Longitudinal Patterns of Community Structure for Stream Fishes in a Virginia Tailwater

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ABSTRACT

I examined the abundance, composition, and distribution of 34 fishes within the first 24 km below Philpott Dam on the Smith River, a hydropeaking system in Virginia. Fish were sampled at 12 sites over 8 time periods ranging from 2000 to 2002 across April, June, and October. I evaluated spatial and temporal change in fish community characteristics. Species demonstrated persistent trends in abundance, diversity, and composition throughout the duration of the study. Fish abundance and diversity generally increased with increasing distance from the dam. Fish composition changed minimally across seasons and years, indicating consistent fish assemblages. Distributional patterns suggested a strong response to thermal gradients and presence of tributaries. I concluded that temperature and tributary location directly influence fish community patterns in the Smith River and that the patterns are persistent over space and time.

I characterized spawning microhabitat use and availability, and tested transferability of spawning microhabitat criteria for *Etheostoma flabellare* and *Nocomis leptocephalus*, two of the most common species in the Smith River. *E. flabellare* preferentially selected small and large cobble size rocks for their spawning rock. *N. leptocephalus* selected areas with slower demersal and mean water column velocities in which to build their mounds. Transferability tests were conducted using spawning microhabitat criteria from the unregulated Roanoke River (Smith 1999). The logistic regression model developed for *E. flabellare* by Smith (1999), using information on the diameter of the spawning rock, silt, and embeddedness, transferred with most success with over half of the spawning sites and available sites correctly classified in the Smith River.

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PROJECT NARRATIVE

Fish community ecology has been studied in a wide array of habitats and ecological regions. Patterns in abundance, diversity, and distribution are the most commonly assessed fish community characteristics. Knowledge of these attributes interests ecologists because of the implications for conservation. Studies of fish community structure in tailwater habitats have been fewer (Bain et al. 1988, Kingsolving and Bain 1993, Travnichek and Maceina 1994) though numerous studies have investigated stream flow in relation to fish communities (Aadland 1993, Poff and Ward 1989, Schlosser 1985, Starrett 1951). Because the most dramatic temporal variability in the physical environment of stream ecosystems is fluctuating flow (Schlosser 1991), understanding patterns of fish community structure in hydropeaking tailwaters is highly important.

The Smith River tailwater, created by the construction of the U.S. Army Corps of Engineers' Philpott Dam in 1952, is home to 36 fish species among which are brown trout, stocked rainbow trout, and 34 non-salmonid species, including one federally endangered species, the Roanoke logperch. These fish are exposed to great environmental variability with high fluctuation in flows and temperature because of Philpott's water releases. I seek to learn about the community of non-salmonid fish and what factors are important for these species to persist in such a changing environment.

I will evaluate fish community structure in terms of abundance, diversity, and distribution as well as assess how longitudinal patterns of fish relate to environmental factors. I will also characterize spawning success for two species, *Etheostoma flabellare* and *Nocomis leptocephalus*, because seasonal and annual variation in stream flow and temperature can induce great variation in reproductive success of fishes, thereby causing change in fish composition and species richness (Angermeier and Schlosser 1989). I will quantify spawning microhabitat and test transferability of spawning microhabitat criteria between the Smith River and the Roanoke River, an unregulated river. Results from these studies are expected to provide information needed to optimally manage the water resources of the Smith River.

CHAPTER 1: Community structure of fishes in the Smith River in relation to longitudinal and environmental gradients below Philpott Dam

ABSTRACT

Community structure of a diverse warmwater fish assemblage was examined in a cool tailwater to discern patterns of abundance, diversity, and distribution in relation to longitudinal and environmental gradients below the dam. I evaluated data across 3 seasons and 3 years during which the peaking flows and temperatures varied. Analyses determined that abundance and diversity did not change significantly between time periods (Kruskal Wallis $p>0.05$). Patterns of abundance and diversity increased with distance from the dam and peaked at tributary junctions. Fish composition was persistent during the study despite changing environmental conditions and faunal similarity increased with increasing distance from the dam. Longitudinal patterns of fish reflected a response to a gradient of increasing temperature and attenuating flows. Multiple linear regression identified mean monthly temperature, temperature depressions, and tributary location as the variables which explained a high level of variability in fish abundance. The observed fish assemblage appears to exist in well-developed patterns under the constructs of high environmental variability. Yet, fish populations do not appear to be stabilized because numbers of individual species highly fluctuated during the study.

INTRODUCTION

Ecological Effects of Flow Regulation

The alteration of flow by hydroelectric dams creates disturbance outside the natural range experienced by stream fishes (Bain et al. 1988). Peaking flow regimes associated with hydroelectric facilities pose unnatural conditions through the frequency of high magnitude flows and the rate of change in flow. Few aquatic organisms are adapted to thrive in this type of environment though some species are more resistant to habitat variability than others such as macrohabitat generalists (Bain et al. 1988). A peaking flow environment is associated with changes in critical habitat variables during water release including changes in depth, width, velocity, water temperature, and water quality (Cushman 1985). Thus, the range in physical habitat parameters is much greater in a regulated river than an unregulated river over a shorter time interval than what might occur naturally.

Fish Response to Flow Variability

Indeed, the quantity and timing of flow are crucial components of ecological function in river systems. It is increasingly recognized that the distribution and abundance of riverine species are limited by the effects of flow regulation (Bain et al. 1988, Marchetti and Moyle 2001). A strong correlation exists between stream flow and a river's physicochemical characteristics such as water temperature and habitat diversity (Poff et al. 1997). Research in the distributional ecology of fishes suggests that fish assemblages form in response to the physicochemical factors of the environment (Matthews 1987). Change in the assemblage structure of stream fishes or species composition is imposed by temporal variation in stream flow (Schlosser 1985). Studies show that flow variability affects use of spatial resources or patterns of microhabitat use (Grossman et al. 1998). Therefore, evidence indicates a strong fish response to flow variability.

Fish Community Structure

The effects of flow regulation operate as a main structuring agent for fish abundance, diversity, and distribution. Thus, understanding how a fish community is structured in flow-regulated rivers has management implications for conservation of biodiversity. It is often impractical to reveal the underlying mechanisms behind community structure, because it requires experimental study of multiple cause-and-effect relationships. However, community patterns can be discerned along longitudinal and environmental gradients providing information about what factors most influence the fish community.

As natural resource management agencies face continued pressure to develop and defend recommendations to preserve aquatic resources, information on fish community structure is necessary, especially in relation to prominent issues such as stream flow regulation. Knowledge about the factors that most affect fish community characteristics can be incorporated into decisions to protect and enhance fisheries below dams. If analyses show adverse affects of varying flows on aquatic biota below a dam, then several mitigation strategies exist to minimize the impact of the dam including operational changes, structural changes, or habitat modification (Cushman 1985).

In this study, I evaluate the patterns of community structure among different populations of non-salmonid fish in a tailwater (Smith River, Henry Co., VA). The Smith River has a hydropeaking flow regime with a hypolimnetic release that dramatically depresses the temperature of the river, providing an excellent template for research on patterns in abundance, diversity, and distribution in relation to longitudinal and environmental gradients in a flow-regulated river. I specifically addressed the following questions:

1) What are the longitudinal patterns in fish abundance, diversity, and distribution?
2) How does fish composition change spatially or temporally?
3) How do environmental variables relate to relative abundance?

STUDY AREA, SPECIES PRESENT, AND CURRENT MANAGEMENT

The Smith River is a sixth order, regulated tributary of the Dan River, located in Virginia's Patrick and Henry Counties. The river is in the Roanoke drainage where forestry and agriculture are the dominant land uses in the upper sections, and urban and industrial development in the Bassett-Martinsville sections. Philpott Dam, constructed in 1952 by the U.S. Army Corps of Engineers and used for hydroelectric generation, flood control, and recreation, created the tailwater of the Smith River. The dam is a peaking hydropower production facility with an operational mode of power generation dictated by energy demands and water availability. The hydrology of the Smith River is dominated by regulated flows 12 months a year.

Discharge from the dam creates a flow regime that can fluctuate from 25 to 1400 cfs. The dam operated under different flow regimes during the sample years of 2000, 2001, and 2002 (Figure 1.1). From January 2000 to May 2001, peaking flows were 1400 cfs, 7 days/wk, for 1-hour duration. From June 2001 to October 2001, peaking generation was at 700 cfs, 5 days/wk, for a duration of 2-10 hours. The peaking flows returned to 1400 cfs from November 2001 to February 2002, 5 days/wk, for a duration of

