Instream Flow Assessment of Mill Creek, a stream draining the Arbuckle-Simpson aquifer.

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Abstract

The availability of water is critical to both humans and ecosystems. Proposals have been made by rapidly expanding municipalities in central Oklahoma and elsewhere 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. The study site Mill Creek, a stream draining the Arbuckle-Simpson aquifer, has existing water uses for industrial gravel mining, which makes further withdrawals a potential danger to the aquatic ecosystem. We conducted an instream flow assessment using Physical Habitat Simulation (PHABSIM) on main channel and springs of Mill Creek using three fish species: one darter (orangethroat darter *Etheostoma spectabile*), one minnow (southern redbelly dace *Phoxinus erthyrogaster*), and one bass (spotted bass *Micropterus punctulatus*). 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 were two sections of main channel in Mill Creek and two spring-fed streams, located adjacent to the channel segments. The spring habitats meet the criteria for groundwater dependent ecosystems because they would not exist without the surface expression of groundwater. A total of 49 transects in the four sites were surveyed for channel elevation, and two sets of water surface elevation and water velocity were measured. Habitat suitability criteria were obtained from existing sources, including a recent study on Spring Creek, a tributary of Pennington Creek in the Arbuckle-Simpson aquifer, for darter and dace and published habitat criteria for spotted bass. 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 three target species. All fish species had losses of habitat when streamflow declined, but headwater and spring dependent species were more sensitive. In the Mill Creek habitats, orangethroat darter was more sensitive than the spotted bass to decreased stream flows. In the spring habitats, darter and minnow species experienced a maximum of 50% decline in WUA, especially in the spring-dependent southern redbelly dace, which had the largest losses of habitat area. Declines in the small quantity of preferred habitat that is available would likely degrade these populations of fishes.

Based on the findings of this study, groundwater removal from the Arbuckle-Simpson aquifer near springs may adversely impact availability of habitat for spring- dependent fish populations. Monitoring the spring flows throughout the year may be useful in developing management plans for maintaining flow in groundwater dependent ecosystems.

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 (e.g. flow regime) 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 1997; Rosenberg et al. 2000; Postel and Richter 2003) 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, 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 (Hamilton et al. 2003; Fitzhugh and Richter 2004). 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 (Reiser et al. 1989; Armour and Taylor 1991). The IFIM provides an organizational framework for evaluating and formulating alternative water management options when managing stream flows (Bovee 1982, 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 Oklahoma Comprehensive Wildlife Conservation Strategy (ODWC 2005) has identified the small rivers of the cross timbers region, including the Arbuckle-Simpson, as very high priory for conservation. The rivers are under stress from altered flow regime for aquifer pumping and water withdrawals, along with habitat alteration from gravel mining. Springs are also identified as moderate priority habitats (ODWC 2005) that are poorly understood but are likely under threat from altered stream flow and reduced water quantities associated with groundwater abstraction. The springs and spring-fed creeks of the Arbuckle-Simpson aquifer are unique habitat types that depend 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 and Froend 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).

Objectives

The objective of this study was to use the Instream Flow Incremental Methodology (IFIM) to assess instream flow requirements of selected fishes in the Mill Creek near Mill Creek, Oklahoma. We used IFIM and Physical Habitat Simulation System (PHABSIM) to model percid (darters), cyprinid (minnows and shiners), and centrarchid (bass) habitat. This study provides information to the Oklahoma Water Resource Board (OWRB) that will allow the agency to account for the impacts of flow reduction resulting from groundwater withdrawal on fish habitat in Mill Creek and its springs. This study will add to the knowledge concerning aquatic physical habitats of the Mill Creek and its springs.

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). Mill Creek is a stream draining the Arbuckle-Simpson. The watershed has water use in the form of groundwater and surface withdrawals for gravel mining operations. Current conditions should be understood before additional water is removed in order to minimize impacts to aquatic communities.

Hydrologic Time Series

An Indicators of Hydrologic Alternation (IHA) analysis by The Nature Conservancy (Tejan and Haase 2008) provided a detailed report on flow regime and altered streamflow in Arbuckle-Simpson springs and streams. They found that general seasonal trends in discharge are similar in streams draining the Arbuckle-Simpson due to the dependence on groundwater flow from a shared aquifer, although watershed size and precipitation that contribute runoff flow have a large influence on discharge in the more riverine sites (e.g. Mill Creek). 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 (Tejan and Haase 2008). Stream sites were found to have the highest median discharge from March to June and lowest median discharge from August to October. In this study we used the USGS stream gage on Mill Creek near the town of Mill Creek, Oklahoma to estimate discharge within the study area. The period of record for this gage is from 9/7/2006, so the dataset was extended using the longer record of the Pennington Creek near Reagan gage (10/1/2003). Because this gage was 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. Baseline is defined as the water supply, habitat values, or population status conditions that occur during a reference (i.e., recent historical) timeframe (Stalnaker et al. 1995).

Target Species

Three fish species were used in the instream flow analysis of the Mill Creek streams and springs. Three families were represented, including the Percidae family (perch and darters, orangethroat darter *Etheostoma spectabile*), the Cyprinidae family (minnows and shiners, southern redbelly dace *Phoxinus erthyrogaster*), and the Centrarchidae family (sunfish and bass, spotted bass *Micropterus punctulatus*). The orangethroat darter is widespread throughout the central United States (Miller and Robison 2004) and was present in the Spring Creek study site of the previous IFIM study (Seilheimer and Fisher 2008). The southern redbelly dace was selected because it is limited to only a few watersheds in southern Oklahoma, which is also the southern limit of its 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). Spotted bass are limited to the eastern half of Oklahoma, with Mill Creek being on the edge of their western boundary of their distribution (Miller and Robison 2004). The spotted bass is considered to be an important game species in the state of Oklahoma (http://www.wildlifedepartment.com/sbass.htm). The three species prefer clear and cool water in spring-fed streams with gravel substrate (Robison and Buchanan 1988; Miller and Robison 2004).

The fish communities of the springs in the Arbuckle-Simpson are 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 nearby springs 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*, and orangethroat darter (Matthews et al. 1985). Study of fish occurrence in Mill Creek is limited to a survey conducted in 1974-75, which found a total of 46 species from 12 families (Binderim 1977). The study classified species within Mill Creek primarily in two types of habitat: lowland stream habitat with higher turbidity and soft sediments (23 species), and upland habitat with clear water flowing over sand, gravel, and bedrock (15 species). Orangethroat darters were found throughout the watershed (21 of 27 sites) but were in highest abundance in the headwaters of Mill Creek, whereas southern redbelly dace only occupied two spring-fed tributaries in the southern part of the watershed (Binderim 1977). Spotted bass were less common than orangethroat darter (11 of 27 sites) but were found in all parts of the watershed.

Methods

Site Description

Mill Creek is located in western Johnston County and eastern Murry County in southern Oklahoma and flows into the Washita River above Lake Texema (Fig. 1). The four study sites were located throughout the watershed with two main channel sites, called Mill1 and Mill2, and two spring sites, called Colvert and Springhouse. Mill1 was located in the middle of the watershed downstream from the Mill2 site and both were near the town of Mill Creek, OK. The springs were separated by a greater distance with Colvert in the upper watershed adjacent to Mill2, and Springhouse was located in the lower watershed. Both stream sites were straight, although Mill1 had a braided section in the middle of the segment (Fig. 2a) and Mill2 had a slight bend near the bottom of the segment (Fig. 2b). Colvert was the most meandering of the study sites (Figure 2b), while Springhouse had a long straight course in the upper portion and some meanders in the bottom portion (Fig. 2c). In the stream sites, Mill1 was the longer (1801.5 ft) than Mill2 (231.8 ft; Table 1). There were also differences in the total length of the spring sites, with Colvert (232.2 ft) being more than two times shorter than Springhouse (661.2 ft; Table 1). The mean wetted width was highest in stream sites where Mill1 (42.9 ft) was wider than Mill2 (34.7 ft). The spring sites were smaller than the stream sites with Colvert (14.2 ft) an average of four feet wider than Springhouse (8.2 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 (gravel and cobble) and three different types of vegetation: emergent vegetation (e.g. terrestrial vegetation, water willow *Justicia americana*), floating vegetation (waterlily *Nymphaea odorata*), and submergent vegetation (coontail *Ceratophyllum demersum*, 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.

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, 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 March-May 2009. 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 boundaries for the Mill1 and Mill2 sites were placed at the head of riffles where the water became shallow. The lower boundary for Colvert and Springhouse sites was placed near the confluence with the Mill Creek at a narrowing of the channel. 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 49 transects were placed within the four study sites (Table 1). In the stream sites, Mill1 had 19 transects and Mill2 had 7 transects. (Table 1). In the spring sites, Colvert had 11 transects, while Springhouse had 12 transects. Transects were placed in the Mill2 site within 300 ft upstream and downstream from the mouth of Colvert.

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 combination of an auto level (Topcon AT-G series) and total station (Topcon GTS-235W Electronic Total Station) with a 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 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).

Water Depth and Temperature

We measured hourly water depth and temperature in Mill1, Colvert, and Springhouse using unattended temperature loggers (Fig. 3). Timing of logger deployment resulted in different start times for monitoring: 159 days at Mill1 (1/1/09-6/8/09), 108 days at Colvert (2/21/09-6/8/09), and only 20 days at Springhouse (5/20/09-6/8/09). The logger was located in Mill1 at the most downstream transect. The loggers in the spring sites were placed in the upper end of the site, near the topmost transect (Fig. 2a). The Colvert logger was located in a pool above a beaver dam (Fig. 2b), while the probe at Springhouse was located in a concrete-sided channel (Fig. 2c). A decrease in depth starting in March was likely the result of a breach of the dam, which lowered the surface elevation of the pool that contained the probe.

Transect Profile Data

Channel cross-sections were described as a series of x and y coordinates called verticals. Verticals were measured across each transect at intervals of 3 feet at Mill1, 2 feet at Mill2, and 1 foot at Colvert and Springhouse. Channel profile data associated with each vertical included a horizontal and vertical distance from a known datum measured to the nearest 0.1 ft, water surface elevation measured to the nearest 0.1 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). Two sets of water surface elevation and velocity measurements at each transect in each site were taken on different days and at different streamflows at Mill1, Mill2, and Colvert, which allowed us to produce a stage-discharge relationship for each site. One set of streamflows were measured at Springhouse.

Fish Sampling

The presence of the target species was confirmed by backpack electrofishing (LR-24 Electrofisher, Smith-Root, Inc., Vancouver, WA). Presence of the species at the study sites allowed us to proceed with the habitat modeling. Although we did not observe juvenile spotted bass at the Colvert site, it was plausible that the species could move from the stream to the spring.

Physical Habitat Simulation

For each variable (depth, velocity, substrate/cover), we used existing species-specific habitat suitability criteria (HSC) curves. A previous study on fish habitat in Arbuckle-Simpson springs developed HSC on nearby Spring Creek for orangethroat darter and southern redbelly dace (Seilheimer and Fisher 2008). HSC for spotted bass were available for multiple life stages in a U S Fish Wildlife Service report (McMahon et al. 1984). We selected the juvenile and adult stages of spotted bass because they had been captured in electrofishing surveys. 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.

Hydrologic 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 the "stage discharge" (STGQ) model and a MANSQ model, with is based on Manning's equation (Waddle 2001). The STGQ model was used in the Mill1, Mill2, and Colvert sites. 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 MANSQ model for the transects at Springhouse because the STGQ model requires more than one set of measurements. The MANSQ model assumes that each cross section is independent of each other and is also tested by comparing simulated with observed water surface elevations (Waddle 2001).

The differences in lowest and highest simulated water surface elevations for the stream transects were larger than the spring transects (Fig. 4). Mill1 had the largest mean difference between the lowest and highest simulated discharge (1.3 feet; Fig. 4a) with a range of 0.6 to 1.7 feet. Mill2 had a slightly lower mean difference of 1.2 feet but a narrower range of 0.9 to 1.4 feet (Fig. 4b). Simulated water surface elevation showed the smallest magnitude (mean difference: 0.3 feet) and variability (0.2 to 0.8 feet) in Colvert (Fig. 4c). Springhouse had a 0.85 foot mean difference between lowest and highest simulated elevations and a range of 0.4 to 1.4 feet (Fig. 4d).

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 0.1 cfs and 100 cfs in Mill1, between 0.5 cfs and 100 cfs in Mill2, between 0.5 cfs and 5.5 cfs in Colvert, and between 1 cfs and 6 cfs in Springhouse. 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. Flow was then simulated for the entire record from 2003 to 2009.

The gage data for Mill Creek was limited to a short time period (October 2006 to present) at the Mill Creek near Mill Creek gage (#07331200). We used linear regression between observed streamflows (and baseflows) at the Mill Creek gage and the Pennington Creek near Reagan, OK gage (#07331300) to extended the streamflow record to 2003. Baseflow for the extended Mill Creek gage was estimated using the BFI software (Bureau of Reclamation; http://www.usbr.gov/pmts/hydraulics_lab/twahl/bfi/). There was a significant relationship between Mill Creek and Pennington Creek for streamflow (R^2 0.74, $P<0.0001$; Fig. 5a):

Log Mill Creek Simulated Flow = -2.241729+1.2920577 * (log(Pennington Creek at Reagan streamflow (cfs)))

and baseflow (R^2=0.73, P<0.0001):

Log Mill Creek baseflow (cfs) = -2.421987 + 1.2358637 * (log(Pennington Creek at Reagan baseflow (cfs)))

Discharge at the study sites were estimated with linear regression between Mill Creek gage and observed measurements. We used measured discharge on three dates from January to June in 2009 for each of the four study sites. Streamflow measurements were used to estimate Mill1 and Mill2, and baseflow to estimate flow in the Colvert and Springhouse. Daily mean discharge was then simulated for the extended record. Significant linear regression between extended Mill Creek data and observed streamflow was used with a y-intercept of zero, which corrected for the negative estimated flows at low flows in Pennington Creek that result from a negative intercept. Regressions to simulate Mill1 (Fig. 5b):

Mill1 steamflow = 0 + 0.7737419 * log(Mill Creek streamflow (cfs))

,and Mill2 (Fig. 5b):

Mill2 steamflow = 0 + 0.5679203 * log(Mill Creek streamflow (cfs))

Simulation of the spring sites used baseflow estimates from the Mill Creek gage. Regression to simulate Colvert (Fig. 5c):

Log Colvert streamflow = -0.105776 + 0.3726947 * log(Mill Creek baseflow (cfs))

,and Springhouse (Fig. 5c):

Log Springhouse streamflow = -0.744475 + 0.5254391 * log(Mill Creek baseflow (cfs))

The estimated flows provide an estimate of site specific discharge that will be used to simulate fish habitat. 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 also collected stage height at Mill1, Colvert, and Springhouse with a Solinst Levelogger (Fig. 3a; Solinst Canada Ltd., Georgetown, Ontario). Beaver activity in Colvert caused changes in water depth that were not related to discharge, so we used the baseflow from the USGS gage at Mill Creek near Mill Creek, OK. The time period of the depth data was not sufficient to estimate stream flow in all the study sites, so alternative methods were selected.

