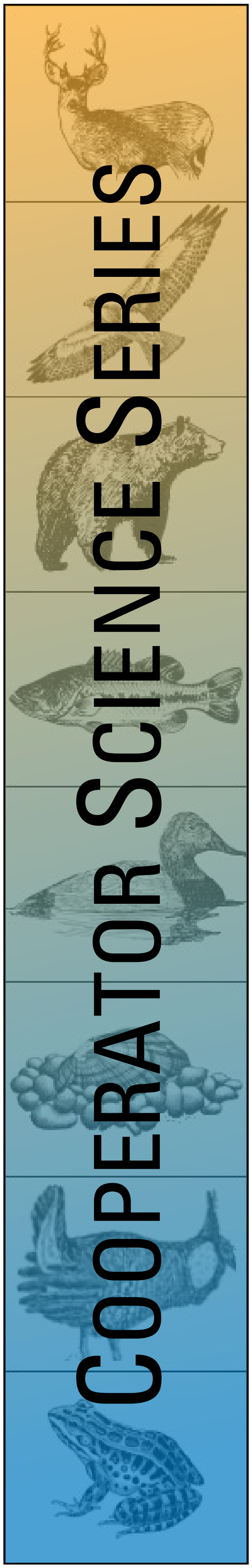


U.S. Fish and Wildlife Service

Habitat Use And Movement By Fishes Using Shallow-Water Habitats In Three Scenic Rivers In Oklahoma

Shannon K. Brewer[1]
Chris Musselman[2]

[1] U.S. Geological Survey, Oklahoma Cooperative Fish and Wildlife Research Unit, Oklahoma State University, Stillwater, OK
[2] Oklahoma Cooperative Fish and Wildlife Research Unit, Oklahoma State University, Stillwater, OK

Cooperator Science Series # 110-2014

About the Cooperator Science Series:

The Cooperator Science Series was initiated in 2013. Its purpose is to facilitate the archiving and retrieval of research project reports resulting primarily from investigations supported by the U.S. Fish and Wildlife Service (FWS), particularly the Wildlife and Sport Fish Restoration Program. The online format was selected to provide immediate access to science reports for FWS, state and tribal management agencies, the conservation community, and the public at large.

All reports in this series have been subjected to a peer review process consistent with the agencies and entities conducting the research. Authors and/or agencies/institutions providing these reports are solely responsible for their content. The FWS does not provide editorial or technical review of these reports. Comments and other correspondence on reports in this series should be directed to the report authors or agencies/institutions. In most cases, reports published in this series are preliminary to publication, in the current or revised format, in peer reviewed scientific literature. Results and interpretation of data contained within reports may be revised following further peer review or availability of additional data and/or analyses prior to publication in the scientific literature.

The Cooperator Science Series is supported and maintained by the FWS, National Conservation Training Center at Shepherdstown, WV. The series is sequentially numbered with the publication year appended for reference and started with Report No. 101-2013. Various other numbering systems have been used by the FWS for similar, but now discontinued report series. Starting with No. 101 for the current series is intended to avoid any confusion with earlier report numbers.

Contractual References:

This document fulfills reporting requirement for U.S. Fish and Wildlife Agreement Number F11AF00016 (F-84-R-1). Previously published documents that partially fulfilled any portion of this contract are referenced within, when applicable. (USGS IPDS #: IP-059620)

Recommended citation:

Brewer, S. K. and C. Mussellman. 2014. Habitat Use And Movement By Fishes Using Shallow-Water Habitats In Three Scenic Rivers In Oklahoma. U.S. Department of Interior, Fish and Wildlife Service, Cooperator Science Series FWS/CSS-110-2014, Washington, D.C.

For additional copies or information, contact:

Shannon Brewer
U.S. Geological Survey
Oklahoma Cooperative Fish and Wildlife Research Unit
Oklahoma State University
404 Life Sciences West
Stillwater, Oklahoma 74078
Phone: (405) 744-6342
E-mail: shannon.brewer@okstate.edu

HABITAT USE AND MOVEMENT BY FISHES USING SHALLOW-WATER HABITATS IN THREE SCENIC RIVERS IN OKLAHOMA (FWS AGREEMENT F11AF00016 (F-84-R-1)

Final report, August 2014

Shannon K. Brewer
U.S. Geological Survey, Oklahoma Cooperative Fish and Wildlife Research Unit
Oklahoma State University
007 Agriculture Hall
Stillwater, Oklahoma 74078
phone: 405-744-9841 | fax: 405-744-5006
e-mail: shannon.brewer@okstate.edu

Chris Musselman
Oklahoma Cooperative Fish and Wildlife Research Unit
Oklahoma State University
007 Agriculture Hall
Stillwater, Oklahoma 74078
phone: 405-744-9841 | fax: 405-744-5006
e-mail: chris.musselman@okstate.edu

EXECUTIVE SUMMARY

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. It is well recognized that stream drying disproportionately affects the shallow-water habitat availability in streams. The two objectives of this study were: 1) determine diel habitat use during baseflow conditions by fishes using shallow-water habitats of three scenic rivers in Oklahoma, and 2) assess movement by a subset of stream fishes and relate those movements to environmental parameters. Habitats most susceptible to loss of area varied by stream but collectively, riffles, runs, and backwater habitats experienced the greatest loss of area. Some linear declines in habitat were observed with some habitat losing up to 60% area over measured discharge conditions. The most shallow-water channel units (riffles, runs, vegetated edgewaters) structured much of the fish assemblage in Barren Fork Creek, particularly benthic fishes. Likewise, benthic fishes in Flint Creek and the Illinois River were positively associated with shallow channel units, but more so higher-velocity habitats. These assemblages had the addition of other fluvial specialists (e.g., cardinal shiner). Diel shifts in habitat use occurred in all streams suggesting movement between channel units to be an important component of the shallow-water fish assemblage. Maximum mean daily water temperatures were: 31.63 ˚C and 29.55 ˚C for the Illinois River and Flint Creek, respectively. A 50% modeled reduction in discharge resulted in a 0.32 ˚C and 0.13 ˚C decrease in maximum water temperature in each of the two streams. Temperature modeling of Barren Fork Creek was difficult to reconcile likely because of a breakdown in the stream width-discharge equation at 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, appears to offer thermal refugia for many species, rarely exceeding published CTM values. Continuous recaptures over about 50 days of four PIT-tagged species in Flint Creek were analyzed using a multistate model in MARK. Survival probability of cardinal shiner and orangethroat darter was related to daily discharge. More importantly, transition probability was also related to daily discharge for three species (cardinal shiner, orangethroat darter, and slender madtom). Transition probability was near zero at approximately 15-20 cfs for several species suggesting movements between channel units were inhibited below this level. Management can use the range of flows

provided to prevent habitat loss for targeted species or habitats. A follow-up analysis identifying the natural frequency of these flow events would be beneficial and allow a means for targeting natural variation in minimum flows. Critical low flow where some fish species are restricted to particular habitats appears to be approximately 20 cfs.

Background

Climate and hydrology are ultimate landscape factors that have substantial influence on stream temperatures, and habitat quality and quantity. Human-induced climate change contributes to a warmer climate with more frequent and intense droughts (IPCC 2007). Rising air temperatures are correlated with rising stream temperatures (Mohseni & Stefan 1999). Increased drought increases water withdrawals from streams for human consumption, sanitation, and irrigation. Lower discharge decreases the buffering capacity of streams to changing temperatures, increases the likelihood of high water temperatures (Sinokrot & Gulliver 2000) and decreases the habitat available to aquatic organisms (Aarts et al. 2004). As water levels decline in streams, water temperatures become more sensitive to climatic conditions (Webb et al. 2003) thereby exacerbating already stressful conditions. The predicted climatic and hydrologic conditions suggest water temperatures in the central United States will continue to rise in the future while some regions will experience more intense drought (e.g., James River Basin, Wu et al. 2012). Understanding the relationship between temperature, discharge, and habitat availability and connectivity is an important first step for incorporating scientific recommendations into environmental-flow decisions associated with a changing climate.

Anthropogenic alterations to flow patterns 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 reduction in species fitness, invasion by exotic species, and extirpation of native species (Bunn & Arthington 2002; Poff & Zimmerman 2010). Unfortunately, flow alteration—a consequence of the ubiquitous use of dams, channels, water withdrawals, and levees to leverage freshwater resources—is 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 (Vorosmarty 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 flows, legally-recognized goals for stream flows, are important for conserving stream biota against current and future threats to lotic environments. Current agricultural, industrial, commercial, and municipal needs already push the limits of available freshwater (Petts 1998; Jackson et al. 2001) and climate change will only serve to exacerbate anthropogenic variations in the natural flow regime (Vorosmarty 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-flow recommendations would benefit from consideration of both water quantity and quality. Water quality (e.g., temperature) governs the water available for human consumption, recreation, and controls overall stream function and the biological integrity of the stream (Karr & Dudley 1981). Properly 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) that society depends upon (Karr & Dudley 1981; Postel et al. 2006; Loomis et al. 2000). Environmental-flow recommendations strive 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 recommendations may not have the desired outcome.

Biological information is needed to determine essential habitats for fishes and identify a threshold for fitness consequences during low flows. We know stream drying disproportionately affects the availability of shallow-water habitats and increases competition or displaces 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 are using shallow-water habitats and at what time to accurately assess essential habitats. For example, juvenile fishes use shallow-water habitats (Schlosser 1987) so we can anticipate lost discharge will 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 life stage. Minimum flows without supporting biological information risk not meeting the needs of aquatic biota.

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-water habitats are essential habitats for the production of macroinvertebrate prey items, spawning and rearing, and the completion of the life histories of many stream fishes (Lobb III and Orth 1991, Aadland 1993, Brewer et al. 2006). For example, invertebrate-prey production in riffles supplements entire downstream reaches (Rabeni & Minshall 1977) suggesting connectivity of habitats is also important. Riffles are habitats with clean substrates and high dissolved oxygen essential for egg survival (Berkman & Rabeni 1987). Other shallow-water habitats, such as edgewaters and backwaters, act as important predator refugia for juvenile fishes (Schlosser 1987). Unfortunately, stream drying is anticipated to disproportionately affect the availability of shallow-water habitats which makes them the most vulnerable habitats to water withdrawals. The importance of connectivity between shallow-water habitats and adjacent habitats would also benefit instream-flow management.

New technologies allow us to supplement available-habitat measures to include information on connectivity of habitats in making instream flow decisions. Historically, obtaining real-time information on small-bodied fish movement and survival was impossible—the technology simply was not available. Recent advances in tag technology (e.g., very small PIT tags) now allows researchers to obtain these data. Assumptions of a relationship between available habitat, connectivity, and fish fitness can now be empirically supported for the stream fish assemblage. Further, we can assess one of the weaknesses of many hydraulic environmental flow methods, functional connectivity.

Objectives:

The project had two objectives:

1) Determine diel habitat use by fishes using shallow-water habitats in the upper Illinois River, Barren Fork Creek, and Flint Creek including "species of conservation need".

2) Determine movement rates by a subset of fishes (from Objective 1), and relate movement to environmental parameters (i.e., discharge, temperature).

Summary of Progress:

OBJECTIVE 1. Determine diel habitat use by fishes using shallow-water habitats in the upper Illinois River, Barren Fork Creek, and Flint Creek including “species of conservation need”

Approach

Study Area

The study catchment is located in the Ozark Highlands and part of the Boston Mountains ecoregions in northeastern Oklahoma, USA. All Ozark Highlands and Boston Mountains streams drain into the Arkansas River, the second largest tributary to the Mississippi River. Mean annual precipitation ranges 100-130 cm and mean annual temperature ranges 13 to 16 ^oC (Adamski et al. 1995). Lithology of the Ozark Highlands is mostly limestone and dolostone with interbedded chert, whereas the Boston Mountains is primarily sandstone and shale (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. 2011). Streams in this area are typical of the Ozark border region with relatively clear-flowing waters and cobble-gravel substrates.

Study sites were located on the upper Illinois River and 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 6th order stream (with a drainage area of 2484 km^2 (959 mi^2) and mean annual discharge of 26.8 m^3/s (946 f^3/s). Flint Creek is a 3rd order tributary with a drainage area of 285 km^2 (110 mi^2) and mean annual discharge of 3.3 m^3/s (118 f^3/s). Barren Fork Creek is a 4th order stream with a drainage area of 936 km^2 (307 mi^2) with a mean annual discharge of 9.3 m^3/s (329 f^3/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). We classified channel units, relevant to stream-fish populations in Ozark streams (Peterson & Rabeni 2001c), following descriptions of Jacobson and Rabeni (1993). Our approach condensed channel units into five major categories (i.e., riffles, runs, pools, backwaters, and vegetated edgewaters) because of their greater availability within the study area. Channel units were selected haphazard through each study reach.

Microhabitat characteristics (i.e., depth, velocity, substrate composition) were measured in each channel unit to verify appropriate classification. We measured depth (0.01 m) with a top-set wading rod and average water-column velocity (0.60 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^2 polyvinyl chloride PVC frame was used to delineate a specific area in each channel unit and we visually estimated the percent of each of the following substrate-size classes (diameter mm): silt (0-0.5), sand (0.5-2), gravel (2-20), cobble (20-200), boulder (> 200), bedrock, and organic material (e.g., aquatic plants, algae, leaves, and woody debris).

We measured discharge at each site and on each sampling occasion using the velocity-area method (Gordon 2004). Measured discharges were checked against the nearest stream gage data for Flint Creek (USGS gage #11110103), Barren Fork Creek (USGS gage #11110103), and the upper Illinois River (USGS gage #07196500) to ensure accuracy of our measurements (Figure 1).

Wetted-area data

We 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 upper Illinois River (upstream of Lake Tenkiller). 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). We used surface area instead of wetted perimeter to get more complete and detailed information on available habitat of individual channel units.

Sample discharges needed for wetted-area analyses were estimated from annual flow-duration curves created in 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.). We only considered flows $\leq 20\%$ exceedance probability in the upper Illinois River because the river is large and flows $> 20\%$ are unsafe for wading. Further, the aim of the project was to locate breakpoints in habitat loss under lower-flow conditions. At selected exceedance probabilities, a range of ± 10% discharge was created to allow for flexibility in field sampling.

We 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 global positioning system (DGPS; Trimble model Juno SB; accuracy ± 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 a base station 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 ($n = 50$) at each site occurred from May 2012 to May 2013 based on discharge (Table 1). Some data points were omitted from the analyses because of obvious discrepancies caused by GPS error. 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 came from USGS gage data. Wetted-area-discharge curves with broken-line regressions and breakpoint estimates for each channel unit are provided in Appendix A.

Habitat use by fishes

Diel habitat use by fishes in Flint Creek, Barren Fork Creek, and the upper Illinois River was evaluated by sampling shallow-water channel units during summer low-flow conditions in 2011 and 2012. Some of the fish sampling in 2011 was completed by the Oklahoma Water Resources Board and all 2012 data were collected by Oklahoma State University. We sampled fishes using a 1-m^2 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 and Rabeni 2001a, Rettig 2003). This gear is most efficient for Cyprinidae, Cottidae, and Percidae but less efficient for Ictaluridae (Peterson & Rabeni 2001a). The quadrat sampler is an effective gear for sampling many small-bodied fishes, 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^2 netted frame by quickly lowering and securing the quadrat sampler to the stream bottom. Trapped fish were "swept" into an attached bag via natural or artificial created downstream velocity while simultaneously disturbing the substrate to dislodge fishes that were stuck within the substrate. 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. 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).

Study sites were sampled for fishes during the day and at night. Day sampling occurred during daylight hours (approximately 0700 to 1800) and night sampling occurred from 2200 to 0400 but only at three downstream study reaches. Night sampling followed the same methods described above, but because sampling conditions were more difficult, we 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). We first created graphical representations of wetted surface area to discharge to visually estimate the probable breakpoints for each channel unit. We used initial visual breakpoint estimates as starting values to iteratively fit broken-line models until model convergence (Muggeo 2008). Significance 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 to address variability inherit in environmental data. Mean breakpoints were then calculated for each channel unit group by study reach and stream because channel-unit geomorphology influences wetted-area analyses (Reinfelds et al. 2004). We did not average breakpoints across all channel units combined because it resulted in a loss of useful information. If the goal is to prevent major loss of habitats, the value for discharge at the breakpoint is the suggested minimum discharge (Robbins et al. 2006).

Fish-community data were analyzed using canonical correspondence analysis (CCA) in CANOCO 4.5 to relate channel unit to fish-community abundance. 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). 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 (2001c) (Figure 2). Runs and pools were relatively deep channel units and riffles, backwaters, and vegetated edgewaters 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 comprised small substrates and vegetated edgewaters that were dominated by organic material (e.g., aquatic macrophytes; Table 3).

Wetted-area breakpoints

Flint Creek, the smallest of the three streams, had a mean breakpoint (± 90% CL) of 2.12 m^3s^{-1} ± 0.74 based on riffle, run, and backwater channel units (Table 4; see also Appendix A). As expected, riffles experienced a significant loss of available area at the highest discharges (2.66 m^3s^{-1} ± 0.83), followed by runs (1.88 m^3s^{-1} ± 0.76), and backwaters (1.23 m^3s^{-1} ± 0.38). Pools had no significant breakpoints ($P > 0.5$), suggesting no major decline in area over the measured discharges. Available pool habitat was best described by mean area. Mean- and median-annual discharges for Flint Creek (based on available U.S. Geological Survey gage data) were 3.29 m^3/s (155% of breakpoint) and 1.59 m^3/s (75% of breakpoint), respectively.

Barren Fork Creek had a mean breakpoint (± 90% CL) at 7.39 m^3s^{-1} ± 4.12 based on upstream (8.42 m^3s^{-1} ± 5.13) and downstream (6.06 m^3s^{-1} ± 2.82) study reaches (Table 5; see also Appendix A). Backwater habitat experienced a significant loss of available area at the highest discharges (9.87 m^3s^{-1} ± 5.87), followed by runs (6.94 m^3s^{-1} ± 3.48), and riffles (6.71 m^3s^{-1} ± 3.92). In a single backwater, the relationship of wetted area-discharge was linear ($r^2 = 0.98$) from the maximum sampled discharge of 21.24 m^3s^{-1} to 0.51 m^3s^{-1} (see Appendix A). This backwater lost 44% of maximum measured habitat at the mean breakpoint for backwater channel units in Barren Fork Creek. Some pools had no significant relationships between area and discharge ($P > 0.5$) and were best described by their means (Table 5) whereas two additional pools showed linear ($r^2 = 0.90$; $r^2 = 0.77$) relationships between area and discharge. However, the linear relationship also suggested that no more than 25% of maximum wetted area was lost over the measured discharges. Mean and median annual discharges for Barren Fork Creek (based on available U.S. Geological Survey gage data) were 9.27 m^3s^{-1} and 3.57 m^3s^{-1}, respectively.

The Illinois River had a mean breakpoint (± 90% CL) of 10.36 m^3s^{-1} ± 5.09 based on the average of upstream (11.05 m^3s^{-1} ± 4.09) and downstream (7.67 m^3s^{-1} ± 2.27) reaches (Table 6; see also Appendix A). Significant ($P < 0.5$) breakpoints for backwaters occurred at 11.59 m^3s^{-1} ± 5.40, riffles at 9.99 m^3s^{-1} ± 2.80 m^3/s, and a single run at 8.53 m^3s^{-1} ± 1.42. Interestingly, this system had pools ($n = 2$) with significant breakpoints (5.72 m^3s^{-1} ± 2.14), but these were excluded from the overall mean breakpoint estimates because these channel units retained the most area compared to all other habitats. Wetted-area discharge relationships that were linear with no significant breakpoints included: runs, pools, backwaters, and a riffle (Table 6). Linear relationships indicated runs lost 20%, 26 %, 38%, and 43% of available habitat, respectively ($r^2 = 0.60$; $r^2 = 0.93$; $r^2 = 0.92$; $r^2 = 0.89$) (see Appendix A). One pool ($r^2 = 0.78$) lost 27% of available habitat at the minimum measured flow of 3.40 m^3s^{-1} and the other pool was best described by mean area (Appendix A). Linear relationships between backwater habitat and discharge suggest these habitats had the greatest slopes and therefore lost the greatest percentage of available habitat (60% and 49%). The single riffle with a linear relationship between area and discharge ($r^2 = 0.96$) lost 21% of available habitat at the mean breakpoint for riffles, and never lost more than 27% at lowest measured discharge of 3.40 m^3s^{-1} (Appendix A). Mean and median discharges for the upper-Illinois River were 26.80 m^3s^{-1} and 12.20 m^3s^{-1}, respectively.

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 & 2012 (Table 7). Most abundant species (89% of catch) were: slender madtom (1,258), Ozark minnow (914), mosquito fish (900), central stoneroller (894), orangethroat darter (634), cardinal shiner (601), and banded darter (560). The remaining 27 species comprised the remaining 11% of the total catch (Table 7). A majority of the catch was adult fishes (79%).

As anticipated due to stream size, species diversity was greatest in the Illinois River (29 species) compared to Barren Fork Creek (23 species) and Flint Creek (18 species); however, overall fish densities were greater in Barren Fork Creek (7.3 fish/m^2) (Table 8). Increased diversity in the Illinois River was due to the occurrences of more minnows (e.g., bigeye chub, creek chub, and wedgespot shiner), suckers (e.g., black redhorse and white sucker), high-velocity species (e.g., gravel chub and logperch), and some sportfish (e.g., bluegill and channel catfish).

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 8). Barren Fork Creek had the only occurrences of suckermouth minnow and yellow bullhead in our samples; however, overall densities were dominated by slender madtom (3.0 fish/m^2). No other species in any stream had densities greater than 1.4 fish/m^2. The less diverse Flint Creek had unique occurrences in our samples including northern studfish and southern redbelly dace. The most abundant species in Flint Creek were mosquito fish, slender madtom, and orangethroat darter (Table 7).

Species diversity and fish densities varied by channel unit (Table 8). For example, pools were the most diverse channel unit (29 fish species), but vegetated edgewaters had the highest fish densities (9.4 fish/m^2. Vegetated edgewaters had densities 1.5 times greater than that of riffles (6.3 fish/m^2), backwaters (5.4 fish/m^2) and pools (5.2 fish/m^2), and twice that of runs (4.3 fish/m^2). Riffles had the second highest densities, but were the least diverse (20 species, same as runs), than vegetated edgewaters (24 species).

Samples taken during each diel period showed similar overall fish densities by species (with few exceptions), but diversity varied by period. Day sampling resulted in higher diversity (32 species) than night sampling (23 species); however, overall fish densities were similar (6.1 fish/m^2 for day and 5.8 fish/m^2 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^2 to 0.3 fish/m^2 at night. Slender madtom densities increased at night by approximately 0.8 fish/m^2. Central stoneroller, sunburst darter, and 10 others also had slightly higher densities at night. We anticipate these density differences to be related to activity level of the fishes (e.g., madtoms more active at night) and thus, more or less susceptible to catch by our gear.

Ordination results for both periods in Barren Fork Creek were significant for the first and all canonical axes (Table 9). All axes combined for the night period explained nearly twice the variation (48%) as the day period (24%). In both periods, the first two axes explained the most variation in species dispersion and represented the strongest environmental gradients. The CCA based on daytime collections (hereafter referred to as ‘day CCA’) indicated the first environmental axis was positively related to the most shallow channel units: riffles, vegetated edgewaters and runs (Table 10). The same axis was negatively related to slackwater channel units: pools (r = -0.6) and backwaters (r = -0.4). Conversely, the second axis appeared to be primarily related to the dominance of macrophyte cover (i.e., 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., only positively related to pools).

Species in Barren Fork Creek were distributed along these environmental gradients and subsequently associated with particular channel units (Figure 3). During the day, the first environmental axis was positively associated with benthic species (e.g., adult banded sculpin and adult banded darter); whereas negatively associated with slackwater species (e.g., all ages of mosquito fish) and minnows (e.g., adult Ozark minnows and adult carmine shiners). 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; 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 habitats from backwaters during the day to pools at night. ‘No fish’ samples were strongly associated with pools during the day and backwaters at night.

The CCA for Flint Creek daytime sampling was significant for the first and all canonical axes, whereas results from nighttime sampling indicated all but the first axis was significant (Table 11). All axes combined to explain 23% of the variation for day sampling and 39% for nighttime samples. The first axis of the daytime CCA was positively associated with higher-velocity channel units: riffles (r = 0.8) and runs (r = 0.3; Table 10). Conversely, the same axis was negatively associated with slackwater habitats. The second day axis appeared to be a depth gradient showing positive associations with deeper-water channel units. We used the night CCA to interpret overall patterns of species and channel-units associations, but we did not interpret individual night axes because of lack of significance.

For daytime collections, Flint Creek had 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) positively associated with the first environmental axis (i.e., riffles and runs; Figure 3). Adult sunburst darter and all ages of mosquito fish 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) and young-of-year cardinal shiner. Some species appeared to use habitat more generally during the day (e.g., adult Ozark minnow and adult orangethroat darter) but shifted to strong channel-unit associations along the second axis at night. Additional day to night shifts were observed in adult sunburst darters from slackwater habitats to a strong association with pools.

The Illinois River results for the first and all axes were significant for both periods (Table 12). All axes combined explained 37% of the total variation at night—more than twice that of day (15%). For daytime samples, environmental gradient appeared to be related to water velocity and was positively correlated with riffles and runs (Table 10). The same axis was negatively correlated with slackwater habitats. Similarly, for nighttime samples, the first axis appeared to be positively related to higher water velocities. The second axis was positively correlated with more open habitats.

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 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 mosquito fish 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 they were not observed at night. The second night axis revealed shifts in habitat use for adult cardinal shiner away from general habitats 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 our gear bias (toward shallower water).

OBJECTIVE 2.-Determine movement rates by a subset of fishes (from Objective 1), and relate movement to environmental parameters (i.e., discharge, temperature)

Approach

Study Area

Summer stream temperatures were modeled at hypothetically lowered discharges for three 1-km reaches on Flint Creek, Barren Fork Creek, and the upper-Illinois River. Flint Creek is a 3rd order tributary with a drainage area of 285 km^2 (110 mi^2) with mean annual discharge of 3.29 m^3/s (116 f^3/s). Barren Fork Creek is a 4th order stream with a drainage area of 936 km^2 (307 mi^2) with a mean annual discharge of 9.27 m^3/s (327 f^3/s). The upper Illinois River is a 6th order stream with a drainage area of 2484 km^2 (959 mi^2) with mean annual discharge of 26.8 m^3/s (946 f^3/s). Study sites were chosen haphazardly based on representativeness of the reach to the entire system. Study reaches were located near Kansas, Oklahoma (OK) for the upper-Illinois River and Flint Creek, and near Baron, OK for Barren Fork Creek.

Fish movement (transition probability) and survival related to discharge was estimated on Flint Creek. A 1-km section of stream, near Kansas, Oklahoma, containing a complex of riffle, run, pool, and backwater channel units was designated as the study reach. The mark-recapture experiments lasted from August 8th to September 25th in 2011 and from June 21st to August 9th in 2012. In 2011, air temperatures (mean ± S.D.) were 23.1 ± 5.6 °C, peaked at 37.8 °C, and went above 37.2 °C (99 °F) only once. Total rainfall was 207.3 mm with a daily maximum of 65.5 mm falling on August 12th. Stream discharge ranged 0.8-6.3 m^3/s, with a mean (± S.D.) of 1.3 ± 1.0 m^3/s. Conversely, summer 2012 was exceptional hot and dry. Over the 50 days in June – August 2012, air temperatures were 29.4 ± 2.6 °C, peaked at 42.2 °C, and went above 37.2 °C 26 times. There was only 26.2 mm of rain in 2012 and 63% (16.5 mm) of that came on the first day of our tagging study. Year two stream discharge was consistently low ranging 0.3-.07 m^3/s, with a mean of 0.4 (± 0.1 m^3/s). Discharge at Flint Creek over the study period was greater in 2011 than 2012 (Figure 4). During 2011, there was one peak in discharge during the start of the tagging season. In 2012, discharge conditions were greatly reduced and relatively constant over the study period.

Water-temperature modeling

Summer stream water temperatures were modeled at hypothetically lowered discharges using the U.S. Geological Survey Stream Segment Temperature Model Version 2.0 (SSTEMP) (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 (elevations, distances, and stream widths) was measured using a rangefinder and GPS (Bartholow 1989). Stream shading was estimated from measured stream characteristics (mean width, azimuth, latitude, and topographic altitude) and riparian vegetation characteristics (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 j/m2/s/C was used for our scenarios. Maximum stream temperatures were estimated from models based on measured stream conditions in summer 2012 (a drought year). Model fit was assessed using a coefficient of determination (r^2) and root-mean-squared error (RMSE).

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 13). We also modeled scenarios of increased discharge (e.g., 200% or higher) to assess the feasibility of lowering stream temperatures via flow management (Table 13). We also modeled the overall mean breakpoint estimates that were provided in the results of

Objective 1. We 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). We 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 ± 10% and time of year (i.e., day length) was set to August 1st.

Model results were compared to published critical thermal maximum (CTM) data for warmwater fishes (literature compiled in 2011; Table 14) to determine if lethal thermal thresholds will be exceeded by the discharge scenarios that we used. CTM is the accepted method for measuring temperature tolerance in fishes (Lutterschmidt & Hutchison 1997). During CTM studies, the water temperature increases at a fast enough rate (1°C per min - 1°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 1997a). 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 °C were excluded from the meta-analysis, as these stream temperatures are considerably lower than expected summer values in our study systems. For those species with multiple published CTMs, the lowest CTM with the highest acclimation temperature was used for our comparison.