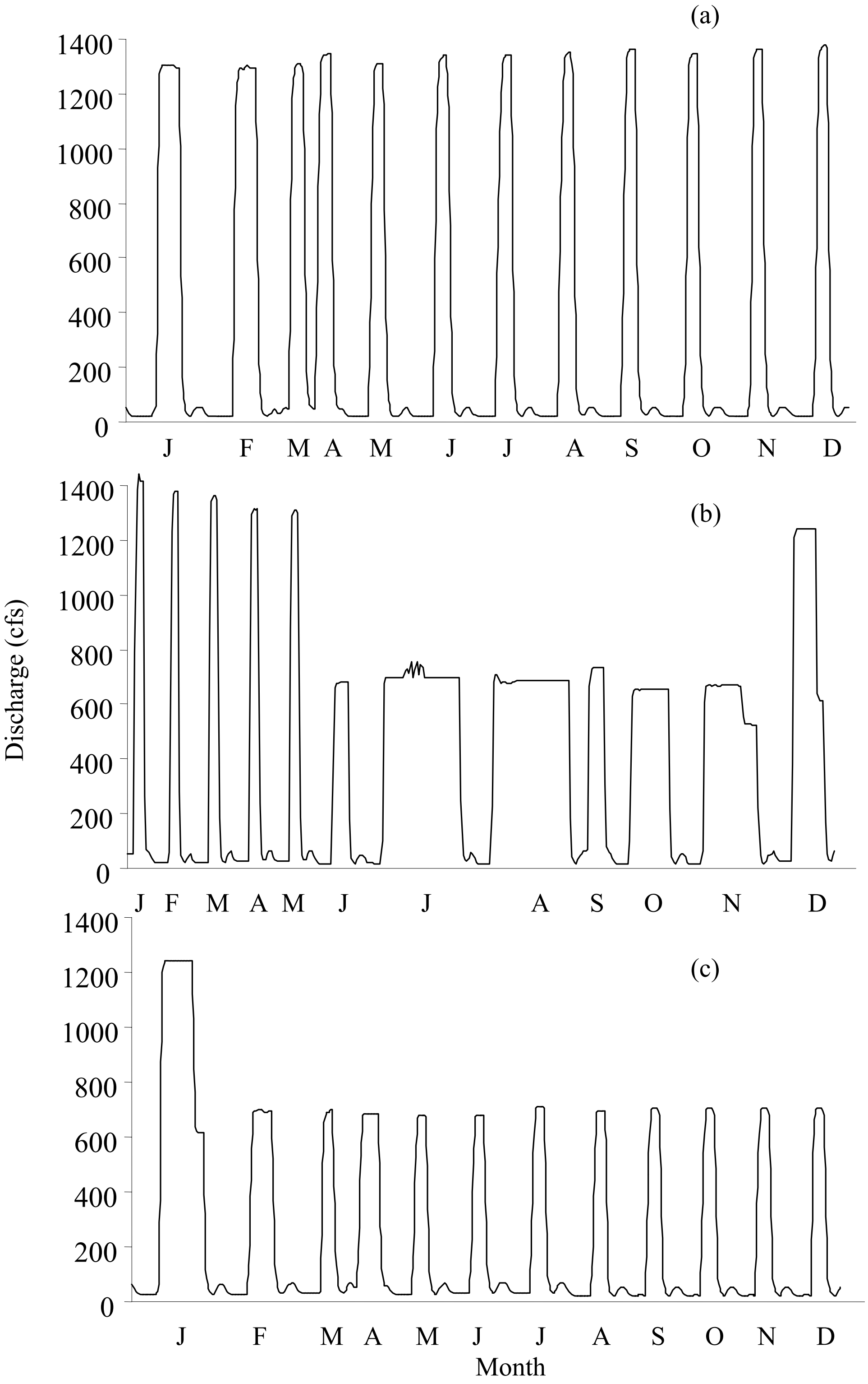


Figure 1.1. Peaking discharge during one 24-hour time period by month for 2000 (a), 2001 (b), and 2002 (c) (USGS Philpott gage # 02072000). Wider peaks represent longer duration releases (i.e. 8 hour release in August, 2001).

3-4 hours. Finally, generation releases were 700 cfs for a 1-hour duration from March 2002 to the end of the study in October 2002.

Because the discharge is hypolimnetic, the water releases influence the thermal regime of the Smith River on a regular basis. Temperatures directly below the dam are less than 10^{o}C and increase with distance from the dam (Figure 1.2). Maximum temperatures occur at the furthest downstream site in the study area. Daily maximum hourly temperature depression ranges from 0.16-8.39^{o}C over all seasons with the greatest flux occurring in the summer. Variation in temperature depends on how greatly the ambient air temperature has warmed the water before the coldwater release.

A substrate profile below Philpott Dam indicates changes in types of substrate from upstream to downstream (Figure 1.3). A high percent composition of bedrock, boulder, and cobble is present in the first 4 km below the dam (sites 1-3). Pebble and gravel, which are used by many fishes for spawning in the Smith River, are found in varying peaks throughout the longitudinal gradient. Sand and silt predominate in the lower reaches between 20 and 24 km below the dam (sites 11-12). Aquatic macrophytes increase as distance increases from the dam. Armoring of the channel exists near the dam due to loss of gravel recruitment, while further downstream the channel receives additions of fines from down-cutting of the channel and sediment input from tributaries. The average width of the channel is 30 m.

Despite such high environmental variability, five families of non-salmonid fish are present in the Smith River with fish boasting a wide range of life history strategies (Table 1.1). The known non-salmonid fish fauna of the SR includes 34 species and is dominated by *Etheostoma flabellare, Nocomis leptocephalus, Notropis hudsonius,* and *Catostomus commersoni*. The most dominant fish group is the cyprinids, followed by the catostomids and centrarchids. Fish that are fairly common are *Clinostomus funduloides* and *Luxilus cerasinus*, while fish that are rare are *Exoglossum maxillingua* and *Percina rex*. *Percina rex* has an extant population in the Smith River and are a federally endangered species (Appendix B). A self-sustaining population of brown trout and stocked rainbow trout are present, creating a highly regarded trout fishery.

The Virginia Department of Game and Inland Fisheries designates a three-mile section of the Smith River to be a special trout regulation area with a 16-inch minimum, 2

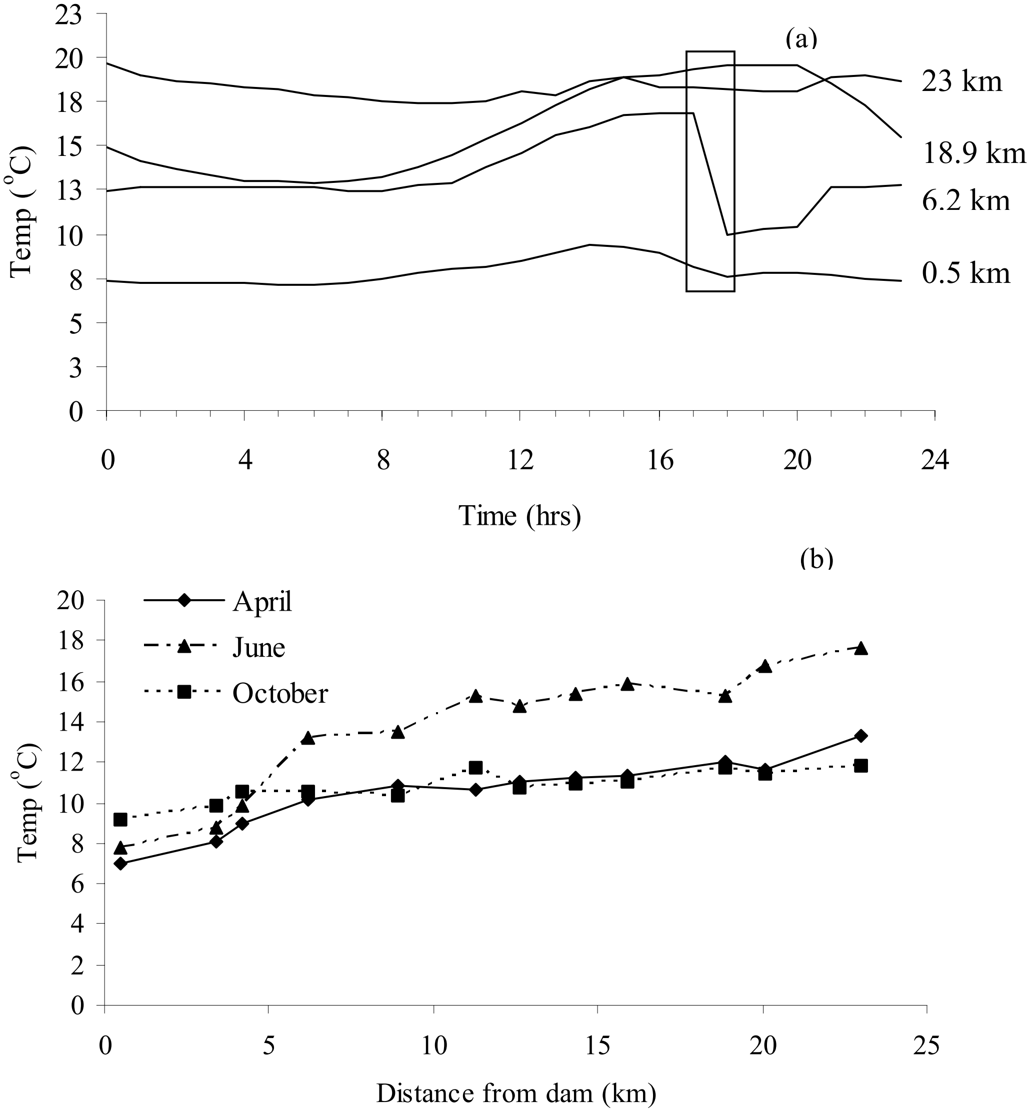


Figure 1.2. Hourly temperature profile (a) for a typical day in June of 2000 at 4 sites, 0.5 km below the dam to 23 km below the dam. The box (a) represents the time of release from 1659 hrs to 1814 hrs (USGS Philpott gage # 02072000). Mean monthly temperature (b) at 12 sites below the dam for the months of April, June, and October in the year 2000 (Krause 2002).