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; Fig. 6). 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 here and in the Arbuckle-Simpson streams (Tejan and Haase 2008). There were pronounced seasonal trends in the flows of Mill1 (Fig. 6a) and Mill2 (Fig. 6b) with higher magnitude of flows downstream at the Mill1 site. 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. Seasonal trends in discharge from the spring sites were nearly 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 (from 0.25 to 0.5 cfs) when groundwater flows are influenced by regional precipitation trends. Discharge was higher in Colvert (Fig. 6c) than Springhouse (Fig. 6d). During dry years, the spring flow fell as low as 0.65 cfs in the Springhouse site.

Groundwater diversion in the Arbuckle-Simpson aquifer would have the greatest impact on aquatic ecosystems if large quantities of water were removed during dry months and years. 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). The extended streamflow records in this study are largely dependent on the years of gage record at the Pennington Creek near Reagan site, which is only 5 years. Thus, the seasonal flows that are estimated for the study sites should reflect the general patterns of the Arbuckle-Simpson region but more field based data would be useful for refining the expected impacts of changes in flow regime and water supply at specific sites.

Habitat Characteristics

There was a large difference in available surface area and volume between and among the stream and spring sites (Fig. 7a-b). Streams had more total habitat with Mill1 being larger than Mill2, which was expected from the general differences in length and width of sites. In the spring sites, there was more total habitat in Colvert than in Springhouse (Fig. 7c-d). All sites showed an increase in surface area and volume with increased discharge and a drop in area during periods of very low flow. Stream sites were faster in than the springs, with Mill1 having a lower mean velocity in spring (1.1 ft/s) compared to Mill2 (1.5 ft/s), while mean depth was similar in both sites (Table 1). Colvert and Springhouse had similar mean velocity (0.7 and 0.6 ft/s, respectively), and depth (0.5 ft and 0.9 ft, respectively; Table 1). The most common substrate types in Mill1 were 4 (cobble, 26%), 6 (bedrock, 24%), 9 (gravel and bedrock, 21%), and 3 (gravel, 15%; Table 3), and there was more area with no cover (95%) than with cover (5%). The most common substrate type in Mill2 was 6 (bedrock, 94%) with no cover being common (96%; Table 3). The most common substrate types at the Colvert site were 6 (bedrock, 37%), 4 (cobble, 18%), 8 (sand and gravel, 16%), and 1 (clay/silt, 12%; Table 3). The Colvert site had the most diverse available cover with large gravel and cobble (29%), submergent aquatic vegetation (18%), and assorted emergent vegetation and woody debris (Table 3). The substrate types in the Springhouse site were mostly smaller substratum with gravel (46%), clay/silt (22%), and gravel/cobble (10%). There was very little available cover in Springhouse (97% no cover; Table 3).

Daily depth measurements were the most variable in Mill1 compared to the spring sites. The low water period early in the year showed stable depths but there was more fluctuation after March when precipitation increased (Fig. 3a). There was less variation in the depth of the spring sites. Higher depths in Colvert corresponded to high flow events in Mill1 in May 2009 (Fig. 3a). Springhouse had minimal variation in depth but the period of data logging was short (Fig. 3a). Mean temperatures were variable in Mill1 because of changes in seasonal temperature (Fig. 3b). Although there was also variation in water temperature at Colvert, the magnitude was much lower than Mill1. The location of the levelogger may have influenced the temperature changes because the measurements were taken in a sunny area, which became shallower following a decline in water depth from a breached beaver dam, but groundwater buffered the changes in early spring. The temperature at Springhouse was very stable (Fig. 3b) and similar to observations of temperature stability in other Arbuckle-Simpson springs (Seilheimer and Fisher 2008). The stable temperatures of the spring habitats can provide refuge habitat for fish from more extreme temperatures that occur in the main channel of Mill Creek.

PHABSIM Modeling

We used two models to describe the relationship between weighted usable area (WUA) and streamflows using PHABSIM. The relationship between flow and WUA at Mill1, Mill2, one of two species at Colvert, and Springhouse 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:

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

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 Mill1 (orangethroat darter [$R^2 = 0.99$; Table 4a], juvenile spotted bass [$R^2 = 0.99$; Table 4b], adult spotted bass [$R^2 = 0.99$; 4c]), Mill2 (orangethroat darter [$R^2 = 0.99$; Table 4d], juvenile spotted bass [$R^2 = 0.99$; Table 4e], adult spotted bass [$R^2 = 0.99$; 4f]), Colvert (juvenile spotted bass [$R^2 = 0.96$; Table 5a]) and Springhouse (orangethroat darter [$R^2 = 0.95$; Table 5b], southern redbelly dace [$R^2 = 0.99$; Table 5c]).

The relationship between flow and WUA using the spline model had similar regression curve shapes between species within sites but curve shapes differed between sites. The WUA for Mill1 was highest for the orangethroat darter at 25.0 cfs (9,162 ft^2; Fig. 8a) with slightly higher WUA at lower discharge for juvenile spotted bass (12.5 cfs, 9,470 ft^2; Fig. 8a) and adult spotted bass (17.5 cfs, 9,709 ft^2; Fig. 8a). There was a steep decline in the amount of habitat as discharge was reduced below the maximum, while the decline was more gradual with increasing discharges. The WUA for Mill2 species showed a similar pattern to Mill1 with orangethroat darter having a maximum WUA at a higher discharge (20 cfs, 11,079 ft^2; Fig. 8b). Spotted bass WUA was much lower than the orangethroat darter at lower discharge (juvenile: 3,071 ft^2 at10 cfs, and adult: 3,382 ft^2 at 15 cfs, ; Fig. 8b). The maximum WUA for the species at Colvert occurred at 5.6 cfs for the juvenile spotted bass (1,114 ft^2; Fig. 9b). The juvenile spotted bass habitat in Colvert showed very little variation with decreases in discharge, which indicates that it does not respond to changes in flow in spring environments. In Springhouse, both species had their maximum WUA at 5.2 cfs (orangethroat darter 952 ft^2, and southern redbelly dace 1,315 ft^2; Fig. 9b). Both species have a threshold flow of 3 cfs where increases in discharge have minimal increases in WUA, while decreases in discharges result in large declines in WUA.

For a single species at the Colvert site, we used a quadratic regression (i.e. parabolic shaped curve) to describe the relationship (orangethroat darter: $R^2 = 0.99$, $P<0.01$; Fig. 9b):

$$\text{Orangethroat darter WUA} = (441.1073 + 169.96292 * x) - 38.398843(x - 2.88571)^2$$

where, x = discharge (cfs). Maximum WUA occurred at the 5.8 cfs for the orangethroat (1704 ft^2). WUA dropped steadily below the maximum to a low amount of habitat at the lowest simulated flows (0.5 cfs). Orangethroat darters seem to have a threshold flow of 3.5 cfs, where lower discharges lead to steady declines in WUA.

Habitat Time Series

We calculated WUA for median monthly streamflows for the study sites using the discharge-WUA regression models (Figures 10 and 11). This habitat time series analysis enabled the establishment of baseline habitat conditions for each species during the low-flow months (August to October) and high-flow months (April to June) for use in alternative analysis.

The WUA for Mill1 and Mill2 species varied seasonally between wet, baseline, and dry years. All three Mill1 species had maximum habitat available in May and the minimum in September (Fig. 10a-c). The orangethroat darter (Figure 10a) had the largest seasonal range in habitat from 5,000 to 7,000 ft^2, while the juvenile and adult spotted bass typically range from 8,000 to 9,000 ft^2 between seasons (Fig. 10b and 10c). In wet years, higher WUA occurred from January to May for all species, and dry years resulted in declines in habitat for all months relative to baseline flows (Fig. 10a-c). The species of Mill2 showed similar trends in WUA with higher seasonal variation in orangethroat darter WUA (5,000 to 8,000 ft^2; Fig. 10d), while both stages of spotted bass had very little variation in WUA (2,500 to 3,000 ft^2; Fig. 10e and 10f). The low WUA in spotted bass and little response to flow change may indicate that the Mill2 habitat is marginal for the species. Wet years resulted in more WUA for the orangethroat darter, especially early in the year but spotted bass did not have large increases in habitat due to increased flows. In dry years, all species had large declines in WUA during the months of February to April, which could have impacts on the availability of spawning and nursery habitat (Sparks 1995).

The weighted usable area in Colvert for orangethroat dater (Fig. 11a) and spotted bass juveniles (Fig. 11b) was highest in the spring and lowest in the late summer. Orangethroat darter had a larger range of WUA than the spotted bass, which showed very little variation with discharge (960 to 970 ft^2; Fig. 9a). The trends in WUA for Springhouse were similar to Colvert (Fig. 11c-d). Trends for orangethroat darter were similar throughout the year and between wet, baseline, and dry years, with more variation between wet and dry years in the spring (Fig. 14c). The southern redbelly dace had a larger range of variation in WUA than the orangethroat darter (Fig. 14d). The southern redbelly dace also had larger differences in WUA between wet and dry years in the early months of the year, while later in the year had less variation between wet and dry conditions. The spring species would be at greatest risk from flow alterations during dry months when flow is naturally lower.

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 Mill1, Mill2, Colvert, and Springhouse. 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 fishes at any level with future diversions from Mill Creek and 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 orangethroat darter and spotted bass in Mill1 (Table 6; Figure 12a-f). In all scenarios, orangethroat darter lost a greater percentage of habitat than spotted bass. The declines in habitat were consistent throughout the season in orangethroat darter but spotted bass had higher percentage of declines in wet months. A 1% decline in baseline discharge resulted in an average 0.4% decline in orangethroat darter WUA, while reductions were less for juvenile (0.2%) and adult (0.2%) spotted bass. A 20% baseline decline resulted in a 9.5% loss of WUA in orangethroat darter, 2.7% loss in juvenile spotted bass, and a 3.7% decline in adult spotted bass. A 50% decline in baseline streamflow resulted in a 28.2% loss of orangethroat darter WUA, a 6.6% loss in juvenile potted bass WUA, and a 9.0% loss of adult spotted bass WUA. In the 70% decline scenario, there is a 45.9% decline in orangethroat darter WUA, 9.2% decline in juvenile spotted bass WUA, and a 10.4% decline in adult spotted bass WUA. The other scenarios of wet and dry seasons had minimal seasonal differences in percentage of habitat loss for orangethroat darter, but were larger differences in the spotted bass WUA percent decline, although the quantity of WUA was similar.

Reductions in baseline streamflows resulted in a decline in WUA for orangethroat darter and spotted bass in Mill2, which were larger than losses in Mill1 (Table 7; Figure 13a-f). In all scenarios, orangethroat darter lost a larger percentage of habitat than spotted bass and the declines in habitat were consistent throughout the season. A 1% decline in baseline discharge resulted in an average 0.6% decline in orangethroat darter WUA, while reductions were less for juvenile (0.2%) and adult (0.2%) spotted bass. A 20% baseline decline resulted in a 13.2% loss of WUA in orangethroat darter, 4.7% loss in juvenile spotted bass, and a 5.1% decline in adult spotted bass. A 50% decline in baseline streamflow resulted in a 36.4% loss of orangethroat darter WUA, a 14.1% loss in juvenile spotted bass WUA, and a 14.9% loss of adult spotted bass WUA. In the 70% decline scenario, there is a 53.3% decline in orangethroat darter WUA, 21.4% decline in juvenile spotted bass WUA, and a 22.3% decline in adult spotted bass WUA.

Reductions in baseline streamflow resulted in larger declines in WUA for orangethroat darter than juvenile spotted bass in Colvert (Table 8; Figure 14a-d). Both species had small declines in WUA at 1% (0.7% orangethroat darter WUA and 0.1% juvenile spotted bass WUA). There were larger differences between species at 20% (13.8% orangethroat darter WUA and 1.3% juvenile spotted bass WUA), and 50% (36.7% orangethroat darter WUA and 4.0% juvenile spotted bass WUA). The worst-case scenario of 70% reduction had a mean decline of 53.3% in orangethroat darter WUA (Table 8). Juvenile spotted bass had only a small decline at 70% with a mean reduction of 6.0%. There was little change between low and high flow months for orangethroat darter (52.9 to 54.0%) and juvenile spotted bass (5.9 to 6.3%; Table 8).

Reductions in baseline streamflow resulted in lower declines in WUA for orangethroat darter than southern redbelly dace in Springhouse (Table 9; Figure 15a-d). Both species had small declines in WUA at 1% (0.4% orangethroat darter WUA and 0.8% southern redbelly dace WUA). There were larger declines and differences at 20% (9.0% orangethroat darter WUA and 15.7% southern redbelly dace WUA), and 50% (22.2% orangethroat darter WUA and 39.0% southern redbelly dace WUA). The worst-case scenario of 70% reduction had a mean decline of 30.6% in orangethroat darter WUA but higher declines in the low flow months (34.5%; Table 9). Southern redbelly dace had large decline at 70% with a mean reduction of 54.1%, which was the largest decline of any species. There was more variation between low and high flow months for orangethroat darter (28.4 to 34.5%), than southern redbelly dace (52.4 to 56.8%; Table 9).

In general, the darters and dace had larger declines in habitat (from 30 to over 50% with a 70% decline in flows). Spotted bass appear to be more generalists in their habitat preferences and prefer deep-water, pool habitat, so maximum declines were lower with 10% declines in Mill1 and 20% declines in Mill2. In Colvert, spotted bass juveniles only had a 6% decline in habitat with a 70% reduction in flows, but the juveniles had low WUA compared to the total amount of habitat. Fish habitat in springs was more sensitive to altered flows, so monitoring may be needed to conserve the species that occur in those habitats

Study Limitations

Several limitations of this study should be outlined for proper interpretation and application of the results. Because of the short timeline associated with the project, only a single field season was available for data collection. Thus, the fish habitat use included in this study was based on previously collected habitat suitability at different sites. The species that use the springs have not been used in many PHABSIM studies, so we were limited to the data that could be used. Additional life stages would be useful for modeling the Arbuckle-Simpson species (i.e. spawning habitat), but they were not available at the time of our study. A macrohabitat- or ecosystem-scale model that looks at other species, in addition to fish, may provide a better framework for modeling the impacts of altered streamflow on the entire Mill Creek watershed.

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 were working with fish species with no historic population data and also have little published information on their habitat ecology. Although we cannot demonstrate a direct relationship between WUA and population, one of the species (southern redbelly dace) is a spring habitat specialist, which indicates that changes in spring habitat quantity and quality are likely to influence their populations.

