>Lepomis macrochirus</i>	bluegill	38	-	36	1.0 °C day <sup>-1</sup>	Cherry et al. (1977)
<i>Micropterus punctulatus</i>	spotted bass	38	-	36	1.0 °C day <sup>-1</sup>	Cherry et al. (1977)
<i>Amerius melas</i>	black bullhead	38.1	0.39	26	2.0 °C hour <sup>-1</sup>	Smale & Rabeni (1995a)
<i>Cyprinella lutrensis</i>	red shiner	38.1	0.42	26	2.0 °C hour <sup>-1</sup>	Smale & Rabeni (1995a)
<i>Fundulus olivaceus</i>	blackstripe topminnow	38.3	0.67	26	2.0 °C hour <sup>-1</sup>	Smale & Rabeni (1995a)
<i>Micropterus salmoides</i>	largemouth bass	38.5	0.34	30	0.3 °C min <sup>-1</sup>	Currie et al. (1998)
<i>Ictalurus punctatus</i>	channel catfish	38.7	0.36	25	0.3 °C min <sup>-1</sup>	Currie et al. (1998)
<i>Carpio cyprinus</i>	common carp	38.8	0.8	24	0.5-0.8 °C min <sup>-1</sup>	Mundahl (1990)
<i>Fundulus notatus</i>	blackspotted topminnow	38.8	0.59	26	2.0 °C hour <sup>-1</sup>	Smale & Rabeni (1995a)
<i>Ictalurus punctatus</i>	channel catfish	39.2	0.58	28	1.0 °C min <sup>-1</sup>	Cheetham et al. (1976)
<i>Micropterus salmoides</i>	largemouth bass	39.2	0.64	32	1.0 °C day <sup>-1</sup>	Fields et al. (1987)
<i>Micropterus salmoides</i>	largemouth bass	40.1	1.33	28	1.0 °C min <sup>-1</sup>	Smith & Scott (1975)
<i>Ictalurus punctatus</i>	channel catfish	40.3	0.29	30	0.3 °C min <sup>-1</sup>	Currie et al. (1998)



<i>Micropterus salmoides</i>	largemouth bass	40.9	0.4	32	0.2 °C min <sup>-1</sup>	Fields et al. (1987)
<i>Ictalurus punctatus</i>	channel catfish	41	0.31	32	1.0 °C min <sup>-1</sup>	Cheetham et al. (1976)
<i>Micropterus salmoides</i>	largemouth bass	41.8	0.38	32	0.2 °C min <sup>-1</sup>	Fields et al. (1987)

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Table 4-4.—Goodness-of-fit testing (i.e., root-mean-squared error RMSE and coefficient-of-determination  $r^2$ ) for SSTEMP models. Model fits for both mean daily temperatures and maximum-daily temperatures are included along with sample sizes.

Stream	n	Mean temp.		Max temp.	
		RMSE	$r^2$	RMSE	$r^2$
Flint Creek	6	0.833	0.98	2.3603	0.83
Barren Fork Creek	8	0.6709	0.99	2.561	0.97
Illinois River	6	0.5498	0.97	0.7083	0.94

