INSTREAM FLOW ASSESSMENT OF STREAMS DRAINING THE ARBUCKLE-SIMPSON AQUIFER

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Abstract

The availability of high quality water is critical to both humans and ecosystems. A recent proposal was made by rapidly expanding municipalities in central Oklahoma to begin transferring groundwater from the Arbuckle-Simpson aquifer, a sensitive sole-source aquifer in south-central Oklahoma. Concerned citizens and municipalities living on and getting their drinking water from the Arbuckle-Simpson lobbied the legislature to pass a temporary moratorium on groundwater transfer to allow for a comprehensive study of the aquifer and its ecosystems. We conducted an instream flow assessment using Physical Habitat Simulation (PHABSIM) on springs and streams with four spring-dependent species: two minnows, southern redbelly dace (*Phoxinus erthyrogaster*) and redspot chub (*Nocomis asper*); and two darters, least darter (*Etheostoma microperca*) and orangethroat darter (*Etheostoma spectabile*). Spring habitats are unique compared to other river habitats because they have constant flow and temperature, small and isolated habitat patches, and a general lack of predators.

Our study sites included two spring-fed streams, one larger stream with high groundwater inputs, and a river with both groundwater and surface water inputs that is adjacent to the small spring-fed streams. These habitats meet the criteria for groundwater dependent ecosystems because they would not exist without the surface expression of groundwater. A total of 99 transects in all four sites were surveyed for channel elevation, and three sets of water surface elevation and water velocity were measured. Habitat suitability criteria were derived for the species at each site using nonparametric confidence limits based on underwater observations made by snorkelers. Simulations of flow were focused on declines in discharge, which is the expected effect of the proposed groundwater diversion.

Our results show that only a small proportion of the total available area in each habitat is considered to be preferred habitat (Weighted Usable Area [WUA]) by the four target species. In the spring habitats, a maximum of 10% of the total area is preferred habitat and that dropped to as little as 3% with decreased flows. The quantity of WUA decreased when lower discharges were simulated for all the target species. Declines in the small amount of habitat that is already available would likely degrade these populations of fishes. In the larger river habitat, the highest WUA occurred at the lowest discharge, which leads us to conclude that in the event of dewatering of the spring habitats, the river should provide some refuge habitat for spring dependent species.

Based on the findings of this study, groundwater removal from the aquifer near springs may have adverse impacts on fish habitat availability for spring dependent fish populations if seasonal trends in spring discharge are not maintained (higher in winter and lower in late summer). Quantifying the relationship of streamflow between gaged and ungaged springs in the Arbuckle-Simpson is a possible method to monitor and maintain flows in springs.

Introduction

The form and function a river naturally exhibits is the result of complex interaction between three broad groups of master parameters—landscape, flow regime, and sediment regime (Leopold 1994). Unperturbed, natural streams typically experience a range of values ("natural range of variability") for each set of three master parameters (Leopold 1994; Thorne et al. 1997). Consistent perturbation of one or more of the master parameters outside the natural range of variability will result in significant adjustments in river structure and morphology as the stream attempts to adjust its form to be consistent with a new range of parameter values (Rosgen 1996; Thorne et al. 1997). Typically, adjustments of rivers to changes in master parameters are not beneficial to sustainable human and ecosystem functions of the river (Rosgen 1996) and will result in sustained and severe degradation of a river system's form and function.

Water development projects create hydrologic alterations to a river that affect the magnitude and timing of natural river flows (Rosenberg et al. 2000). These alterations modify both the structure and function of river ecosystems (Poff et al. 1997; Postel and Richter 2003; Rosenberg et al. 2000) impacting the habitat and survival of aquatic organisms including fishes, invertebrates, and plants. For example, withdrawals from the Edwards Aquifer in southcentral Texas, the sole source of water for San Antonio, have quadrupled from the early 1930s to the 1980s and are threatening the survival of the endangered fountain darter *Etheostoma fontinala* as well as other federally threatened and endangered aquatic species that are spring dependent (Fitzhugh and Richter 2004; Hamilton et al. 2003). To maintain the ecological integrity of rivers, their flows should be managed to mimic the natural flow regime (Poff et al. 1997; Richter et al. 2003; Richter et al. 2006).

Methods for assessing the impacts of flow alterations from water development projects on stream habitats have evolved over the past 30 years from standard-setting techniques (e.g., minimum flow, Tennant method, wetted perimeter), which develop a low flow standard or seasonal standards that may or may not have particular aquatic habitat benefits, to incremental techniques (i.e., the Instream Flow Incremental Methodology, IFIM) in which aquatic habitats beneficial to fish and other aquatic organisms are quantified as a function of stream discharge (Stalnaker et al. 1995). The most commonly applied and comprehensive instream flow assessment technique used by state and federal agencies is the IFIM (Armour and Taylor 1991; Reiser et al. 1989). The IFIM provides an organizational framework for evaluating and formulating alternative water management options when managing stream flows (Bovee 1982; Bovee 1986). It consists of several phases including: (1) legal-institutional analysis which involves problem identification and analysis of the physical system, (2) study plan development, (3) study implementation through macrohabitat (water quality, temperature, channel morphology, discharge) and microhabitat (depth, velocity, substratum, cover) suitability modeling, (4) project alternatives analysis and evaluation and (5) problem resolution through negotiation (Stalnaker et al. 1995). Successful implementation of IFIM requires sequential execution of all five phases.

The springs and spring-fed creeks of the Arbuckle-Simpson aquifer are unique habitat types that depend extensively on groundwater. Groundwater dependent ecosystems are those habitats that depend on the surface expression of groundwater to maintain their species composition and habitat quality (Eamus et al. 2006; Sophocleous 2007). Spring habitats differ from other lotic (i.e. flowing water) systems in four crucial factors: constant flow and temperature, existing as small and isolated habitat areas, and a general lack of large predators (Glazier 1991). Spring habitats can provide important refuge habitat for fish species that are sensitive to high temperatures or low water clarity, and springs may also provide feeding habitat and an escape from predators for species living in the larger, adjacent rivers (Meyer et al. 2007). This study provides descriptions of physical habitat along with ecological information on the use of springs as fish habitat in groundwater dependent ecosystems in Oklahoma.

Objective

The objective of this study was to use the Instream Flow Incremental Methodology (IFIM) to assess instream flow requirements of selected fishes in the Blue River near Connersville, Oklahoma and Pennington Creek near Reagan, Oklahoma. We used IFIM and Physical Habitat Simulation System (PHABSIM) to model cyprinid (minnows and shiners) and percid (perch and darters) habitat. This study provides information to the Oklahoma Water Resource Board (OWRB) that will allow the OWRB to account for the impacts of groundwater withdrawal on fish habitat in streams of the Arbuckle-Simpson. We will also present information on requirements and habitat use by spring dependent species, which will improve the understanding of groundwater dependent ecosystems in Oklahoma.

Problem identification

Problem statement

The Arbuckle-Simpson aquifer encompasses over 500 mi^2 in southcentral Oklahoma and is the primary source of water for Ada, Sulphur, and other towns in the region (OWRB 2003). In early 2002, the Central Oklahoma Water Authority proposed to pump up to 80,000 acre-feet of water from the aquifer to communities in Canadian County. In 2003, the Oklahoma State Legislature passed Senate Bill 288 (SB 288), which imposed a moratorium on the issuance of temporary groundwater permits for municipal and public water supplies outside of any county in the state that overlays in whole or in part a sensitive sole source groundwater basin. A specific requirement for permit approval, as stated in SB 288, was that the proposed use of water would not degrade or interfere with springs or streams emanating from the Arbuckle-Simpson aquifer. Although the ecological services of streams (recreational use and habitat for biological species) emanating from the aquifer were not identified in SB 288, their value is being considered in the development of a water management plan for the Arbuckle-Simpson aquifer (OWRB 2003).

Hydrologic time series

The Indicators of Hydrologic Alternation (IHA) analysis of the Arbuckle-Simpson springs and streams completed by The Nature Conservancy (see IHA section, this report) provides a detailed report on spring and stream flows. The general seasonal trends in discharge are similar at all of the sites due to the dependence on groundwater flow from a shared aquifer, although watershed size and precipitation that contribute runoff flow also have a large influence on discharge in the more riverine sites (e.g. Spring Creek and Blue River). Springs (e.g., Byrds Mill Spring near Fittstown) exhibit a modest seasonal trend in median discharge, which is slightly higher in March to May and lower during the remainder of the year (IHA: Fig. 41). Pennington Creek (near Spring Creek) and Blue River have the highest median discharge from March to June and lowest median discharge from August to October (IHA: Fig. 31 and Fig. 10, respectively). The USGS stream gages (see IHA section, this report) at Byrds Mill Spring (Spring 2 and Spring 3), Pennington Creek at Reagan (Spring Creek), and Blue River at Connerville (Blue River) estimate discharge within the study area. Because these gages are not located within the boundaries of the study sites, we estimated discharge to better reflect the conditions occurring at the study sites. We used median discharge to represent "baseline" conditions at each site, the 25th Quartile as dry years, and 75th Quartile as wet years. Stalnaker et al. (1995) defined "baseline" as the water supply, habitat values, or population status conditions that occur during a reference (i.e., recent historical) timeframe.

Target species

Four fish species were used in the instream flow analysis of the Arbuckle-Simpson springs and streams. Two species are from the Cyprinidae family (minnows and shiners), the southern redbelly dace *Phoxinus erthyrogaster* and the redspot chub *Nocomis asper*, and two species from the Percidae family (perch and darters), least darter *Etheostoma microperca* and orangethroat darter *Etheostoma spectabile* (Miller and Robison 2004). Three of the species (southern redbelly dace, redspot chub, and least darter) were selected because they are limited to only a few watersheds in southern Oklahoma, which is also the southern limit of their distribution. Southern redbelly dace range from the Northeast to Midwest, and south to Mississippi but they have been reported in a few watersheds in southern Oklahoma (Miller and Robison 2004). Redspot chub have a small geographic distribution, primarily in the upland drainage of the Arkansas River and in the Ozark Mountains, but are found as an isolated population in the Blue River watershed (Robison and Buchanan 1988). The least darter is primarily found in the Great Lake states, with populations in the Ozark Mountains. The only watershed containing least darter in southern Oklahoma is the Blue River (Miller and Robison 2004). The fourth species, orangethroat darter, is widespread throughout the central United States and was present in the Spring Creek study site, so, in the absence of least darters, it was included in the analysis (Miller and Robison 2004). These species prefer clear and cool water in spring-fed streams with southern redbelly dace, redspot chub, orangethroat darter on gravel substrate, and least darter in areas of dense vegetation (Miller and Robison 2004; Robison and Buchanan 1988).

The fish community of the springs is composed primarily of small-bodied fishes, such as minnows, darters, and mosquitofish. Matthews et al. (1985) sampled 50 springs located throughout Oklahoma, finding only 19 that contained fishes, of which five were located in the Connerville area of the Blue River watershed. Species collected in the Arbuckle-Simpson springs included: central stoneroller *Camostoma anomalum*, southern redbelly dace, western mosquitofish *Gambusia affinis*, green sunfish *Lepomis cyanellus*, least darter, and orangethroat darter (Matthews et al. 1985). Two other studies found southern redbelly dace, least darter, and orangethroat darter at springs in the Connerville area in the late 1940s (Linder 1955) and more recently in the late 1990s by Chad Stinson (Personal Communication, William Stark, Fort Hays State University, Kansas). Other nearby streams containing southern redbelly dace include Byrds Mill Creek, a spring-fed creek flowing from the Arbuckle-Simpson and located east of the Blue River (Stewart et al. 1992), and Mill Creek, a stream located to the west of Pennington Creek (Binderim 1977). In Byrds Mill Creek, southern redbelly dace were only present near the spring and not downstream (Stewart et al. 1992) and in Mill Creek, southern redbelly dace only occupied spring-fed tributaries (Binderim 1977). Orangethroat darters were found in the highest abundance in the headwaters of Byrds Mill Creek and Mill Creek (Binderim 1977; Stewart et al. 1992). Neither redspot chub nor least darter were reported in either study.

Methods

Site establishment

Establishment of the PHABSIM study sites consisted of four activities: (1) defining the lower and upper site boundaries of the study stream segment (i.e., a relatively long stream section with a geographically homogeneous flow regime); (2) subdividing the segment into reaches, or sites (i.e., short stream sections that contains multiple mesohabitat [i.e., riffles, runs and pools] types) in which microhabitat (depth, velocity, substrate, cover, stream bed and water surface elevation) variables were measured across transects; (3) establishing horizontal control; and (4) establishing vertical control (Bovee 1986; Bovee 1994).

We determined segment boundaries by first mapping the channel with Trimble GeoXT GPS. After defining the segment boundaries, we visually classified mesohabitat types while walking the stream segment (Toepfer et al. 2000) in May 2007. Next, we identified distinct habitat types and their boundaries within each study site. The downstream site boundaries are the most important (Bovee 1994). The lower boundary of each site was placed near a hydraulic control. A hydraulic control is a feature in the stream channel (e.g., narrowing of the channel below a pool) that creates a backwater effect on upstream transects. The lower boundary for Spring 2 and Spring 3 sites was placed near the confluence with the Blue River at a narrowing of the channel. The lower boundaries for the Spring Creek and Blue River sites were placed at the head of riffles where the water became shallow. These sites allowed us to sample different sets of mesohabitats along the length of each study site.

Transects were placed within each site to identify available microhabitat characteristics needed to describe and model all the habitat features. From 2-5 transects were systematically placed across each mesohabitat type to describe the longitudinal stream cells based on depth, velocity, cover, and substrate characteristics. A total of 99 transects were placed within the four study sites (Table 1). Spring 2 and Spring 3 each had 25 transects, while Spring Creek had 29 transects and Blue River had 20 transects (Table 1). Transects were placed in the Blue River site within 300 ft upstream and downstream from the mouth of Spring 2 or Spring 3.

Horizontal control measurements for PHABSIM modeling are the distance between transects and the relative length of stream cells that define a site. We obtained these data by measuring the distance between pins of one transect to those of another transect and to an established benchmark. Distances and angles to different transect pins or benchmarks were measured with a total station (Topcon GTS-235W Electronic Total Station) and prism pole. Total stations use a combination of an electromagnetic distance meter (using infra-red radiation to measure distance), and electronic theodolite (to measure horizontal and vertical angles), and have the ability to log data on the instrument (Schofield 1993).

Vertical control measurements within a site are critical for PHABSIM modeling. These measurements are used to calculate slopes and energy transfer between transects. All of the elevations in a site must be referenced to a common datum. This process involved the installation of multiple permanent benchmarks at a site and relating their elevations by differential leveling. The purpose of benchmarks is to allow a backsight to a known elevation from anywhere in the site. The downstream-most benchmark at each site was arbitrarily set at 100.00 feet. Completion of the vertical control measurements involved conducting a level loop. When the last benchmark in each site was surveyed, the complete survey of benchmarks was completed in reverse. This was done to check for errors in elevations, or to "close the loop" (Bovee 1994).

Site Description

The springs and streams we studied were located in north central Johnston County in southern Oklahoma (Figures 1 and 2). Although the Blue River and Pennington Creek sites are only 6 miles apart (by air), the Blue River flows into the Red River and Pennington Creek empties into the Washita River. Spring 2 is the most meandering of the study sites (Figure 1, inset A), while Spring 3 (Figure 1, inset B), Blue River (Figure 1), and Spring Creek (Figure 2) had a straight course. The spring sites had similar total lengths (Spring 2, 714.1 ft and Spring 3, 862.0 ft), while Spring Creek was longer (1335.2 ft) and the Blue River site was the longest at 1512.1 ft (Table 1). Mean wetted width in the springs was similar at 13.7 ft (Spring 2) and 13.5 ft (Spring 3) with similar range of width in both springs (Table 1). Spring Creek was wider than the springs (32.6 ft), while the Blue River had the widest channel (53.4 ft; Table 1).

We identified 10 substrate types (6 single substrates and 4 combinations of substrates) and 6 groups of instream cover (Table 2). Cover types included rocks (cobble and boulders) and three different types of vegetation: emergent vegetation (e.g. arrowhead *Sagittaria* spp, water willow *Justicia americana*), floating vegetation (waterlily *Nymphaea odorata*), and submergent vegetation (coontail *Ceratophyllum demersum*, watercress *Rorippa* sp., pondweed *Potamogeton* spp.). Woody debris included roots, stumps, and piles of small woody debris (i.e. small sticks; Table 2). We created a channel index code for each cell on a transect and each observation of habitat use. The channel index uses the number to the left of the decimal point to represent the substrate code and the number to the right of the decimal place to represent the cover.

Water Temperature

We measured hourly water temperature in Spring 2, Spring 3, and Spring Creek using a Solinst Levelogger (Figure 3). Timing of the probe deployment resulted in different start times for temperature measurement: 405 days at Spring 3 (4/19/07-5/27/08), 372 days at Spring 2 (5/22/07-5/27/08), and 292 days at Spring Creek (8/10/07-5/27/08).

Transect Profile Data

Channel cross-sections were described as a series of x and y coordinates called verticals. Channel profile data associated with each vertical included a horizontal and vertical distance from a known datum measured to the nearest 0.001 ft, water surface elevation measured to the nearest 0.001 ft, and descriptions of the cover and substrate in that cell (Bovee 1994). Cover and substrate information was coded and transformed into channel index codes during data entry (Table 2). In addition to these measurements, velocity was measured with a flowmeter (Marsh-McBirney Model 2000) attached to a top-setting wading rod at each vertical point. For depths less than 2.46 ft (0.75 m), a single velocity measurement was made (40 second interval) at 60% of the depth at that vertical. For depths over 2.46 ft (0.75 m), two measurements were taken, one at 20% of the total depth and one at 80%, and these two velocity measurements were averaged to obtain a single value for that vertical (Bovee 1994). Three sets of measurements at each transect in each site were taken on different days and at different streamflows, which allowed us to produce a stage-discharge relationship for each site.