The flow record for the Mill Creek watershed is limited to 3 years (5.5 years extended), compared to the suggested 10 years (Armstrong et al. 2008) or 20 years (Poff 1996) that are needed to properly describe flow regime. We simulated flows in Mill Creek based on Pennington Creek, but the dataset is not sufficient to show the longer term climatic patterns that affect Mill Creek and its springs. In addition to the short term of the flow records, existing alteration of streamflow from groundwater and surface water withdrawals associated with gravel mining in the watershed are not addressed directly in this study. The long term effects of these and other impacts (e.g., land use) may affect aquatic health and lead to unexpected outcomes from increased water removal from Mill Creek and its springs. Habitat modeling alone is not sufficient to fully understand the effects of these and future water withdrawal impacts on the Mill Creek ecosystem.

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. The southern redbelly dace occurs at the southern end of their distribution in the spring habitats, which provide a temperature refuge and excellent water quality. The southern redbelly dace is also interesting because it appears to occupy only springs in the lower Mill Creek watershed. Movement upstream may be limited due to water conditions in the main channel. This species is also isolated from Spring Creek populations by the Washita River, and from the Blue River populations by Lake Texoma. Local extinction of the Mill Creek populations would not likely be repopulated through migrations from adjacent streams.

Critical flows (where maximum WUA occurred) for the stream sites were higher for the orangethroat darter (20-25 cfs) than juvenile spotted bass (10-12.5 cfs) and adult spotted bass (15-17.5 cfs). In the spring sites, orangethroat darter (5.2-5.8 cfs), southern redbelly dace (5.2 cfs), and juvenile spotted bass (5.6 cfs) had similar critical flows. This study found that reductions in streamflow in the Mill Creek would reduce habitat by as much as 53% for orangethroat darter, 21% for the juvenile spotted bass, and 22% for the adult spotted bass. Loss of habitat in the spring sites was more pronounced in the orangethroat darter (53.3%) and southern redbelly dace (54.1%), than the juvenile spotted bass (6%). 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.

Monitoring the spring flows throughout the year may be useful in developing management plans for 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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Works Cited

Armour, C.L. & Taylor, J.G. 1991. Evaluation of the instream flow incremental methodology by united-states fish and wildlife service field users. Fisheries 16: 36-43.

Armstrong, D.S., Parker, G.W. & Richards, T.A. 2008. Characteristics and classification of least altered streamflows in Massachusetts. Reston, Va.: U.S. Dept. of the Interior, U.S. Geological Survey. viii, 113 p. pp.

Binderim, G.E. 1977. Fishes of Mill Creek, a tributary of the Washita River, Johnston and Murray Counties, Oklahoma. Proceedings of the Oklahoma Academy of Science 57: 1-11.

Bovee, K.D. 1982. A guide to stream habitat analysis using the instream flow incremental methodology. Washington, D.C.: U.S. Fish and Wildlife Service, Instream Flow Information Paper 12, FWS/OBS-82/26.

Bovee, K.D. 1986. Development and evaluation of habitat suitability criteria for use in the instream flow incremental methodology. Washington, D.C.: U.S. Fish and Wildlife Service, Instream Flow Information Paper 21, Biological Report 86(7).

Bovee, K.D. 1994. Data collection procedures for the physical habitat simulation system. Fort Collins, CO: National Biological Service, Midcontinent Ecological Science Center. 159 pp. pp.

Bunn, S.E. & Arthington, A.H. 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. Environmental Management 30: 492-507.

Eamus, D. & Froend, R. 2006. Groundwater-dependent ecosystems: the where, what and why of GDEs. Australian Journal of Botany 54: 91-96.

Fitzhugh, T.W. & Richter, B.D. 2004. Quenching urban thirst: growing cities and their impacts on freshwater ecosystems. Bioscience 54: 741-754.

Glazier, D.S. 1991. The fauna of North American temperate cold springs - patterns and hypotheses. Freshwater Biology 26: 527-542.

Hamilton, J.M., Johnson, S., Esquilin, R., Thompson, E.L., Luevano, G., Wiatrek, A., Mireles, J., Gloyd, R., Sterzenback, J. & Hoyt, J.R. 2003. Edwards aquifer authority hydrogeologic data report for 2002. San Antonio, Texas: Edwards Aquifer Authority.

Leopold, L.B. 1994. A view of the river. Cambridge, Massachusetts: Harvard University Press.

Matthews, W.J., Hoover, J.J. & Milstead, W.B. 1985. Fishes of Oklahoma springs. Southwestern Naturalist 30: 23-32.

McMahon, T.E., Gebhart, G., Maughan, O.E. & Nelson, P.C. 1984. Habitat suitability index models and instream flow suitability curves: spotted bass. U.S. Fish Wildl. Servo FWS/OBS-82/10.72. 41 pp.

Meyer, J.L., Strayer, D.L., Wallace, J.B., Eggert, S.L., Helfman, G.S. & Leonard, N.E. 2007. The contribution of headwater streams to biodiversity in river networks. Journal of the American Water Resources Association 43: 86-103.

Miller, R.J. & Robison, H.W. 2004. Fishes of Oklahoma. Norman, Oklahoma: University of Oklahoma Press.

Nehring, R.B. & Anderson, R.M. 1993. Determination of population-limiting critical salmonid habitats in Colorado streams using the Physical Habitat Simulation System. Rivers 4: 1-19.

ODWC. 2005. Oklahoma comprehensive wildlife conservation strategy. [Oklahoma City, Okla.]: Oklahoma Dept. of Wildlife Conservation. x, 421 p. pp.

Poff, N.L. 1996. A hydrogeography of unregulated streams in the United States and an examination of scale-dependence in some hydrological descriptors. Freshwater Biology 36: 71-91.

Poff, N.L. 1997. Landscape filters and species traits: Towards mechanistic understanding and prediction in stream ecology. Journal of the North American Benthological Society 16: 391-409.

Poff, N.L., Allan, J.D., Bain, M.B., Karr, J.R., Prestegaard, K.L., Richter, B.D., Sparks, R.E. & Stromberg, J.C. 1997. The natural flow regime. Bioscience 47: 769-784.

Postel, S.L. & Richter, B.D. 2003. Rivers for life: managing water for people and nature. Washington, D.C.: Island Press.

Reiser, D.W., Wesche, T.A. & Estes, C. 1989. Status of instream flow legislation and practices in North America. Fisheries 14: 22-29.

Richter, B.D., Mathews, R. & Wigington, R. 2003. Ecologically sustainable water management: Managing river flows for ecological integrity. Ecological Applications 13: 206-224.

Richter, B.D., Warner, A.T., Meyer, J.L. & Lutz, K. 2006. A collaborative and adaptive process for developing environmental flow recommendations. River Research and Applications 22: 297-318.

Robison, H.W. & Buchanan, T.M. 1988. The fishes of Arkansas. Fayetteville, Arkansas: The University of Arkansas Press.

Rosenberg, D.M., McCully, P. & Pringle, C.M. 2000. Global-scale environment effects of hydrological alterations: introduction. Bioscience 50: 746-751.

Rosgen, D.L. 1996. Applied river morphology. Pagosa Springs, Colorado: Wildland Hydrology.

Schofield, W. 1993. Engineering surveying: theory and examination problems for students. Oxford, England: Butterworth-Heinemann Ltd.

Seilheimer, T.S. & Fisher, W.L. 2008. Instream flow assessment of streams draining the Arbuckle-Simpson aquifer. Submitted to the Oklahoma Water Resources Board. 49 pp.

Sophocleous, M. 2007. The science and practice of environmental flows and the role of hydrogeologists. Ground Water 45: 393-401.

Sparks, R.E. 1995. Need for Ecosystem Management of Large Rivers and Their Floodplains. Bioscience 45: 168-182.

Stalnaker, C.B., Lamb, B.L., Henriksen, J., Bovee, K.D. & Bartholow, J.M. 1995. The instream flow incremental methodology: a primer for IFIM. Washington, D.C.: National Biological Service, Biological Report 29.

Tejan, E.C. & Haase, C.S. 2008. Indicators of Hydrologic Alteration (IHA) analysis of selected streams of the Arbuckle-Simpson aquifer, south central Oklahoma. Submitted to the Oklahoma Water Resource Board. 113 pp.

Thorne, C.R., Hey, R.D. & Newson, M.D. 1997. Applied fluvial geomorphology for river engineering and management. West Sussex, England: John Wiley & Sons, Ltd.

Toepfer, C.S., Fisher, W.L. & Warde, W.D. 2000. A multi-stage approach to estimate fish abundance in streams in streams using geographic information systems. North American Journal of Fisheries Management 20: 634-645.

Waddle, T.J. 2001. PHABSIM for Windows: user's manual and exercises. Fort Collins, Colorado: U.S. Geological Survey. 288 pp.

Table 1: Site and physical characteristics describing the four study sites in the Mill Creek watershed.

	Mill1	Mill2	Colvert	Springhouse
Site Characteristics				
Site Length (ft)	1801.5	231.8	232.2	661.2
Transects (N)	19	7	11	12
Observed Discharge (N)	2	2	2	1
Simulated Discharge (N)	12	7	5	5
Physical Characteristics (Transect)				
Mean Wetted Width (ft)				
Low Flow	38.3	30.9	12.8	-
High Flow	43.0	34.7	14.2	8.2
Velocity (ft/s)				
Low Flow				
Mean	0.1	0.0	0.3	-
Range	-0.7 - 2.8	-0.1 - 1.2	-0.1 - 1.4	-
High Flow				
Mean	1.1	1.5	0.7	0.6
Range	-0.4 - 7.5	-0.2 - 3.5	-0.2 - 2.4	-0.2 - 2.9
Depth (ft)				
Low Flow				
Mean	1.0	0.6	0.5	-
Range	0.0 - 3.2	0.0 - 1.3	0.0 - 2.2	-
High Flow				
Mean	1.7	1.5	0.5	0.9
Range	0.0 - 4.1	0.3 - 2.6	0.0 - 1.7	0.1 - 2.6

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

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

	Frequency of Occurrence			
	Mill1	Mill2	Colvert	Springhouse
Substrate Type				
1	0.07	0.01	**0.12**	**0.22**
2	0.03	0.01	0.01	0.08
3	**0.15**	0.04	0.03	**0.46**
4	**0.26**		**0.18**	0.07
5	0.02		0.01	
6	**0.24**	**0.94**	**0.37**	0.02
7	0.01		0.05	
8	0.01		**0.16**	0.05
9	**0.21**		0.07	**0.10**
Cover Type				
0.0	**0.95**	**0.96**	**0.40**	**0.97**
0.1		0.04	**0.29**	
0.2				
0.3	0.04		0.05	0.02
0.4				
0.5			**0.18**	
0.6	0.01		0.08	0.01
N=	422	81	124	87

Table 4: Regression equation coefficients of WUA-discharge models for Mill1: A) orangethroat darter, B) spotted bass juvenile, and C) spotted bass adult; and Mill2: D) orangethroat darter, E) spotted bass juvenile, and F) spotted bass adult.

A)

Discharge (cfs)		Coefficients		
Interval (O)	Intercept (A)	B	C	D
0.1	862.73	1821.06	0.00	-292.45
0.5	1572.44	1680.68	-350.94	66.08
1.3	2726.22	1246.06	-192.34	15.36
5.0	5481.42	453.49	-21.87	0.62
10.0	7280.15	281.62	-12.50	0.14
20.0	8980.96	72.05	-8.45	0.25
30.0	9110.10	-20.84	-0.84	-0.08
40.0	8732.95	-63.08	-3.39	0.08
50.0	7841.01	-107.57	-1.06	0.05
65.6	6085.06	-105.95	1.17	-0.16
70.0	5627.94	-104.92	-0.93	0.05
80.0	4532.00	-109.62	0.46	0.05
90.0	3534.04	-84.78	2.02	-0.07
100.0	2820.98	0.00	0.00	0.00

B)

Discharge (cfs)		Coefficients		
Interval (O)	Intercept (A)	B	C	D
0.1	4020.80	6452.89	0.00	-4396.60
0.5	6320.57	4342.52	-5275.92	2210.57
1.3	7549.81	145.36	29.46	-3.00
5.0	8338.89	240.09	-3.85	-1.14
10.0	9299.88	115.69	-21.03	0.79
20.0	9146.06	-67.27	2.73	-0.14
30.0	8604.99	-55.09	-1.51	0.07
40.0	7972.00	-64.58	0.56	-0.01
50.0	7373.00	-56.17	0.28	0.00
65.6	6565.00	-47.39	0.28	0.01
70.0	6363.00	-44.19	0.44	0.00
80.0	5963.00	-36.06	0.37	-0.01
90.0	5630.00	-31.48	0.09	0.00
100.0	5321.00	0.00	0.00	0.00

C)

Discharge (cfs)		Coefficients		
Interval (O)	Intercept (A)	B	C	D
0.1	3917.15	7253.28	0.00	-5338.37
0.5	6476.81	4690.87	-6406.05	2767.99
1.3	7546.84	-244.26	237.13	-26.80
5.0	8532.09	409.98	-60.31	3.52
10.0	9514.10	70.81	-7.52	0.20
20.0	9675.01	-18.13	-1.37	0.03
30.0	9388.00	-36.12	-0.43	0.02
40.0	9001.00	-39.58	0.08	-0.01
50.0	8606.00	-40.14	-0.14	0.01
65.6	7971.00	-39.70	0.17	-0.01
70.0	7799.00	-38.61	0.08	0.00
80.0	7424.00	-36.09	0.17	0.00
90.0	7078.00	-33.32	0.11	0.00
100.0	6752.00	0.00	0.00	0.00

D)

Discharge (cfs)		Coefficients		
Interval (O)	Intercept (A)	B	C	D
0.5	2016.21	1684.57	0.00	-27.35
2.9	5681.08	1211.95	-196.93	22.39
5.0	7565.02	681.01	-55.90	2.30
10.0	9860.57	294.80	-21.34	0.40
25.0	10846.11	-72.32	-3.13	0.04
50.0	7760.01	-147.44	0.13	0.02
75.0	4423.00	-108.71	1.42	0.00
86.1	3397.01	-75.68	1.55	-0.04
100.0	2544.99	0.00	0.00	0.00

E)

Discharge (cfs)		Coefficients		
Interval (O)	Intercept (A)	B	C	D
0.5	2055.94	366.37	0.00	-9.78
2.9	2799.98	197.31	-70.44	10.30
5.0	2999.03	37.68	-5.57	0.18
10.0	3071.03	-4.29	-2.82	0.07
25.0	2617.02	-39.95	0.44	0.00
50.0	1824.00	-26.28	0.11	0.00
75.0	1242.00	-19.92	0.15	0.00
86.1	1045.00	-15.06	0.29	-0.01
100.0	873.00	0.00	0.00	0.00

F)

Discharge (cfs)		Coefficients		
Interval (O)	Intercept (A)	B	C	D
0.5	2056.69	377.92	0.00	-8.98
2.9	2839.57	222.76	-64.65	9.11
5.0	3106.64	71.78	-7.24	0.32
10.0	3324.08	23.09	-2.50	0.04
25.0	3253.01	-22.86	-0.56	0.01
50.0	2548.01	-24.85	0.48	-0.01
75.0	2115.00	-14.31	-0.06	0.00
86.1	1954.00	-14.23	0.07	0.00
100.0	1765.00	0.00	0.00	0.00

Table 5: A Colvert MIPU JUV, B-C Springhouse Regression equation coefficients of WUA-discharge models for Colvert: A) spotted bass juvenile; and Springhouse: B) orangethroat darter and C) southern redbelly dace.