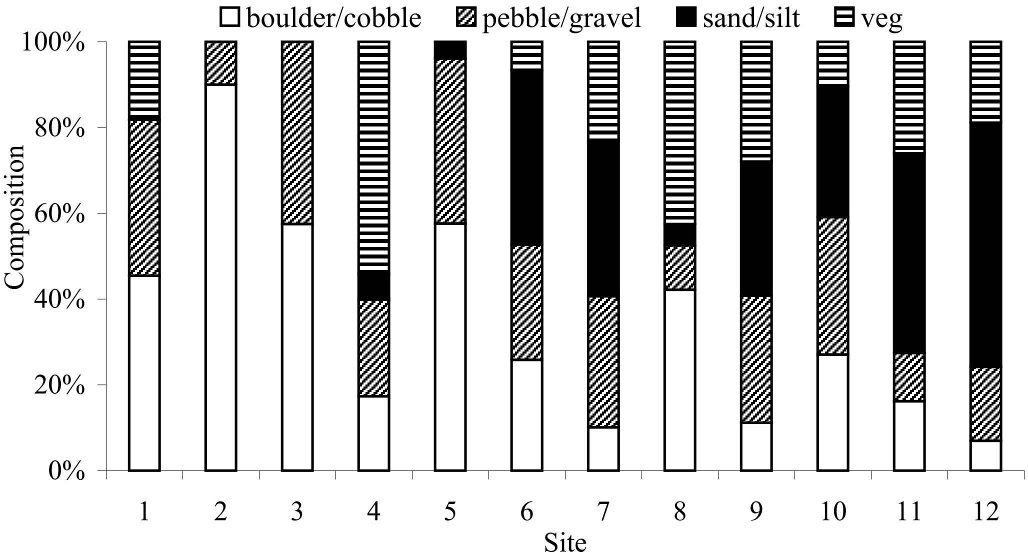


Figure 1.3. Percent composition of substrate categories at each of the 12 sampling sites (Orth 2001).

Table 1.1. Species list and age distribution categories for juvenile and adult fish based on Jenkins and Burkhead (1993). Scientific and common name given with accompanying acronym.

Species		Acronym	Juvenile Length (mm)	Adult Length (mm)
Catostomidae				
Catostomus commersoni	White sucker	WHS	0-179	≥180
Hypentelium nigricans	Northern hogsucker	NHS	0-124	≥125
H. roanokense	Roanoke hogsucker	RHS	0-59	≥60
Moxostoma erythrurum	Golden redhorse	GOR	0-199	≥200
M. pappilosum	V-lip redhorse	VLR	0-229	≥230
Scartomyzon cervinus	Black jumprock	BLJ	0-84	≥85
Cyprinidae				
Nocomis leptocephalus	Bluehead chub	BHC	0-69	≥70
Semotilus atromaculatus	Creek chub	CRC	0-79	≥80
Campostoma anomalum	Central stoneroller	CES	0-59	≥60
Exoglossum maxillingua	Cutlips minnow	CUT	0-64	≥65
Clinostomus funduloides	Rosyside dace	RSD	0-49	≥50
Luxilus cerasinus	Crescent shiner	CRS	0-49	≥50
Phoxinus oreas	Mountain redbelly dace	MRD	0-39	≥40
Notropis hudsonius	Spottail shiner	SPS	0-59	≥60
N. procne	Swallowtail shiner	SWS	0-39	≥40
N. chiliticus	Redlips shiner	RES	0-39	≥40
Notemigonus chrysoleucas	Golden shiner	GOS	0-69	≥70
Luxilus albeolus	White shiner	WS	0-64	≥65
Cyprinella galactura	Whitetail shiner	WTS	0-49	≥50
Lythrurus ardens	Rosefin shiner	ROS	0-44	≥45
Centrarchidae				
Micropterus salmoides	Largemouth bass	LMB	0-229	≥230
M. dolomieu	Smallmouth bass	SMB	0-199	≥200
Lepomis auritus	Redbreast sunfish	RBS	0-89	≥90
L. cyanellus	Green sunfish	GSF	0-69	≥70
L. macrochirus	Bluegill	BLG	0-79	≥80
Ambloplites cavifrons	Roanoke bass	ROB	0-199	≥200
Pomoxis nigromaculatus	Black crappie	BLC	0-99	≥100
Percidae				
Etheostoma flabellare	Fantail darter	FND	0-34	≥35
E. vitreum	Glassy darter	GLD	0-34	≥35
E. podostemone	Riverweed darter	RWD	0-29	≥30
Percina roanoka	Roanoke darter	RND	0-34	≥35
P. rex	Roanoke logperch	ROL	0-79	≥80
Ictaluridae				
Ameirus nebulosus	Brown bullhead	BRB	0-139	≥140
Noturus insignis	Margined madtom	MAM	0-89	≥90

fish-per-day limit. In 1995, anglers spent over 36,000 hours trout fishing in the Smith River, generating a total economic value of $440,000/yr (Hartwig 1998). During the 1970s, the Smith River became well-known for the catch of a state record brown trout. Today, there are smaller trout and fewer trophy catches. Biologists with the Virginia Department of Game and Inland Fisheries and the public would like to see more trophy catches due to the high economic and recreational value of the resource. Biologists also understand the ecological value of the entire fish community and seek to understand assemblage characteristics in relation to distance from the dam.

METHODS

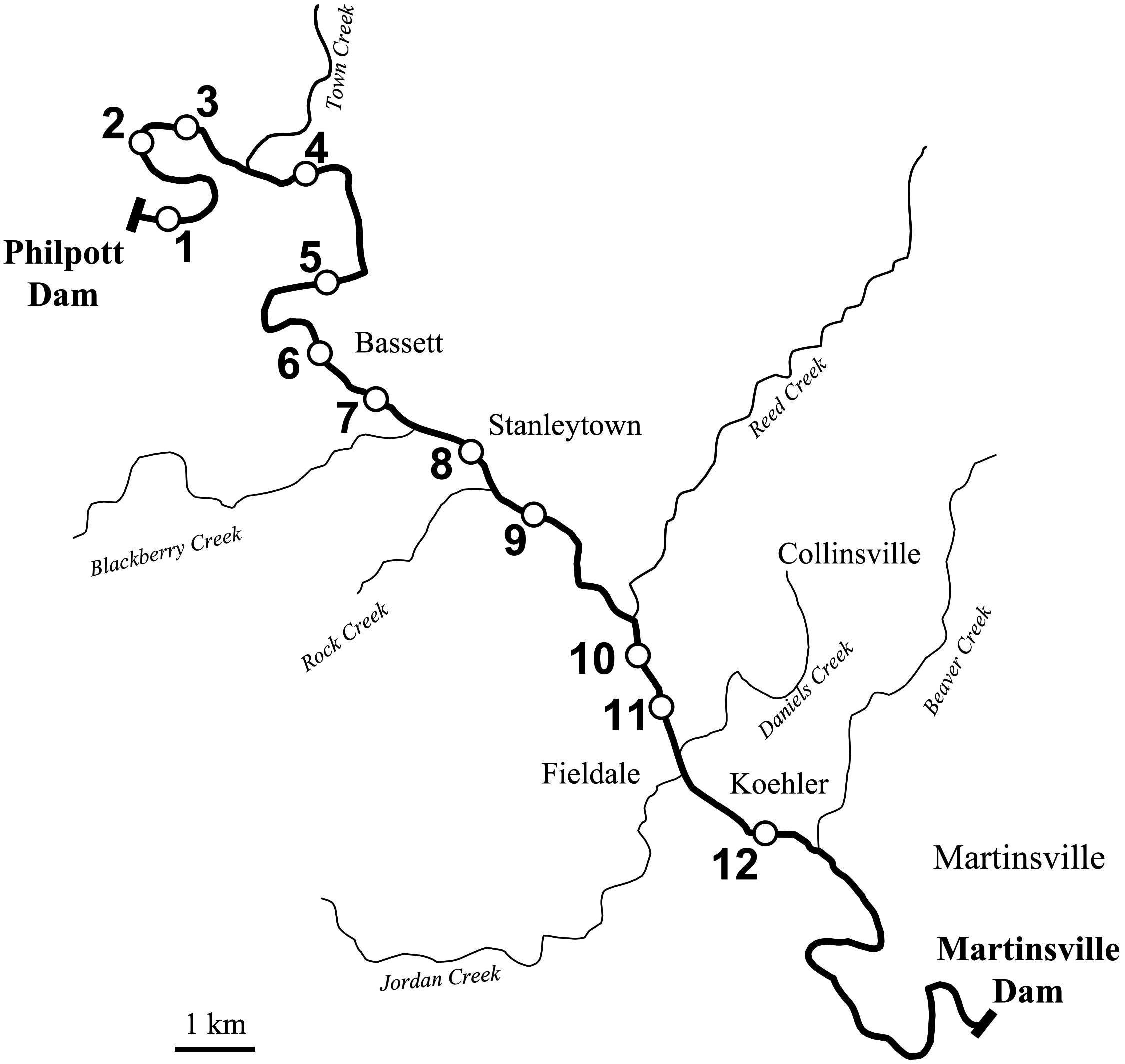
Data Collection

Fish were sampled at 12 locations below Philpott Dam to 24 km downstream during the spring, summer, and fall (Figure 1.4). The 12 locations were established as permanent sampling sites because of suitable access. Paired multiple anode, pulsed DC barge electrofishers were used to capture fish. Within the 12 sites, single-pass depletion sampling was performed in the spring and fall while three-pass depletion sampling was performed during the summer. All non-salmonid fish were identified to species, counted, and a subsample were measured to the nearest mm (total length), weighed to the nearest tenth of a gram, and released. Efforts were made to subsample each species across the range of sizes captured, measuring a representative sample of small to large individuals.

