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# EFFECT OF RIPARIAN VEGETATION ON THE SPATIAL DISTRIBUTION OF SLIMY SCULPIN *COTTUS COGNATUS* IN SOUTHWESTERN WISCONSIN STREAMS

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# EFFECT OF RIPARIAN VEGETATION ON THE SPATIAL DISTRIBUTION OF SLIMY SCULPIN *COTTUS COGNATUS* IN SOUTHWESTERN WISCONSIN STREAMS

By Katri Helena Laukkanen

We recommend acceptance of this thesis in partial fulfillment of the candidate's requirements for the degree of Master of Science in Biology-Aquatic Science.

The candidate has completed the oral defense of the thesis.

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# ABSTRACT

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The distribution of freshwater sculpin (*Cottus* spp.), are frequently associated with the quantity and quality of macroinvertebrate prey, which are often influenced by the presence of riparian vegetation. I hypothesized that open- canopies would increase primary and secondary production in riffles, and would result in greater densities of slimy sculpin (*Cottus cognatus*) than in riffles underlying closed- canopies. The quantity of periphyton and macroinvertebrates, and the density and size-frequency distribution of slimy sculpin were monitored for three months in riffles with open- and closed- canopy in three streams in the Coon Creek watershed, Wisconsin. Sculpin densities were not significantly different between riffles with open- or closed- canopies, nor were there differences in periphyton and macroinvertebrate standing crop. However, the size-frequency distribution of sculpin varied by canopy type; a larger proportion of juvenile and small-bodied sculpin were collected in riffles with open- canopies than in riffles with closed- canopies. Many juvenile sculpin were sampled from within macrophyte beds growing in open- canopy riffles. These results suggest that the quantity of photosynthetically active radiation may not influence the density of sculpins through enhancement of periphyton and subsequent increases in macroinvertebrate prey, but rather, may influence the size distribution of sculpin by increasing the growth of macrophytes, which serve as an important habitat for juvenile slimy sculpin.

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# INTRODUCTION

Freshwater sculpin (*Cottus* spp.) are small-bodied, benthic-feeding fish found throughout the northern hemisphere. More than 60 species of freshwater sculpin have been identified (Kinziger et al. 2005) and, in North America, are distributed from as far north as British Columbia and Alaska to the southern coldwater streams of Virginia (Adams and Schmetterling 2007; Kinziger et al 2005). Although freshwater sculpin may inhabit a variety of environments ranging from high gradient streams and rivers (Jenkins and Burkhead 1994) to low gradient coastal streams (Rohde and Arndt 1981) and lakes (Madenjian and Bunnell 2008), slimy sculpin *Cottus cognatus* frequently inhabit riffle habitats of freshwater streams. Within these riffle habitats, slimy sculpin maintain a highly cryptic lifestyle, hiding beneath cobble and large substrates (Gray et al 2004). Numerous studies have found that sculpin are sedentary and tend to have small home ranges (Goto 1998; Gray et al. 2004; Hudy and Shiflet 2009; Petty and Grossman 2004; 2007). Their limited mobility and susceptibility to anthropogenic perturbations make sculpin well suited as bioindicators of habitat and water quality (Spencer et al. 2008).

Sculpin frequently dominate the fish assemblages, both in number of individuals and in biomass, of many cool- and coldwater streams (Adams and Schmetterling 2007, Freeman et al. 1988). As benthic invertivores, sculpin may significantly influence invertebrate biomass. For example, in laboratory studies with reticulate sculpin *(Cottus perplexus)* and cutthroat trout *(Salmo clarki),* sculpin reduced the availability of invertebrate prey which, in turn, resulted in reduced food consumption and production of trout (Brocksen et al. 1968). Some research suggests that freshwater sculpin (primarily lake inhabitants) prey on salmonid eggs and larvae of other sport fishes (Foote and Brown 1998, Mirza and Chivers 2002, Tabor et al 2004). This predatory behavior may be amplified in stream systems impacted by anthropogenic alterations of the habitat (Gadomski and Parsley 2005). For example, experiments in which artificial light was introduced to riverine systems, greater predation of sockeye salmon fry by cottids was observed (Tabor 2004). In addition, increased turbidity and alterations to canopy cover have been shown to increase predation efficiency of sculpin on sport fishes in some systems (Gadomski and Parsley 2005).

In addition to reducing salmonid biomass, sculpin may have a significant role in structuring trophic interaction among lower trophic levels within streams (Konishi et al. 2001; Ruetz et al. 2004). For example, during summer months when high metabolism requires greater feeding activity, sculpin have been found to alter diatom communities via the consumption of invertebrate grazers (Koetsier 2005). Due to their abundance, sculpin also are an important prey for sport fishes, birds, reptiles and mammals (Hodgens et al 2004, Madanjian et al 2005, Koczaja et al. 2005, Kortan 2010, Poe et al 1991) and are likely important intermediary transporters of nutrients and energy within drainages (Goto 1998; Ruzycki and Wurtsbaugh 1999).

The distribution and abundance of sculpin may be affected by a myriad of variables, including physical parameters such as temperature, dissolved oxygen, stream bed stability, discharge patterns and the availability of suitable shelter, such as large rocks or debris (Edwards and Cunjak 2007; Facey and Grossman 1992). In addition to abiotic factors, biotic factors such as food availability, interspecific and intraspecific competition, also influence the occurrence and density of sculpin (Petty and Grossman 2010). For example, mottled sculpin (*Cottus bairdi*) in a southern Appalachian stream selected feeding locations with a high abundance of benthic macroinvertebrates and selected patches that increased access to prey. In turn, the selection of high-quality patches likely increased the individual fitness of sculpin through increased energy gain (Petty and Grossman 1996). Petty and Grossman (2010) also demonstrated that mottled sculpin exhibit hierarchical competitive exclusion from high quality patches, with large adult sculpin accessing patches with the greatest density and volume of prey.

In aquatic environments where prey may be limiting, non-native fishes, such as brown trout *Salmo trutta,* may compete with sculpin for food and habitat (Zimmerman and Vondracek 2006). A lack of terrestrial prey in the drift, for example, may cause brown trout to shift to benthic feeding, which potentially results in the non-native trout out-competing sculpin for ideal habitat patches or feeding positions. Zimmerman and Vondracek (2006) suggest that trout may displace sculpin from resting locations, thus forcing sculpin into higher velocity habitats with corresponding increased activity costs. Therefore, in stream systems where sculpin are ill-adapted to compete with non-native species, their choice of habitats may not be optimal.

In many drainage systems, the physical attributes of streams, such as temperature, significantly influence sculpin distribution. Freshwater sculpin are physiologically adapted to temperatures between 11° C and 22° C, with temperatures of 25° C acutely lethal (Edwards and Cunjak 2007; Otto and Rice 1977). Water temperature was previously cited as the single most important factor affecting sculpin distribution (Edwards and Cunjak 2007, Willock 1969) and sculpin density typically decreases with increasing water temperature and longevity of high summer temperatures (Edwards and Cunjak 2007).

As streams are altered, particularly through the removal of riparian cover, temperature fluctuations often increase because streams are exposed to both ambient air temperatures and direct sunlight. Large diel changes in temperature, and associated changes in dissolved oxygen concentrations, are important limiting factors structuring the species composition within some streams (Wehrly et al. 2003). In these streams, species richness is often reduced, especially the richness of stenothermic invertebrates (Wehrly et al. 2003). As a consequence, food resources for predatory fish, such as sculpin, may become limited. However, stream temperatures are spatially heterogeneous, and vary longitudinally along stream gradients, with lower temperatures in the upper reaches, and higher temperatures downstream (Poole and Berman 2001). Therefore, in low-order streams with stable base flows, temperature fluctuations may be relatively mild, and therefore may not limit species richness.