A)

Discharge (cfs)		Coefficients		
Interval (O)	Intercept (A)	B	C	D
0.5	911.28	68.48	0.00	-13.56
1.1	949.44	53.83	-24.41	1.91
2.0	979.50	14.52	-19.27	8.16
2.8	982.96	-0.64	0.32	9.03
3.8	991.66	27.07	27.39	1.81
4.5	1024.65	68.08	31.19	-10.40
5.5	1113.52	0.00	0.00	0.00

B)

Discharge (cfs)		Coefficients		
Interval (O)	Intercept (A)	B	C	D
1.0	491.86	226.34	0.00	-13.95
2.0	704.25	184.48	-41.86	1.30
3.0	848.17	104.66	-37.97	6.57
4.2	930.44	41.92	-14.31	-5.91
5.0	951.79	7.68	-28.49	9.50
6.0	940.49	0.00	0.00	0.00

C)

Discharge (cfs)		Coefficients		
Interval (O)	Intercept (A)	B	C	D
1.0	484.59	385.28	0.00	-15.20
2.0	854.68	339.69	-45.59	-12.42
3.0	1136.35	211.24	-82.86	11.13
4.2	1289.75	60.44	-42.80	7.88
5.0	1314.74	7.08	-23.89	7.96
6.0	1305.89	0.00	0.00	0.00

Table 6: Quantity and percent change (%) in weighted usable area (WUA; ft^2/1000ft) for orangethroat darter, spotted bass juvenile, and spotted bass adult in relation to incremental reductions in baseline streamflow (Q) of Mill1.

		Discharge	Orangethroat Darter		Spotted Bass Juv.		Spotted Bass Ad.	
Scenario	Flow	(cfs)	WUA (ft^2)	%	WUA (ft^2)	%	WUA (ft^2)	%
Baseline	Low	3.96	4939.6	0.0	8089.6	0.0	9190.9	0.0
	Annual Average	4.98	5375.9	0.0	8321.8	0.0	8077.9	0.0
	High	7.74	6453.6	0.0	8875.9	0.0	8404.1	0.0
Baseline-1%	Low	3.92	4917.6	-0.4	8080.3	-0.1	9177.4	-0.2
	Annual Average	4.93	5352.5	-0.4	8310.9	-0.1	8061.2	-0.2
	High	7.66	6427.0	-0.4	8861.4	-0.2	8389.0	-0.1
Baseline-5%	Low	3.76	4827.5	-2.3	8043.1	-0.6	9119.8	-1.0
	Annual Average	4.74	5256.7	-2.2	8267.3	-0.7	7995.0	-0.9
	High	7.35	6319.0	-2.1	8801.8	-0.8	8327.5	-0.8
Baseline-10%	Low	3.56	4709.7	-4.7	7997.1	-1.1	9038.6	-2.0
	Annual Average	4.49	5132.0	-4.5	8212.2	-1.3	7914.2	-1.8
	High	6.96	6179.5	-4.2	8724.3	-1.7	8249.1	-1.7
Baseline-20%	Low	3.16	4454.2	-9.8	7907.0	-2.3	8841.1	-3.9
	Annual Average	3.99	4864.3	-9.5	8100.7	-2.7	7763.8	-3.7
	High	6.19	5884.8	-8.8	8560.9	-3.5	8089.3	-3.8
Baseline-30%	Low	2.77	4166.8	-15.6	7820.5	-3.3	8593.3	-5.5
	Annual Average	3.49	4568.0	-15.0	7989.1	-4.0	7637.5	-5.7
	High	5.42	5565.9	-13.8	8388.8	-5.5	7929.2	-6.5
Baseline-40%	Low	2.37	3841.3	-22.2	7738.9	-4.3	8298.9	-6.6
	Annual Average	2.99	4236.8	-21.2	7879.3	-5.3	7545.9	-7.5
	High	4.64	5215.0	-19.2	8211.9	-7.5	7777.0	-9.7
Baseline-50%	Low	1.98	3471.8	-29.7	7663.2	-5.3	7979.2	-7.2
	Annual Average	2.49	3860.4	-28.2	7773.6	-6.6	7499.6	-9.0
	High	3.87	4815.5	-25.4	8035.0	-9.5	7648.5	-13.2
Baseline-60%	Low	1.58	3052.0	-38.2	7594.8	-6.1	7695.8	-7.0
	Annual Average	1.99	3423.7	-36.3	7674.8	-7.8	7509.5	-9.9
	High	3.09	4331.4	-32.9	7865.4	-11.4	7569.6	-16.3
Baseline-70%	Low	1.19	2575.1	-47.9	7504.5	-7.2	7518.6	-6.6
	Annual Average	1.50	2909.6	-45.9	7557.9	-9.2	7547.9	-10.4
	High	2.32	3722.7	-42.3	7711.0	-13.1	7533.1	-18.2

Table 7: Quantity and percent change (%) in weighted usable area (WUA; ft^2/1000ft) for orangethroat darter, spotted bass juvenile, and spotted bass adult in relation to incremental reductions in baseline streamflow (Q) of Mill2.

		Discharge	Orangethroat Darter		Spotted Bass Juv.		Spotted Bass Ad.	
Scenario	Flow	(cfs)	WUA (ft^2)	%	WUA (ft^2)	%	WUA (ft^2)	%
Baseline	Low	2.90	7843.2	0.0	3012.2	0.0	3134.7	0.0
	Annual Average	3.66	5647.2	0.0	2787.1	0.0	2827.9	0.0
	High	5.68	6276.6	0.0	2847.4	0.0	2912.7	0.0
Baseline-1%	Low	2.87	7808.7	-0.4	3010.3	-0.1	3131.0	-0.1
	Annual Average	3.62	5612.7	-0.6	2781.6	-0.2	2821.6	-0.2
	High	5.62	6242.3	-0.5	2843.0	-0.2	2907.2	-0.2
Baseline-5%	Low	2.76	7666.9	-2.2	3002.1	-0.3	3115.8	-0.6
	Annual Average	3.48	5471.6	-3.1	2758.5	-1.0	2795.6	-1.1
	High	5.39	6102.1	-2.8	2824.6	-0.8	2884.7	-1.0
Baseline-10%	Low	2.61	7481.0	-4.6	2990.5	-0.7	3095.1	-1.3
	Annual Average	3.29	5288.8	-6.3	2727.4	-2.1	2761.0	-2.4
	High	5.11	5919.9	-5.7	2799.7	-1.7	2854.6	-2.0
Baseline-20%	Low	2.32	7077.0	-9.8	2961.3	-1.7	3047.6	-2.8
	Annual Average	2.93	4901.3	-13.2	2657.4	-4.7	2684.6	-5.1
	High	4.54	5531.1	-11.9	2742.9	-3.7	2787.9	-4.3
Baseline-30%	Low	2.03	6618.1	-15.6	2919.2	-3.1	2987.1	-4.7
	Annual Average	2.56	4486.6	-20.6	2577.6	-7.5	2599.3	-8.1
	High	3.98	5107.2	-18.6	2675.7	-6.0	2711.2	-6.9
Baseline-40%	Low	1.74	6082.6	-22.4	2854.0	-5.2	2905.0	-7.3
	Annual Average	2.20	4048.7	-28.3	2489.5	-10.7	2506.4	-11.4
	High	3.41	4643.9	-26.0	2595.6	-8.8	2622.6	-10.0
Baseline-50%	Low	1.45	5451.9	-30.5	2757.1	-8.5	2793.3	-10.9
	Annual Average	1.83	3592.0	-36.4	2394.8	-14.1	2407.3	-14.9
	High	2.84	4137.8	-34.1	2500.9	-12.2	2520.4	-13.5
Baseline-60%	Low	1.16	4716.4	-39.9	2623.2	-12.9	2647.6	-15.5
	Annual Average	1.46	3120.8	-44.7	2294.8	-17.7	2303.5	-18.5
	High	2.27	3589.6	-42.8	2391.2	-16.0	2404.5	-17.4
Baseline-70%	Low	0.87	3893.0	-50.4	2457.8	-18.4	2473.1	-21.1
	Annual Average	1.10	2639.4	-53.3	2191.2	-21.4	2196.3	-22.3
	High	1.70	3008.0	-52.1	2269.6	-20.3	2277.7	-21.8

Table 8: Quantity and percent change (%) in weighted usable area (WUA; ft^2/1000ft) for orangethroat darter and spotted bass juvenile in relation to incremental reductions in baseline streamflow (Q) of Colvert.

		Discharge	Orangethroat Darter		Spotted Bass Juv.	
Scenario	Flow	(cfs)	WUA (ft^2)	%	WUA (ft^2)	%
Baseline	Low	1.37	585.6	0.0	962.2	0.0
	Annual Average	1.49	617.3	0.0	966.0	0.0
	High	1.75	681.7	0.0	973.9	0.0
Baseline-1%	Low	1.36	581.7	-0.7	961.6	-0.1
	Annual Average	1.47	613.2	-0.7	965.5	-0.1
	High	1.74	677.2	-0.7	973.4	0.0
Baseline-5%	Low	1.30	565.8	-3.4	959.3	-0.3
	Annual Average	1.41	596.6	-3.4	963.3	-0.3
	High	1.67	659.1	-3.3	971.5	-0.2
Baseline-10%	Low	1.23	545.7	-6.8	956.1	-0.6
	Annual Average	1.34	575.5	-6.8	960.3	-0.6
	High	1.58	635.9	-6.7	968.8	-0.5
Baseline-20%	Low	1.10	504.3	-13.9	949.2	-1.4
	Annual Average	1.19	531.9	-13.8	953.6	-1.3
	High	1.40	587.9	-13.8	962.5	-1.2
Baseline-30%	Low	0.96	461.5	-21.2	941.4	-2.2
	Annual Average	1.04	486.6	-21.2	945.8	-2.1
	High	1.23	537.6	-21.1	954.8	-2.0
Baseline-40%	Low	0.82	417.2	-28.8	932.9	-3.0
	Annual Average	0.89	439.6	-28.8	937.1	-3.0
	High	1.05	484.9	-28.9	945.7	-2.9
Baseline-50%	Low	0.68	371.5	-36.6	923.9	-4.0
	Annual Average	0.74	390.9	-36.7	927.7	-4.0
	High	0.88	430.0	-36.9	935.4	-4.0
Baseline-60%	Low	0.55	324.4	-44.6	914.6	-4.9
	Annual Average	0.59	340.4	-44.9	917.7	-5.0
	High	0.70	372.9	-45.3	924.1	-5.1
Baseline-70%	Low	0.41	275.8	-52.9	905.2	-5.9
	Annual Average	0.45	288.2	-53.3	907.6	-6.0
	High	0.53	313.4	-54.0	912.4	-6.3

Table 9: Quantity and percent change (%) in weighted usable area (WUA; ft^2/1000ft) for orangethroat darter and southern redbelly dace in relation to incremental reductions in baseline streamflow (Q) of Springhouse.

		Discharge	Orangethroat Darter		Southern Redbelly Dace	
Scenario	Flow	(cfs)	WUA (ft^2)	%	WUA (ft^2)	%
Baseline	Low	0.86	460.1	0.0	430.5	0.0
	Annual Average	0.97	484.7	0.0	472.3	0.0
	High	1.22	534.1	0.0	556.5	0.0
Baseline-1%	Low	0.85	458.2	-0.4	427.2	-0.8
	Annual Average	0.96	482.5	-0.4	468.6	-0.8
	High	1.21	531.4	-0.5	552.0	-0.8
Baseline-5%	Low	0.82	450.4	-2.1	414.0	-3.8
	Annual Average	0.92	473.8	-2.2	453.7	-3.9
	High	1.16	520.7	-2.5	533.7	-4.1
Baseline-10%	Low	0.77	440.8	-4.2	397.5	-7.7
	Annual Average	0.87	462.9	-4.5	435.2	-7.9
	High	1.10	507.3	-5.0	510.9	-8.2
Baseline-20%	Low	0.69	421.6	-8.4	364.7	-15.3
	Annual Average	0.77	441.1	-9.0	398.1	-15.7
	High	0.97	480.5	-10.0	465.2	-16.4
Baseline-30%	Low	0.60	402.6	-12.5	332.1	-22.9
	Annual Average	0.68	419.6	-13.4	361.1	-23.5
	High	0.85	453.7	-15.1	419.5	-24.6
Baseline-40%	Low	0.52	383.8	-16.6	299.7	-30.4
	Annual Average	0.58	398.2	-17.8	324.4	-31.3
	High	0.73	427.1	-20.0	374.1	-32.8
Baseline-50%	Low	0.43	365.4	-20.6	267.7	-37.8
	Annual Average	0.48	377.1	-22.2	288.1	-39.0
	High	0.61	400.8	-25.0	329.0	-40.9
Baseline-60%	Low	0.34	347.3	-24.5	236.1	-45.2
	Annual Average	0.39	356.5	-26.4	252.1	-46.6
	High	0.49	375.0	-29.8	284.5	-48.9
Baseline-70%	Low	0.26	329.6	-28.4	204.9	-52.4
	Annual Average	0.29	336.3	-30.6	216.7	-54.1
	High	0.37	349.9	-34.5	240.6	-56.8