PIT-tag retention

Tag loss and impacts of tagging on fish fitness (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) recommends PIT tags be evaluated for each species prior to conducting field studies to avoid violating mark-recapture assumptions. This is especially an important consideration for juveniles and other small-bodied fishes where negative tag effects are likely (Prentice et al. 1990b).

To address these potential issues, we conducted laboratory experiments to assess the fitness consequences of intraperitoneal-placed PIT tags in six fish species (cardinal shiner, central stoneroller, greenside darter, orangethroat darter, slender madtom, and juvenile smallmouth bass, Table 15). Each fish (a minimum of 55 mm TL) was injected with a 12-mm PIT tag and held in a 10-gallon aquarium with a control fish of the same species (no tag) for 38-45 days. Because of their larger size, smallmouth bass (mean TL >125 mm) were injected with a 23-mm PIT tag and held in 2400-L tanks in groups of ten (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 ($\alpha = 0.05$) differences 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, we also provide Pearson's correlation coefficient (r) as an estimate of effect size (Field et al. 2012).

<u>Fish sampling and tagging</u>

Mark-recapture efforts focused on four species that occupied shallow-water channel units. Stream fishes were captured using multiple-pass seining techniques. Sampling with a low-mortality gear reduced extraneous factors that may have influenced survival of tagged fishes. Seine methods varied depending on channel-unit characteristics (i.e., low velocity and deep waters with a typical seine haul and swift waters by kick seine; Bonnar et al. 2009). Each channel unit was sampled until 20 fish were captured (the number of fish it took roughly 30 min to tag), so the maximum holding time of untagged fish was < 1 h. Steps were taken to reduce fish stress during sampling by limiting scale abrasion and exposure to air. 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.

Stream fishes ≥ 55-mm TL were PIT tagged using techniques to minimize stress during the tagging process (Table 16). Fish were transferred from holding cages to an anesthetic bath (2.5 mL MS-222 stock solution/liter water) for five minutes or until fish lost 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 (mm), and tagged. All fish were handled with wet gauze dipped in a slime-coat protectant to limit loss of slime coat and scales. We avoided double-handling fish because of cumulative stress (Barton et al. 1986). We used a 12 gauge needle to inject 12-mm HDX tags (Oregon RFID, Portland, OR) into fish < 120 mm. The angle of the needle was 45^o above the belly of the fish and inserted anteriorly into the peritoneum following methods from Prentice et al. (1990). 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 > 120 mm were surgically implanted with 23-mm HDX tags using a scalpel and finger pressure to gently place the tag. 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 on the downstream end. Fish were allowed to recover for a minimum of 24 h (Hauer and Lamberti 2006). Fish were then checked for mortalities, normal respiration, and normal swimming prior to release into the channel unit of origin.

Tagged fish recaptures

A series of long-range receivers with multiple antennas was used as remote-monitoring stations to passively detect fish movements among channel units. Antennas were constructed with 10 – 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 (i.e., shallow riffle-run interface) and were wired to nearby battery-powered receivers in groups of three to four. Additionally, one receiver and a single-antenna station were located at the downstream extent of the study reach for a total of eight antennas. 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 (i.e., length and depth) of the antenna, the size of the PIT tag (i.e., 12 mm or 23 mm), and the angle of the fish passing through the antenna, but ranged 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, issues with stream shade and overcast conditions interfered with solar recharge and reduced maintenance power to receivers for 11 days in August 2011, 5 days in September 2011, and 6 days in August 2012; however, days with missing data were addressed in the analyses.

Analysis

Fish-capture histories were analyzed using the multistate model in Program MARK (White and 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 2014). 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 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. We 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 approximate 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 the multistate models to estimate transition probabilities between habitats. All individual recaptures were summed over 24 hours and given a single capture history: captured in state A, captured in state B, or not captured. Fish that moved outside of their original channel unit (i.e., the capture and release location) at any point were given a state of B for that day. Any fish that was continuously recaptured at a single antenna for more than 48 hours was removed from the analyses (likely to be a tag loss or mortality). 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 overall model complexity (i.e., the number of estimable parameters) which can quickly grow beyond available computing power for multistate models.

Candidate models for apparent survival (S), recapture probability (p), and transition probability (Psi) were built based on a few variables of interest to maintain simplicity. We 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, we also included a model covariate for "effort" in our 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 17).

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., c-hat) goodness-of-fit approach in U-CARE (Cooch 2006). This test measured for and corrected model overdispersion or "noise" present in the data. Typically, overdispersion is caused by too many model parameters (Cooch 2006); however, our models only contained a few variables so any overdispersion was most likely caused by violating model assumptions (see results). If the most general model adequately fit the data, reasonable values for c-hat (e.g., ≤ 3.0) were used to adjust model AICc scores for each model set. We followed a model averaging approach as outlined by Burnham and Anderson (2002) to eliminate ambiguity when selecting the best model based on AIC ranks. Weighted models (i.e., DeltaQAICc < 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 movements to discharge. Additionally, the overall importance of the covariables was determined by summing support over all weighted models (Burnham and Anderson 2002).

Results

Water temperature modeling

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 upper-Illinois River (Table 18). Accuracy of maximum temperatures models was also excellent for the upper-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 (100%, 3.09 m^3/s). Reducing discharges by 10%, 25%, and 50% increased maximum water temperatures by 0.06 °C, 0.15 °C, and 0.32 °C (Figure 5). Likewise, reduced flows slightly increased the probability of higher water temperatures (Figure 6). For example, a 50% decrease in summer discharge increased the probability of exceeding 33.0 °C from 0.11 to 0.24. Flint Creek followed a similar pattern as the upper Illinois River, except maximum water temperatures were much cooler. Maximum water temperatures were: 29.55 at mean daily discharge (100%, 0.46 m^3/s), and at 10%, 25%, and 50% discharge reductions increased maximum temperatures by only 0.01 °C, 0.05 °C, and 0.13 °C (Figure 4). Most importantly, under all considered discharges, the probability of maximum temperatures exceeding 31 °C (an upper thermal limit for many fish species) was low (< 0.10) (Figure 7). Overall variation in temperatures under multiple flow scenarios was small. For example, at an exceedance probability of 0.10, maximum temperatures ranged from 30.33 – 30.81 (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 mean daily discharge (0.40 m^3/s). Unexpectedly, reductions to 90%, 75%, and 50% of discharge *decreased* maximum water temperatures by 0.02 ^{o}C, 0.07 ^{o}C, and 0.23 ^{o}C (Figure 5). This indicates lower discharges have the unrealistic potential for reducing maximum water temperatures (e.g., exceedance probabilities that intersected below 32 ^{o}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 upper 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 ^{o}C from 0.13 to 0.18. Flows increased to 200% of the mean for July and August (0.80 m^3/s) had only a 0.01 ^{o}C decrease in maximum water temperature and a probability of exceeding 34.0 ^{o}C near 0.10. However, at 3.27 m^3/s (our extreme increase of 818% of the mean annual discharge and the lower limit of our breakpoint estimate from Objective 1), we predicted a decrease 0.55 ^{o}C in maximum water temperature and had a very small probability of exceeding 34.0 ^{o}C (< 0.05).

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 6 and Figure 7). For example, at 100% discharge (mean daily July-August discharge), there was a probability of 0.39 of exceeding CTM for duskystripe shiner (sister species to cardinal shiner), a probability of 0.31 – 0.25 of exceeding CTM for southern redbelly dace, banded sculpin, and redfin shiner, and a probability of 0.11 of exceeding CTM for central stoneroller and bluntnose minnow. A 50% decrease in discharge increased exceedance probabilities to 0.49 for duskystripe shiner; 0.41 – 0.36 for southern redbelly dace, banded sculpin, and redfin shiner; and 0.25 for central stoneroller and bluntnose minnow. At this extreme low discharge, CTM for roseyface shiner, northern hogsucker, greenside darter, and banded sculpin were approximated 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 7). 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 0.10 at 100% to 0.15 at 50%, a difference of only 0.05. Increased discharge reduced the chance of exceeding temperature thresholds, but all discharges retained potential to exceed CTM. 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 0.10 probability of exceeding CTM for any species even at the lowest flows.

PIT tag retention

Experimental fishes ranged in TL (mm) from 63 – 107 for cardinal shiner *Luxilis cardinalis*, 70 – 126 for central stoneroller *Campostoma annomalum*, 82 – 95 for greenside darter *Etheostoma blennioides*, 56 – 72 for orangethroat darter *Etheostoma spectabile*, 72 – 95 for slender madtom *Noturus exilis*, and 120 – 180 for juvenile-smallmouth bass *Micropterus dolomieu* (Table 15). All species growth data fit the assumption of normality. Differences in growth between trials for orangethroat darter were not significant $t(_{13.418}) = 1.41$, $P = 0.18$ so trials were combined into one analysis.

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 lab study duration (Table 19). 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 19). Low survival by orangethroat darter may be due to improper diet because negative growth was seen for both treatment and control fishes (Figure 8). All mortalities were excluded from growth and retention analyses.

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 8).

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 20). 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 c-hat overdispersion factors were all reasonable (e.g., < 2.0) and well below acceptable thresholds (Table 20); therefore, all model results ranked by QAICc scores were deemed appropriate.

Capture probabilities were adequate for 2011 and 2012 for most of the species included in the analyses. All species in 2012 had mean capture probabilities ≥ 0.25. In 2011, all species except smallmouth bass met the cutoff criteria of 0.25. Mean capture probability for smallmouth bass in 2011 was 0.16.

Cardinal shiner.—Model results for cardinal shiner indicated an effect of discharge on the probability of survival and movements in both years. For 2011 data, the best fit model suggested survival was influenced by discharge, capture probability by effort, and transition probability was constant through time (Table 21). However, this model had only 39% of the Akaike weight and seven other models were also possible (though with less weight, Table 21). Summed weights over model parameters indicated daily discharge had the most support (0.60) as an important predictor of survival (Table 22). Model-averaged results estimated high survival over daily-time steps and was typically above 0.8 in 2011 and above 0.9 in 2012; however, much wider confidence bands are observed as discharge increases above 60 cfs in 2011 (Figure 9). Conversely, transition probability being constant (0.75) was weighed more heavily than the influence of discharge (0.25). In 2012, only two models were weighted, each having support for discharge in predicting survival and transition probability (Table 21). Summed weights showed daily discharge (1.0) as the better predictor of both survival and transition probability (Table 22). Capture probability had more support as a constant term in our 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 ~.04 above 20 cfs (Figure 10). Transition probabilities were near zero below 15 cfs suggesting movement was limited below this flow.

Central stoneroller.—Model results for central stoneroller showed relatively constant survival and transition probabilities regardless of sample year. Each year had eight weighted models and no model was given overwhelming support. In 2011, the best fit model suggested survival was constant, capture probability was influence by effort, and transition probability was constant through time (Table 23). 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 11). Mean probability of survival was higher in 2012 at ~ 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 24). Mean transition probability was relatively low in both sample years (~ 0.25-0.30) with much greater variation in confidence limits in 2011 (Figure 12). Capture probability appeared to be relatively constant. Again in 2012, no model showed substantial support as being the best model (Table 23). Across all models, survival and transition probability were relatively constant over time (Table 24). There was a slight positive trend in transition probability above 25 cfs. Capture probability was influenced by effort (antennas operating at any one time).

Orangethroat darter.— Seven models were weighted for orangethroat darter, but the best-fit model had more support (58% Akaike weight) than the next best model (20%, Table 25). The best-fit model for 2012 data supported discharge as an important predictor of survival and transition probability and constant capture probability. Summed weights over model parameters indicated discharge (0.82) was the best predictor of survival (Table 26). Survival in 2012 was relatively high throughout the study (> 0.80, Figure 13). Discharge was strongly related to transition probability of the darter (0.95). Transition probability increased above 20 cfs suggesting more movement by the species when discharge increased above this level (Figure 14). Capture probability was relatively constant throughout 2012 (above 0.30).

Slender madtom.—Model results suggest discharge may relate to probability of survival, but there is full support for the relationship between discharge and transition probability (Table 27). Only two models were weighted for slender madtom. The best-fit model had 61% Akaike weight and 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; however, did not support discharge as a predictor of survival (Table 27). Summed weights over model parameters support constant survival (0.61, Table 28). Survival in 2012 was consistently high across measures discharges (>0.9, Figure 13). Discharge, however, was fully supported as an important predictor of transition probability (Table 28). Transition probability and associated variation increased with increasing discharge, particularly at discharges > 20 cfs (Figure 14). As expected effort was the best predictor of recapture probability with decreasing recaptures when there were fewer active antennas (similar to findings for cardinal shiner and central stoneroller).

Smallmouth bass.—Model results for smallmouth bass showed little relationship between daily discharge and survival or transition probability in either sample year (Table 29). Each year had eight weighted models and the best-fit model in each year suggested model parameters were constant. In 2011, the model with constant parameters had 38% of the Akaike weight and in 2012 it was supported with 40% of the weight. Summed weights over model parameters indicated very similar results for both 2011 and 2012 (Table 30). Constant parameter estimates had roughly 70%-75% of the model weight. Mean survival probability was high in both sample years though in 2011, there was much greater variation in the parameter estimate when discharge exceeded approximately 60 cfs (Figure 15). Discharge influenced transition probability slightly more in 2011 (0.31) than in 2012 (0.26), but the range of discharge encountered by fish during the 2011 study period was much greater. In 2012, transition probability appeared stable over time (~ 0.3, Figure 16), whereas there was a very slight linear increase in 2011 and much wider confidence bands above 60 cfs (Figure 16). Recapture probabilities were less than 0.2 in 2011, but about 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 and Recommendations

Minimum-flow theory assumes a relationship between stream discharge, habitat availability, and fish fitness (Annear & Conder 1984; Jowett 1997). If flow is the major determinant of physical habitat in aquatic ecosystems (Jowett 1997; Bunn and Arthington 2002) and habitat is a major determinant of biotic composition (Schlosser 1982), then the relationship between flow and available habitat may be 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 1984; Gippel et al. 1998). For example, the wetted-area approach used in this study measured the relationship between stream discharge and channel unit area. We have shown these measurements to be stream specific and therefore only apply to the locations where data were collected and associated reaches with similar stream geometries. Unfortunately, fish abundance is not necessarily linearly related to available habitat so it is still difficult to predict the biotic responses to flow (Jowet 1997). However, evidence suggests that even species occupying the deeper, slackwater environments of streams benefit from the persistence of shallow, high-velocity channel units throughout the reach (e.g., smallmouth bass, Sowa and Rabeni 1995; Johnson et al. 2009; Brewer 2013).

Connectivity among habitats is an important component of fish fitness in lotic ecosystems. Riffle drying reduces downstream drift, decreases water quality, and eliminates movements into and across these habitats (Mathews & Marsh-Mathews 2003; Arthington 2012). Lack of connection between habitats may affect growth of drift-feeding fishes (e.g., rainbow trout *Oncorhynchus mykiss*, Rimmer 1985; smallmouth bass, Paragamian and 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-Mathews & Mathews 2010) and may result in a decline in fish diversity (Perkin et al. *In Press*). Increases in mortality have been observed during seasonal stream drying for nongame (e.g., Ouachita madtom *Noturus lachneri*, Gagen et al. 1998) and game fish (e.g., smallmouth bass, Hafs et al. 2010). 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 for important life-history events (Jaeger et al. 2014). Transition probabilities shown in the current study indicate movements are very likely restricted at flows < ~20 cfs in Flint Creek. Based on channel morphology, we would expect restricted movements to occur above this threshold in the Barren Fork Creek and Illinois River.

Stream fishes use specific habitats that vary with season and diel period. This study was conducted during the warmwater, low-flow season. However, we show fish species are associated with particular channel units but that diel shifts in habitat use are not usual. The most interesting diel shift was associated with movements between backwater and other channel units. In Barren Fork Creek and Flint Creek, adult sunburst darter moved from pools to backwater habitats at night. Adult orangethroat darter also made this shift in the Illinois River. In addition, several juveniles moved to backwater habitats at night (e.g., central stoneroller and slender madtom). Conversely, adult longear sunfish in the Illinois River appeared to use higher-velocity water at night compared to the day. Backwaters were previously not considered important habitats because they were off channel and did not affect stream connectivity (Gippel & Stewardson 1998). However, backwaters act as rearing habitats for young-of-year fishes avoiding high flows (Moore & Gregory 1988a) and escaping predation (Schlosser 1987; Peterson & Rabeni 2001). Loss of a stream's backwaters has led to decreased young-of-year survival (Falke et al. 2010) and eventual extirpation from stream reaches (Moore & Gregory 1988b). Connection of backwater habitats within these systems appears important during juvenile rearing. Documenting fish shifting among channel units suggests transitioning between habitats is important to the ecology of these species.

A weakness of the wetted-area method is that is does not relate habitat connectivity (i.e., functional connectivity) to usable habitat for stream fishes (Annear et al. 2004). Wetted-area methods determine the discharge needed to maintain water in channel units above an obvious break, but depending on the character of the stream, this level of discharge may be unsuitable for some species. The breakpoints we identified in this study are associated with large confidence intervals suggesting high variation in habitat loss of individual channel units. The lower end of these confidence intervals paired with a frequency analysis of historical occurrence of these flow events would provide a reasonable negotiation point for minimum flows. Additionally, this would allow managers to establish a range of variation in low flows that would benefit fish populations. Our results also suggest habitat connectivity is lost near 15-20 cfs for several small-bodied fishes. Reduced habitat connectivity and loss of available habitat may confer a disadvantage to overall fish fitness; however, when these events occur (i.e., how many days of disconnected habitat results in decreased fitness) and the magnitude of effect on fish are often unknown.

Breakpoint estimates, in many cases, are higher than would be expected for a minimal flow designation. In particular, Flint Creek and Barren Fork Creek are more sensitive to changes in discharge. Bias in the wetted-area approach can result in higher breakpoint estimates for small streams and lower estimates for large-stable streams (Annear & Conder 1984; Jowett 1997). Channel braiding, as occurs on Barren Fork Creek, can also introduce an upward bias because flow is dissipated laterally (Jowett 1997). Most likely, the upward bias is due to the breakpoint definition: 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 situation to which native species have evolved (Ward 1983; Resh et al. 1988; Lake 2003). Our breakpoint estimates are best viewed as the ideal situation and minimum flow expectations would gain most by focusing between the lower confidence band of these estimates and the discharge where connectivity is drastically reduced. Further, these estimates would be useful for examining necessary flows during other seasons when maximizing habitat would be biologically beneficial. 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 and Spacie 1984) and to maintain adequate connectivity for downstream migrations in late fall (Grabowski 2006).

The results found in this study indicate daily fish survival is related to discharge for many species, but cardinal shiner and orangethroat darter were most strongly influenced. Increased discharge decreased the probability of survival at daily time steps. Similar increases in mortality with discharges have been documented in stream fish which were hypothesized to be vulnerable to high-flow events during particular seasons (Grossman et al. 1998). This appears to be the case in this study, as spikes in discharge following rain events resulted in a corresponding dip in survival probability for cardinal shiner and orangethroat darter. Short-term increases in flow magnitude could potentially displace fish, separate individuals from a school, increase access to refuge habitats for predators, or cause physical stress from high-velocity flows or shifting substrates. However, daily time steps are too short to detect the long-term implications of reduced flows on fish fitness (Marsh-Mathews & Mathews 2010). We expect survival to increase with 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 (Gagen et al. 1998), loss of drying riffle habitats lowered survival of bigeye shiner (Hodges & Magoulick 2011), and stream drying appeared to lower the survival of smallmouth bass (Hafs et al. 2010). Finally, alteration of the flow regime does not directly equate to discharge conditions a fish would experience in one year, particularly at a daily time step. The effects of flow regime alterations during the low-flow period can only be viewed by examining data over a much more coarse time step. However, the power of this analysis comes from identifying transition probabilities so managers know at what discharge fish become restricted to certain habitats.

Stream connectivity has implications for stream temperatures at multiple spatial scales. Continuous discharge links instream habitats, 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 ^{o}C cooler into the Illinois River during summer 2012. Lost discharge reduced horizontal connectivity at the channel-unit scale 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 become isolated in these habitats when waters recede and consequently have no mode of escape if water temperatures turn fatal. Under drought conditions of summer 2012, measured temperatures in an isolated backwater exceeded the highest predicted maximum water temperatures (e.g., 34.1 ^{o}C) in the upper 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. *In Press*). 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 (Zekster et al. 2005). Out 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 substantial areas of 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.

Overexposure of stream fishes to high temperatures may reduce the fitness of populations. Critical thermal maxima (i.e., survival temperatures) are conservative thermal thresholds that are typically much higher than preferred temperatures (i.e., those optimal for growth and reproduction) (Hasnain et al. 2013). 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 1995). 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. Shifts in community composition and abundance are likely due to thermal stress at multiple temporal scales (Mohseni et al. 2003). Finally, exposure to high temperature combined with other stressors may result in increased mortality (Coutant 1976). It appears that fishes in the Illinois River and Barren Fork Creek are most susceptible to thermal stress and therefore, the addition of other stressors may exacerbate associated fitness consequences.

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.

Supplementing minimum streamflow estimates with direct measures of fish fitness would help make recommended flows biologically relevant and likely improve the success of stream conservation and management efforts. In particular, assessing survival over a greater temporal period (weeks or months rather than daily) would provide a more complete understanding of how changes in overall flow patterns affect fish survival. Survival is a straightforward way to measure fish fitness, however, connectivity has also been shown to be a significant factor controlling species persistence through time (Fullerton et al. 2010). For example, isolating stream-fish communities has reduced fish biodiversity (Perkin and Gido 2012) and made species more susceptible to extinction (Fagan et al. 2002). Consideration of stream connectivity at multiple spatial scales (e.g., channel unit and tributary to tributary) would be beneficial if protection of fishes from water withdrawals and altered flow dynamics is the goal. This would allow for persistence of thermal refugia that might be important to the persistence of fish populations in the future (e.g., 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. Alternatively, linking a population model to flow scenarios would be beneficial to improving our understanding of the effects of flow alteration on fishes.

Acknowledgements

This research was supported by the Oklahoma Department of Wildlife Conservation and was conducted under the auspices of the Oklahoma State University Animal Care and Use Committee (ACUP AG1110 and AG118). Special thanks to Nicole Farless and Jahna Hill for

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

I. Recommendations

Management recommendations are provided above in the discussion and recommendations section.

II. Significant Deviations

None to report

III. Equipment

None purchased

IV. Prepared by:

Shannon Brewer and Chris Musselman

Oklahoma Cooperative Fish and Wildlife Research Unit,
Oklahoma State University Stillwater, Oklahoma

Date: 28 August 2014

Approved by:____
Fisheries Division Administration
Oklahoma Department of Wildlife Conservation

____
Andrea Crews, Federal Aid Coordinator
Oklahoma Department of Wildlife Conservation

References

Aadland, L. P. (1993). "Stream habitat types: their fish assemblages and relationship to flow." North American Journal of Fisheries Management 13(4): 790-806.

Aarts, B. G. W., F. W. B. Van Den Brink, et al. (2004). "Habitat loss as the main cause of the slow recovery of fish faunas of regulated large rivers in Europe: the transversal floodplain gradient." River Research and Applications **20**(1): 3-23.

Acreman, M. and M. J. Dunbar (2004). "Defining environmental river flow requirements- a review." Hydrology and Earth System Sciences 8(5): 861-876.

Adamski, J. C., et al. (1995). "Environmental and hydrologic setting of the Ozark Plateaus study unit, Arkansas, Kansas, Missouri, and Oklahoma." U.S. Geological Survey Water-Resources Investigation Report: 94-4022.

Annear, T. C. and A. L. Conder (1984). "Relative bias of several fisheries instream flow methods." North American Journal of Fisheries Management 4 (4B): 531-539.

Annear, T., et al. (2004). Instream Flows for Riverine Resource Stewardship - Revised Edition. Cheyenne, WY, Instream Flow Council.

Arthington, A. (2012). Environmental Flows-Saving Rivers in the Third Millennium, University of California Press.

Arthington, A. H., et al. (2006). "The Challenge of Providing Environmental Flow Rules to Sustain River Ecosystems." Ecological Applications 16(4): 1311-1318.

Bain, M. B., et al. (1985). "Quantifying stream substrate for habitat analysis studies." North American Journal of Fisheries Management 5(3B): 499-500.

Bartholow, J. (1999). "Stream Segment Temperature Model (SSTEMP), Version 1.0. 0, Revised December 1999. Computer program and user documentation available at http://www.mesc.usgs.gov/rsm/rsm_download.html#TEMP."

Bartholow, J. (2000). "Estimating cumulative effects of clearcutting on stream temperatures." Rivers 7(4): 284-297.

Bartholow, J. M. (1989). "Stream temperature investigations: field and analytical methods." Biological report.

Bartholow, J. M. (2010). "Constructing an Interdisciplinary Flow Regime Recommendation1." JAWRA Journal of the American Water Resources Association 46(5): 892-906.

Barton, B. A., et al. (1986). "Multiple acute disturbances evoke cumulative physiological stress responses in juvenile chinook salmon." Transactions of the American Fisheries Society 115(2): 245-251.

Becker, C.D. and R.G. Genoway. (1979). Evaluation of the critical thermal maximum for determining thermal tolerance of freshwater fish. Environmental Biology of Fishes 4: 245 – 256.

Berkman, H. E. and C. F. Rabeni (1987). "Effect of siltation on stream fish communities." Environmental Biology of Fishes 18(4): 285-294.

Bolland, J. D., et al. (2009). "Evaluation of VIE and PIT tagging methods for juvenile cyprinid fishes." Journal of Applied Ichthyology 25(4): 381-386.

Bonar, S. A., et al. (2009). Standard methods for sampling North American freshwater fishes, American Fisheries Society Bethesda, Maryland.

Brewer, S. K. (2013). Channel unit use by Smallmouth Bass: do land-use constraints or quantity of habitat matter? North American Journal of Fisheries Management 33(2):351–358.

Brewer, S. K., et al. (2006). "Spawning habitat associations and selection by fishes in a flow-regulated prairie river." Transactions of the American Fisheries Society 135(3): 763-778.

Brock, F. V., et al. (1995). "The Oklahoma Mesonet: a technical overview." Journal of Atmospheric and Oceanic Technology 12(1): 5-19.

Brownie, C., J. E. Hines, et al. (1993). "Capture-recapture studies for multiple strata including non-Markovian transitions." Biometrics 49(4): 1173-1187.

Brunke, M. and T. Gonser (1997). "The ecological significance of exchange processes between rivers and groundwater." Freshwater Biology 37(1): 1-33.

Bunn, S. E. and A. H. Arthington (2002). "Basic Principles and Ecological Consequences of Altered Flow Regimes for Aquatic Biodiversity." Environmental Management 30(4): 492-507.

Burnham, K. P., et al. (1987). "Design and analysis methods for fish survival experiments based on release-recapture." American Fisheries Society Monograph 5. American Fisheries Society, Bethesda Maryland. 1987. 737.

Chen, C. W., et al. (2011). "A comparison of estimators for regression models with change points." Statistics and Computing 21(3): 395-414.

Cooch, E. and G. White (2006). "Program MARK: a gentle introduction." Colorado State University Fort Collins USA: 120-138.

Cooch E., and G. C. White (2014). Program MARK: A Gentle Introduction, 12th edn. [Cited 4 May 2014.] Available from URL:http://www.phidot.org/software/mark/docs/book/

Coutant, C. (1976). "Thermal effects on fish ecology." Encyclopedia of environmental engineering 2.

Cucherousset, J., et al. (2005). "A continuum of life history tactics in a brown trout (Salmo trutta) population." Canadian Journal of Fisheries and Aquatic Sciences 62(7): 1600-1610.

Cucherousset, J., et al. (2005). "The Use of Two New Portable 12-mm PIT Tag Detectors to Track Small Fish in Shallow Streams." North American Journal of Fisheries Management 25(1): 270-274.

Curry, K. D. and A. Spacie (1984). "Differential use of stream habitat by spawning catostomids." American Midland Naturalist 111(2): 267-279.

Dauwalter, D. C., et al. (2006). "Mapping stream habitats with a global positioning system: accuracy, precision, and comparison with traditional methods." Environmental Management 37(2): 271-280.

Dudgeon, D., et al. (2006). "Freshwater biodiversity: importance, threats, status and conservation challenges." Biological Reviews 81(2): 163-182.

Fagan, W. F. (2002). "Connectivity, fragmentation, and extinction risk in dendritic metapopulations." Ecology 83(12): 3243-3249.

Falke, J. A., K. R. Bestgen, et al. (2010). "Streamflow Reductions and Habitat Drying Affect Growth, Survival, and Recruitment of Brassy Minnow across a Great Plains Riverscape." Transactions of the American Fisheries Society 139(5): 1566-1583.

Fenneman, N. M. (1938). Physiography of eastern United States. New York, New York, McGraw-Hill Book Co., Inc.

Field, A., et al. (2012). Discovering Statistics Using R, SAGE Publications.

Fox, G. et al. (2011). Evaluation of a stream-aquifer analysis test for deriving reach-scale streambed conductance. Transactions of the American Society of Agricultural and Biological Engineers 54(2): 473-479.

Fox, G.A. (2004). Evaluation of a stream aquifer analysis test using analytical solutions and field data. Journal of the American Water Resources Association 40(3):755-763.

Frissell, C. A., et al. (1986). "A hierarchical framework for stream habitat classification: viewing streams in a watershed context." Environmental Management 10(2): 199-214.

Fullerton, A. H., K. M. Burnett, et al. (2010). "Hydrological connectivity for riverine fish: measurement challenges and research opportunities." Freshwater Biology 55(11): 2215-2237.

Gagen, C. J., R. W. Standage, and J. N. Stoeckel. (1998).Ouachita madtom (*Noturus lachneri*) metapopulation dynamics in intermittent Ouachita Mountain streams. Copeia 1998(4):874–882.

