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Habitat Suitability Criteria for Fishes of the South Fork of the Shenandoah River and an Investigation into Observer Effects Associated with Two Techniques of Direct Underwater Observation

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A thesis submitted in partial fulfillment of the requirements for the degree of Master of Science at Virginia Commonwealth University.

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Acknowledgment

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Abstract

HABITAT SUITABILITY CRITERIA FOR FISHES OF THE SOUTH FORK OF THE SHENANDOAH RIVER AND AN INVESTIGATION INTO OBSERVER EFFECTS ASSOCIATED WITH TWO TECHNIQUES OF DIRECT UNDERWATER OBSERVATION.

By Robert Clayton Ramey, M.S.

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This study constructed habitat suitability criteria for fishes of South Fork of the Shenandoah River, in Virginia. The criteria will be used in an IFIM study to produce estimates of the discharge required by fishes in the South Fork. Chi-square tests were used to evaluate whether criteria described habitat use to a statistically significant degree. Secondly, chi-square tests were used to test transferability. The criteria described the habitat use of seven taxa commonly found in the South Fork to a statistically significant degree. Habitat criteria for two taxa did not describe their habitat use to a statistically significant degree. One set of criteria from the North Fork of the Shenandoah transferred to the fish observed in the South Fork.

Secondly, this paper examined observer effects of underwater observation. It was of interest to explore how observer effects influenced habitat suitability criteria.

Introduction to Thesis Project

Anthropogenic demands and drought limit the availability of surface and groundwater for use by stream ecosystems. Water users in Virginia for example used an estimated 1,408 million gallons of freshwater a day (MGD) in 2007 (DEQ 2008). Of the total, approximately 1,206 MGD was surface water and 784 MGD of that was stream water (DEQ 2008). While drought is obviously not subject to regulatory oversight, water withdraws for municipal and commercial uses are (DEQ 2009). With stress on water resources high and rising, to the extent that stream biodiversity and the viability of native species have been impacted (Minckley and Deacon 1991) not to mention human well-being (UNEP 2007), resource managers need tools that allow them to make informed and biologically based decisions to protect the integrity of stream ecosystems.

The Instream Flow Incremental Methodology (IFIM) is one such tool (Bovee 1986; DEQ 2008). The Physical Habitat Simulation model (PHABSIM) is the component of an IFIM analysis that attempts to link habitat criteria for biota to increments of discharge (Bovee 1986). The products of PHABSIM are thus estimates of the area of usable habitat available to biota at each simulated discharge (Krstolic and Hayes 2004). The estimates of useable area that PHABSIM produces are based on habitat suitability criteria for the target organisms (Bovee et al. 1998). Habitat suitability criteria have been developed for aquatic macroinvertebrates (Gore et al. 2001), fish (Aadland et al. 1991; Strakosh et al. 2003), and even canoe paddlers (Krstolic and Hayes 2004). Habitat criteria for fish are generally based on depth, velocity, substrate sizes,

embeddedness, and some measure of cover (Bovee et al. 1998). Habitat suitability criteria are often univariate, accounting for only one habitat parameter at a time, while other methods for developing criteria are multivariate (Thomas and Bovee 1993).

Habitat criteria for fish are generally constructed on the level of the species though in some instances genus are used (Bovee 1986). Criteria based on habitat guilds (Leonard and Orth 1988; Vadas and Orth 2001) have been sought as a way to economize data collection.

There are several types of habitat criteria. Criteria based on the literature and professional opinion are known as category I criteria (Bovee et al. 1998). Category II, or use, criteria produce univariate curves that illustrate the relative suitability of ranges of habitat parameters and are based on observations of fish habitat use obtained in the field (Bovee et al. 1998). To gather the data needed to assemble use curves, undisturbed fish are observed or sampled and the physical habitat parameters of the microhabitats that the sampled fish occupied are measured. Category III preference curves are based on habitat use observations, though random measurements are also obtained from the stream to provide a dataset that describes the habitats available (Bovee et al. 1998). A preference index is then derived that can suggest habitat preference based on frequencies of use disproportionate to availability. Univariate curves are also produced from preference criteria that plot preference index scores on the y-axis and the habitat variable along the x-axis.

One-sided chi-square (X^2) tests have also been used to test habitat suitability criteria (Thomas and Bovee 1993; Freeman et al. 1997). These tests provide several advantages to curve-based criteria. Among those advantages are that statistically based

X^2 tests allow for decisive comparisons of criteria developed by other researchers or from different streams, and tests can be multivariate. Chi-square tests can also be used to test the defensibility of a given set of criteria (Bovee 2008).

The goal of each of these types of criteria is to estimate the quality and quantity of habitat available at specific ranges of discharge (Bovee 1986). Once habitat criteria are established, the range of discharge required to maintain the fish community can be estimated. Developing criteria for as broad a subset of the community as feasible can improve the odds that recommended discharges account for the habitat needs of that community (Bovee et al. 1998).

An important question is whether habitat criteria are consistent within species throughout their range or at least within regions. If this were the case, the costly process of determining site-specific criteria could be avoided and resource managers could quickly evaluate the potential ecological effects of different stream water withdrawal scenarios. So called "transferable" criteria, as in transferable from one stream to another, would also be inferred to be accurate and biologically significant (Freeman et al. 1997) if they could consistently predict habitat use. Tests of transferability have had mixed results. Different researchers classify the quality of habitats in different ways and comparisons of use and preference curves developed in different streams or by different researchers are notoriously imprecise and arbitrary and therefore unreliable. Different rivers can have different available habitats and different ecological contexts i.e. inter- and intraspecific competition, predation, or food sources, any of which could result in differences in habitat selection and the failure of habitat criteria to transfer. It is also possible that factors not included in conventional habitat suitability criteria could have a

significant influence on habitat use or that influential drivers are not discernable from point measurements of physical habitat (Freeman et al. 1997).

It is also possible that the techniques used to gather the data used to create habitat suitability criteria affect those criteria. It has been shown that different sampling methods can produce different habitat criteria within the same species and river (Persinger et al. 2004). Responses that observers provoke in the behavior of their subjects are known as observer effects. If observer effects are substantial enough, they could compromise the accuracy and thus the transferability of habitat criteria. Observer effects associated with a particular sampling protocol that inadvertently pushed fish into deeper water would produce habitat criteria that do not accurately reflect the habitat use of those fish and inaccurate habitat suitability criteria would not be expected to transfer (Freeman et al. 1997). Secondly, observer effects can influence the frequency with which specific species are or are not sampled as different species respond differently to given stimuli (Thresher and Gunn 1986). When the goal of an IFIM analysis is to account for a broad swath of a diverse fish community, a combination of sampling tools and techniques is the preferred way to gather data to construct habitat criteria so that the sampling biases of each method are compensated for by the strengths of the other (Bovee et al. 1998; Persinger et al. 2004). There are explorations of observer effects and other sampling biases in the literature (Samoilys and Carlos 2000; Walsh et al. 2002), though few address those associated with arguably the most common method of habitat use data collection, direct underwater observation (Thurow 1994). Direct underwater observation has been compared to other sampling tools (Mullner et al. 1998; Persinger et al. 2004),

but rarely have variations of direct underwater observation been tested against one another, particularly in the context of warmwater streams.

Unknowns about the observer effects associated with direct underwater observation protocols thus left open the possibility that observer effects could be contributing factors in the failure of habitat suitability criteria developed from data collected via snorkeling to transfer between rivers. Part I of this thesis document describes the processes and results of statistical testing of the hypothesis that two types of underwater observers would each produce habitat suitability criteria that were not different from those produced by the other type of observer.

The North and South Forks of the Shenandoah River in the Valley and Ridge physiographic province of Virginia (Krstolic and Hayes 2004) have been and are currently subjects of an IFIM investigation by the U.S. Geological Survey Water Science Center (USGS) in Richmond, Virginia. Data collection for a pending IFIM analysis for the South Fork of the Shenandoah (South Fork) also presented a unique opportunity to test the concept of transferability.

The North and South Forks flow north-northeast, along the west and east flanks of Massanutten Mountain respectively, in the northwest portion of Virginia, and join at Front Royal, to become the main stem of the Shenandoah River. At Harper's Ferry, West Virginia, the main stem joins the Potomac River which continues into the Chesapeake Bay. Increasing human demands for surface water and reoccurring and persistent drought led to the initiation of instream flow studies on the Shenandoah River system. A pilot study was conducted on the Main Stem to establish the utility of IFIM modeling in the Shenandoah (Zappia and Hayes 1998), and an IFIM analysis was published for the

North Fork in 2006 (Krstolic et al. 2006). Because of the physical and historical similarities between the North and South Forks and because of the similarity of their fish communities, it was suspected that if transferable habitat criteria are a realistic possibility, they would be demonstrable in the North and South Forks of the Shenandoah River.

There are several obstacles to determining if a given set of habitat suitability criteria, based on data collected in a source stream, are transferable to conspecifics in other destination streams. Controlling independent variables in transferability tests is difficult if not impossible because of the number of factors that influence fish habitat use including competition among species (Gunckel et al. 2002), predation (Power and Matthews 1983), competition among conspecifics (Petty and Grossman 2007), the time of day (Banish et al. 2008), the season (Minckley and Deacon 1991), and mesohabitat (Vadas and Orth 2000) and microhabitat characteristics of specific rivers. The North and South Forks are uniquely qualified for hosting transferability tests because their degree of similarity in effect controls for, albeit incompletely, many if not all of the factors suggested above. Part II of this thesis document contains a description of the processes involved in and the results of a study to assemble habitat suitability criteria for fishes of the South Fork of the Shenandoah River. Transferability tests were conducted to test the applicability of habitat suitability criteria developed by other researchers to the fish observed during this study.

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Part I:

Comparisons of fish habitat use data collected by roving and stationary underwater observers to identify observer effects

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Introduction to Observer Comparisons

An "observer effect" describes the tendency for the presence of an observer to influence the behavior of the observed. Observer effects have been demonstrated in semaphore crabs (Macfarlane and King 2002), stream invertebrates (Baker and McGuffin 2007), song birds (Fletcher et al. 2000), and fish (Chapman et al. 1974; Kulbiki 1998). The nature or severity of observer effects on fish behavior can be influenced by underwater visibility (Peterson et al. 2005) and by the temperament of specific taxa. Some fishes exhibit observer effects by approaching observers while other species flee (Kulbicki 1998; De Girolamo and Mazzoldi 2001), either of which can influence the accuracy of density estimates, for example (Byerly and Bechtol 2005).

If a researcher is pursuing a specific species with known behavioral tendencies, an observation protocol can probably be found that is particularly well suited to that species (Thresher and Gunn 1986; Thurow and Schill 1996). The difficulty of selecting a sampling technique or protocol is increased in the context of a diverse fish community that includes fishes with a variety of behavioral adaptations and patchy distributions (Samoilys and Carlos 2000). In trying to sample or observe many members of the community, by definition, a protocol that samples some species most effectively does not sample others with the same precision, accuracy, or efficiency (Samoilys and Carlos 2000). Different observation protocols might for example produce different estimates of relative abundance over repeated measurements at the same locations (Thresher and Gunn 1986) as a result of their respective observer effects.

Investigations of fish habitat based on empirical data rely on an assumed ability to collect data from fish whose habitat use is unaffected by the presence of an observer (Bovee 1986). Habitat criteria for fish are critical components of the Instream Flow Incremental Methodology (IFIM), an analytical tool available to resource mangers faced with water allocation decisions (Bovee 1986; Bovee et al. 1998). The Physical Habitat Simulation model (PHABSIM) is the component of an IFIM analysis that links the habitat needs of biota, as defined by habitat suitability criteria, to stream discharge (Bovee 1986). The output of PHABSIM provides resource managers with estimates of the area of habitat available for the target fishes at increments of discharge (Bovee 1986; Krstolic et al. 2006). The accuracy and biological relevance of PHABSIM estimates are only as credible as the habitat criteria used in the model (Freeman et al. 1997; Beecher et al. 2002).

Direct underwater observation is commonly used to gather the data needed to construct habitat criteria for fish (Thurow 1994; Bovee et al. 1998). Among the advantages of direct underwater observation is that it is considered to be minimally disruptive, facilitating observation of undisturbed fish. Secondly, observers are able to determine the locations of fish precisely (Thurow 1994) which is required to define habitat criteria at a microscale (Bovee et al. 1998).

There are different strategies of underwater observation and there are observer effects associated with each of them (Heggenes et al.1990). Though the biases associated with underwater observation may be minimal in comparison with those of other methods, those biases remain to some extent. Methods for underwater observation can be subdivided into roving and stationary techniques. The most common technique utilizes a

snorkeler or SCUBA diver, or a team of several (Schill and Griffith 1984), swimming

upstream, observing the entire width and depth of the stream or following a randomly

placed study lane in a larger stream (Bovee et al. 1998). This paper will refer to these

swimming observers as "roving observers". Roving observers have been used in

relatively clear and fast streams to sample salmonids (Thurow and Schill 1996; Strakosh

et al. 2003) in depauperate systems. Underwater visibility can approach several meters in

typical salmonid stream habitats which facilitates underwater observation (Goldstein

1978; Mullner et al. 1998). Salmonids in high water velocities can be approached from

downstream by a slowly approaching observer (Heggenes et al. 1990), probably as a

consequence of their drift feeding behavior (Jenkins and Burkhead 1993), as their

attention tends to be focused into the water column ahead of them and the fast currents

and turbulence of the microhabitats that they hold station in obscure disturbances that

originate from outside the immediate area. Observers are encouraged to move slowly so

as to minimize spooking fish (Thurow 1994). It is thus the movement of the observers

that is presumed to create the most disturbance rather then their mere presence (Thurow

1994).

Though roving observers have been used in diverse warmwater streams (Leonard

and Orth 1988; Lobb and Orth 1991), the observer effects of doing so have not been

examined, with the exception of a paper that compared the observer effects of a roving

observer to those of throwable anode electrofishing (Persinger et al. 2004). Sampling

biases associated with different types of fish sampling equipment and techniques have

been discussed (Walsh et al. 2002; Peterson et al. 2005), as have the observer effects of

roving versus stationary observers on density (Thresher and Gunn 1986; Byerly and

Bechtol 2005) and relative abundance estimates (Samoilys and Carlos 2000) of reef fish, and the influence of the speed of roving observers has been explored (Lincoln-Smith 1988; De Girolamo and Mazzoldi 2001). Few studies regarding the observer effects of roving observers on the development of fish habitat suitability criteria have been published. Although using several types of gear or techniques to sample a diverse fish community is often the ideal way to sample many species (Persinger et al. 2004), this paper is concerned only with exploring the observer effects of different snorkeling protocols.

This study used direct underwater observation to gather fish habitat use data. A roving observer followed a transect sampling protocol (Bovee 1986) and a stationary observer recorded the habitat use of fish in randomly selected microhabitats over 15 minute intervals. This paper sought to identify observer effects through statistical comparisons of the fish habitat use data gathered by roving and stationary observers. Specific comparisons included comparisons of individual and event sample sizes, the number of individual fish per sampling event, taxa and life stage richness, the relative abundance of specific taxa, and differences in fish behavior observed by each type of observer. Finally, the distributions of each observer's fish habitat use observations were statistically compared as a proxy for the habitat suitability criteria that could be derived from each observer's data.

Study Area

Fish habitat use data were collected in the South Fork of the Shenandoah River (South Fork), in the Valley and Ridge physiogeographic province in northwest Virginia. The Shenandoah River system drains approximately 6,070 km^2 in Frederick, Clarke, Rockingham, Shenandoah, Augusta, Page, and Warren Counties in Virginia and Jefferson County, West Virginia (PWP 2009), and includes the South Fork, the North Fork of the Shenandoah River (North Fork), and the main stem of the Shenandoah River (main stem). The South Fork begins near Port Republic at the confluence of the North, South, and Middle Rivers. The South Fork flows north-northeast for approximately 167 km until its juncture with the North Fork near Front Royal, Virginia (DGIF 2009). The watershed of the South Fork is approximately 4,300 km^2, captures runoff from the Blue Ridge and Allegheny Mountains, and Massanutten Mountain, and ground water from the karst areas in the Shenandoah and Page Valleys (DGIF 2009). It is an unregulated stream, though there are three low-head hydropower dams in Shenandoah, Virginia, Newport, Virginia, and Luray, Virginia (DGIF 2009). The South Fork is a 5th order stream, and bedrock run habitat constitutes 34% of its length; natural pools, particle runs, bedrock riffles, and artificial pools make up 21, 18, 7, and 7% of that length, respectively (Krstolic, personal communication 2009; see Table 1). The South Fork had an average annual discharge of 38 m^3/sec based on USGS records from 1926 to 2008 from stream gages at Lynnwood, Virginia, Luray, and Front Royal.

The North Fork is also a 5th order stream and flows approximately 172 km, parallel to the South Fork in the Shenandoah Valley floor along the western edge of Massanutten Mountain, from Rockingham County to Front Royal, Virginia. The North Fork and has a watershed of approximately 2,675 km^2 (Krstolic et al. 2006). The average annual discharge of the North Fork was approximately11 m^3/sec from 1926 to 2008 based on records from USGS stream gages at Cootes Store, Virginia, Mt. Jackson, Virginia, and Strasburg, Virginia.

Downstream of Front Royal, the North and South Forks join to form the Main Stem. The Main Stem joins the Potomac River in Harper's Ferry, West Virginia, which flows into the Chesapeake Bay.