During June of 2000, 2001, and 2002, three-pass depletion electrofishing was conducted in 100-m sections enclosed by block nets at each location. From this data, population estimates and 95% confidence intervals were computed from maximum-likelihood estimates using the Microfish program (Van Deventer and Platts 1989). In April of 2001, 2002 and October of 2000, 2001, 2002 single-pass depletion electrofishing was performed in 100-400 m reaches without block nets.

Data Analysis

I calculated the relative abundance of fish per 100 m for each site in each sampling period. For all June samples, first-pass numbers of fish were used in analyses. To qualitatively discern the longitudinal pattern below the dam to 24 km downstream, relative abundance and species richness were plotted for each site across sampling



Site #	1	2	3	4	5	6	7	8	9	10	11	12
km below Philpott Dam	0.5	3.4	4.2	6.2	8.9	11.3	12.6	14.3	15.9	18.9	20.1	23.0

Figure 1.4. Map of the Smith River tailwater between Philpott Dam and Martinsville Dam with sampling sites numbered upstream to downstream.

periods. A Kruskal-Wallis test was used to determine if a significant difference in relative abundance was present across the eight sampling periods (Zar 1996). Likewise, I tested species richness across sampling periods using Kruskal-Wallis. A coefficient of variation (CV) was calculated for relative abundance and species richness at each site across sampling periods and is reported as a percentage. To classify population variability for relative abundance and species richness, an arbitrary scheme proposed by Freeman et al. (1988) using CV values was followed: (1) CV ≤ 25% = stable; (2) 26% < CV ≤ 50% = moderately stable; (3) 51% < CV ≤ 75% = moderately fluctuating; (4) ≥ 76% = fluctuating.

Length data was used to show the age distribution over time and on a longitudinal basis. All species were classified as adult or juvenile based on information from Jenkins and Burkhead (1993). April, June, and October samples from 2001 and 2002 were used to show age distribution. Length data from 2000 was not used because it was not representative of more recent sampling periods. During June 2000, lengths were measured for only a few species. During October 2000, lengths were taken for fewer individuals of fish, because the distance sampled was set at 100 m, shorter than all other time periods.

I also used information from Jenkins and Burkhead (1993) to classify each of the 13 most common or numerically dominant species as fluvial specialists or habitat generalists. A fluvial specialist (FS) was considered to be the type of fish that is obligate to a river, sensitive to stress, and a microhabitat specialist. A habitat generalist (G) was considered to be a fish with broad habitat requirements or a high stress tolerance. These definitions were derived from Bain and Boltz (1989) and follow the theoretical framework of Kinsolving and Bain (1993) and Travnichek and Maceina (1994). Because fluvial specialists are sensitive to changes in flow, measures of their relative abundance are practical for assessing the effects of flow on community structure (Travnichek et al. 1995). Using relative abundance (fish per 100 m) of each species by site, I averaged the number of fluvial specialists and habitat generalists by site within each season and plotted the longitudinal patterns with distance from the dam.

Similarity of fish assemblages was estimated using Morisita's Index (Morisita 1959). The index was used to compare consistency of fish composition between time periods at a specific site as well as across sites in one time period. An index value was calculated for each site across successive surveys (i.e. June 2000 to October 2000) and like seasons (i.e. June 2000 to June 2001). The original measure of Morisita's index (I_m) was used because it is found to be independent of sample size and diversity (Wolda 1981). Smith and Zaret (1982) measured bias of such indices in terms of sample size, diversity, and evenness and found that the original measure of Morisita's index gives the most accurate results. Values calculated from Morisita's Index range from zero, suggesting no assemblage similarity, to approximately one, suggesting identical assemblages.

The Spearman rank correlation coefficient (r_s) was used to compare significant changes in fish assemblage structure across space and time (Siegel 1956). Relative abundance of all species combined was used to compare assemblages across sites between successive samples (i.e. June 2000 and October 2000) and like seasons (i.e. June 2000 and June 2001). Relative abundance of each of the 13 most common species was used to compare assemblages between successive samples and like seasons. Multiple comparisons were performed using Kendall's coefficient of concordance (W), after correcting for ties (Siegel 1956). Multiple comparisons across all sampling periods for each of nine sites were made using the relative abundance of the 13 most common species. The first three sites were omitted from the concordance analysis because of the number of zeros present in the data. Significance of r_s and W was tested by using the large-sample method and χ^2 values respectively, after Siegel (1956). Because rank correlation is susceptible to Type I error (Grossman et al. 1982), or rejection of a null hypothesis that is true, a conservative critical value was set at $p = 0.01$ following the approach of Schlosser (1987).

Multiple linear regression was used to help discern the relationship between non-salmonid abundance and other biotic and abiotic variables in the Smith River. The primary utility of this analysis was inference about mechanisms that most influence relative abundance. Variables were carefully screened in order to avoid "data dredging" or "overfitting" commonly associated with multiple linear regression (Burnham and Anderson 1998).

Relative abundance of all non-salmonid fish (number of fish per 100 m) for each of the 12 sites was estimated for time periods June 2000, October 2000, and April 2001 and served as the dependent variable for the regression model. Data for brown trout abundance, density of macroinvertebrates (Newcomb et al. 2001), chlorophyll *a* content (Appendix A), temperature (Krause 2002), and substrate composition (Orth 2001) were input into the model as the independent, predictor variables. Each independent variable had site specific values coinciding with the 12 fish sampling sites.

A correlation matrix of all the variables in the full model was used to reveal a collinearity problem if one of the pairwise correlations exceeded 0.9 or several exceeded 0.7. A second test for collinearity was performed by running a variance inflation factor (VIF) test in SAS for the full model (version 8 SAS). If the calculated VIF is 10, then the variable is most likely collinear with another variable. Variables which were determined to be collinear were removed from the model to prevent inclusion of redundant variables, or those sharing too much information. Finally, a stepwise regression procedure was used to obtain the final model (version 8 SAS).

To help summarize the relationship between the environmental variables selected by the regression model and the 13 most common species, canonical correspondence analysis (CCA) was performed using an Excel© macro developed by Lipkovich and Smith (2002). Relative abundance of the 13 species was used along with the corresponding environmental data. CCA has been used to relate species abundances to measured variation in the environment (Taylor 1993 and 2000). The technique selects the linear combination of environmental variables that maximizes the dispersion of the species scores (Jongman et al. 1987). By looking at the perpendicular intersection of a species with each environmental vector, I estimated the center of that species distribution along that environmental gradient (Taylor 1993). Likewise, species and sites that were grouped together were noted.

RESULTS

Overall Patterns

A total of 14,245 non-salmonid fish were caught in the Smith River tailwater, representing 5 families and 34 species. The longitudinal distribution of fishes exhibited much spatial variation and minimal temporal variation in terms of abundance and species richness. Longitudinal patterns of relative abundance (Figure 1.5) and species richness occurred consistently over time across the 12 sites. Neither relative abundance nor number of species changed significantly between time periods (Kruskal Wallis, $p > 0.05$). There was a general trend of increasing abundance and numbers of species as distance increases from the dam. Marked increases or peaks in relative abundance and species richness consistently occurred in areas of the mainstem that are adjacent to four main tributaries, which I refer to as tributary junction sites (Figure 1.6). The four main tributary junctions are found at 5.4, 13.4, 18.6, and 21.1 km from the dam.

June population estimates per 100 m ranged from a total of 2 individuals at site 1, 0.5 km downstream from the dam, to 1809 individuals at site 12, 24 km below the dam (Figure 1.7). Numbers of individuals were generally higher across sites in the June 2002 sample compared to June 2000 and June 2001 with 8 out of 12 sites having more fish, on the order of 828 more individuals per 100 m at one site. The longitudinal pattern of population estimates was similar to relative abundance in that numbers of individuals increased with increasing distance from the dam. A comparison between the years showed that a distinctive peak in population estimates occurred at site 4, 6.2 km below the dam, for June 2002, and a distinctive drop in population estimates occurred at site 12, 24 km below the dam, for June 2000. Site 4 is adjacent to the largest tributary that flows into the mainstem, Town Creek.

The number of species per sampling period ranged from a high of 29 in June 2001 and October 2002 to a low of 24 in October 2000. The highest number of species occurred in site 11 with 26 species during October 2001, and the lowest with 0 species in site 2 during April 2001. The greatest variation in species richness occurred at site 2, followed by sites 1,3, and 5 with CVs at 76, 47, 42, and 43 percent, respectively (Table 1.2).