Gippel, C. J. and M. J. Stewardson (1998). "Use of wetted perimeter in defining minimum environmental flows." Regulated Rivers: Research & Management 14(1): 53-67.

Gordon, N. D. (2004). Stream hydrology: an introduction for ecologists, John Wiley & Sons Inc.

Grabowski, T. B. and J. J. Isely (2006). "Seasonal and diel movements and habitat use of robust redhorses in the lower Savannah River, Georgia and South Carolina." Transactions of the American Fisheries Society 135(5): 1145-1155.

Grossman, G. D., et al. (1998). "Assemblage organization in stream fishes: effects of environmental variation and interspecific interactions." Ecological Monographs 68(3): 395-420.

Hafs, A. W., C. J. Gagen, and J. K. Whalen. (2010). Smallmouth Bass summer habitat use, movement, and survival in response to low flow in the Illinois Bayou, Arkansas. North American Journal of Fisheries Management 30(2): 604–612.

Halleraker, J., et al. (2007). "Application of multiscale environmental flow methodologies as tools for optimized management of a Norwegian regulated national salmon watercourse." River Research and Applications 23(5): 493-510.

Harvey, J. (2008). "Analysis and Interpretation of Freshwater Fisheries Data." Freshwater Biology 53(10): 2130-2131.

Hasnain, S. S., et al. (2013). "Phylogeny influences the relationships linking key ecological thermal metrics for North American freshwater fish species." Canadian Journal of Fisheries and Aquatic Sciences 70(7): 964-972

Hauer, F. R. and G. A. Lamberti (2006). Methods in stream ecology, Academic Press.

HDR (2007). WRIA 59 -- Colville River Watershed Plan Instream Flow Study Report, HDR, Inc. Pasco, Washington. April 2007.

Hodges, S. and D. Magoulick (2011). "Refuge habitats for fishes during seasonal drying in an intermittent stream: movement, survival and abundance of three minnow species." Aquatic Sciences - Research Across Boundaries 73(4): 513-522.

Jackson, R. B., et al. (2001). "Water in a Changing World." Ecological Applications 11(4): 1027-1045.

Jaeger, K. L., Olden, J. D., & Pelland, N. A. (2014). “Climate change poised to threaten hydrologic connectivity and endemic fishes in dryland streams”. Proceedings of the National Academy of Sciences, doi: 10.1073/pnas.1320890111.

Johnson, R. L., et al. (2009). Distribution, population characteristics, and physical habitat associations of black bass (*Micropterus*) in the lower Eleven Point River, Arkansas. Southeastern Naturalist 8(4): 653–670.

Jowett, I. G. (1997). "Instream flow methods: a comparison of approaches." Regulated Rivers: Research & Management 13(2): 115-127.

Junk, W. J., P. B. Bayley, et al. (1989). "The flood pulse concept in river-floodplain systems." Canadian Special Publication of Fisheries and Aquatic Sciences 106: 110-127.

Karl, T., et al. (2012). "US temperature and drought: Recent anomalies and trends." Eos, Transactions American Geophysical Union 93(47): 473-474.

Karr, J. R. and D. R. Dudley (1981). "Ecological perspective on water quality goals." Environmental Management 5(1): 55-68.

Laake, J. L. (2013). RMark: An R Interface for Analysis of Capture-Recapture Data with MARK. AFSC Processed Rep 2013-01, 25p. Alaska Fish. Sci. Cent., NOAA, Natl. Mar. Fish. Serv., 7600 Sand Point Way NE, Seattle WA 98115.

Lake, P. (2003). "Ecological effects of perturbation by drought in flowing waters." Freshwater Biology 48(7): 1161-1172.

Leathe, S. A. and F. A. Nelson (1986). A literature evaluation of Montana's wetted perimeter inflection point method for deriving instream flow recommendations, Montana Dept. of Fish, Wildlife and Parks.

Leopold, L. B., et al. (1995). Fluvial processes in geomorphology, Dover Pubns.

Lobb III, M. D. and D. J. Orth (1991). "Habitat use by an assemblage of fish in a large warmwater stream." Transactions of the American Fisheries Society 120(1): 65-78.

Lonzarich, D. G., M. L. Warren Jr, et al. (1998). "Effects of habitat isolation on the recovery of fish assemblages in experimentally defaunated stream pools in Arkansas." Canadian Journal of Fisheries and Aquatic Sciences 55(9): 2141-2149.

Loomis, J., et al. (2000). "Measuring the total economic value of restoring ecosystem services in an impaired river basin: results from a contingent valuation survey." Ecological Economics 33(1): 103-117.

Lutterschmidt, W.I. and V.H. Hutchison. (1997). The critical thermal maximum: history and critique. Canadian Journal of Zoology 75(10): 1561 – 1574.

Lyons, J. (1996). "Patterns in the species composition of fish assemblages among Wisconsin streams." Environmental Biology of Fishes 45(4): 329-341.

Macdonald, J., J. Morrison, et al. (2012). "The efficacy of reservoir flow regulation for cooling migration temperature for sockeye salmon in the Nechako River watershed of British Columbia." North American Journal of Fisheries Management 32(3): 415-427.

Magoulick, D. D. and R. M. Kobza (2003). "The role of refugia for fishes during drought: a review and synthesis." Freshwater Biology 48(7): 1186-1198.

Marsh-Matthews, E. and W. J. Matthews (2010). Proximate and residual effects of exposure to simulated drought on prairie stream fishes. Community ecology of stream fishes: concepts, approaches, and techniques. American Fisheries Society, Symposium.

Matthews, W. J. and E. Marsh Matthews (2003). "Effects of drought on fish across axes of space, time and ecological complexity." Freshwater Biology 48(7): 1232-1253.

Mohseni, O. and H. Stefan (1999). "Stream temperature/air temperature relationship: a physical interpretation." Journal of Hydrology 218(3): 128-141.

Mohseni, O., et al. (2003). "Global warming and potential changes in fish habitat in US streams." Climatic Change 59(3): 389-409.

Moore, K. M. and S. V. Gregory (1988a). "Summer habitat utilization and ecology of cutthroat trout fry (Salmo clarki) in Cascade Mountain streams." Canadian Journal of Fisheries and Aquatic Sciences 45(11): 1921-1930.

Moore, K. M. and S. V. Gregory (1988b). "Response of young-of-the-year cutthroat trout to manipulation of habitat structure in a small stream." Transactions of the American Fisheries Society 117(2): 162-170.

Moyle, P. B., et al. (2003). "Alien fishes in natural streams: fish distribution, assemblage structure, and conservation in the Cosumnes River, California, USA." Environmental Biology of Fishes 68(2): 143-162.

Muggeo, V. M. (2008). "Segmented: an R package to fit regression models with broken-line relationships." Using Sweave with LyX: 20.

Palmer, M. W. (1993). "Putting things in even better order: the advantages of canonical correspondence analysis." Ecology 74(8): 2215-2230.

Paragamian, V. L., and M. J. Wiley. (1987). Effects of variable streamflows on growth of smallmouth bass in the Maquoketa River, Iowa. North American Journal of Fisheries Management 7(3): 357–362

Perkin, J. S. and K. B. Gido (2012). "Fragmentation alters stream fish community structure in dendritic ecological networks." Ecological Applications 22(8): 2176-2187.

Perkin, J., et al. (*In press*). Fragmentation and dewatering transform Great Plains stream fish communities. Ecological Monographs. http://dx.doi.org/10.1890/14-0121.1

Peterson, J. T. and C. F. Rabeni (2001a). "Evaluating the efficiency of a one-square-meter quadrat sampler for riffle-dwelling fish." North American Journal of Fisheries Management 21(1): 76-85.

Peterson, J. T. and C. F. Rabeni (2001b). "The Relation of Fish Assemblages to Channel Units in an Ozark Stream." Transactions of the American Fisheries Society 130(5): 911 - 926.

Poff, N. L. and J. K. H. Zimmerman (2010). "Ecological responses to altered flow regimes: a literature review to inform the science and management of environmental flows." Freshwater Biology 55(1): 194-205.

Poff, N. L. R., et al. (2003). "River flows and water wars: emerging science for environmental decision making." Frontiers in Ecology and the Environment 1(6): 298-306.

Poff, N. L., et al. (1997). "The Natural Flow Regime." BioScience 47(11): 769-784.

Poole, G. C. and C. H. Berman (2001). "An Ecological Perspective on In-Stream Temperature: Natural Heat Dynamics and Mechanisms of Human-Caused Thermal Degradation." Environmental Management 27(6): 787-802.

Postel, S. and B. D. Richter (2003). Rivers for life: managing water for people and nature, Island Press.

Power, G., et al. (1999). "Groundwater and fish—insights from northern North America." Hydrological processes 13(3): 401-422.

Power, M. E., et al. (1995). "Hydraulic Food-Chain Models." BioScience 45(3): 159-167.

Prentice, E. F., et al. (1990). Feasibility of using implantable passive integrated transponder (PIT) tags in salmonids. American Fisheries Society Symposium. 7: 317–322.

Prentice, E. F., et al. (1990). PIT tag monitoring systems for hydroelectric dams and fish hatcheries. American Fisheries Society Symposium. 7: 323-334.

R Core Team, Ed. (2012). R: A language and environment for statistical computing. Vienna, Austria, R Foundation for Statistical Computing.

Rabeni, C. and G. Minshall (1977). "Factors affecting microdistribution of stream benthic insects." Oikos 29(1): 33-43.

Rabeni, C. F. and S. P. Sowa (1996). "Integrating biological realism into habitat restoration and conservation strategies for small streams." Canadian Journal of Fisheries and Aquatic Sciences 53(S1): 252-259.

Reinfelds, I., et al. (2004). "Refinement of the wetted perimeter breakpoint method for setting cease-to-pump limits or minimum environmental flows." River Research and Applications 20(6): 671-685.

Resh, V. H., A. V. Brown, et al. (1988). "The role of disturbance in stream ecology." Journal of the North American Benthological Society 7(4): 433-455.

Rettig, A. V. (2003). Associations between benthic fishes and habitat at multiple spatial scales in headwater streams of the Missouri Ozarks., M.S. Thesis. University of Missouri, Columbia: 189 pp.

Ricciardi, A. and J. B. Rasmussen (1999). "Extinction rates of North American freshwater fauna." Conservation Biology 13(5): 1220-1222.

Richter, B. D. (2010). "Re-thinking environmental flows: from allocations and reserves to sustainability boundaries." River Research and Applications 26(8): 1052-1063.

Richter, B. D., et al. (1996). "A method for assessing hydrologic alteration within ecosystems." Conservation Biology: 1163-1174.

Robbins, K., et al. (2006). "Estimation of nutrient requirements using broken-line regression analysis." Journal of Animal Science 84(13 suppl): E155-E165.

Rosenfeld, J. (2003). "Assessing the habitat requirements of stream fishes: an overview and evaluation of different approaches." Transactions of the American Fisheries Society 132(5): 953-968.

Schlosser, I. J. (1982). "Fish community structure and function along two habitat gradients in a headwater stream." Ecological monographs 52(4): 395-414.

Schlosser, I. J. (1987). "The role of predation in age-and size-related habitat use by stream fishes." Ecology 68(3): 651-659.

Sinokrot, B. A. and J. S. Gulliver (2000). "In-stream flow impact on river water temperatures." Journal of Hydraulic Research **38**(5): 339-349.

Smale, M. A. and C. F. Rabeni (1995a). "Hypoxia and hyperthermia tolerances of headwater stream fishes." Transactions of the American Fisheries Society 124(5): 698-710.

Smale, M. A. and C. F. Rabeni (1995b). "Influences of hypoxia and hyperthermia on fish species composition in headwater streams." Transactions of the American Fisheries Society 124(5): 711-725.

Sowa, S. P., and C. F. Rabeni. (1995). Regional evaluation of the relation of habitat to distribution and abundance of Smallmouth Bass and Largemouth Bass in Missouri streams. Transactions of the American Fisheries Society 124(2): 240–251.

Stanford, J. A., and F. R Hauer. (1992). "Mitigating the impacts of stream and lake regulation in the Flathead River catchment, Montana, USA: An ecosystem perspective." Aquatic Conservation: Marine and Freshwater Ecosystems 2(1): 35-63.

Ter Braak, C. J. (1986). "Canonical correspondence analysis: a new eigenvector technique for multivariate direct gradient analysis." Ecology 67(5): 1167-1179.

Vannote, R. L., et al. (1980). "The river continuum concept." Canadian Journal of Fisheries and Aquatic Sciences 37(1): 130-137.

Vörösmarty, C. J., et al. (2000). "Global water resources: vulnerability from climate change and population growth." Science 289(5477): 284.

Ward, J. (1989). "The four-dimensional nature of lotic ecosystems." Journal of the North American Benthological Society 8(1): 2-8.

Ward, J. and J. Stanford (1983). "Intermediate-disturbance hypothesis: an explanation for biotic diversity patterns in lotic ecosystems." Dynamics of Lotic Systems, Ann Arbor Science, Ann Arbor MI. 1983. 347-356 p, 2 fig, 35 ref.

Webb, B. W., and D. E. Walling. (1997). "Complex summer water temperature behaviour below a UK regulating reservoir." Regulated Rivers: Research & Management 13(5): 463-477.

Webb, B. W., P. D. Clack, et al. (2003). "Water–air temperature relationships in a Devon river system and the role of flow." Hydrological Processes **17**(15): 3069-3084.

Wehrly, K. E., et al. (2007). "Field-based estimates of thermal tolerance limits for trout: incorporating exposure time and temperature fluctuation." Transactions of the American Fisheries Society 136(2): 365-374.

White, G. C. and K. P. Burnham (1999). "Program MARK: survival estimation from populations of marked animals." Bird Study 46(01): 120-139.

Wu, Y., S. Liu, et al. (2012). "Predicting impacts of increased CO2 and climate change on the water cycle and water quality in the semiarid James River Basin of the Midwestern USA." Science of The Total Environment **430**(0): 150-160.

Zeigler, M. P., A. S. Todd, et al. (2012). "Evidence of recent climate change within the historic range of Rio Grande cutthroat trout: implications for management and future persistence." Transactions of the American Fisheries Society 141(4): 1045-1059.

Zektser, S., et al. (2005). "Environmental impacts of groundwater overdraft: selected case studies in the southwestern United States." Environmental Geology 47(3): 396-404.

Zydlewski, B. G., et al. (2001). "Performance of stationary and portable passive transponder detection systems for monitoring of fish movements." Journal of Fish Biology 58(5): 1471-1475.

Table 1.—Number of wetted area-discharge samples taken over the range of flow-exceedance percentages. Each trip included measurements of multiple channel units.

	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
upper-Illinois River	1	2	4	2	4	1	1	-	1	-	16
											50

Table 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 were unavailable. Sources of published data are provided.

Species		Length freq. (TL)	Publication (TL)	Sister species	Publication
Etheostoma zonale	banded darter	30	30.5	-	Pflieger (1975)
Cottus carolinae	banded sculpin	30	27.9	mottled sculpin	Pflieger (1975)
Notropis amblops	bigeye chub	50	50.8	-	Pflieger (1975)
Notropis boops	bigeye shiner	30	30.5	-	Pflieger (1975)
Moxostoma duquesni	black redhorse	90	88.9	-	Pflieger (1975)
Fundulus olivaceus	blackspotted topminnow	30	30.5	plains killifish	Pflieger (1975)
Lepomis macrochirus	bluegill	35	38.1	-	Pflieger (1975)
Labidesthes sicculus	brook silverside	60	63.5	-	Pflieger (1975)
Luxilus cardinalis	cardinal shiner	40	43.2	duskystripe shiner	Pflieger (1975)
Notropis percobromus	carmine shiner	35	40.6	roseyface shiner	Reed (1957)
Campostoma anomalum	central stoneroller	60	58.4	-	Pflieger (1975)
Ictalurus punctatus	channel catfish	66	66	-	Pflieger (1975)
Semotilus atromaculatus	creek chub	65	63.5	-	Pflieger (1975)
Erimystax x-punctatus	gravel chub	60	63.5	-	Pflieger (1975)
Lepomis cyanellus	green sunfish	40	43.2	-	Pflieger (1975)
Etheostoma blenniodes	greenside darter	60	63.5	-	Pflieger (1975)

Percina caprodes	logperch	70	73.7	-	Pflieger (1975)
Lepomis megalotis	longear sunfish	35	33	-	Pflieger (1975)
Gambusia affinis	mosquito fish	15	*variable*	-	Pyke (2005)
Hypentilium nigracans	northern hogsucker	100	86.4	-	Pflieger (1975)
Fundulus catenatus	northern studfish	40	42.3	-	Fisher (1981)
Etheostoma spectabile	orangethroat darter	30	27.9	-	Pflieger (1975)
Notropis nubilus	Ozark minnow	35	30.5	-	Pflieger (1975)
Nocomis asper	redspot chub	50	48.3	hornyhead chub	Pflieger (1975)
Ambloplites rupestris	rock bass	60	40.6	-	Pflieger (1975)
Noturus exilis	slender madtom	40	48.3	-	Pflieger (1975)
Micropterus dolomieu	smallmouth bass	90	88.9	-	Pflieger (1975)
Phoxinus erthrogaster	southern-redbelly dace	30	27.9	-	Pflieger (1975)
Phenacobius mirabilis	suckermouth minnow	40	36	-	Etnier (1993)
Etheostoma mihileze	sunburst darter	35	35	stippled darter	Hotalling (1987)
Lepomis gulosus	warmouth	60	40.6	-	Pflieger (1975)
Notropis greenei	wedgespot shiner	40	45.7	-	Pflieger (1975)
Catostomus commersonii	white sucker	100	96.5	-	Pflieger (1975)
Amerius natalis	yellow bullhead	175	177.8	-	Pflieger (1975)

Table 3—Substrate composition (% mean coverage ± 95% C.I.) by channel unit for summer 2012 habitat-use samples. Bold values are substrates with greater than 10% surface-area coverage per m^2.

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 4.—Breakpoint estimates (90% C.L.) for wetted area-discharge relationships by channel unit in Flint Creek. Breakpoint is discharge when significant habitat area is lost (m^3/s and ft^3/s). Significance of each breakpoint was tested using a Davie's Test. If there was no significant breakpoint, a linear-model fit was determined.

Channel Unit	Breakpoint ± 90% C.I.				Davie's Test	R^2
	m^3/s	±	ft^3/s	±	*p*-value	
Backwater	1.23	0.38	43.44	13.42	<0.001	-
Pool	-	-	-	-	0.410	*mean*
Pool	-	-	-	-	0.640	*mean*
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	-
Reach Mean	*2.12*	*0.74*	*74.92*	*26.08*		
Stream Mean	2.12	0.74	74.92	26.08		

Table 5.—Breakpoint estimates (90% C.L.) for wetted area-discharge relationships by channel unit for Barren Fork Creek. Breakpoint is discharge when significant habitat area is lost (m^3/s and ft^3/s). Significance of each breakpoint was tested using a Davie's Test. If there was no significant breakpoint, a linear-model fit was determined.

Channel Unit	Breakpoint ± 90% C.I.				Davie's Test	R^2
	m^3/s	±	ft^3/s	±	*p*-value	
<u>Lower Reach</u>						
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	*mean*
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	-
Reach Mean	*6.06*	*2.82*	*213.86*	*99.54*		
<u>Upper Reach</u>						
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	*mean*
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	-
Reach Mean	*8.42*	*5.13*	*297.47*	*181.28*		
Stream Mean	7.39	4.12	260.89	145.52		

Table 6.—Breakpoint estimates (90% C.L.) for wetted area-discharge relationships by channel unit for reaches of the upper Illinois River (upper and lower reaches are both above Lake Tenkiller). Significance of each breakpoint was tested using a Davie's Test. If there was no significant breakpoint, a linear model was fitted.

Channel Unit	Breakpoint ± 90% C.I.				Davie's Test	R^2
	m^3/s	±	ft^3/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	*mean*
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
Reach Mean	*7.67*	*5.40*	*271.01*	*190.84*		
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
Reach Mean	*11.05*	*4.09*	*390.35*	*144.38*		

Stream Mean	*10.36*	*5.09*	*365.90*	*179.71*

Table 7.—Total abundances of fishes sampled from Flint Creek and Barren Fork Creek and the upper-Illinois River. Species codes used for CCA are the three-letter abbreviations to the right of the common names.

Species	Species Code	Flint Creek	Barren Fork	Illinois River	Totals
banded darter	bnd	17	33	510	560
banded sculpin	bsc	49	37	9	95
bigeye chub	bgc	-	1	11	12
bigeye shiner	bgs	-	4	2	6
black redhorse	brh	-	-	3	3
blackspotted topminnow	btp	3	5	-	8
bluegill	blg	5	2	12	19
brook silverside	bks	-	-	2	2
cardinal shiner	cds	87	152	362	601
carmine shiner	cms	-	7	148	155
central stoneroller	cst	107	226	561	894
channel catfish	cct	-	-	49	49
creek chub	crk	-	-	6	6
gravel chub	gvc	-	-	17	17
green sunfish	gsf	4	2	1	7
greenside darter	gsd	5	4	34	43
logperch	lgp	-	-	1	1
longear sunfish	lgs	20	2	74	96
mosquito fish	mqf	319	211	370	900
northern hogsucker	nhg	-	5	11	16
northern studfish	nsf	12	-	-	12
orangethroat darter	otd	164	167	303	634
ozark minnow	ozk	55	221	638	914
redspot chub	rdc	19	3	3	25
rock bass	rkb	14	4	6	24

slender madtom	sld	249	746	263	1258
smallmouth bass	smb	-	1	2	3
southern redbelly dace	srd	3	-	-	3
suckermouth minnow	skm	-	1	-	1
sunburst darter	sbd	26	47	16	89
warmouth	war	-	-	1	1
wedgespot shiner	wdg	-	-	18	18
white sucker	wts	-	-	1	1
yellow bullhead	ybh	-	1	-	1
Total		*1,158*	*1,882*	*3,434*	*6,474*

Table 8.—Fish densities (#fish/m^2) by stream and channel unit based on 2011 to 2012 habitat-use data. Empty (no fish) samples and data from fall 2011 are not included, Abbreviated channel units are: BW=backwater and Veg=vegetated edgewater.

Species		n	Densities			Densities				
			Barren Fork	Flint Creek	Illinois River	BW	Pool	Riffle	Run	Veg
Etheostoma zonale	banded darter	560	0.141	0.054	0.724	0.011	0.028	1.194	0.402	0.226
Cottus carolinae	banded sculpin	95	0.136	0.165	0.014	0.016	0.022	0.285	0.042	0.072
Notropis amblops	bigeye chub	12	0.003	-	0.016	0.009	0.031	-	-	-
Notropis boops	bigeye shiner	6	0.018	-	0.004	0.007	0.022	-	-	0.006
Moxostoma duquesni	black redhorse	3	-	-	0.005	-	0.008	0.004	-	-
Fundulus olivaceus	blackspotted topminnow	8	0.022	0.014	-	0.014	0.006	-	0.006	0.025
Lepomis macrochirus	bluegill	19	0.015	0.013	0.034	0.007	0.015	-	-	0.098
Labidesthes sicculus	brook silverside	2	-	-	0.003	-	0.007	-	-	-
Luxilus cardinalis	cardinal shiner	601	0.526	0.277	0.634	0.561	0.447	0.442	0.665	0.410
Notropis percobromus	carmine shiner	155	0.026	-	0.224	0.048	0.290	0.015	0.126	0.055
Campostoma anomalum	central stoneroller	894	0.807	0.395	1.185	0.905	0.708	0.916	0.721	1.124
Ictalurus punctatus	channel catfish	49	-	-	0.110	-	0.012	0.031	0.009	0.206
Semotilus atromaculatus	creek chub	6	-	-	0.015	-	-	-	0.002	0.032
Erimystax x-punctatus	gravel chub	17	-	-	0.022	-	0.009	0.023	0.015	-
Lepomis cyanellus	green sunfish	7	0.011	0.018	0.003	0.005	0.012	-	-	0.030
Etheostoma blenniodes	greenside darter	43	0.017	0.018	0.054	-	0.006	0.115	0.025	0.023
Percina caprodes	logperch	1	-	-	0.003	-	-	0.006	-	-

Lepomis megalotis	longear sunfish	96	0.008	0.091	0.134	0.062	0.148	0.011	0.059	0.165
Gambusia affinis	mosquito fish	900	0.883	1.219	0.802	1.503	0.873	0.012	0.184	2.337
Hypentilium nigracans	northern hogsucker	16	0.022	-	0.023	0.014	0.016	0.009	0.034	0.006
Fundulus catenatus	northern studfish	12	-	0.040	-	0.006	0.012	-	0.028	0.006
Etheostoma spectabile	orangethroat darter	634	0.737	0.551	0.503	0.317	0.535	0.465	0.337	1.278
Notropis nubilus	Ozark minnow	914	0.701	0.163	1.415	0.709	1.530	0.142	0.427	1.664
Nocomis asper	redspot chub	25	0.014	0.089	0.014	-	0.006	-	0.004	0.167
Ambloplites rupestris	rock bass	24	0.021	0.056	0.016	0.007	0.003	0.005	0.010	0.122
Noturus exilis	slender madtom	1258	3.017	0.837	0.444	1.080	0.251	2.638	1.122	1.144
Micropterus dolomieu	smallmouth bass	3	0.005	-	0.005	-	0.006	0.006	-	0.006
Phoxinus erthrogaster	southern redbelly dace	3	-	0.014	-	0.021	-	-	-	-
Phenacobius mirabilis	suckermouth minnow	1	0.005	-	-	-	0.006	-	-	-
Etheostoma mihileze	sunburst darter	89	0.167	0.093	0.022	0.019	0.167	0.011	0.076	0.127
Lepomis gulosus	warmouth	1	-	-	0.003	0.007	-	-	-	-
Notropis greenei	wedgespot shiner	18	-	-	0.036	0.013	0.027	0.008	-	0.039
Catostomus commersonii	white sucker	1	-	-	0.001	-	0.003	-	-	-
Amerius natalis	yellow bullhead	1	0.008	-	-	0.011	-	-	-	-
Total	*34*	*6474*	*7.311*	*4.106*	*6.470*	*5.351*	*5.207*	*6.337*	*4.294*	*9.368*

Table 9.—CCA results from Barren Fork Creek sampling (daytime and nighttime) with channel units as explanatory variables. Eigenvalue for each ordination axis are provided along with total explained variance and tests for significance.

Daytime sampling					
Axes	1	2	3	4	Total inertia
Eigenvalues:	0.221	0.175	0.107	0.037	2.234
Species-environment correlations:	0.842	0.856	0.797	0.654	
Cumulative percentage variance					
of species data:	9.9	17.7	22.5	**24.2**	
of species-environment relation:	40.9	73.3	93.1	100	
Sum of all eigenvalues:					2.234
Sum of all canonical eigenvalues:					0.54

Test of significance of first canonical axis: eigenvalue = 0.221
F-ratio = 2.634
P-value = 0.0038

Test of significance of all canonical axes : Trace = 0.540
F-ratio = 1.914
P-value = 0.0001

Nighttime sampling					
Axes	1	2	3	4	Total inertia
Eigenvalues:	0.416	0.207	0.133	0.044	1.683
Species-environment correlations:	0.965	0.923	0.898	0.879	
Cumulative percentage variance					
of species data:	24.7	37.1	44.9	**47.6**	
of species-environment relation:	52.0	77.9	94.5	100	
Sum of all eigenvalues:					1.683
Sum of all canonical eigenvalues:					0.8

Test of significance of first canonical axis: eigenvalue = 0.416
F-ratio = 3.287
P-value = 0.0001

Test of significance of all canonical axes : Trace = 0.800
F-ratio = 2.267
P-value = 0.0001

Table 10.—Correlation coefficients between channel units and CCA axes (i.e., environmental gradients) for Barren Fork Creek (upper panel), Flint Creek (middle panel), and the upper Illinois River (lower panel).

Channel Units	Day Sampling		Night Sampling	
	Axis 1	Axis 2	Axis 1	Axis 2
Barren Fork Creek				
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
Flint Creek				
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
Illinois River				
Backwater	-0.1186	0.5463	-0.1310	0.0488
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

Table 11.— CCA results from Flint Creek sampling (daytime and nighttime) with channel units as explanatory variables. Eigenvalue for each ordination axis are provided along with total explained variance and tests for significance.

Daytime sampling

Axes	1	2	3	4	Total inertia
Eigenvalues :	0.33	0.156	0.064	0.044	2.539
Species-environment correlations :	0.912	0.836	0.559	0.571	
Cumulative percentage variance					
of species data :	13.0	19.1	21.7	**23.4**	
of species-environment relation:	55.6	81.8	92.7	100	
Sum of all eigenvalues					2.539
Sum of all canonical eigenvalues					0.593

Test of significance of first canonical axis: eigenvalue = 0.330
F-ratio = 3.734
P-value = 0.0001

Test of significance of all canonical axes : Trace = 0.593
F-ratio = 1.906
P-value = 0.0001

Nighttime sampling

Axes	1	2	3	4	Total inertia
Eigenvalues:	0.359	0.163	0.111	0.089	1.853
Species-environment correlations:	0.907	0.933	0.697	0.902	
Cumulative percentage variance					
of species data:	19.4	28.2	34.2	**39.0**	
of species-environment relation:	49.7	72.3	87.7	100	

Sum of all eigenvalues:	1.853
Sum of all canonical eigenvalues:	0.723

Test of significance of first canonical axis: eigenvalue = 0.359
F-ratio = 2.164
P-value = 0.0558

Test of significance of all canonical axes : Trace = 0.723
F-ratio = 1.438
P-value = 0.0346

Table 12.— CCA results from Illinois River sampling (daytime and nighttime) with channel units as explanatory variables. Eigenvalue for each ordination axis are provided along with total explained variance and tests for significance.