The study reaches were located at established U.S. Geological Survey (USGS) sites. A table containing the station numbers, names, and coordinates of the USGS sites sampled and the nearest river gages is contained in Appendix I. Each study reach contained multiple transects (3 to 5), each transect at a discrete mesohabitat type. Specific transects to be sampled at each study reach were stratified by mesohabitat type and randomly selected in proportion to the relative abundance of each habitat type. A single study reach could thus contain more than one transect sampled. As field work progressed, established USGS sites were exhausted and two additional sites were selected on the basis of their mesohabitat type and accessibility. Six transects at five study reaches were sampled by both types of observer.

Table 1. Summary of Relative Frequencies of Mesohabitat Types in the North Fork and the South Fork Rivers, by Percentage of River Length

Mesohabitat Type	South Fork (%)	North Fork (%)
Artificial Pool	7	7
Backwater	5	2
Glide	3	na
Natural Pool	21	10
Bedrock Riffle	7	4
Particle Riffle	5	1
Pocket Run	2	2
Bedrock Run	34	35
Particle Run	18	31

North Fork mesohabitat data from Krstolic et al. 2006

South Fork mesohabitat data and table from Krstolic, personal communication, preliminary data

Methods

Data Collection

Data to construct habitat suitability criteria for as many species as feasible in the

South Fork were collected by a roving observer (Bovee 1986) and by a stationary

observer. Both methods used direct underwater observation (Goldstein, 1978; Thurow

1994) with snorkels to observe microhabitat use by fishes.

Following Bovee (1986), a roving observer swam six, 30 x 1 m lanes

perpendicular to the selected transects. Fish observed were identified, counted, and

recorded on an underwater note pad. Fish locations were marked with weighted markers

and physical habitat parameters were measured for a 1 m^2 area with the fish marker at its

center. Habitat measurements were also obtained every 6 m in each lane to account for

available habitat. The roving observer sampled 960 m^2.

A stationary observer recorded the taxa and the number of fish that occupied

randomly selected microhabitats over 15-minute intervals. Ten 2 x 1 m cells were

randomly selected from each transect with one cell at the rivers edge; the long axis of

each cell was parallel to the transect, perpendicular to the current. Two-meter cell widths

were observed by the stationary observer because that was a conservative estimate of

approximately twice the maximum distance at which the observer could identify all fishes

present. The stationary observer sampled 120 m^2.

Both types of observers flipped ten rocks in each sample cell and each sample

lane where feasible to look for cryptic taxa including *Cottus* spp. and *Noturus insignis*.

At the center of each cell and at each available habitat and fish marker, physical habitat variables were measured including depth and mean column velocity at 0.6 depth (Bovee 1986). Depths and velocities were measured either with a FlowTracker (Sontek) on a graduated top-setting wading rod or with a StreamPro Acoustic Doppler Current Profiler (ADCP). Depths and velocities were measured and recorded in feet, though reported here in cm and cm/s, respectively. Numerical codes for substrate size based on a modified Wentworth scale (Bovee 1986), cover, and embeddedness, were used so that substrate characterizations could be summarized in a single channel index (Krstolic et al. 2006). Where two or more cover types were present at a particular location, cover was characterized as "complex" in the channel index. Embeddedness was estimated visually by percent embedded (between 0 and 25% embedded, between 25 and 50%, between 50 and 75%, or between 75 and 100% embedded).

With few exceptions, fishes were identified to species. *Cyprinella spiloptera* and *C. analostana* were indistinguishable from one another while swimming and were recorded as *Cyprinella* spp. *Notropis rubellus* and *N. amoenus* were recorded as *Notropis* spp. As young-of-the-year (YOY) of all species were plausibly using specific habitat types, YOY were recorded as such without finer-scale distinction. Visually apparent life stages were recorded when feasible. Only division into juvenile, sub-adult, and adult was possible, and generally only for the species that attained the largest adult sizes. Conspecific life stages are noted as juvenile (J) or sub-adult (SA), and adults are referred to by a species abbreviation.

Fishes' vertical positions in the water column were noted because those locations represented measureable facets of fish behavior and were related to the nose velocities

used (Baker and Coon 1997). Vertical positions were recorded for individual fish as being at 0.2 depth (approximately the upper $1/3^{rd}$ of the water column), 0.6 depth (central $1/3^{rd}$), or 0.8 (lower $1/3^{rd}$; at, near, or within the substrate). It was also noted if fish moved from one vertical stratum to another. If vertical movements were observed in a particular individual, the vertical position the fish moved from, its holding position, was recorded, as was the vertical position it moved to, its feeding position; feeding being the presumed activity that was encouraging vertical movement.

Underwater visibility was considered adequate for sampling if an observer with mask and snorkel could see at least 1 m to each side and from the water's surface to the substrate. Conscientious efforts were made not to count the same fish repeatedly, not to count fish that were obviously attracted to the observer, nor to count fish that were using microhabitats outside of the boundaries of sample cells or lanes. The time spent observing cells and swimming lanes at each site was approximately equal.

Data Management and Analysis

The dataset that the methods comparisons were based on included only data collected by snorkeling from six transects at five sites where both field methods were used. This dataset is referred to as the comparison dataset. All statistical analyses were conducted using SAS's JMP 7.0, at $\alpha = 0.05$.

There are in effect, two sample sizes relevant to fish habitat data (Kinzie and Ford 1988). Single events where one or more conspecifics were observed are referred to as "events". Sample sizes referred to as "n (individual)" reflect the number of individual fish observed. Event sample sizes were used to select the appropriate statistical tests, although most of the actual analyses were conducted with individual sample sizes to

reflect the frequency that specific ranges of the habitat parameter in question were utilized. It was of interest, for example, to account for the fact that a single individual of a particular species was at a particular location with specific habitat parameter values, while multiple individuals of that same species were at a different location with different habitat parameter values. The assumption was that a location hosting many individuals may have more desirable habitat than a location where only a single individual was observed.

Wilcoxon/Kruskal-Wallis tests (Daniel 2005) were used to identify differences in sample sizes, the number of individual fish per event, the number of taxa and life stages observed, and differences in habitat use among conspecific life stages. Relative abundances of taxa and life stages sampled on at least five events by both the roving and stationary observers were compared graphically and statistically. The relative abundances of each taxa and life stage that each observer estimated were compared to determine if either observer systematically and differentially sampled specific taxa i.e. whether the observer effects associated with each observer resulted in a sampling bias. Relative abundances were compared for juvenile smallmouth bass (*Micropterus dolomieu*; J MDO), sub-adult smallmouth bass (SA MDO), sub-adult redbreast sunfish (*Lepomis auritus*; SA LAU), adult redbreast sunfish (LAU), *Cyprinella* spp., common shiners (*Luxillus cornutus*; LCO), margined madtoms (*Noturus insignis*; NIN), river chub (*Nocomis micropogon*; NMI), and YOY. The number of individuals of each taxa or life stage was divided by the area sampled to derive the number of individuals of each taxa or life stage per 10 m^2 sampled by each type of observer. The number of individuals of each taxa or life stage per 10 m^2 was divided by the total estimated number of individual fish

of all taxa and life stages per 10 m^2 sampled to generate estimates of relative abundance. The relative abundances estimated by the means just described were graphically compared because the point estimates of relative abundance for each taxon and lifestage and observer precluded statistical comparisons. Statistical comparisons via Wilcoxon/Kruskal-Wallis tests analyzed the relative abundance of each taxa and life stage estimated at each transect by each observer.

Wilcoxon/Kruskal-Wallis tests were also used to compare the distributions of fish observations recorded by each type of observer. The distributions of the most frequently sampled taxa and life stages were compared; the distributions of J and SA MDO, *Cyprinella* spp., NMI, and LCO. Distributions of fish observations throughout ranges of depths and mean column velocities were analyzed as proxies for the habitat suitability criteria that could be produced from the data collected by the roving and snorkeling observers. The tests comparing the distributions of fish observations are referred to as "habitat suitability criteria comparisons" in the results. Tukey-Kramer honestly significant difference (HSD) tests were used to locate specific differences in all tests (Daniel 2005). Because of the similarities in the channel indices that each type of observer identified, the observer comparisons were conducted based on depths and mean column velocities. The vertical orientations of fish in the water column (at 0.2, 0.6, or 0.8 depths) are reported where relevant.

Nose velocities at 0.8 depths were estimated for selected fishes observed at three morphologically similar riffle sites. The hydrologic conditions and the relationships between the velocities of each stratum were assumed to be reasonably uniform among the three riffle sites.

Results

In all of the data collected by each observer, there were no statistically significant differences in the depths sampled (p > 0.05). The mean depth sampled was 67.4 cm (SD = 46.9). The mean column velocities sampled by each observer were significantly different (2-sample test, normal approximation (S = 61919.5, Z = -3.7583, p = 0.0002), 1-way test, X^2 approximation (X^2 = 14.1265, df = 1, p = 0.0002)). A Tukey-Kramer HSD test confirmed that the roving observer sampled mean column velocities that were an average of 6 cm/s faster than those sampled by the stationary observer (95% CI on difference (2, 9); roving observer measured a mean column velocity of 27 cm/s (SD = 22) vs. the stationary observer's 22 cm/s (SD = 20)). The mean velocity sampled was 26 cm/s (SD = 22). Discharge was relatively low and consistent throughout the sampling period as is ideal for habitat criteria development (Bovee 1986). Table 2 contains a summary of discharge by month and by USGS stream gauge for the sampling period in the summer of 2008. The percentiles given are based on 46 to 77 years of USGS records. Discharge on the days spent in the field (n = 31) was representative of the discharges observed throughout the four month sampling period.

Table 3 contains a summary of the statistically significant differences from the observer comparisons, while Figures 1 and 2 contain plots of the data that were collected by each observer and compared.

Individual Sample Sizes

There was a statistically significant difference in the average number of individual fish observed per 10 m^2 by the roving observer and the stationary observer (2-sample test, normal approximation (S = 53, Z = 2.1617, p = 0.0306), 1-way test, chi square (X^2) approximation (X^2 = 5.0256, df = 1, p = 0.0250)). The stationary observer noted an average of 37 individual fish per 10 m^2 (SD = 17; 95% confidence interval (CI) (19, 55)), while the roving observer recorded an average of 10 individual fish per 10 m^2 (SD = 4; 95% CI 6, 14). A Tukey-Kramer HSD test confirmed that the stationary observer recorded an average of 27 more fish per 10 m^2 (95% CI on difference (11, 43)).

Event Sample Sizes

There was a statistically significant difference in the average number of sampling events per 10 m^2 recorded by each type of observer (2-sample test, normal approximation (S = 57, Z = 2.8022, p = 0.0051), 1-way test, X^2 approximation (X^2 = 8.3077, df = 1, p = 0.0039)). The stationary observer recorded an average of 11 events per 10 m^2 (SD = 4; 95% CI 6, 16), while the roving observer noted an average of 3 events per 10 m^2 (SD = 1; 95% CI (2, 3)). A Tukey-Kramer HSD test confirmed that the stationary observer recorded a mean of 8 more events per 10 m^2 sampled (95% CI on difference (4, 12)).

Number of Fish per Event

There were no statistically significant differences in the average number of individual fish observed at each event recorded by each type of observer (Wilcoxon-Kruskal/Wallis test, (2-sample test, normal approximation (S = 38, Z = -0.0806, p = 0.9357), 1-way test, X^2 approximation (X^2 = 0.0260, df = 1, p = 0.8719)).

Taxa and Life Stage Richness

There was a statistically significant difference in the average number of taxa and life stages sampled per 10 m^2 by each type of observer (2-way test, normal approximation (S = 57, Z = 2.8221, p = 0.0047), 1-way test, X^2 approximation (X^2 = 8.4255, df = 1, p = 0.0036)). The stationary observer recorded an average of 4 taxa and life stages per 10 m^2 (SD = 2; 95% CI 2, 6), while the roving observer recorded an average of 1 taxon or life stage per 10 m^2 (SD = 0; 95% CI 0, 1). A Tukey-Kramer HSD test confirmed that the stationary observer recorded 3 more taxa and life stages per 10 m^2 sampled than the roving observer (95% CI on difference (2, 5)).

Possible Observer Effect on the Relative Abundance of Specific Taxa

There were no statistically significant differences in the relative abundance of each taxon per 10 m^2 as estimated from the data collected by the roving and stationary observers (Wilcoxon/Kruskal-Wallis tests, all p-values > 0.05; Table 4. Graphical comparisons of the relative abundance of each taxon and life stage estimated by each observer did not suggest substantial differences, with the exception of disparity in the relative abundances estimated for YOY (Figure 3). Table 5 provides a summary of the data displayed in Figure 3. The relative abundance values included in Table 5 do not sum to 100% because all species sampled were included in the calculation of relative abundance, while Table 5 reflects the relative abundance of only the most frequently sampled fishes.

Possible Observer Effect on Fish Behavior and Habitat Use

Because it is not known to what extent the fish observed in this study altered their behavior or habitat use in the presence of an observer, this study has only behaviors noted by one observer and not the other to draw conclusions from. Only the stationary observer

noted vertical feeding behavior. The stationary observer recorded *Cyprinella* spp. holding at 0.8 depths and feeding at 0.6 depths on 8 out of 16 events, representing 63% of the individuals observed (41 out of 65). In the data collected by both observers, *Cyprinella* spp. were sampled on 49 events in riffles (out of 61 total sampling events). Nose velocities at 0.8 depth at the riffle sites were estimated using an equation derived from a linear regression of a positive linear relationship between measured velocities at 0.6 depth and measured velocities at 0.8 depth obtained from one of the riffle sites (n = 162; $V_{0.8} = -0.1297 + 0.6698*V_{0.6}$, $r^2 = 0.56$). Estimates of velocities at 0.2 depth were derived from the equation $V_{0.2} = 0.1911 + 1.1432*V_{0.6}$, $r^2 = 0.85$ (n = 162) that described the positive linear relationship between velocities at 0.6 and at 0.2 depths. The estimated mean nose velocity for *Cyprinella* spp. at 0.8 depth at the three riffle sites was 17 cm/s (95% CI (12, 23)). The mean column velocity at 0.6 depth used by *Cyprinella* spp. at the three riffle sites was 29 cm/s (95% CI (22, 37)). These estimates suggest that at the three riffle sites *Cyprinella* spp. were foraging in velocities that were on average 12 cm/s faster than the velocities they were holding in (95% CI on difference (3, 21) (Wilcoxon-Kruskal/Wallis test (2-sample test, normal approximation (S = 1381, Z = -2.9680, p = 0.0030), 1-way test, X^2 approximation ($X^2 = 8.8368$, df = 1, p = 0.0030); A Tukey-Kramer HSD test specified the difference).

Though observers repel some fishes, they attract others (Kulbicki 1998; Byerly and Bechtol 2005). The *Cyprinella* spp. were attracted to both roving and stationary observers and were repeatedly observed feeding on fine particulate organic matter agitated by the observer. Because the relative abundance of *Cyprinella* spp. per 10 m^2 estimated by the roving and stationary observers was identical at 14%, there was no

evidence to suggest that *Cyprinella* spp. were more attracted to one type of observer than to the other type.

The stationary observer also recorded common shiners moving from lower vertical strata (0.6 depth) to higher strata (0.2 depth), in pursuit of forage, though only four individuals at one event did so. The velocity at 0.6 depth at that location was measured to be 9 cm/s and the velocity at 0.2 depth was estimated to be 11 cm/s.

Habitat Suitability Criteria Comparisons for Selected Taxa and Life Stages

Smallmouth bass

The stationary observer found one J MDO at 0.2 depth. The roving observer noted 5 J MDO holding at 0.6 depths, while 15 were observed at 0.8 depths. The stationary observer recorded 12 individual J MDO (out of 12) at 0.8 depths (LR X^2 suspect because of small sample sizes (LR X^2 = 5.244, df = 1, p = 0.0220)). The stationary observer noted three J MDO exhibiting vertical feeding behavior, holding at 0.8 depth and foraging at 0.6 depth.

There were no differences in the vertical positions the observers noted for SA MDO (LR X^2 = 0.619, df = 1, p = 0.4312). On average, 13% of SA MDO were occupying 0.6 depths, while 87% occupied 0.8 depths.

The distributions of the depths and velocities recorded by each type of observer were not significantly different for either MDO life stage (all p > 0.05). Table 5 contains a summary of statistical differences in habitat use, by taxa, for all taxa. The roving observer did however find J MDO and SA MDO in significantly different depths (2-sample test, normal approximation (S = 555, Z = -3.4001, p = 0.0007), 1-way test, X^2 approximation (X^2 = 11.5943, df = 1, p = 0.0007). A Tukey-Kramer HSD test confirmed

that the roving observer found SA MDO at depths that were on average 34 cm deeper than J MDO (95% CI on difference (12, 55)). No statistically significant differences were found in the mean column velocities used by J MDO and SA MDO in the data collected by either observer (all p-values > 0.05). The stationary observer did not observe J MDO and SA MDO using different depths (p > 0.05).

Satinfin and Spotfin shiner

The vertical positions of the *Cyprinella* spp. were different among the observers (LR $X^2 = 28.151$, df = 2, p < 0.0001). The roving observer noted 64% of *Cyprinella* spp. holding at 0.6 depths, while the stationary observed recorded 65% of *Cyprinella* spp. holding at 0.8 depths. The relative feeding depths for the *Cyprinella* spp. were not different across methods (LR X^2 p > 0.05). On average, 70% of *Cyprinella* spp. were feeding at 0.6 depth. The velocities associated with those vertical strata were discussed previously.

There were differences in the depths that each observer found *Cyprinella* spp. occupying (2-sample test, normal approximation (S = 7795.5, Z = 3.8465, p = 0.0001), 1-way test, X^2 approximation ($X^2 = 14.8057$, df = 1, p = 0.0001)). The stationary observer found *Cyprinella* spp. at depths that were on average 11 cm deeper than did the roving observer (95% CI (6, 17)). There were no differences in the mean column velocities used by *Cyprinella* spp. (p > 0.05).