Variability in species richness was moderately stable to stable for all sites across sampling periods with the exception of site 2, which highly fluctuated. Across sampling periods, I found relative abundance to be moderately fluctuating to fluctuating for sites 1-4 and 8-12 (Table 1.3). However, sites 5,6, and 7 were moderately stable. CV values for relative abundance of the 13 most common species (Table 1.4) were classified as

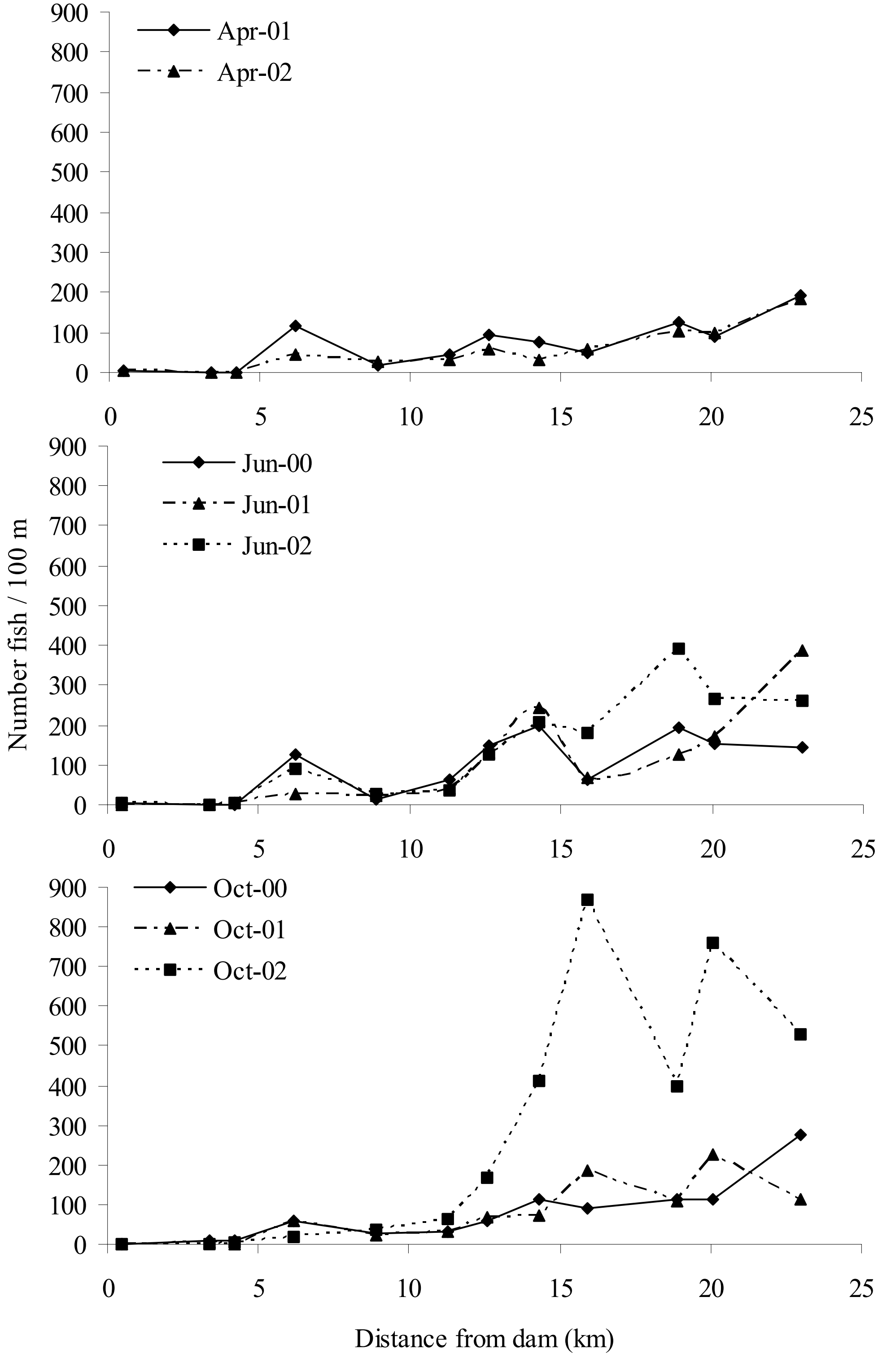


Figure 1.5. Relative abundance for all non-salmonid fish as distance increases from the dam for all sampling periods.

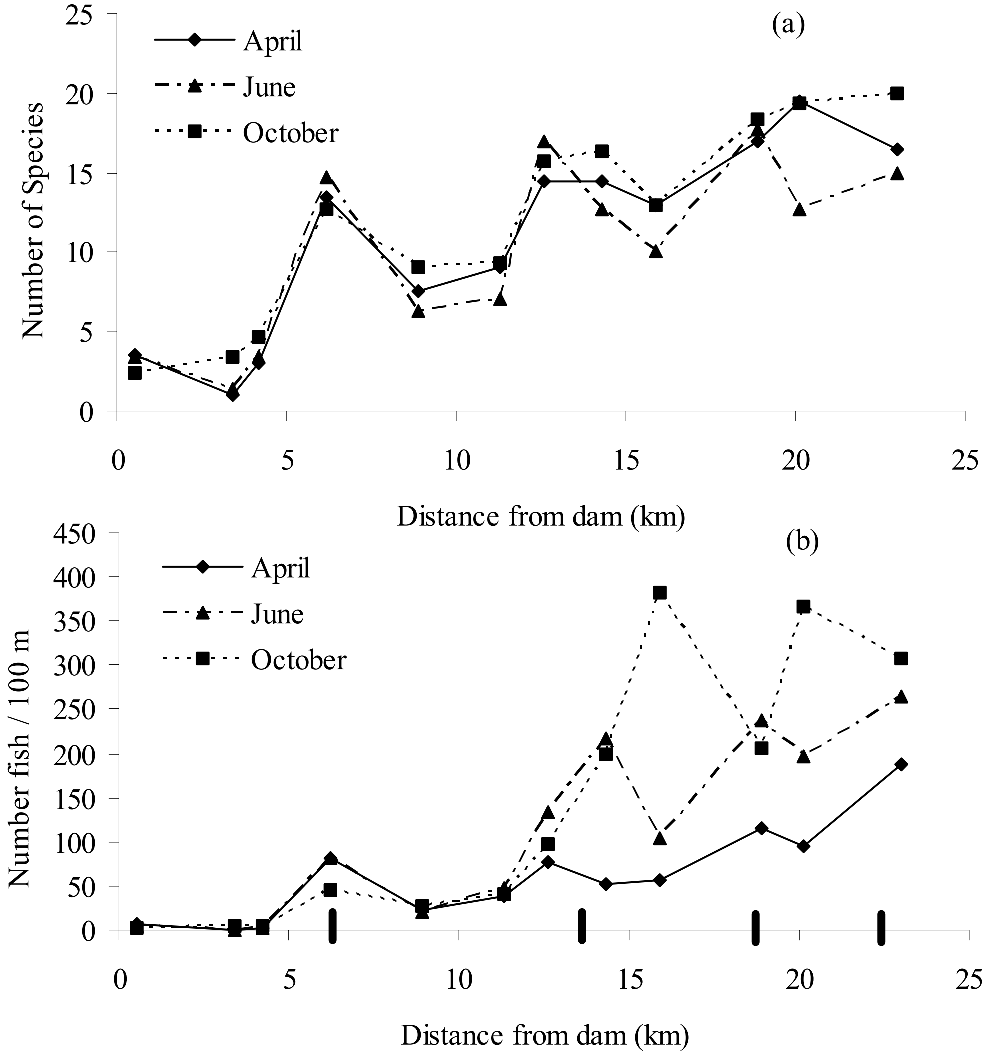


Figure 1.6. Species richness (a) and relative abundance (b) for all species averaged within each season as distance increases from the dam for sampling periods in 2000, 2001, and 2002. Vertical bars (b) represent tributary junctions.