Daytime sampling					
Axes	1	2	3	4	Total inertia
Eigenvalues:	0.265	0.167	0.065	0.044	3.635
Species-environment correlations:	0.822	0.821	0.64	0.616	
Cumulative percentage variance					
of species data:	7.3	11.9	13.7	**14.9**	
of species-environment relation:	49.0	79.8	91.8	100	
Sum of all eigenvalues:					3.635
Sum of all canonical eigenvalues:					0.541
Test of significance of first canonical axis: eigenvalue = 0.265					
F-ratio = 4.244					
P-value = 0.0001					
Test of significance of all canonical axes : Trace = 0.541					
F-ratio = 2.359					
P-value = 0.0001					

Nighttime sampling					
Axes	1	2	3	4	Total inertia
Eigenvalues:	0.36	0.194	0.126	0.068	2.038
Species-environment correlations:	0.958	0.833	0.863	0.726	

Cumulative percentage variance					
of species data:	17.7	27.2	33.4	**36.7**	
of species-environment relation:	48.2	74.0	90.9	100	
Sum of all eigenvalues:					2.038
Sum of all canonical eigenvalues:					0.748

Test of significance of first canonical axis: eigenvalue = 0.360
F-ratio = 2.148
P-value = 0.0085

Test of significance of all canonical axes : Trace = 0.748
F-ratio = 1.450
P-value = 0.0170

Table 13. —Range of discharges (m^3/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^3/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.I. for mean breakpoint estimates (mean of all habitats) obtained from objective one

Table 14.—Summary of known critical thermal maxima (CTM) for species found in the Illinois River watershed or related species. The state indicates where experimental fish originated.

	Species	Mean (^{o}C)	SD	Acclimation (^{o}C)	Rate of increase	Publication	State
Notropis pilsbyri	duskystripe shiner	32.0	-	22.5	2.0 ^{o}C day^{-1}	Hickman & Dewey (1973)	-
Phoxinus erthrogaster	southern-redbelly dace	32.3	-	21.5	0.5 ^{o}C min^{-1}	Scott (1987)	NM
Cottus carolinae	banded sculpin	32.4	0.204	20	0.3 ^{o}C min^{-1}	Walsh et al. (1997)	FL/GA
Lythrurus umbratilis	redfin shiner	32.5	-	-	1.0 ^{o}C min^{-1}	Maness & Hutchison (1980)	OK
Cottus carolinae	banded sculpin	32.8	0.16	20	0.3 ^{o}C min^{-1}	Walsh et al. (1997)	FL/GA
Campostoma anomalum	central stoneroller	33.0	-	30	1.0 ^{o}C day^{-1}	Cherry et al. (1977)	VA
Pimephales notatus	bluntnose minnow	33.0	-	30	1.0 ^{o}C day^{-1}	Cherry et al. (1977)	VA
Notropis rubellus	roseyface shiner	34.0	-	30	1.0 ^{o}C day^{-1}	Cherry et al. (1977)	VA
Hypentilium nigracans	hogsucker	34.0	-	33	1.0 ^{o}C day^{-1}	Cherry et al. (1977)	VA
Etheostoma	greenside darter	34.2	0.40	20	1.0 ^{o}C min^{-1}	Hlohowskyj & Wissing	OH

blenniodes						(1984)	
Cottus carolinae	banded sculpin	34.2	0.10	25	0.3 °C min^{-1}	Walsh et al. (1997)	FL/GA
Cottus carolinae	banded sculpin	34.3	0.10	25	0.3 °C min^{-1}	Walsh et al. (1997)	FL/GA
Etheostoma blenniodes	greenside darter	34.9	0.70	20	1.0 °C min^{-1}	Hlohowskyj & Wissing (1984)	OH
Micropterus dolomieu	smallmouth bass	35.0	-	33	1.0 °C day^{-1}	Cherry et al. (1977)	VA
Luxilus zonatus	bleeding shiner	35.3	0.50	26	2.0 °C hour^{-1}	Smale & Rabeni (1995)	MO
Notropis rubellus	rosyface shiner	35.3	0.23	26	2.0 °C hour^{-1}	Smale & Rabeni (1995)	MO
Micropterus salmoides	largemouth bass	35.4	0.47	20	0.3 °C min^{-1}	Currie et al. (1998)	OK
Etheostoma flabellare	fantail darter	35.5	1.1	20	1.0 °C min^{-1}	Hlohowskyj & Wissing (1984)	OH
Ictalurus punctatus	channel catfish	35.5	0.38	20	1.0 °C min^{-1}	Cheetham et al. (1976)	-
Lepomis macrochirus	bluegill	35.5	-	21.5	2.0 °C day^{-1}	Hickman & Dewey (1973)	-
Nocomis biguttatus	hornyhead chub	35.6	0.55	26	2.0 °C hour^{-1}	Smale & Rabeni (1995)	MO
Semotilus	creek chub	35.7	0.44	26	2.0 °C hour^{-1}	Smale & Rabeni (1995)	MO

atromaculatus							
Ictalurus punctatus	channel catfish	35.8	0.47	20	0.15 °C min^{-1}	Bennett et al. (1987)	-
Phoxinus erthrogaster	southern-redbelly dace	35.9	0.42	26	2.0 °C hour^{-1}	Smale & Rabeni (1995)	MO
Etheostoma flabellare	fantail darter	36.0	0.50	20	1.0 °C min^{-1}	Hlohowskyj & Wissing (1984)	OH
Etheostoma flabellare	fantail darter	36.0	0.54	26	2.0 °C hour^{-1}	Smale & Rabeni (1995)	MO
Labidesthes sicculus	brook silversides	36.0	0.44	26	2.0 °C hour^{-1}	Smale & Rabeni (1995)	MO
Lythrurus umbratilis	redfin shiner	36.2	0.54	26	2.0 °C hour^{-1}	Smale & Rabeni (1995)	MO
Notropis nubilus	Ozark minnow	36.2	0.62	26	2.0 °C hour^{-1}	Smale & Rabeni (1995)	MO
Micropterus salmoides	largemouth bass	36.3	0.6	26	2.0 °C hour^{-1}	Smale & Rabeni (1995)	MO
Ictalurus punctatus	channel catfish	36.4	0.25	20	0.3 °C min^{-1}	Currie et al. (1998)	OK
Micropterus salmoides	largemouth bass	36.5	0.51	24	0.2 °C min^{-1}	Fields et al. (1987)	-
Etheostoma spectabile	orangethroat darter	36.5	0.15	26	2.0 °C hour^{-1}	Smale & Rabeni (1995)	MO
Noturus exilis	slender madtom	36.5	0.24	26	2.0 °C hour^{-1}	Smale & Rabeni (1995)	MO

Pimephales notatus	bluntnose minnow	36.6	0.48	26	2.0 °C $hour^{-1}$	Smale & Rabeni (1995)	MO
Micropterus salmoides	largemouth bass	36.7	0.76	20	1.0 °C min^{-1}	Smith & Scott (1975)	-
Micropterus salmoides	largemouth bass	36.7	0.59	25	0.3 °C min^{-1}	Currie et al. (1998)	OK
Notemigonus crysoleucas	golden shiner	36.8	0.37	26	2.0 °C $hour^{-1}$	Smale & Rabeni (1995)	MO
Micropterus dolomieu	smallmouth bass	36.9	0.31	26	2.0 °C $hour^{-1}$	Smale & Rabeni (1995)	MO
Fundulus sciadicus	plains topminnow	37.0	0.3	26	2.0 °C $hour^{-1}$	Smale & Rabeni (1995)	MO
Ambloplites rupestris	rock bass	37.0	-	36	1.0 °C day^{-1}	Cherry et al. (1977)	VA
Campostoma anomalum	central stoneroller	37.2	0.33	26	2.0 °C $hour^{-1}$	Smale & Rabeni (1995)	MO
Micropterus salmoides	largemouth bass	37.3	0.6	32	1.0 °C day^{-1}	Fields et al. (1987)	-
Ictalurus punctatus	channel catfish	37.5	0.52	24	1.0 °C min^{-1}	Cheetham et al. (1976)	-
Micropterus salmoides	largemouth bass	37.5	0.64	24	0.2 °C min^{-1}	Fields et al. (1987)	-
Campostoma anomalum	central stoneroller	37.7	0.5	26	0.5-0.8 °C min^{-1}	Mundahl (1990)	OH

Etheostoma flabellare	fantail darter	37.7	0.5	27	0.5-0.8 °C min^{-1}	Mundahl (1990)	OH
Lepomis megalotis	longear sunfish	37.8	0.84	26	2.0 °C hour^{-1}	Smale & Rabeni (1995)	MO
Pimephales notatus	bluntnose minnow	37.9	0.50	25	0.5-0.8 °C min^{-1}	Mundahl (1990)	OH
Amerius natalis	yellow bullhead	37.9	0.44	26	2.0 °C hour^{-1}	Smale & Rabeni (1995)	MO
Lepomis cyanellus	green sunfish	37.9	0.75	26	2.0 °C hour^{-1}	Smale & Rabeni (1995)	MO
Lepomis macrochirus	bluegill	37.9	0.68	26	2.0 °C hour^{-1}	Smale & Rabeni (1995)	MO
Ictalurus punctatus	channel catfish	38.0	0.39	20	0.3 °C min^{-1}	Watenpaugh et al. (1995)	-
Lepomis macrochirus	bluegill	38.0	-	36	1.0 °C day^{-1}	Cherry et al. (1977)	VA
Micropterus punctulatus	spotted bass	38.0	-	36	1.0 °C day^{-1}	Cherry et al. (1977)	VA
Amerius melas	black bullhead	38.1	0.39	26	2.0 °C hour^{-1}	Smale & Rabeni (1995)	MO
Cyprinella lutrensis	red shiner	38.1	0.42	26	2.0 °C hour^{-1}	Smale & Rabeni (1995)	MO
Fundulus olivaceus	blackstripe topminnow	38.3	0.67	26	2.0 °C hour^{-1}	Smale & Rabeni (1995)	MO

Micropterus salmoides	largemouth bass	38.5	0.34	30	0.3 °C min^{-1}	Currie et al. (1998)	OK
Ictalurus punctatus	channel catfish	38.7	0.36	25	0.3 °C min^{-1}	Currie et al. (1998)	OK
Carpio cyprinus	common carp	38.8	0.8	24	0.5-0.8 °C min^{-1}	Mundahl (1990)	OH
Fundulus notatus	blackspotted topminnow	38.8	0.59	26	2.0 °C hour^{-1}	Smale & Rabeni (1995)	MO
Ictalurus punctatus	channel catfish	39.2	0.58	28	1.0 °C min^{-1}	Cheetham et al. (1976)	-
Micropterus salmoides	largemouth bass	39.2	0.64	32	1.0 °C day^{-1}	Fields et al. (1987)	-
Micropterus salmoides	largemouth bass	40.1	1.33	28	1.0 °C min^{-1}	Smith & Scott (1975)	-
Ictalurus punctatus	channel catfish	40.1	0.66	30	0.15 °C min^{-1}	Bennett et al. (1987)	-
Ictalurus punctatus	channel catfish	40.3	0.29	30	0.3 °C min^{-1}	Currie et al. (1998)	OK
Micropterus salmoides	largemouth bass	40.9	0.4	32	0.2 °C min^{-1}	Fields et al. (1987)	-
Ictalurus punctatus	channel catfish	41.0	0.31	32	1.0 °C min^{-1}	Cheetham et al. (1976)	-
Micropterus salmoides	largemouth bass	41.8	0.38	32	0.2 °C min^{-1}	Fields et al. (1987)	-

Ictalurus punctatus	channel catfish	42.1	0.25	35	0.15 °C min^{-1}	Bennett et al. (1987)	-

Table 15.—Number of individuals, mean initial-total length (TL; mm), mean initial weight (WT; g) and mean PIT-tag size (length x diameter; mm) for PIT-tag retention trials by species.

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

Table 16.—Number of tagged individuals, range of total lengths (mm), mean total length, and standard deviation (S.D.) of the mean total length for all species that were included in multistate-model analyses.

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

Table 17. —Models and parameter descriptions used in multistate mark-recapture analyses.

#	Model	Parameter Description:		
		Survival (S)	Recapture (p)	Transition (Psi)
1	S(~1)p(~1)Psi(~1)	**All Constant**		
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)		Constant.	Variable by day.
		Variable with		

		discharge.		
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)	Global Model		

Table 18.—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	0.8330	0.98	2.3603	0.83
Barren Fork Creek	8	0.6709	0.99	2.5610	0.97
Illinois River	6	0.5498	0.97	0.7083	0.94

Table 19.—Survival (S) and PIT-tag retention (R) (both expressed as percentages) for treatment fishes after 30, 60, and 90 days. Growth (G = weight $_{\text{final}}$ – weight $_{\text{initial}}$) is represented as the mean weight gain (+/-) after 35 - 43 days.

Species		n	30 days		60 days		90 days		
			S	R	S	R	S	R	G
Luxilus cardinalis	cardinal shiner	12	100	100	100	100	100	100	+
Compostoma annomalum	central stoneroller	8	100	100	88	100	88	100	+
Noturus exilis	slender madtom	13	100	100	100	100	n/a	n/a	-
Etheostoma spectabile	orangethroat darter	9	89	88	56	88	n/a	n/a	-
Etheostoma blennioides	greenside darter	3	100	100	100	100	n/a	n/a	-
Micropterus dolomieu	smallmouth bass	25	96	100	96	100	n/a	n/a	+

Table 20. —Goodness-of-fit testing and estimate of the variance inflation factor (i.e., c-hat) for each global model (i.e., time varying model) used in multistate mark-recapture analyses.

Year	Species	X^2	df	Estimate of c-hat	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	1.50	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 21.—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	c-hat
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	194.20	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	5.80	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 22.—Summed weights of parameter covariates across all weighted models.

Cardinal Shiner				
	covariates	S	p	Psi
2011	~1	0.40	0.26	0.75
	time	0.00	0.00	0.00
	Q	0.60	-	0.25
	effort	-	0.74	-
	covariates	S	p	Psi
2012	~1	0.00	0.74	0.00
	time	0.00	0.00	0.00
	Q	1.00	-	1.00
	effort	-	0.26	-

Table 23—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	c-hat
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

Table 24. —Summed weights of parameter covariates across all weighted models.

Central Stoneroller				
	covariates	S	p	Psi
2011	~1	0.41	0.72	0.53
	time	0.00	0.00	0.00
	Q	0.59	-	0.47
	effort	-	0.28	-
	covariates	S	p	Psi
2012	~1	0.72	0.30	0.76
	time	0.00	0.00	0.00
	Q	0.28	-	0.24
	effort	-	0.70	-

Table 25.—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	c-hat
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 26. —Summed weights of parameter covariates across all weighted models.

Orangethroat Darter				
	covariates	S	p	Psi
2012	~1	0.18	0.74	0.05
	time	0.00	0.00	0.00
	Q	0.82	-	0.95
	effort	-	0.26	-

Table 27.—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	c-hat
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	0.922	0.39	574.30	1.03

Table 28. —Summed weights of parameter covariates across all weighted models.

Slender Madtom				
	covariates	S	p	Psi
2012	~1	0.61	0.00	0.00
	time	0.00	0.00	0.00
	Q	0.39	-	1.00
	effort	-	1.00	-

Table 29.—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	c-hat
2011	1	S(~1)p(~1)Psi(~1)	3	313.67	0	0.38	251.70	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	1.50
	4	S(~1)p(~effort)Psi(~1)	4	695.82	1.98	0.15	608.49	1.50
	2	S(~1)p(~1)Psi(~Q)	4	695.87	2.03	0.14	608.54	1.50
	10	S(~Q)p(~1)Psi(~1)	4	695.92	2.07	0.14	608.59	1.50
	5	S(~1)p(~effort)Psi(~Q)	5	697.87	4.03	0.05	608.44	1.50
	13	S(~Q)p(~effort)Psi(~1)	5	697.92	4.07	0.05	608.49	1.50
	11	S(~Q)p(~1)Psi(~Q)	5	697.97	4.13	0.05	608.54	1.50
	14	S(~Q)p(~effort)Psi(~Q)	6	699.99	6.15	0.02	608.44	1.50

Table 30. —Summed weights of parameter covariates across all weighted models.

Smallmouth Bass				
	covariates	S	p	Psi
2011	~1	0.74	0.74	0.69
	time	0.00	0.00	0.00
	Q	0.26	-	0.31
	effort	-	0.26	-
	covariates	S	p	Psi
2012	~1	0.74	0.73	0.74
	time	0.00	0.00	0.00
	Q	0.26	-	0.26
	effort	-	0.27	-

Table 31.—Summed weights of parameter covariates across all weighted models for each species and year. Orangethroat darter and slender madtom did not have enough tagged individuals and overall number of recaptures to include in the 2011 analyses.

		cardinal shiner			central stoneroller			orangethroat darter			slender madtom			smallmouth bass		
	Variables	S	p	Psi	S	p	Psi	S	p	Psi	S	p	Psi	S	p	Psi
2011	~1	0.40	0.26	0.75	0.41	0.72	0.53	-	-	-	-	-	-	0.74	0.74	0.69
	time	0.00	0.00	0.00	0.00	0.00	0.00	-	-	-	-	-	-	0.00	0.00	0.00
	Q	0.60	-	0.25	0.59	-	0.47	-	-	-	-	-	-	0.26	-	0.31
	effort	-	0.74	-	-	0.28	-	-	-	-	-	-	-	-	0.26	-
2012	~1	0.00	0.74	0.00	0.72	0.30	0.76	0.18	0.74	0.05	0.61	0.00	0.00	0.74	0.73	0.74
	time	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Q	1.00	-	1.00	0.28	-	0.24	0.82	-	0.95	0.39	-	1.00	0.26	-	0.26
	effort	-	0.26	-	-	0.70	-	-	0.26	-	-	1.00	-	-	0.27	-

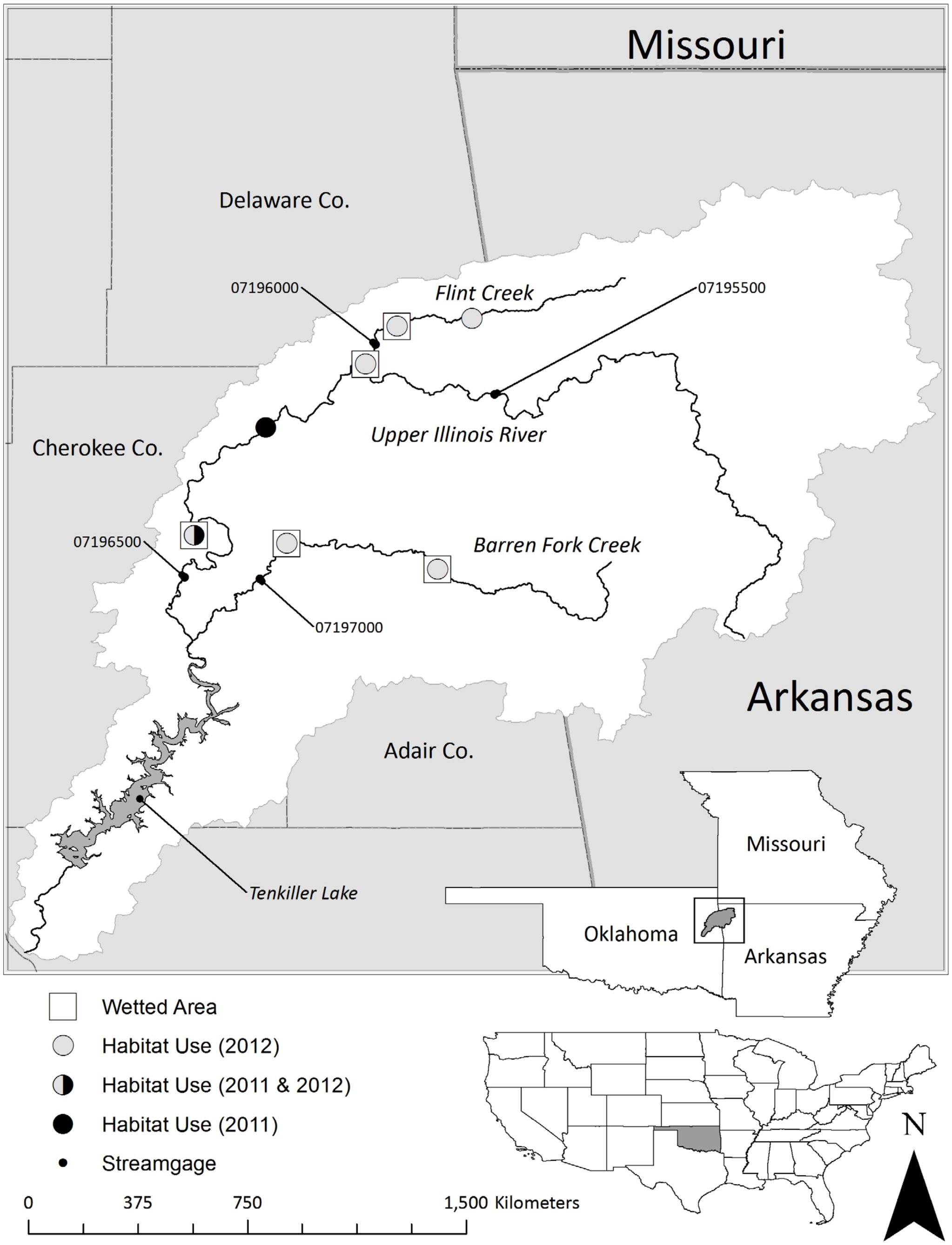


Figure 1. Study sites in the Illinois River catchment for Objective 1.

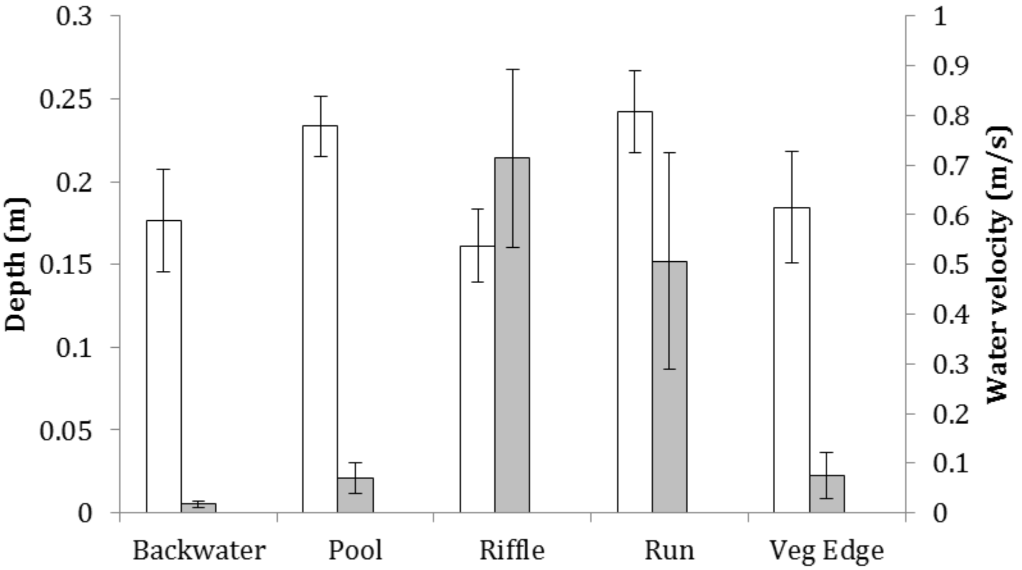


Figure 2.—Mean (± 95% confidence limits) depth (m) and velocity (m/s) by channel-unit for all habitat-use samples. Hollow bars represent depth and grey bars represent velocity.

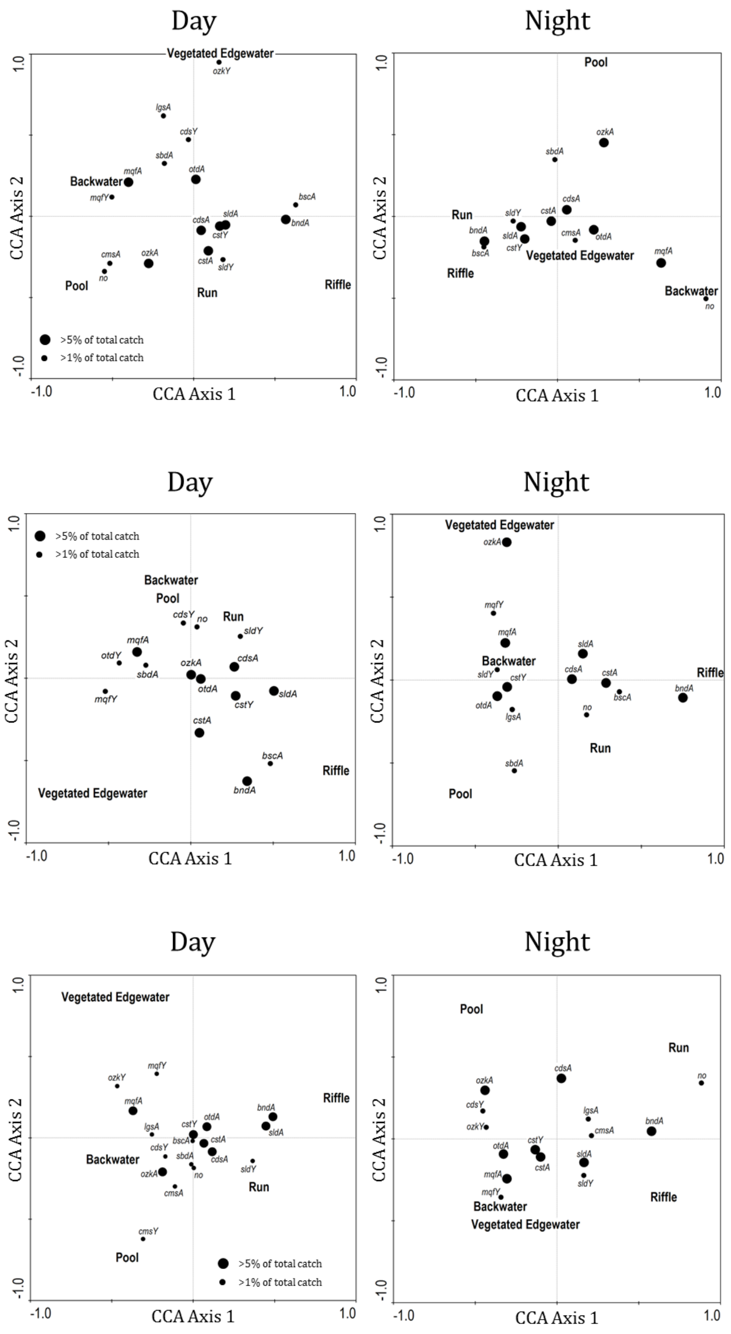


Figure 3.—Canonical Correspondence Analysis (CCA) plots for samples taken in the day and night periods from Barren Fork Creek (upper panel), Flint Creek (middle panel) and the upper-Illinois River (lower panel). Species codes are listed in Table 7.

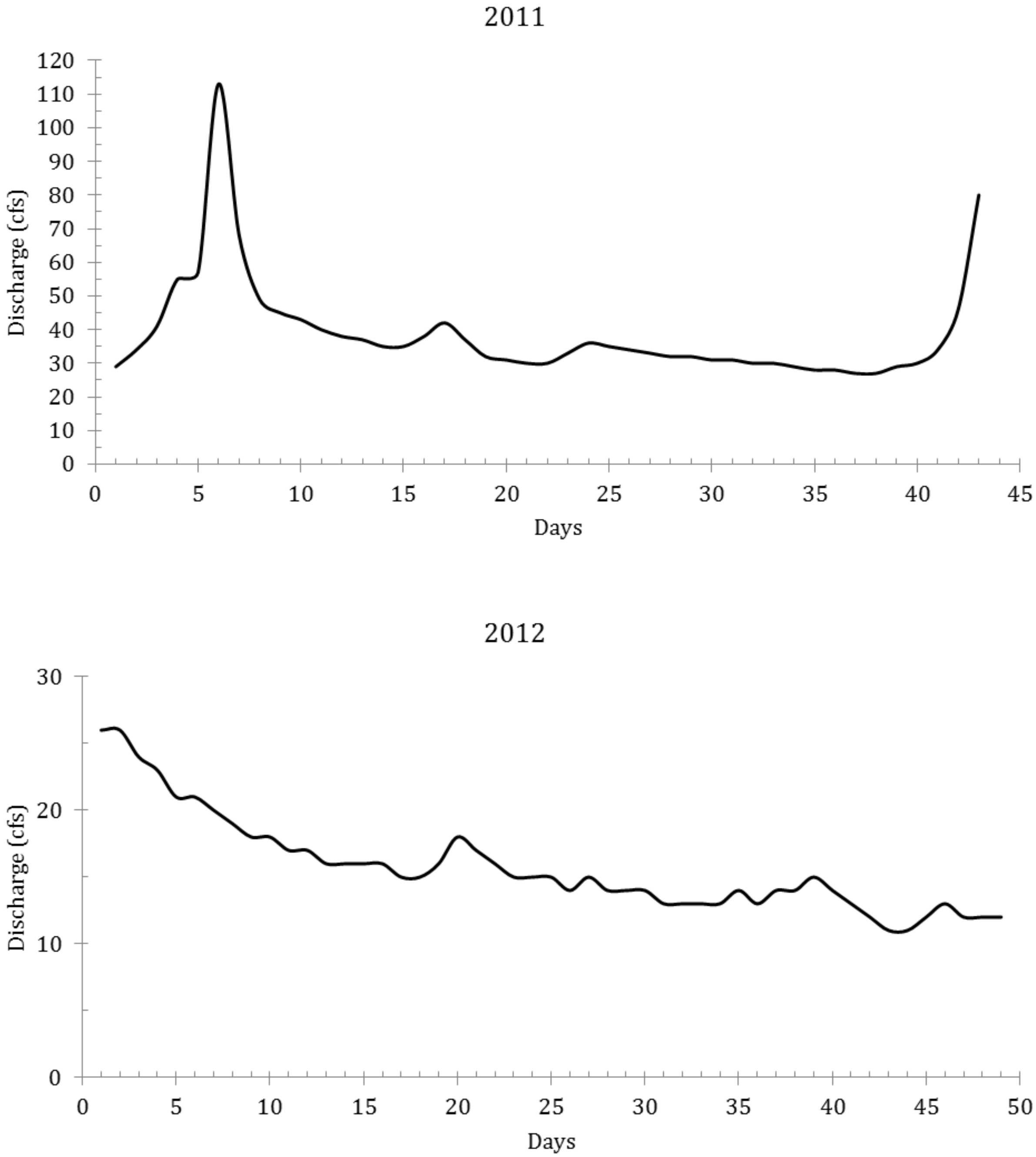


Figure 4.—Discharge (cubic feet per second) of Flint Creek during the 2011 and 2012 field season.