River chub

All river chub (126 out of 126 individuals) were observed at 0.8 depths. No differences in depth were identified between methods (p > 0.05).

A Wilcoxon/Kruskal-Wallis test found a statistically significant difference in the mean column velocities that each observer found NMI in (2-sample test, normal approximation (S = 2806, Z = -2.8667, p = 0.0041), 1-way test, X^2 approximation (X^2 = 8.2319, df = 1, p = 0.0041)). A subsequent Tukey-Kramer HSD test however failed to acknowledge a difference (95% CI on difference 0.00, 0.12 m).

Common shiner

There were no significant differences in the vertical strata that each observer noted LCO occupying (LR X^2 p > 0.05). The roving observer found LCO at deeper depths than the stationary observer (2-sample test, normal approximation (S = 1097, Z = -2.8080, p = 0.0050), 1-way test, X^2 approximation (X^2 = 7.9104, df = 1, p = 0.0049); mean difference = 9 cm, 95% CI on difference (3, 15); Tukey-Kramer HSD). The statistical difference in depth is likely too small to represent a biologically significant one and may be an artifact of the habitats randomly sampled by each observer.

The roving observer also found LCO in faster mean column velocities (2-sample test, normal approximation (S = 909.5, Z = -4.5326, p < 0.0001), 1-way test, X^2 approximation (X^2 = 20.5865, df = 1, p < 0.0001); mean difference in velocities = 19.2 cm/s, 95% CI on difference 11.2, 27.1; Tukey-Kramer HSD).

Table 2. Discharge Summary for Sampling Period on the South Fork

Month	Site/Gage	Mean (m³/sec)	Range		Percentile[1]
June	Lynnwood	11	8	18	25th
	Luray	16	11	29	50th
	Front Royal	21	14	34	50th
July	Lynnwood	8	6	13	25th
	Luray	12	9	19	25th
	Front Royal	13	8	23	25th
August	Lynnwood	7	5	14	25th
	Luray	9	6	35	25th
	Front Royal	8	6	25	10th
September	Lynnwood	8	5	30	50th
	Luray	13	9	49	75th
	Front Royal	15	11	42	75th
Mean		12			

Mean in header column is (m^3/sec).

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[1] USGS WSC 2009

Table 3. Partial Summary of Observer Comparisons

Comparison	X^2 p-value	Observer with Higher Value	Mean Difference	95% CI on Difference	
Fish per 10 m^2	**0.0250**	Stationary	27	11	43
# Events per 10 m^2	**0.0039**	Stationary	8	4	12
# Fish per event	0.8719	na			
Taxa per 10 m^2	**0.0036**	Stationary	3	2	5

Statistically significant at alpha = 0.05
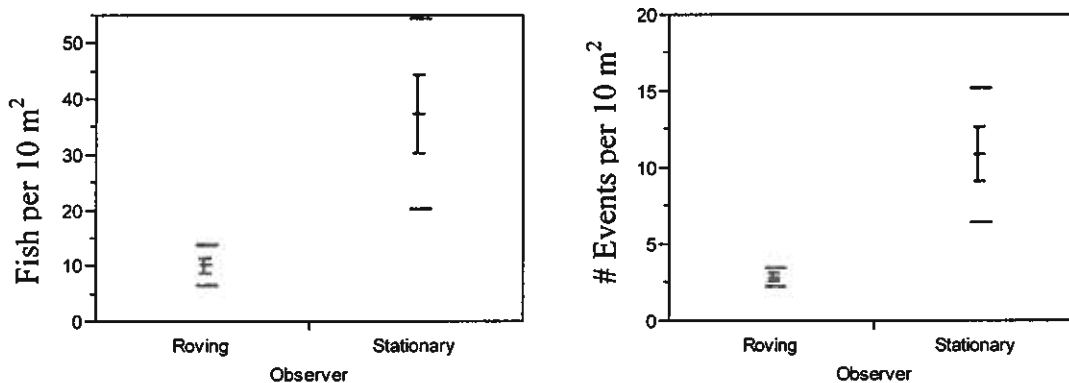
Figure 1. Number of Individual Fish and Number of Sampling Events per 10 m^2 Estimated by Each Type of Observer. Horizontal lines illustrate means, SE, and 95% CI's.
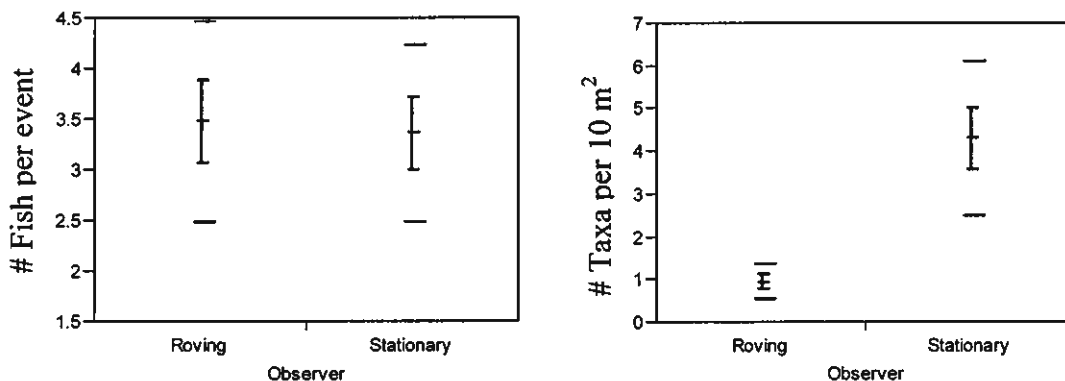
Figure 2. Number of Fish per Sampling Event and Number of Taxa per 10 m^2 Estimated by Each Type of Observer. Horizontal lines illustrate means, SE, and 95% CI's.

Table 4. Summary of Wilcoxon/Kruskal-Wallis Tests Comparing the Relative Abundances Estimated by Each Observer for Frequently Sampled Taxa and Life Stages

Taxa/Lifestage	Roving n	Stationary n	2-sample, normal approx.			1-way test, X^2 approx.		
			S	Z	p-value	X^2	df	p-value
J MDO	5	3	17.5	1.0760	0.2819	1.5122	1	0.2188
SA MDO	6	5	35.0	0.8253	0.4092	0.8410	1	0.3591
LAU	5	2	8.5	0.0000	1.0000	0.0382	1	0.8451
SA LAU	5	2	9.0	0.1954	0.8451	0.1527	1	0.6959
Cyp. spp.	5	5	25.0	-0.4178	0.6761	0.2727	1	0.6015
NMI	5	4	26.0	1.3472	0.1779	2.1600	1	0.1416
NIN	4	3	13.0	0.1835	0.8544	0.1346	1	0.7137
LCO	4	2	10.0	1.1573	0.2472	1.9286	1	0.1649
YOY	4	2	6.0	-0.2315	0.8170	0.2143	1	0.6434

Statistically significant at alpha = 0.05

sample sizes reflect the number of transects where each taxa and life stage were observed
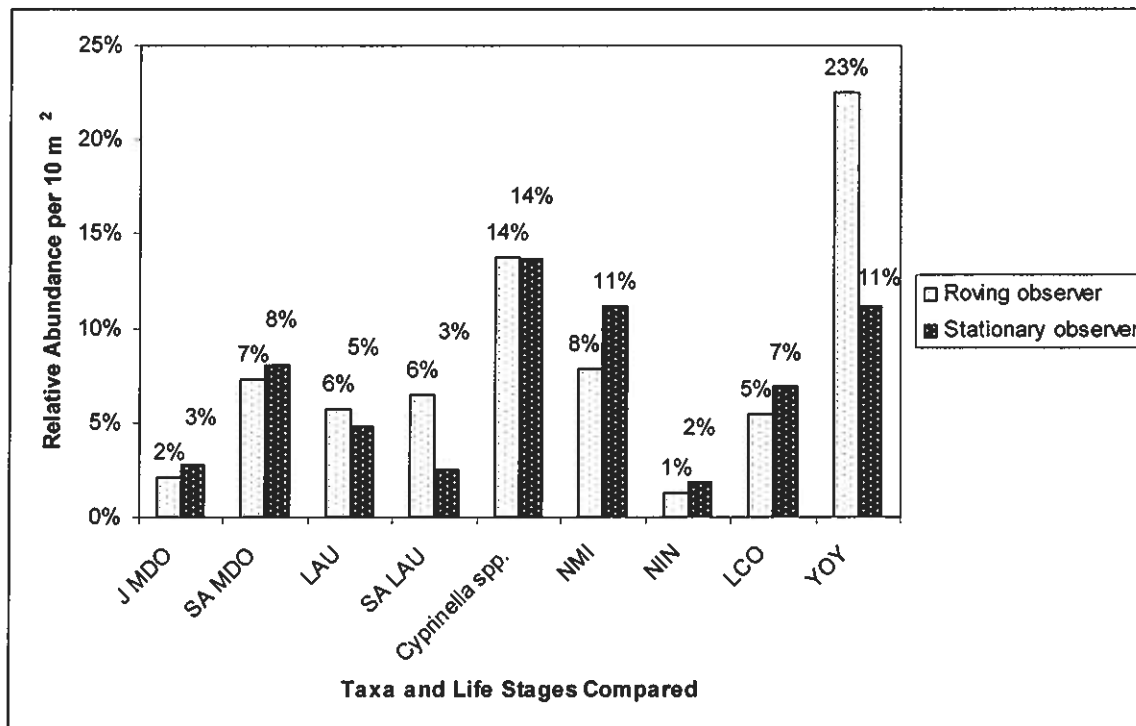
Figure 3. Visual Comparison of Relative Abundance per 10 m² by Taxa and Life Stage[2]

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[2] Percentages represent the proportional representation of each taxa and life stage among all data collected by each observer

Table 5. Summary of Fish per 10 m^2 and Relative Abundance for Common Taxa and Life Stages by Type of Observer[3]

Taxa/ lifestage	Roving				Stationary			
	n (events)	n (individual)	per 10m^2	Rel Abun	n (events)	n (individual)	per 10m^2	Rel Abun
J MDO	11	20	0.2	2%	8	13	1.1	3%
SA MDO	38	69	0.7	7%	19	38	3.2	8%
LAU	16	54	0.6	6%	6	23	1.9	5%
SA LAU	15	61	0.6	6%	5	12	1.0	3%
Cyp. spp.	35	130	1.4	14%	16	65	5.4	14%
NMI	23	74	0.8	8%	22	53	4.4	11%
NIN	11	12	0.1	1%	6	9	0.8	2%
LCO	10	51	0.5	5%	9	33	2.8	7%
YOY	23	213	2.2	23%	6	53	4.4	11%
Totals	182	684	na[4]	72%	97	299	na	63%

[3] The number of individuals of each taxa or life stage was divided by the area sampled to derive the number of individuals of each taxa or life stage per 10 m^2 sampled by each type of observer. The number of individuals of each taxa or life stage per 10 m^2 was divided by the total estimated number of individual fish of all species and life stages per 10 m^2 sampled to generate estimates of relative abundance.

[4] Estimates of fish per 10 m^2 used in the calculation of relative abundances included all fishes sampled, many of which are not included in this table.

Table 6. Summary of Differences in Habitat Use by Taxa and Life Stage

Taxa/Life Stage	Habitat Parameter	p-value	Observer with Higher Value	Mean Diff. (cm or cm/s)	95% CI on Diff.	
J MDO	Depth	p > 0.05	na			
	Mean col. vel.	p > 0.05	na			
SA MDO	Depth	p > 0.05	na			
	Mean col. vel.	p > 0.05	na			
Cyp. spp.	Depth	**p = 0.0001**	Stationary	12	6	15
	Mean col. vel.	p > 0.05	na			
NMI	Depth	p > 0.05	na			
	Mean col. vel.	**p = 0.0041**	*	--	--	--
LCO	Depth	**p = 0.0050**	Roving	9	3	15
	Mean col. vel.	**p < 0.0001**	Roving	1.6	.9	2.2

Statistically significant at alpha = 0.05

* no Tukey-Kramer HSD difference identified

Discussion

In one study, bull trout (*Salvelinus confluentus*) were observed to move upstream 28% of the time in response to a roving, snorkeling observer vs. only 18% of the time in response to a wading observer with a backpack electroshocker (Peterson et al. 2005). In the same study, bull trout fled from a roving observer that was 10 to 20 m distant in clear water. The greater amount of disturbance created by the snorkeling observer swimming upstream was the explanation for the differential response of the bull trout to each type of sampling. Because the disturbance of the stationary observer in this study was assumed to be less than that of the roving observer, it is plausible that the stationary observer recorded more fish per 10 m^2 because his level of disturbance was less threatening. A similar negative relationship between reef fish counts and the speed of the observer has also been documented (Lincoln-Smith 1988) that indicated that quickly swimming observers simply overlooked more fish.

Although nearly all individual fish left the immediate vicinity of either observer upon his entry into the water, many returned within the first several minutes of the observer remaining still. Because the disturbance of the roving observer was constant as he continued upstream fish that spooked initially may not have returned as the disturbance was ongoing.

Counts of individual fish by area sampled are essentially estimates of density (Watson et al. 1995). Estimates of density are influenced by the ease of locating and identifying certain taxa (Lincoln-Smith 1988), the mobility and degree of wariness or

attraction of the target species to observers (Byerly and Bechtol 2005; Peterson et al. 2005), the direction and speed of approach of fish relative to the diver (Watson et al. 1995), the speed of the observer (Lincoln-Smith 1988; De Girolamo and Mazzoldi 2001), and the time spent observing (Watson et al. 1995). The estimates of fish per 10 m^2 produced by each type of observer in this study should not be interpreted as rigorous estimates of density because calculations of the area sampled by each observer did not take into account all of the above influences (Byerly and Bechtol 2005). Because of the length of time that the observers monitored each sample area and because some of the most active taxa could cover significant distance in that length of time, it is likely that some percentage of the fish noted in particular cells or lanes occupied multiple microhabitats over the observation period. Instead of observing a single 2 x 1 m cell, the stationary snorkeler arguably observed an area with a radius equal in length to the distance that each observed fish could travel in 15 minutes. The estimates of density, expressed as the number of fish or the number of taxa per 10 m^2, are thus likely overestimated for the stationary observer as mobile and curious fish moved into the observer's sample cell (Lincoln-Smith 1988; Byerly and Bechtol 2005). If fish fled from the disturbance created by the roving observer, or if fish swam parallel to and in the same direction as the roving snorkeler (Watson et al. 1995), those estimates of density may be underestimated. Provided that the bias inherent to each type of observer is consistent however, even biased estimates of density remain useful means of comparison (Watson et al. 1995; Byerly and Bechtol 2005).

Evidence that has contributed to the threat sensitive predator avoidance hypothesis (Helfman 1989; Brown et al. 2006) suggests that fish base the intensity of

predator avoidance responses on their recent experiences with a similar predator (Ferrari and Chivers 2006) and that a combination of chemical and visual cues inform the reactions of fish to specific perceived threats (Chivers et al. 2001). Because there is no reason to believe that the fish community that resides in the study area is conditioned to avoid snorkelers per se, as divers don't spearfish there for example (Kulbicki 1998), it is likely that fish would base their responses primarily on the levels of disturbance that the roving and stationary observers presented. It was probably fair to assume that the disturbance created by the roving observer would be perceived as a greater threat and thus elicit a greater avoidance response than would the disturbance generated by the stationary observer.

Because fright responses of rainbow trout (*Oncorhynchus mykiss*) are negatively correlated with substrate size (Peterson et al. 2005), it is possible that substrate size had some influence on the observer effects associated with each type of observer in this study. However, because each observer collected data over many types of substrates, the effect of substrates is likely minimal or at least equally distributed among the observer types.

The observer effects of the roving and stationary observers likely affected the number of sampling events per 10 m^2 in the same ways that they affected the number of individual fish per 10 m^2.

As the stationary observer noted an average of 27.3 more individual fish per 10 m^2 sampled (Table 3), it was not surprising that the stationary observer encountered greater taxa richness per 10 m^2 also. Given heterogeneous microhabitats and a diverse fish community, it was likely that many taxa would use a given microhabitat. In the

study planning stage, one of the perceived strengths of the stationary observer was that an observer producing minimal disturbance and observing for 15 minutes would witness more taxa than would an observer creating greater disturbance and spending less time at each microhabitat. The data produced by this study suggests that assumption was correct and consistent with studies of reef fishes that have found a negative relationship between the speed of a roving observer and estimates of taxa richness (Lincoln-Smith 1988).

The higher counts of individuals and taxa per area sampled recorded by the stationary observer were most pronounced at study sites with higher estimates of fish or taxa richness. At sites where the estimated fish density was high (represented by the mean of each observer's estimates of the total number of individual fish per 10 m^2), the ratio of the roving observer's density estimate and the stationary observer's density estimate tended to be low. That is, the higher the density of fish, or taxa, the larger the disparity between the density estimates of each observer, with the roving observer finding from 9 to 58% of the number of fish that the stationary observer noted per 10 m^2. There was thus a moderately strong, negative linear relationship between the estimated mean number of fish per 10 m^2 and the ratio of the roving observer's density estimate and the stationary observer's density estimate (ratio roving/stationary = $0.7524 - 0.0177*$mean # fish per 10 m^2; n = 6, r^2 = 0.75; Figure 4).