Habitat availability may also be significantly reduced with the removal of riparian vegetation. Riparian vegetation provides stable habitat for aquatic macroinvertebrates in the form of large woody roots and leaf litter, which serves as both habitat and food for macroinvertebrate shredders (Sweeney 1993). In addition, large woody debris provides cover for sculpin, as well as larger fish species. In turn, loss of riparian vegetation may result in reduced food availability for fish species, as well as reduced habitat for sculpin.

In addition to altering the habitat availability and thermal regime of streams, riparian vegetation also influences stream bank structure and the geomorphology of streams. For example, stream width changes significantly in response to changes in the type of vegetation bordering the stream. Sweeney (1993) noted that forested streams tended to be 2.5 times wider than meadow streams. Stream size, in turn, further affects water temperature. For example, in small streams, riparian forest moderates stream temperature by reducing the amount of solar radiation reaching the water surface (Beschta et al. 1986).

Accelerated soil erosion due to agriculture has been the greatest contributor of increased turbidity in most rivers (Walling and Fang 2003), and has decreased benthic light availability by enhancing the attenuation of light within the water column (Julian et al. 2008b). Light availability is often the driving force behind many fundamental processes within aquatic systems, including photosynthesis, photochemical reactions, thermal fluctuations and animal behaviors (Julian et al. 2008; Sweeney 1993).

Increased sunlight in sections of open-canopy streams increases fauna and flora (Murphy et al. 1981; Murphy and Hall 1981). Removal of streamside vegetation by logging results in an increase in aquatic production at all trophic levels through the augmentation of periphyton production (Murphy et al. 1981; Murphy and Hall 1981). Mulholland et al. (2009) studied the indirect effects of a late spring freeze on Walker Branch, a stream in Tennessee. The death of newly formed leaf tissues on riparian plants, as a result of freezing temperatures, reduced riparian canopy cover throughout the summer months. Consequently, significantly greater quantities of photosynthetically active radiation reached the stream bed of Walker Branch, resulting in higher gross primary production (GPP), which then lead to a cascade of effects including increased productivity within all trophic levels of the stream.

Much research focuses on the effects of geomorphology and hydrological processes within stream communities on nutrient cycling and primary production (Peterson and Grimm 1992; Stockner and Shortreed 1978). However, the effects of light availability on production and trophic interactions have received much less attention (Julian, et al. 2008a; Julian, et al. 2008b). Studies to date, which focused on food webs dominated by salmonids, provide inconclusive evidence that light availability affects the abundance and production of salmonids. For example, Murphy et al. (1981), and Hawkins et al. (1983) noted that canopy removal was associated with increased salmonid biomass. In addition, Murphy et al. (1981) reported that open, clear-cut habitats within streams exhibited greater rates of respiration, higher densities of benthic algae, and greater salamander populations than those with closed canopies. Tait et al. (1994), however, found a decrease in both steelhead (*Oncorhychus myskiss*) and sculpin as a result of increased solar radiation. Tait et al. attribute the decrease in steelhead and sculpin to lethal water temperatures, exceeding 30 C at open sites. In contrast, stream temperatures did not exceed 22 C in streams studied by Murphy et al. (1981), and therefore did not reach incipient lethal levels for salmonids.

Because of sculpin's limited vagility and reliance upon benthic invertebrates, the habitat choice of sculpin may be indicative of profitable or poor habitat conditions. My objective was to examine the effects of riparian cover on primary production, aquatic invertebrates, and slimy sculpin. Specifically, I assessed differences between riffles under closed- and open- canopy in primary production, standing crop of aquatic invertebrates, and the density and size-frequency distribution of slimy sculpin. Due to the greater availability of light, I hypothesized that riffles under open canopy would have a greater standing crop of periphyton than riffles under closed canopy. In turn, greater primary production would be expected to have a positive bottom-up effect on the rest of the food web, with greater availability of food for grazing aquatic macroinvertebrates which, consequently, would support higher densities of sculpin.

## METHODS

### Study Area

This study was conducted during June, July, and August 2009. The study area, located in southwestern Wisconsin, included three streams in the Coon Creek Watershed: Rullands Creek, Bohemian Valley Creek, and Timber Coulee Creek (Figure 1). The Coon Creek watershed (196 $km^2$) flows through the unglaciated Driftless Zone and discharges into the Mississippi River near the city of Stoddard, WI (Cavanaugh et al., 2004). The upper stretches of the watershed are dominated by deciduous forests and limestone bedrock, and the lower reaches consist of wide valleys surrounded by limestone bluffs. Much of the lower reaches of this watershed are surrounded by row-crop agriculture and cattle pastures. Discharge is primarily from groundwater flow, resulting in stable temperatures and significant base-flows within the streams.

Four riffles in each stream were selected as study reaches based on their physical attributes and included two closed-canopy sites with riparian vegetation blocking more than 75% of the sunlight directly striking the stream, and two open-canopy sites with the stream bed exposed to direct sunlight. Thus, for all streams combined, a total of six closed-canopy reaches, and six open-canopy reaches were used. Riparian vegetation and light exposure within sites were determined with a spherical crown densitometer (Forest Densiometers Spherical Densiometer Concave - Model C, Forestry Suppliers, Inc, Jackson, MS) each month at the upstream, middle, and downstream end of the riffle.

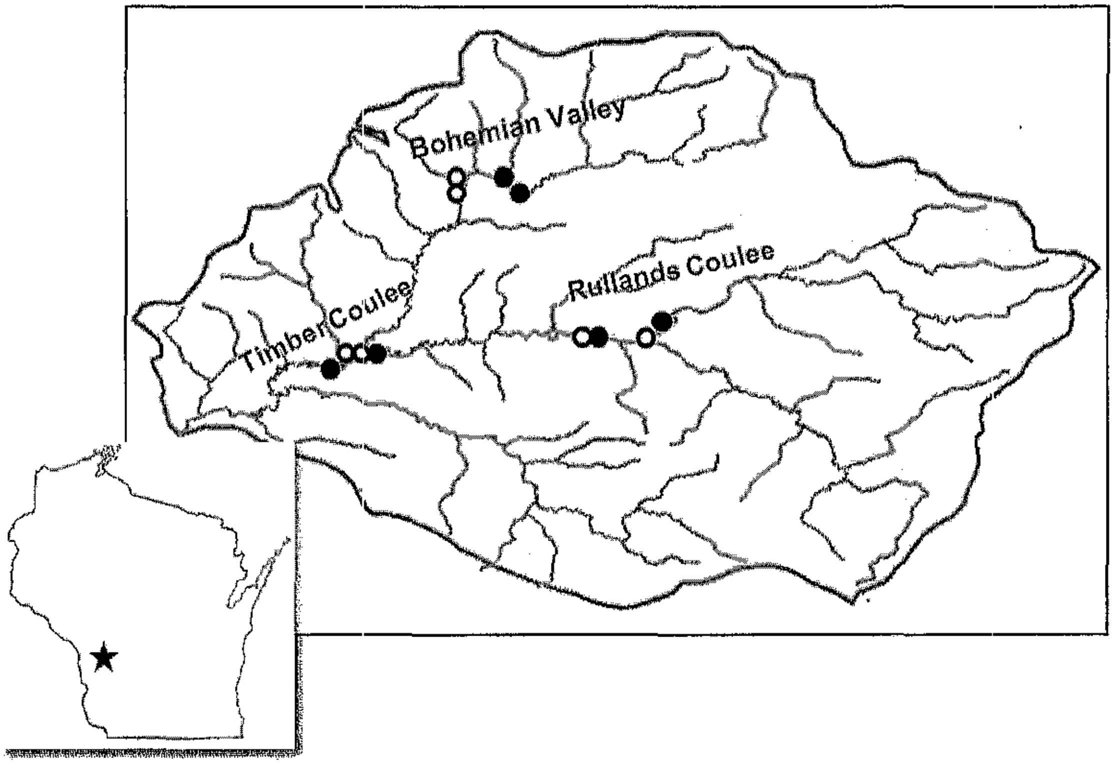

Figure 1. Location of the 3 study streams (Timber Coulee Creek, Bohemian Valley Creek, and Rullands Coulee Creek) in Coon Creek watershed, Wisconsin. Two ● closed-canopy sites, and two ○ open-canopy sites were sampled per stream.