Fish sampling

Habitat use by the target species was identified by visual observation with snorkeling gear (Thurow 1994). Snorkel surveys were well suited for the springs and streams of the Arbuckle-Simpson because high water clarity made observing habitat use feasible and a single snorkeler could survey the small, narrow habitats. The species were visually distinct from the other species present and could be correctly identified underwater. At the Spring Creek site, southern redbelly dace were divided into juvenile (<40mm) and adult (>40mm) size classes. The snorkeler would move upstream in a zigzag pattern and drop brightly colored markers at each location that a target species was occupying. A second researcher would follow, at a sufficient distance to avoid disturbing the fish behavior, to collect depth, velocity, and channel index data at each point that fish used. To increase the power of our habitat use data, we conducted multiple snorkeler passes over the entire site (i.e. "pseudo-replicates"). We made three passes for southern redbelly dace (163 observations) and four passes for redspot chub (143 observations) at Spring 2. Although we completed three passes for least darter in Spring 2, only 13 observations of habitat use were recorded, so least darter was not modeled at that site. In Spring 3, we completed three passes for southern redbelly dace (65 observations) and three passes for least darter (251 observations). In Spring Creek, three passes were completed for adult southern redbelly dace (254 observations) and juvenile southern redbelly dace (157 observations), while only one pass was needed for the orangethroat darter (190 observations).

Physical habitat simulations

For each variable (depth, velocity, substrate/cover), habitat suitability criteria (HSC) curves were constructed for each group using non-parametric tolerance limits. For small sample sizes, non-parametric tolerance limits are preferred over nonlinear regression techniques (Bovee 1986). Habitat suitability index curves were classified into three quality classes (optimal, usable and suitable) using the following equation:

$$NSI=2(1-P),$$

where NSI is the normalized suitability index and P is the proportion of the population under the curve (i.e., the 50%, 75%, and 95% ranges). Optimal habitat encompassed the central 50% (NSI=1.0), usable habitat encompassed the broader central 75% (NSI=0.5) of the observations, and the broadest range of habitat, suitable, contained the locations within the 95% (NSI=0.1) range (Bovee 1986). These suitability curves were then entered into the physical habitat simulation model (PHABSIM, Windows version beta-2) to determine habitat quality and quantity during microhabitat simulation. Discrete HSC were developed for the target species at Spring 2, Spring 3, and Spring Creek. Habitat use data were not collected in the Blue River, so existing HSC were used. Because HSC for southern redbelly dace were developed at all three sites, we used the maximum and minimum values for each of the optimal, suitable, and usable values when modeled in the Blue River.

Hydraulic simulations

We modeled microhabitat at each site with PHABSIM. The PHABSIM was used to predict hydraulic conditions at unmeasured discharges. Water surface elevations were determined for simulation discharges using a combination of a "stage discharge" (STGQ) model and a "step-backwater" (WSP) model. The STGQ model was used in the lower reaches of Spring 2 and Spring 3, and for all transects in Spring Creek and Blue River. The STGQ method predicts water surface elevation by deriving constants from a regression between the log of discharge and the log of water surface elevation (minus stage zero flow). Each cross section is considered independent of all other cross sections and is modeled as such. The model is tested by comparing the simulated and observed water surface elevations at the field measured discharges (Waddle 2001).

We used the WSP model for the upper transects of Spring 2 (transects: 637, 651, 714) and Spring 3 (transects: 614, 638, 682, 746, 823). The WSP model more closely simulates flows in pool habitats. The WSP model assumes that each cross section is affected by a downstream hydraulic control that creates a backwater effect. All cross sections in a site must be tied together to a common benchmark to effectively use this method. The process of using the WSP model involves selecting coefficient values, "Manning's n", that best fit the water surface elevations at each cross section for the highest calibration flow (Waddle 2001).

The simulated water surface elevations for Spring 2 were variable and at one transect ranged from as little as 0.5 ft to as much as 1.8 ft between lowest and highest simulated discharge. The mean elevation difference between lowest and highest discharge was 0.8 ft (Figure 4a). There was a smaller range between lowest and highest simulated water surface elevation in Spring 3, from 0.2 ft to 1.0 ft and a mean difference of 0.6 ft (Figure 4b). Spring 2 has a lower channel gradient over its length than Spring 3, but Spring 2 has several deeper pools (Figure 4). During the period of data collection in Spring 3, beavers *Castor canadensis* modified water surface elevation through dam construction. Because this occurred late in the season, the highest water surface elevation occurred when discharge was lowest. To correct this problem, we reordered the recorded data to reflect the trend of higher water surface elevation and discharge earlier in the season followed by a decline in to the fall. Thus, we are able to model the conditions that would generally occur in Spring 3 and other springs in the Arbuckle-Simpson region.

Spring Creek had the largest difference in water surface elevation between lowest and highest simulated discharge with a mean difference of 1.7 ft and range of 1.1 to 2.3 ft (Figure 5a). Blue River had a mean difference between lowest and highest simulated discharge of 1.2 ft and a range from 0.7 to 1.5 ft (Figure 5b).

Weighted Usable Area (WUA) is the area of the stream in the wetted channel weighted to the suitability of the habitat (i.e. depth, velocity, and channel index) of the species of interest (Stalnaker et al. 1995). The WUA is standardized as square feet per 1,000 feet ($ft^2/1000ft$) of the stream. WUA for each site was determined after simulating flows between 0.5 cfs and 14 cfs in the Springs, between 0.5 cfs and 100 cfs in Spring Creek, and between 5 cfs and 165 cfs in Blue River. Each WUA site estimate was then weighted by site length to obtain a single WUA value for each flow and target species. These values were then plotted to determine the maximum habitat available and at what flow this maximum occurred. The point at which this maximum occurred would then be considered the critical flow for future microhabitat simulations.

Hydrographic time series estimate

Our study sites were not located adjacent to USGS stream gage stations, so we estimated flow in the sites. We used the relationship between the gage data and measured discharge to estimate discharge for the time periods of the gages and to calculate monthly statistics for each site. This is a simplistic estimate of discharge but for our purposes it is appropriate for the range of conditions we were interested in (i.e., low flows and monthly mean conditions). We collected habitat use data for the target species from a narrow range of conditions, so these data are not directly applicable to higher flows and simulated higher flows. There is also a threshold of high water velocity where it becomes dangerous to collect velocity and water surface elevation data in the stream, and we did not collect data at these higher flows.

We collected stage height at Spring 2, Spring 3, and Spring Creek with a Solinst Levelogger (Solinst Canada Ltd., Georgetown, Ontario). Beaver activity in Spring 2 and Spring 3 caused changes in water depth that were not related to discharge, so we used the baseflow from the USGS gage at Blue River at Connerville (#07332390 Baseflow for the Blue River gage was estimated using the BFI software (Bureau of Reclamation; http://www.usbr.gov/pmts/hydraulics_lab/twahl/bfi/). We measured discharge on four dates from July to October in 2007 for each of the three sites. There was a significant relationship between Blue River baseflow and spring discharge for Spring 2 ($p = 0.02$):

$$\text{Spring 2 discharge (cfs)} = -0.290573 + 0.0464166 \times \text{Blue River at Connerville baseflow (cfs)}$$

, and Spring 3 ($p = 0.03$):

$$\text{Spring 3 discharge (cfs)} = 1.8038409 + 0.0262495 \times \text{Blue River at Connerville baseflow (cfs)}$$

The estimated flow in the springs follows a similar pattern to the baseflow in the Blue River (Figure 6a). Estimated discharge at the Spring Creek site used gage data from the Pennington Creek at Reagan gage (#07331300). A regression between measured discharge at the site and the Pennington Creek gage was significant (< 0.01):

Spring Creek Discharge (cfs) = 0 + 0.2305838 x Pennington Creek gage discharge (cfs)

We forced the regression through the origin so when zero flow theoretically occurred at the Pennington Creek gage, there would be zero flow at the Spring Creek site (Figure 6b).

Results and Discussion

Hydrologic trends

Streamflow in the study sites was highest in the spring (April to June) and lowest in the late summer (August to October; Figure 7). Seasonal trends in discharge from the spring sites were fairly constant throughout a normal (median) year. However, there was a more pronounced seasonal trend in discharge in wet (75Q) years, while dry (25Q) years had only a minimal change throughout the year. Discharge was lower in the Spring 2 (Figure 7a) than Spring 3 (Figure 7b). During dry years, the spring flow fell as low as 1 cfs in the Spring 2 site. We will refer to the period of March to May as the high flow period of the year and August to October as the low flow period of the year because these are the general patterns observed in the Arbuckle-Simpson streams (Figure 7). Groundwater diversion in the Arbuckle-Simpson aquifer would have the greatest impact on aquatic ecosystems if large quantities of water were removed during dry years.

There was a more pronounced seasonal trend in the flows of Spring Creek (Figure 7c) and Blue River (Figure 7d). The difference between wet, normal, and dry years was greatest in the spring, when flows were higher. The difference between wet, normal, and dry years was smaller in the dry months of the late summer. As in the springs, fish habitat during the dry months would be the most impacted by water removal. The seasonal trend in discharge was low in dry years, which has the potential to disrupt fish behaviors that require specific types of cues related to flow (Bunn and Arthington 2002).

Habitat characteristics

Physical characteristics were different in the springs, with Spring 2 having a lower mean velocity (0.8 ft/s) compared to Spring 3 (1.2 ft/s), while mean depth was similar in both sites (Table 1). Spring Creek and Blue River had similar mean velocity, 1.3 and 1.2 ft/s, respectively, but Spring Creek was shallower than Blue River (0.8 ft and 1.4 ft; Table 1). The most common channel index types in Spring 2 were 3.0 (gravel, no cover), 8.0 (sand and gravel, no cover), and 9.0 (gravel and cobble, no cover; Table 3), and there were more cells with no cover (55.4%) than with cover (44.6%). The most common channel index types in Spring 3 were 1.3 (clay/silt, emergent vegetation) and 2.0 (sand, no cover; Table 3). Spring 3 was the most densely vegetated of all the study sites with nearly a quarter of the streambed containing emergent and submergent vegetation (24.9 % and 20.9%, respectively). The most common channel index types at the Spring Creek site were 6.0 (bedrock, no cover), 9.0 (gravel and cobble, no cover) and 10.0 (gravel and bedrock, no cover; Table 3). There was little cover present at the Spring Creek site (61.2% no cover; Table 3), with only 21.3% of the site having emergent vegetations and 10.8% with submergent vegetation. The major channel index types in the Blue River site were 6.0 (bedrock, no cover (51.3%)) and 10.0 (gravel and bedrock, no cover; Table 3). For all substrate types, 77.5% did not have cover and only 13.1% of cells had submergent vegetation, which was located on the edges of the river channel in areas of low velocity.

Mean temperatures were similar at three sites: Spring 2 (63.5° F), Spring 3 (63.3° F), and Spring Creek (63.0° F). Mean daily temperature for all sites ranged from a maximum of 68.0° F to a minimum of 59.0° F (Figure 3). The maximum temperatures were primarily due to high flow events that brought warmer water from the Blue River into Spring 2 and Spring 3, and warmer run-off from the watershed into Spring Creek. Winter temperatures were cooler in the sites but did not approach freezing. To compare spring temperatures with those in the Blue River, we obtained water temperature data from 10/16/2003 to 09/30/06 recorded at the USGS stream gage on the Blue River at Connerville (#07332390). During that time period, the mean daily temperature was 65.5° F, but, unlike the springs, ranged from 40.8° F to 84.0° F. The Blue River also demonstrated a distinct seasonal pattern in temperature related to air temperature, with lowest temperatures from December to January temperatures (50.0° F) to the highest temperatures from July to August (75.2° F).

Habitat suitability criteria

Habitat suitability criteria (HSC) for southern redbelly dace and redspot chub in Spring 2 (Figure 8) reflected differences in habitat use between species. Southern redbelly dace preferred shallower water than redspot chub (Figure 8a) but both species used low velocity habitats (Figure 8b) with sand/gravel or gravel/cobble and no cover (Figure 8c).

HSC for southern redbelly dace and least darter in Spring 3 (Figure 9) reflected differences in habitat use between species. Southern redbelly dace preferred deeper water than least darter (Figure 9a) but both species used low velocity areas (Figure 9b). Substrate and cover preferences were different between the species. Southern redbelly dace used gravel substrates with overhanging banks or woody debris, while least darter’s preferred habitats with clay/silt substrate and submergent vegetation or woody debris (Figure 9c).

HSC for adult southern redbelly dace, juvenile southern redbelly dace, and orangethroat darter (Figure 10) indicate each size class and species uses a different habitat. Adult southern redbelly dace used deeper water (Figure 10a) with an intermediate velocity (pools; Figure 10b), juvenile southern redbelly dace preferred the shallowest and slowest habitats (channel edge), while orangethroat darters preferred intermediate depths with the highest velocities (riffles). Substrate and cover preferences differed between species with both size classes of southern redbelly dace preferring bedrock with no cover (the most common channel index type), while orangethroat darters were most common in habitat with gravel/cobble (Figure 10c).

There currently are few studies on habitat selection of these species and this study is the first to develop HSC. The information on habitat use will not only be useful for future PHABSIM projects but are also an important contribution to the understanding of these species and their ecology.

PHABSIM modeling

We used two models to describe the relationship between weighted usable area (WUA) and simulated streamflows using PHABSIM. For Spring 2, we used a quadratic regression (i.e. parabolic shaped curve) to describe the relationship (southern redbelly dace: $R^2 = 0.98$, $P<0.01$ and redspot chub: $R^2 = 0.99$, $P<0.01$; Figure 11a):

$$\text{Southern Redbelly Dace WUA} = (432.13824 + 160.99688 * x) - 13.53764(x - 4.85)^2$$

and,

$$\text{Redspot Chub WUA} = (401.68608 + 165.85259 * x) - 12.421331 * (x - 4.85)^2$$

where, x = discharge (cfs). Maximum WUA occurred at the 11.0 cfs for the southern redbelly dace (1704 ft^2) and redspot chub (1815 ft^2). WUA dropped steadily below the maximum to a low quantity of habitat at the lowest simulated flows (0.5 cfs).

The relationship between flow and WUA at Spring 3, Spring Creek, and Blue River were described using a cubic spline model. The cubic spline model achieves a smooth regression by splicing a set of cubic polynomial regressions at “knot points” (this analysis uses the simulated discharges as knot points (Marsh and Cormier 2002). This is a “natural” cubic spline because the coefficients of the maximum endpoints are set to 0, thus providing only the maximum modeled estimates of WUA for all discharges higher than the maximum endpoint rather than extrapolating beyond the limits of this model. The cubic spline is expressed as:

$$\text{WUA} = A + B(x - O) + C(x - O)^2 + D(x - O)^3$$

where, x = discharge (cfs), O = discharge at start of interval x (knot point), A = intercept, B = linear coefficient, C = quadratic coefficient, and D = cubic coefficient (Tables 4 and 5). Coefficients are provided for Spring 3 (southern redbelly dace [$R^2 = 0.99$; Table 4a] and least darter [$R^2 = 0.91$; Table 4b]), Spring Creek (adult southern redbelly dace [$R^2 = 0.99$; Table 4c], juvenile southern redbelly dace [$R^2 = 0.97$; Table 4d], and orangethroat darter [$R^2 = 0.99$; Table 4e]), and Blue River (adult southern redbelly dace [$R^2 = 0.99$; Table 5a], juvenile southern redbelly dace [$R^2 = 0.99$; Table 5b], southern redbelly dace [$R^2 = 0.99$; Table 5c], least darter [$R^2 = 0.98$; Table 5d], and orangethroat darter [$R^2 = 0.99$; Table 5e]).

The relationship between flow and WUA using the spline model had similar regression shapes between species within sites but regression shapes differed between sites. The WUA for Spring 3 was highest for both species at 7.7 cfs (southern redbelly dace 1553 ft^2 and least darter 1238 ft^2; Figure 11b). There is a small decline in the amount of habitat as discharge is reduced because as the amount of habitat in one pool decreased, the amount of preferred habitat in the upper pool increased (due to more favorable depth). Spring Creek species had the highest WUA for the adult southern redbelly dace and orangethroat darter at 30 cfs (9597 ft^2 and 10,545 ft^2, respectively; Figure 12a). The maximum WUA for the juvenile southern redbelly dace occurred at 13.1 cfs and was higher than the other species (13,116 ft^2, Figure 12a). WUA dropped steadily below the maximum to a small amount of habitat at the lowest simulated flows. In the

Blue River, three species had their maximum WUA at 10 cfs (adult southern redbelly dace 11,310 ft^2, redspot chub 1,370 ft^2, and least darter 1,025 ft^2; Figure 12b). The other two species had maximum WUA at higher flows, with juvenile southern redbelly dace at 25 cfs (7,713 ft^2; Figure 12b) and orangethroat darter at 67 cfs (6,533 ft^2; Figure 12b).

The weighted usable area in the study sites represents only a small proportion of the total aquatic habitat that is present at any point in time (Figure 13). This trend demonstrates how selective these species are for specific habitat types. Although changes in their habitat may appear to be small, these species are already selecting and utilizing the most suitable habitat in these systems and losses of WUA could be detrimental to the species' health (Rabeni and Jacobson 1993). A positive finding is that the WUA in the Blue River was highest for most species at lower flows. In a scenario where groundwater flows are reduced and habitat is lost in the springs, the spring dependent species would move into the Blue River at the time when the most suitable physical habitat is present. There would be a trade-off for moving into the river, however, because of the higher temperatures and the greater number of predators (Harvey and Stewart 1991).

Habitat time series

We related WUA to median monthly streamflows for the period of record for the study sites using the discharge-WUA regression models (Figures 11 and 12) for all target species. This habitat time series analysis enabled us to establish baseline habitat conditions for each species during the low-flow months (August to October) and high-flow months of April, May, and June for use in alternative analysis.

The weighted usable area in Spring 2 for southern redbelly dace (Figure 14a) and redspot chub (Figure 14b) was highest in the spring and lowest in the late summer. Southern redbelly dace and redspot chub have a large drop in WUA from September values (i.e. month of lowest flow) in a wet year of 712.0 ft^2 and 699.9 ft^2 to dry year WUA of 405.6 ft^2 and 396.4 ft^2, respectively. The trends in WUA for Spring 3 are different than Spring 2 (Figure 14c-d). The seasonal trends for southern redbelly dace are similar throughout the year and between wet, baseline, and dry years (Figure 14c). The least darter has a seasonal trend similar to Spring 2 in the wet years and a drop in habitat during the late summer in dry years (Figure 14d).