Because different fishes respond differently to observers (Peterson et al. 2005), snorkelers and divers specifically (Kulbicki 1998; DeGirolamo and Mazzoldi 2001), this study tested the hypothesis that the roving observer's and the stationary observer's estimates of relative abundance per 10 m^2 for given taxa would not be significantly different. The lack of statistically significant differences in the relative abundance of

each taxon (Table 4) suggested that observer effects on the specific taxa and life stages sampled were shared by both the roving and the stationary observers. Statistically significant differences in the relative abundances of taxa per 10 m^2 could have indicated that there were observer effects on the frequencies that specific taxa were sampled by each type of observer. The graphical comparison of relative abundance (Figure 3 and Table 5) also illustrated the similarities in the estimates that each observer produced. Young-of-the-year were observed sporadically and sometimes in large groups. It's likely that the roving observer encountered YOY more often because the roving observer sampled a larger area. Secondly, because YOY were often observed in large groups, the more frequently that YOY were encountered, the greater the likelihood that very large groups would be noted which subsequently increased the relative abundance of YOY. There was little to suggest that the two observation techniques produced divergent estimates of relative abundance for fishes frequently sampled in diverse warmwater streams.

This study thus found limited evidence to suggest that observer effects associated with a roving observer are more pronounced than those of a stationary observer. My opinion is that relative abundances per 10 m^2 of each taxon, as graphically compared in Figure 3, provided the best measure of an observer effect because they integrated variation around sample sizes, the areas sampled by each observer, and different sites. By this measure, each observer produced similar descriptions of the fish community. This was important to determine because in developing habitat criteria for a fish community, criteria for many taxa are needed (Bovee 1986). An observer effect

associated with a particular snorkeling technique that biased the taxa sampled could thus be problematic.

The roving observer noted 0.03 margined madtoms (NIN) per rock flipped vs. the stationary observer's 0.01 (12 NIN for 360 rocks flipped vs. 9 NIN for 630 rocks flipped, respectively). This suggests that cursory sampling for margined madtoms over a large area was more productive than more extensive sampling in a small area. Underestimation of cryptic reef fishes is common of all visual sampling methods (Lincoln-Smith 1988) and it's likely that both types of observers in this study underestimated the true abundance of NIN. Slowing the speed of the observer through the sample area is the measure generally taken to compensate for underestimation (De Girolamo and Mazzoldi 2001), though the numbers of NIN sampled in this study do not reflect an increase with time spent observing per area sampled.

Though observers repel some fishes, they attract others (Kulbicki 1998; Byerly and Bechtol 2005). The *Cyprinella* spp. were attracted to both roving and stationary observers and were repeatedly observed feeding on fine particulate organic matter agitated by the observer. Although observations were not recorded from the eddy to the lee of the stationary observer, *Cyprinella* spp. were almost always present there, feeding and riding in the swirling currents there. A possible explanation for the differences in the relative holding depths that *Cyprinella* spp. were observed in is that the agitation and velocity shelter created by the roving observer constantly provided *Cyprinella* spp. opportunities to forage at 0.6 depths. Other *Cyprinella* species (*C. cardinalis*) have demonstrated selection of microhabitats with lower than average velocities and an ability to find and exploit those areas while swimming at high speed (Scott and Magoulick

2008); the implication being that *Cyprinella* spp. are most definitely attracted to snorkelers. Because the relative abundance of *Cyprinella* spp. per 10 m^2 estimated by the roving and stationary observers was identical at 14%, there was no evidence to suggest that *Cyprinella* spp. were more attracted to one type of observer than to the other type.

Because the stationary observer consistently noted vertical feeding behavior while the roving observer never did (with the exception of a single bluegill sunfish that snatched a cicada out of the surface tension in a pool), there does seem to be some evidence that a less threatening observer effect was associated with the stationary observer. Though the vertical feeding behaviors observed may not be particularly useful in the construction of habitat suitability criteria, observer effects seem to have some bearing on the vertical positions and observed behavior of drift feeding cyprinids.

That neither snorkeling technique seemed to illicit strong observer effects or fright bias on the fishes observed was consistent with previous characterizations of snorkeling as an un-biased method with the exception of snorkeling in very shallow water (Bovee 1986; Heggenes et al. 1990). That few rare or skittish fishes were sampled frequently by either observer suggested that there are sampling biases common to all types of underwater observations that influence the specific taxa sampled (Lincoln-Smith 1988).

The primary objective of this investigation was to determine whether or not these two underwater observation techniques would produce statistically different habitat suitability criteria. The statistically significant differences in habitat use observed by each observer were small enough to be of questionable biological significance with the possible exception of the difference in mean column velocities for LCO. The difference

in the estimates of the mean column velocities used by common shiners was 19.2 cm/s (Table 6). In tests used to determine the critical swimming speeds of fish, velocity is increased in increments of 0.5 to 1 body length per second (BL/s) for a proscribed length of time (Plaut 2001). Congeneric cardinal shiners (*Luxillus cardinalis*) have demonstrated critical swimming speeds of 4.5 BL/s (Scott and Magoulick 2008). For an adult common shiner approximately 8 cm in total length, 19.2 cm/s represents approximately 2.4 BL/s. The roving observer estimated mean column velocities for LCO of approximately 4.8 BL/s (38.1 cm/s) while the stationary observer noted LCO in velocities of approximately 2.4 BL/s (18.9 cm/s). The difference in velocity of 2.4 BL/s, representing an increase from 2.4 BL/s to 4.8 BL/s, to a velocity slightly greater than the critical velocity of a congenera, suggests that a difference of 2.4 BL/s or 19.2 cm/s could be energetically consequential and thus biologically significant. There were no comparable means to evaluate biological significance with regard to depths.

That the observers did not generally produce different criteria suggested that observer effects were of similar magnitude regardless of the observation method. It is possible that differences in the available habitats that each observer sampled at the site scale are reflected in differences in the habitat criteria for LCO and the *Cyprinella* spp.

Because of small sample sizes, small statistical differences of subsequently dubious biological significance, the complexities of habitat suitability curve comparisons (Thomas and Bovee 1993), and because the Physical Habitat Simulation model (PHABSIM) was not run with the habitat criteria developed in this study, the practical differences in the habitat suitability criteria that each observer could have developed are not clear. Some extent of difference is possible as the depth and velocity observations for

LCO and the *Cyprinella* spp. demonstrate. The stationary observer offers advantages in efficiency (i.e. fish per area sampled for example) in habitat types with high fish or taxa densities or in situations where fish behavior is a principle objective. But because the burden of proof is on the new method and it's not clear that burden has been met here, the standard method of a roving observer is probably the most easily defended.
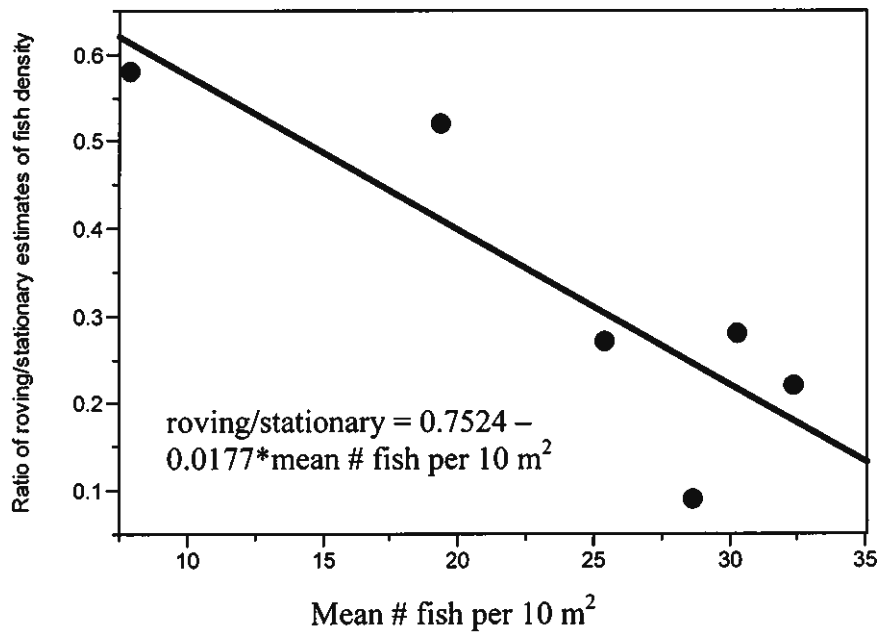
Figure 4. Plot of the Negative Linear Relationship Between Mean Estimates of Fish Density and the Ratio of the Observers' Estimates of Fish Density at each Study Site

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Appendix I.

Table 7. Table of station numbers, names, and coordinates of USGS study sites and the nearest stream gages.

Station No.	Station Name	Latitude (NAD83)	Longitude (NAD83)	PID (local name)
01629510	S F SHENANDOAH RIVER BELOW ROUTE 211 NR HAMBURG, VA	383920	-783230	Kauffman Mill
01630555	SOUTH FORK SHENANDOAH RIVER AT OAK HILL, VA	384457	-782542	Route 611
01630600	S F SHENANDOAH RIVER ABOVE DRY RUN AT COMPTON, VA	384701	-782234	Compton Pool
01630790	S F SHENANDOAH RIVER BL GOONEY RUN NR LIMETON, VA	385238	-781526	Thunderbird Farms
01628500	S F SHENANDOAH RIVER NEAR LYNNWOOD, VA	381921	-784518	Lynnwood Gage
01629500	S F SHENANDOAH RIVER NEAR LURAY, VA	383846	-783206	Luray Gage
01631000	S F SHENANDOAH RIVER AT FRONT ROYAL, VA	385450	-781240	Front Royal Gage

USGS 2009

Part II:

Habitat suitability criteria for selected fishes of the South Fork of the Shenandoah River

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Introduction

The Instream Flow Incremental Methodology (IFIM) is an analytical tool designed to assist resource managers faced with increasing municipal, industrial, and agricultural demands on surface water supplies (Bovee 1986). Among the considerations of resource managers in making water allocation decisions are the effects of changes in stream discharge on biota, including the resident fish community, as changes in discharge alter the available habitat (Bain et al. 1988). To estimate the effects of changes in discharge on fish, knowledge of the specific habitats that fish use is required. The Physical Habitat Simulation Model (PHABSIM) is the component of the IFIM that links habitat use to increments of discharge (Bovee 1986; Aadland et al. 1991). Habitat suitability criteria for the target species are among the inputs required for PHABSIM (Bovee 1986). The estimates of usable area that PHABSIM produces are only as reliable as the habitat criteria those estimates are based on (Freeman et al. 1997; Maki-Petays et al. 2002).

The availability of specific habitat types changes as discharge increases or decreases (Krstolic et al. 2006). Consequently, changes in discharge will affect different fishes differently (Bain et al. 1988). Instream flow analyses and published habitat criteria often focus on salmonids (Strakosh et al. 2003) and other game fish (Probst et al. 1984; Groshens and Orth 1994). A problem associated with using game fish as indicator species in the context of instream flow analyses is that in warmwater streams game fish tend to be pool-oriented and facultative riverine fishes (Jenkins and Burkhead 1993). As the

area of pools tends to change less in response to discharge than the area of riffles and runs (Krstolic et al. 2006), the habitats available to pool oriented fishes are less sensitive to changes in discharge than are the habitats of other fishes (Orth and Maughan 1982; Bain et al. 1988). As a result, resource managers could underestimate the discharge required to maintain the habitats used by the majority of the fish community if only habitat criteria for game fish are considered in making decisions that affect instream flows (Bovee 1986; Aadland 1993). Ecological considerations, such as situations where pool dwellers prey on riffle dwellers (Leonard and Orth 1988B; Aadland 1991), also require that the habitat criteria of many members of the fish community be determined to maintain the biological integrity of the stream. While habitat criteria for habitat guilds and indicator species have been pursued as a way to minimize field work (Leonard and Orth 1988A; Persinger 2003), it remains desirable to construct habitat suitability criteria for as many members of the fish community as feasible (Leonard and Orth 1988 A and B; Aadland 1993).

There are at least three types of habitat suitability criteria (Bovee 1986). Category I criteria are based on the literature regarding target species. Category II criteria delineate the ranges of habitat parameters used by the target species based on observations, while Category III criteria adds an available habitat component to determine habitat preference, or use disproportionate to availability. Each of these three types of criteria are univariate, considering only one habitat variable at a time.

Researchers also use one-sided chi square tests (X^2) to evaluate the applicability of specific habitat suitability criteria (Bovee 2008). Chi-square tests can be used to create composite suitabilities that account for multiple habitat parameters in single tests.

Habitats with measurements of multiple parameters, depth and velocity and a channel index, for example, each within the range of habitats defined as optimal, would be defined as optimal habitats. If the value for any one parameter was outside the range of that parameter defined as optimal, that habitat might be defined as marginal. If the habitat is deemed unsuitable in regard to any single habitat parameter, that habitat is unsuitable. The final ranking of a particular habitat, as optimal or marginal, or suitable or unsuitable, is thus determined by the lowest ranking of each of the component habitat parameters.

Chi-square tests are also used to test the transferability of habitat criteria among streams (Thomas and Bovee 1993). Habitat suitability criteria that were developed on a source stream but can account for habitat use in another stream are said to be transferable (Bovee 1986; Thomas and Bovee 1993). Transferable criteria would suggest that those criteria were accurate and biologically relevant (Freeman et al. 1997) and that the needs or preferences of target species are relatively uniform at scales larger than the local (Beecher et al. 2002; Maki-Petays et al. 2002). Transferable criteria could also alleviate the necessity of gathering site-specific habitat use data which would create substantial cost savings (Bovee et al. 1998), while also shortening the time required by resource managers to gather the information needed to make informed decisions (Freeman et al. 1997).

Among of the strengths of using X^2 tests is that they incorporate a decision rule that allows researchers to evaluate the transferability of habitat criteria in an objective way (Thomas and Bovee 1993). Visual comparisons of curve based criteria by what

Bovee (1986) called the "abbreviated convergence" method are imprecise (Conklin et al. 1995; Maki-Petays et al. 2002).

The realities of local drivers of habitat use including differences in habitat availability (Beecher et al. 2002; Guay et al. 2003) and species interactions (Power and Mathews 1983) confound the determination of the feasibility of truly transferable habitat criteria. The North Fork and South Fork of the Shenandoah River, in Virginia, present a unique opportunity to test the viability of the concept of transferability. Because of their shared climates, history, geology, and the similarities of their fish communities, if transferable habitat suitability criteria are attainable, transferability should be demonstrable between the North and South Forks of the Shenandoah River.

This study thus sought to construct habitat suitability criteria for as many fish species and lifestages as feasible from the South Fork of the Shenandoah. The criteria developed by this study will be input into PHABSIM as part of an IFIM analysis of the South Fork. The South Fork IFIM study is a cooperative effort between the U.S. Geological Survey (USGS) and the Northern Shenandoah Valley Regional Commission and the Central Shenandoah Planning District Commission. One-sided X^2 tests were used to evaluate whether the criteria developed represented the habitat use of fishes in the South Fork to a statistically significant degree. Secondly, X^2 tests were used to explore the issue of transferability, with a particular interest in the transferability of habitat criteria developed on the North Fork of the Shenandoah (Persinger 2003) to the fishes of the South Fork. Because of the limited availability of data, one-way tests were conducted that tested other researchers' habitat criteria (Aadland et al. 1991; Groshens and Orth 1994; Persinger 2003) against the fish use data obtained in this study.

Study Area

Fish habitat use data were collected in the South Fork of the Shenandoah River

(South Fork), in the Valley and Ridge physiogeographic province in northwest Virginia.

The Shenandoah River system drains approximately 6,070 km^2 in Frederick, Clarke,

Rockingham, Shenandoah, Augusta, Page, and Warren Counties in Virginia and Jefferson

County, West Virginia (PWP 2009), and includes the South Fork, the North Fork of the

Shenandoah River (North Fork), and the Main Stem of the Shenandoah River (main

stem). The South Fork begins near Port Republic at the confluence of the North, South,

and Middle Rivers. The South Fork flows north-northeast for approximately 167 km

until its juncture with the North Fork near Front Royal, Virginia (DGIF 2009). The

watershed of the South Fork is approximately 4,300 km^2, captures runoff from the Blue

Ridge and Allegheny Mountains, and Massanutten Mountain, and ground water from the

karst areas in the Shenandoah and Page Valleys (DGIF 2009). It is an unregulated

stream, though there are three low-head hydropower dams in Shenandoah, Virginia,

Newport, Virginia, and Luray, Virginia (DGIF 2009). The South Fork is a 5th order

stream, and bedrock runs constitute 34% of its length; natural pools, particle runs,

bedrock riffles, and artificial pools make up 21, 18, 7, and 7% of that length, respectively

(Krstolic, personal communication 2009; see Table 1). The South Fork had an average

annual discharge of 38 m^3/sec based on USGS records from 1926 to 2008 from stream

gages at Lynnwood, Virginia, Luray, and Front Royal.

The North Fork is also a 5th order stream and flows approximately 172 km, parallel to the South Fork, from Rockingham County to Front Royal in the Shenandoah Valley floor at the western foot of Massanutten Mountain and has a watershed of approximately 2,675 km^2 (Krstolic et al. 2006). The average annual discharge of the North Fork was approximately 11 m^3/sec from 1926 to 2008 based on records from USGS stream gages at Cootes Store, Virginia, Mt. Jackson, Virginia, and Strasburg, Virginia.

Downstream of Front Royal, the two rivers form the Main Stem. The Main Stem joins the Potomac River in Harper's Ferry, West Virginia, which flows into the Chesapeake Bay.