## Physical Habitat

The length and width of each riffle was measured. Water depth and velocity were also measured at the upstream, middle, and downstream end of the riffle during each sample period. Water velocity was measured with a velocity meter (Marsh-McBirney Flow Meter ®2000, Marsh-McBirney, Frederick, MD). Due to the strong link between sculpin habitat and substrate characteristics (Davey et al. 2005, Brown 1991), substratum size analysis was conducted at each site. The most abundant substrate size was determined by classifying pebbles into categories according to a modified Wentworth scale: particulate organic matter =0, silt=1 (diameter < 0.06 mm), sand=2 (0.06 – 2 mm), fine gravel =3 (2 - 8 mm), medium gravel = 4 (8 - 32 mm), coarse gravel=5 (32 – 64 mm), small cobble= 6 (64 - 128), large cobble =7 (128 – 256 mm), boulder = 8 (>256 mm) (Wolman 1954).

Although I did not quantitatively assess the presence or density of aquatic macrophytes in riffles, I noted that there was a substantial amount of growth and expansion of aquatic plants in nearly all open-canopy sites over the three summer months. Those plants most abundant at open sites included *Elodea canadensis*, *Ceratophyllum spp*., *Potamgogeton spp.* and *Myriophyllum spp*. There were no to very few macrophytes present at closed-canopy sites.

### Sculpin

Once each month, sculpin were collected at each riffle with a kicknet (1 m in length; 2 mm bar mesh). The net was placed perpendicular to the current, and an area of 1 $m^2$ directly upstream was disturbed by shuffling substrates for 1 minute. This was done in an upstream direction within each riffle to minimize disturbance of un-sampled areas. As substrates were disturbed, sculpin were swept along with the stream current and were caught in the screen of the kick-net. The entire stream width of each riffle segment was sampled. The density of sculpin within each site was determined by dividing the total number of sculpin collected in each riffle by the total area sampled.

Collected sculpin were placed in a bucket of stream water and lightly anesthetized with tricaine methanesulfonate (MS-222, Argent Chemical Laboratories, Inc., Redmond, WA). Total length of each sculpin was measured, and the gut contents of the sculpin were removed by gastric lavage. Sculpin were allowed to recover from the anesthetic and return to the stream riffle. Gut contents were flushed into a fine-mesh net (0.5 μm) and preserved in 95% ethanol. Invertebrates found within stomach contents were later identified to lowest taxon when possible (genus) in the laboratory and Ivlev's Electivity Index (1961) was calculated to evaluate potential sculpin preferences for different types of invertebrate prey in the diet.

### Macroinvertebrates

To estimate the biomass of macroinvertebrates in each riffle, I collected invertebrates from cobble clusters with a D-frame dip net (150 µm nylon mesh) following procedures by Cavenaugh et al. (2004). Each invertebrate sample consisted of a three-cobble cluster, which was obtained at three points along a transect within each riffle. Invertebrates were removed from cobbles and preserved in 95% ethanol, and identified to lowest practical taxonomic category in the laboratory. The density of larvae of Chironomids, *Simulium* sp., *Baetis* sp., and *Gammarus psuedolimnaeus* (not age-specific), four invertebrates common in the diet of sculpin, were estimated by digitally photographing cobbles, and collecting and counting the number of individuals from the surfaces of each rock. The number of invertebrates was then divided by the exposed surface area of cobble samples to yield densities. In addition, the body surface area of these four groups of larval invertebrates were digitally measured with ImagePro® software (MediaCybernetics, Bethesda, MD). Areal measurements of the four invertebrate taxa were used to calculate individual dry mass (mg) and standing crop with formulas from Benke et al. (1999) for *Baetis*, *Gammarus*, and *Simulium*, and from R. J. Haro (University of Wisconsin-La Crosse, *unpublished data*) for Chironomidae.

*Baetis (larval dry mass (mg)) = 0.0076 * (body length) ^ 2.691*

*Gammarus (larval dry mass (mg)) = 0.0049 * (body length) ^ 3.001*

*Simulium* (larval dry mass (mg)) = 0.004 * (body length) ^ 2.807

*Chironomidae* (larval dry mass(mg)) = 0.001 * (body area) ^ 2.761

### Periphyton and Ash Free Dry Mass

Similar to procedures used for sampling invertebrates, periphyton samples were obtained from three-rock clusters within each riffle. Rock clusters were selected randomly from three points along a transect at each site. A plastic pipe (area 5.68 $cm^2$) was secured to each rock's surface with plumber's putty, and periphyton was removed with a stiff brush and deionized water. Periphyton was washed into a plastic pan, and transferred into opaque bottles that were stored on ice for transport to the laboratory. Samples were stored at 7 C until they were filtered within 24 h of collection.

Periphyton in the samples were concentrated on a pre-weighed Whatman®GF/F (47 mm; Piscataway, NJ) glass fiber filter and analyzed for ash-free dry mass (AFDM). Ash-free dry mass was measured by drying organic matter from each sample to a constant weight in a pre-ashed aluminum boat at 75 C and then combusting the samples in a muffle furnace for 4 h at 550 C (Rosenfeld 2000, Steinman et al. 2007).

### Data Analysis

Data were natural log or rank transformed, if necessary, prior to statistical analysis to meet the assumptions of normality and homogeneity of variances. Differences in water depth, velocity, discharge, temperature, dissolved oxygen, invertebrate density and standing crop, AFDM, and sculpin density between open- and closed canopy, and among streams and months, were analyzed with a repeated-measures split-plot analysis of variance (ANOVA). The 4 sites within each of the 3 streams were each considered as an experimental unit for the purposes of statistical analysis. The main plot evaluated the effects of canopy type (open or closed) and stream, as well as the interaction between canopy and stream. The sub-plot evaluated the effects of sampling period and the interaction of sampling period with canopy type, stream, and the three-way interaction between sampling period, canopy type and stream. Because wetted widths were measured only once at the start of the study, a two-factor ANOVA was used to assess the effects of this variable. In addition, a single-factor ANOVA was used to evaluate each individual taxon density, as sample size was insufficient to use multiple-factor ANOVAs. Also, because data on sculpin total length still did not meet assumptions of normality and homogeneity of variances after transformation, differences in the monthly size-frequency distribution of sculpin between canopy types were evaluated with a Kolmogorov-Smirnov (K-S) two-sample test.

Regression analysis was used to examine the relationship between sculpin density, stream depth, stream discharge, potential invertebrate prey (standing crop) and the total length of sculpin. With the exception of the K-S test, a P-value of 0.05 was used to reject null hypotheses. Because a separate K-S test was used for each of the 3 sampling periods, a P-value of 0.017 was used to reduce the probability of making a Type II error. All statistical analyses for these data were conducted with SAS® software version 9.1.3 (SAS Institute Inc, Cary, NC).