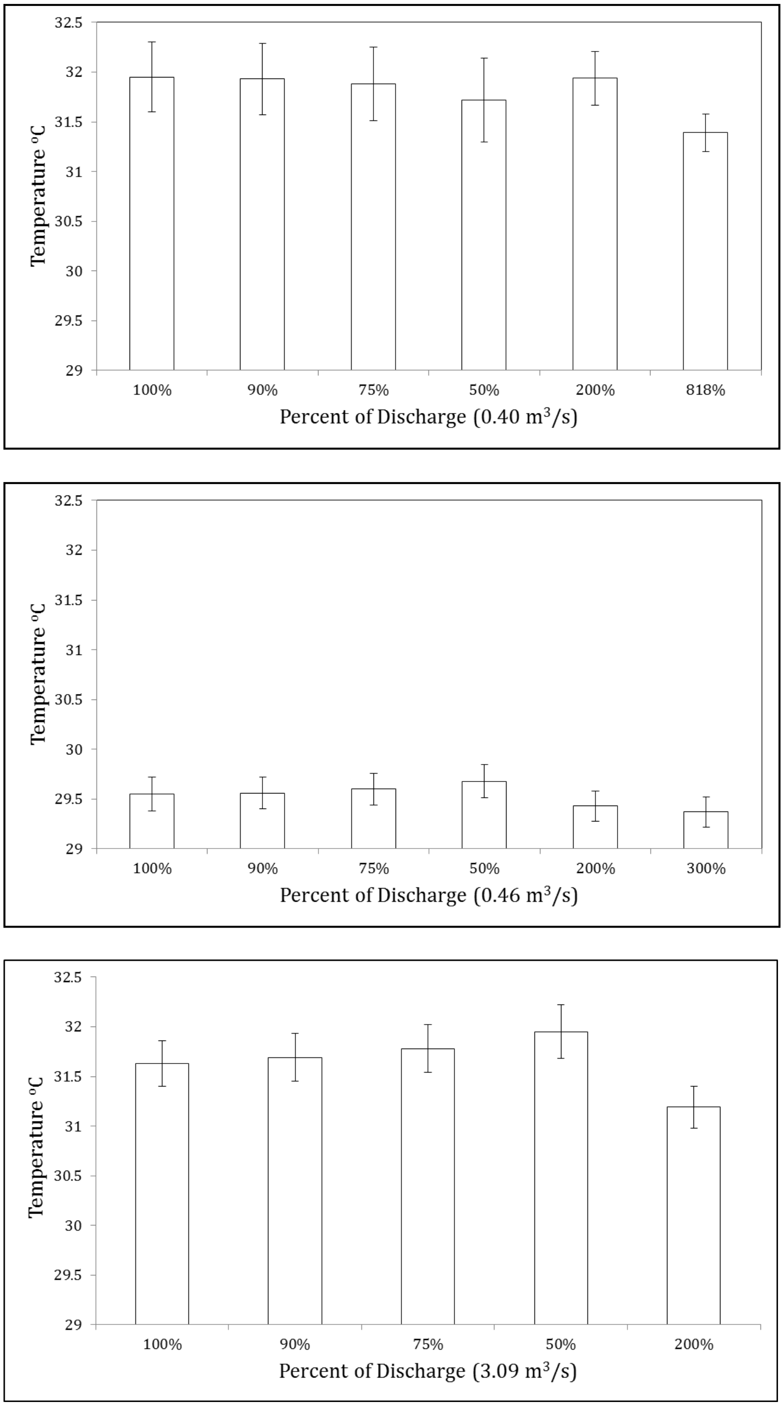


Figure 5. —Predicted mean maximum water temperatures (° C) (± 95% C.I.) in Barren Fork Creek (upper panel), Flint Creek (middle panel) and the upper Illinois River (lower panel) at mean July-August discharge (100%), hypothetical reduced discharges (e.g., 90%, 75%, 50% of the mean), and a hypothetical increases (200%, 300%, 818%). Values of percentages are provided in the methods section.

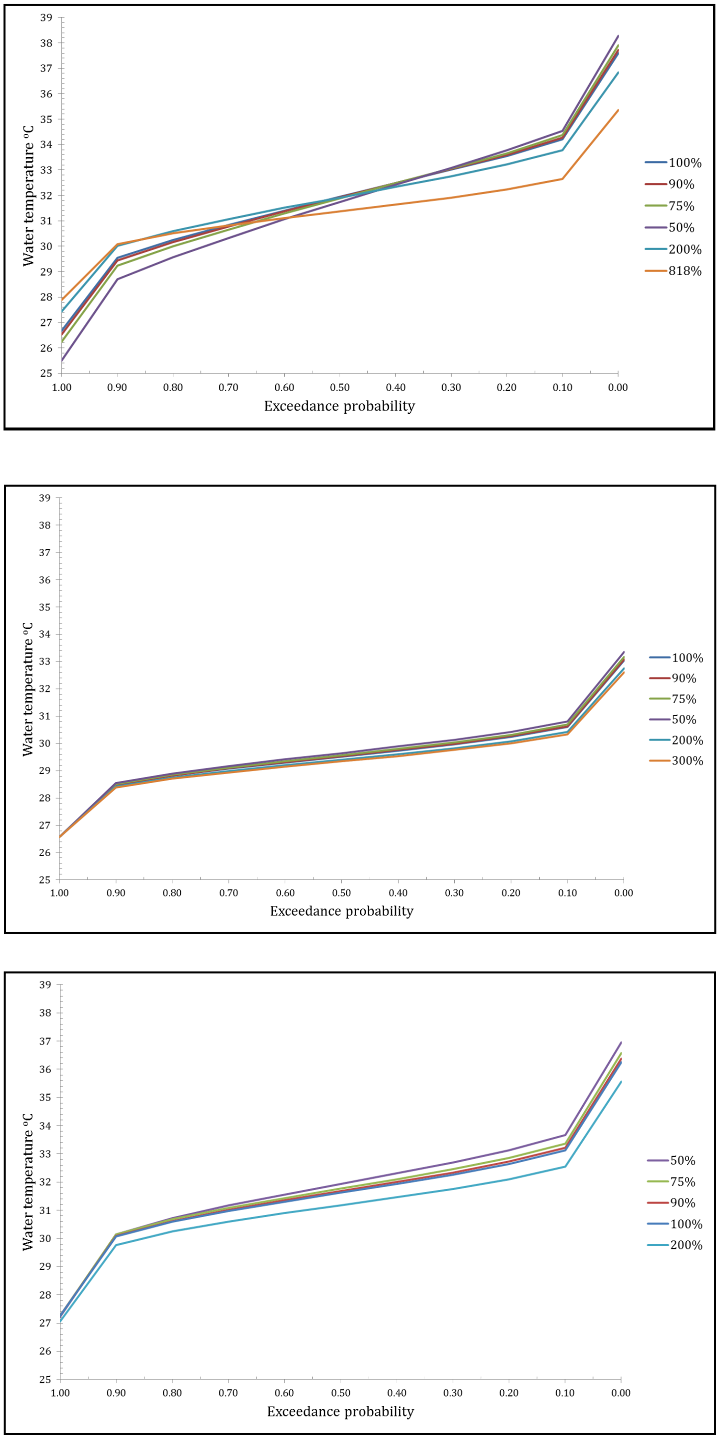


Figure 6.—Exceedance probabilities of maximum-water temperatures under different discharge scenarios for conditions present during July – August 2012 in Barren Fork Creek (upper panel), Flint Creek (middle panel) and the upper Illinois River (lower panel). Hypothetical discharges are the same as provided in Figure 4.

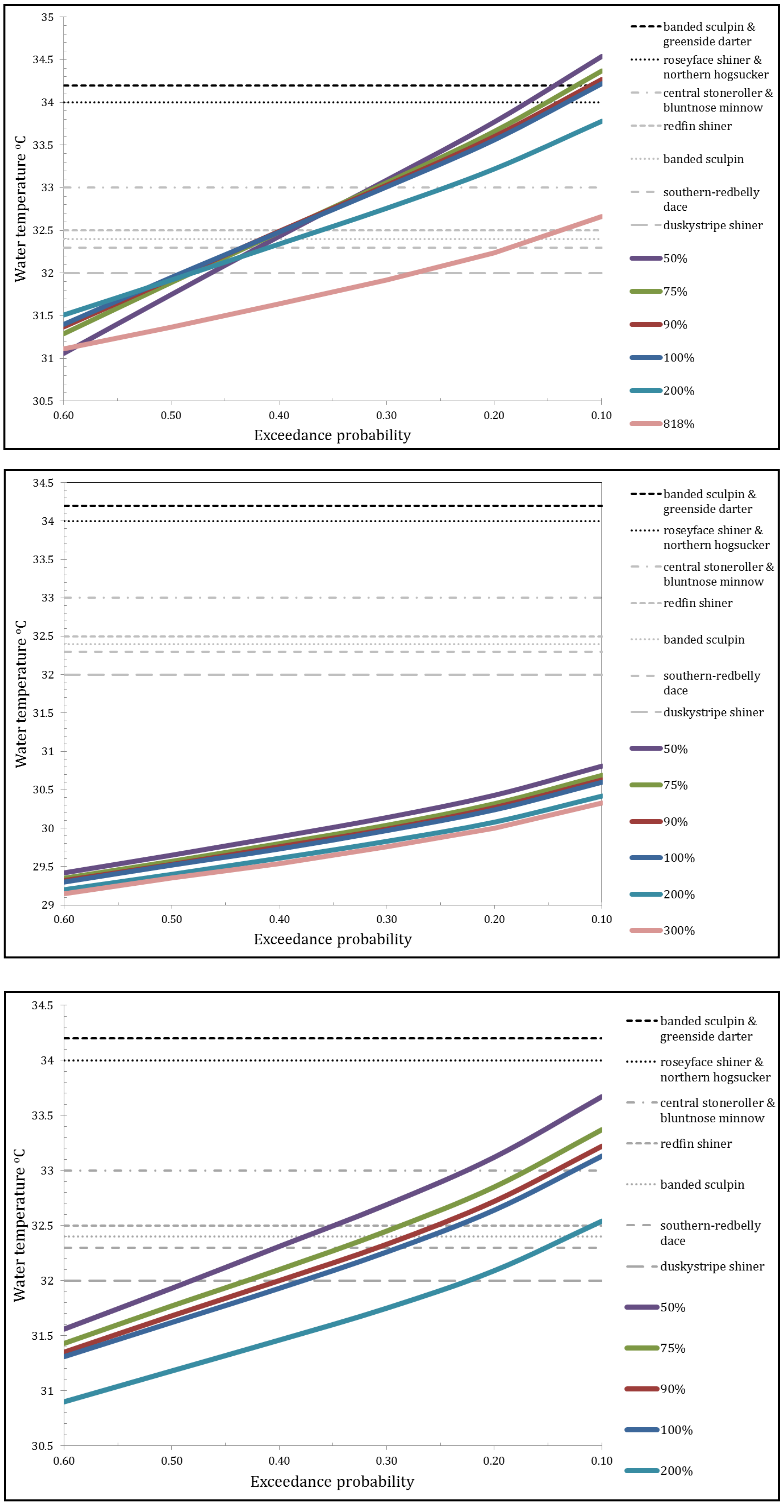


Figure 7.—Discharge scenarios with the probability to exceed critical-thermal maxima for fishes in Barren Fork Creek (upper panel), Flint Creek (middle panel) and the upper Illinois River (lower panel). Hypothetical discharges are the same as provided in Figure 4.

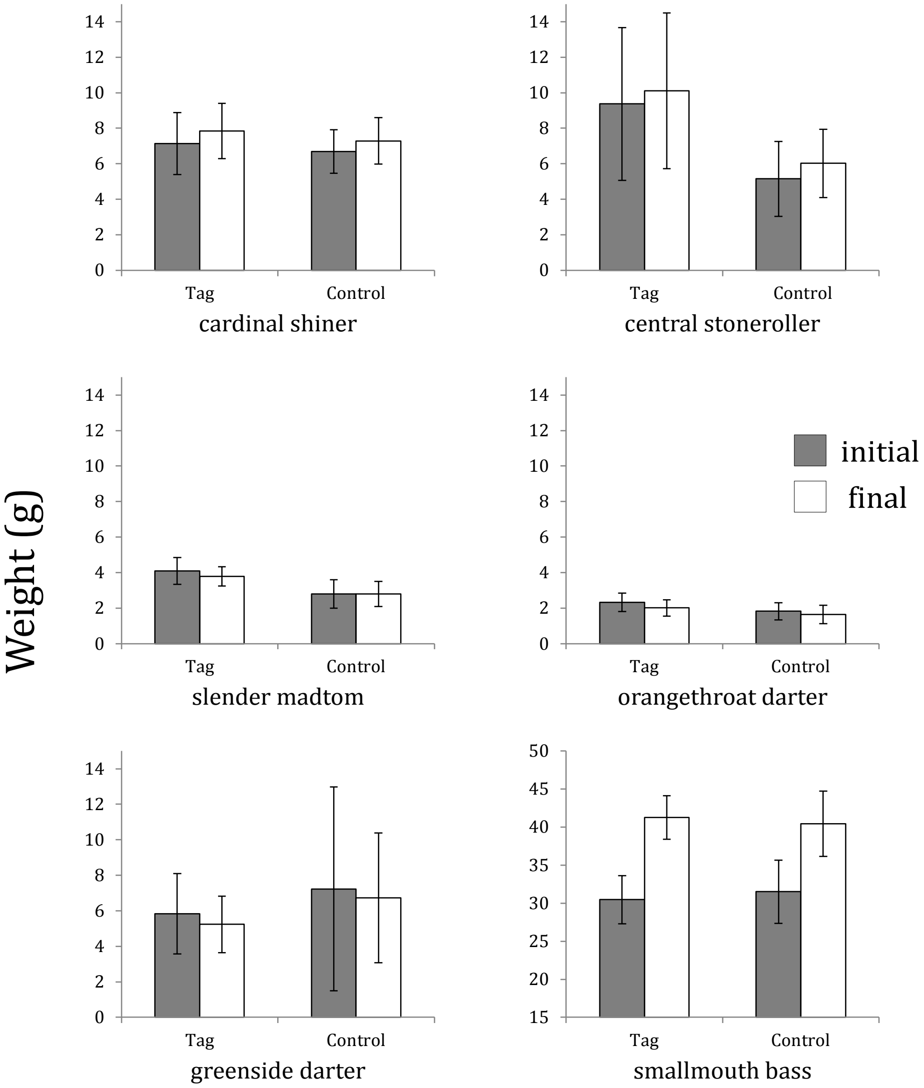


Figure 8.—Mean growth (mean weight ± 95% confidence limits) of treatment and control fishes during PIT-tag retention experiments. Growth was not significantly different between PIT-tag treatment and control groups for any species.

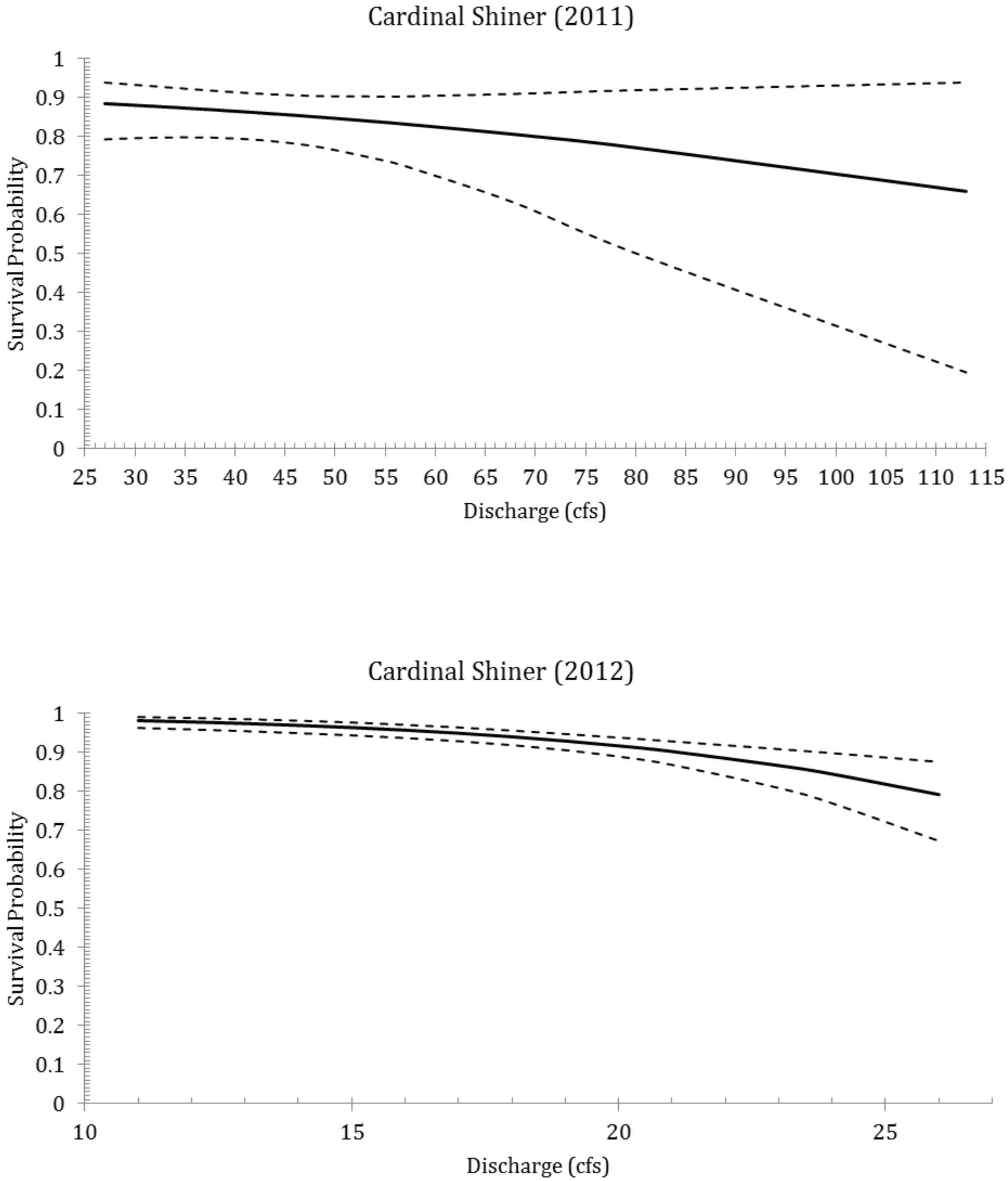


Figure 9.—Survival probability of cardinal shiner related to discharge during 2011 (upper panel) and 2012 (lower panel).

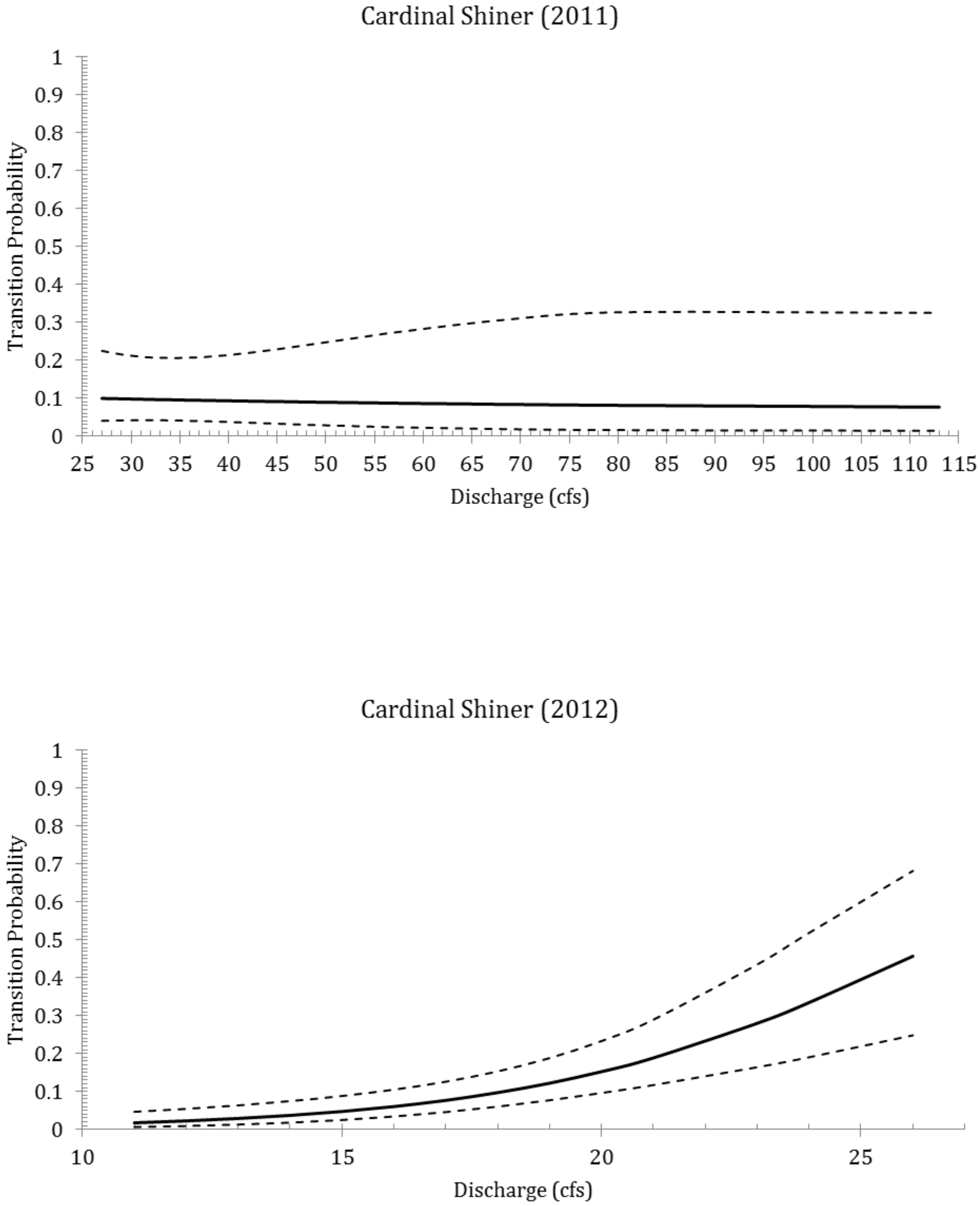


Figure 10.—Probability of cardinal shiner transitioning to adjacent channel units with changing discharge during 2011 (upper panel) and 2012 (lower panel).

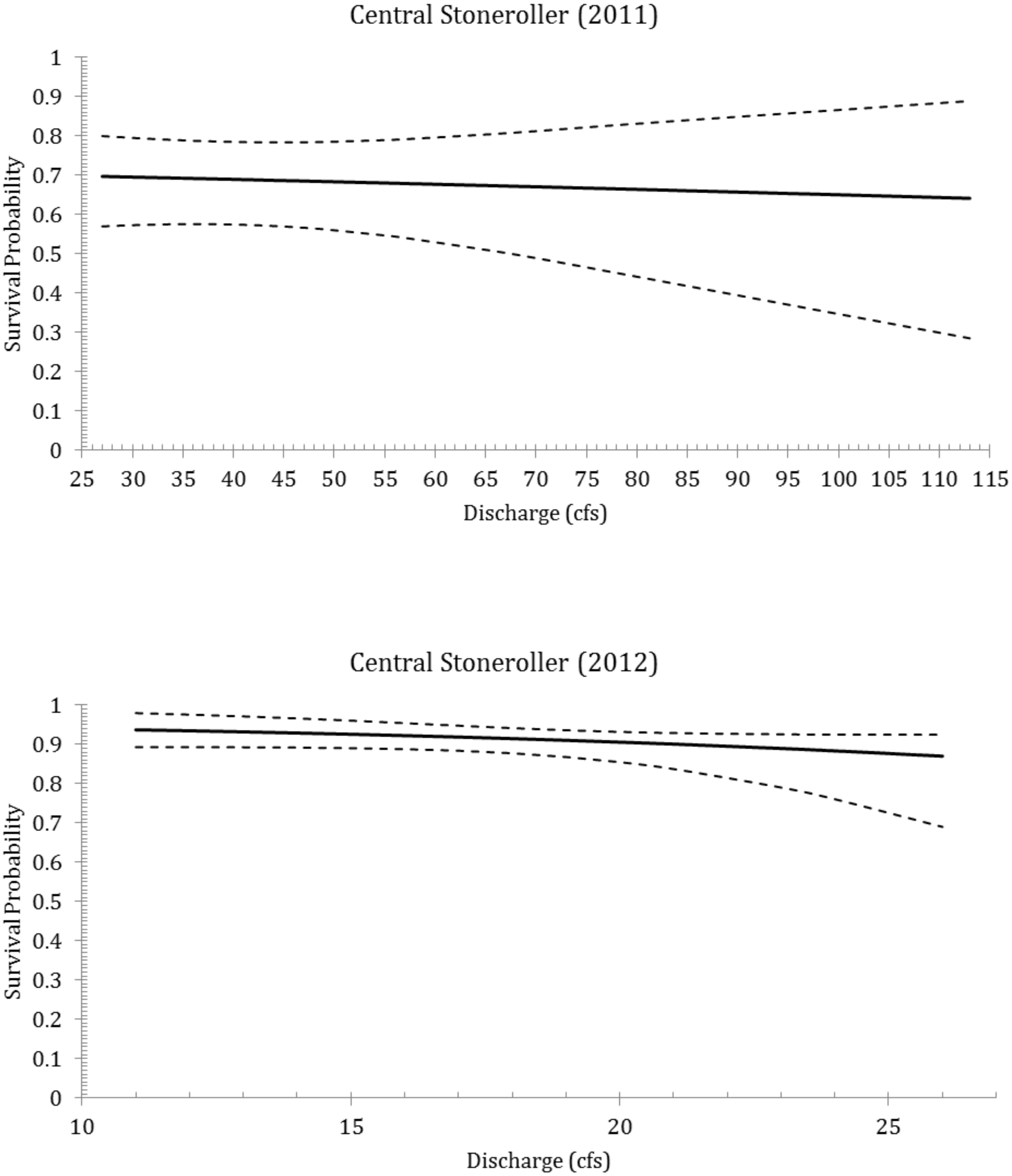


Figure 11.—Survival probability of central stoneroller related to discharge during 2011 (upper panel) and 2012 (lower panel).