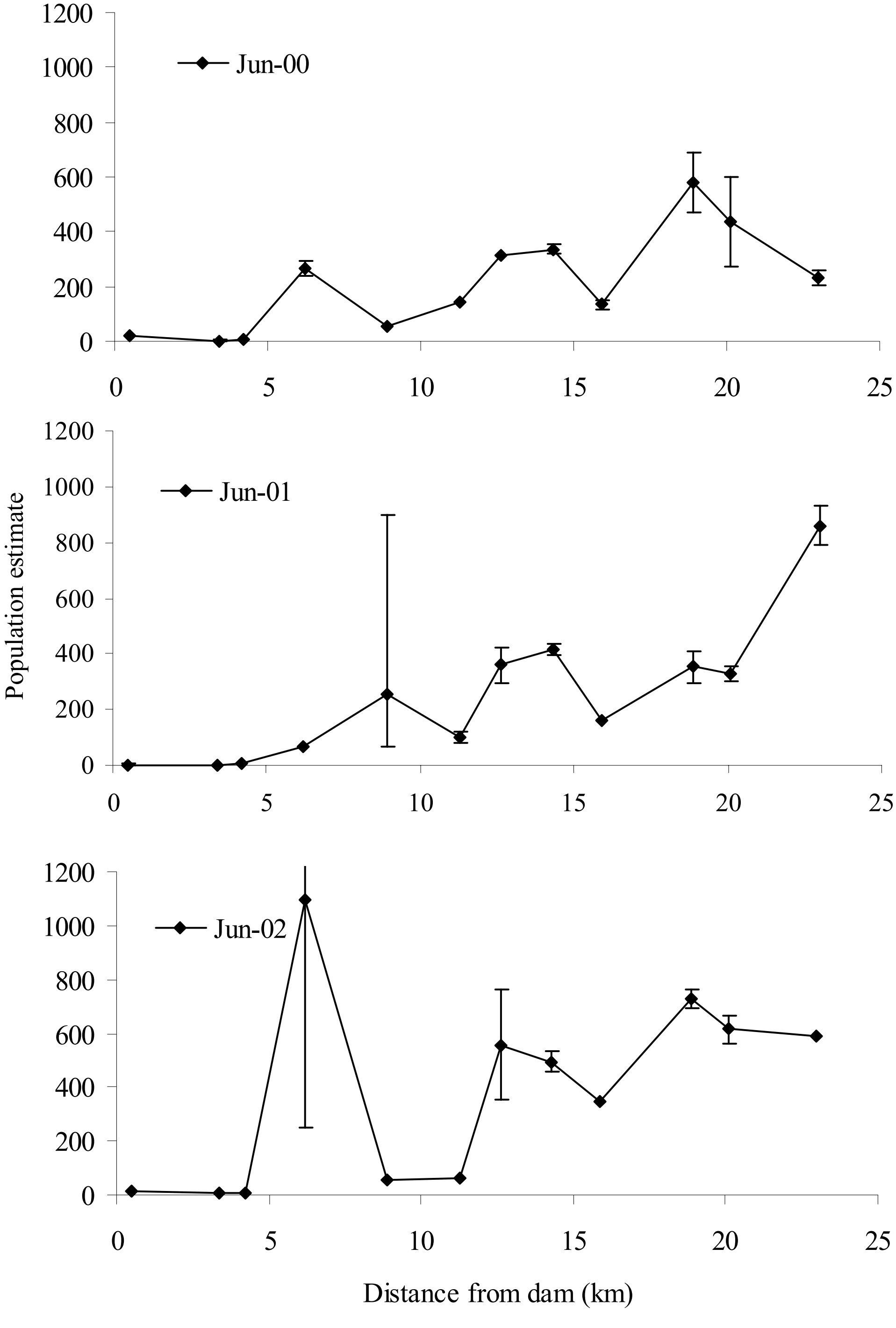


Figure 1.7. Population estimates for fish per 100 m with 95% confidence intervals for June 2000, 2001, and 2002 using 3-pass depletion catch data (Microfish 3.0) for 12 sites. Total catch was used for sites with non-descending catch data (those without error bars).

Table 1.2. Species richness for non-salmonid fish in the Smith River, VA over eight sampling periods, where CV is the coefficient of variation for species richness across time periods within each site. There is no significant difference in species richness among time periods (Kruskal Wallis, $p > 0.05$).

Site	Distance from dam (km)	June 2000	October 2000	April 2001	June 2001	October 2001	April 2002	June 2002	October 2002	CV
1	0.5	5	1	3	1	3	4	4	3	0.47
2	3.4	1	5	0	1	3	2	2	2	0.76
3	4.2	2	5	4	6	4	2	2	5	0.42
4	6.2	17	14	16	12	17	11	15	7	0.25
5	8.9	4	4	7	6	14	8	9	9	0.43
6	11.3	10	6	9	6	11	9	5	11	0.28
7	12.6	15	11	16	22	18	13	14	18	0.22
8	14.3	11	13	16	14	17	13	13	19	0.18
9	15.9	8	11	14	12	13	12	10	15	0.19
10	18.9	17	14	16	17	21	18	19	20	0.13
11	20.1	12	11	20	15	26	19	11	21	0.33
12	23	8	20	19	20	17	14	17	23	0.27
	Total species	26	24	26	29	28	26	27	29	

Table 1.3. Abundance data for non-salmonid fish in the Smith River, VA over eight sampling periods where N = total abundance, RA = relative abundance or number of individuals per 100 m, and CV is the coefficient of variation for relative abundance across time periods within each site. There is no significant difference in abundance among time periods (Kruskal Wallis, $p > 0.05$).

Site	Distance from dam (km)	June 2000		October 2000		April 2001		June 2001		October 2001		April 2002		June 2002		October 2002		CV
		N	RA	N	RA	N	RA	N	RA	N	RA	N	RA	N	RA	N	RA	
1	0.5	18	6	2	2	24	6	1	1	3	1	17	6	11	4	6	2	0.66
2	3.4	1	0	7	7	0	0	2	1	7	3	3	1	8	1	3	1	1.32
3	4.2	5	0	9	9	5	2	9	3	4	1	2	1	10	4	9	3	0.99
4	6.2	228	128	57	57	351	117	68	28	141	58	122	46	250	90	47	18	0.59
5	8.9	53	12	25	25	56	17	67	24	68	21	73	27	58	25	100	37	0.31
6	11.3	145	64	32	32	151	44	82	40	128	31	115	33	62	36	222	62	0.31
7	12.6	311	149	59	59	333	95	267	129	260	67	256	60	301	124	552	168	0.40
8	14.3	311	197	113	113	265	74	383	243	263	72	138	32	425	209	1777	413	0.73
9	15.9	119	64	92	92	153	51	162	69	382	187	216	60	350	181	1779	868	1.41
10	18.9	393	193	111	111	468	126	267	126	372	107	359	103	650	391	949	399	0.65
11	20.1	254	153	113	113	336	91	391	171	800	227	371	99	504	264	1902	761	0.94
12	23	202	146	277	277	709	192	684	387	418	115	583	182	591	263	1716	528	0.53

Table 1.4. Abundance data summed over all sites for the 13 most common or numerically dominant non-salmonid fish in the Smith River, VA over eight sampling periods where FS = fluvial specialist, G = habitat generalist, N = total abundance, RA = relative abundance or number of individuals per 100 m, and CV is the coefficient of variation for relative abundance across time periods within each site.

13 Most Common Species	FS or G	June 2000		October 2000		April 2001		June 2001		October 2001		April 2002		June 2002		October 2002		CV
		N	RA	N	RA	N	RA	N	RA	N	RA	N	RA	N	RA	N	RA	
N.leptocephalus	FS	406	266	224	224	534	151	468	230	536	155	219	63	278	278	923	309	0.39
C. anomalum	FS	34	32	21	21	7	2	5	2	42	12	12	3	2	2	108	35	1.02
L. cerasinus	G	38	21	17	17	17	5	22	12	103	32	9	3	5	5	78	25	0.71
P. oreas	G	12	9	9	9	18	5	42	9	43	13	9	2	13	13	151	40	0.91
C. funduloides	G	48	37	49	49	84	24	123	85	173	51	71	18	62	62	398	92	0.51
N. hudsonius	G	1	0	48	48	641	174	431	296	525	167	733	216	114	108	2787	1132	1.35
C. commersoni	G	48	209	228	228	533	150	176	76	162	49	452	131	228	234	1226	386	0.58
H. nigricans	G	9	8	27	27	35	10	11	7	40	11	29	7	18	18	88	33	0.65
M. erythrurum	G	20	19	21	21	74	22	3	0	19	9	52	14	27	27	4	1	0.70
E.flabellare	FS	818	332	117	117	753	228	651	347	646	268	463	132	649	649	1576	466	0.56
E. podostemone	FS	26	17	59	59	29	9	136	48	236	81	77	21	47	47	805	280	1.25
P. roanoka	FS	118	53	19	19	32	9	58	42	50	15	22	6	43	74	350	121	0.94
N. insignis	FS	52	21	1	1	18	5	25	7	9	3	7	2	8	8	16	6	0.95

moderately fluctuating to fluctuating for all species except *N. leptocephalus*, a moderately stable species.

Patterns in fish abundance were not driven by juvenile numbers during any sampling period. Variability in fish abundance was most attributable to spatial and temporal variation in adult fish numbers, primarily because adults comprised 75% to 90% of the population (Figure 1.8). However, both age groups followed the same trend within each season such that no contrasting peaks occurred between adults and juveniles (Figure 1.9). The lack of accentuated change in either age group indicates stability of age structure within each season. Though change in juvenile abundance in response to high flow regimes is a potential source of variation in community structure (Schlosser 1985), this appears not to be a mechanism in the Smith River.

Of the 13 most common species, 6 were classified as fluvial specialists and 7 as habitat generalists (Table 1.4). Averaging relative abundance (number of fish per 100 m) of each macrohabitat class within each season, April and October were evenly split with 50% of the fish being fluvial specialists and 50% being habitat generalists. The June sample mean indicated 69% of the fish were fluvial specialists while 31% were habitat generalists. The raw data indicates that June 2002 had the highest number of fluvial specialists at 3 sites compared to all other time periods. October 2002 had the highest number of habitat generalists at 5 sites compared to all other time periods. By plotting fluvial specialists and habitat generalists separately, higher numbers of fluvial specialists were seen at tributary junctions, and increasing numbers of both classes with increasing distance from the dam (Figure 1.10).