Substrate heterogeneity was determined from pebble count data using a Shannon-Wiener diversity index in which the relative abundances of particles were determined by size category. Hutcheson t-test (1970) was then used to determine differences in the diversity index (size categories) between open- and closed- canopy sites.

# RESULTS

## Physical Habitat

Cover at closed-canopy sites exceeded 59%, with relatively low variation in cover over time. Canopy cover (mean ± SE) among the 6 closed sites was 85.3 ± 2.42 %. With the exception of one site in the Rullands Coulee stream (1.4 % cover), all open-canopy sites had no cover (0.14% ± 0.1%).

With the exception of water temperature, physical attributes (mean ± SE) were not significantly different between open- and closed-canopy sites or among streams (Table 1). Stream depth (0.22 ± 0.01 m) was not significantly different among canopy, streams, or months. Wetted width (4.03 ± 0.17 m) also was not significantly different between open- and closed-canopy sites, nor among streams. Water velocity was not different between open- and closed canopy sites, but was different among months. Mean water velocity was greater in July (0.78 ± 0.05 m/s) than in June (0.71 ± 0.03 m/s) or August (0.58 ± 0.02 ms/). Water temperature was significantly different between open- and closed-canopy sites, among streams, and between months (Table 1). Mean temperatures at open-canopy sites were approximately 2 C greater than at closed-canopy sites. In addition to having greater mean temperatures at open-canopy sites, open sites also exhibited a greater ranges in temperatures (12.8 to18.4 C), than closed-canopy sites (12.5 to16.5 C). The interactive effects of canopy type and stream on water temperature were also significant. Water temperature was greater in July (15.22 ± 0.49 C) than August (14.49 ± 0.61). Dissolved oxygen (11.7 ± 0.2 mg/L) was also only measured in

July and August, but did not significantly differ by canopy type, stream or period (Table 1). In addition, the size of the rock substrate did not significantly differ by canopy type ($t_{0.05(2)} = 2.131$).

Table 1. Results of repeated-measures analysis of variance for effects of canopy type, stream, and month on (A) stream depth, (B) wetted width, (C) water velocity, (D) water temperature, and (E) dissolved oxygen.

| Source | df | F | *P* |
|---|---|---|---|
| (A) Stream depth | | | |
| Main Plot | | | |
| Canopy (C) | 1 | 1.36 | 0.287 |
| Stream (S) | 2 | 1.24 | 0.354 |
| C x S | 2 | 0.49 | 0.637 |
| Error | 6 | | |
| Subplot | | | |
| Month (M) | 2 | 2.3 | 0.142 |
| M x C | 2 | 0.11 | 0.894 |
| M x S | 4 | 0.37 | 0.828 |
| M x C x S | 4 | 0.55 | 0.704 |
| Error | 12 | | |
| B) Wetted width | | | |
| Main Plot | | | |
| Canopy (C) | 1 | 5.25 | 0.062 |
| Stream (S) | 2 | 4.47 | 0.065 |
| C x S | 2 | 10.39 | 0.011 |
| Error | 6 | | |
| C) Water velocity | | | |
| Main Plot | | | |
| Canopy (C) | 1 | 1.25 | 0.306 |
| Stream (S) | 2 | 1.24 | 0.355 |
| C x S | 2 | 0.16 | 0.858 |
| Error | 6 | | |
| Subplot | | | |
| Month (M) | 2 | 7.31 | < 0.001 |
| M x C | 2 | 1.74 | 0.178 |
| M x S | 4 | 4.03 | 0.003 |
| M x C x S | 4 | 2.17 | 0.073 |

| | | | |
|---|---|---|---|
| Error | 12 | | |
| D) Water temperature | | | |
| Main Plot | | | |
| Canopy (C) | 1 | 24.76 | 0.003 |
| Stream (S) | 2 | 10.71 | 0.011 |
| C x S | 2 | 10.36 | 0.011 |
| Error | 6 | | |
| Subplot | | | |
| Month (M) | 1 | 8.51 | 0.027 |
| M x C | 1 | 3.48 | 0.111 |
| M x S | 2 | 9.96 | 0.012 |
| M x C x S | 2 | 2.32 | 0.179 |
| Error | 6 | | |
| E) Dissolved oxygen | | | |
| Main Plot | | | |
| Canopy (C) | 1 | 0.31 | 0.599 |
| Stream (S) | 2 | 0.80 | 0.49 |
| C x S | 2 | 2.88 | 0.133 |
| Error | 6 | | |
| Subplot | | | |
| Month (M) | 1 | 0.52 | 0.498 |
| M x C | 1 | 1.04 | 0.348 |
| M x S | 2 | 0.20 | 0.82 |
| M x C x S | 2 | 2.15 | 0.20 |
| Error | 6 | | |

## Macroinvertebrates

The overall mean density for the sum total of all invertebrates collected in stream habitats did not differ by canopy type or by stream. However, the mean density of invertebrates did differ by month (F = 7.19, df = 2, P = 0.0013), with the least number of individuals/m$^2$ collected in June (mean 9765 ± 934.6 individuals/m$^2$), and the greatest densities in August (mean 14060 ± 1175 individuals/m$^2$).

Density and standing crop (Table 2) of Chironomids did not differ significantly between open- and closed- canopies, nor among months. However, there was a significant difference among streams, with the mean density ranging from 1052 to 2987 individuals/m$^2$, and standing crop ranging from 1.97 to 9.20 mg/m$^2$.

Table 2. Results of repeated-measures analysis of variance for effects of canopy type, stream, and month on standing crop of (A) Chironomidae, (B) *Baetis* sp, (C) *Gammarus pseudolimnaeus*, and (D) *Simulium* sp.

| Source | df | F | *P* |
|---|---|---|---|
| (A) Chironomidae | | | |
| Main Plot | | | |
| Canopy (C) | 1 | 2.18 | 0.191 |
| Stream (S) | 2 | 29.16 | <0.001 |
| C x S | 2 | 14.98 | 0.005 |
| Error | 6 | | |
| Subplot | | | |
| Month (M) | 2 | 0.24 | 0.792 |
| M x C | 2 | 1.71 | 0.222 |
| M x S | 4 | 1.32 | 0.318 |
| M x C x S | 4 | 0.82 | 0.534 |
| Error | 12 | | |
| (B) *Baetis sp.* | | | |
| Main Plot | | | |
| Canopy (C) | 1 | 2.1 | 0.198 |
| Stream (S) | 2 | 6.49 | 0.032 |
| C x S | 2 | 0.19 | 0.829 |
| Error | 6 | | |
| Subplot | | | |
| Month (M) | 2 | 7.77 | 0.007 |
| M x C | 2 | 0.37 | 0.698 |
| M x S | 4 | 1.4 | 0.292 |
| M x C x S | 4 | 1.37 | 0.301 |
| Error | 12 | | |
| (C) *Gammarus pseudolimnaeus* | | | |
| Main Plot | | | |
| Canopy (C) | 1 | 1.1 | 0.342 |
| Stream (S) | 2 | 2.29 | 0.197 |
| C x S | 2 | 0.03 | 0.971 |
| Error | 5 | | |

|  |  |  |  |
|---|---|---|---|
| Subplot |  |  |  |
| Month (M) | 2 | 0.07 | 0.932 |
| M x C | 2 | 0.04 | 0.959 |
| M x S | 4 | 1.26 | 0.36 |
| M x C x S | 4 | 0.19 | 0.939 |
| Error | 12 |  |  |
| (D) *Simulium* sp. |  |  |  |
| Main Plot |  |  |  |
| Canopy (C) | 1 | 0.5 | 0.506 |
| Stream (S) | 2 | 1.46 | 0.303 |
| C x S | 2 | 0.18 | 0.84 |
| Error | 6 |  |  |
| Subplot |  |  |  |
| Month (M) | 2 | 2.24 | 0.149 |
| M x C | 2 | 0.42 | 0.669 |
| M x S | 4 | 1.04 | 0.425 |
| M x C x S | 4 | 1.31 | 0.322 |
| Error | 12 |  |  |