The study reaches were located at established USGS sites. A table containing the station numbers, names, and coordinates of the USGS sites sampled and the nearest river gages is contained in Appendix I. Each study reach contained multiple transects (3 to 5), each transect at a discrete mesohabitat type. Specific transects to be sampled at each study reach were stratified by mesohabitat type and randomly selected in proportion to the relative abundance of each habitat type. A single study reach could contain more than one transect sampled. As field work progressed, existing USGS sites were exhausted and two additional sites were established on the basis of their mesohabitat type and accessibility. Eight transects at 6 study reaches were sampled by both types of observer (Appendix I).
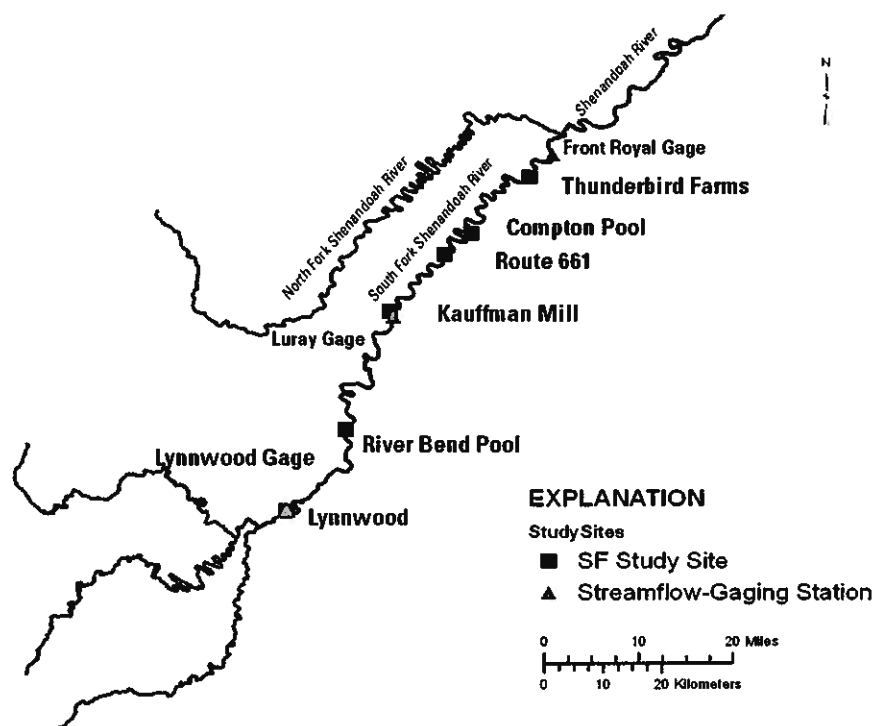
Figure 1. Map of the Shenandoah River Basin and Study Sites[1]

Table 1. Summary of Relative Frequencies of Mesohabitat Types in the North Fork and the South Fork Rivers, by Percentage of River Length[2]

Mesohabitat Type	South Fork (%)	North Fork (%)
Artificial Pool	6.8	6.6
Backwater	5.1	2.0
Glide	2.9	na
Natural Pool	20.5	10.1
Bedrock Riffle	7.2	3.9
Particle Riffle	4.5	10.1
Pocket Run	1.7	1.7
Bedrock Run	33.8	34.7
Particle Run	17.5	30.9

[1] Map from Krstolic, personal communication 2009
[2] North Fork mesohabitat data from Krstolic et al. 2006, South Fork mesohabitat data and table from Krstolic 2009

Methods

Data Collection

Data to construct habitat suitability criteria for as many species as feasible in the

South Fork were collected by a roving observer (Bovee 1986) and by a method that

utilized what will be referred to as a stationary observer. Both methods used direct

underwater observation (Goldstein 1978; Thurow 1994) with snorkels to observe

microhabitat use by fishes.

The roving observer swam 6 randomly placed, 30 x 1 m lanes, perpendicular to

the selected transects (Bovee 1986). The roving observer marked fish locations and

random habitats every 6 m along the length of each lane with weighted markers and

physical habitat parameters were measured for a 1 m^2 area with the marker at its center.

The random habitat measurements were obtained as a means to account for available

habitat (Bovee 1986). A stationary observer recorded how many of what fishes occupied

a randomly selected microhabitat over a 15-minute interval. Ten 2 x 1 m cells were

randomly selected from each transect with one cell at the rivers edge; the long axis of

each cell was parallel to the transect, perpendicular to the current. Two-meter cell widths

were observed by the stationary observer because that was a conservative estimate of

approximately twice the maximum distance at which the observer could identify all fishes

present. Underwater visibility was considered adequate for sampling if an observer with

mask and snorkel could see at least 1 m to each side and from the water surface to the

substrate.

Fish observed were identified, counted, and recorded on an underwater note pad. Depth (measured in feet, reported here in centimeters) and mean column velocity (measured in ft/s, reported here in cm/s; measured at 0.6 depth) were measured either with a FlowTracker (Sontek) on a graduated top-setting wading rod or with a StreamPro Acoustic Doppler Current Profiler (ADCP) at the locations of the rover observer's fish markers and at the center of each of the ten sample cells observed by the stationary observer.

Numerical codes for substrate size based on a modified Wentworth scale (Bovee 1986), cover, and embeddedness, were used so that substrate characterizations could be summarized in a single channel index (Krstolic et al. 2006). Where two or more cover types were present at a particular location, cover was characterized as "complex" in the channel index. Embeddedness was estimated visually by percent embedded (between 0 and 25%, between 25 and 50%, between 50 and 75%, or between 75 and 100% embedded).

Ten rocks were flipped in each sample lane where feasible and in each sample cell at the conclusion of the 15 minute observation period, to look for cryptic taxa including *Cottus* spp. and *Noturus insignis*. Also noted were the vertical positions of fish in the water column. Vertical positions were recorded for individual fish as being at 0.2 depth (approximately the top $1/3^{rd}$ of the water column), 0.6 depth (central $1/3^{rd}$), or 0.8 (lower $1/3^{rd}$; at, near, or within the substrate). It was also noted if fish moved from one vertical stratum to another. If vertical movements were observed, the vertical position the fish moved from, its holding position, was recorded, as was the vertical position it moved to,

its feeding position; feeding being the presumed activity that was encouraging vertical movement.

Conscientious efforts were made not to count the same fish repeatedly, not to count fish that were obviously attracted to the observer, nor to count fish that were observed in microhabitats outside of the boundaries of sample cells or lanes. The time spent observing cells and swimming lanes at each site was approximately equal.

Each of the ten sample cells were also sampled with pre-positioned electroshocking equipment (PPES). A Smith-Root SR-6 tote barge floating a 5.0 GPP and generator was used to provide power to a modified anode. Two circular anodes from two anode poles were replaced with a single 50' length of 8 gauge, 3-conductor, water-resistant insulated wire. Two conductor wires were striped and fitted with vinyl insulated bolt terminals and bolted into the distal ends of the two anode poles which were duct-taped to each other and to the tote barge to minimize their movement. At the wet end of the anode wire, the same two conductor wires that were attached to the anode poles were stripped and connected to 1 m lengths of un-insulated 5/16 inch braided stainless steel cable via brass connectors. All connection points were reinforced with shrink tubing and electrical tape.

Once the components of the tote barge were assembled at the river bank, the barge was pushed to a point approximately 3 m downstream of the downstream edge of the cell to be sampled and secured in place. The anode wires were placed on the river bottom along the sides of the cell and left undisturbed for 15 minutes to allow the area within the cell to return presumably to a representative occupancy (Bain et al. 1985; Peterson et al. 2005). After 15 minutes, the tote barge operator started the generator and

delivered a high-voltage shock for 30 seconds, during which time 1 or 2 netters in waders

would net the fishes shocked within the cell, avoiding capturing fishes shocked from

outside the cell. Netted fishes were put into a bucket of water until netting was complete

whereby fishes were identified, counted, recorded, and replaced into the river. This

process was repeated at the other cells along the transect. All fish were identified by the

same researcher.

With few exceptions, fishes were identified to species. *Cyprinella spiloptera* and

C. analostana were indistinguishable from one another while swimming and were

recorded as *Cyprinella* spp. *Notropis rubellus* and *N. amoenus* were recorded as *Notropis*

spp. As young-of-the-year (YOY) of all species were plausibly using specific habitat

types, YOY were recorded as such without finer-scale distinction. Visually apparent life

stages were recorded when feasible. Only division into juvenile, sub-adult, and adult was

possible, and generally only for the species that attained the largest adult sizes.

Conspecific life stages are noted as juvenile (J) or sub-adult (SA), and adults are referred

to by a species abbreviation.

Data Management and Analysis

Statistical analyses were conducted using JMP 7.0 (SAS), though Excel

(Microsoft Office 2003) was used for X^2 tests, all at $\alpha = 0.05$. Non-parametric

Wilcoxon/Kruskal-Wallis tests (Daniel 2005) were used to identify differences in habitat

use among conspecific lifeststages. Tukey-Kramer tests were used to identify specific

differences (Daniel 2005).

There are in effect two sample sizes relevant to fish habitat data (Kinzie and Ford

1988). Single events where one or more conspecifics were observed are referred to as

"events". Sample sizes referred to as "n (individual)" reflect the number of individual fish observed and were used in comparisons of habitat use among conspecific lifestages. Chi-square tests were conducted with event sample sizes (Thomas and Bovee 1993).

Although the velocity at 0.6 depth is generally accepted as a good estimate of mean column velocity, measured velocities at 0.6 depth were divided by 1.020 to estimate mean column velocities (Rantz et al. 1982).

One-way X^2 tests using 2 x 2 contingency tables were conducted (Thomas and Bovee 1993) to evaluate the defensibility of the criteria this study developed and to perform transferability tests of other researcher's habitat criteria to the observed habitat use of fish in the South Fork. Habitat Suitability Criteria were tested with X^2 tests for taxa sampled on 30 or more events (Maki-Petays et al. 2002), with the exception of MDO, which had an event sample size of only 18. Habitat suitability criteria were described for sub-adult and adult smallmouth bass (*Micropterus dolomieu*, MDO), juvenile, sub-adult, and adult redbreast sunfish (*Lepomis auritus*, LAU), *Cyprinella* spp. (*Cyprinella analostana* and *C. spiloptera*), margined madtoms (*Noturus insignis*, NIN), river chubs (*Nocomis micropogon*, NMI), and young-of-the-year (YOY).

Ranges of optimal habitat were defined as those used by the central 50% of the taxa or lifestage sampled. Marginal habitats were those used by the fish outside the range of the central 50% but within the central 80% of the taxa or lifestage sampled. Habitats used by the central 80% of each taxa and life stage sampled (optimal + marginal) were considered suitable, while unsuitable habitats were those with habitat parameters with ranges outside of those used by the central 80% of each taxa and life stage.

In an effort to create an unoccupied or available habitat dataset of sufficient size

to conduct the X^2 analyses (n > 200; per Thomas and Bovee 1993), available habitat was

defined as all habitats measured where the target species was not observed, as it was

reasonable to consider availability at the scale of the population (Bovee et al. 1998).

Because the habitat use of prey is influenced by the presence of predators (Power and

Matthews 1983), the available habitat for prey fishes (redbreast sunfish (*Lepomis auritus*,

LAU), *Cyprinella* spp. (*Cyprinella analostana* and *C. spiloptera*), margined madtoms

(*Noturus insignis*, NIN), river chubs (*Nocomis micropogon*, NMI), and young-of-the-year

(YOY)) did not include habitats where largemouth bass (*Micropterus salmoides*, MSA),

smallmouth bass (*Micropterus dolomieu*, MDO), or channel catfish (*Ictalaurus*

punctatus, IPU) were present. The random available habitat measurements taken in the

field following Bovee (1986) were also excluded from the available habitat dataset

because many of those habitats were recorded multiple times, as available habitats and as

locations of one or more fish, in violation of the assumption that available habitats were

unoccupied (Bovee 1986). Because this available habitat strategy resulted in more

available habitat measurements than necessary, available habitat measurements were

randomly selected (by SAS JMP 7.0) and deleted until approximately 250 remained for

use in the X^2 tests. This last step was taken because there was some concern that a

dataset of available habitat that was much larger than the dataset of fish observations

could confound the X^2 tests (Bovee, personal communication 2008).

The analyses were performed using an Excel spreadsheet made available by

Bovee (written communication, 2009). Critical T-statistic values were found in a

standard normal distribution table (Conover 1980). The T-statistic as given in Thomas

and Bovee (1993; from Conover 1980) is defined as:

$$T = [N^{0.5} (ad - bc)] / [(a+b)(c+d)(a+c)(b+d)]^{0.5}$$

where a = number of occupied optimal (or suitable) cells; b = number of occupied

marginal (or unsuitable) cells; c = number of unoccupied optimal (or suitable) cells; d =

number of unoccupied marginal (or unsuitable) cells; and N = total number of cells

(Thomas and Bovee 1993).

The X^2 tests tested two null hypotheses (Thomas and Bovee 1993):

Ho_1: That optimal habitats were occupied with the same frequency as marginal

habitats, and

Ho_2: that suitable habitats were occupied with the same frequency as unsuitable

habitats.

Optimal habitat and marginal habitat are subdivisions of suitable habitat.

Generally, X^2 tests of optimal vs. marginal habitats are conducted to test if a set of criteria

can differentiate between optimal habitats and those that are marginal (Freeman et al.

1997). Optimal vs. marginal tests thus exclude data from habitats with parameters

outside the bounds of suitability; in this case, outside the habitats used by the central 80%

of each taxa and life stage. Due to the size of the available habitat data pool this study

had available, there was insufficient available habitat to conduct X^2 tests if all unsuitable

habitats were excluded (n < 200; Thomas and Bovee 1993). Therefore, this study tested

optimal habitats against "non-optimal" habitats to test the first hypothesis (Ho_1) to

determine if the habitat criteria could identify a narrow range of preferred optimal habitat

(Maki-Petays et al. 2002). Chi-square tests of suitable versus unsuitable habitat criteria

were also conducted to test Ho$_2$. Both hypotheses must be rejected for the tested criteria to be considered viable (Bovee, written communication 2009).

Transferability tests involved entering the Habitat suitability criteria of other researchers (Aadland et al. 1991; Groshens and Orth 1994; Persinger 2003) into the X^2 spreadsheet to test the applicability of those criteria against the data collected by this study. Adequately defined or transferable habitat criteria are generally expected to reject both null hypotheses (Thomas and Bovee 1993; Strakosh et al. 2003). Because Aadland et al. (1991) reported habitat preference criteria in the form of preference curves, this study used Groshens and Orth's (1994) definition of optimal habitat as the range of a habitat parameter with normalized habitat suitability index scores ≥ 0.7 to estimate values from the habitat criteria of Aadland et al. (1991). Marginal habitats from Aadland et al. (1991) were defined as those with ranges of habitat parameters with normalized habitat suitability index scores > 0.0 (Groshens and Orth 1994).

Results

Habitat Suitability Criteria

Appendix II lists all fishes sampled and sample sizes. The mean depth and mean column velocity sampled was 67.4 cm (SD = 46.9) and 25.6 cm/s (SD = 21.6), respectively. Discharge was relatively low and consistent throughout most of the sampling period with a mean of 12 m^3/s (SD = 4; range from 5 to 49). Table 2 contains a summary of discharge by month and by USGS stream gage for the 31 days spent collecting data in the field. The percentiles given are based on 46 to 77 years of USGS records. Discharge on the field days was representative of discharge over the entire four month period during which sampling took place.

Habitat criteria tested with X^2 tests are contained in Tables 3 through 5. Composite suitability criteria are multivariate, accounting for several physical habitat parameters (depth, mean column velocity, and the parameters included in channel indices), and are included in Tables 6 and 7 contain T-statistics that summarize the X^2 tests that were performed. Table 8 contains a summary of the composite habitat suitability tests and the conclusions drawn from the X^2 tests. Criteria that successfully rejected Ho_1 and Ho_2 were considered to be statistically defensible and referred to as "validated" in Table 8. Appendix III contains and example of the X^2 tables produced by the tests of optimal vs. non-optimal and suitable vs. unsuitable habitats. Tables 9 and 10 contain means-based summary statistics for depth and mean column velocity for each species and lifestage.

Nose velocities at 0.8 depths were estimated for selected fishes observed at three morphologically similar riffle sites, though sample sizes were not sufficient to permit habitat criteria to be constructed from them. Nose velocities at 0.8 depths were estimated using an equation derived from a linear regression of a positive linear relationship between measured mean column velocities (measured at 0.6 depth) and measured velocities at 0.8 depths (Kaufman's Mill transect 2) (n = 162; $V_{0.8}$ = -0.1297 + 0.6698*$V_{0.6}$, r^2 = 0.56). Estimates of velocities at 0.2 depth were derived from the equation $V_{0.2}$ = 0.1911 + 1.1432*$V_{0.6}$, r^2 = 0.85 (n = 162) that describes the positive linear relationship between measured velocities at 0.6 and at 0.2 depths at the same three riffle sites. The hydrologic conditions and the relationships between velocities at each velocity strata were assumed to be reasonably uniform among the three riffle sites. The velocity data to estimate velocities at 0.8 depths in pools were insufficient to construct habitat criteria that included nose velocities.

Smallmouth bass (MDO)

Smallmouth bass lifestages were comparable in their vertical orientation, with a 18% of all MDO at 0.6 depth, and 81% at 0.8 depth. Wilcoxon/Kruskal-Wallis tests confirmed that MDO lifestages were using different depths (1-way test, X^2 approximation (X^2 = 36.5301, df = 2, p < 0.0001). Tukey-Kramer tests confirmed that adult MDO (MDO) were using the deepest depths, sub-adult MDO (SA MDO) intermediate depths, and juvenile MDO (J MDO; not included in habitat criteria because of small sample sizes) were in the shallowest depths (Table 9). There were no significant differences in mean column velocities among MDO lifestages (p > 0.05; Table 10).