Morisita's Index of Similarity

The fish assemblages of successive surveys at each site ranged from no similarity to almost identical assemblages with I_m values ranging from 0 to >1.00 (Table 1.5). However, similarities between sampling periods did not differ significantly across time (single-factor ANOVA, $p = 0.99$). Thus, no interval occurred between sampling periods where there was a complete change of the fish assemblage, despite the change in magnitude and duration of water releases during the interim of the study. The most variable fish assemblages were found within sites 1, 2, and 3, nearest the dam. These sites had I_m values ranging from 0 to >1.00 across sampling periods. I_m values of 0

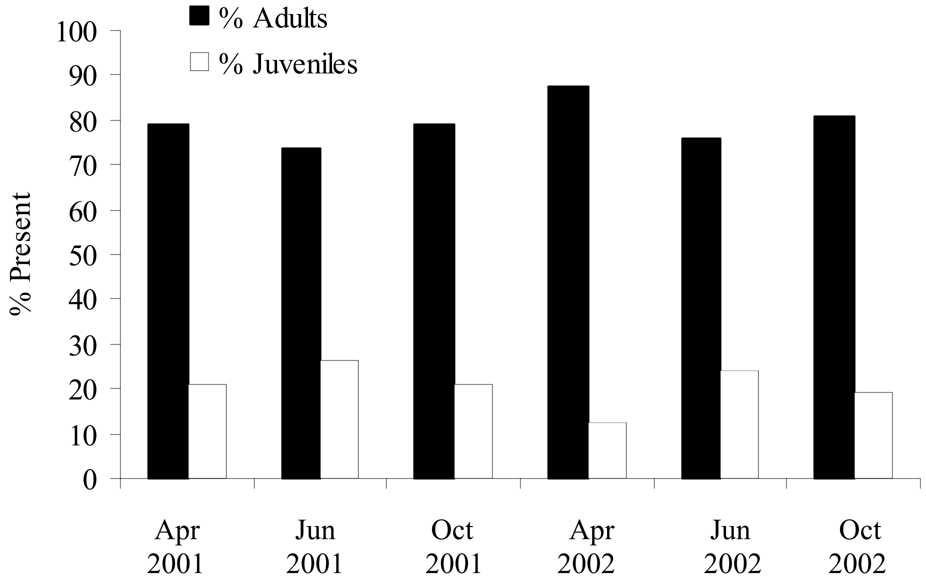


Figure 1.8. Age distribution of the percentage of adults and juveniles in each sampling period for which length measurements were taken.

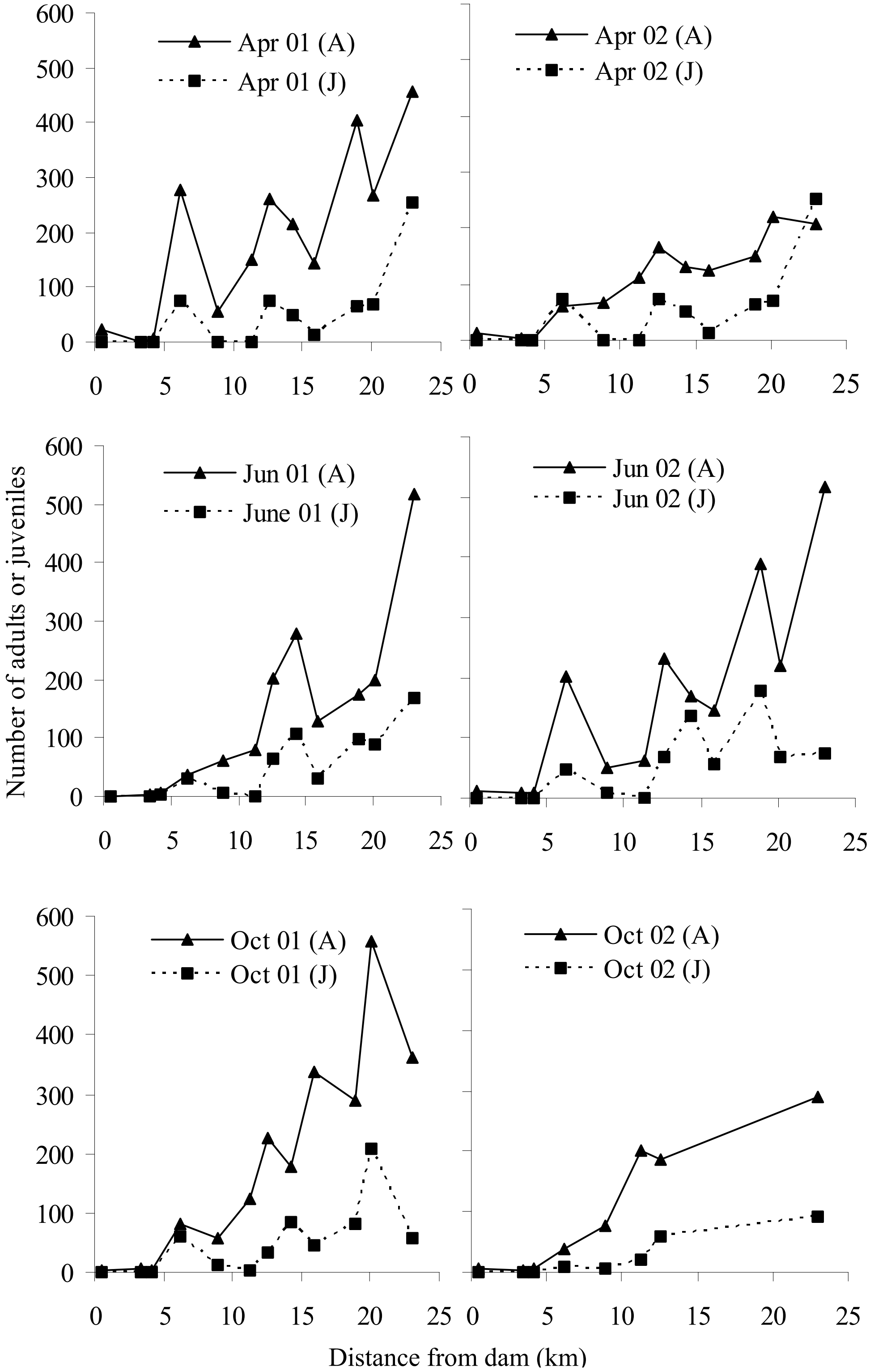


Figure 1.9. Age distribution per 100 m as distance increases from the dam using a subsample of fish for which length measurements were taken in 3 seasons of 2001 and 2002. Length was not taken for sites 8-11 for October 2002. Note — A = Adults, J = Juveniles.

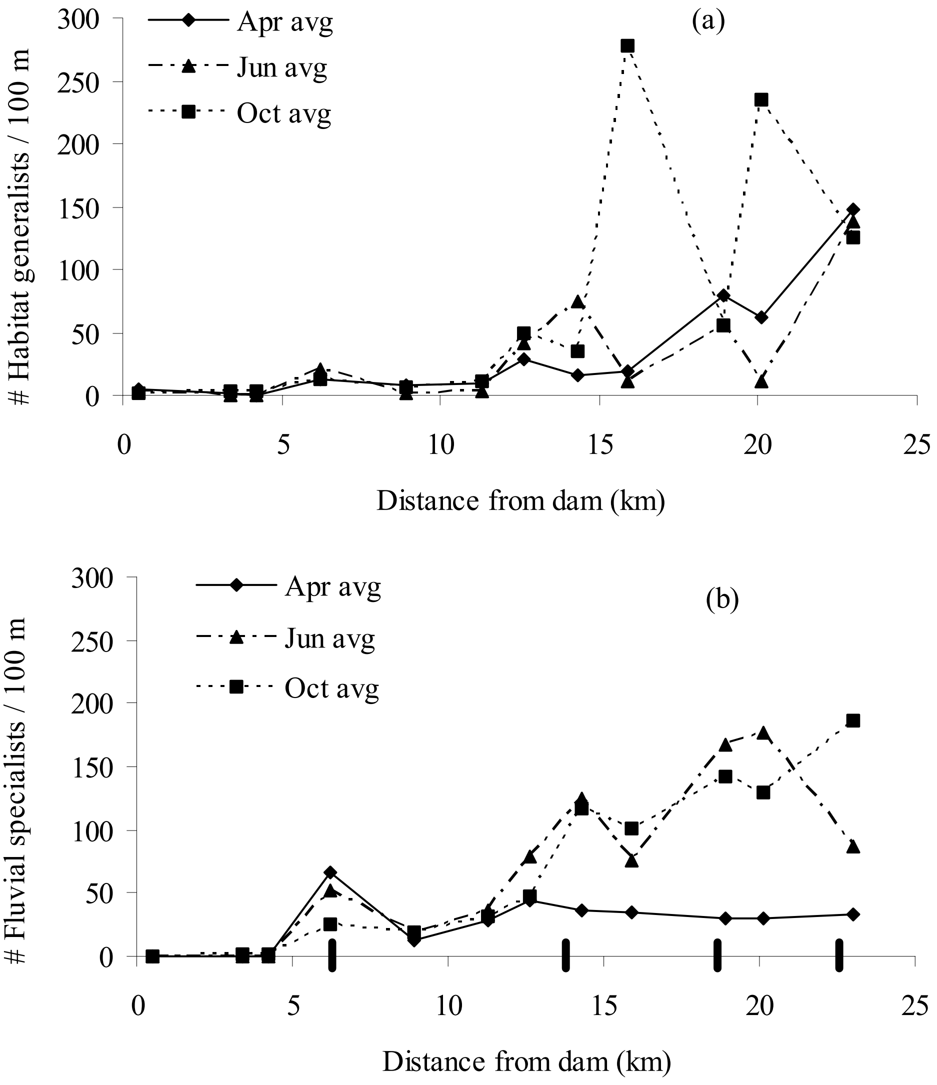


Figure 1.10. Number of habitat generalists (a) and fluvial specialists (b) per 100 m for the 13 most common species averaged within each season across all sampling periods. Vertical bars (b) represent tributary junctions.

Table 1.5. Morisita's index of similarity (I_m) of fish assemblages within 12 sites of the Smith River, VA across successive sampling periods where SD = standard deviation.