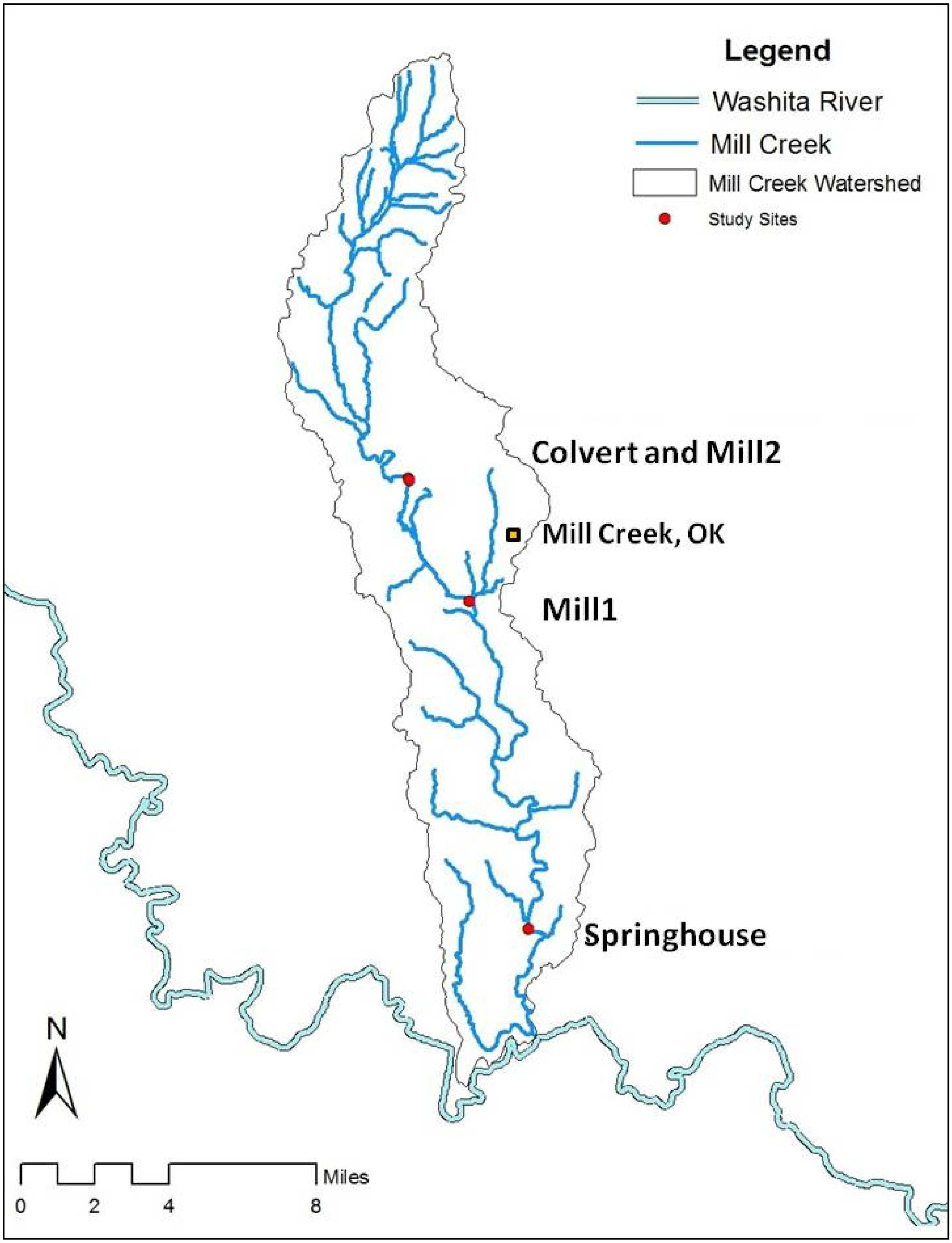

Figure 1: Map of the Mill Creek watershed with the location of the study sites.

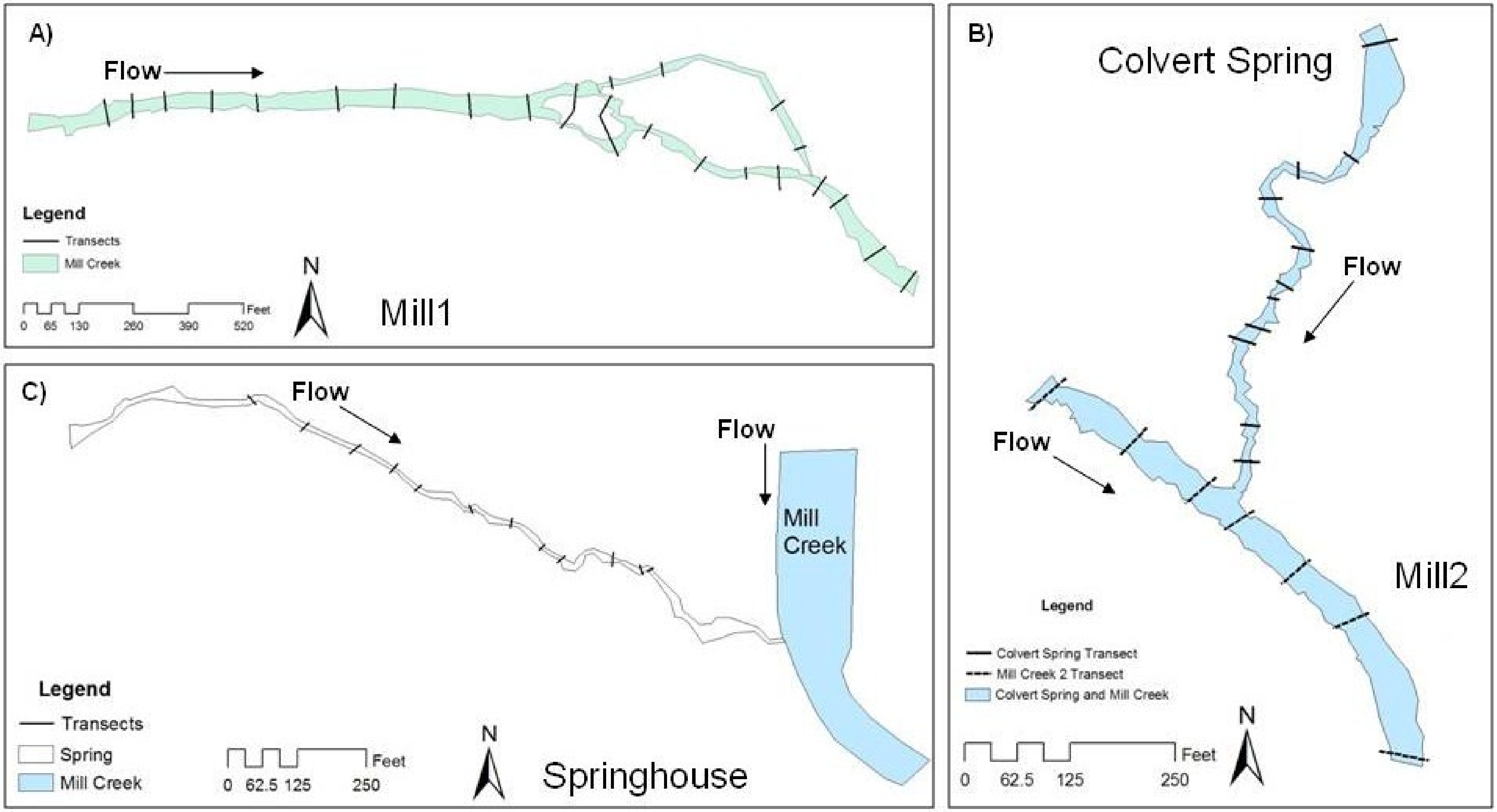

Figure 2: Stream channel and transect location in A) Mill1, B) Mill2 and Colvert Spring, and C) Springhouse spring.

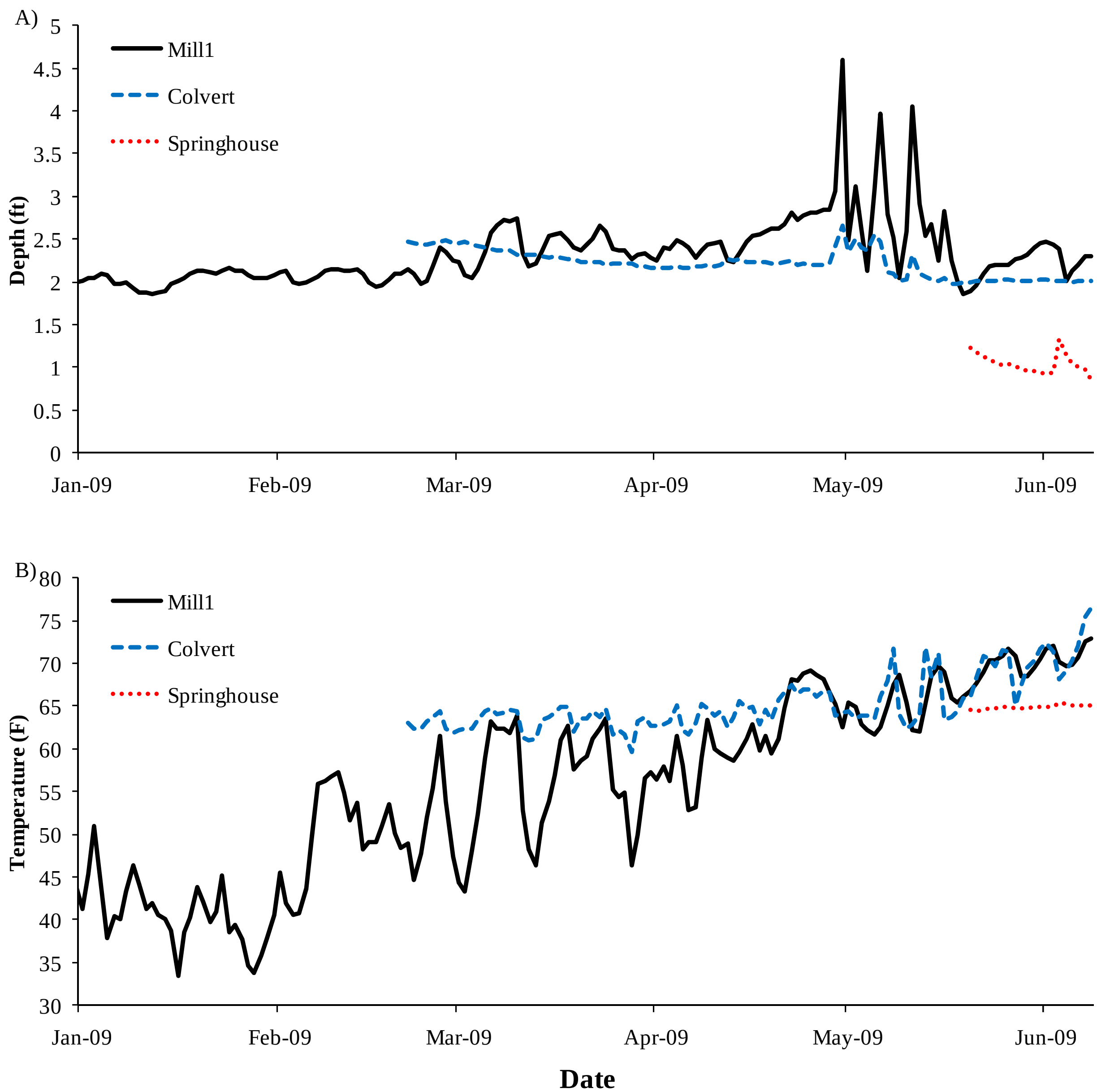

Figure 3: Mean daily A) depth and B) water temperature in Mill1 (solid line; January 1, 2009 to June 8, 2009), Colvert Spring (dash line; February 21, 2009 to June 8, 2009), and Springhouse (dotted line; May 20, 2009 to June 8, 2009).

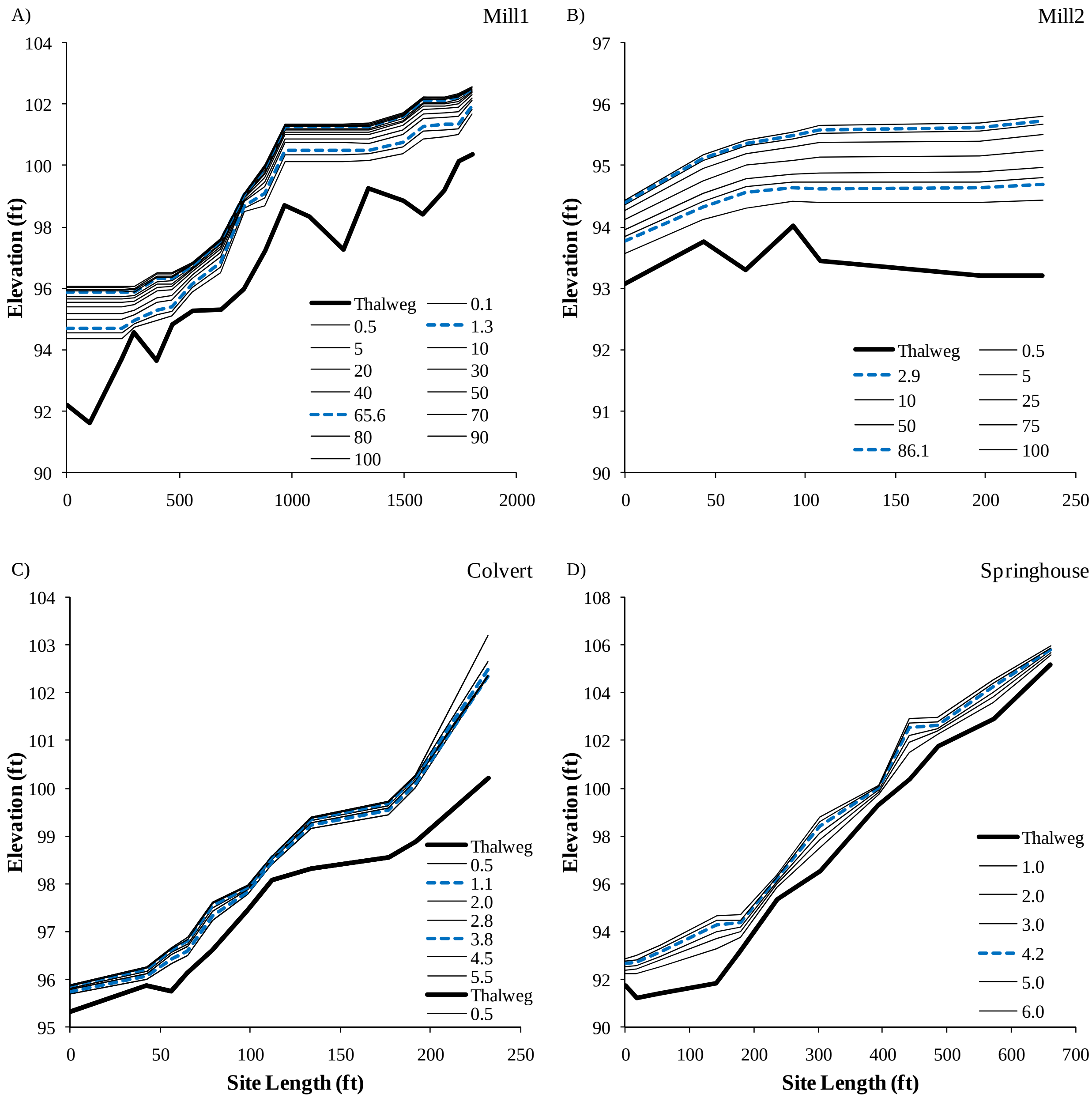

Figure 4: Water surface elevation (feet) for simulated (solid line) and observed (dash line) discharges with thalweg elevation (feet) at A) Mill1, B) Mill2, C) Colvert, and D) Springhouse.