Among the most extensively sampled taxa and lifestages with n = 67 sampling events observing n = 128 individuals, SA MDO was observed over 43 channel indices (X^2 analyses of SA MDO criteria included only n = 61, as 6 events were recorded in the same location and would have resulted in repeated measurements of the same habitat had they been included). Fifty-one percent (51%) of channel indices that SA MDO were observed to use included bedrock, while large gravel and small cobble were included in 37 and 35%, respectively. Silt, sand, and large cobbles were included in 16, 13, and 10%, respectively. More than one type of cover was present at 73% of sampling events and at least 1 type of cover was present at all events. Forty-two percent (42%) of channel indices scored as 0 to 25% embedded, 23% were 25 to 50% embedded, 6% were 50 to 75% embedded, and 29% were 75 to 100% embedded. A channel index that represented large gravel, small cobble, multiple cover types, and low embeddedness accounted for the most SA MDO sampling events (n = 6).

On 18 sampling events, n = 45 MDO were observed in habitats described by 17 channel indices. Bedrock was included in the channel indices where 47% of individual MDO were observed (42% of events). Bedrock, small cobble (36%), silt (29%), large cobble (24%), and large gravel (20%) dominated the substrates where MDO were observed. Cover was extensive as 84% of individual MDO were observed in the presence of multiple cover types and all were observed near at least one type of cover. The (relatively low) velocity preference of adult smallmouth bass manifested itself in channel indices with relatively high degrees of embeddedness. Only 22% of individual MDO were found in areas 0 to 25% embedded, while 42% and 33% were found in areas either 25 to 50%, or 75 to 100% embedded, respectively.

Redbreast sunfish (LAU)

Redbreast sunfish (LAU) were tested for differences in habitat use by lifestage. There were differences in depth (1-way test, X^2 approximation (X^2 = 143.1577, df = 2, p < 0.0001). A Tukey-Kramer HSD test confirmed that SA LAU were at the deepest depths, adult LAU were intermediate, while J LAU were in the shallowest depths (Table 9).

There were differences in the mean column velocities used by LAU lifestages (1-way test, X^2 approximation (X^2 = 56.8913, df = 2, p < 0.0001)). Juvenile LAU were in the fastest waters, LAU were in intermediate velocities, and SA LAU used the slowest mean column velocities (Tukey-Kramer HSD; Table 10).

On 31 sampling events, 126 individual J LAU were observed in habitats described by 27 channel indices. Large gravel was present in the habitats where 50% of J LAU were observed; small cobble in 44%, sand in 31%, silt in 20%, and large cobble in 11%. Multiple cover types were present at the locations where 64% of individual J LAU were observed, while submerged vegetation and interstitial spaces in the substrate were present in the habitats where 14% and 12% of individual J LAU were observed, respectively. While 33% of J LAU were observed in areas with low embeddedness (0 to 25%), 52% were found in areas of very high embeddedness (75 to 100% embedded).

On 31 sampling events, 132 individual SA LAU were observed in habitats described by 24 channel indices. Bedrock was the dominant substrate at 18% of the observed locations of SA LAU, while the prevalence of silt (at 64% of SA LAU observations) revealed the preference of SA LAU for relatively deep and low velocity habitats. Large substrates including large cobble and small boulders, present at 25% and

41% of SA LAU locations, were consistent with that fish's use of complex cover (at 82% of SA LAU locations). Only 3% of SA LAU were observed at locations with no cover. High degrees of embeddedness were prevalent at SA LAU sampling events; 75% of SA LAU locations were > 50% embedded, 25% were < 50% embedded.

Sand was a frequent component of the microhabitats where adult LAU were observed on 30 sampling events, included in 48% of channel indices. Twenty-two percent (22%) of LAU were found over bedrock, while areas where silt (21% of observations) was mixed with large gravel (32% of observations), and small cobble (37% of observations), tended to hold most LAU. Adult redbreast sunfish were also associated with complex cover (74% of observations) and microhabitats where titled bedrock provided velocity shelter (11% of observations). High embeddedness was also estimated at the locations of LAU sampling events, with only 20% of the locations where LAU were observed having embeddedness less than 50%; 80% of LAU observations were recorded in areas with embeddedness > 50%.

Satinfin and Spotfin shiners (*Cyprinella* spp.)

Sixty-one sampling events yielded observations of n = 229 *Cyprinella* spp. *Cyprinella* spp. tended to hold position at 0.6 depth (53%, 106 out of 200), or 0.8 depth (40%, 79 out of 200), but feed higher in the water column (3% feeding at 0.2 depth, 16% moving from 0.8 to feed at 0.6 depth). Based on the nose velocities at 0.8 depths estimated for the three riffle sites, *Cyprinella* spp. were feeding in velocities that were approximately 14 cm/s faster than the velocities in which they were holding (the measured mean column velocity at 0.6 depth at the riffle sites was 33 cm/s (95% CI 27, 40), while the estimated mean nose velocity for *Cyprinella* spp. at 0.8 depth at the three

riffle sites was 19 cm/s (95% CI 15, 23), for a difference of 14 cm/s (95% CI on

difference 7, 22); (Wilcoxon-Kruskal/Wallis test (2-sample test, normal approximation (S

= 1902, Z = -3.7167, p = 0.0002) (1-way test, X^2 approximation (X^2 = 13.8402, df = 1, p

= 0.0002)), a Tukey-Kramer HSD test specified difference)).

Thirty-four channel indices described the 61 locations where *Cyprinella* spp. were

observed. The moderate depths used by *Cyprinella* spp. (mean = 46 cm, SD = 21; Table

9), and relatively high mean column velocities (mean at all sites where *Cyprinella* spp.

were sampled = 25 cm/s, SD = 20; Table 10), are evident in the distribution of the

substrate types that were characteristic. Bedrock was present at 36% of observations,

while large gravel and small cobble were most prevalent (at 54 and 52% of observations,

respectively). Multiple cover types were at 57% of observation sites, while interstitial

space was also common (included in 28% of channel indices), a cover typology

consistent with the depth and velocity preferences of *Cyprinella* spp. Low embeddedness

was a subsequent characteristic (70% of observations were in locations with

embeddedness < 25%, while only 8% of *Cyprinella* spp. were observed where

embeddedness > 75%).

Margined madtom (NIN)

Margined madtoms (NIN) were sampled on 30 sampling events for a total of n =

43 individual NIN. Electroshocking accounted for almost half of all NIN sampled (20

out of 43 individuals or 47%).

All NIN sampled were observed at the substrate, occupying 0.8 depths. Nose

velocities in the riffles where 18 NIN sampling events occurred were estimated to have a

mean of 24 cm/s (SD = 22). Mean column velocities at those same locations were

measured to have a mean of 43 cm/s (SD = 29). Nose velocities for NIN in the riffle sites were thus estimated to be 18 cm/s slower than mean column velocities (95% CI on difference 1, 36; Wilcoxon/Kruskal-Wallis tests (2-sample test, normal approximation (S = 268, Z = -2.0415, p = 0.0412)) (1-way test, X^2 approximation (X^2 = 4.2325, df = 1, p = 0.0397)); a Tukey-Kramer HSD test specified difference).

Nineteen channel indices were used to describe the substrates at 30 NIN sampling events. Bedrock was present at 47% of sampling events, small cobble at 60%, large gravel at 50%, large cobble at 23%, and small gravel at 10% of NIN sampling events. Multiple cover types were present at 87% of sample events, interstitial space at 13%. Embeddedness tended to be low (86% of observations occurred in embeddedness < 50%; 13% > 50% embeddedness). Eight sampling events occurred in habitats described by very similar channel indices (bedrock and either gravel or small cobble, complex cover, and low embeddedness (< 25%)).

River chub (NMI)

Fifty-four sampling events included n = 144 individual NMI. Most (n = 127) NMI were observed while snorkeling although n = 17 were electroshocked via prepositioned electroshocking.

All NMI observed via snorkeling (n = 127) were occupying 0.8 depths. River chubs were observed at 35 sampling events at the three riffle sites. Nose velocities at those locations were estimated to have a mean of 23 cm/s (SD = 18), which was approximately 16 cm/s slower than mean column velocities (95% CI on difference 6, 27; mean column velocities at riffle locations had a mean = 39 cm/s, SD = 24; Wilcoxon/Kruskal-Wallis tests (2-sample test, normal approximation (S = 996, Z = -

2.8899, p = 0.0039)) (1-way test, X^2 approximation (X^2 = 8.3853, df = 1, p = 0.0038)), a Tukey-Kramer HSD test specified difference).

River chubs were relatively consistent in the habitats they used with large gravel, bedrock, small cobble, large cobble, and small gravel regularly occurring substrate types (at 60, 55, 36, 28, and 15% of NMI sampling events, respectively). These five substrate types accounted for the substrates at 98% (52 out of 53) of all NMI sampling events. Complex cover was present at 77% of observations, interstitial space at 15%. Ninety-four percent (94%) of NMI observations were recorded over embeddedness of < 50%; 83% in < 25% embeddedness. Seven out of 53, or 13%, of NMI sampling events occurred in areas with bedrock, large gravel, complex cover, and low (< 25%) embeddedness; the conditions described by the single most frequently noted channel index.

Young-of-the-Year (YOY)

Young-of-the-year (YOY) were observed on 37 occasions, with an average of 9 individuals present at each event. Approximately 318 individual YOY were observed, though counts of YOY were especially problematic given their small size, schooling, and movement. Two-hundred and eighty-six individual YOY were observed by snorkeling and thus had relative depths estimated. Thirty-seven percent (37%, 106 out of 286) of YOY were observed at 0.2 depth, 57% (163 out of 286) at 0.6, and the remaining 6% (17 out of 286), at 0.8 depth.

The locations of the 37 sampling events for YOY were described by 28 channel indices. Only bedrock (43%), large gravel (38%), small cobble (32%), and sand (22%), were present at more than 20% of sampling events. The other substrate types were

present at 13% or fewer sampling events. Complex cover was available at 86% of the

locations where YOY were observed; only 3% of locations had no cover. Embeddedness

was inconsistent as 51% of sampling events occurred in < 25% embeddedness, while

32% recorded embeddedness > 75%, with the 2 central categories having 8% each.

Bedrock and large gravel, with complex cover and low embeddedness, was the most

frequently occurring combination, accounting for the context at 5 out of 37 observations.

Transferability Tests

Because compatible channel indices were not always available, transferability

tests involved only criteria for depths and mean column velocities and composite

suitabilities based on those two parameters. Tables 11 – 13 contain summaries of the T-

statistics generated by the transferability tests. No Data (ND) in Tables 12 and 13 for the

suitable vs. unsuitable tests of Aadland et al.'s (1991) SA MDO criteria indicate that the

range of mean column velocities was sufficiently wide as to render all available habitats

in the South Fork suitable; thus the test failed due to the supposed non-existence of

unsuitable mean column velocities. No Data (ND) in rows of Persinger (2003) data in

those same tables indicate that the data necessary to test criteria for suitable and

unsuitable habitats was unavailable.

Habitat suitability criteria developed on the North Fork of the Shenandoah River

have been successfully transferred to the South Fork for NIN, *Cyprinella* spp., and NMI

(Persinger 2003). Habitat criteria for *Cyprinella* spp. developed from data collected in

the North Fork of the Shenandoah (Persinger 2003) transferred to the *Cyprinella* spp.

observed in this study (composite suitability T >2.5570; see Table 13).

This study tested optimal habitat criteria for *Cyprinella* spp. *developed on the South Fork* in 2003 (Persinger 2003), and those criteria rejected the hypothesis in the optimal vs. non-optimal test for depth and the composite suitability (composite = depth and mean column velocity; T-statistics ≥ 2.6270; Table 11), though the data to test the suitable vs. unsuitable hypothesis was not available. Optimal habitat criteria developed in the South Fork in 2003 for NIN (Persinger 2003) rejected the hypothesis in the optimal vs. non-optimal test against this study's data (T = 1.7755; Table 11), though the data required for the suitable vs. unsuitable test was not available. The results of these two tests were of interest because habitat suitability criteria built by one researcher do not necessarily account for the habitat use observed by another researcher, even in the same river (see Strakosh et al. 2003).

Persinger's (2003) NIN habitat criteria from the North Fork failed to reject the hypothesis of either composite suitability test when applied to the data collected by this study (T < 1.2030; Table 13). Persinger's (2003) North Fork habitat criteria for NMI rejected the hypothesis in the composite (depth and mean column velocity) suitable vs. unsuitable test (T = 1.7938), though not in the optimal vs. non-optimal test (T = 0.3856; Table 13). Transferability tests of Persinger's (2003) LAU and adult MDO habitat criteria from the North Fork were both unsuccessful (all composite suitability T-statistics < 1.6449; Table 13).

Smallmouth bass have been a difficult species to transfer habitat criteria for given their adaptability and the wide range of habitats they can use. Groshens and Orth (1994) were not able to transfer velocity criteria among streams for a size class of MDO equivalent to the SA MDO observed in this study, with only 3 out of 8 X^2 tests

suggesting transferability. Chi-square tests of the mean column velocity criteria from this study also failed to reject the hypothesis that SA MDO was simply using the available velocities in both the optimal vs. non-optimal and the suitable vs. unsuitable tests (T < 1.5970; Tables 6 and 7), though the composite suitabilities were both significant (T > 2.7767; Table 8).

Depth and mean column velocity criteria from the North Anna River (Groshens and Orth 1994) for SA MDO successfully transferred to the SA MDO observed by this study, rejecting both hypotheses (both composite T-statistics > 1.8589), though criteria from Craig Creek (Groshens and Orth 1994) did not (T < 1.4988; Table 13).

Groshens and Orth (1994) found that generalized habitat criteria for SA MDO assembled from several studies did transfer well in their datasets, as it encompassed sufficient variation to accommodate the habitat use of SA MDO. Their finding however did not extend to data collected in this study as the test of suitable vs. unsuitable habitats failed to reject the hypothesis that SA MDO occupied suitable habitats with the same frequency as those habitats defined as unsuitable (T < 1.1519; Table 13).

Depth and mean column velocity criteria for SA MDO aggregated from three Minnesota streams (Aadland et al. 1991) did not transfer to the data collected by this study (suitable vs. unsuitable composite T = 0.8429; Table 13).

Groshens and Orth's (1994) North Anna criteria for adult MDO rejected the hypothesis in the optimal vs. non-optimal test, but failed to reject the hypothesis in the suitable vs. unsuitable test (all T-statistics ≤ 1.4341; Tables 11 and 12). Habitat criteria for MDO in Craig Creek (Groshens and Orth 1994) rejected the null hypothesis of the optimal vs. non-optimal test (composite T = 4.7651), but failed to reject the hypothesis in

the suitable vs. unsuitable test (composite $T = 1.1668$; Table 13). Groshens and Orth's

(1994) generalized criteria for MDO failed to reject either null hypothesis (composite $T \leq$

1.6298; Table 13).

Table 2. Discharge Summary for Sampling Period on the South Fork

Month	Site/Gage	Mean (m^3/s)	Range		Percentile[3]
June	Lynnwood	11	8	18	25th
	Luray	16	11	29	50th
	Front Royal	21	14	34	50th
July	Lynnwood	8	6	13	25th
	Luray	12	9	19	25th
	Front Royal	13	8	23	25th
August	Lynnwood	7	5	14	25th
	Luray	9	6	35	25th
	Front Royal	8	6	25	10th
September	Lynnwood	8	5	30	50th
	Luray	13	9	49	75th
	Front Royal	15	11	42	75th
Mean		12			

[3] USGS WSC 2009

Table 3. Habitat Suitability Criteria for Depth for 9 Frequently Sampled Fish Taxa and Life Stages

Optimal = range within central 50% of data

Marginal = range within central 80% of data

| | | Suitable Range of Depths (cm) | | | |
| | | Lower | | Upper | |
Taxa/Life Stage	n (event)	Marginal	Optimal		Marginal
SA MDO	61	**40**	**52**	**88**	**168**
MDO	*45	**64**	**85**	**168**	**189**
J LAU	31	**24**	**34**	**58**	**73**
SA LAU	31	**40**	**55**	**110**	**134**
LAU	30	27	49	107	137
Cyprinella spp.	61	**27**	**34**	**55**	**73**
NIN	30	21	**27**	**49**	88
NMI	54	3	**34**	**64**	88
YOY	37	27	34	61	113

*Used individual sample size (event n = 18)

Statistically significant at alpha = 0.05[4]

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[4] Statistical significance in Tables 3 through 5 refers to the statistical significance of the T-statistics generated by the criteria contained in Tables 3 through 5. The T-statistics generated by each X^2 test are summarized in Tables 6 through 8. A statistically significant T-statistic indicated that there was a high probability (significance probability $\geq$ 0.95) that the habitat suitability criteria tested accurately described non-random habitat use of the fish sampled. In other words, the habitat criteria in **bold** described the habitat use of fish to a statistically significant degree.