The mean density of *Baetis* sp. (F =14.12, df = 1, P = 0.009), differed significantly between open- and closed- canopy types, with a mean density of 1812 ± 181 individuals/m$^2$ at open- canopy sites, and 1128 ± 134 individuals/m$^2$ at closed- canopy sites. Standing crop (42.54 ± 14.64 mg/m$^2$; Table 2) of *Baetis* sp. did not significantly differ between canopy types, however, did vary among streams, ranging from 31.04 ± 16.32 to 57.84 ± 27.29 mg/m$^2$. Density and standing crop also differed among months, with the greatest number (2107 ± 267 individuals/m$^2$) and biomass (26.82 ± 3.39 mg/m$^2$) of *Baetis* sp. observed in July, and the least (1052 ± 123 inidividuals/m$^2$; 16.48 ± 2.97 mg/m$^2$) observed in June.

There was not a significant difference between canopy type, among streams, or among months in the density and standing crop (Table 2) of *Gammarus pseudolimnaeus.* The mean ± SE density and standing crop was 183 ± 22 individuals/m$^2$ and 25.50 ± 3.69

g mg/m$^2$ for *Gammarus pseudolimnaeus.* While there was not a significant difference in the standing crop of Simulium by canopy type, stream, or by month (Table 2), *Simulium* sp. densities did vary by month (F = 3.26, df= 2, P = 0.043), although were not significantly different between canopy types or among streams. The mean ± SE density and standing crop was 2371 ± 352 individuals/m$^2$ and 71.40 ± 11.32 mg/m$^2$ for *Simulium* sp.

Of the 40 taxa collected, (excluding the four most common taxa: *Baetis* sp, Chironomidae spp, *Gammarus psuedolimnaeus* and *Simulium* sp.) only densities *Hydroptilidae* larva (F =15.03, df = 1, P = 0.008), and *Limnephilidae* pupa (F =10484.4, df = 1, P < 0.0001) differed significantly by canopy type.

The four most abundant taxa in the sculpin diet included *Baetis* sp., Chironomidae spp., *Gammarus pseudolimnaeus* and *Simulium* sp. (Figure 2). A calculation of Ivlev's Index indicated that sculpin selected *Gammarus psuedolimnaeus* ($E_i$ =0.80), *Simulium* sp. ($E_i$ =0.34) and *Baetis* sp. ($E_i$ = 0.31) at open sites, and selected *Gammarus psuedolimnaeus* ($E_i$ =0.77), *Simulium* sp. ($E_i$ =0.47) and *Optioservus* sp. larva ($E_i$ =0.36) at closed sites (Table 3). The size (mg dry weight) of *Baetis, Gammarus pseudolimnaeus*, and *Simulium* in the gut of the sculpin did not differ between canopy types or among streams and months. However, Chironomids in the guts of sculpin at closed- canopy sites were larger ($F_{1,3}$ = 47.24, p=.006; least-square mean = 0.0148 ± 0.008 g dry weight) than those at open-canopy sites (least-square mean = -0.009 ± 0.014 g dry weight).

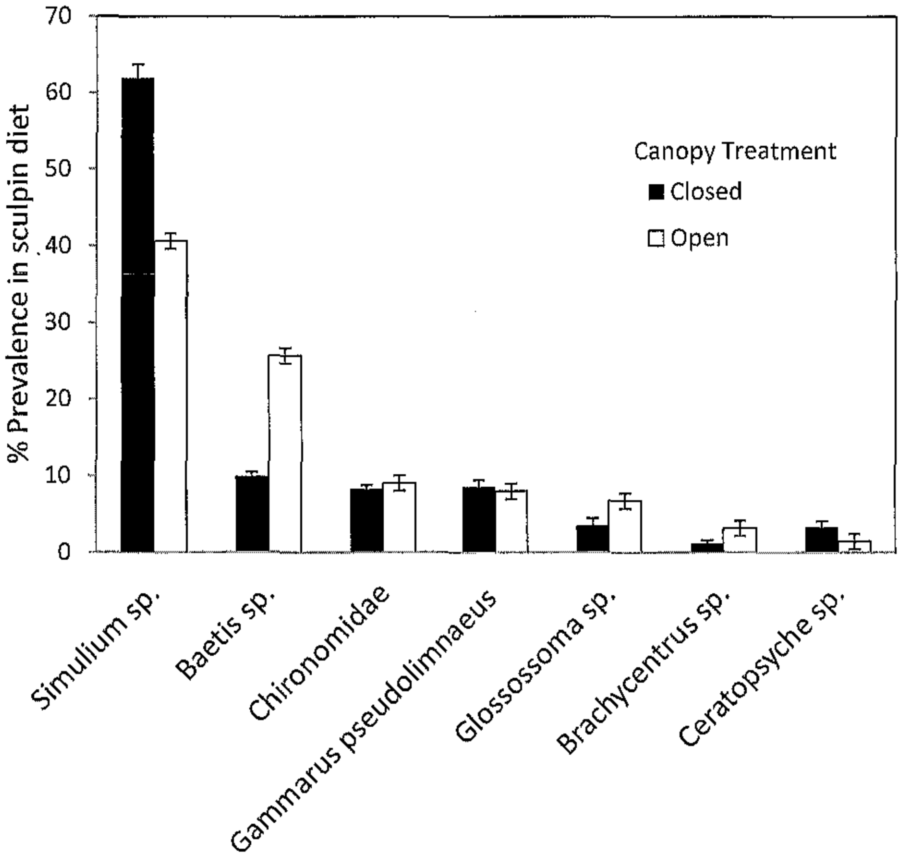

Figure 2. Mean prevalence (% ± SE) of the 7 most common invertebrate taxa in sculpin diet in closed- and open- canopy riffles.

Table 3. Results of Ivlev's Electivity Index ($E_i$) for major prey taxa consumed by sculpin at closed- and open- canopy sites. Positive values indicate active selection by sculpin. Negative $E_i$ values indicate no selection.

|  | $E_i$ |  |
|---|---|---|
| Taxon | Closed | Open |
| *Gammarus pseudolimneus* | 0.77 | 0.80 |
| *Simulium* sp. | 0.47 | 0.34 |
| *Optioservus* sp. | 0.36 | -- |
| *Baetis* sp. | -0.05 | 0.31 |
| *Glossossoma* sp. | -0.62 | -0.33 |
| Chironomidae *spp.* | -0.49 | -0.34 |
| *Ceratopsyche* sp. | -0.06 | -0.45 |
| Hydracarina | -0.35 | -0.66 |
| *Brachycentrus* sp. | -0.88 | -0.67 |

### Periphyton and Ash Free Dry Mass

Ash free dry mass (AFDM) was not significantly different between canopy cover (F =0.34, df = 1, P = 0.56), streams (F =0.07, df = 2, P = 0.93), or by month (F =0.59, df = 2, P = 0.56). However, there was a significant canopy x stream interaction (F =11.99, df = 2, P = 0.008). The mean AFDM ranged from 6.9 to 19.7 mg/cm$^2$ at open- and closed-canopy sites in the three streams.