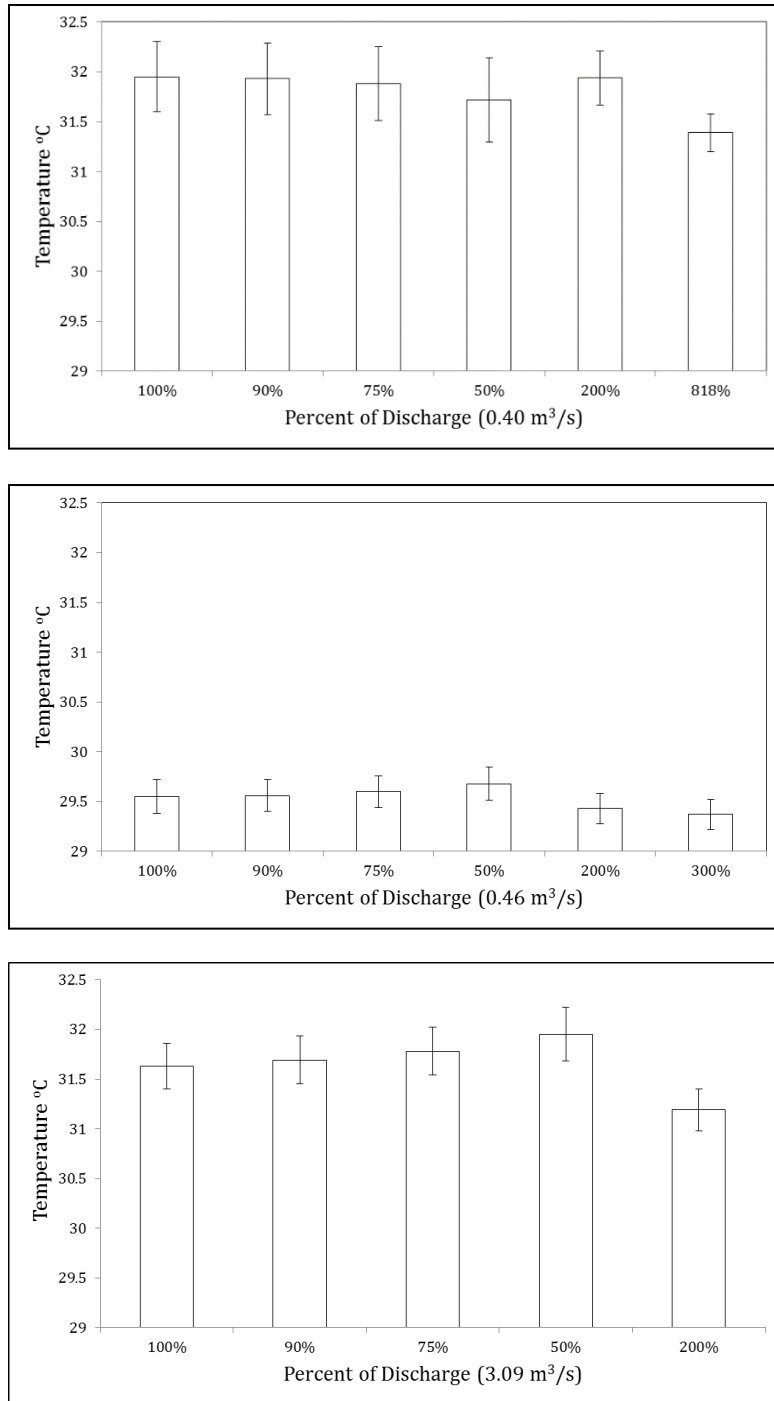


Figure 4-2.—Predicted mean maximum water temperatures (° C; ± 95% C.I.) in Barren Fork Creek (upper panel), Flint Creek (middle panel) and the Illinois River (lower panel) at mean July-August discharge (100%), hypothetical reduced discharges (e.g., 90%, 75%, 50% of the mean), and a hypothetical increases (200%, 300%, 818%). Values of percentages are provided in the methods section.

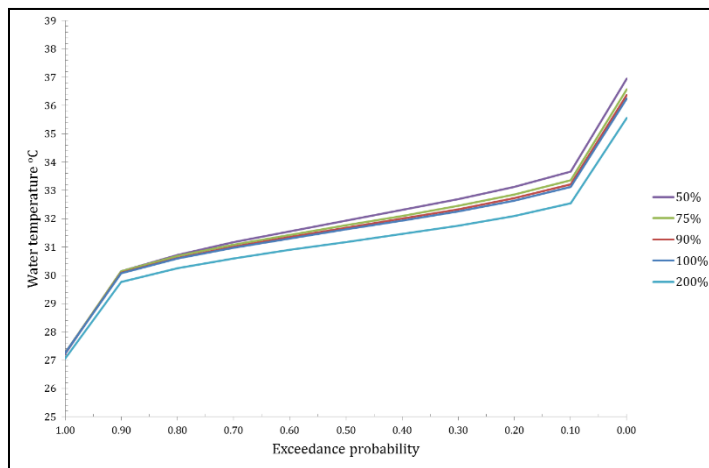
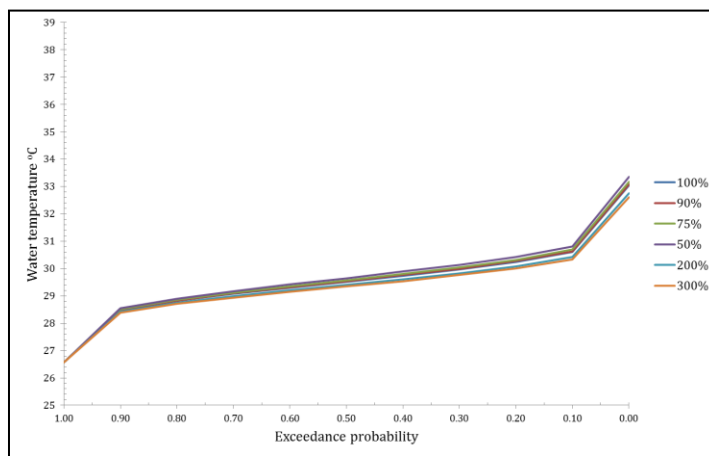
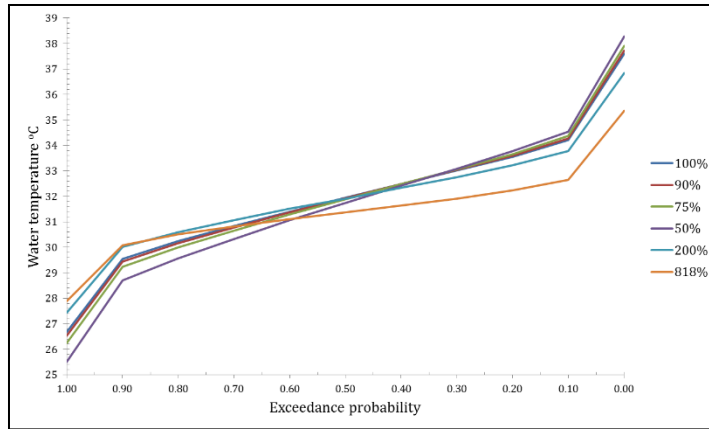