The WUA for Spring Creek species varies seasonally between wet, baseline, and dry years. All three species have the maximum habitat available in May and the minimum in September in baseline years and August in dry years (Figure 15a-c). The adult southern redbelly dace (Figure 15a) and orangethroat darter (Figure 15c) typically range from a high of 6,000 ft^2 to 5,000 ft^2, respectively, in a baseline year but WUA are reduced to 4,000 ft^2 in dry years. The juvenile southern redbelly dace (Figure 15b) have higher WUA than the other two species in all years (ranging from over 10,000 to 13,000 ft^2). The juvenile southern redbelly dace had the lowest WUA in dry years when flow was lowest but median years were similar to dry years.

The Blue River time series results are different from the other three sites. The WUA for adult and juvenile southern redbelly dace and redspot chub (Figure 16a-c) have the highest available habitat for all months during the dry years and the lowest available habitat in the wet years. The spring species use shallow habitats with slow flows, which are only available in the dry years in the Blue River. The WUA for the least darter (Figure 16d) also shows a similar trend to the minnows but the orangethroat darter (Figure 16e) has a more typical relationship of highest habitat in the spring and in wet years followed by lowest habitat in late summer and dry years. The orangethroat darter occupies riffle habitats with faster water velocities than the other species, so it is better adapted to use the Blue River as a refuge. Redspot chub and least darter had the lowest WUA of all the species, which indicates that these species have little suitable habitat available. The Blue River is providing the most habitat area for the spring dependent species in the dry months and dry years, so it would likely serve as a refuge in the event of dewatering of the springs-fed creeks.

Alternatives analysis

We analyzed the effects of alternative streamflows on the target species at each study site using WUA estimates from the seasonal time series as the baseline condition. For our analysis, we modeled incremental reductions in streamflow based on median monthly streamflow (normal year) for the period of record. We did not run the analysis on wet year or dry year monthly streamflows. During wet years, weighted usable area rarely fell below the critical levels during the late summer period. The incremental analysis encompassed or approached dry year monthly streamflows. Alternatives are included for low and high flow periods of the year (e.g. seasonal high flows in spring and seasonal low flows in the late summer), and the annual average of all twelve months is also included.

For our analysis, we decreased monthly baseline streamflows by increments from 1% to 70%. These increments were selected to reflect a minimal reduction in baseflow of 1% and 5%. We then included WUA at 10% increments from 10% to 70% for Spring 2, Spring 3, and Spring Creek. This should provide sufficient information for decision making from minimal changes to a worst-case scenario of 70% reduction in baseline streamflow. These increments provide a large range of possibilities, which should allow managers to determine the loss of WUA for spring fishes at any level with future diversions from the Arbuckle-Simpson aquifer. Additional discharges may be modeled for sites and target species using the regression models for the sites and species in the PHABSIM modeling section.

Reductions in baseline streamflows resulted in a decline in WUA for southern redbelly dace and redspot chub in Spring 2 (Table 6; Figure 17a-d). In all scenarios, southern redbelly dace lost a smaller percentage of habitat than redspot chub and the declines in habitat were consistent throughout the season. This is the result of the parabolic shape of the model used to describe the relationship between WUA and discharge in Spring 2 (Figure 11a). A 1% decline in baseline discharge resulted in an average 0.7% decline in southern redbelly dace WUA and redspot chub WUA. The decline was greatest in the low flow months of August to October (Table 6). The other scenarios had minimal seasonal differences in percentage of habitat loss. A 20% baseline decline resulted in a 15.1% loss of WUA in southern redbelly dace and a 14.8% decline in the redspot chub. A 50% decline in baseline streamflow resulted in a 38.9% loss of southern redbelly dace WUA and a 37.8% loss of redspot chub WUA. In the 70% decline scenario, there is a 55.8% decline in southern redbelly dace WUA and a 53.7% decline in redspot chub WUA.

Reductions in baseline streamflow resulted in modest declines in WUA for southern redbelly dace and least darter in Spring 3 (Table 7; Figure 18a-b). Southern redbelly dace and least darter had small declines in WUA at 1% (0% southern redbelly dace WUA and 0.1% least darter WUA), 20% (0.4% southern redbelly dace WUA and 1.0% least darter WUA), and 50% (2.3% southern redbelly dace WUA and 2.5% least darter WUA). At the 60% reduction of the stream flow, least darters pass a threshold and there is a more pronounced decline in the late summer with 5.8% reduction of WUA in the low flow period; Table 7). The worst-case scenario of 70% reduction had a mean decline of 4.4% in southern redbelly dace WUA but higher declines in the low flow months (5.4%; Table 7). Least darter has a greater decline at 70% with a mean reduction of 11.9% and high reductions in the low flow months (14.3%; Table 7).

Reductions in baseline streamflow in Spring Creek resulted in small reduction in WUA at 1% to 30% and larger reductions in WUA at 40% to 70% (Table 8; Figure 19a-f). At a 1% reduction of baseline flow, there were 0.5% reduction in adult southern redbelly dace and orangethroat darter, while there was a 0.2% reduction in juvenile southern redbelly dace habitat. At 20% below baseline flow, there was reduced WUA for all species in the dry months of late summer. At the 20% increment, adult southern redbelly dace had 11.2% less WUA (Table 8), juvenile southern redbelly dace had 5.4% less WUA (Table 8), and orangethroat darter had 9.9% less WUA (Table 8c). At 50% and 70% below baseline flow, adult southern redbelly dace have 29.3% and 35.8% less WUA, respectively. Although the percent reduction of habitat was higher in the high flow months, the total amount of WUA was lowest in the low flow months (Table 8). Juvenile southern redbelly dace had the smallest decrease in habitat at the 50% reduction in flow (15.5%) but had larger reductions at 70% less flow (27.4%), especially in the low flow months (Table 8). The orangethroat darter had a 24.8% reduction in WUA at 50% below the baseline streamflow but only 23.2% less WUA at the 70% reduction in streamflow (Table 8).

In the Blue River, minnow species WUA increased when streamflow decreased (Table 9). Only the juvenile southern redbelly dace had a small decrease in WUA in low flow months at 20% reduction (0.2% decline), 50% reduction (1.6% reduction), and 70% reduction (2.7% reduction; Table 9). The least darter WUA was little changed at 1% reduction in baseline stream flow but above 1%, WUA tended to be constant through the year at 1,000 ft^2 (Table 9). Although there is little change in the amount of WUA for redspot chub and least darter with reduced flows, there is very little habitat available compared to other species, especially when considering the size of total available habitat. The orangethroat darter, which prefers fast flowing water, was the only species with consistently reduced WUA with reduced streamflow (Table 9). This species had a minimal decline in WUA at 1% reduction (0.1%) and at 20% reduction (3.1%; Table 9). At 50% and 70% reduction in baseline flow there was the greatest decline in WUA of 10.3% and 16.7%, respectively.

Study Limitations

Several limitations of this study should be outlined for proper interpretation and application of the results. Because of the timeline associated with the project funding, only a single field season was available for data collection. Thus, the fish habitat use included in this study was for summer habitat use (i.e. feeding and shelter in 2007). The species that use the springs have not been used in previous PHABSIM studies, so we were limited to the data that we collected. Additional life stages would be useful for the Arbuckle-Simpson species (i.e. spawning habitat), but are not available at this time.

The use of the PHABSIM model assumes that there is a relationship between WUA and the target species population. It is preferable to have long term data that can directly show a causative relationship between WUA and fish populations (Nehring and Anderson 1993). For this study, we are working with fish species with no historic population data and also have little published research on their ecology. Although we cannot demonstrate a direct relationship between WUA and population, three of the species (southern redbelly dace, redspot chub, and least darter) are spring habitat specialists, which indicates that changes in spring habitat quantity and quality are likely to influence their populations.

Several other factors may have compromised some of the data in the study. High flow in the flow in the Blue River on June 30, 2007 resulted in the scouring of vegetation and altered habitat in Spring 3. These changes occurred before the surveying of the channel, velocity measurements, and habitat use data collection, but it may have had residual effects on the available habitat and habitat selection by least darters. We observed least darters in the upper pool using cobbles for habitat in an area that previously contained dense vegetation. Thus it is unclear if the cobble habitat was preferred or if the least darters were using the available habitat after the high flow event. Another factor that affected the PHABSIM modeling in Spring 3 was the alteration of water surface elevation from beaver dam construction. Dams increased the water surface elevation in fall 2007 when discharge had been reduced, resulting in an unexpected relationship between water surface elevation and discharge. Although we adjusted the data to reflect the expected flow and water surface elevation relationship (decrease in water surface elevation associated with a decrease in discharge), the results from the WUA and discharge are questionable and should not be used for management purposes (Figure 11b). The results from Spring 2 could be used to represent small spring-runs and Spring Creek can be used to represent larger creeks in the Arbuckle-Simpson.

Conclusions

The objective of this study was to quantify the effect of reduced streamflows on fish habitat in spring-fed streams of the Arbuckle-Simpson. These spring habitats are considered to be groundwater dependent ecosystems because they require the surface expression of groundwater or they would no longer exist in their current form. The species assemblages of the springs are unique in southern Oklahoma because spring habitats provide a consistent source of clean and clear water with minimal temperature fluctuation. Three of the species (southern redbelly dace, redspot chub, and least darter) occur at the southern end of their distribution in the spring habitats, which provide a temperature refuge and excellent water quality.

This study found that reductions in streamflow in the Spring 2, Spring 3, and Spring Creek would reduce habitat by as much as 58% for southern redbelly dace and redspot chub, 14% for the least darter, and 35% for the orangethroat darter. Reduced habitat area would lead to a greater influence of air temperature on water temperature (as volume of groundwater decreases), loss of preferred spawning and nursery habitat, increased competition for resources, and increased predation. In an Arbuckle-Simpson context, results from Spring 2 and Spring 3 could be used as a model for other small spring-runs in the Arbuckle-Simpson, while Spring Creek could be used as a model for larger creeks with high groundwater inputs. In small spring-runs, a reduction of baseline flow from 10 to 20% will result in modest reductions in WUA (8-15%). In larger streams, slightly larger reductions in flow of 20 to 30% will result in a similar degree of WUA reduction (11-17%). The regression models for discharge and WUA included in this report can be used to predict WUA at different target discharges. The models can also be used if future seasonal flow data is simulated or collected from the study sites (e.g. Spring 2, Spring 3, and Spring Creek).

We found that the greatest amount of usable habitat in the Blue River would occur at lower streamflows. The Blue River could potentially serve as refuge habitat for spring fish species if spring flows were reduced to the level that fishes move from springs to the river. However, because we did not have temperature data from the Blue River, we do not know if the temperature regime of the river would be suitable for the spring-dependent species. Furthermore, movement of spring fish species into the Blue River would likely expose them to a larger number of predators and, therefore, a greater risk to predation. In contrast to the springs flowing into the Blue River, species in groundwater dependent sites like Spring Creek could experience lower survival under reduced flows because there is no adjacent refuge habitat in Spring Creek causing species to move longer distances downstream to Pennington Creek where the habitat might be less suitable and predation risk greater.

Monitoring the spring flows throughout the year may be useful in maintaining flow in groundwater dependent ecosystems. The relationship between groundwater depth and spring flow throughout the Arbuckle-Simpson could be used in the maintenance of flows in springs. If acceptable thresholds in streamflow can be maintained and seasonal patterns in flow preserved, impact to the survival and reproduction spring-dependent fish species may be minimal. Water allocation in the Arbuckle-Simpson aquifer could provide an opportunity to use adaptive management during groundwater removal to ensure minimal impact on spring habitat and fishes. A balance between the human and ecosystem needs for water would ultimately benefit humans because the needs of the environment provide goods and services indirectly to humans.

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Table 1: Site and physical characteristics describing the four study sites in the Arbuckle-Simpson aquifer.

	Spring 2	Spring 3	Spring Creek	Blue River
	Site Characteristics			
Site Length (ft)	714.1	862.0	1335.2	1512.1
Transects (N)	25	25	29	20
Observed Discharge (N)	3	3	3	3
Simulated Discharge (N)	6	7	8	8
	Physical Characteristics (Transect)			
Wetted Width (ft)				
Mean	13.7	13.5	32.6	53.4
Min	6.0	4.7	11.0	32.5
Max	33.5	29.4	53.3	75.9
Mean Transect Velocity (ft/s)				
Mean	0.8	1.2	1.3	1.2
Min	0.1	0.1	0.4	0.3
Max	2.1	4.5	5.5	3.5
Mean Transect Depth (ft)				
Mean	0.6	0.5	0.8	1.4
Min	0.2	0.1	0.3	0.6
Max	1.7	1.2	1.4	2.7

Table 2: Categories of substrate and cover and their codes.

Category	Code
Substrate	
Clay/Silt (0.0005-0.02625 mm)	1
Sand (0.0625-2 mm)	2
Gravel (2-64 mm)	3
Cobble (64-256 mm)	4
Boulder (>256 mm)	5
Bedrock (flat and fractured)	6
Clay/Silt + Sand	7
Sand + Gravel	8
Gravel + Cobble	9
Gravel + Bedrock	10
Cover	
None	0.0
Gravel and Cobble	0.1
Bedrock	0.2
Emergent Vegetation	0.3
Floating Vegetation	0.4
Submergent Vegetation	0.5
Woody Debris	0.6
Overhanging Bank	0.7

Table 3: Percentage of cells containing each channel index type in the four study sites. Bold numbers indicate greater than 10%. N indicates the total number of cells surveyed for channel index.

Channel Index	Percent of Channel Index in Category			
(Sub.Cov)	Spring 2	Spring 3	Spring Creek	Blue River
1.0	7.8	1.8		
1.3	5.0	**11.1**		
1.5	5.6	5.2		
1.6	1.7	1.2		
1.7	1.4	0.3		
2.0	4.2	9.8		0.4
2.3	6.1	6.5	3.6	1.1
2.5	5.3	5.2	0.3	1.1
2.6	1.7	3.4		
2.7		1.2		
3.0	9.7	7.1	4.2	2.5
3.3	0.3	0.9	4.2	0.4
3.5	0.6	2.5	1.4	0.4
3.6	0.3	2.5	0.3	0.7
3.7	0.6	0.9	0.0	0.0
4.0	1.4	0.0	3.0	1.1
4.6	0.3	0.0		
6.0	1.9	3.4	**28.5**	**51.3**
6.3	0.3	0.6	2.2	1.5
6.5		0.6	4.4	8.7
6.6		0.3		1.1
6.7		0.3		1.5
7.0	3.6	1.5	0.3	0.4
7.3	3.1	3.4	3.6	1.1
7.5	3.1	1.2	0.6	1.1
7.6	0.8	0.0		
7.7	0.3	0.0		
8.0	**16.7**	8.0	0.6	2.9
8.3	1.7	2.5	5.0	1.5
8.5	2.5	5.5	0.6	1.5
8.6	1.1	3.1		
8.7	0.6	0.6		0.4
9.0	**10.0**	4.0	**11.4**	3.3
9.3			2.8	0.0
9.5	0.6	0.6	3.6	0.4
9.6	1.1	1.5		
9.7	0.8	0.3		
10.0		2.8	**13.3**	**15.6**
10.3			2.2	0.0
10.5			4.2	0.4
N =	359	325	361	275

Table 4: Regression equation coefficients of WUA-discharge models for Spring 3: A) southern redbelly dace and B) least darter; and Spring Creek: C) adult southern redbelly dace, D) juvenile southern redbelly dace, and E) orangethroat darter.

A)

Discharge (cfs)		Coefficients		
Interval (O)	Intercept (A)	B	C	D
0.0	41.75	2266.37	0.00	-1391.52
0.5	1000.99	1222.73	-2087.28	1308.85
1.0	1254.14	117.08	-124.01	104.22
1.5	1294.71	71.24	32.32	-52.68
2.0	1331.82	64.05	-46.69	19.92
2.5	1354.66	32.30	-16.82	-2.23
3.5	1367.92	-8.02	-23.50	33.87
4.4	1366.36	31.99	67.95	-44.74
5.5	1424.22	19.08	-79.68	14.56
7.7	1235.56	-120.13	16.40	-4.21
9.0	1097.87	0.00	0.00	0.00

B)

Discharge (cfs)		Coefficients		
Interval (O)	Intercept (A)	B	C	D
0.0	190.85	1197.70	0.00	-424.12
0.5	736.69	879.62	-636.17	141.02
1.0	1035.08	349.21	-424.65	198.61
1.5	1128.35	73.52	-126.73	86.73
2.0	1144.27	11.84	3.37	3.91
2.5	1151.52	18.14	9.23	-23.90
3.5	1154.99	-35.10	-62.47	42.80
4.4	1104.00	-43.54	53.08	-10.54
5.5	1106.31	34.99	18.31	-13.44
7.7	1128.83	-79.55	-70.38	18.05
9.0	946.12	0.00	0.00	0.00

C)

Discharge (cfs)		Coefficients		
Interval (O)	Intercept (A)	B	C	D
0.0	29.34	3268.03	0.00	-977.89
0.5	1541.11	2534.61	-1466.84	284.98
2.5	3022.83	87.04	243.05	-42.69
5.0	4092.42	501.80	-77.15	9.89
7.5	5019.29	301.51	-2.96	0.28
10.0	5758.97	292.01	-0.84	1.17
13.1	6691.07	320.62	10.07	-1.32
21.0	9201.96	232.71	-21.20	0.02
30.0	9596.98	-142.98	-20.55	0.56
50.0	3017.02	-290.10	13.19	-0.19
77.5	1161.99	15.09	-2.09	0.03
100.0	795.00	0.00	0.00	0.00

D)

Discharge (cfs)		Coefficients		
Interval (O)	Intercept (A)	B	C	D
0.0	1654.90	5596.22	0.00	-1103.26
0.5	4315.10	4768.78	-1654.90	238.67
2.5	9142.43	1013.24	-222.87	32.38
5.0	10788.55	506.06	20.00	-9.99
7.5	12022.69	418.84	-54.89	2.89
10.0	12771.80	198.48	-33.25	1.69
13.1	13117.75	40.91	-17.58	0.52
21.0	12599.33	-139.82	-5.30	-0.37
30.0	10642.65	-324.88	-15.26	0.53
50.0	2304.18	-295.78	16.72	-0.25
77.5	1569.53	51.69	-4.08	0.06
100.0	1355.09	0.00	0.00	0.00

E)

Discharge (cfs)		Coefficients		
Interval (O)	Intercept (A)	B	C	D
0.0	56.37	5566.54	0.00	-1878.85
0.5	2604.78	4157.40	-2818.27	561.90
2.5	4141.72	-372.84	553.16	-95.46
5.0	5175.31	603.10	-162.78	27.42
7.5	6094.11	303.32	42.87	-9.65
10.0	6969.57	336.72	-29.51	4.68
13.1	7869.19	288.64	14.00	-1.62
21.0	10224.93	206.69	-24.38	0.60
30.0	10545.01	-87.31	-8.29	0.20
50.0	7059.01	-182.45	3.53	-0.03
77.5	3992.00	-66.87	0.67	-0.01
100.0	2713.00	0.00	0.00	0.00

Table 5: Regression equation coefficients of WUA-discharge models for Blue River: A) adult southern redbelly dace and B) juvenile southern redbelly dace, C) redspot chub, D) least darter, and E) orangethroat darter.