### Sculpin Density and Size Distribution

Over three months, a total of 468 sculpin were sampled at the 12 sites (mean density = 0.75 ± 0.10 fish /m$^2$). Fish density was not significantly affected by canopy cover (Table 4). However, sculpin density did differ among streams and months, and there was also a significant interactive effect of month and canopy type on sculpin density. The density of sculpin in Bohemian Coulee Creek (1.05 ± 0.23 fish/m$^2$) was greater than that in Rullands Coulee Creek (0.87 ± 0.11 fish/ $m^2$) or Timber Coulee Creek (0.33 ± 0.04 fish/ $m^2$). Fish density increased during the summer and ranged from 0.55 ± 0.09 fish/ $m^2$ in June to 0.95 ± 0.21 fish/ $m^2$ in August. Sculpin density remained relatively constant at closed canopy sites during the summer with means ranging from 0.57 ± 0.11 to 0.73 ± 0.18 sculpin/$m^2$ (Figure 3). In contrast, sculpin density at open canopy sites increased from 0.42 ± 0.09 fish/ $m^2$ in June to 1.19 ± 0.38 fish/ $m^2$ in August.

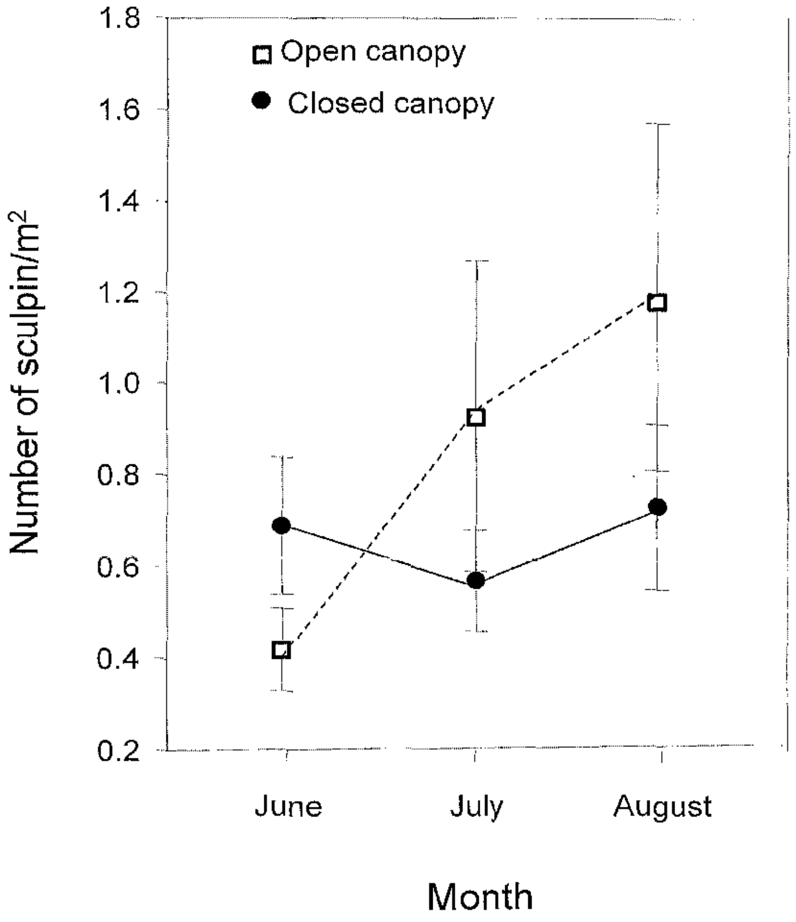

Figure 3. Density of sculpin in open- and closed-canopy sites in June, July, and August 2009.

Table 4. Results of repeated-measures analysis of variance for effects of canopy type, stream, and month on density of sculpin.

| Source | df | F | *P* |
|---|---|---|---|
| Main Plot | | | |
| Canopy (C) | 1 | 0.07 | 0.794 |
| Stream (S) | 2 | 17.31 | 0.003 |
| C x S | 2 | 2.24 | 0.187 |
| Error | 6 | | |
| Subplot | | | |
| Month (M) | 2 | 5.11 | 0.025 |
| M x C | 2 | 4.54 | 0.034 |
| M x S | 4 | 1.06 | 0.416 |
| M x C x S | 4 | 2.67 | 0.084 |
| Error | 12 | | |

The total length of sculpin ranged from 1.5 cm to 10 cm and the size distribution of sculpin varied between canopy types. There was not a significant difference in the size distribution of sculpin between open-and closed-canopy sites in June (KSa = 1.104, p = 0.174, Figure 3A), but there was a significant difference in July (KSa = 3.303, $p < 0.00001$, Figure 4B) and in August (KSa = 1.962, p = 0.0009, Figure 4C). In June, sculpin at open-canopy and closed-canopy were generally 4 to 8 cm long. However, in July sculpin 2 to 3 cm long comprised more than 50% of the population at open-canopy sites, whereas sculpin 4.5 to 7.5 cm comprised most of the population at closed-canopy sites. Similarly, in August small sculpin comprised most of the population at open-canopy sites, while large sculpin comprised a greater percentage of the population at closed-canopy than at open-canopy sites.