Table 4. Habitat Suitability Criteria for Mean Column Velocity for 9 Frequently Sampled Fish Taxa and Life Stages

Optimal = range within central 50% of data

Marginal = range within central 80% of data

| Taxa/Life Stage | n (event) | Suitable Range of Mean Column Velocities (cm/s) | | | |
| | | Lower | | Upper | |
		Marginal	Optimal		Marginal
SA MDO	61	4	8	37	54
MDO	*45	**2**	**5**	**21**	**36**
J LAU	31	0	5	27	39
SA LAU	31	2	**4**	**20**	54
LAU	30	**2**	6	22	**36**
Cyprinella spp.	61	5	**13**	**39**	67
NIN	30	2	20	44	90
NMI	54	4	18	50	72
YOY	37	2	5	33	54

*Used individual sample size (event n = 18)

Statistically significant at alpha = 0.05

Table 5. Habitat Suitability Criteria for Dominant and Subdominant Substrate Size, Presence and Type of Cover, and Embeddedness, Characterized by Channel Indices, for 9 Frequently Sampled Fish Taxa and Life Stages

Optimal = range within central 50% of data

Marginal = range within central 80% of data

| | | Suitable Range of Channel Indices | | | |
| | | Lower | | Upper | |
Taxa/Life Stage	n (event)	Marginal	Optimal		Marginal
SA MDO	61	36.84	**67.92**	**119.92**	611.94
MDO	*45	**37.94**	**76.92**	**210.94**	**211.94**
J LAU	31	29.94	**64.93**	**87.51**	611.91
SA LAU	31	**28.94**	67.51	117.91	**210.94**
LAU	30	33.94	63.53	117.91	711.91
Cyprinella spp.	61	**56.91**	67.52	117.91	**611.91**
NIN	30	**67.83**	**76.63**	**117.91**	**611.91**
NMI	54	**67.51**	**76.91**	**118.91**	**611.92**
YOY	37	28.94	37.94	116.91	611.91

*Used individual sample size (event n = 18)

Statistically significant at alpha = 0.05

Table 6. T-statistics for Tests of Optimal vs. Non-optimal Habitats, as Defined by the Habitat Suitability Criteria Developed by this Study

Optimal = range within central 50% of data

Marginal = range within central 80% of data

Critical Values for T^5	Significance Probability
2.3263	0.99
1.6449	0.95

Taxa/Life Stage	n (event)	Optimal vs. Non-optimal			
		Depth	Mean Column Velocity	Channel Index	Composite Suitability
SA MDO	61	**4.1070**	1.5970	**1.8288**	**2.7767**
MDO	*45	**6.0810**	**1.6940**	**3.0848**	**5.4048**
J LAU	31	**1.8647**	0.0710	**2.0975**	**2.4790**
SA LAU	31	**3.0790**	**1.7262**	0.1292	**2.2340**
LAU	30	1.5495	1.6129	0.0818	1.4890
Cyprinella spp.	61	**3.0832**	**1.6820**	1.1699	**3.1302**
NIN	30	**1.8946**	1.2026	**1.7483**	**3.3359**
NMI	54	**1.9732**	1.3086	**2.7325**	**2.5955**
YOY	37	0.7507	-0.2815	0.1648	0.3855

*Used individual sample size (event n = 18)

Statistically significant at alpha = 0.05[6]

[5] From Conover 1980

[6] Statistically significant T-statistics in Tables 6 through 8 indicated high probabilities (significance probability ≥ 0.95) that the habitat suitability criteria tested, contained in Tables 3 through 5 accurately described non-random habitat use of the fish sampled.

Table 7. T-statistics for Tests of Suitable vs. Unsuitable Habitats, as Defined by the Habitat Suitability Criteria Developed by this Study

Taxa/Life Stage	n (event)	Suitable vs. Unsuitable			
		Mean Column			Composite
		Depth	Velocity	Channel Index	Suitability
SA MDO	61	**4.4550**	0.7244	1.3143	**4.1564**
MDO	*45	**6.7712**	**1.9673**	**3.0698**	**6.8713**
J LAU	31	**2.4067**	1.0623	0.6275	**2.2529**
SA LAU	31	**3.2791**	0.3163	**1.6945**	**2.9689**
LAU	30	1.2595	**2.2717**	0.3384	**2.2074**
Cyprinella spp.	61	**3.3034**	0.6758	**2.1738**	**3.0768**
NIN	30	1.0918	-1.2904	**2.6721**	**1.8558**
NMI	54	**2.7231**	-0.0128	**2.7861**	**2.0971**
YOY	37	1.0376	-0.5283	0.1245	0.4711

*Used individual sample size (event n = 18)

Statistically significant at alpha = 0.05

Table 8. Summary of T-Statistics Generated by X^2 Tests and Conclusions for the Composite Habitat Suitability Criteria Developed by this Study

Taxa/Life Stage	n (event)	Optimal vs. Non-optimal Composite Suitability	Suitable vs. Unsuitable Composite Suitability	Validated
SA MDO	61	2.7767	4.1564	Yes
MDO	*45	5.4048	6.8713	Yes
J LAU	31	2.4790	2.2529	Yes
SA LAU	31	2.2340	2.9689	Yes
LAU	30	1.4890	2.2074	No
Cyprinella spp.	61	3.1302	3.0768	Yes
NIN	30	3.3359	1.8558	Yes
NMI	54	2.5955	2.0971	Yes
YOY	37	0.3855	0.4711	No

*Used individual sample size (event n = 18)

Statistically significant at alpha = 0.05

Table 9. Means-Based Summary Statistics for Depth[7]

Taxa/Life Stage	n (event)	n (individual)	Depth (cm)			
			Mean	SD	95% CI	
SA MDO	61	128	88	43	79	94
MDO	18	45	122	52	107	137
J LAU	31	126	52	24	49	55
SA LAU	31	132	110	34	104	116
LAU	30	132	98	46	88	104
Cyprinella spp.	61	229	46	21	43	49
NIN	30	43	43	27	34	52
NMI	54	144	55	21	49	58
YOY	37	318	67	37	61	70

Table 10. Summary Statistics for Mean Column Velocity

Taxa/Life Stage	n (event)	n (individual)	Mean Column Velocity (cm/s)			
			Mean	SD	95% CI	
SA MDO	61	128	21	18	17	24
MDO	18	45	14	12	10	17
J LAU	31	126	16	14	14	19
SA LAU	31	132	8	11	6	9
LAU	30	132	12	12	10	14
Cyprinella spp.	61	229	25	20	23	28
NIN	30	43	30	24	23	37
NMI	54	144	35	22	31	39
YOY	37	318	17	15	15	19

[7] Means were calculated using individual sample sizes

Table 11. Summary of Transferability Tests of Optimal vs. Non-optimal Habitat Suitability Criteria

Statistically significant at alpha = 0.05[8]

Taxa/Lifestage	Author	River	T-statistics		
			Optimal vs. Non-optimal		
			Depth	Mean Column Velocity	Composite Suitability
Cyprinella spp.	Persinger (2003)	South Fork, Shenandoah	**3.9609**	1.5547	**2.6270**
Cyprinella spp.	Persinger (2003)	North Fork, Shenandoah	**3.9218**	**2.0076**	**3.0468**
NIN	Persinger (2003)	South Fork, Shenandoah	0.6185	**2.2338**	**1.7755**
NIN	Persinger (2003)	North Fork, Shenandoah	-0.4776	**2.0963**	1.2030
NMI	Persinger (2003)	North Fork, Shenandoah	-0.2680	1.6380	0.3856
LAU	Persinger (2003)	North Fork, Shenandoah	0.0457	1.3654	1.3248
MDO	Persinger (2003)	North Fork, Shenandoah	-5.1180	-0.1930	-2.7670
SA MDO	Groshens and Orth (1994)	North Anna	**4.4684**	0.5585	**3.2777**
SA MDO	Groshens and Orth (1994)	Craig's Creek	-0.7820	**2.2126**	-1.0920
SA MDO	Groshens and Orth (1994)	Generalized Criteria	**3.2322**	**1.8124**	**2.4401**
SA MDO	Aadland (1991)	Minnesota, 3 streams	**3.9225**	**1.7045**	**2.3900**
MDO	Aadland (1991)	Minnesota, 3 streams	**4.0096**	-0.2920	**2.3673**
MDO	Groshens and Orth (1994)	North Anna	**2.3909**	**2.3769**	**2.7058**
MDO	Groshens and Orth (1994)	Craig's Creek	**5.2085**	0.2022	**4.7651**
MDO	Groshens and Orth (1994)	Generalized Criteria	**4.0096**	0.8574	**1.6298**

[8] Statistically significant T-statistics in Tables 11 through 13 indicated that the habitat suitability criteria developed by the authors included in Tables 11 through 13 described the habitat use of fish observed by this study in the South Fork to a statistically significant degree ($\alpha \leq 0.05$).

Table 12. Summary of Transferability Tests of Suitable vs. Unsuitable Habitat Suitability Criteria

Statistically significant at alpha = 0.05[6]

			T-statistics Suitable vs. Unsuitable		
Taxa/Lifestage	Author	River	Depth	Mean Column Velocity	Composite Suitability
Cyprinella spp.	Persinger (2003)	South Fork, Shenandoah	ND	ND	ND
Cyprinella spp.	Persinger (2003)	North Fork, Shenandoah	**2.8649**	-0.3543	**2.5570**
NIN	Persinger (2003)	South Fork, Shenandoah	ND	ND	ND
NIN	Persinger (2003)	North Fork, Shenandoah	0.9420	-0.6700	0.1049
NMI	Persinger (2003)	North Fork, Shenandoah	**1.7222**	0.7535	**1.7938**
LAU	Persinger (2003)	North Fork, Shenandoah	-1.2370	**2.5145**	1.2614
MDO	Persinger (2003)	North Fork, Shenandoah	-6.4854	**2.4970**	-4.2704
SA MDO	Groshens and Orth (1994)	North Anna	1.5323	1.0916	**1.8589**
SA MDO	Groshens and Orth (1994)	Craig's Creek	1.0455	1.2957	1.4988
SA MDO	Groshens and Orth (1994)	Generalized Criteria	0.7773	0.7071	1.1519
SA MDO	Aadland (1991)	Minnesota, 3 streams	0.8429	ND	0.8429
MDO	Aadland (1991)	Minnesota, 3 streams	0.6020	0.8543	1.0500
MDO	Groshens and Orth (1994)	North Anna	-3.0210	1.4341	-1.6240
MDO	Groshens and Orth (1994)	Craig's Creek	1.0124	1.2927	1.1668
MDO	Groshens and Orth (1994)	Generalized Criteria	-6.5300	**2.7329**	-3.1600

Table 13. Summary of Transferability Tests of Composite Habitat Suitability Criteria and Transferability Decisions

Statistically significant at alpha = 0.05[b]

Taxa/Lifestage	Author	River	T-statistics Composite Suitability (Depth and Vel.)		
			Optimal vs. Non-optimal	Suitable vs. Unsuitable	Transferable
Cyprinella spp.	Persinger (2003)	South Fork, Shenandoah	**2.6270**	ND	ND
Cyprinella spp.	Persinger (2003)	North Fork, Shenandoah	**3.0468**	**2.5570**	**Yes**
NIN	Persinger (2003)	South Fork, Shenandoah	**1.7755**	ND	ND
NIN	Persinger (2003)	North Fork, Shenandoah	1.2030	0.1049	No
NMI	Persinger (2003)	North Fork, Shenandoah	0.3856	**1.7938**	No
LAU	Persinger (2003)	North Fork, Shenandoah	1.3248	1.2614	No
MDO	Persinger (2003)	North Fork, Shenandoah	-2.7670	-4.2704	No
SA MDO	Groshens and Orth (1994)	North Anna	**3.2777**	**1.8589**	**Yes**
SA MDO	Groshens and Orth (1994)	Craig's Creek	-1.0920	1.4988	No
SA MDO	Groshens and Orth (1994)	Generalized Criteria	**2.4401**	1.1519	No
SA MDO	Aadland (1991)	Minnesota, 3 streams	**2.3900**	0.8429	No
MDO	Aadland (1991)	Minnesota, 3 streams	**2.3673**	1.0500	No
MDO	Groshens and Orth (1994)	North Anna	**2.7058**	-1.6240	No
MDO	Groshens and Orth (1994)	Craig's Creek	**4.7651**	1.1668	No
MDO	Groshens and Orth (1994)	Generalized Criteria	1.6298	-3.1600	No

Discussion

Habitat Suitability Criteria

Because the event sample sizes of this study were lower than 55 some extent of error was likely present in the analyses (Thomas and Bovee 1993). Type I errors improperly rejecting nonsignificant statistical null hypotheses increase when occupied habitats drop below 55 (Thomas and Bovee 1993). Type II errors that fail to reject significant statistical null hypotheses and thus fail to detect patterns in the data increase when fewer than 200 unoccupied habitats are included in X^2 tests (Thomas and Bovee 1993). Because there were less than 200 unoccupied habitats within the suitable range (suitable = optimal + marginal), X^2 tests of Ho_1 (that optimal habitats were occupied with the same frequency as marginal habitats) tested optimal habitats against non-optimal habitats rather than optimal vs. marginal habitats.

Event sample sizes were used for X^2 analyses with the exception of those for adult MDO. Though event sample sizes are the norm (Thomas and Bovee 1993; Freeman et al. 1997), Aadland et al. (1991) point out that in situations where there is reason to believe that specific microhabitat conditions lead to higher concentrations of specific taxa, the use of weighted sample sizes is appropriate. There was no reason to believe that MDO were schooling or otherwise selecting habitat dependently on their conspecifics based on observations made by this study or reported in the literature. It was assumed therefore that individual MDO were selecting habitats independently from one another and that

many individual MDO in a given microhabitat reflected preference for the conditions found in that habitat.

Because of the association between low sample sizes and more frequent type I errors (Thomas and Bovee 1993), it is possible that the type I error rate for the adult MDO criteria was relatively high (there were no differences in the statistical decisions reached when the same tests were performed using the event sample size of n (un-weighted) = 18). The T-statistics for the MDO criteria tended to be much higher than those generated in the other tests which may be indicative of type I errors. However, since the X^2 tests relied on single sets of minimum and maximum values for each habitat parameter, because the ranges of the habitat parameters that MDO in this study were observed in were expectedly wide, and because MDO were found to consistently occupy only habitats within these ranges, the MDO criteria can be viewed as viable despite the potential statistical weakness.

In the X^2 tests used to confirm the habitat criteria generated by this study (Table 8), tests of optimal vs. non-optimal habitats often resulted in instances where occupied-non-optimal counts were higher than occupied-optimal counts even in instances when the T-statistic was significant. Because these were tests of a narrow range of optimal habitats, significant T-statistics identified instances where even a relatively low percentage of occupancy occurred in a set of habitat conditions that were very seldom available, suggesting a high degree of preference for those conditions. Thus the number of unoccupied-optimal habitats was just as important as the number of occupied-optimal habitats in determining the T-statistic. For example, in the composite suitability table of the test of optimal vs. non-optimal habitats for SA MDO (see Table 14), the T-statistic =

2.7767 was statistically significant at a significance probability of 0.99 (p < 0.01; Conover 1980). While 13 of 36 optimal habitats were occupied, and 48 of 285 non-optimal habitats were also occupied, 36% (13 out of 36) of the available optimal habitats were occupied vs. only 17% (48 out of 285) of the available non-optimal habitats that were occupied. This suggested that while only 13 of n = 61 SA MDO were occupying habitats defined as optimal, those fish were demonstrating a preference for those rare habitats by occupying them with a frequency disproportionate to their availability.

This pattern of occupied- optimal and non-optimal was not seen in the tests of suitable vs. unsuitable, where in coincidence with significant T-statistics, occupied suitable always outnumbered occupied unsuitable, as would be expected of appropriately defined suitable habitat. For SA MDO (Table 15), of 125 suitable habitats available, 38 (or 30%), were occupied vs. only 23 out of 196 unsuitable habitats that were occupied (12%). The T-statistic for the composite suitability criteria was statistically significant at 4.1564 (at a significance probability of 0.99; Conover 1980). This test result suggested that the habitat criteria tested represent the habitat use of SA MDO in the South Fork to a statistically significant degree. Secondly, these results suggested that SA MDO exhibited preference for the range of habitats defined as suitable because the use of those habitats by SA MDO was disproportionate to the availability of those habitats (Freeman et al. 1997).

There were also incidents when the use and availability of a given range of a single habitat variable did not generate a significant T-statistic. In those instances, while sometimes many fish occupied the suitable range of mean column velocities for example, those velocities were widely available (see Appendix III for an example). In coincidence

with significant composite suitability T-statistics, these results suggested (though did not statistically test), that the selection of mean column velocities was not the most important determinant of the habitat use of SA MDO; that other habitat variables, interactions between several habitat variables, or variables not included in this analysis, were steering the habitat selection of SA MDO in the South Fork. Other studies have found that the mean column velocities used by SA MDO were not significantly different from those available (Groshens and Orth 1994).

In all instances but two (to be discussed), composite suitability tests of the habitat criteria produced by this study were statistically significant despite insignificant T-values for univariate criteria. With 15 out of 18 X^2 tests significant at $\alpha = 0.05$ (7 of those 15 significant at $\alpha = 0.01$; Table 8), the habitat suitability criteria developed in this study successfully described multivariate habitat preferences of SA MDO, MDO, J LAU, SA LAU, the *Cyprinella* spp., NIN, and NMI, in the South Fork of the Shenandoah River. The composite test of optimal vs. non-optimal habitats for LAU was not statistically significant (T = 1.4890; Appendix III), while the test result for the composite suitable vs. unsuitable habitats for LAU was statistically significant at $\alpha = 0.05$ (T = 2.2074; Appendix III).

The composite test of optimal vs. non-optimal habitats for YOY was not statistically significant (T = 0.3855; Appendix III), nor was the result for the composite suitable vs. unsuitable habitats for YOY (T = 0.4711; Appendix III). Young-of-the-year were recorded in the field because they could be, but were expected to be problematic as multiple species fell under that label. Criteria for YOY were widely ranging and

inconsistent. That the YOY criteria failed to reject either hypothesis in the X^2 tests demonstrated that the X^2 tests could prevent type I errors.

Noting the argument that testing habitat criteria on the data used to construct them is questionable (Guay et al. 2003), the X^2 tests used here to confirm the criteria developed in this study were merely a means to confirm the plausibility of those criteria. The rejection of the criteria for LAU and YOY illustrated the utility of this tactic. The X^2 tests were used to vet the criteria to be used in the pending IFIM analysis for the South Fork.