A)

Discharge (cfs)		Coefficients		
Interval (O)	Intercept (A)	B	C	D
5	11272.05	-24.70	0.00	-0.06
10	11141.49	-28.93	-0.85	0.00
15	10975.69	-37.40	-0.85	0.06
20	10774.96	-41.41	0.05	0.03
25	10573.40	-38.35	0.56	-0.02
37	10167.66	-31.51	0.01	-0.03
52	9601.48	-50.30	-1.26	0.05
67	8721.58	-56.50	0.85	-0.03
97	6848.46	-99.68	-2.28	0.05
127	3270.94	-89.86	2.61	-0.02
165	2370.27	0.00	0.00	0.00

B)

Discharge (cfs)		Coefficients		
Interval (O)	Intercept (A)	B	C	D
5	7425.09	13.66	0.00	0.00
10	7493.81	13.91	0.05	0.00
15	7564.05	14.08	-0.01	-0.02
20	7631.33	12.28	-0.34	-0.03
25	7680.74	6.80	-0.75	-0.02
37	7626.00	-18.28	-1.34	-0.02
52	6985.27	-71.50	-2.21	0.09
67	5703.32	-80.24	1.63	-0.03
97	3890.53	-69.59	-1.27	0.03
127	1548.48	-56.89	1.69	-0.01
165	1018.40	0.00	0.00	0.00

C)

Discharge (cfs)		Coefficients		
Interval (O)	Intercept (A)	B	C	D
5	1358.17	-0.46	0.00	-0.01
10	1354.84	-1.07	-0.12	0.00
15	1346.10	-2.50	-0.17	0.00
20	1329.72	-3.99	-0.13	0.00
25	1306.95	-5.03	-0.08	0.00
37	1240.48	-5.66	0.02	0.00
52	1159.41	-5.25	0.00	0.00
67	1083.12	-4.82	0.02	0.00
97	881.11	-11.28	-0.24	0.01
127	490.10	-9.35	0.30	0.00
165	427.01	0.00	0.00	0.00

D)

Discharge (cfs)		Coefficients		
Interval (O)	Intercept (A)	B	C	D
5	1009.93	0.50	0.00	0.00
10	1012.23	0.38	-0.02	0.00
15	1013.82	0.33	0.01	0.00
20	1015.99	0.57	0.03	0.00
25	1019.42	0.74	0.00	-0.01
37	1014.84	-2.63	-0.28	0.00
52	899.50	-13.59	-0.45	0.02
67	657.80	-14.42	0.39	0.00
97	449.40	-3.84	-0.04	0.00
127	333.17	-2.69	0.08	0.00
165	307.89	0.00	0.00	0.00

E)

Discharge (cfs)		Coefficients		
Interval (O)	Intercept (A)	B	C	D
5	4889.56	60.15	0.00	-0.03
10	5186.51	57.86	-0.46	-0.01
15	5463.55	52.78	-0.56	0.00
20	5713.33	47.09	-0.58	0.00
25	5934.07	41.17	-0.60	-0.01
37	6330.64	24.01	-0.83	0.01
52	6542.78	6.85	-0.31	0.00
67	6575.42	-2.46	-0.31	-0.01
97	5849.50	-58.50	-1.56	0.03
127	3576.40	-63.45	1.40	-0.01
165	2510.24	0.00	0.00	0.00

Table 6: Quantity and percent change (%) in weighted usable area (WUA; ft^2/1000ft) for southern redbelly dace and redspot chub in relation to incremental reductions in baseline streamflow (Q) of Spring 2.

		Discharge	Southern redbelly dace		Redspot chub	
Scenario	Flow	(cfs)	WUA (ft^2)	%	WUA (ft^2)	%
Baseline	Low	1.47	514.6	0.0	763.0	0.0
	Annual Average	1.99	641.8	0.0	514.6	0.0
	High	2.50	763.0	0.0	641.8	0.0
Baseline-1%	Low	1.45	510.8	-0.7	757.3	-0.7
	Annual Average	1.97	637.1	-0.7	510.8	-0.7
	High	2.48	757.3	-0.7	637.1	-0.7
Baseline-5%	Low	1.39	495.9	-3.6	734.6	-3.8
	Annual Average	1.89	618.1	-3.7	495.9	-3.7
	High	2.38	734.6	-3.7	618.1	-3.7
Baseline-10%	Low	1.32	477.2	-7.3	705.8	-7.6
	Annual Average	1.79	594.1	-7.4	477.2	-7.3
	High	2.25	705.8	-7.5	594.1	-7.5
Baseline-20%	Low	1.17	439.2	-14.7	646.9	-15.4
	Annual Average	1.59	545.2	-15.1	439.2	-14.8
	High	2.00	646.9	-15.2	545.2	-15.2
Baseline-30%	Low	1.03	400.6	-22.2	586.3	-23.4
	Annual Average	1.39	495.2	-22.8	400.6	-22.3
	High	1.75	586.3	-23.2	495.2	-23.1
Baseline-40%	Low	0.88	361.4	-29.8	524.0	-31.6
	Annual Average	1.20	444.1	-30.8	361.4	-30.0
	High	1.50	524.0	-31.3	444.1	-31.1
Baseline-50%	Low	0.73	321.6	-37.5	459.9	-40.1
	Annual Average	1.00	391.9	-38.9	321.6	-37.8
	High	1.25	459.9	-39.7	391.9	-39.3
Baseline-60%	Low	0.59	281.2	-45.4	394.1	-48.7
	Annual Average	0.80	338.5	-47.3	281.2	-45.7
	High	1.00	394.1	-48.3	338.5	-47.6
Baseline-70%	Low	0.44	240.2	-53.3	326.6	-57.6
	Annual Average	0.60	284.0	-55.8	240.2	-53.7
	High	0.75	326.6	-57.2	284.0	-56.1

Table 7: Quantity and percent change (%) in weighted usable area (WUA; ft^2/1000ft) for southern redbelly dace and least darter in relation to incremental reductions in baseline streamflow (Q) of Spring 3.

Scenario	Flow	Discharge (cfs)	Southern redbelly dace WUA (ft^2)	Southern redbelly dace %	Least darter WUA (ft^2)	Least darter %
Baseline	Low	2.80	1303.2	0.0	1014.7	0.0
	Annual Average	3.10	1302.8	0.0	1024.4	0.0
	High	3.39	1302.5	0.0	1034.3	0.0
Baseline-1%	Low	2.77	1303.2	0.0	1014.0	-0.1
	Annual Average	3.07	1302.9	0.0	1023.3	-0.1
	High	3.36	1302.5	0.0	1033.1	-0.1
Baseline-5%	Low	2.66	1303.0	0.0	1011.1	-0.4
	Annual Average	2.94	1303.0	0.0	1019.4	-0.5
	High	3.22	1302.6	0.0	1028.4	-0.6
Baseline-10%	Low	2.52	1302.2	-0.1	1008.2	-0.6
	Annual Average	2.79	1302.9	0.0	1014.9	-0.9
	High	3.05	1302.9	0.0	1022.5	-1.1
Baseline-20%	Low	2.24	1298.4	-0.4	1004.8	-1.0
	Annual Average	2.48	1301.3	-0.1	1008.0	-1.6
	High	2.71	1303.1	0.0	1012.4	-2.1
Baseline-30%	Low	1.96	1290.0	-1.0	1004.3	-1.0
	Annual Average	2.17	1296.1	-0.5	1004.8	-1.9
	High	2.37	1300.7	-0.1	1006.0	-2.7
Baseline-40%	Low	1.68	1277.4	-2.0	1003.1	-1.1
	Annual Average	1.86	1285.4	-1.3	1004.0	-2.0
	High	2.03	1292.8	-0.7	1004.2	-2.9
Baseline-50%	Low	1.40	1267.2	-2.8	992.2	-2.2
	Annual Average	1.55	1272.3	-2.3	998.9	-2.5
	High	1.70	1278.1	-1.9	1003.4	-3.0
Baseline-60%	Low	1.12	1258.7	-3.4	955.6	-5.8
	Annual Average	1.24	1262.6	-3.1	974.1	-4.9
	High	1.36	1266.0	-2.8	988.7	-4.4
Baseline-70%	Low	0.84	1233.3	-5.4	870.1	-14.3
	Annual Average	0.93	1245.1	-4.4	902.5	-11.9
	High	1.02	1253.5	-3.8	931.1	-10.0

Table 8: Quantity and percent change (%) in weighted usable area (WUA; ft^2/1000ft) for adult southern redbelly dace, juvenile southern redbelly dace, and orangethroat darter in relation to incremental reductions in baseline streamflow (Q) of Spring Creek.

		Discharge	Adult southern redbelly dace		Juvenile southern redbelly dace		Orangethroat darter	
Scenario	Flow	(cfs)	WUA (ft^2)	%	WUA (ft^2)	%	WUA (ft^2)	%
Baseline	Low	4.84	4009.5	0.0	10709.3	0.0	5072.4	0.0
	Annual Average	6.40	4606.7	0.0	11462.4	0.0	5694.4	0.0
	High	8.07	5258.7	0.0	12317.7	0.0	6362.4	0.0
Baseline-1%	Low	4.79	3984.3	-0.6	10684.8	-0.2	5041.3	-0.6
	Annual Average	6.33	4582.8	-0.5	11434.0	-0.2	5667.9	-0.5
	High	7.99	5234.0	-0.5	12289.4	-0.2	6333.5	-0.5
Baseline-5%	Low	4.60	3881.3	-3.2	10586.7	-1.1	4912.7	-3.1
	Annual Average	6.08	4484.9	-2.6	11317.8	-1.3	5559.9	-2.4
	High	7.67	5134.9	-2.4	12169.7	-1.2	6221.1	-2.2
Baseline-10%	Low	4.36	3749.9	-6.5	10462.1	-2.3	4746.9	-6.4
	Annual Average	5.76	4358.3	-5.4	11166.6	-2.6	5420.7	-4.8
	High	7.26	5009.9	-4.7	12004.5	-2.5	6089.3	-4.3
Baseline-20%	Low	3.87	3492.0	-12.9	10195.7	-4.8	4429.6	-12.7
	Annual Average	5.12	4090.6	-11.2	10844.5	-5.4	5128.3	-9.9
	High	6.46	4749.2	-9.7	11626.0	-5.6	5845.2	-8.1
Baseline-30%	Low	3.39	3264.8	-18.6	9887.8	-7.7	4185.8	-17.5
	Annual Average	4.48	3806.2	-17.4	10496.8	-8.4	4818.3	-15.4
	High	5.65	4450.1	-15.4	11206.4	-9.0	5565.2	-12.5
Baseline-40%	Low	2.91	3097.6	-22.7	9516.3	-11.1	4080.7	-19.6
	Annual Average	3.84	3516.2	-23.7	10117.9	-11.7	4509.1	-20.8
	High	4.84	4079.2	-22.4	10779.4	-12.5	5156.2	-19.0
Baseline-50%	Low	2.42	3018.9	-24.7	9058.5	-15.4	4178.0	-17.6
	Annual Average	3.20	3257.8	-29.3	9681.4	-15.5	4279.7	-24.8
	High	4.04	3637.8	-30.8	10351.0	-16.0	4606.9	-27.6
Baseline-60%	Low	1.94	2998.7	-25.2	8455.0	-21.0	4424.5	-12.8
	Annual Average	2.56	3078.6	-33.2	9123.7	-20.4	4240.4	-25.5
	High	3.23	3237.4	-38.4	9838.3	-20.1	4162.7	-34.6
Baseline-70%	Low	1.45	2868.8	-28.4	7559.5	-29.4	4489.6	-11.5
	Annual Average	1.92	2959.7	-35.8	8326.7	-27.4	4370.9	-23.2
	High	2.42	3024.8	-42.5	9129.5	-25.9	4151.8	-34.7

Table 9: Quantity and percent change (%) in weighted usable area (WUA; $ft^2/1000ft$) for adult southern redbelly dace, juvenile southern redbelly dace, redspot chub, least darter, and orangethroat darter in relation to incremental reductions in baseline streamflow (Q) of Blue River.

		Discharge	Adult southern redbelly dace		Juvenile southern redbelly dace		Redspot chub		Least darter		Orangethroat darter	
		(cfs)	WUA (ft^2)	%	WUA (ft^2)	%	WUA (ft^2)	%	WUA (ft^2)	%	WUA (ft^2)	%
Baseline	Low	30.30	10382.6	0.0	7692.4	0.0	1278.3	0.0	1022.1	0.0	6135.3	0.0
	Annual Average	39.40	10076.6	0.0	7491.3	0.0	1227.6	0.0	990.0	0.0	6346.3	0.0
	High	48.10	9766.7	0.0	7201.9	0.0	1180.0	0.0	939.8	0.0	6503.6	0.0
Baseline-1%	Low	30.00	10392.9	0.1	7693.1	0.0	1280.0	0.1	1022.1	0.0	6124.8	-0.2
	Annual Average	39.01	10091.0	0.1	7503.8	0.2	1229.8	0.2	992.2	0.2	6338.5	-0.1
	High	47.62	9787.4	0.2	7229.0	0.4	1182.6	0.2	944.8	0.5	6498.7	-0.1
Baseline-20%	Low	24.30	10601.9	2.1	7675.0	-0.2	1310.5	2.5	1018.9	-0.3	5903.4	-3.8
	Annual Average	31.60	10350.4	2.7	7650.7	2.1	1270.9	3.5	1016.5	2.7	6150.9	-3.1
	High	38.58	10119.9	3.6	7582.4	5.3	1232.3	4.4	1007.5	7.2	6357.1	-2.3
Baseline-50%	Low	18.15	10969.2	5.6	7566.4	-1.6	1345.6	5.3	1013.9	-0.8	5472.4	-10.8
	Annual Average	23.60	10786.4	7.0	7623.0	1.8	1329.1	8.3	1016.2	2.6	5692.1	-10.3
	High	28.81	10611.6	8.7	7671.3	6.5	1311.3	11.1	1018.8	8.4	5892.5	-9.4
Baseline-70%	Low	17.60	11166.8	7.6	7481.4	-2.7	1355.7	6.1	1011.9	-1.0	5134.2	-16.3
	Annual Average	22.88	11081.6	10.0	7519.5	0.4	1351.8	10.1	1012.8	2.3	5288.4	-16.7
	High	27.93	10995.4	12.6	7556.1	4.9	1347.2	14.2	1013.7	7.9	5432.8	-16.5

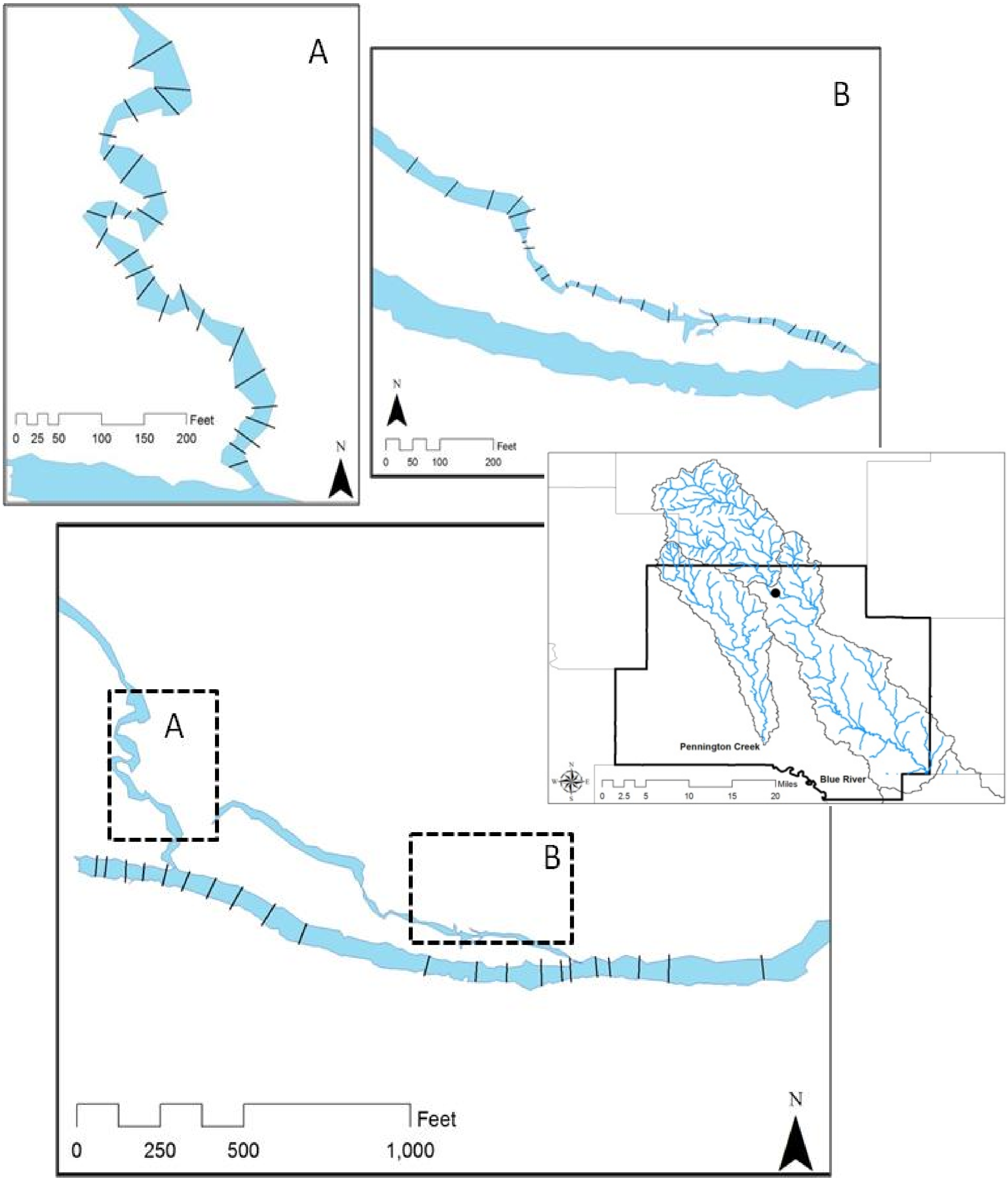

Figure 1: Map of Blue River channel and transect locations with Spring 2 channel and transects (inset A) and Spring 3 channel and transects (inset B).