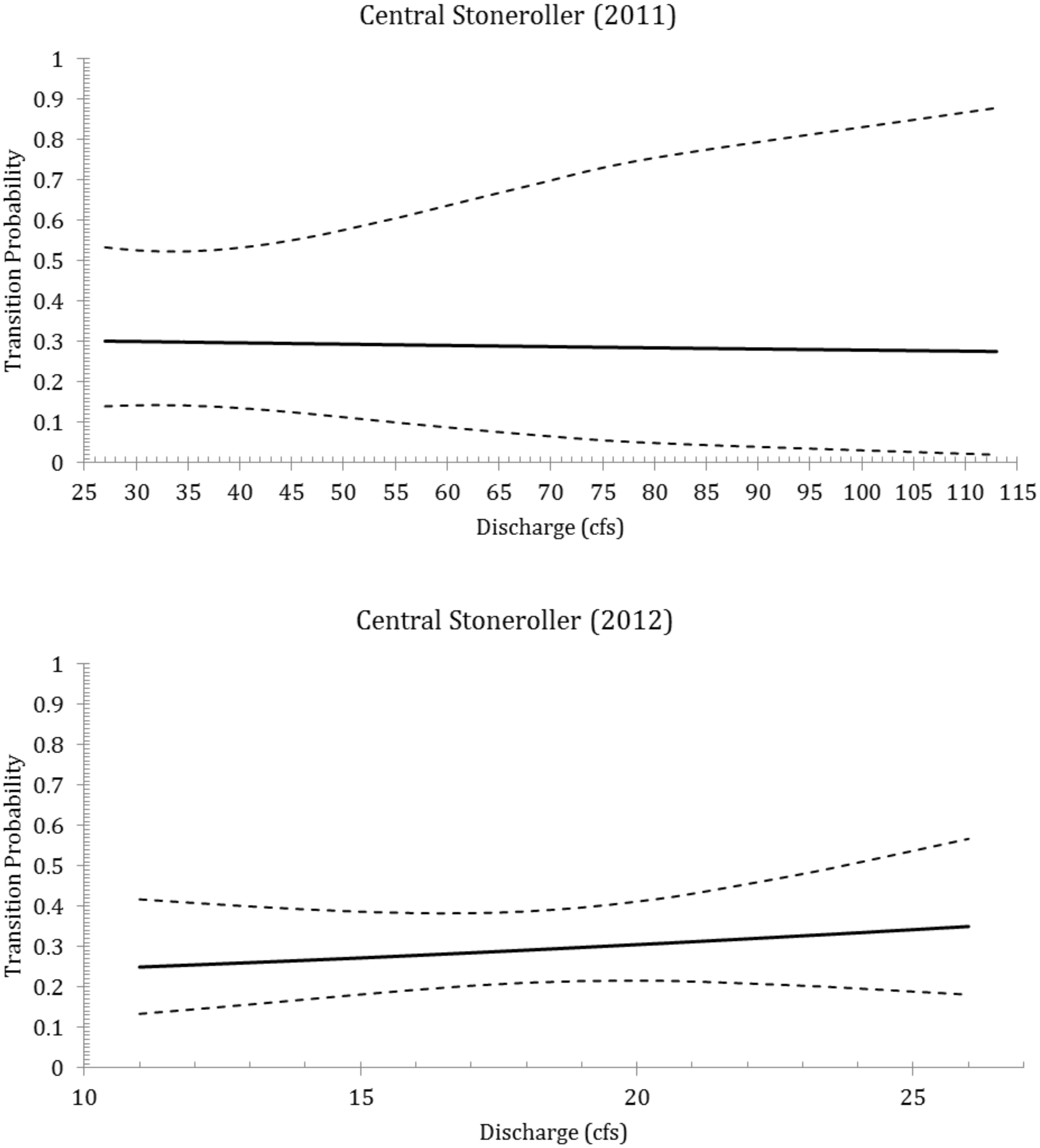


Figure 12.—Probability of central stoneroller transitioning to adjacent channel units with changing discharge during 2011 (upper panel) and 2012 (lower panel).

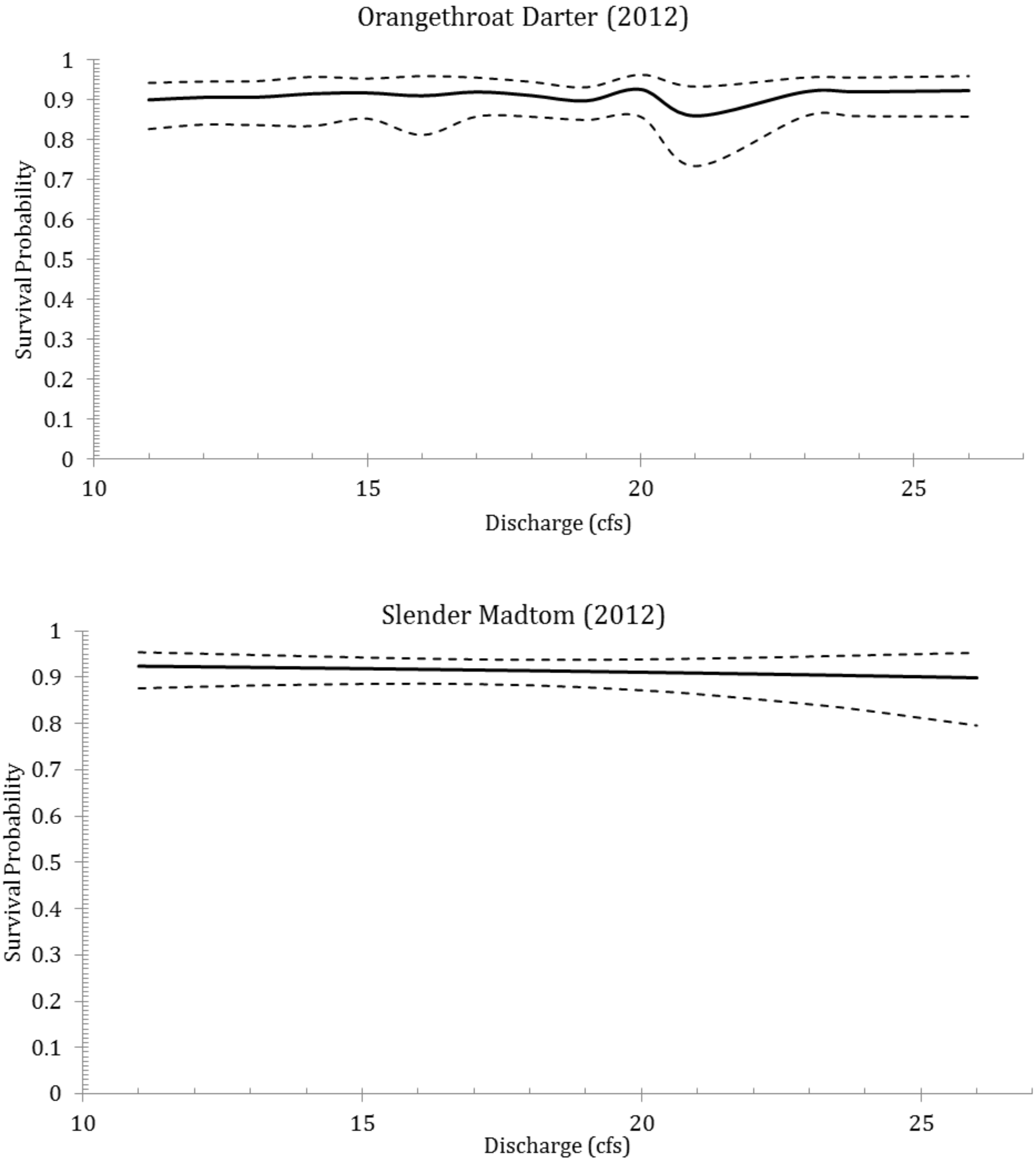
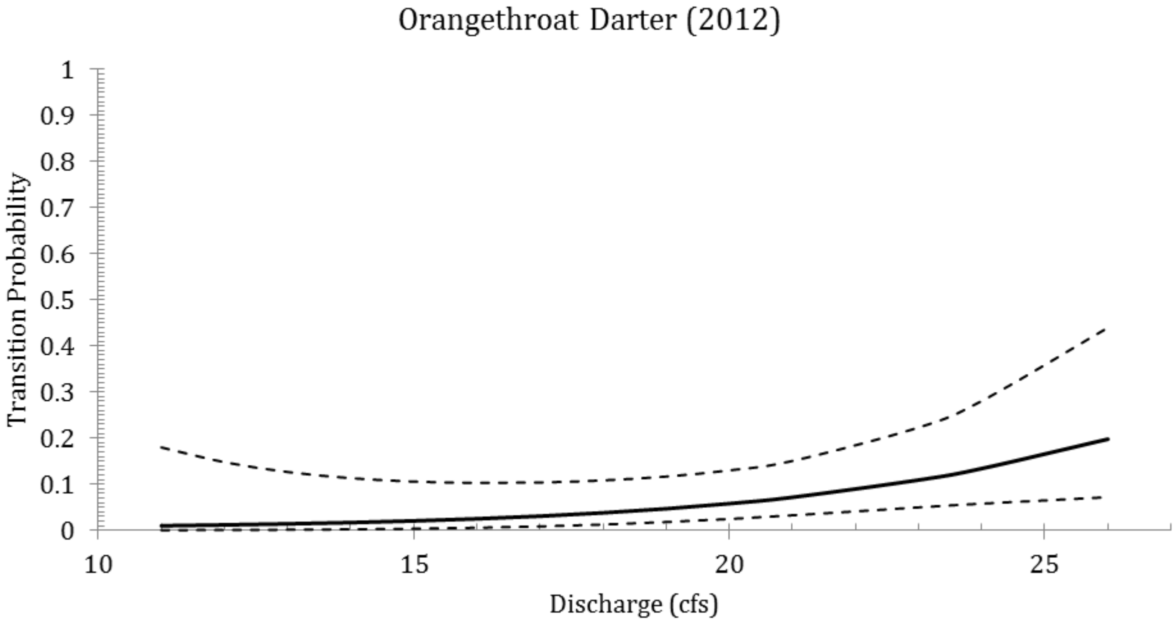


Figure 13.—Survival probability of orangethroat darter (upper panel) and slender madtom (lower panel) related to discharge during the 2012 field season.



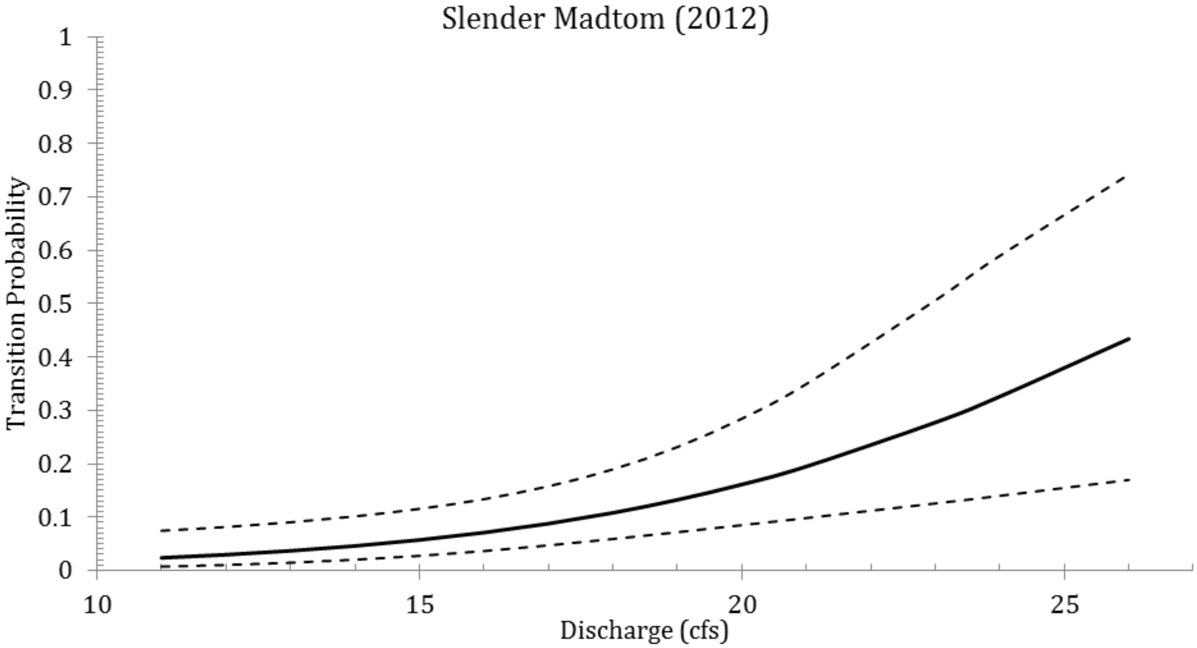


Figure 14.—Probability of orangethroat darter (upper panel) and slender madtom (lower panel) transitioning to adjacent channel units with changing discharge during 2012.

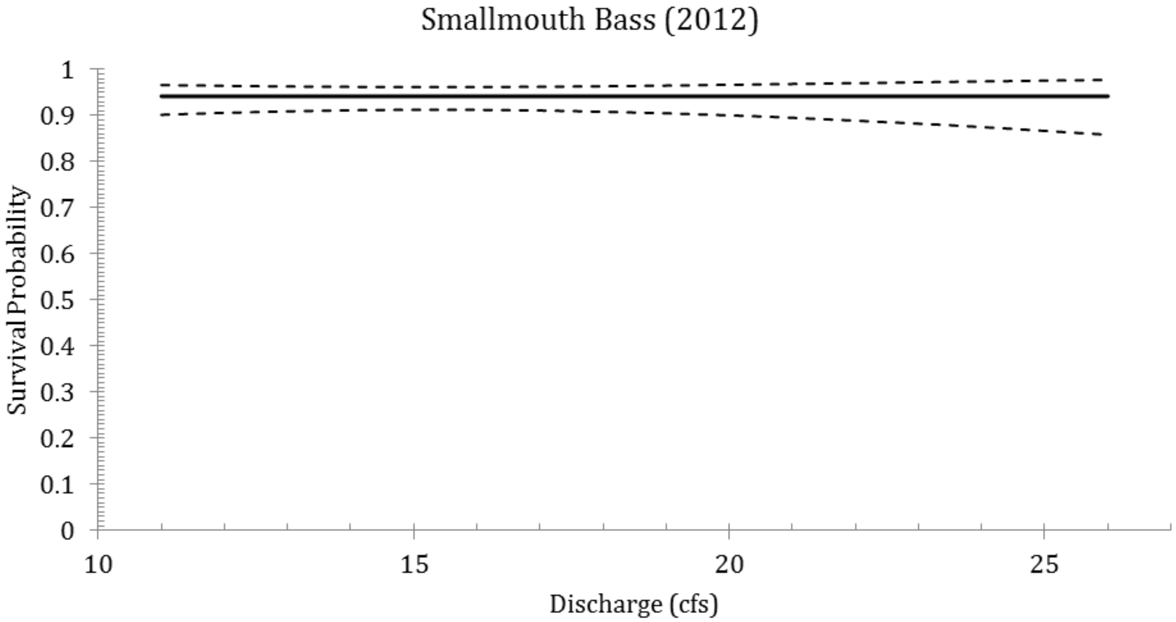


Figure 15.—Survival probability of smallmouth bass related to discharge during 2011 (upper panel) and 2012 (lower panel).

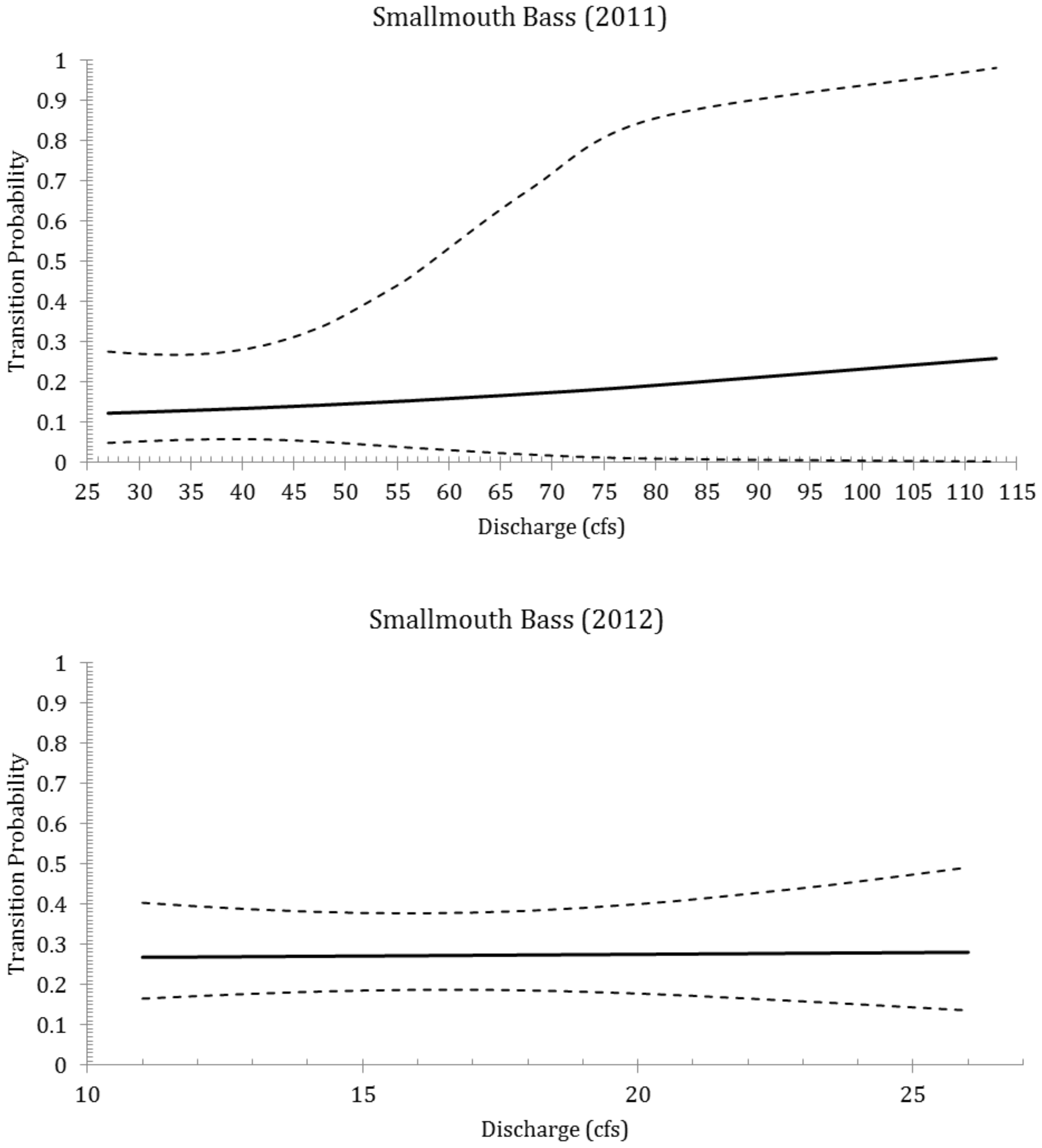


Figure 16.—Probability of smallmouth bass transitioning to adjacent channel units with changing discharge during 2011 (upper panel) and 2012 (lower panel).

Appendix A. — Wetted area (m^2) to discharge (m^3/s) relationships by channel unit used in broken-line regressions. Each figure represents a single channel unit and panels (A-D on individual figures) are different channel units that were measured over a range of discharge conditions.

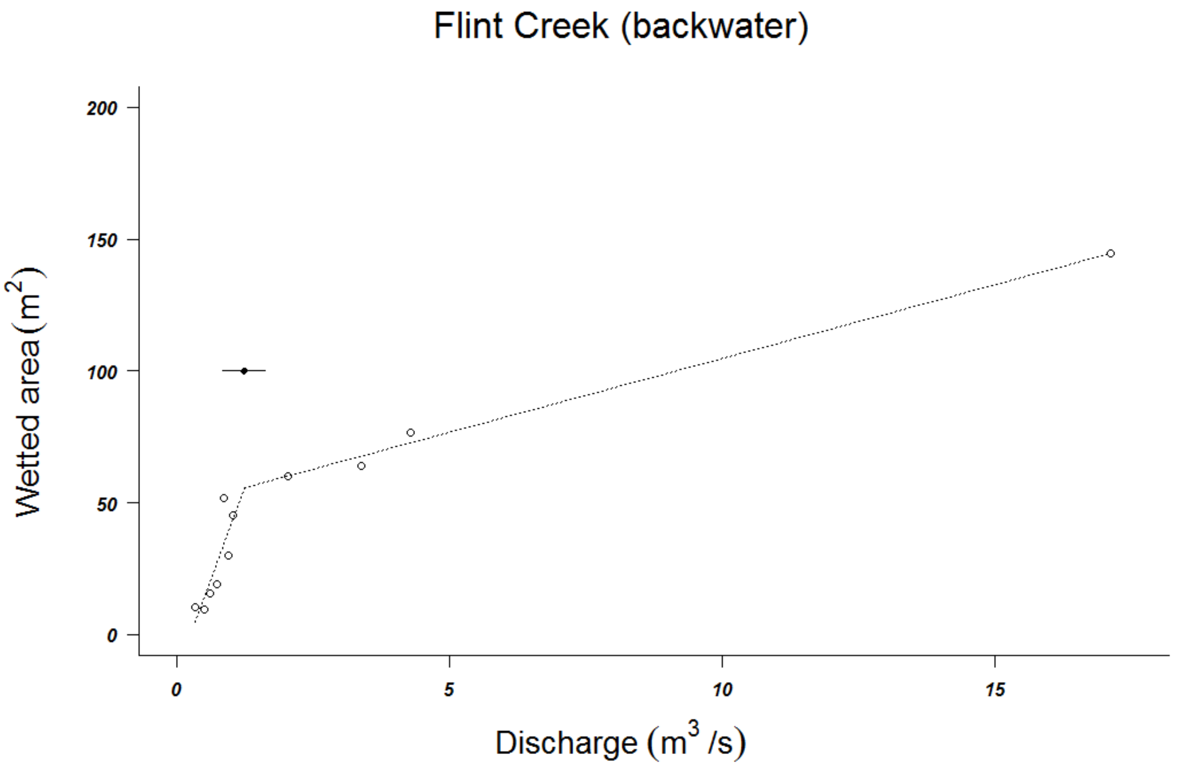


Figure 1.—Wetted area (m^2) to discharge (m^3/s) by channel unit used in broken-line regressions. Solid lines are breakpoint estimates with 90% confidence levels. Dashed lines are segmented-linear models or linear models (where no significant breakpoint was found).

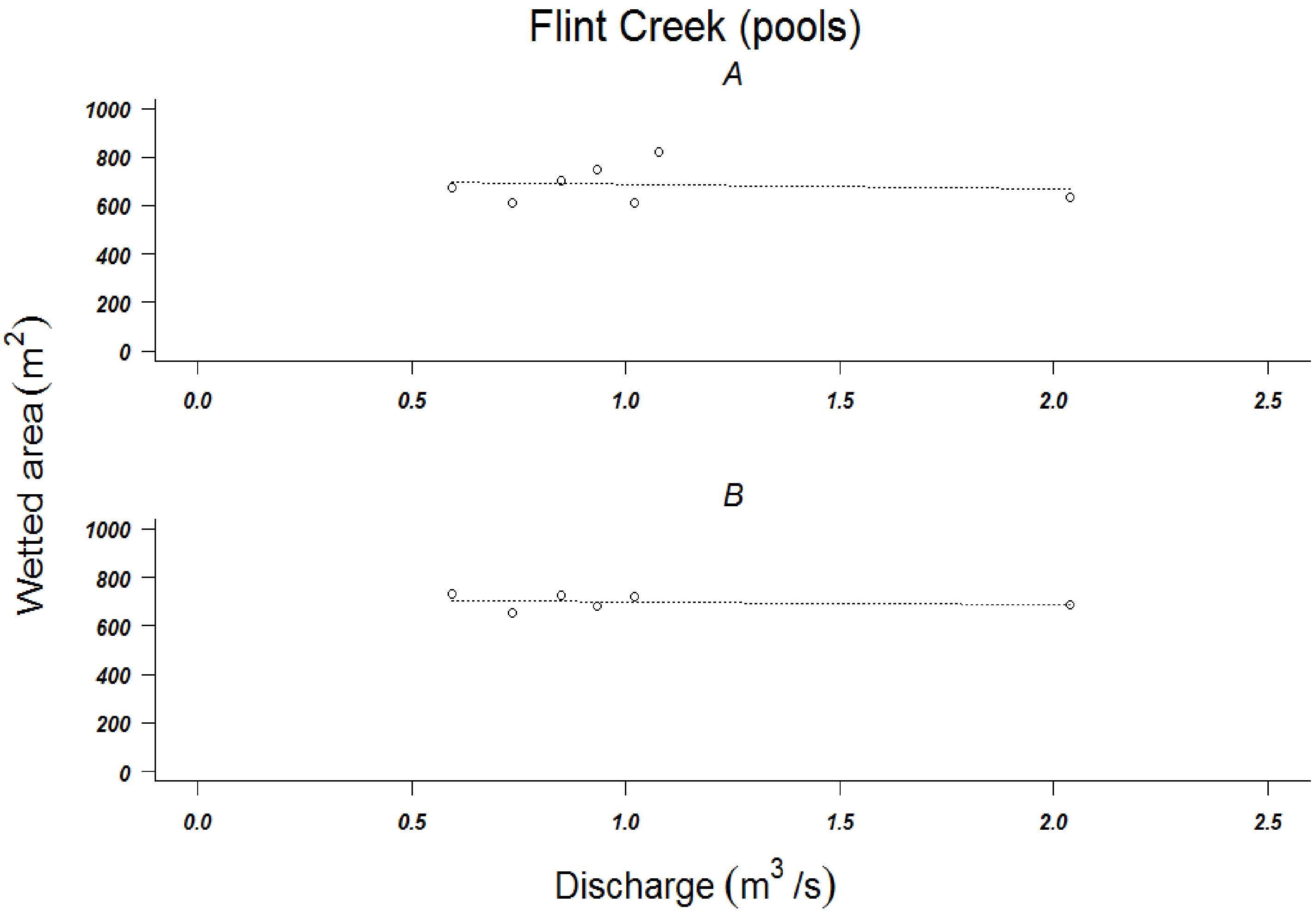


Figure 2.—Wetted area (m^2) to discharge (m^3/s) by channel unit used in broken-line regressions. Solid lines are breakpoint estimates with 90% confidence levels. Dashed lines are segmented-linear models or linear models (where no significant breakpoint was found).

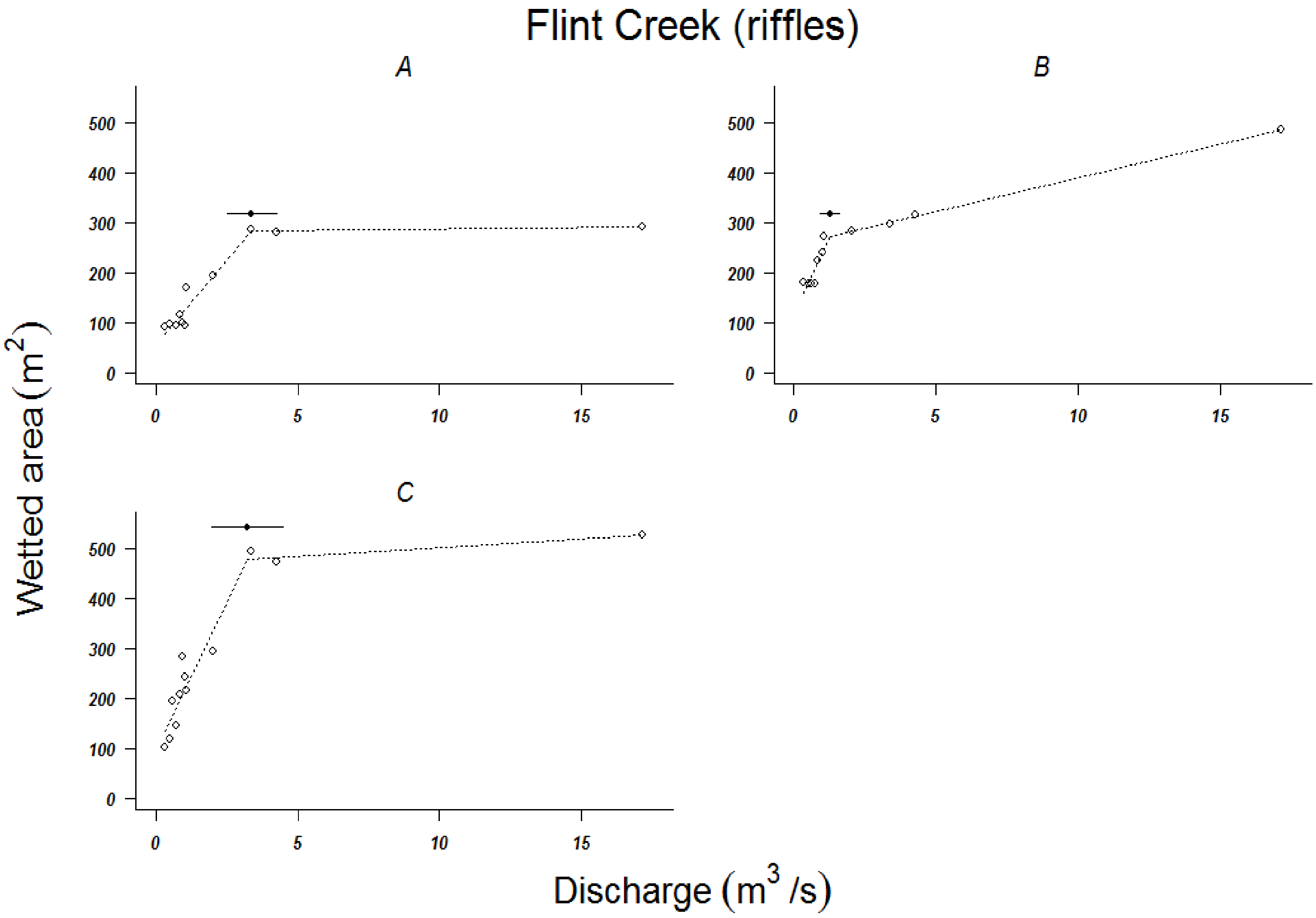


Figure 3.—Wetted area (m^2) to discharge (m^3/s) by channel unit used in broken-line regressions. Solid lines are breakpoint estimates with 90% confidence levels. Dashed lines are segmented-linear models or linear models (where no significant breakpoint was found).