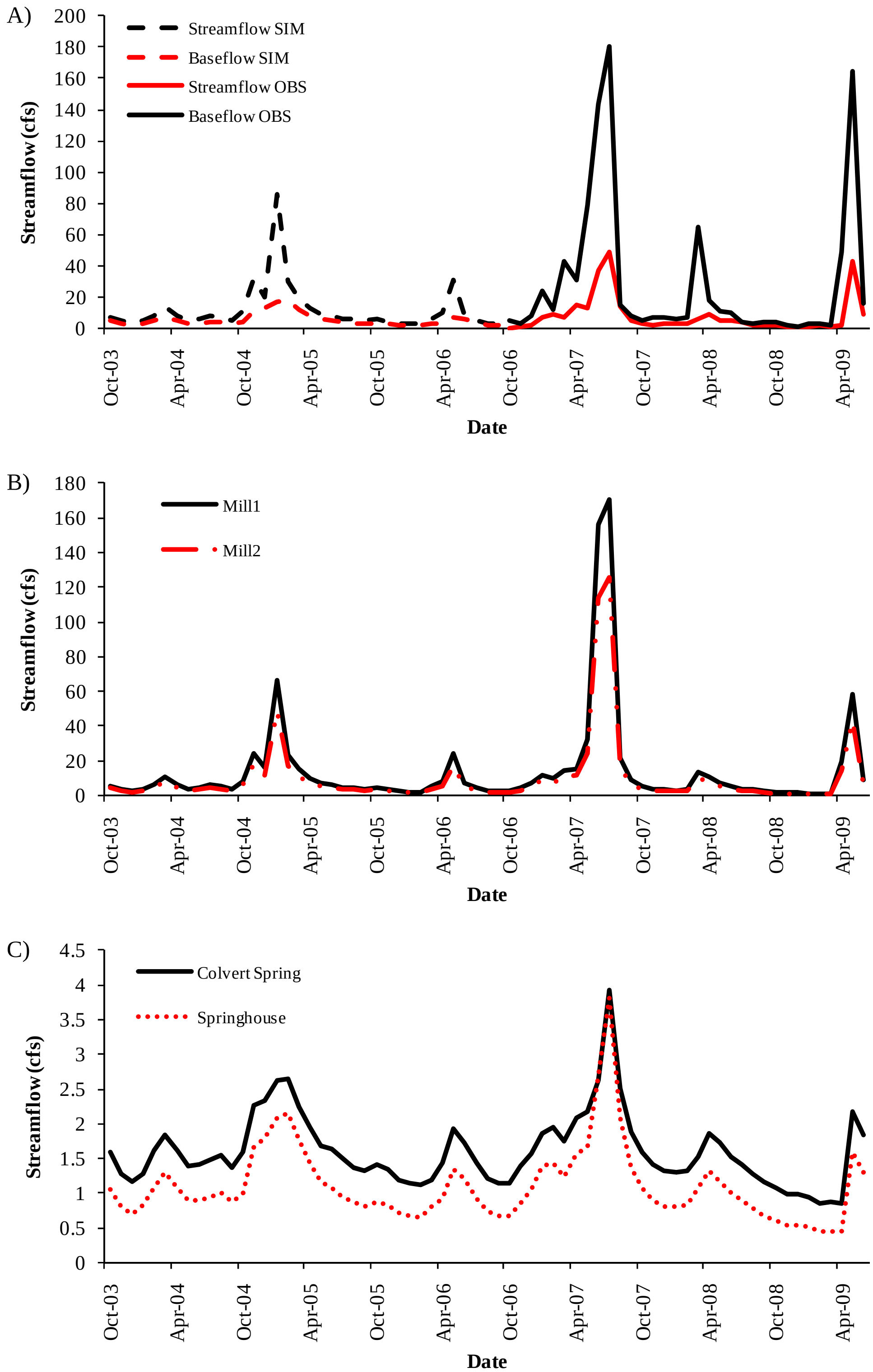

Figure 5: Streamflow and baseflow for A) the Mill Creek USGS gage (dashed simulated from USGS Pennington near Reagan and solid indicate observed Mill Creek gage data). Simulated streamflow based on Mill Creek gage for B) Mill1 and Mill2, and C) Colvert and Springhouse.

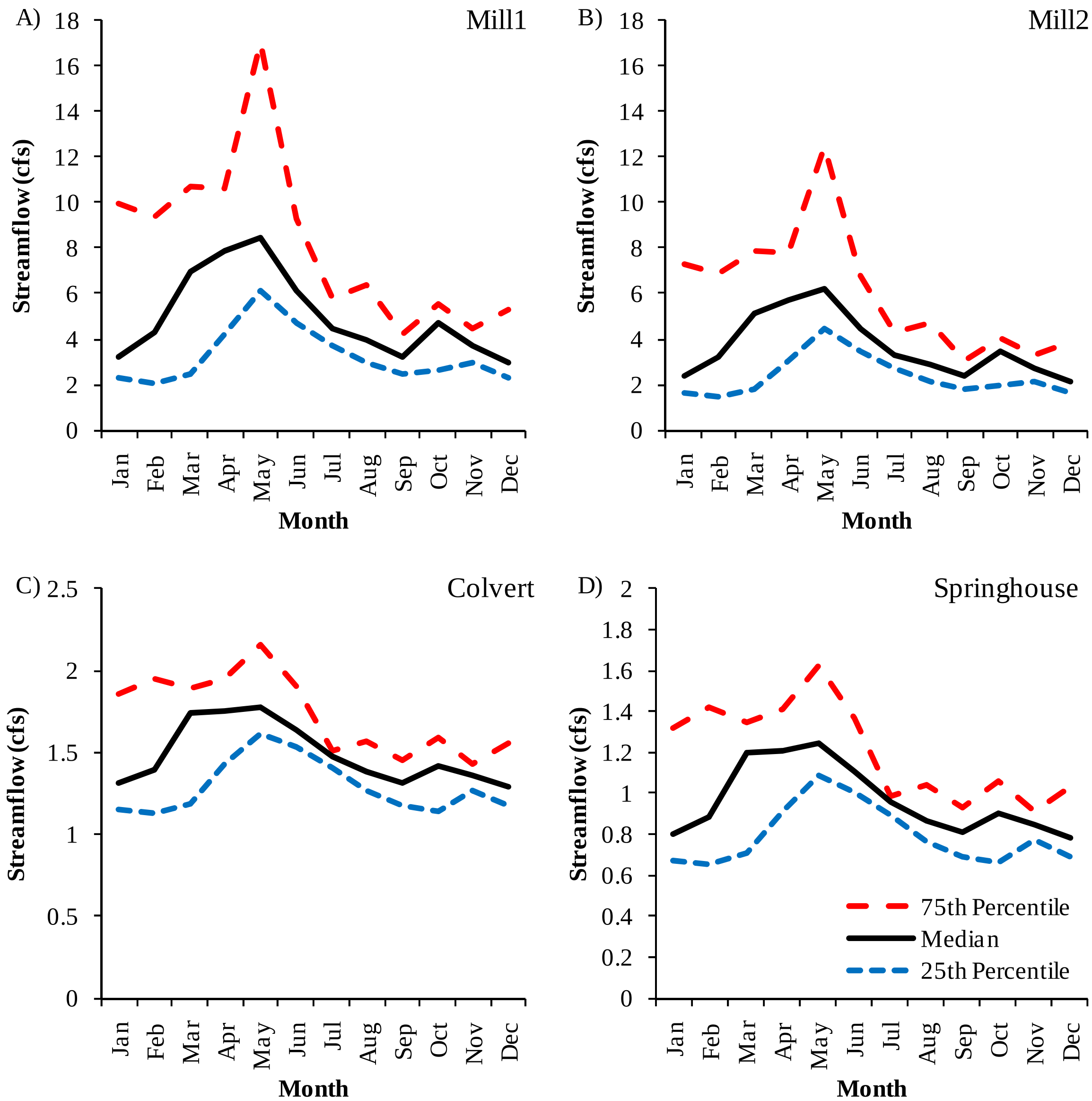

Figure 6: Monthly median (black solid line), 25th percentile (blue short dashed line), and 75th percentile (red long dashed line) discharge in A) Mill1, B) Mill2, C) Colvert, and D) Springhouse.

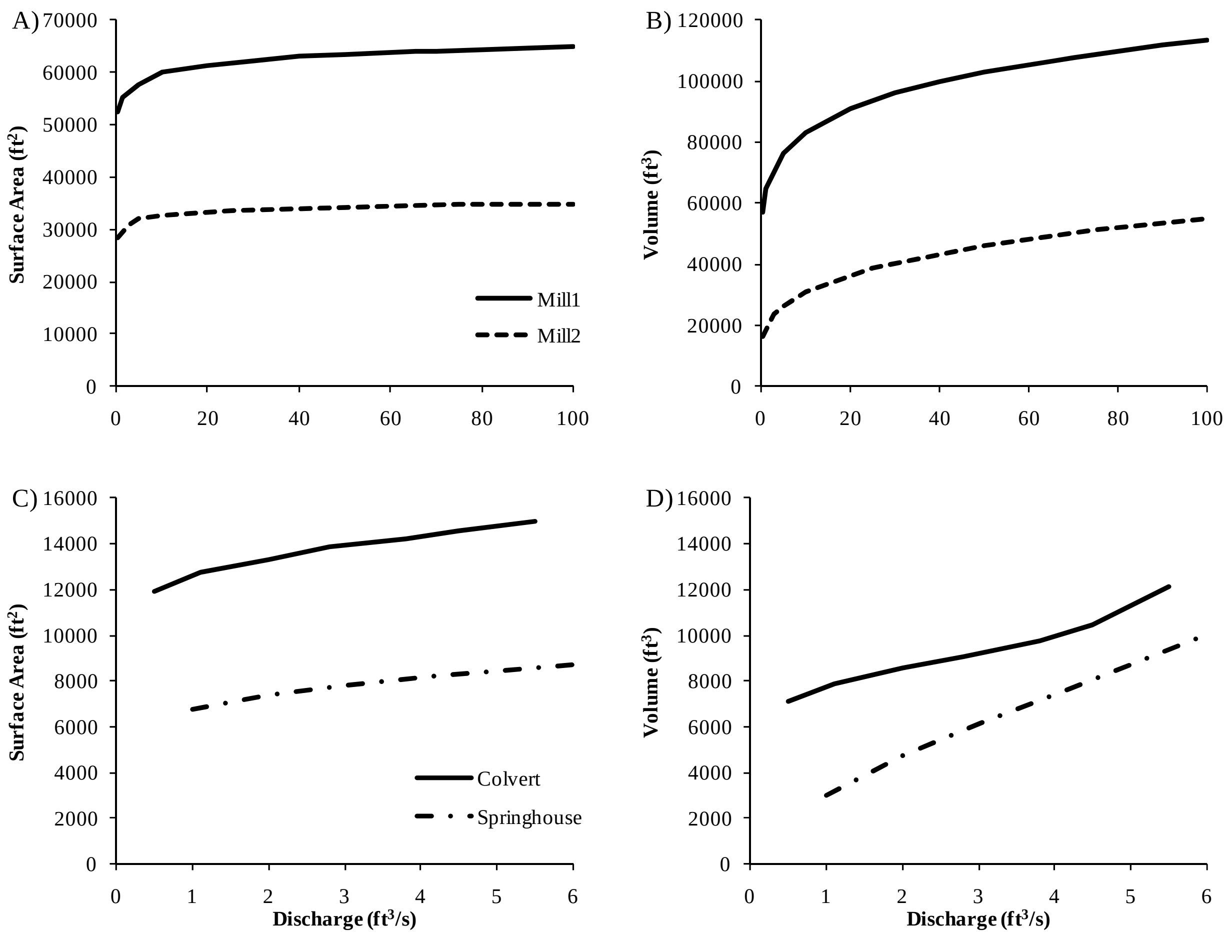

Figure 7: Total available habitat for Mill1 and Mill2 A) surface area and B) volume, and Colvert and Springhouse C) surface area and D) volume.

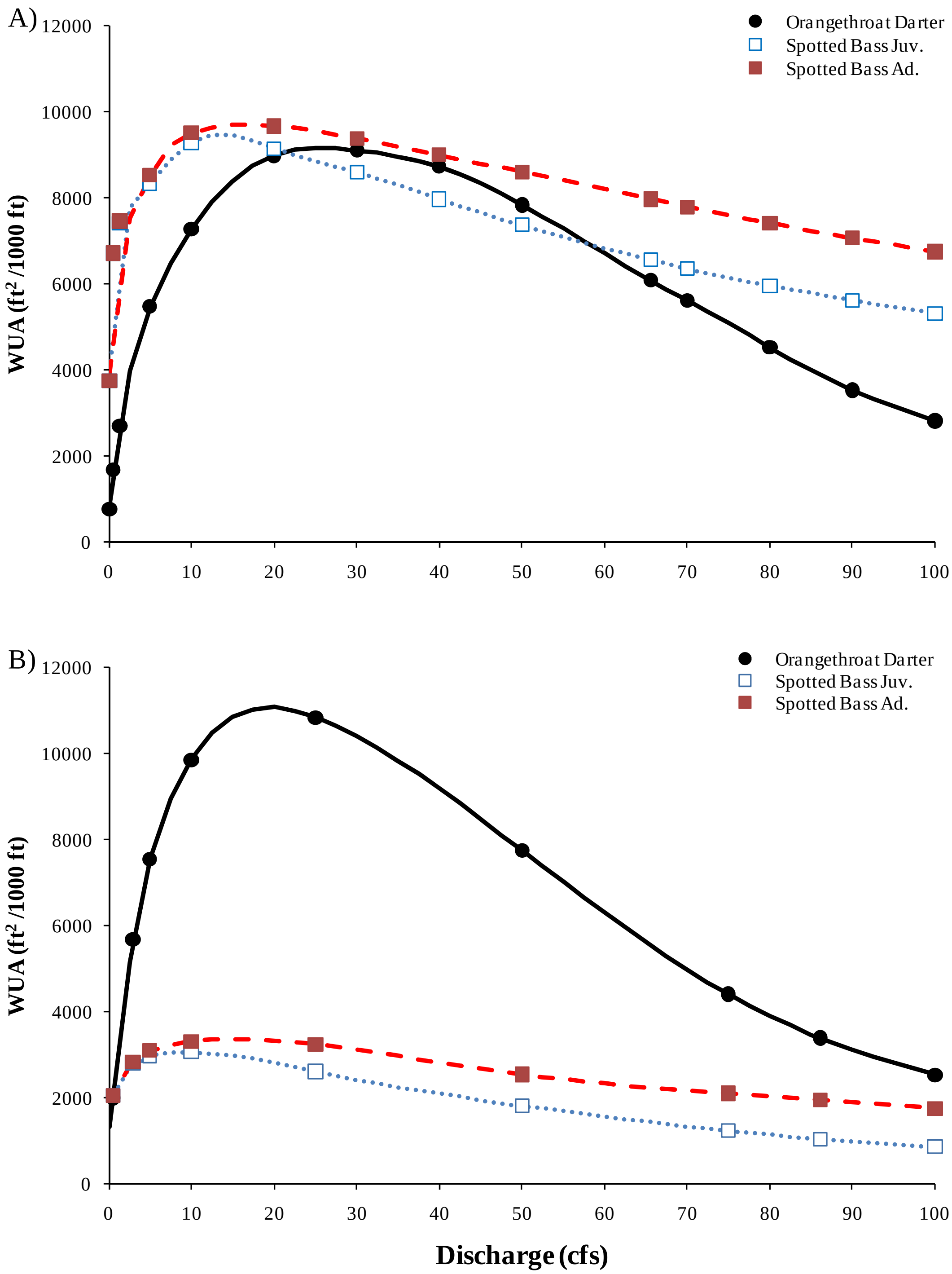

Figure 8: Relationship between weighted usable area (WUA ft^2/1000ft) and discharge for A) Mill1 (orangethroat darter [black circle and solid line], spotted bass juvenile [hollow square and dotted blue line]), spotted bass adult [solid square and dashed red line]); and B) Mill2 (orangethroat darter [black circle and solid line], spotted bass juvenile [hollow square and dotted blue line]), spotted bass adult [solid square and dashed red line]).