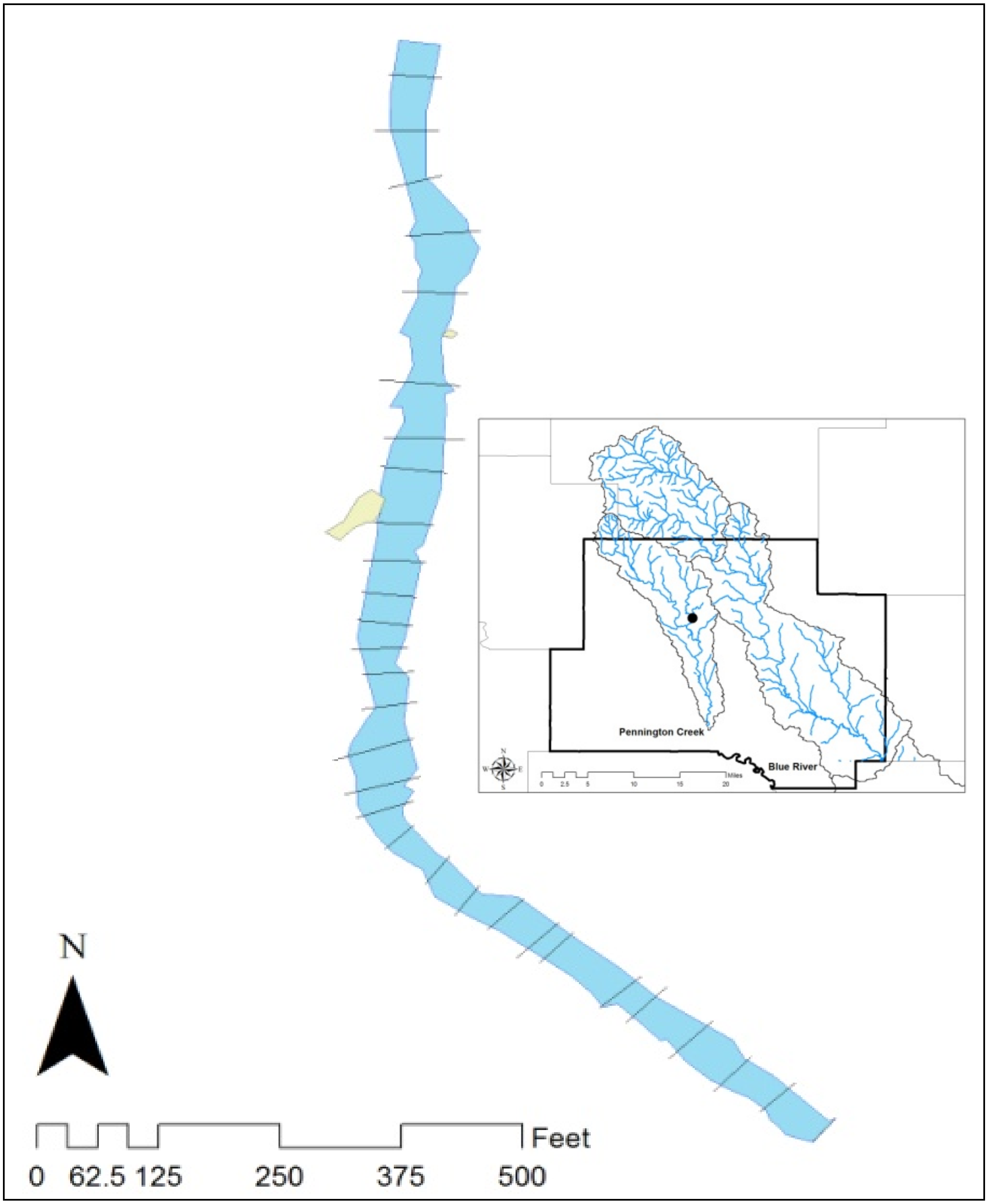

Figure 2: Map of Spring Creek channel and transects.

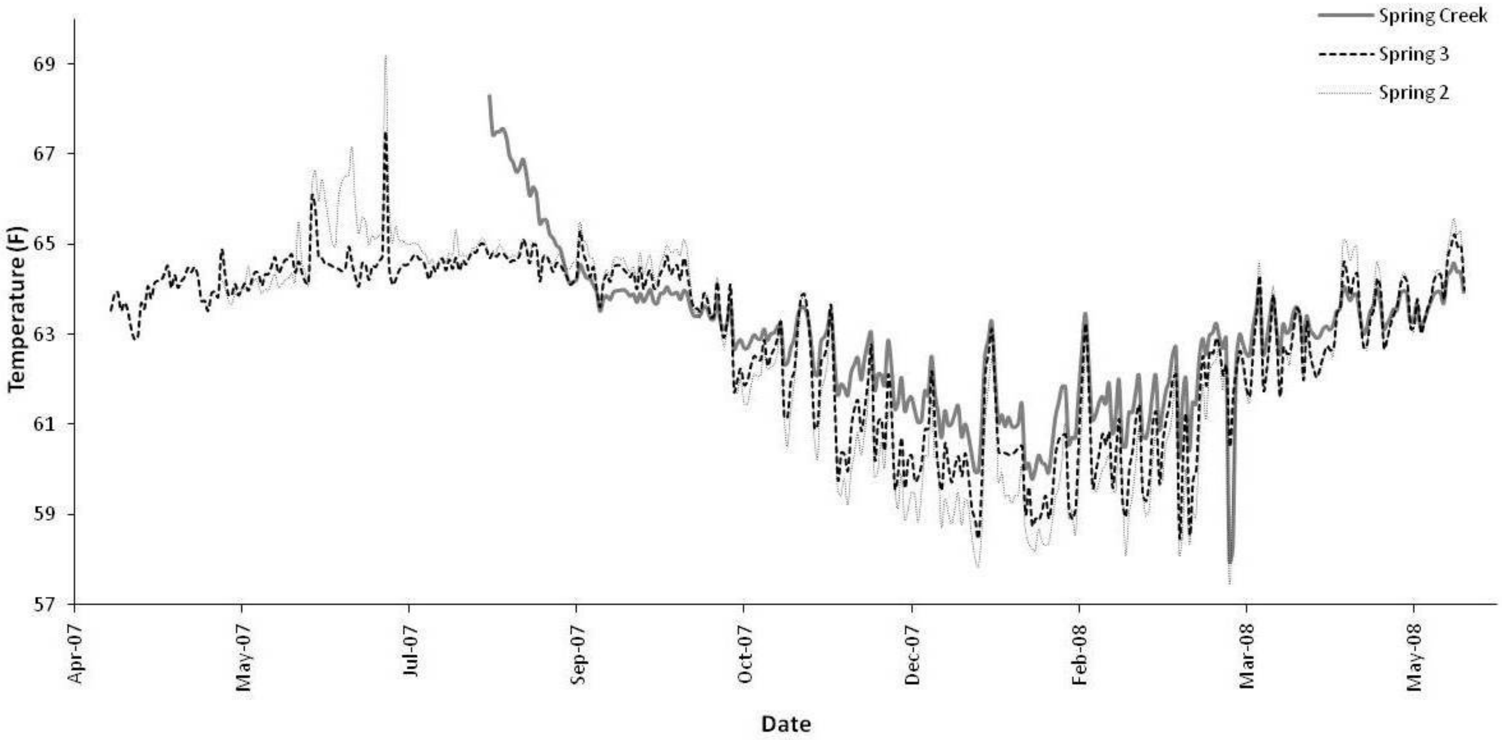

Figure 3: Water temperature in Spring 2 (dot line; May 22, 2007 to May 27, 2008), Spring 3 (dash line; April 19, 2007 to May 27, 2008), and Spring Creek (solid line; August 10, 2007 to May 27, 2008).

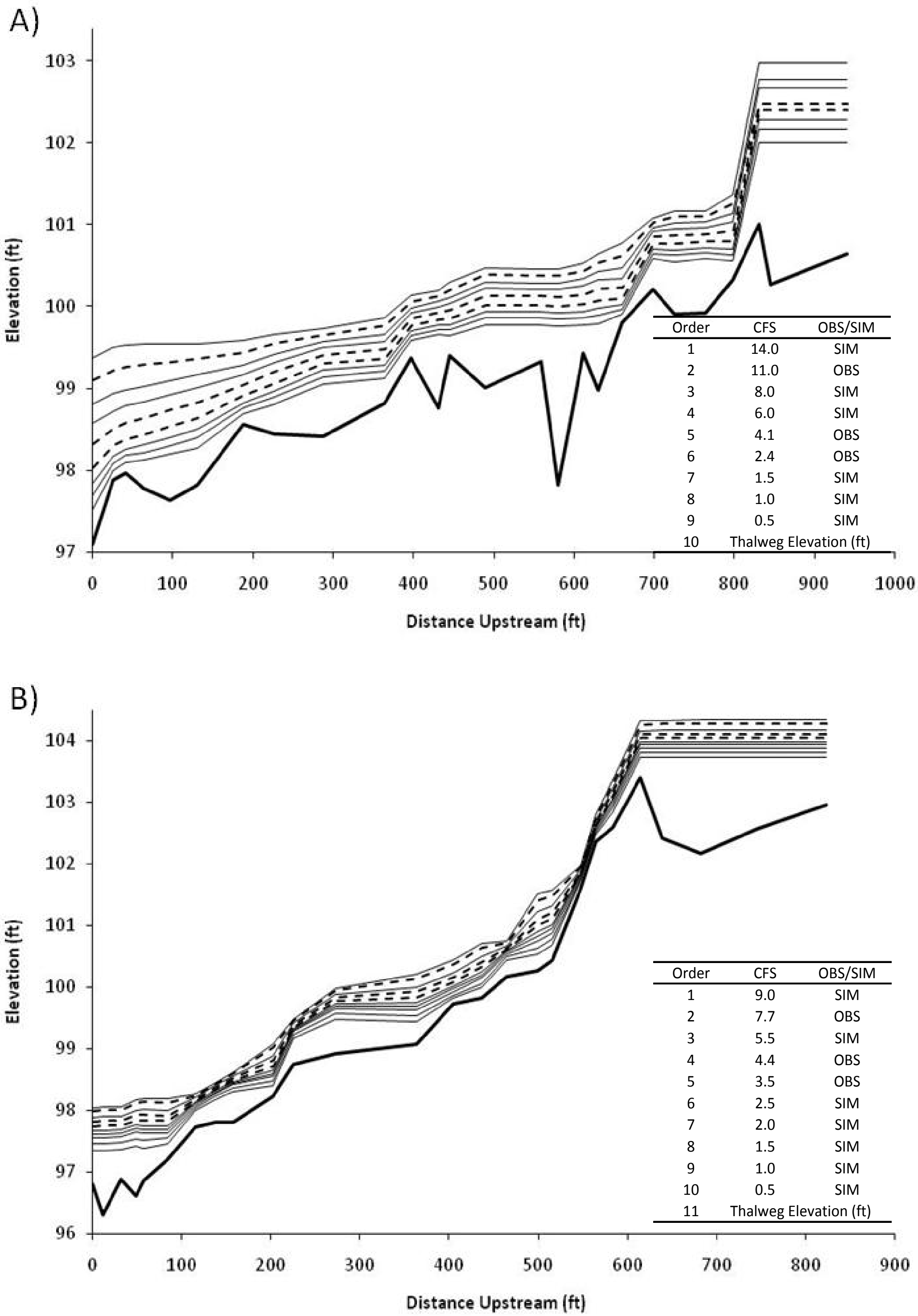

Figure 4: Water surface elevation (feet) for simulated (solid line) and observed (dash line) discharges with thalweg elevation (feet) at A) Spring 2 and B) Spring 3.

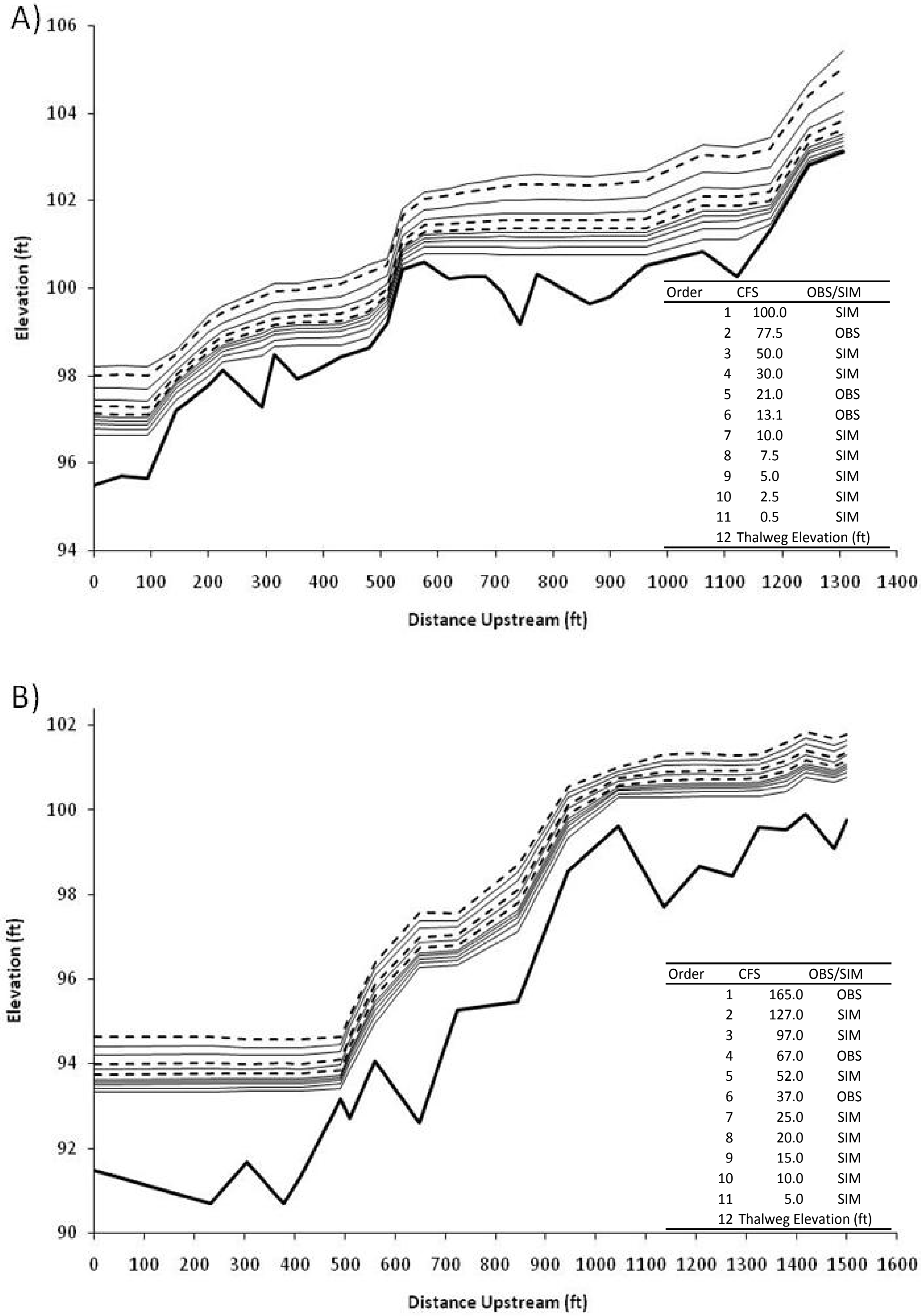

Figure 5: Water surface elevation (feet) for simulated (solid line) and observed (dash line) discharges with thalweg elevation (feet) at A) Spring Crrek and B) Blue River.