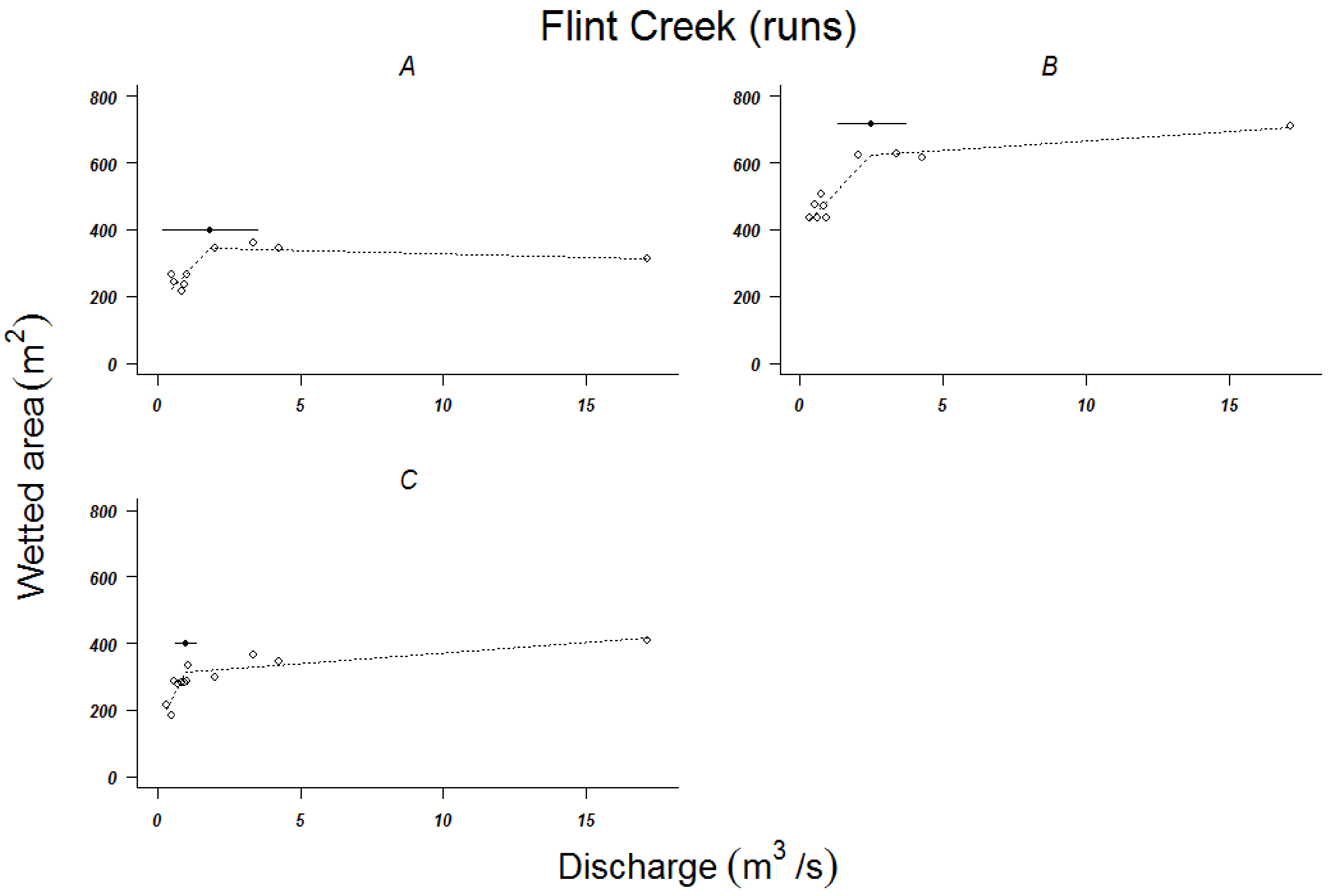


Figure 4.—Wetted area (m^2) to discharge (m^3/s) by channel unit used in broken-line regressions. Solid lines are breakpoint estimates with 90% confidence levels. Dashed lines are segmented-linear models or linear models (where no significant breakpoint was found)

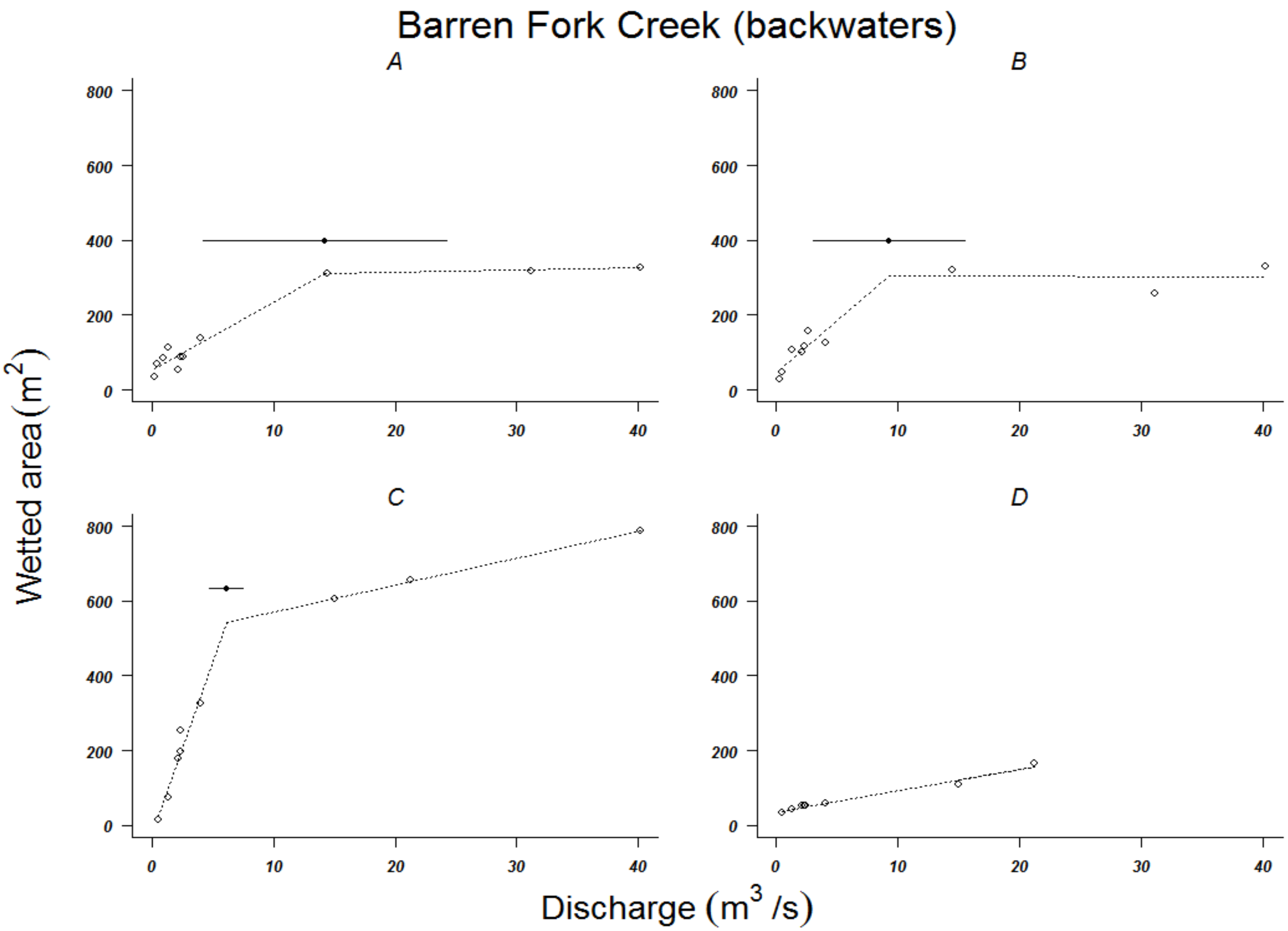


Figure 5.—Wetted area (m^2) to discharge (m^3/s) by channel unit used in broken-line regressions. Solid lines are breakpoint estimates with 90% confidence levels. Dashed lines are segmented-linear models or linear models (where no significant breakpoint was found).

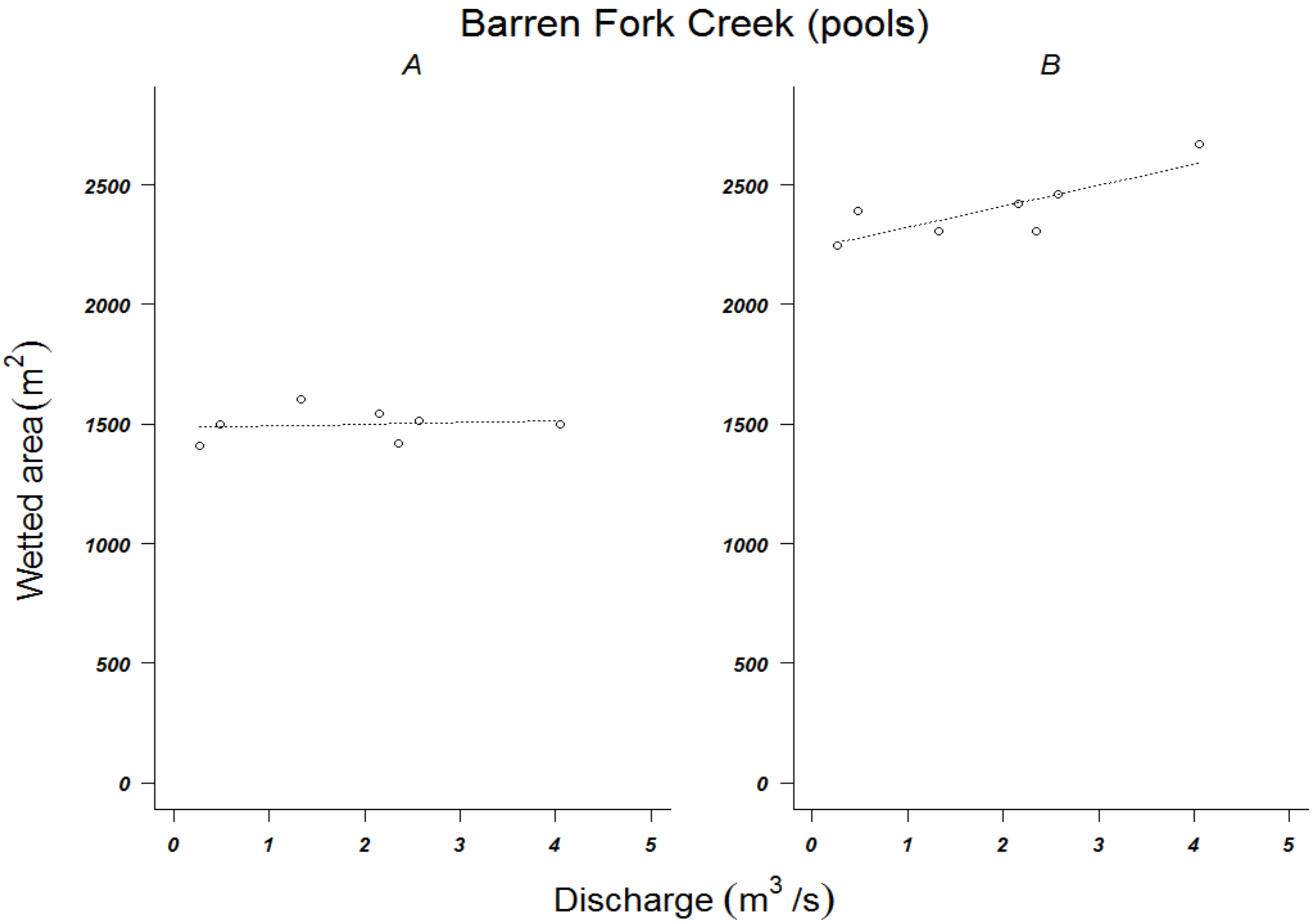


Figure 6.—Wetted area (m^2) to discharge (m^3/s) by channel unit used in broken-line regressions. Solid lines are breakpoint estimates with 90% confidence levels. Dashed lines are segmented-linear models or linear models (where no significant breakpoint was found).

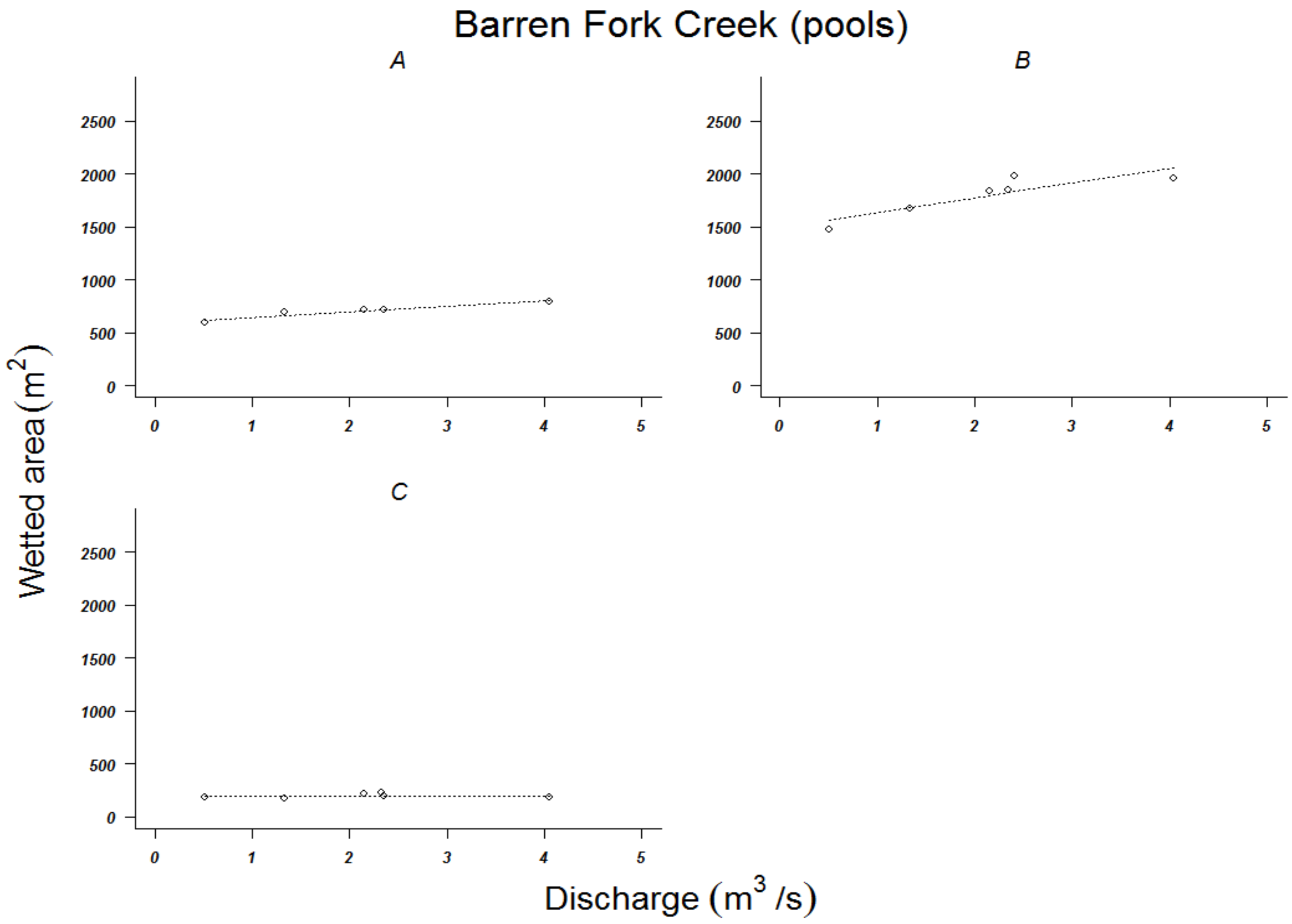


Figure 7.—Wetted area (m^2) to discharge (m^3/s) by channel unit used in broken-line regressions. Solid lines are breakpoint estimates with 90% confidence levels. Dashed lines are segmented-linear models or linear models (where no significant breakpoint was found).

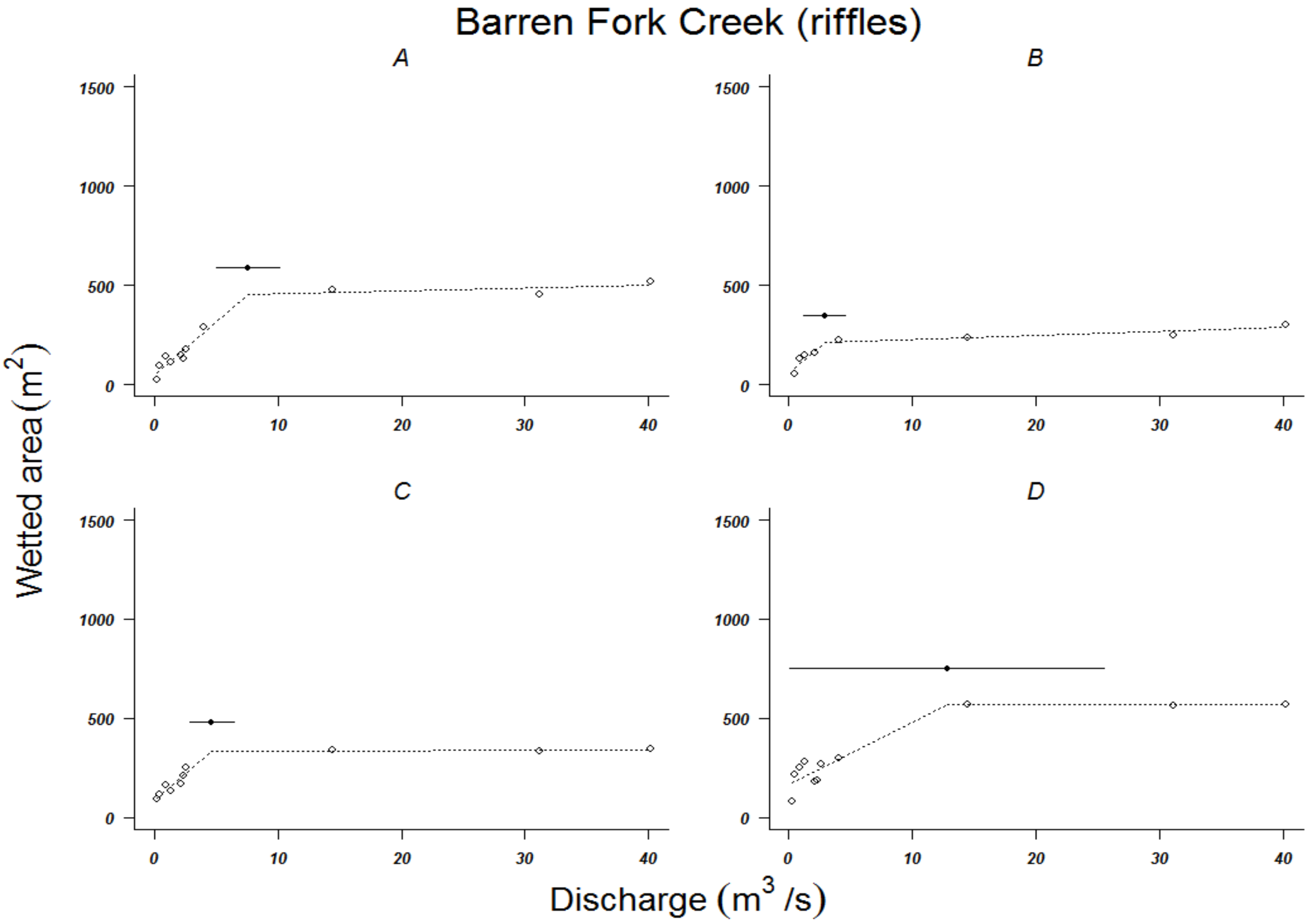


Figure 8.—Wetted area (m^2) to discharge (m^3/s) by channel unit used in broken-line regressions. Solid lines are breakpoint estimates with 90% confidence levels. Dashed lines are segmented-linear models or linear models (where no significant breakpoint was found).

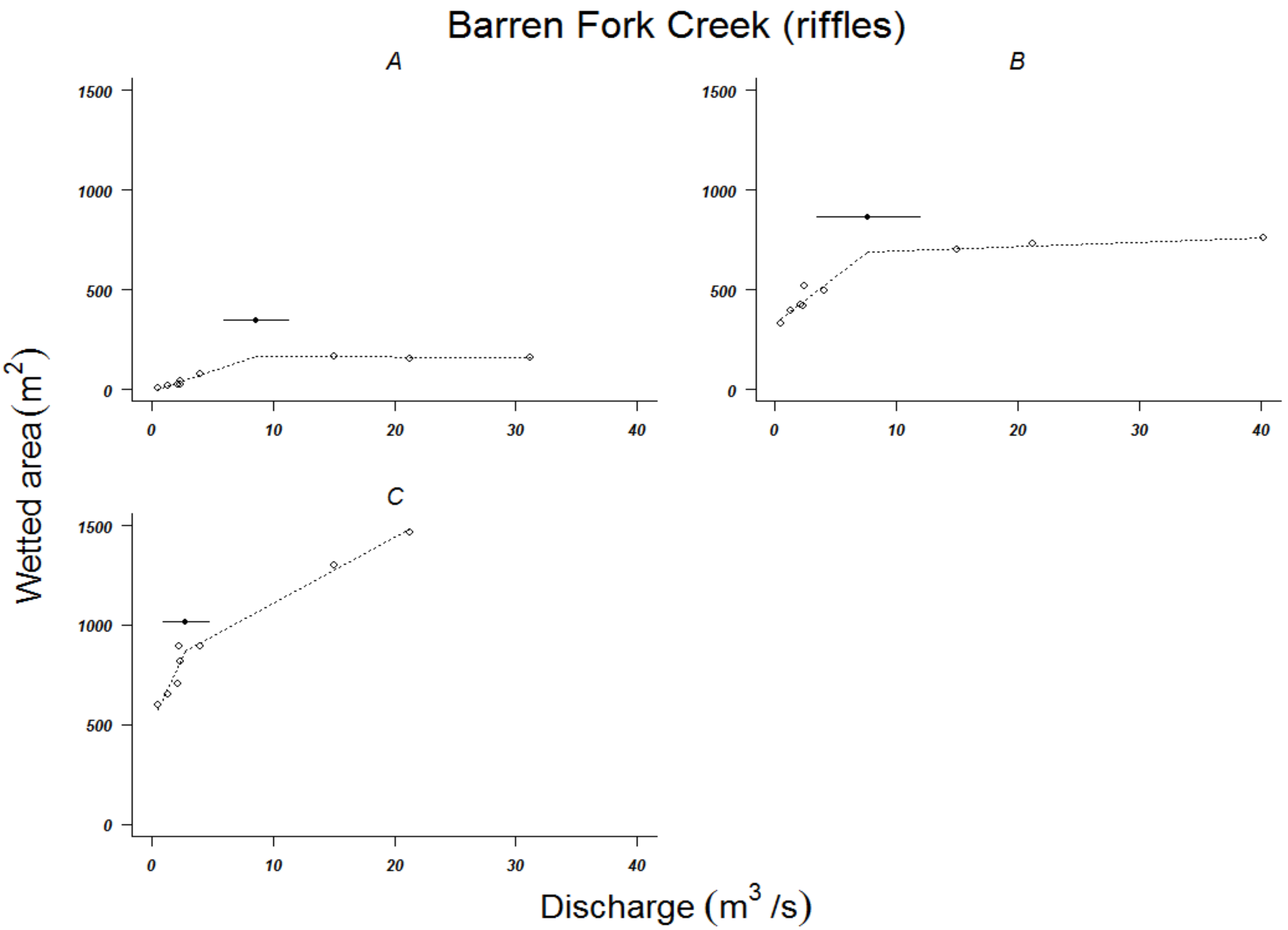


Figure 9.—Wetted area (m^2) to discharge (m^3/s) by channel unit used in broken-line regressions. Solid lines are breakpoint estimates with 90% confidence levels. Dashed lines are segmented-linear models or linear models (where no significant breakpoint was found).

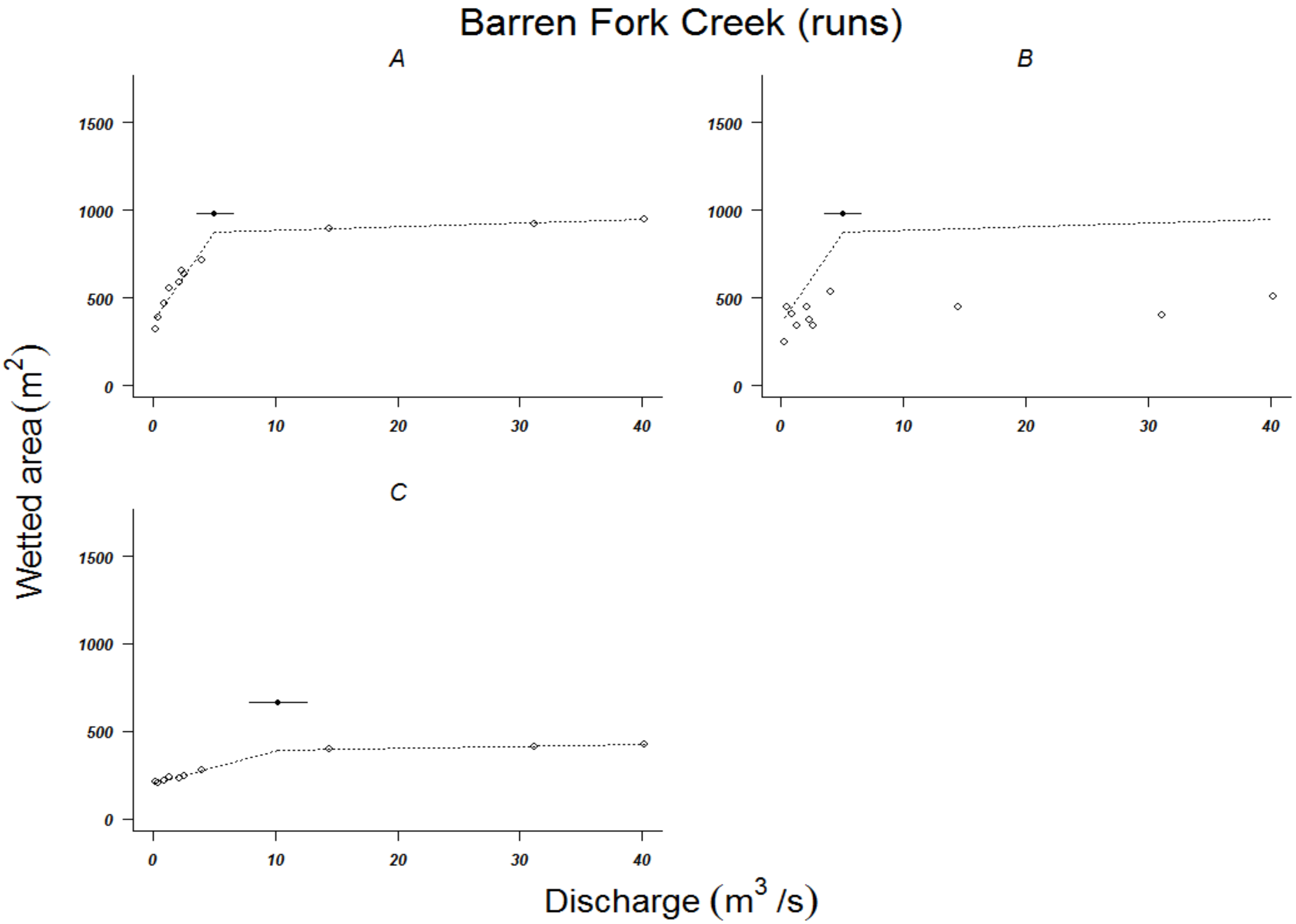


Figure 10.—Wetted area (m^2) to discharge (m^3/s) by channel unit used in broken-line regressions. Solid lines are breakpoint estimates with 90% confidence levels. Dashed lines are segmented-linear models or linear models (where no significant breakpoint was found).

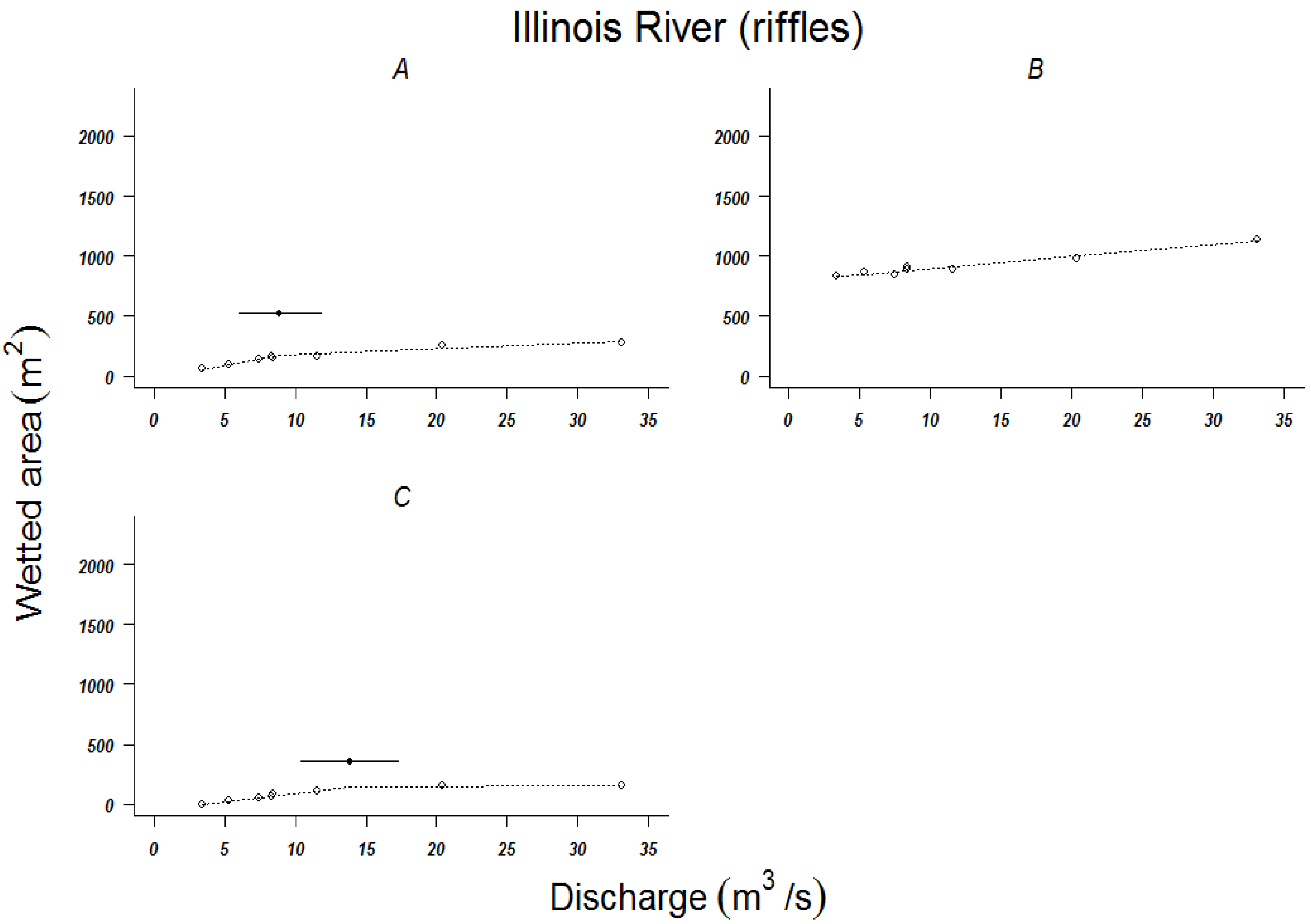


Figure 15.—Wetted area (m^2) to discharge (m^3/s) by channel unit used in broken-line regressions. Solid lines are breakpoint estimates with 90% confidence levels. Dashed lines are segmented-linear models or linear models (where no significant breakpoint was found).