Figure 4-3.—Exceedance probabilities of maximum-water temperatures under different discharge scenarios for conditions present during July – August 2012 in Barren Fork Creek (upper panel), Flint Creek (middle panel) and the Illinois River (lower panel). Hypothetical discharges are the same as provided in Figure 4-2.

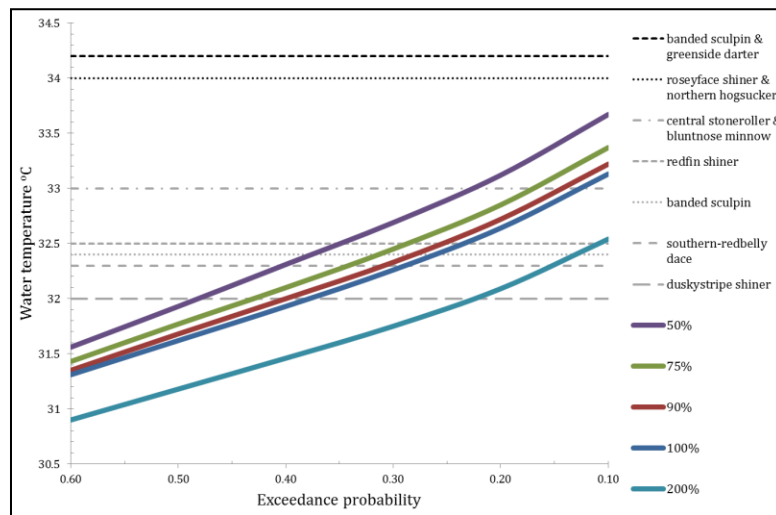
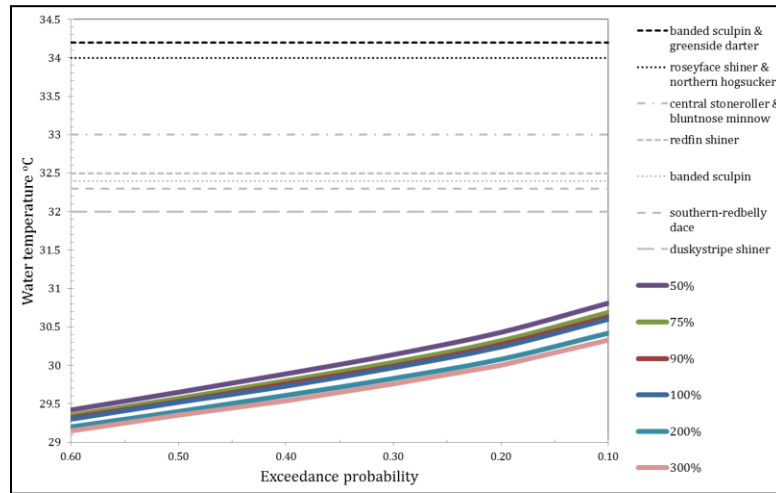
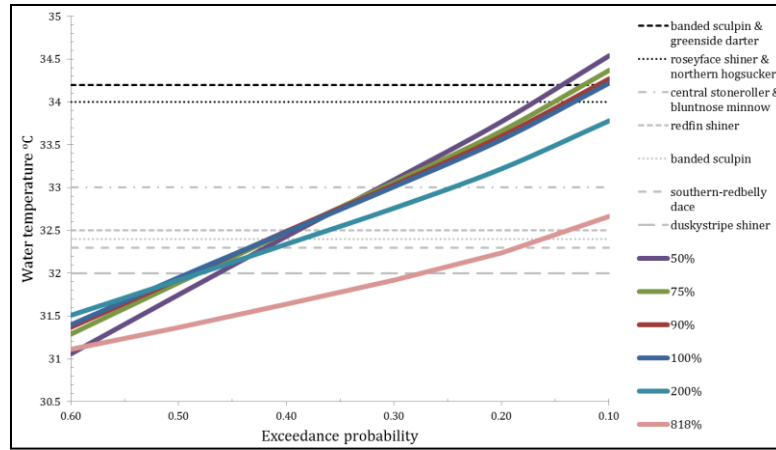


Figure 4-4.—Discharge scenarios with the probability to exceed critical thermal maxima for fishes in Barren Fork Creek (upper panel), Flint Creek (middle panel) and the Illinois River (lower panel). Hypothetical discharges are the same as provided in Figure 4-2.



VITA

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