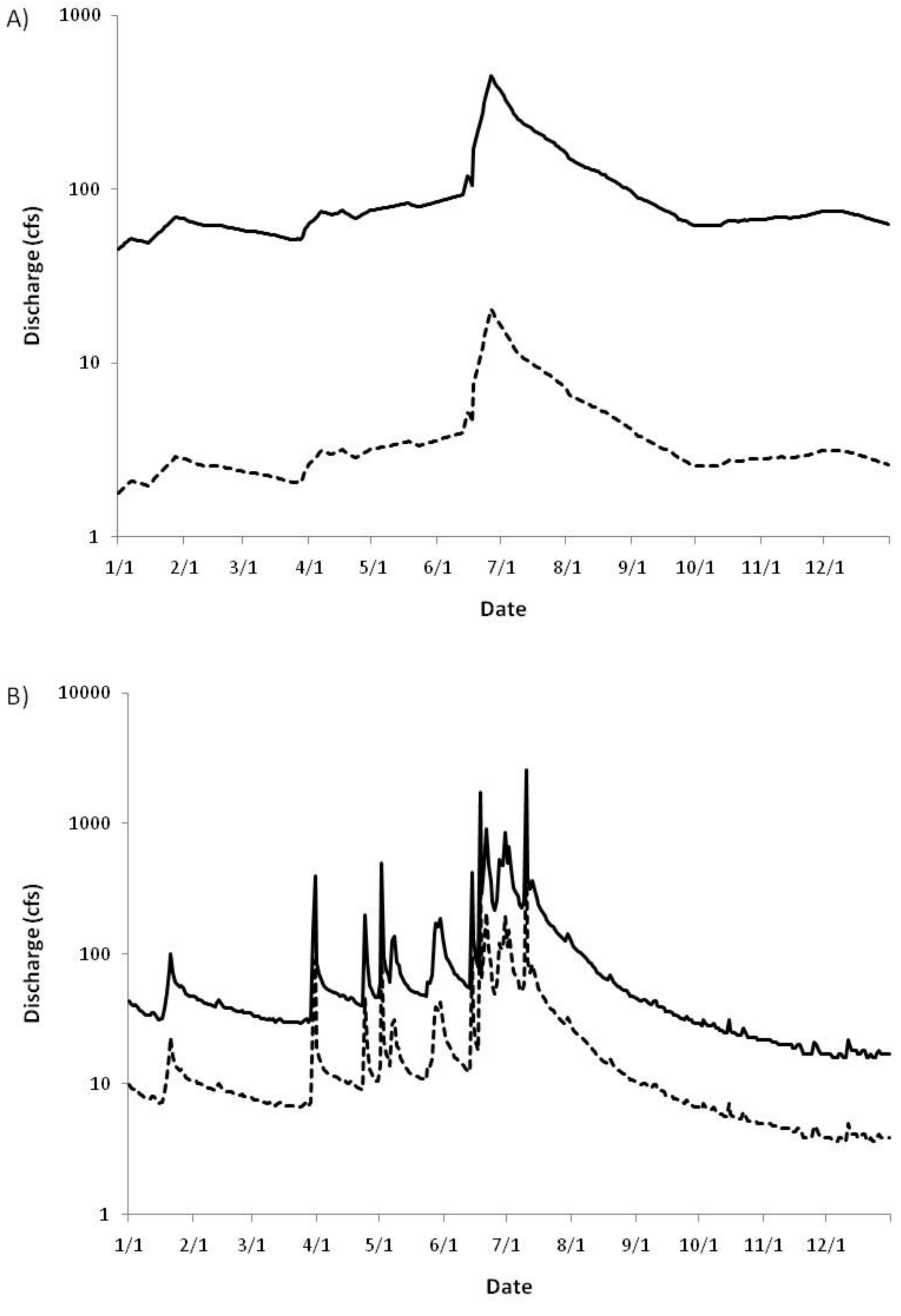

Figure 6: USGS gage daily mean discharge (solid line) at A) baseflow for Blue River at Connerville and B) Pennington Creek at Reagan and estimated daily mean discharge at the study site (dash line) at A) Spring 2 and B) Spring Creek during 2007.

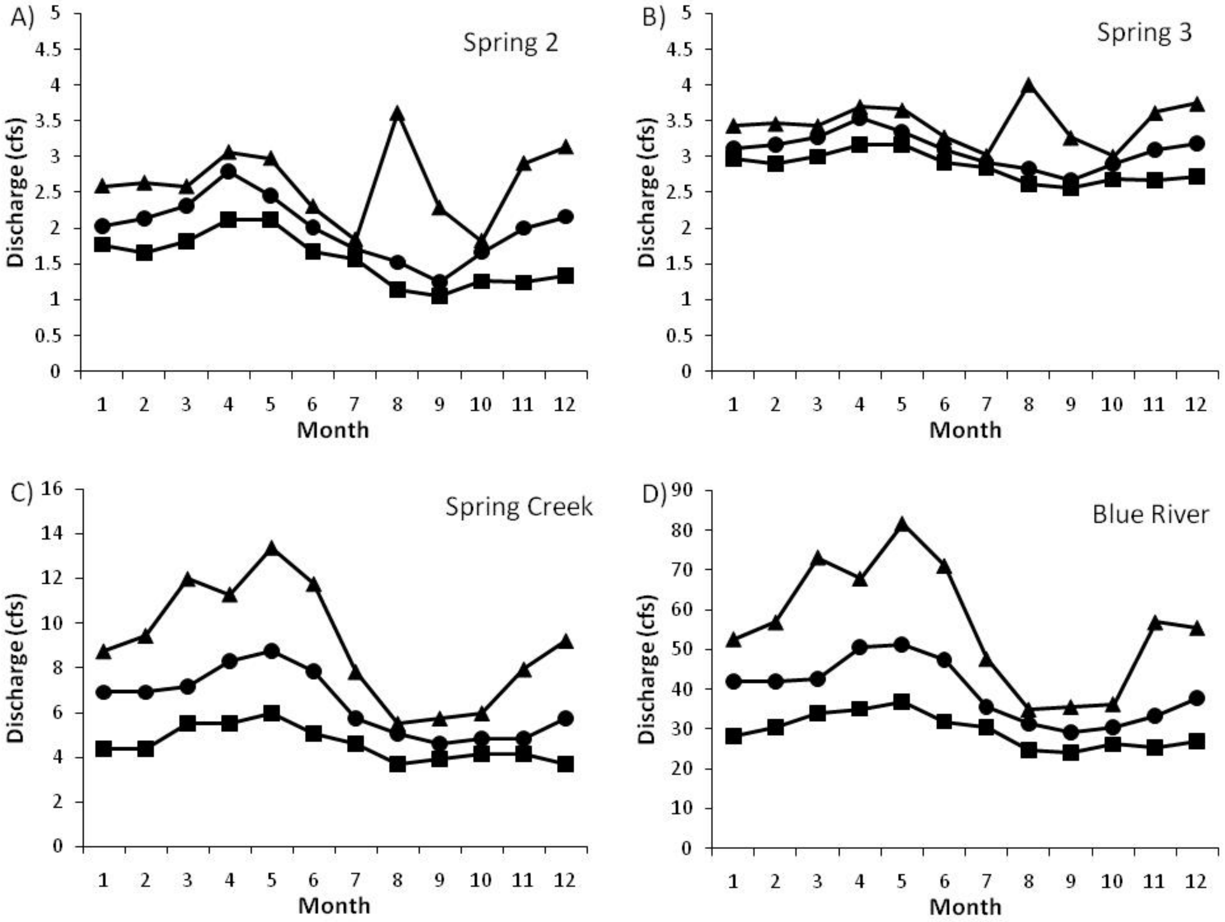

Figure 7: Monthly median (black circle), 25th percentile (black square), and 75th percentile (black triangle) discharge in A) Spring 2, B) Spring 3, C) Spring Creek, and D) Blue River.

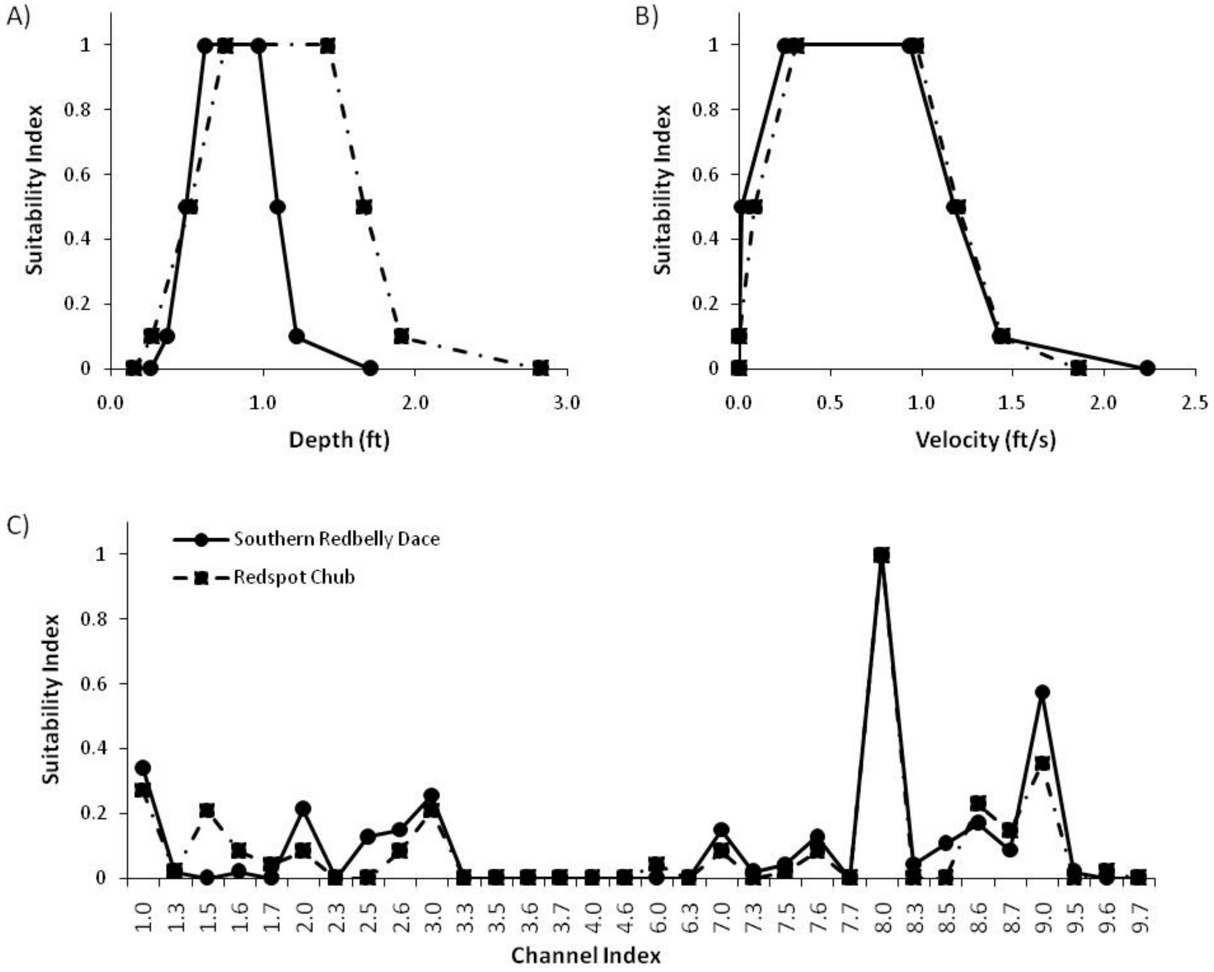

Figure 8: Habitat suitability criteria for southern redbelly dace (black circle and solid line) and redspot chub (black square and dot-dash line) in Spring 2.

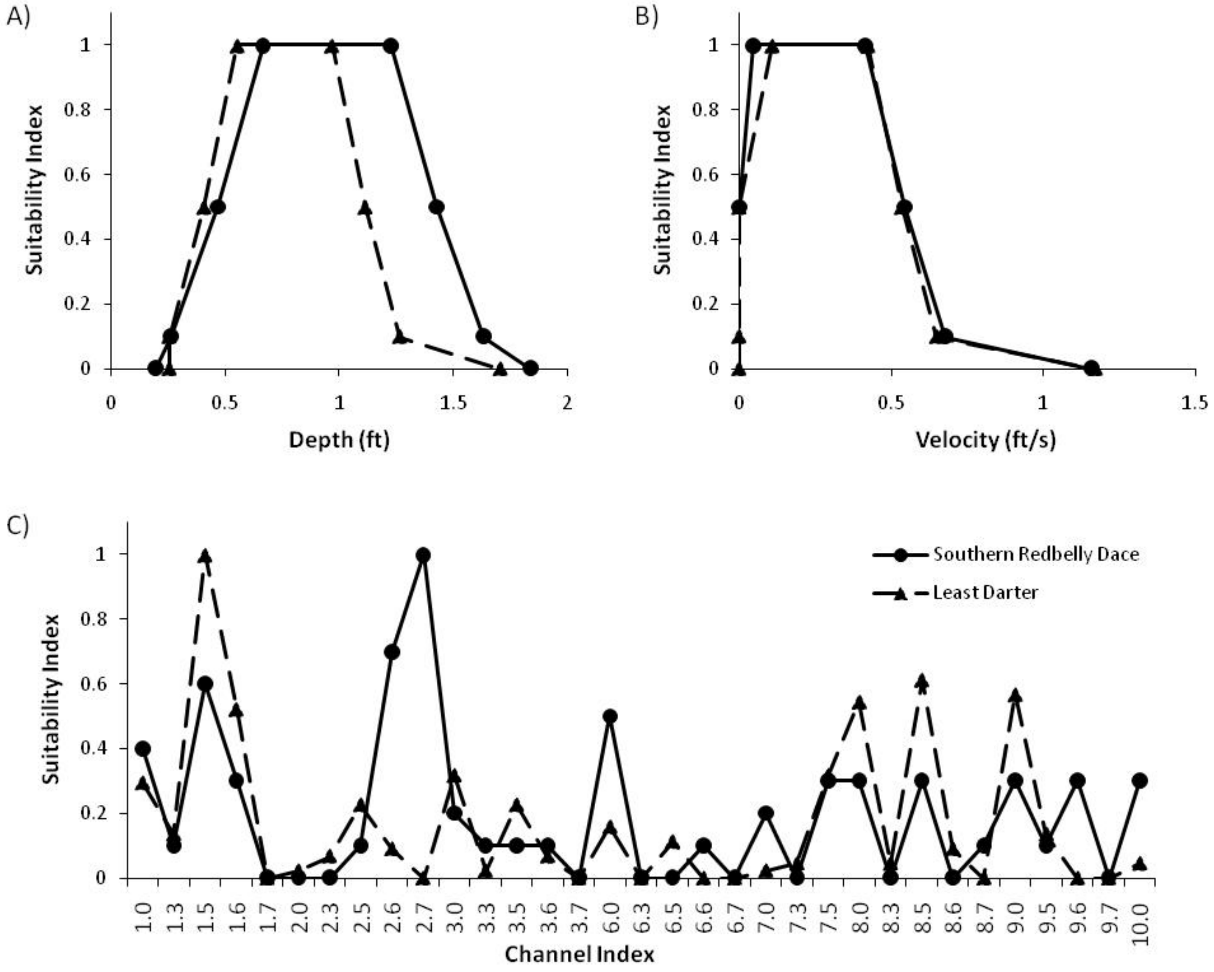

Figure 9: Habitat suitability criteria for southern redbelly dace (black circle and solid line) and least darter (black triangle and long dash line) in Spring 3.

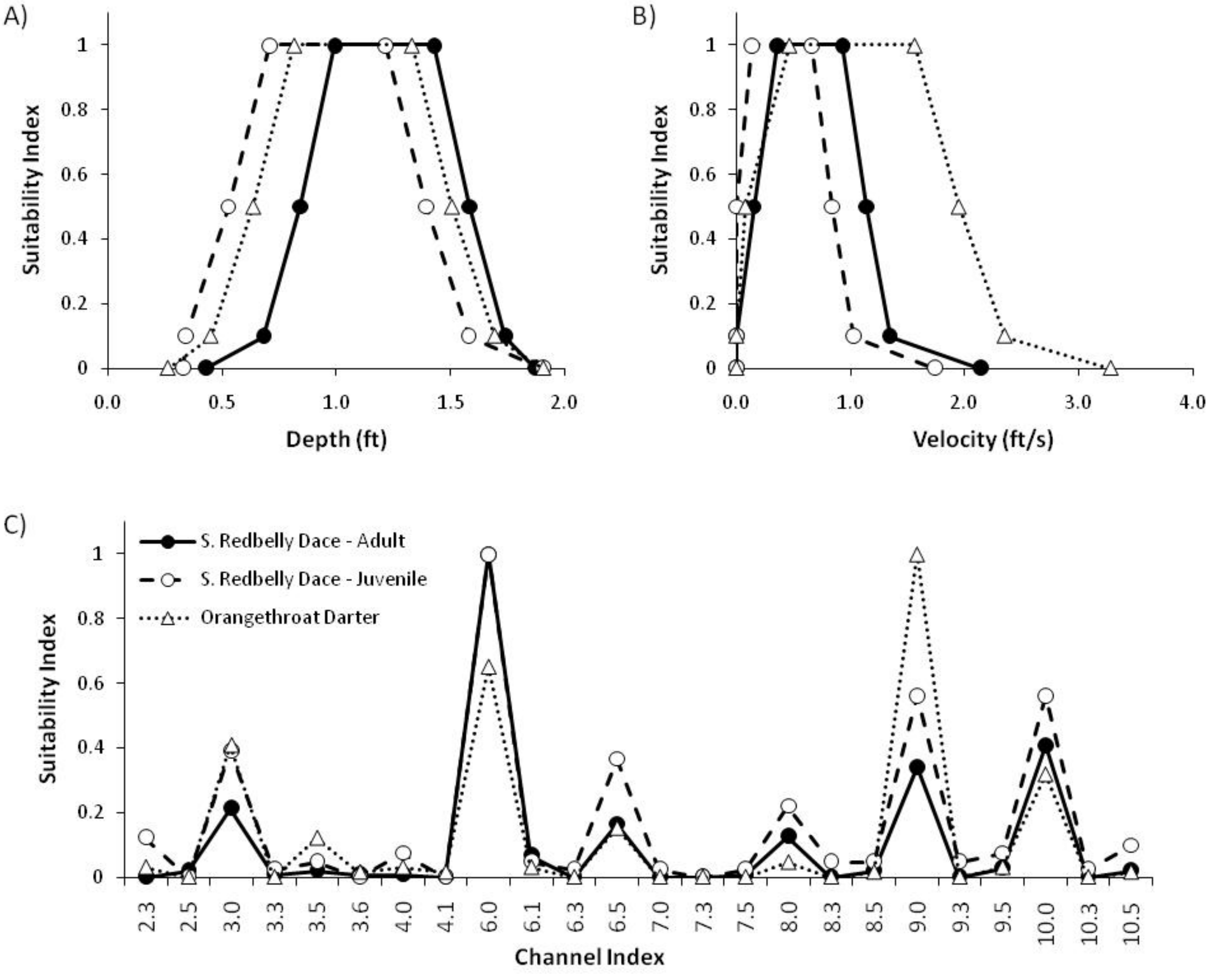

Figure 10: Habitat suitability criteria for adult southern redbelly dace (black circle and solid line), juvenile southern redbelly dace (hollow circle and short dash line), and orangethroat darter (hollow triangle and dot line) in Spring Creek.