One of the goals of this study was to construct habitat criteria based on nose velocities and relative depths. A thorough analysis of nose velocities will require that velocity measurements be obtained at or just above the substrate at each habitat measurement, including those for available habitat. This study estimated velocities at 0.8 depth at the three riffle sites based on an equation derived from measurements obtained at 0.2, 0.6, and 0.8 depths at a representative riffle site. Velocities at 0.8 depths at the pool sites were not retrievable from the ADCP measurements as was expected, likely because the measured velocities at that depth in those habitats were within the range of measurement error for the ADCP. The cost of not measuring velocities at 0.8 depths was the loss of meaningful nose velocities that could have made a significant contribution to the criteria assembled. Relative depth being a function of depth, the later measurement sufficiently delineated preferred depths.

Recent and severe fish kills on the South Fork of the Shenandoah River have had an unknown effect on the habitat use of the fish observed during this study. It is known that at least the populations of MDO, Largemouth bass (*Micropterus salmoides*, MSA),

and LAU, suffered extensive mortality in widespread fish kill events in the first half of this decade. It is not known to what extent, if any, non-game fishes suffered higher than background rates of mortality during the fish kills. Secondly, as many of the non-game species are preyed upon by gamefish, reduced predator densities, at least in the short-term, had unknown affects on the habitat use of non-game fishes. Different population densities of central stonerollers (*Campostoma anomalum*) have been shown to affect the habitat use of that species and the relative importance of the physical habitat parameters that steered it (Orth and Maughan 1982). Largemouth bass are known to affect the habitat use of central stonerollers (Power and Matthews 1983), thus fish kills that substantially reduced populations of piscivores, or at least altered the age structure of the piscivorous community, would be expected to affect the behavior and habitat selection of forage fishes.

It is plausible that the ranges of habitats used by the fish community became more narrow as fish were able to exploit and concentrate in optimal habitats due to reduced inter- and intraspecific competition and predation. It is equally plausible that the habitat use of the fish community could have become more variable if reduced inter- and intraspecific competition ceased to constrain habitat selection to the same extent than it did when standing stock (Moyle and Baltz 1985), and the standing stock of predators, was higher.

It is also unknown what condition with regard to standing stock and carrying capacity that the fish community was in at the time field work was conducted. There is reason to suspect that the MDO and LAU populations are rebounding due to high recruitment success during the fish kill years and there were no fish kills during the

spring of 2008 on the scale of previous years (Reeser 2008). The effects of the presence

of strong age classes of SA MDO on the habitat use of the cyprinids, for example, or

even on SA MDO themselves, are unknown.

Transferability Tests

Transferable habitat suitability criteria suggest that those criteria accurately reflect

microhabitat use or preference and that those preferences apply at the species level, at

least in the streams where the criteria were tested (Freeman et al. 1997). Transferability

is generally defined as the ability of habitat criteria to differentiate optimal from merely

marginal habitats, as well as to differentiate suitable habitats from unsuitable habitats

(Thomas and Bovee 1993; Strakosh et al. 2003).

Nine out of 15 composite optimal vs. non-optimal transferability tests were

statistically significant (Table 13), while only 3 of 13 of the composite suitable vs.

unsuitable tests were (Table 13); a pattern of results opposite from what Groshens and

Orth (1994) reported. The range of optimal habitat, particularly when defined as the

range used by only a percentage of observations as it was here and elsewhere (Bovee et

al. 1998; Strakosh et al. 2003), is relatively narrow by definition. The subsequent

inference is that if fishes genuinely prefer a specific range of habitat that is optimal i.e.

habitat that increases their reproductive success to its maximum, that preference would

not be expected to differ among streams to the same extent as the range of suitable

habitat; suitable habitats being those merely sufficient to maintain some extent of

reproduction.

Because the range of suitable habitat for a species will always be at least slightly

wider and more variable than the range of optimal habitat, it is also likely that differences

in the available habitats in different streams are more manifest in suitable vs. unsuitable tests than in tests of optimal vs. non-optimal habitat. The differences between the habitats used and those not used by a species are likely to be larger than the differences between the habitats used by a species and those that are available (Guay et al. 2003). Tests of suitable vs. unsuitable habitats are more like tests of used vs. not used, live or die, reproduction or no reproduction, while tests of optimal vs. non-optimal are more like tests of better vs. good, or habitats used vs. habitats available. The inference that a narrow range of optimal habitat could be consistent for a given species throughout its geographic range (or at least in more than one stream within a region for example), combined with smaller differences between optimal and marginal habitats, might be factors that help explain why more transferability tests of optimal composite habitat suitability criteria rejected the null hypothesis than did tests of suitable composite habitat criteria (Table 13).

Habitat specialists, with narrower habitat suitability criteria by definition, and riffle species in particular, might tend to transfer criteria better than generalists (Freeman et al. 1997; Maki-Petays et al. 2002). One explanation for this is that the narrow ranges of habitats used by riffle fishes and other habitat specialists are *required* by those species wherever they are found; they're inflexible, discrete criteria that those fishes have adapted to throughout their evolution. Habitat generalists are able to use a wide range of habitats. Because a variety of habitats will support habitat generalist species, it is reasonable to expect to find members of generalist species in a variety of microhabitats throughout their geographic range. The likelihood of transferring habitat suitability criteria among streams could be reduced for generalists because of the variety of habitats

that they can inhabit. Other research suggests that wider habitat criteria might facilitate transferability (Groshens and Orth 1994; Conklin et al. 1995) and that assembling generalized habitat criteria from several sources to construct a wide set of criteria (Aadland et al. 1991; Maki-Petays et al. 2002) could increase transferability.

That only 2 out of 13 attempts to reject the two hypotheses that define transferability were successful (Table 13) added further dubious evidence to the concept of transferable habitat suitability criteria. Habitat criteria can fail to adequately describe habitat use if the wrong variables are measured or if biotic interactions such as predation (Power and Matthews 1983; Baker and Coon 1997) are steering it.

Only 1 of the 5 tests of habitat criteria from the North Fork of the Shenandoah (Persinger 2003) rejected both hypotheses to transfer to the South Fork; criteria for the *Cyprinella* spp. Persinger's (2003) unpublished species list from the North Fork includes all of the fishes sampled by this study and several species not sampled by this study. Those fishes noted in the North Fork and not the South Fork, bluehead chub (*Nocomis leptocephalus*) and rosyside dace (*Clinostomus funduloides*) for example, are thought to prefer smaller streams closer in scale to the North Fork (Jenkins and Burkhead 1994), thus their absence from the South Fork samples was not surprising. It was assumed that the similarity of the fish communities of each river would favor transferable criteria because the biotic influences that influence habitat selection in each river would be comparable. Secondly, while the average discharge is higher in the South Fork, the relative frequencies of each mesohabitat type in each river are very similar as Table 1 illustrated, thus the habitats that are available in each river are also very similar.

That habitat criteria from the North Fork (Persinger 2003) did not transfer for 4 out of 5 fishes casts doubt on the feasibility of transferable habitat criteria. Larger sample sizes from each stream and access to the complete datasets from each river so that transferability could have been tested in both directions might have suggested other conclusions, but this study was not able to support an exemplary degree of transferability between the North Fork and South Fork of the Shenandoah River. In accordance with Freeman et al.'s (1997) findings that the criteria of riffle species might tend to transfer better, it was Persinger's (2003) *Cyprinella* spp. criteria from the North Fork that rejected both hypotheses to transfer to the South Fork. The *Cyprinella* spp. criteria produced by this study found *Cyprinella* spp. in slightly deeper and slower water than what Freeman et al. (1997) considered riffle criteria, but *Cyprinella* spp. tended to occupy shallower and faster habitats relative to the other fishes observed in this study.

Only a preliminary investigation was made into the presence of habitat guilds in the South Fork. The dataset collected for this study did not contain adequate sample sizes of enough species to make the identification of habitat guilds feasible.

More work is needed to establish a link between the weighted usable area that PHABSIM produces based on habitat criteria and standing stock (Orth and Maughan 1982; Freeman et al. 1999). Determining if there is a relationship between standing stock and optimal and suitable habitats could support or reject the validity and usefulness of evaluating fish habitat in those terms.

Table 14. Results of the X^2 Test of Composite Suitability Criteria (depth + mean column velocity + channel index) of Optimal vs. Non-optimal Habitats for SA MDO (n = 61). Values in the Table are Counts of Either Fish (Occupied) or Habitat Data Points (Unoccupied).

	Composite suitability table		
	Optimal	Non-optimal	TOTAL
OCCUPIED	13	48	61
UNOCCUPIED	23	237	260
TOTAL	36	285	321
T =	2.7767		

(Table reproduced from Bovee, personal communication 2008)
Statistically significant at α = 0.05

Table 15. Results of the X^2 Test of Composite Suitability Criteria (depth + mean column velocity + channel index) of Suitable vs. Unsuitable Habitats for SA MDO (n = 61). Values in the Table are Counts of Either Fish (Occupied) or Habitat Data Points (Unoccupied).

	Composite suitability table		
	SUITABLE	UNSUITABLE	TOTAL
OCCUPIED	38	23	61
UNOCCUPIED	87	173	260
TOTAL	125	196	321
T =	4.1564		

(Table reproduced from Bovee, personal communication 2008)
Statistically significant at α = 0.05

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Appendix I. Table of Station Numbers, Names, and Coordinates of USGS Study Sites and the Nearest Stream Gages

Table 16. Table of station numbers, names, and coordinates of USGS study sites and the nearest stream gages.

Station No.	Station Name	Latitude (NAD83)	Longitude (NAD83)	PID (local name)
0162910630	S F SHENANDOAH RIVER AB NAKED CR NR SHENANDOAH, VA	382729	-783800	River Bend Pool
01629510	S F SHENANDOAH RIVER BELOW ROUTE 211 NR HAMBURG, VA	383920	-783230	Kauffman Mill
01630555	SOUTH FORK SHENANDOAH RIVER AT OAK HILL, VA	384457	-782542	Route 611
01630600	S F SHENANDOAH RIVER ABOVE DRY RUN AT COMPTON, VA	384701	-782234	Compton Pool
01630790	S F SHENANDOAH RIVER BL GOONEY RUN NR LIMETON, VA	385238	-781526	Thunderbird Farms
01628500	S F SHENANDOAH RIVER NEAR LYNNWOOD, VA	381921	-784518	Lynnwood Gage
01629500	S F SHENANDOAH RIVER NEAR LURAY, VA	383846	-783206	Luray Gage
01631000	S F SHENANDOAH RIVER AT FRONT ROYAL, VA	385450	-781240	Front Royal Gage

Appendix II. List of all Species Sampled

Table 17. Abbreviations, Common and Scientific Names, and Sample Sizes for all Fishes Sampled

Abbreviation	Common Name	Scientific Name	n (event)	n (individual)
ANA	yellow bullhead	*Ameiurus natalis*	3	4
ARO	American eel	*Anguilla rostrata*	1	1
ARU	rock bass	*Ambloplites rupestris*	4	5
CAN	central stoneroller	*Campostoma anomalum*	14	52
CCA	common carp	*Cyprinus carpio*	4	34
CCO	white sucker	*Catostomus commersoni*	2	21
Cottus spp.	Blueridge sculpin	*Cottus caeruleomentum*	1	2
Cyprinella spp.	satinfin shiner	*Cyprinella analostana*		
Cyprinella spp.	spotfin shiner	*Cyprinella spiloptera*	61	229
EBL	greenside darter	*Etheostoma blennioides*	2	2
HNI	northern hogsucker	*Hypentelium nigricans*	2	3
IPU	channel catfish	*Ictalurus punctatus*	6	38
J ANA	juvenile yellow bullhead	*Ameiurus natalis*	1	1
J ARU	juvenile rock bass	*Ambloplites rupestris*	4	5
J EBL	juvenile greenside darter	*Etheostoma blennioides*	3	5
J HNI	juvenile northern hogsucker	*Hypentelium nigricans*	1	1
J IPU	juvenile channel catfish	*Ictalurus punctatus*	1	1
J LAU	juvenile redbreast sunfish	*Lepomis auritus*	31	126
J LCO	juvenile common shiner	*Luxilus cornutus*	1	2
J LGI	juvenile pumpkinseed	*Lepomis gibbosus*	2	2
J LMA	juvenile bluegill sunfish	*Lepomis macrochirus*	4	9

J MDO	juvenile smallmouth bass	*Micropterus dolomieu*	21	35
J PNT	juvenile bluntnose minnow	*Pimephales notatus*	1	5
LAU	redbreast sunfish	*Lepomis auritus*	30	132
LCO	common shiner	*Luxilus cornutus*	23	91
LCY	green sunfish	*Lepomis cyanellus*	5	18
LGI	pumpkinseed	*Lepomis gibbosus*	6	13
LMA	bluegill sunfish	*Lepomis macrochirus*	8	24
MDO	smallmouth bass	*Micropterus dolomieu*	19	45
MSA	largemouth bass	*Micropterus salmoides*	11	18
NCR	golden shiner	*Notemigonus crysoleucas*	1	1
NIN	margined madtom	*Noturus insignis*	30	43
NMI	river chub	*Nocomis micropogon*	54	144
Notropis spp.	rosyface shiner	*Notropis rubellus*		
Notropis spp.	comely shiner	*Notropis amoenus*	16	64
NPR	swallowtail shiner	*Notropis procne*	1	2
NHU	spottail shiner	*Notropis hudsonius*	4	34
PMI	black crappie	*Pomoxis nigromaculatus*	1	20
PNT	bluntnose minnow	*Pimephales notatus*	13	76
RCA	longnose dace	*Rhinichthys cataractae*	7	10
SA IPU	sub-adult channel catfish	*Ictalurus punctatus*	2	2
SA LAU	sub-adult redbreast sunfish	*Lepomis auritus*	31	132
SA LCY	sub-adult green sunfish	*Lepomis cyanellus*	2	10
SA LGI	sub-adult pumpkinseed	*Lepomis gibbosus*	1	4
SA LMA	sub-adult bluegill sunfish	*Lepomis macrochirus*	1	6
SA MDO	sub-adult smallmouth bass	*Micropterus dolomieu*	67	128
SA MSA	sub-adult largemouth bass	*Micropterus salmoides*	5	17
SCO	fallfish	*Semotilus corporalis*	4	12
Total			512	1629

Appendix III. Chi-square Table

Figure Critical T-values (Conover, 1980; reproduced from Bovee 2008).

Critical Values of T	Significance probability
2.3263	0.99
1.6449	0.95
1.2816	0.9
1.0364	0.85
0.8416	0.8

Smallmouth bass (MDO)

Sub-adult MDO

Results of Chi-Square Tests of Optimal vs. Non-optimal Depth for SA MDO (n = 61).

	Depth suitability table		
	Optimal	Non-optimal	TOTAL
OCCUPIED	31	30	61
UNOCCUPIED	63	197	260
TOTAL	94	227	321
T =	4.1070		

Results of Chi-Square Tests of Optimal vs. Non-optimal Mean Column Velocities for SA MDO (n = 61).

	Velocity suitability table		
	Optimal	Non-optimal	TOTAL
OCCUPIED	31	30	61
UNOCCUPIED	103	157	260
TOTAL	134	187	321
T =	1.5970*		

*not significant at $\alpha = 0.05$

Results of Chi-Square Tests of Optimal vs. Non-optimal Channel Indices for SA MDO (n = 61).

	Channel Index suitability table		
	Optimal	Non-optimal	TOTAL
OCCUPIED	32	29	61
UNOCCUPIED	103	157	260
TOTAL	135	186	321
T =	1.8288		

Results of Chi-Square Tests of Optimal vs. Non-optimal Composite Suitability for SA MDO (n = 61).

	Composite suitability table		
	Optimal	Non-optimal	TOTAL
OCCUPIED	13	48	61
UNOCCUPIED	23	237	260
TOTAL	36	285	321
T =	2.7767		

Results of Chi-Square Tests of Suitable vs. Unsuitable Depths for SA MDO (n = 61).

	Depth suitability table		
	SUITABLE	UNSUITABLE	TOTAL
OCCUPIED	54	7	61
UNOCCUPIED	151	109	260
TOTAL	205	116	321
T =	4.4550		

Results of Chi-Square Tests of Suitable vs. Unsuitable Mean Column Velocities for SA MDO (n = 61).

	Velocity suitability table		
	SUITABLE	UNSUITABLE	TOTAL
OCCUPIED	48	13	61
UNOCCUPIED	193	67	260
TOTAL	241	80	321
T =	0.7244*		

*not significant at $\alpha = 0.05$

Results of Chi-Square Tests of Suitable vs. Unsuitable Channel Indices for SA MDO (n = 61).

	Channel Index suitability table		
	SUITABLE	UNSUITABLE	TOTAL
OCCUPIED	51	10	61
UNOCCUPIED	197	63	260
TOTAL	248	73	321
T =	1.3143*		

*not significant at $\alpha = 0.05$

Results of Chi-Square Tests of Suitable vs. Unsuitable Composite Suitability for SA MDO (n = 61).

	Composite suitability table		
	SUITABLE	UNSUITABLE	TOTAL
OCCUPIED	38	23	61
UNOCCUPIED	87	173	260
TOTAL	125	196	321
T =	4.1564		

Vita

Robert Clayton ("Clay") Ramey was born in Fairfax, Virginia in 1979. He lived in Falls Church, Virginia, until 1992, when the family moved to Woodstock, Virginia, near the North Fork of the Shenandoah River. The author received a B.S. from Radford University, in Radford, Virginia, in English, with a minor in geography, in May of 2001. The author received a M.S. in Environmental Studies from the Virginia Commonwealth University Center for Environmental Studies in May of 2009. Clay and his lovely wife Julie headed west shortly after the completion of this thesis.