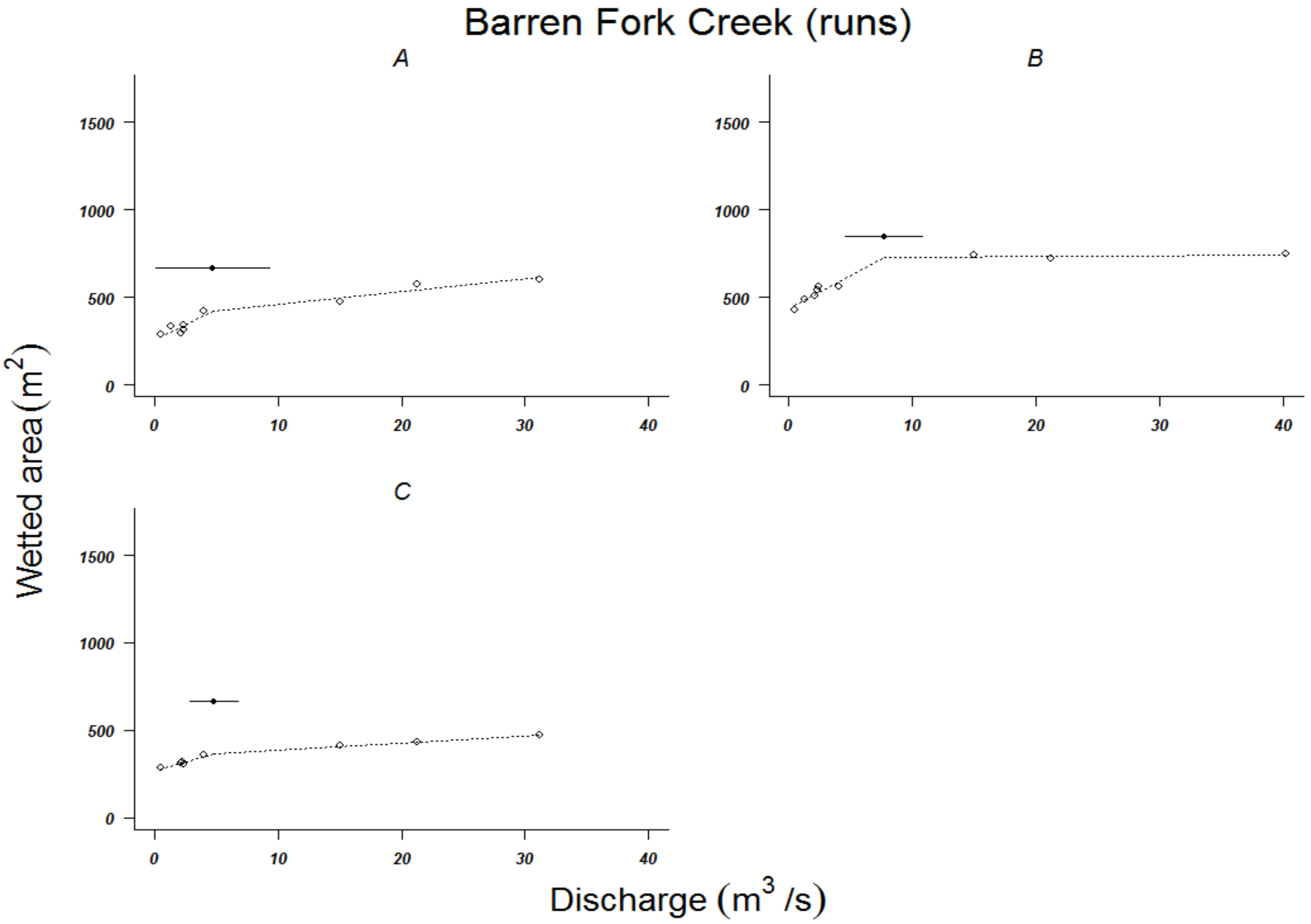


Figure 11.—Wetted area (m^2) to discharge (m^3/s) by channel unit used in broken-line regressions. Solid lines are breakpoint estimates with 90% confidence levels. Dashed lines are segmented-linear models or linear models (where no significant breakpoint was found).

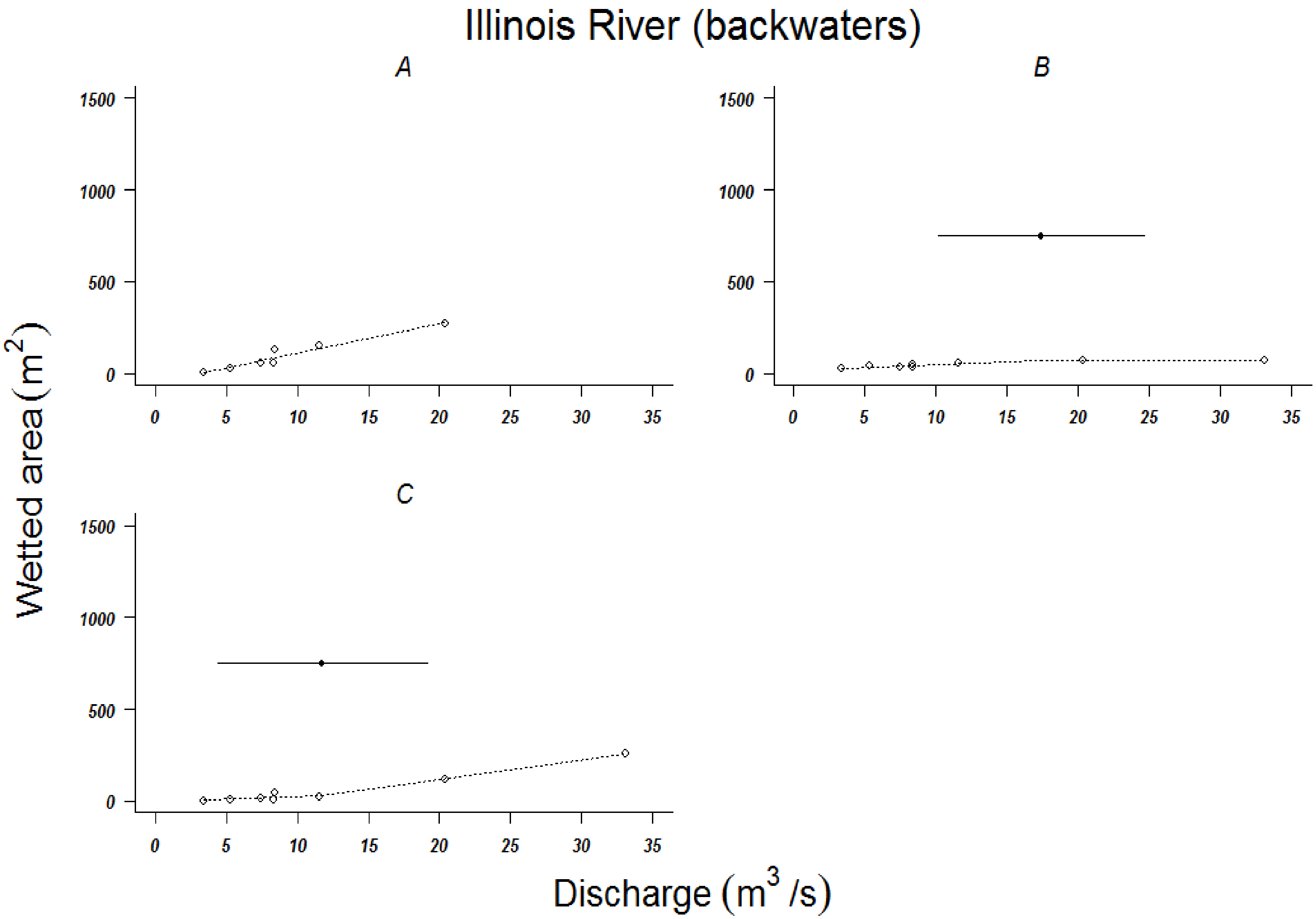


Figure 12.—Wetted area (m^2) to discharge (m^3/s) by channel unit used in broken-line regressions. Solid lines are breakpoint estimates with 90% confidence levels. Dashed lines are segmented-linear models or linear models (where no significant breakpoint was found).

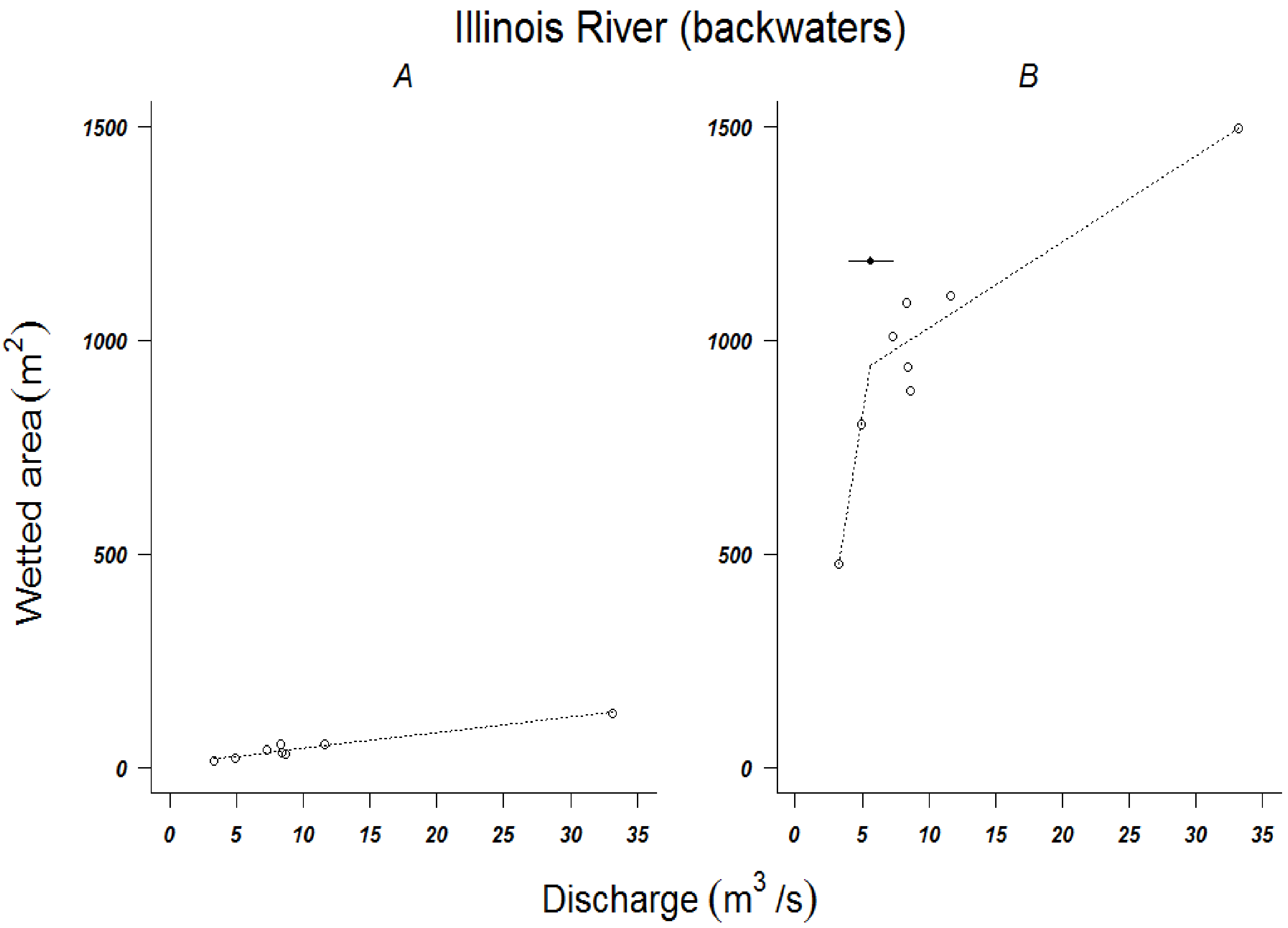


Figure 13.—Wetted area (m^2) to discharge (m^3/s) by channel unit used in broken-line regressions. Solid lines are breakpoint estimates with 90% confidence levels. Dashed lines are segmented-linear models or linear models (where no significant breakpoint was found).

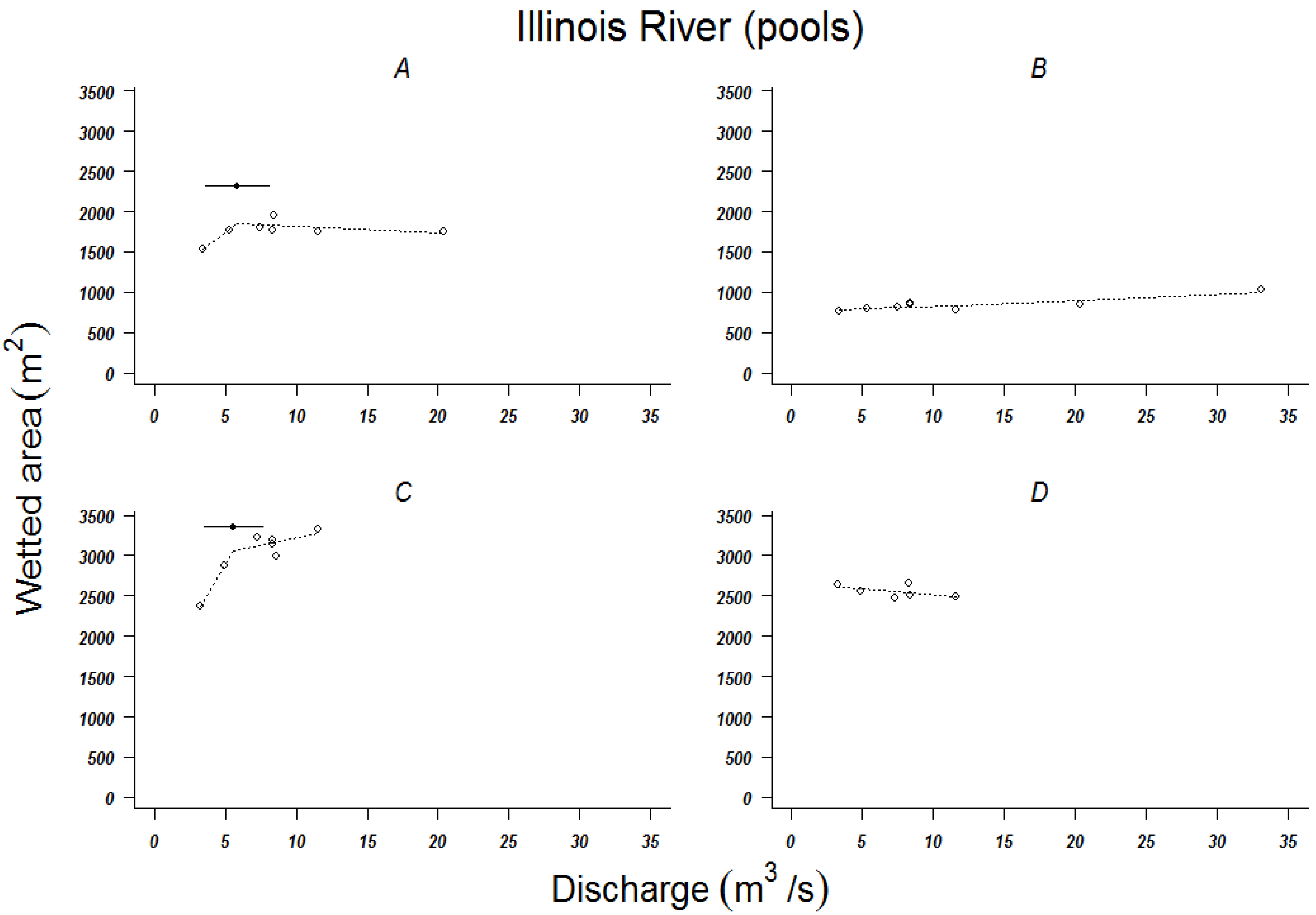


Figure 14.—Wetted area (m^2) to discharge (m^3/s) by channel unit used in broken-line regressions. Solid lines are breakpoint estimates with 90% confidence levels. Dashed lines are segmented-linear models or linear models (where no significant breakpoint was found).

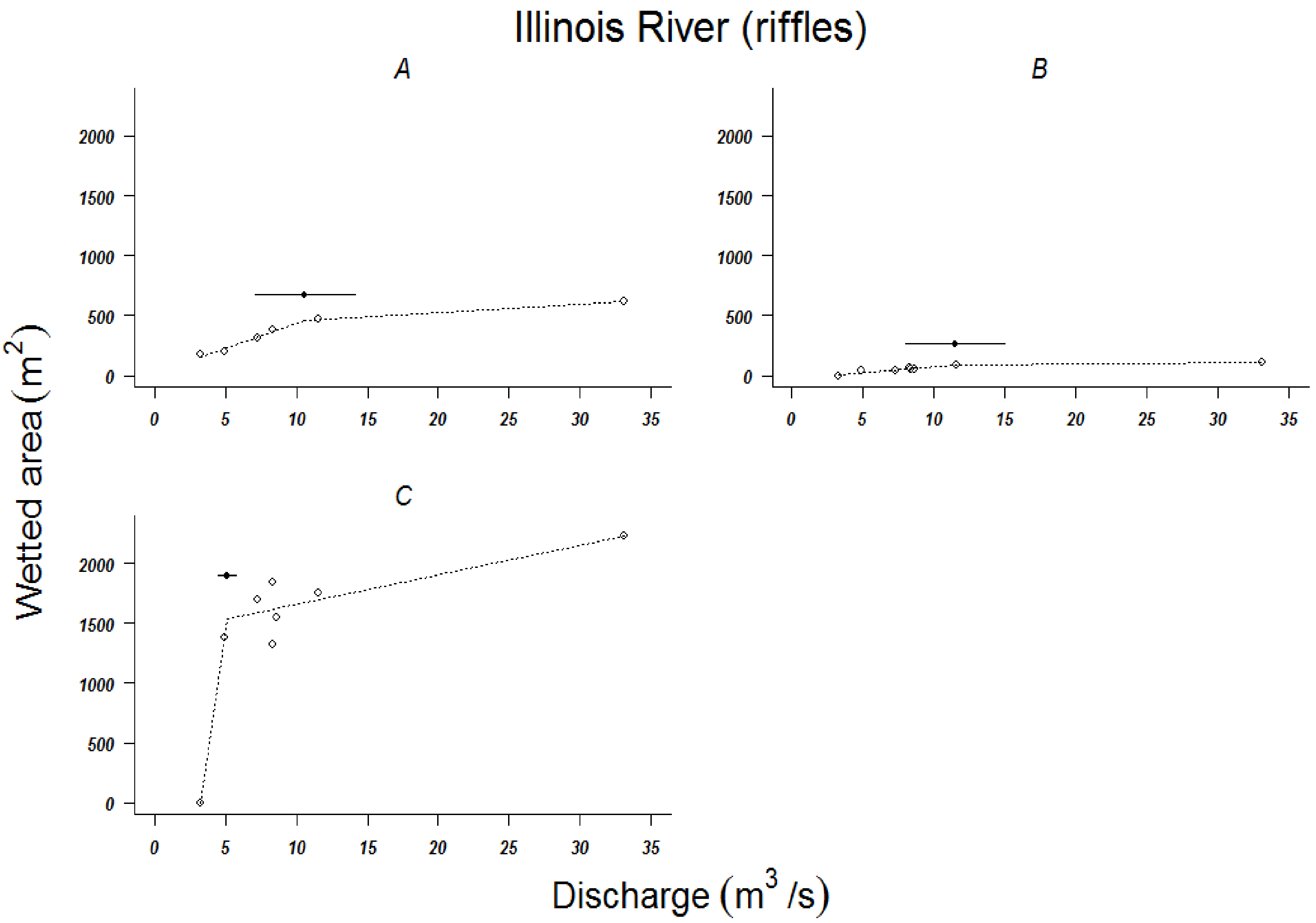


Figure 16.—Wetted area (m^2) to discharge (m^3/s) by channel unit used in broken-line regressions. Solid lines are breakpoint estimates with 90% confidence levels. Dashed lines are segmented-linear models or linear models (where no significant breakpoint was found).

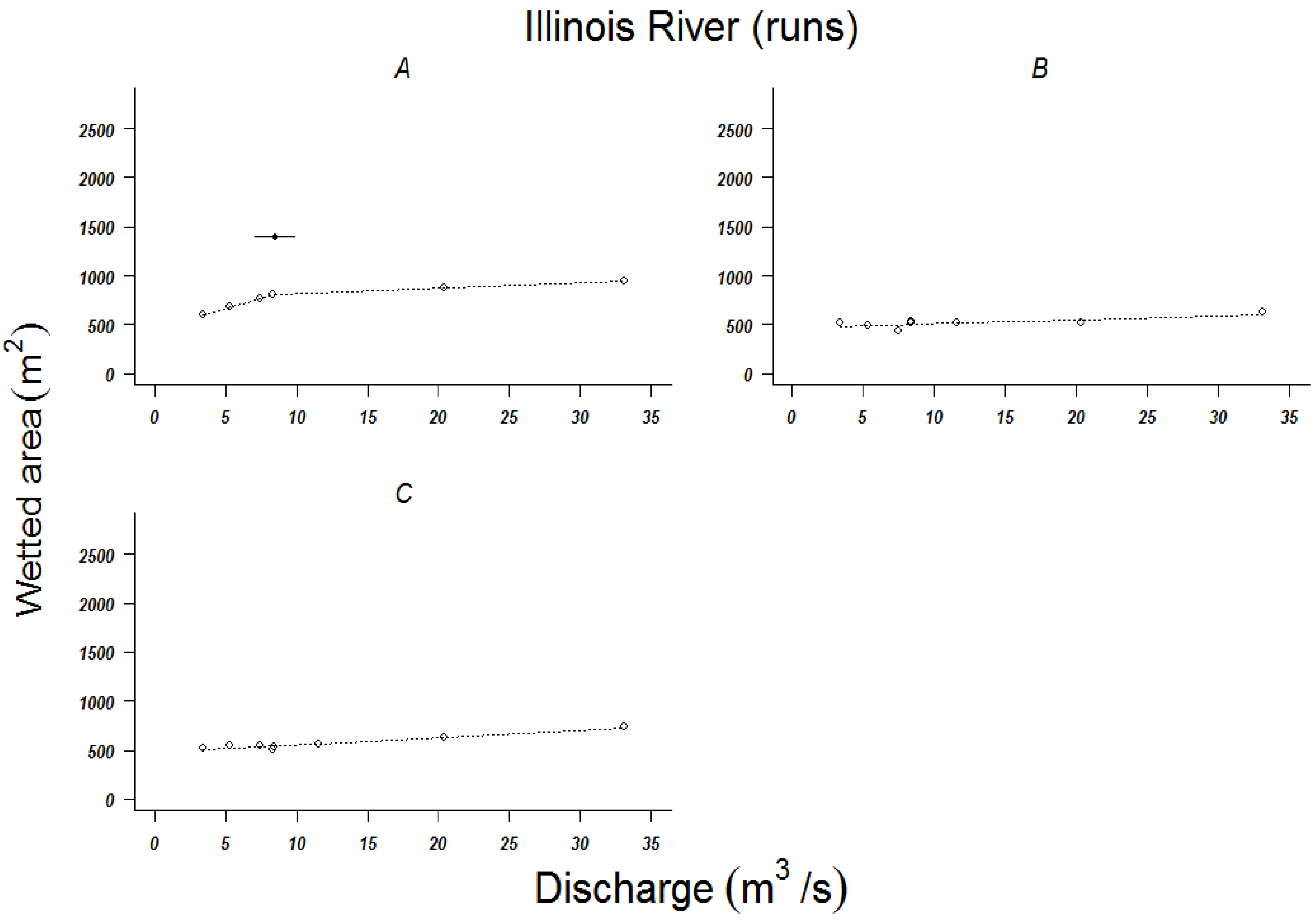


Figure 17.—Wetted area (m^2) to discharge (m^3/s) by channel unit used in broken-line regressions. Solid lines are breakpoint estimates with 90% confidence levels. Dashed lines are segmented-linear models or linear models (where no significant breakpoint was found).

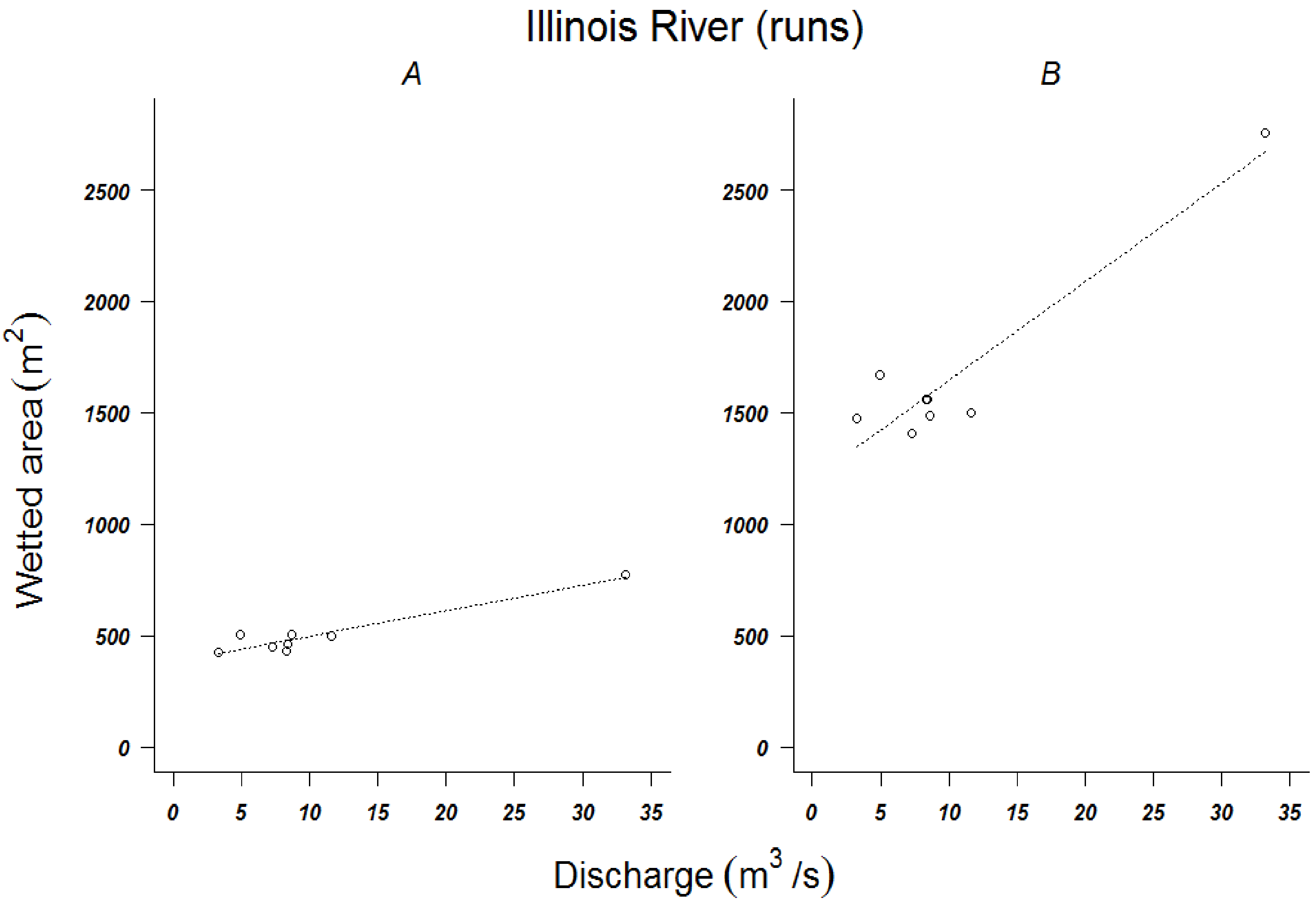


Figure 18.—Wetted area (m^2) to discharge (m^3/s) by channel unit used in broken-line regressions. Solid lines are breakpoint estimates with 90% confidence levels. Dashed lines are segmented-linear models or linear models (where no significant breakpoint was found).