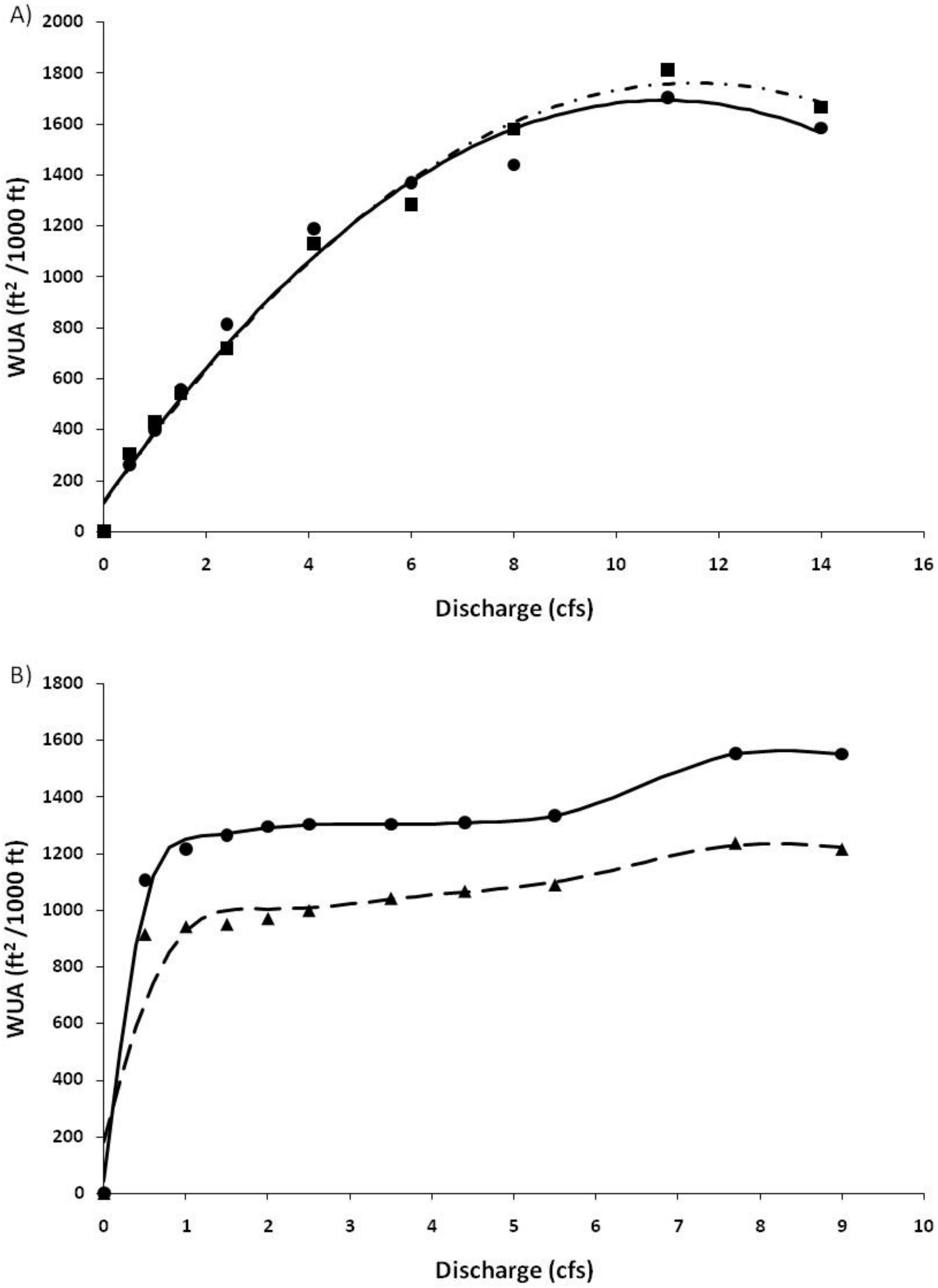

Figure 11: Relationship between weighted usable area (WUA ft^2/1000ft) and discharge for A) Spring 2 (southern redbelly dace [black circle and solid line] and redspot chub [black square and dot-dash line]) and B) Spring 3 (southern redbelly dace [black circle and solid line] and least darter [black triangle and long dash line]).

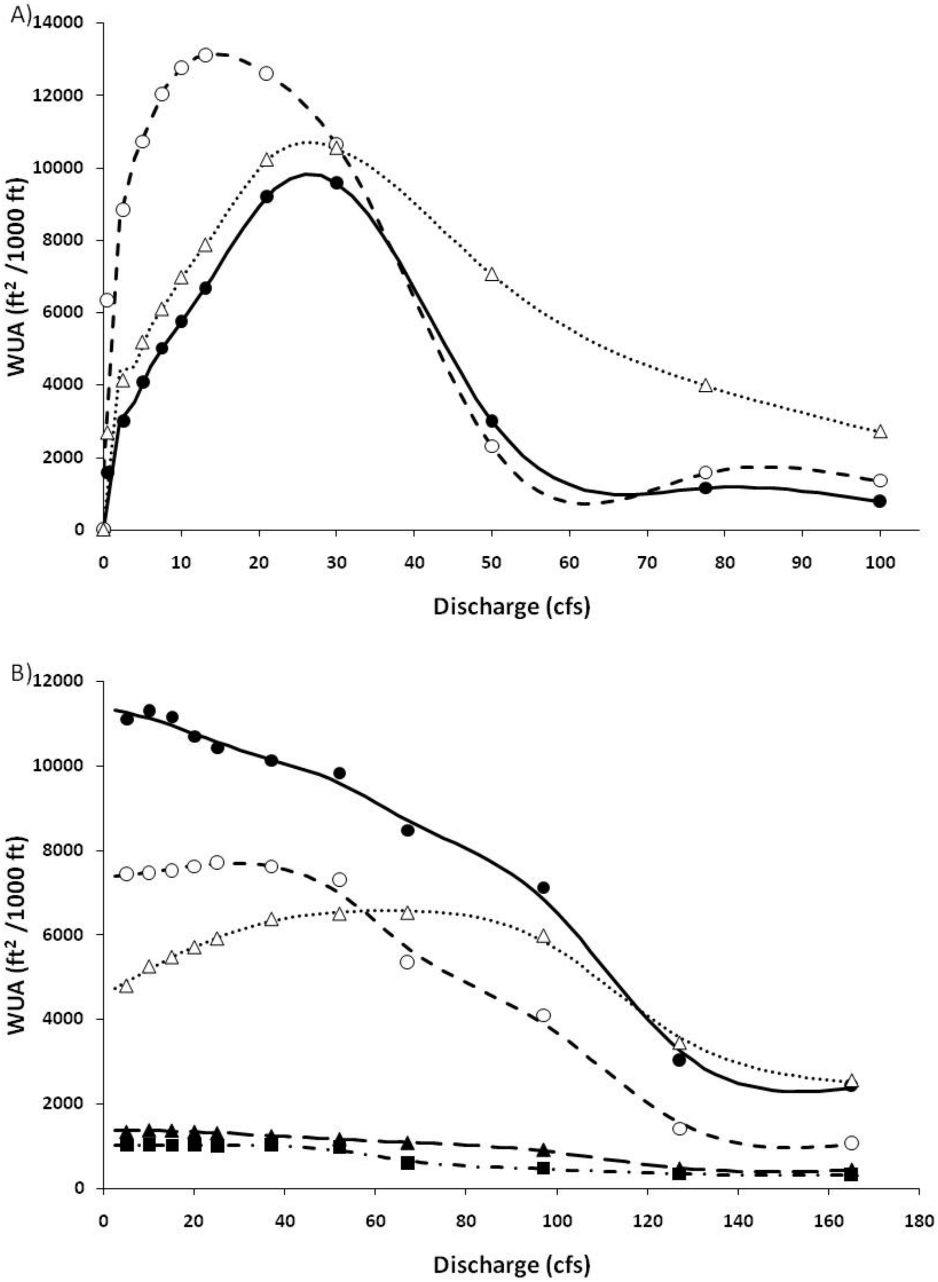

Figure 12: Relationship between weighted usable area (WUA ft^2/1000ft) and discharge for A) Spring Creek (adult southern redbelly dace [black circle and solid line], juvenile southern redbelly dace [hollow circle and short dash line], and orangethroat darter [hollow triangle and dot line]), and B) Blue River (adult southern redbelly dace [black circle and solid line], juvenile southern redbelly dace [hollow circle and short dash line], and orangethroat darter [hollow triangle and dot line], least darter [black triangle and long dash line], and orangethroat darter [hollow triangle and dot line]).

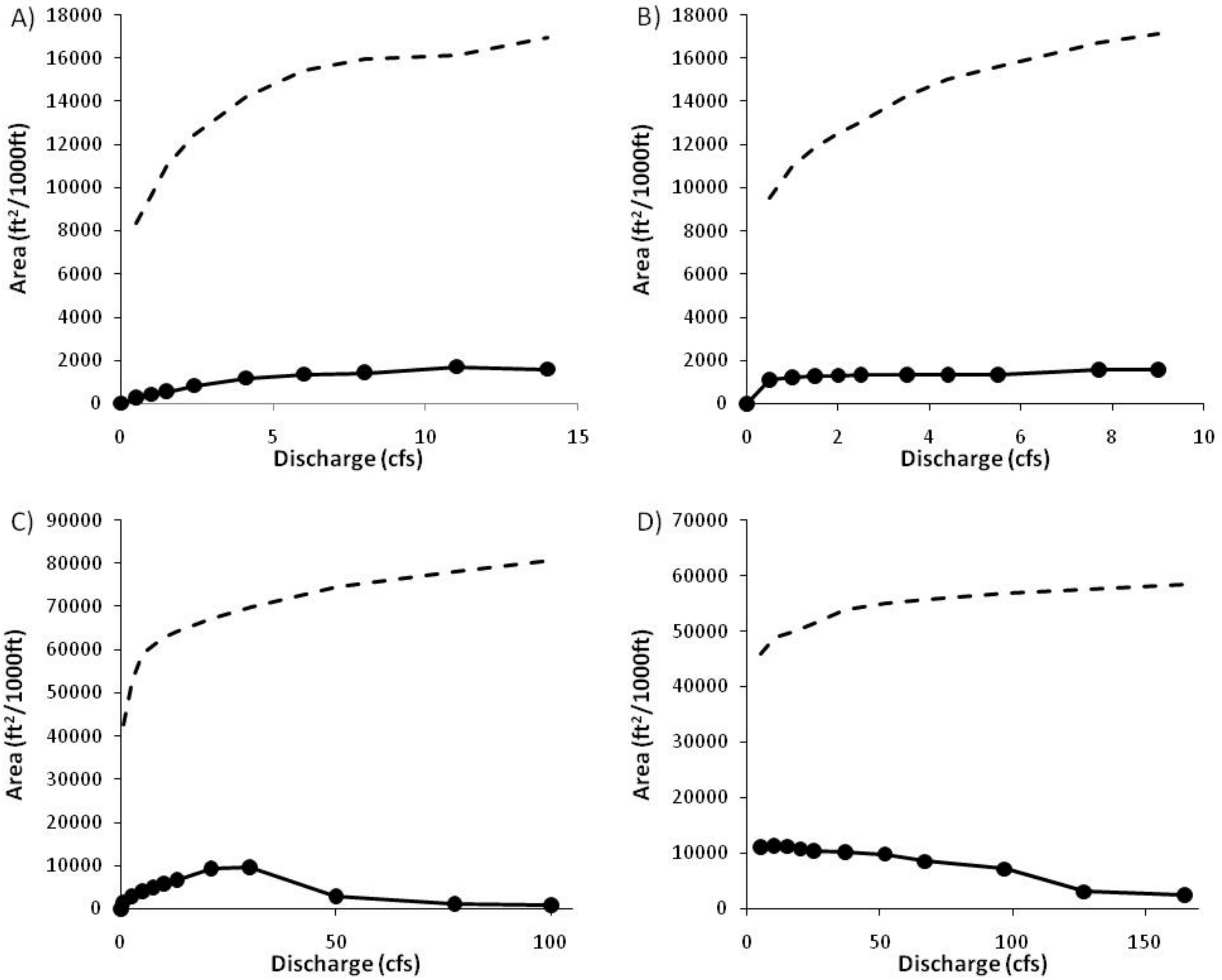

Figure 13: Total habitat area (dash line) and weighted usable area (WUA ft^2/1000ft) for southern redbelly dace (black circle and solid line) at A) Spring 2, B) Spring 3, C) Spring Creek, and D) Blue River.

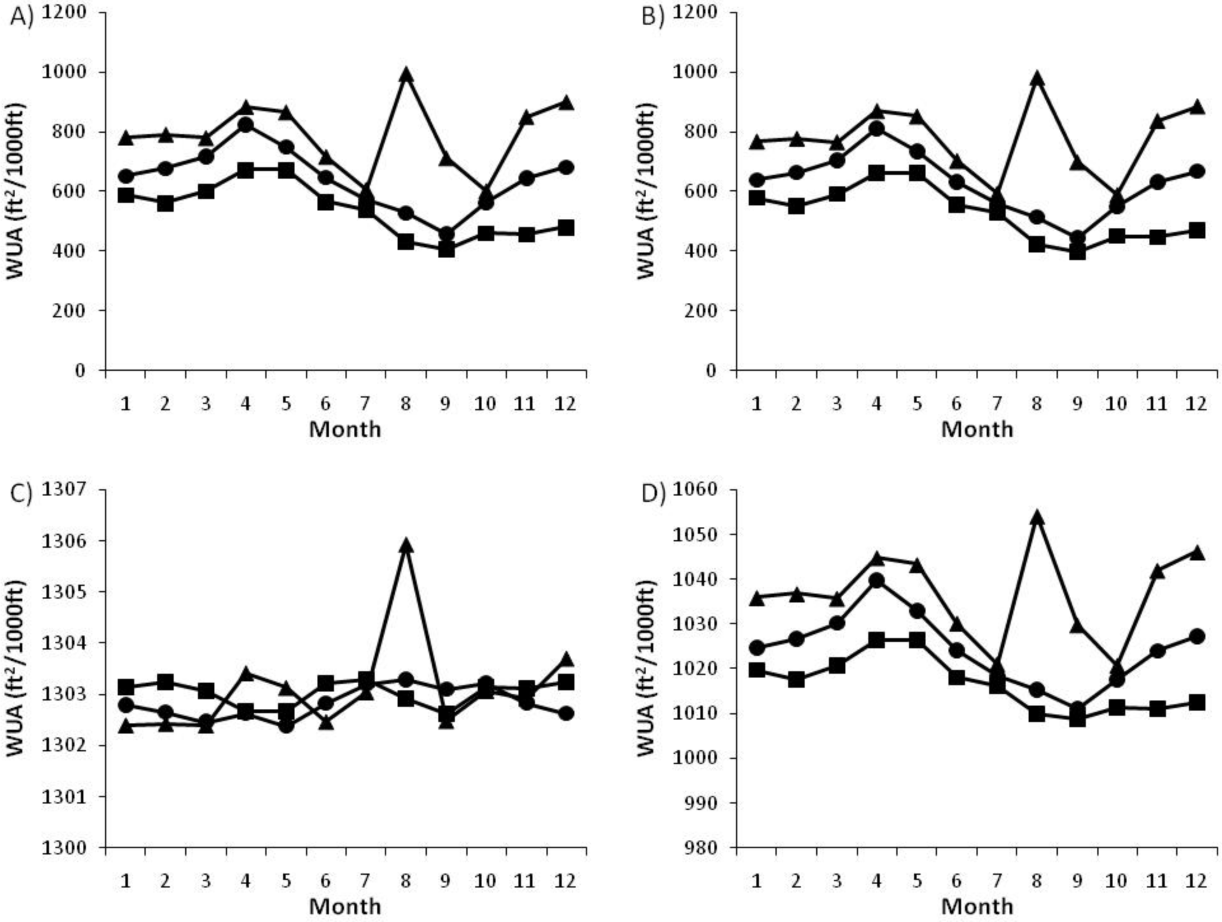

Figure 14: Time series analysis at monthly median (black circle), 25th percentile (black square), and 75th percentile (black triangle) discharge for Spring 2: A) southern redbelly dace and B) redspot chub; and Spring 3: C) southern redbelly dace and D) least darter.