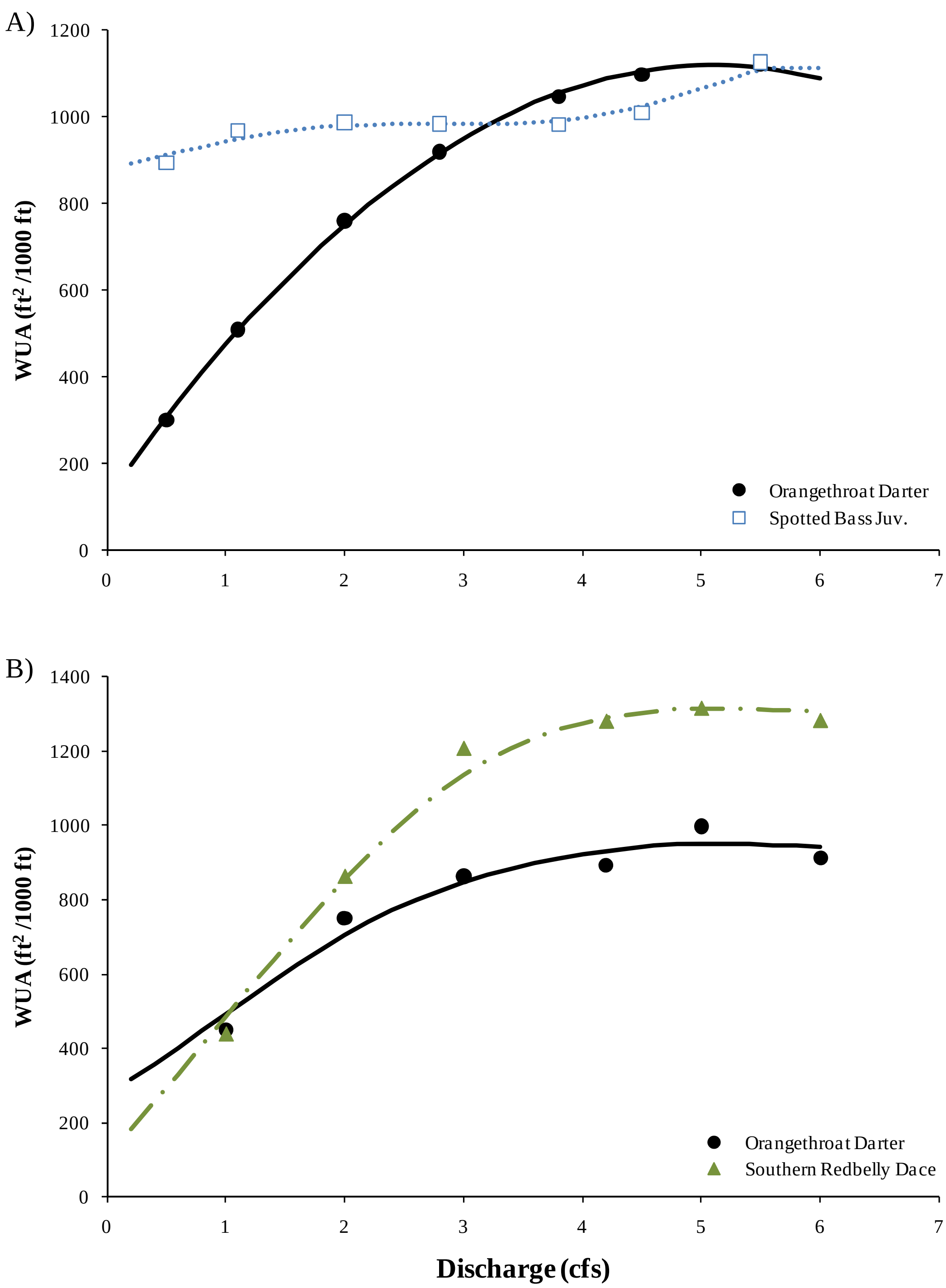

Figure 9: Relationship between weighted usable area (WUA ft^2/1000ft) and discharge for A) Colvert (orangethroat darter [black circle and solid line] and spotted bass juvenile [hollow square and dotted blue line]); and Springhouse (orangethroat darter [black circle and solid line] and southern redbelly dace [solid triangle and dot-dashed green line]).

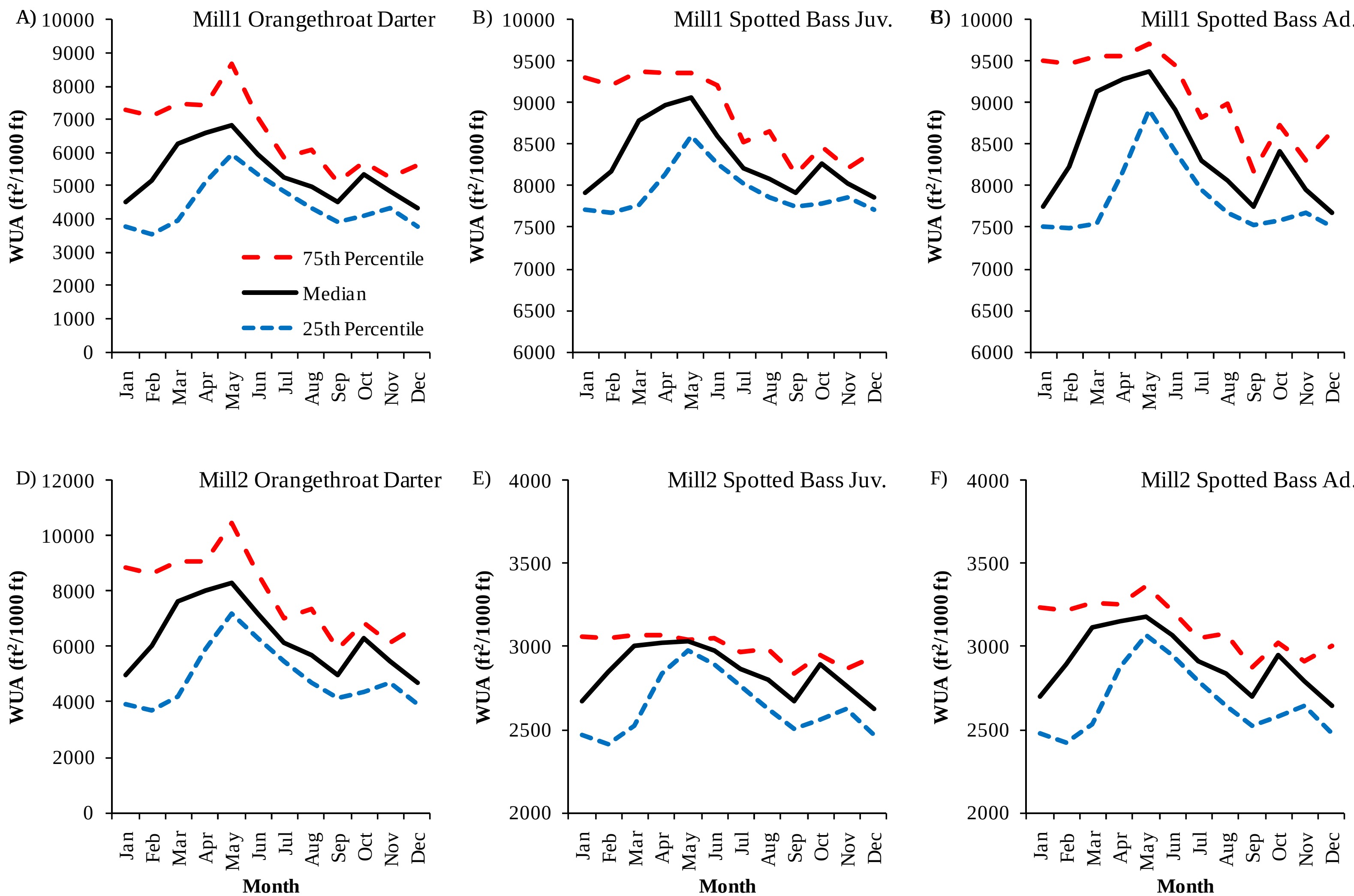

Figure 10: Time series analysis at monthly median (black line), 25th percentile (blue short dashed line), and 75th percentile (red long dashed line) discharge for Mill1: A) orangethroat darter, B) spotted bass juvenile, and C) spotted bass adult; and Mill2: D) orangethroat darter, E) spotted bass juvenile, and F) spotted bass adult..

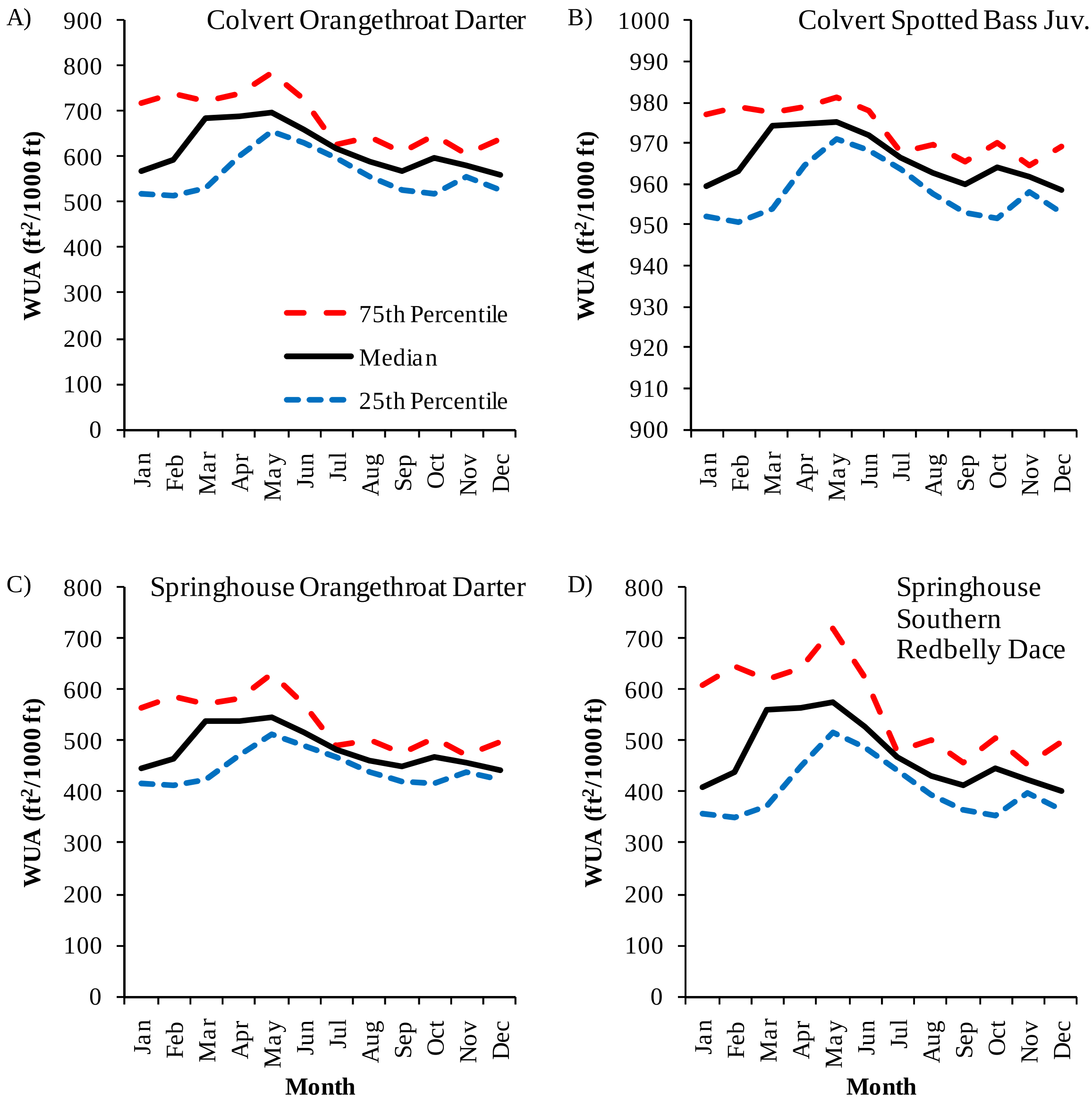

Figure 11: Time series analysis at monthly median (black line), 25th percentile (blue short dashed line), and 75th percentile (red long dashed line) discharge for Colvert: A) orangethroat darter and B) spotted bass juvenile; and Springhouse: C) orangethroat darter, and D) southern redbelly dace.