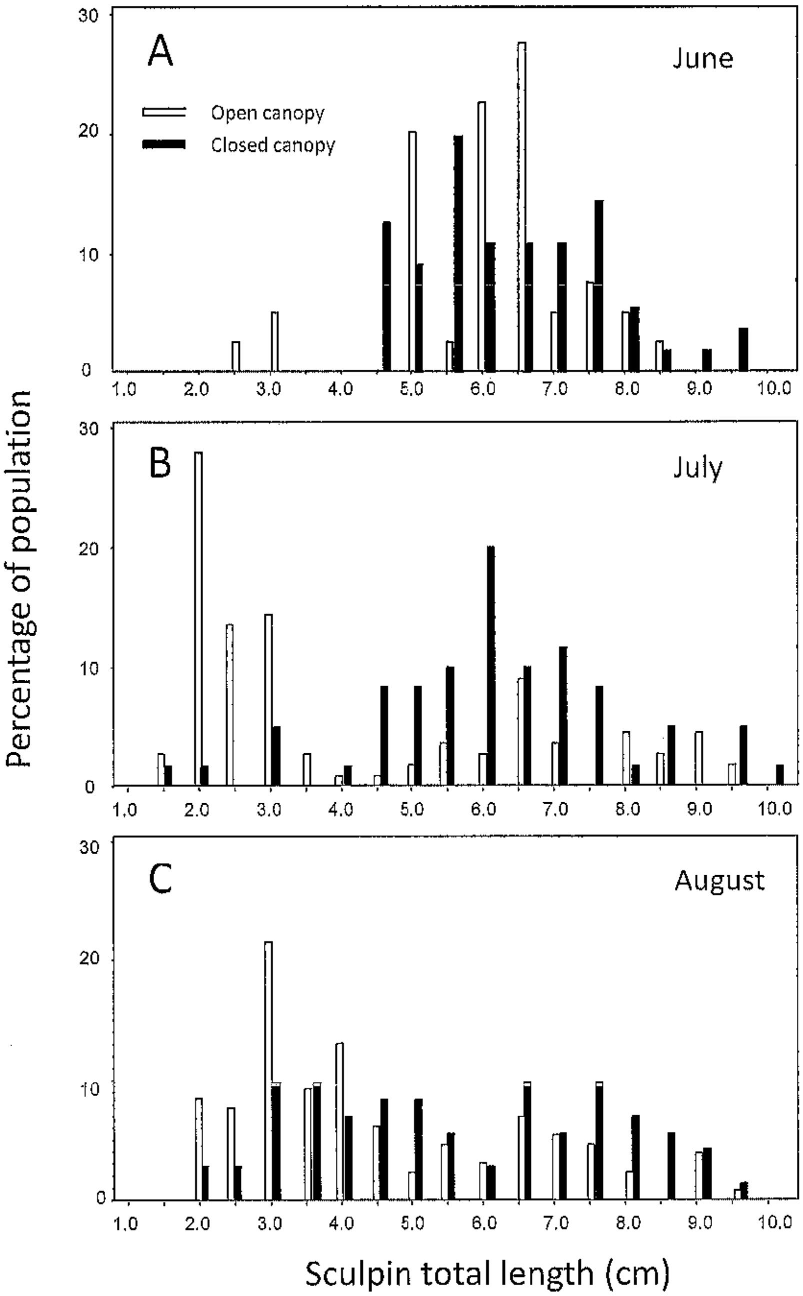

Figure 4. Size-frequency (percentage of population) of sculpin in (A) June, (B) July, and (C) August 2009 at open- and closed-canopy sites.

The size of sculpin at each site were related to a number of factors. Sculpin size decreased with increasing sculpin density at both open- (Regression: df= 1, 16, $R^2 = 0.656$, $p<0.0001$) and closed- (df= 1, 16, $R^2= 0.240$, $p=0.039$) canopy sites. At closed-canopy sites, sculpin size was influenced by stream depth (df = 1, 16, $R^2 = 0.172$, $p < 0.0001$) and water velocity (df= 1, 16, $R^2 = 0.361$ $p < 0.0001$). Larger sculpin tended to inhabit deep areas with swifter currents. There was no significant correlation between site-specific factors and sculpin size in open- canopy sites.

# DISCUSSION

Riparian canopy cover did not affect the density of slimy sculpin, as originally hypothesized. However, canopy cover did significantly impact the size-distribution of sculpin. In this study, I observed that small, juvenile slimy sculpin were most prevalent in riffles with extensive beds of macrophytes that were established in July and August at open-canopy sites. To my knowledge, slimy sculpin occupation of macrophyte beds in streams is undocumented elsewhere in the literature. This was unexpected given the large body of research suggesting sculpin are tightly linked to gravel and cobble-substrate habitats for life history processes, food acquisition, and protection from predation (Edwards and Cunjak 2007; Haro and Brusven 1994; Keeler and Cunjak 2007; Petrosky and Waters 1975). Under experimental conditions, sculpin have been shown to select pebbles and cobbles instead of macrophytes, potentially due to higher foraging efficiency on bare substrata (Welton et al. 1983).

Initially, I hypothesized that the biomass of invertebrate prey was a driving force in shaping the distribution and densities of sculpin. Because of increased sunlight at open-canopy sites, I expected a greater biomass of periphyton and invertebrates to be found at open-canopy sites, which would subsequently result in greater densities of sculpin at open- canopy sites than at closed-canopy sites. Riparian vegetation has been shown by previous studies to impact primary production within stream systems, as well as benthic macroinvertebrate communities (Hetrick et al. 1998). However, in this study, the amount of periphyton was not significantly different between riffles with open- and closed- canopies.

Several factors may explain the indiscernible differences in periphyton biomass between the two canopy types. Firstly, while AFDM is frequently used to estimate periphyton biomass, it is important to note that this methodology is a measurement of both the photosynthetic biomass as well as the heterotrophic biomass of periphyton. Therefore, if the heterotrophic community (bacteria, fungi) was substantial, differences in photosynthetic biomass between open- and closed-canopy sites may have been masked. Secondly, the presence of strong invertebrate grazers, such as snails, and the caddisfly *Glossossoma* spp., may inhibit the growth of periphyton (Steinman 1996). In the case of this study, substantial populations of *Glossossoma* sp. were present at all study sites. While *Glossossoma* sp. densities were not found to be greater at open-canopy sites than closed-canopy sites, due to their sheer numbers, *Glossossoma* sp. likely impacted periphyton growth at both open- and closed- canopy sites. Lastly, and perhaps most notably, temperate, fresh-water streams exhibit heterogeneous characteristics due to variable physical and biological factors (Petty and Grossman 1996, Thompson et al. 2001; Petty and Grossman 2010). For example, physical factors such as nutrients, temperature, substrate type and flow regimes are variable within streams, but also vary at a smaller scale, in this case, within riffles. A similar study conducted by Inoue and Nunokawa (2005) measured periphyton biomass (as AFDM), invertebrate dry mass, and sculpin density in Japanese meadow streams and forested streams. Similarly, the researchers found no difference in the biomass of periphyton between canopy treatments, and attributed their results to site-specific factors influencing primary production, including nutrient availability, grazing pressures, substrate stability and flow regimes. Consequently, neither invertebrate dry mass nor sculpin density were found to differ between open meadow streams and forested streams.

Likewise, the periphyton biomass in this study did not appear to directly, or clearly influence invertebrate biomass via a cascade effect, as originally hypothesized. While there were marked differences in the invertebrate communities between streams and by month, the overall invertebrate food availability (standing crop), and sum of invertebrate densities were not significantly different between canopy types. Nor were there significant differences in individual taxon biomass, with the exception of three taxa: *Baetis* sp., *Limnephilidae* and *Hydroptilidae*. However, neither *Limnephilidae*, nor *Hydroptilidae* were found in sculpin diet; therefore, neither taxa likely contributed to sculpin habitat choice.

*Baetis* sp. however, did contribute substantially to sculpin diet, particularly at open- canopy sites where sculpin selected for the mayfly nymphs. As herbivores, *Baetis* spp. often congregate in high numbers where periphyton abundance is greatest (Behmer and Hawkins 1985, Richards and Minshall 1988). In their evaluation of the effects of canopy cover on invertebrate abundance, Richards and Minshall (1988) found that *Baetis* species occupied open stream reaches significantly more often than in shaded reaches. This conclusion agrees with my own findings that densities of *Baetids* were most abundant in open-canopy sites, and may be a contributing factor to sculpin habitat selection. Given that small and juvenile sculpin were more commonly found in open-canopy sites, it may be logical to conclude that there is a preference for *Baetis* sp. by small-bodied sculpin. However, it also remains possible that sculpin selected *Baetis* sp.

more commonly in open- canopy sites by default due to *Baetis'* greater occupancy of open- canopy sites. Either way, it is difficult to draw a final conclusion based on the methodology used in this experiment since sculpin gut contents were pooled together at each study site. Therefore, I did not track the size of individual sculpin in relation to the species found in their gut.

While *Baetis* sp. constituted a significant percentage of prey items found in sculpin diet, *Simulium* sp. larva was the most dominant invertebrate collected from sculpin diet at both open- and closed- canopy sites. In addition, *Simulium* sp. and *Gammarus pseudolimneaus* were also selected for by sculpin at both canopy types. Unexpectedly, the riffle beetle, *Optioservus* sp. (larva) was selected for at closed-canopy sites. *Optioservus* sp. beetles are considered to be herbivore-detritivores and scrape algae and detritus from hard surfaces (Seagle 1982), and would plausibly be found in shaded reaches. While *Optioservus'* presence in sculpin diet should be noted, sculpin consumed only 4 individuals total at closed-canopy sites. Therefore, the importance of *Optioservus'* influencing sculpin distribution is likely limited.