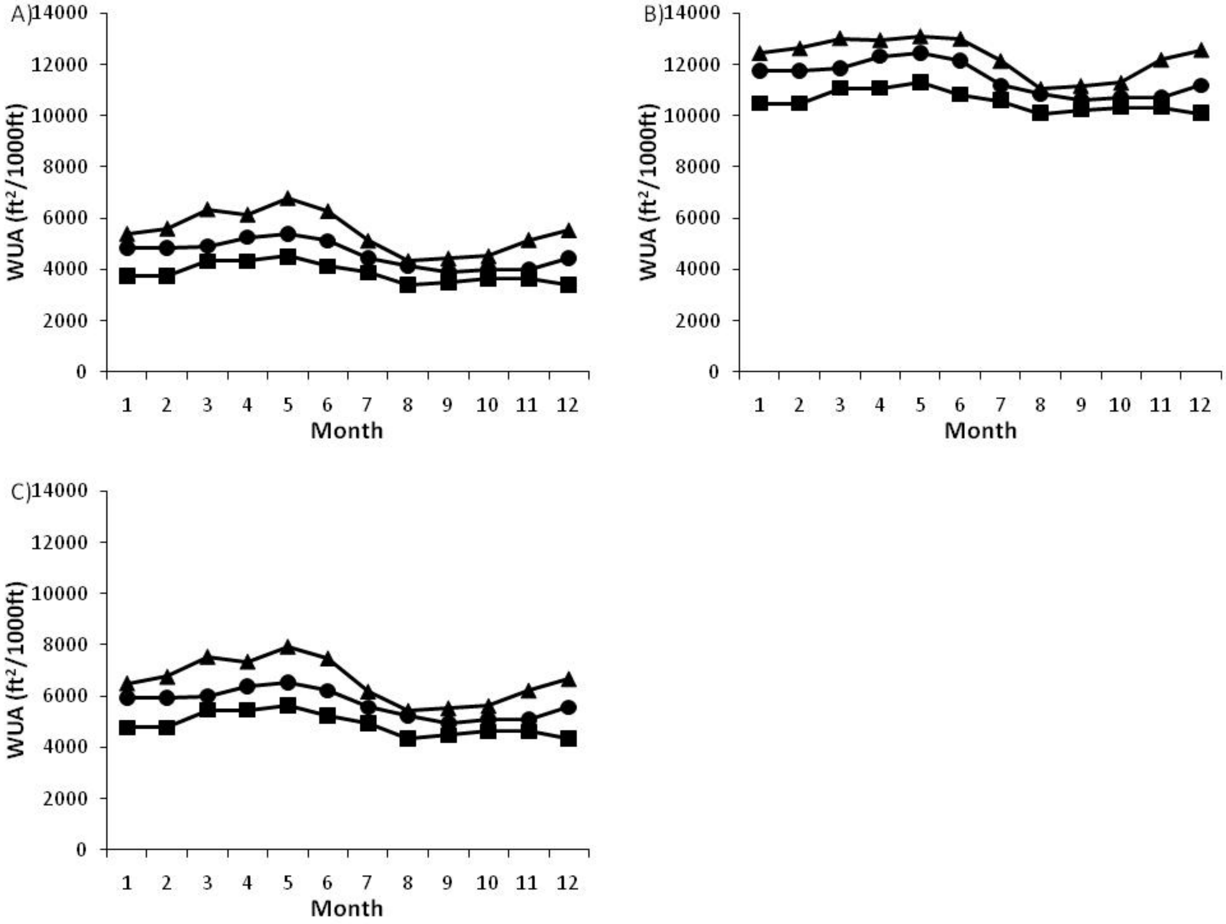

Figure 15: Time series analysis at monthly median (black circle), 25th percentile (black square), and 75th percentile (black triangle) discharge for Spring Creek: A) adult southern redbelly dace, B) juvenile redbelly dace, and C) orangethroat darter.

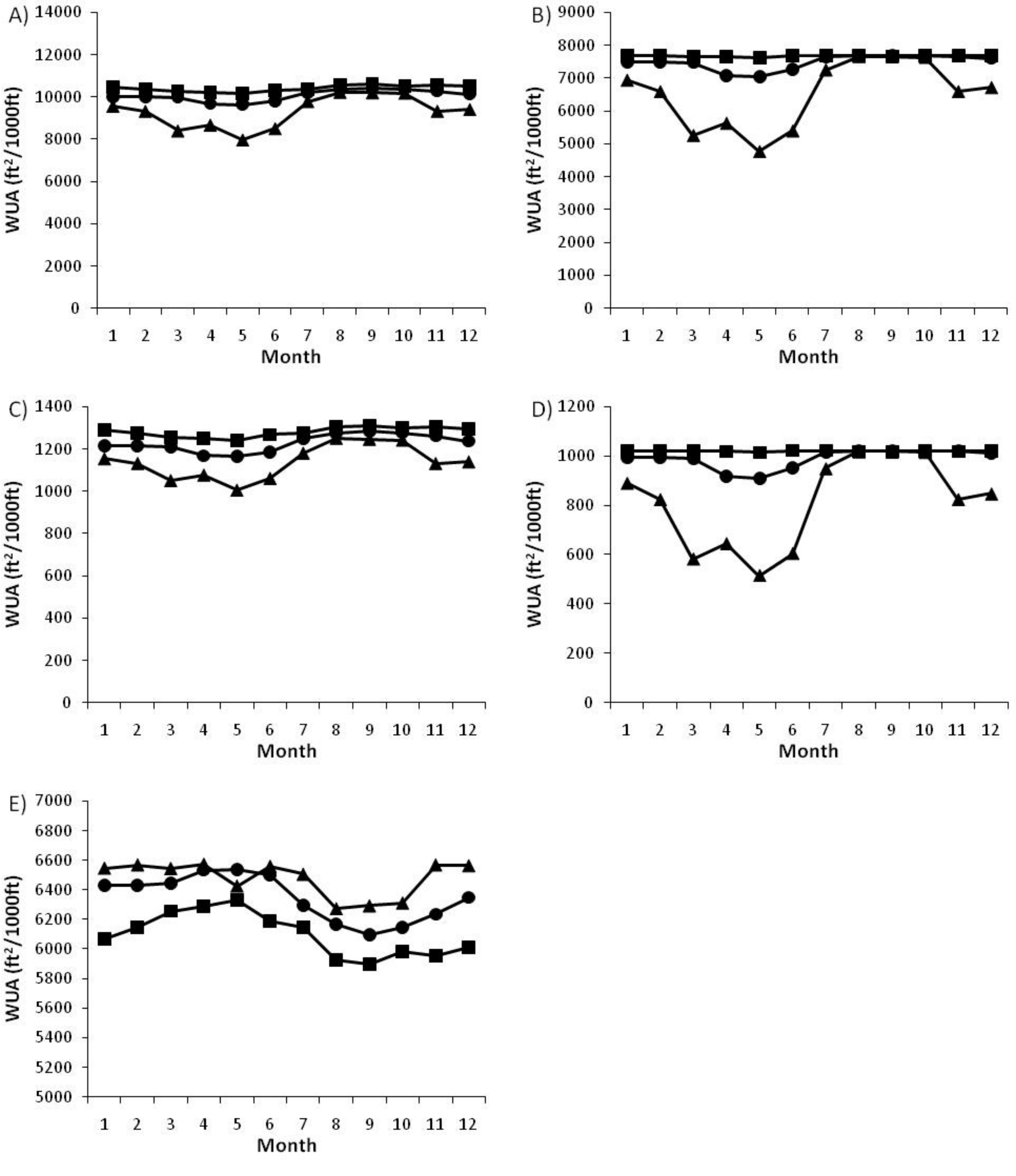

Figure 16: Time series analysis at monthly median (black circle), 25th percentile (black square), and 75th percentile (black triangle) discharge for Blue River: A) adult southern redbelly dace, B) juvenile redbelly dace, C) redspot chub, D) least darter, and E) orangethroat darter.

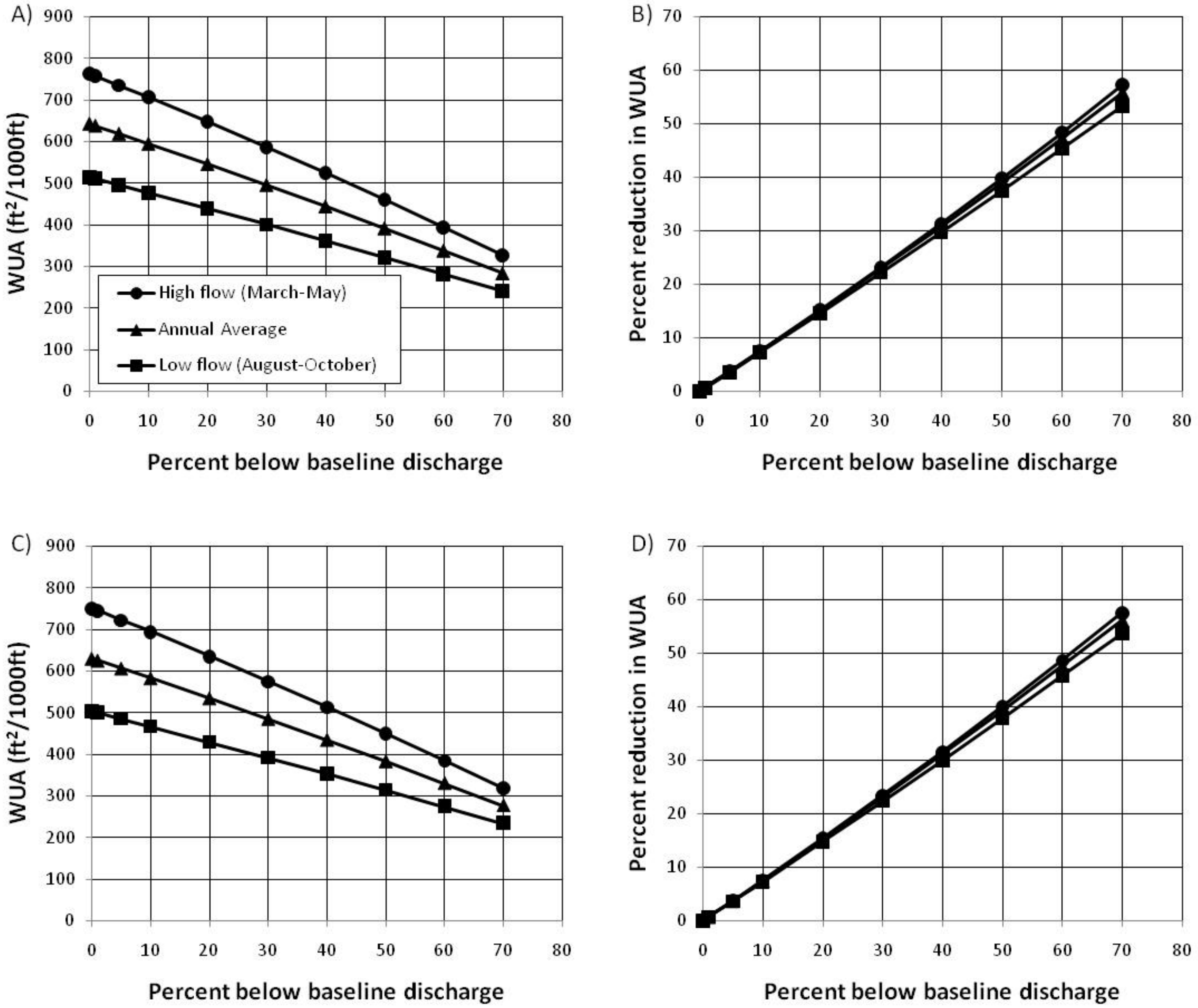

Figure 17: Incremental streamflow reduction scenarios (percent below baseline/median flow) and weighted usable area (WUA ft^2/1000ft) and percent reduction in WUA for southern redbelly dace (A and B) and redspot chub (C and D) in Spring 2.

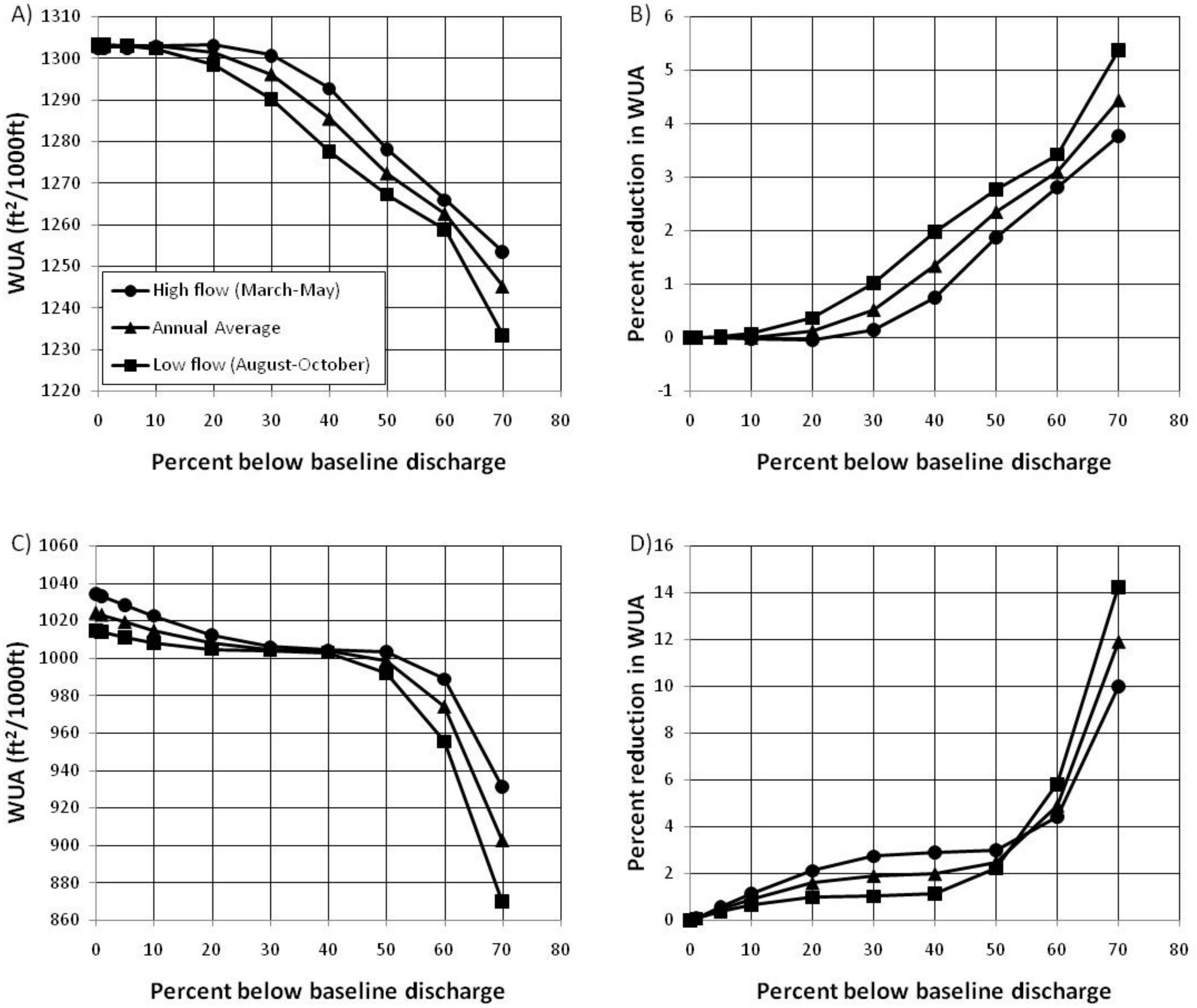

Figure 18: Incremental streamflow reduction scenarios (percent below baseline/median flow) and weighted usable area (WUA ft^2/1000ft) and percent reduction in WUA for southern redbelly dace (A and B) and least darter (C and D) in Spring 3.

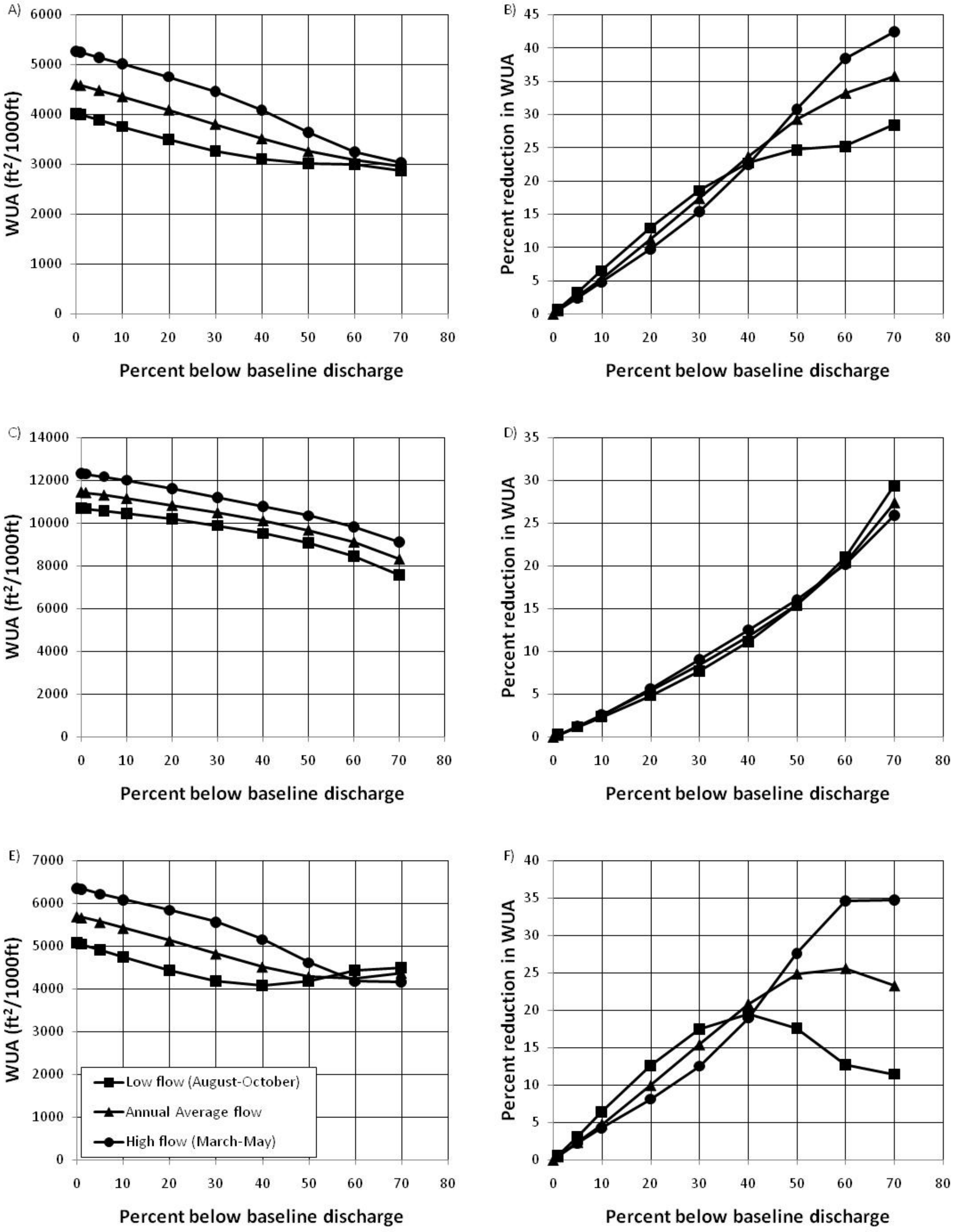

Figure 19: Incremental streamflow reduction scenarios (percent below baseline/median flow) and weighted usable area (WUA ft^2/1000ft) and percent reduction in WUA for adult southern redbelly dace (A and B), juvenile southern redbelly dace (C and D), and orangethroat darter (E and F) in Spring Creek.