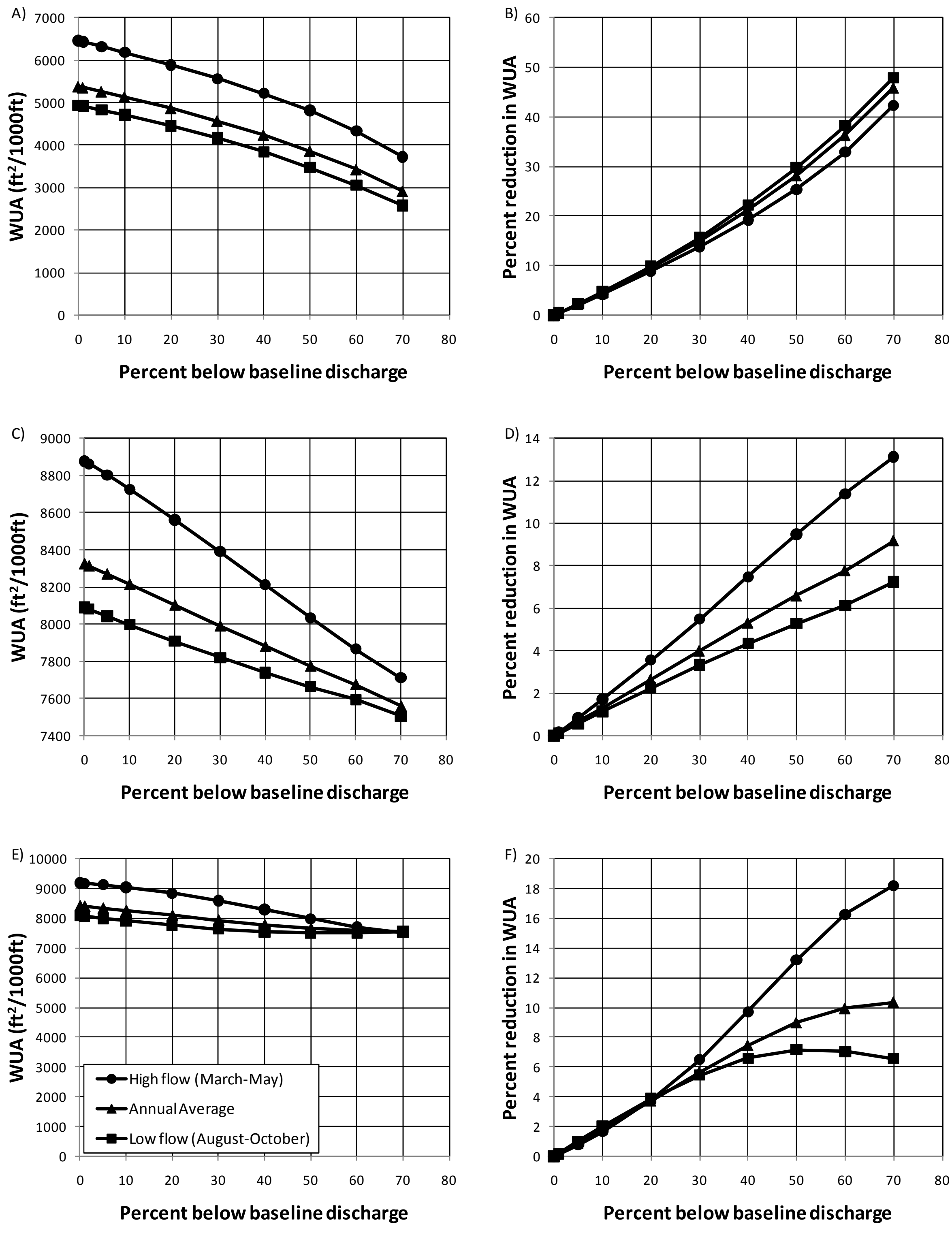

Figure 12: Incremental streamflow reduction scenarios (percent below baseline/median flow) and weighted usable area (WUA ft^2/1000ft) and percent reduction in WUA for orangethroat darter (A and B), spotted bass juvenile (C and D), and spotted bass adult (E and F) in Mill1.

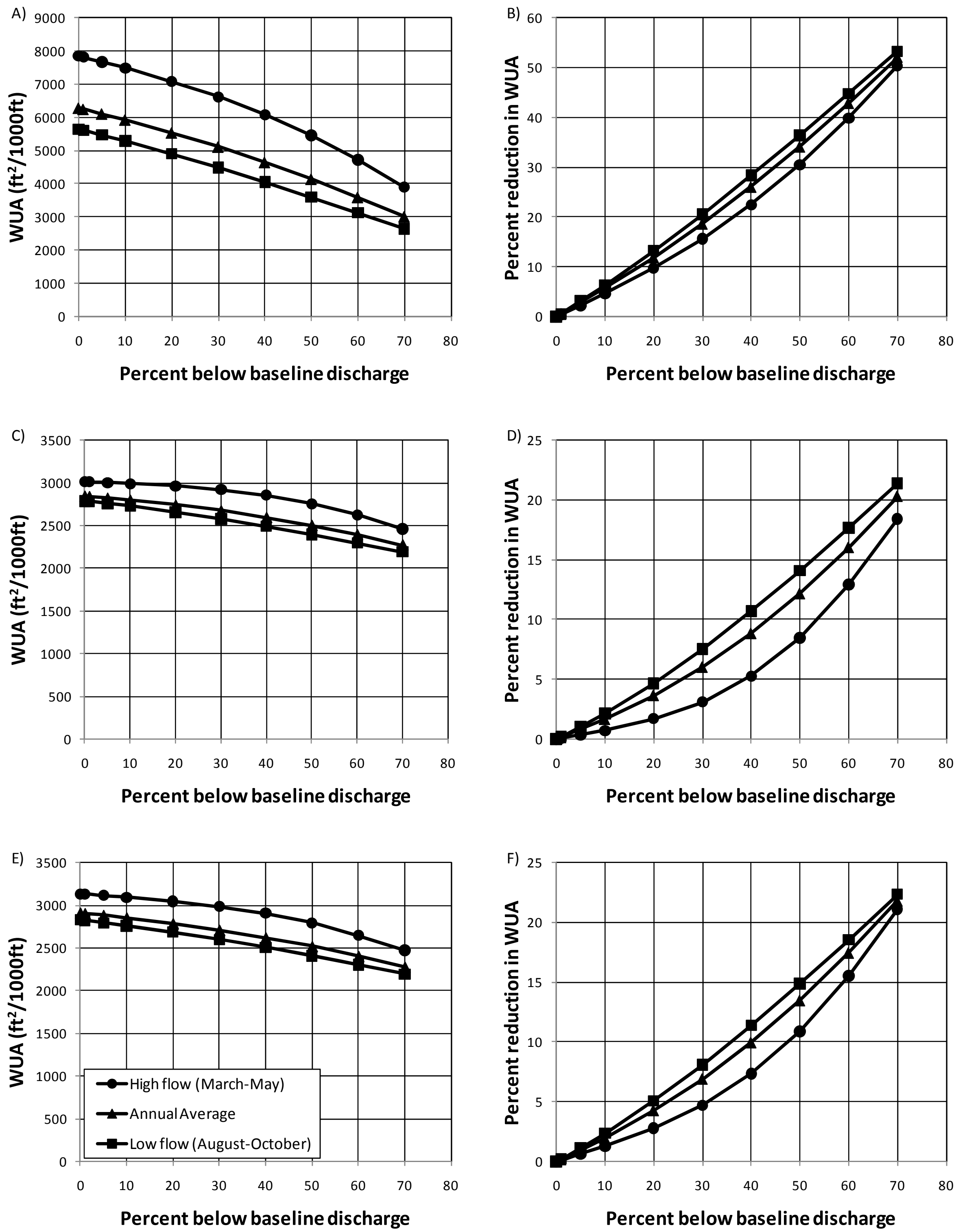

Figure 13: Incremental streamflow reduction scenarios (percent below baseline/median flow) and weighted usable area (WUA $ft^2/1000ft$) and percent reduction in WUA for orangethroat darter (A and B), spotted bass juvenile (C and D), and spotted bass adult (E and F) in Mill2.

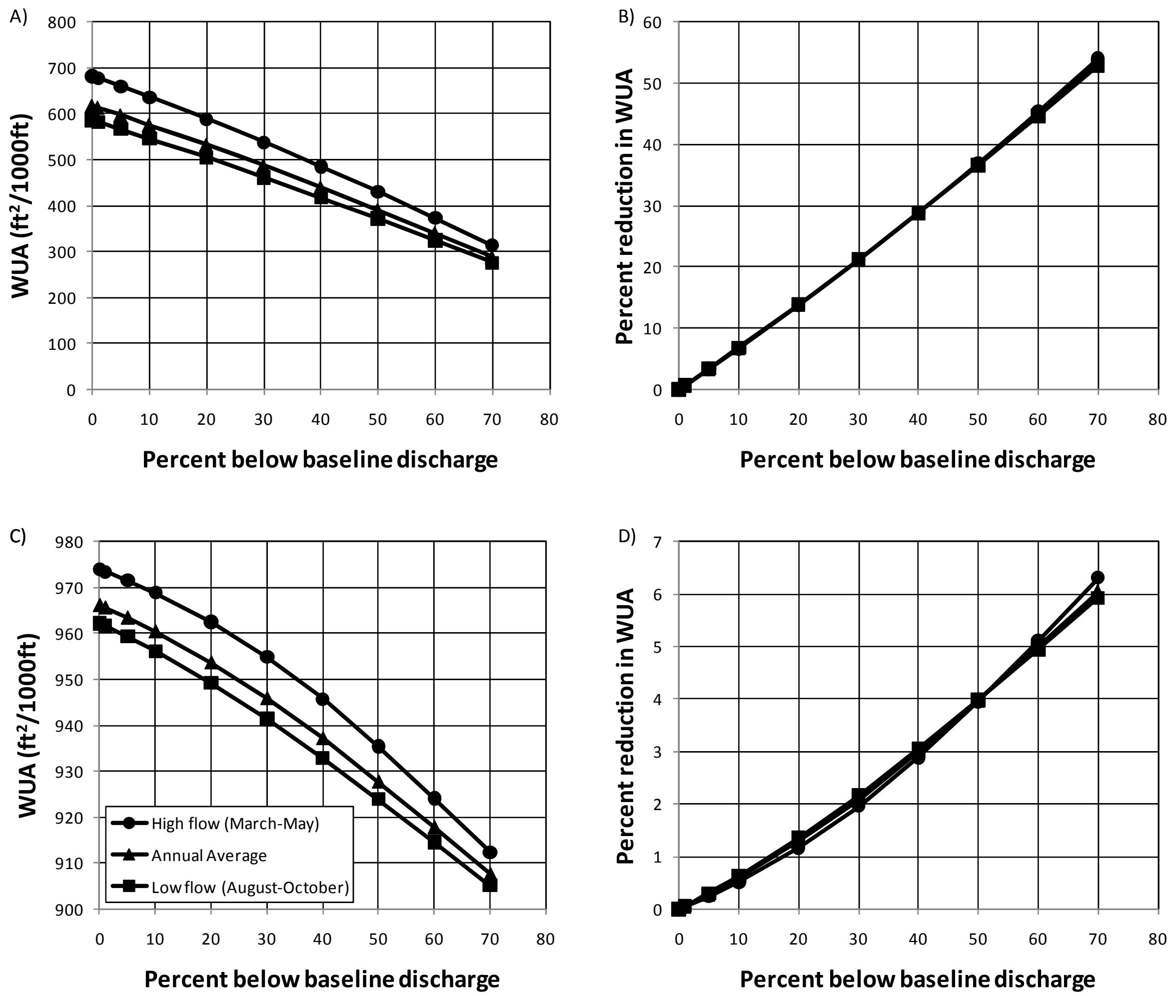

Figure 14: Incremental streamflow reduction scenarios (percent below baseline/median flow) and weighted usable area (WUA ft^2/1000ft) and percent reduction in WUA fororangethroat darter (A and B) and spotted bass juvenile (C and D) in Colvert.

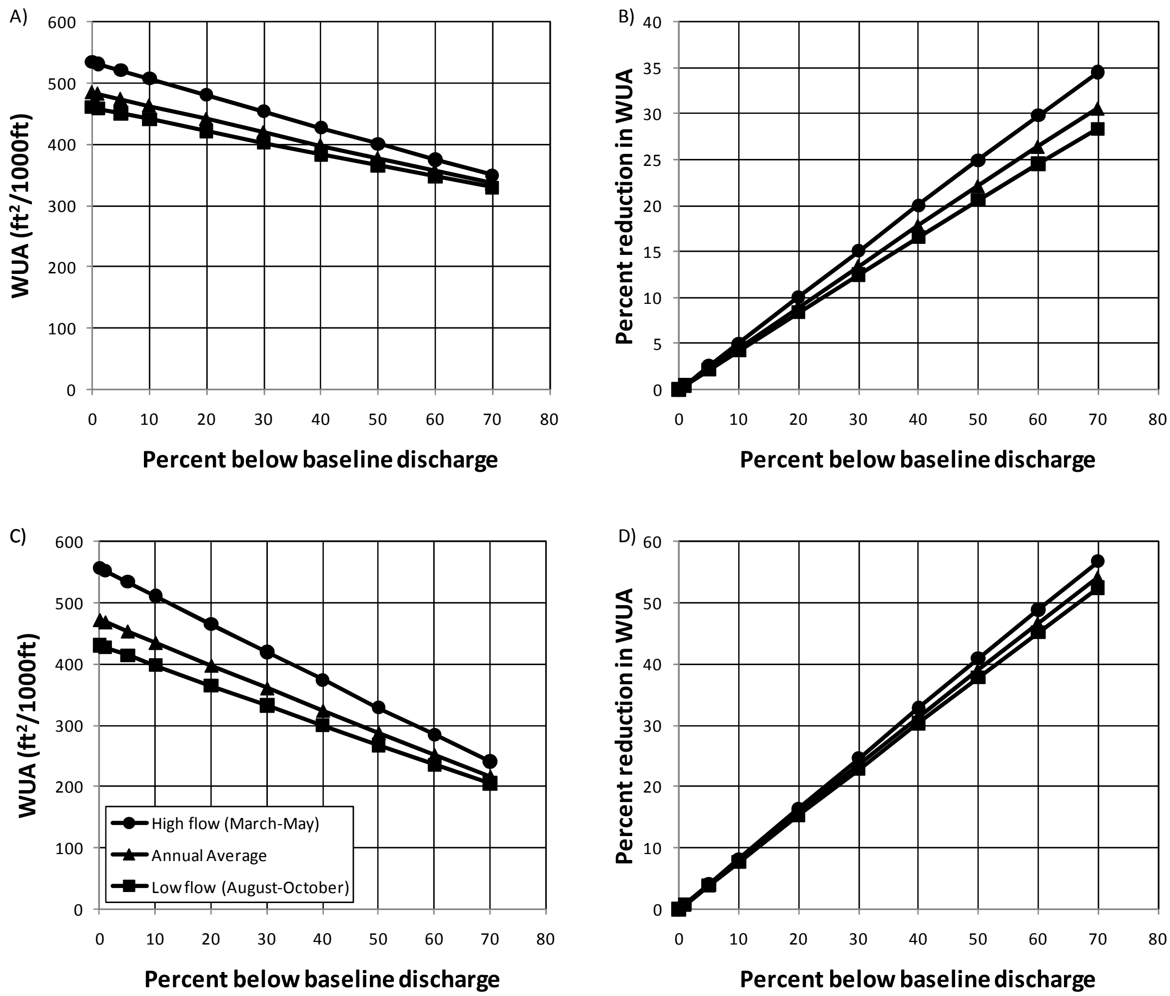

Figure 15: Incremental streamflow reduction scenarios (percent below baseline/median flow) and weighted usable area (WUA ft^2/1000ft) and percent reduction in WUA for orangethroat darter (A and B) and southern redbelly dace (C and D) in Springhouse.