While differences in the invertebrate populations are interesting to note, they alone likely do not account for sculpin habitat choice. Previous research on sculpin microhabitat choice have shown that sculpin select patches with the greatest macroinvertebrate abundance, biomass and accessibility (Petty and Grossman 1996, 2010). In this study, I found no evidence that the overall invertebrate community differed in abundance (standing crop) or by density in open- and closed- canopy sites. Rather, the Coon Valley watershed system is highly productive, and it is unlikely that sculpin chose habitat based on the presence of a single invertebrate taxon on a macro-scale. Therefore, the results of this study suggest that the quantity of photosynthetically active radiation may not influence the density of sculpin through the enhancement of periphyton, and subsequent increases in macroinvertebrate prey. However, several other factors may explain why large sculpin were found more frequently in closed-canopy sites, while small and juvenile sculpin were more prevalent at open- canopy sites.

Likely factors contributing to slimy sculpin population distribution (or size discrimination) include intraspecific competition, physical habitat characteristics, and predation avoidance. For example, large-bodied sculpin have been shown to display aggressive behavior toward juvenile and small co-specifics (Davey et al. 2005; Freeman and Stouder 1989; Grossman et al. 2006), indicating intraspecific competition may play an important role in habitat acquisition.

In addition to aggressive behavior, competition for ideal substrates may contribute to the habitat discrimination amongst different age-classes in the Coon Creek Watershed. Because larger substrates may be a sought-after attribute for protection from predation and nesting (Balon 1975), juvenile sculpin may be forced out of large substrate habitats by aggressively dominant individuals. While substrate size was not significantly different between canopy cover treatments in this study of the Coon Creek watershed, I did find that juvenile sculpin occupied the littoral zones of the open-canopy sites where macrophytes were prevalent. Similarly, juvenile bullhead sculpin (*Cottus gobio*) in New Brunswick streams were shown to more frequently select cover in the form of macrophytes than adult sculpin (Davey et al. 2005). Adult mottled sculpin in southern Appalachian streams displayed territorial behavior, and established habitats that were more geologically stable than surrounding microhabitats. Contrastingly, juvenile mottled sculpin were found to occupy less-stable, depositional habitats, and did not display territorial behavior (Petty and Grossman 2007).

In addition to intraspecific competition, physical factors, such as stream depth, discharge, velocity, substrate size, and temperature have all been shown to impact sculpin populations (Edwards and Cunjak 2007, Wehrly et al. 2003) , and likely influence slimy sculpin habitat choice in the Coon Creek watershed as well. For example, based on my results, larger sculpin were more frequently associated with greater wetted widths, while stream depth may be linked to reduced predation for small-bodied individuals. Sweeney (1993) noted that streamside forests tend to have wider reaches than open meadow streams, thus our findings may indicate a combination of co-factors contributing to sculpin size distribution.

Perhaps most significant among the physical factors impacting slimy sculpin, was the temperature differences between open-and closed-canopy sites. While temperatures fluctuated substantially at different times of day among all sites, open-canopy sites had greater temperatures for all three streams. In addition, the range of temperature fluctuation was significantly greater in open-canopy sites compared to closed-canopy sites. Despite temperature variation however, all temperatures remained well- below the physiological limits of slimy sculpin tolerance, and dissolved oxygen did not significantly vary by canopy type. Due to the heterogenous nature of streams, it is difficult to pinpoint the exact causal effects that such physical attributes may have on fish populations. Thus, such factors are worth further exploration in future studies.

In addition to intraspecific competition and physical characteristics influencing sculpin habitat occupation, avoidance of predation appears to be the strongest explanation for size segregation amongst this sculpin population. In this study, the highest proportion of juvenile sculpin occurred at sites with substantial macrophyte growth, where they were visually inaccessible to both piscivorous fish as well as terrestrial predators, namely birds. Davey et al. (2005) observed that bullhead sculpin showed preference for cover in the form of macrophytes, and attributed their unexpected findings to bullhead sculpin's pursuit of visual isolation from predators. Hyslop (1982) noted that 0+ juvenile bullhead sculpin inhabit beds of *Ranunculus fluitans* and *Ranunculus aquatilis* during summer months, which Davey et al. (2005) propose is likely due to protection from predators. Furthermore, I also found a correlation between stream depth and sculpin size, where juvenile and small-bodied sculpin were more frequently found in deeper habitats (associated with open-canopy sites). Harvey and Stewart (1991) found that predation risk for a multitude of fish species was reduced in deeper aquatic habitats than in shallow pools. Therefore, in addition to macrophyte cover reducing predation risk for juveniles, stream depth may also contribute to habitat choice.

In addition to reducing competition and predation risk, beds of macrophytes may reduce energetic costs associated with stream discharge that may be encountered by sculpin. While I measured water velocity across six points within each riffle, macrophytes occurred in patches and differences in water velocity among microhabitats may not have been adequately measured. Champion and Tanner (2000) showed that macrophyte beds in streams significantly impact stream velocities during summer months, and in turn, were an important structuring mechanism for streams. Due to reduced water velocity within macrophyte beds, Champion and Tanner suggest that such beds act as semi-permeable dams, providing increased habitat heterogeneity for aquatic species. Therefore, despite some of the costs of occupying macrophyte beds (presumably reduced food availability), juvenile sculpin may benefit by exerting less energy to maintain position in addition to avoiding predation.

While little is known concerning ontogenetic shifts in microhabitat amongst aquatic benthic fish (Davey 2005), there have been a large number of studies in other fields of ecology exploring this phenomenon (Brown 1999; Brown and Kotler 2004; Lima and Dill 1990; Polivka 2007). For example, immature backswimmers (*Notonecta* sp.) shift their feeding habitat in the presence of larger, cannibalistic backswimmers (Sih 1982). In addition, juvenile perch (*Perca fluviatilis*) had reduced encounter rates with large predators in littoral zones of rivers than in pelagic zones (Byström et al. 2003).

Predation is an important factor contributing to habitat selection, particularly when more than one predator species is present (Power 1987). Furthermore, ontogenetic habitat shifts may be reflective of habitat preferences based on changing susceptibility to predators with respect to body size (Davey et al. 2005; Persson and Greenberg 1990; Schlosser 1987). While many models have been created and used to predict foraging behavior in response to the costs of predation, different animal species undoubtedly vary considerably with foraging effort and "risky" behavior. Juvenile species are susceptible to predation due to their small size and reduced mobility and/or speed. In the case of juvenile slimy sculpin, it is likely that the costs of predation in the Coon Creek watershed outweigh the energetic benefits of occupying deeper, more exposed habitats.

# CONCLUSIONS

Stream systems throughout the United States are continually being altered by removal of riparian vegetation. While traditionally the removal of forest surrounding streams has been viewed as detrimental to native species, such alterations may serve an ecologically important role, whereby species occurrence and interactions can be studied. Reduced overhead canopy cover allows for greater photosynthesis to reach the stream bed, potentially increasing habitat opportunity due to increased heterogeneous resource availability for many aquatic organisms. In this study, increased sunlight to the stream bed did not directly appear to influence periphyton or macroinvertebrate biomass, but did result in the growth of large macrophyte beds at open-canopy sites, wherein juvenile sculpin were found to inhabit. The discovery that juvenile slimy sculpin use macrophyte beds for habitat is noteworthy in that it has been undocumented elsewhere. Use of macrophyte beds likely serves a critical role in habitat selection for juvenile sculpin by reducing interaction with more dominant individuals, reducing encounters with predators, and reducing energetic costs associated with higher velocity habitats.

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