Site	Jun 2000 vs. Oct 2000	Oct 2000 vs. Apr 2001	Apr 2001 vs. Jun 2001	Jun 2001 vs. Oct 2001	Oct 2001 vs. Apr 2002	Apr2002 vs. Jun 2002	Jun 2002 vs. Oct 2002	Mean across time	SD
1	0.53	0.96	0.00	0.00	0.14	0.71	>1.00	0.39	0.40
2	>1.00	0.00	0.00	0.26	>1.00	0.54	0.54	0.27	0.27
3	0.20	0.19	0.21	>1.00	0.00	0.00	>1.00	0.12	0.11
4	0.68	0.59	0.85	0.91	0.90	0.93	0.91	0.83	0.13
5	0.88	0.84	0.66	0.76	0.77	0.68	0.75	0.76	0.08
6	0.82	0.71	0.97	1.00	0.86	0.87	0.78	0.86	0.10
7	0.59	0.81	0.98	0.78	0.74	0.91	0.78	0.80	0.12
8	0.72	0.66	0.72	0.87	0.75	0.88	0.95	0.79	0.11
9	0.93	0.89	0.82	0.91	0.84	0.94	0.17	0.79	0.27
10	0.74	0.37	0.68	0.73	0.62	0.61	0.56	0.62	0.13
11	0.12	0.58	0.51	0.60	0.86	0.24	0.14	0.43	0.28
12	0.64	0.71	0.96	0.86	0.58	0.83	0.76	0.76	0.13
Mean across sites	0.62	0.61	0.61	0.70	0.64	0.68	0.64		
SD	0.26	0.29	0.36	0.31	0.30	0.30	0.28		

Single-Factor ANOVA across sampling periods:
F = 0.17 p = 0.99

Single-Factor ANOVA across sites:
F = 9.23 *p < 0.001

resulted when no fish were caught, none of the same species were caught, or no more than 1 individual of each species was caught. I_m values > 1 resulted from very low sample sizes in each sample. I omitted comparisons from analyses with I_m values over one.

Fish assemblages were significantly different among sites (single-factor ANOVA, $p < 0.001$). The lowest similarity in ichthyofauna was found near the dam in contrast to high similarity farther away from the dam, producing a longitudinal gradient of increased consistency in composition downstream of the dam with the exception of site 11. Though site 11 is one of the furthest downstream, it has a low mean I_m value of 0.43. In the June 2002 to October 2002 comparison, site 9 is unique in that it has an I_m value of 0.17 or very low similarity between sampling periods. The most obvious difference between the two sampling periods was that October had 9 more species present and had far greater numbers of individuals present than the June sample (i.e. 1347 *N. hudsonius* in October 2002 vs. 5 *N. hudsonius* in June 2002).

The comparison of fish assemblages at each site across like seasons provided information on the annual variability of fish assemblages (Table 1.6). Across all comparisons, 61% of the I_m values were > 0.70 such that more than half of all sites had high annual similarity. Those comparisons that had low I_m values were within the first 3 sites below the dam. Additionally, several comparisons across site 9 and 11 had low I_m values including the lowest at 0.14 for the site 9 comparison of October 2000 and October 2002. Annual variation was highest between the June 2000 and June 2001 sampling periods with low I_m values for 5 sites.

Spearman Rank Correlation Coefficient

The relative abundance of all non-salmonid fish during successive surveys and like seasons showed significant associations in site ranks across time (Table 1.7). The significance of site ranks over all comparisons illustrates consistency of a longitudinal pattern in fish abundance. Using relative abundance to compare species ranks for the 13 most common species demonstrated significant correlations between successive samples and like seasons except for the comparisons between June 2000 and October 2000, June 2000 and June 2001, and June 2000 and June 2002 (Table 1.7). It is likely that this analyses was driven by the fact that 0 *N. hudsonius* were caught in June 2000, which differs greatly from all other sampling periods. The comparison between relative

Table 1.6. Morisita's index of similarity (I_m) of fish assemblages within 12 sites of the Smith River, VA across like seasons where SD = standard deviation.

Site	Jun 2000 vs. Jun 2001	Jun 2001 vs. Jun 2002	Jun 2000 vs. Jun 2002	Apr 2001 vs. Apr 2002	Oct 2000 vs. Oct 2001	Oct 2001 vs. Oct 2002	Oct 2000 vs. Oct 2002	Mean across time	SD
1	>1.00	>1.00	>1.00	1.01	0.00	0.42	0.53	0.49	0.41
2	0.00	0.57	0.00	0.00	0.11	0.33	0.00	0.14	0.22
3	0.00	>1.00	>1.00	0.00	0.80	>1.00	0.53	0.33	0.40
4	0.93	0.79	0.91	0.97	0.71	0.85	0.63	0.83	0.12
5	1.00	0.99	0.98	1.03	0.94	0.93	0.83	0.96	0.07
6	0.96	0.99	1.00	0.88	0.71	0.76	0.89	0.88	0.11
7	0.90	0.84	0.68	0.93	0.77	0.82	0.59	0.79	0.12
8	0.78	0.72	0.95	0.88	0.74	0.86	0.54	0.78	0.13
9	0.44	0.95	0.43	0.95	0.63	0.55	0.14	0.58	0.29
10	>1.00	0.98	0.91	0.99	0.58	0.75	0.67	0.81	0.17
11	0.32	0.85	0.96	0.67	0.32	0.83	0.55	0.64	0.26
12	0.35	0.90	0.65	0.94	0.79	0.83	0.91	0.77	0.21
Mean across sites	0.57	0.86	0.75	0.77	0.59	0.72	0.57		
SD	0.39	0.14	0.32	0.37	0.29	0.20	0.27		

Table 1.7. Spearman rank correlation tests between species ranks for the 13 most common species and site ranks (* $p < 0.05$).

Successive samples compared	Site Ranks (r_s)	t	Successive samples compared	Species Ranks (r_s)	t
Jun 2000 & Oct 2000	0.86*	5.29	Jun 2000 & Oct 2000	0.44	1.62
Oct 2000 & Apr 2001	0.82*	4.47	Oct 2000 & Apr 2001	0.77*	3.99
Apr 2001 & Jun 2001	0.79*	4.10	Apr 2001 & Jun 2001	0.76*	3.92
Jun 2001 & Oct 2001	0.84*	4.98	Jun 2001 & Oct 2001	0.95*	9.72
Oct 2001 & Apr 2002	0.87*	5.69	Oct 2001 & Apr 2002	0.80*	4.37
Apr 2002 & Jun 2002	0.90*	6.61	Apr 2002 & Jun 2002	0.85*	5.45
Jun 2002 & Oct 2002	0.87*	5.46	Jun 2002 & Oct 2002	0.81*	4.63
Like seasons compared			**Like seasons compared**		
Jun 2000 & Jun 2001	0.87*	5.60	Jun 2000 & Jun 2001	0.40	1.43
Jun 2000 & Jun 2002	0.90*	6.61	Jun 2000 & Jun 2002	0.49	1.88
Jun 2001 & Jun 2002	0.88*	5.94	Jun 2001 & Jun 2002	0.83*	4.85
Apr 2001 & Apr 2002	0.91*	7.06	Apr 2001 & Apr 2002	0.92*	7.58
Oct 2000 & Oct 2001	0.91*	7.06	Oct 2000 & Oct 2001	0.71*	3.35
Oct 2000 & Oct 2002	0.89*	6.17	Oct 2000 & Oct 2002	0.72*	3.45

abundance of specific species demonstrates that the fish assemblages were not always consistent across time on a scale of all sites combined.

Kendall's Coefficient of Concordance

Within individual sites, multiple comparisons of species ranks were made across all sampling periods, showing significant correlations of fish assemblage over time (Table 1.8). The previous species rank tests between successive surveys and like seasons masked the spatial structure of the data, yielding inconsistency of fish assemblages among three comparisons. The multiple comparisons analysis includes both the spatial and temporal elements of the data and indicates no overall change in fish composition within a site over all sampling periods. Though the concordance analysis was not performed for the first three sites, the same species were often present in low numbers. Thus, there exists a consistent grouping of assemblages on a longitudinal basis over time.

Regression Analyses

Specific variables used in the full model included: brown trout relative abundance (number of fish per 100 m), mean monthly temperature, maximum hourly temperature flux, tributary junction (binary), chlorophyll *a* in mg/m^2, mean number of macroinvertebrates per 0.1 m, percent composition of sand and silt (< 2 mm), percent composition of aquatic macrophytes. I ran a correlation matrix for the eight variables (Table 1.9) in the regression model to reveal the linear relationship between the variables as well as the strength of that relationship (Table 1.10). The relationships marked by the strongest correlations with non-salmonid fish abundance were those of tributary junction, mean monthly temperature, macroinvertebrate density, and percent composition of sand and silt. Strong correlations also existed between the following variables: percent composition of sand and silt and mean monthly temperature, percent composition of sand and silt and brown trout abundance, percent composition of aquatic macrophytes and macroinvertebrate density. The correlation coefficients were as high as 0.7 for aquatic macrophytes and macroinvertebrate density as well as macroinvertebrate density and tributary junction. These variables were not dropped from the full regression model based on the correlation matrix, but were further tested. The VIF test in SAS gave values for each variable in the full model much lower than 10, so no variables were dropped from the original full regression model.

Table 1.8. Kendall's coefficient of concordance (W) for the 13 most common species across all sampling periods (** $p < 0.001$).

Site	Distance from dam (km)	W	χ^2
4	6.2	0.49**	47.46
5	8.9	0.44**	42.11
6	11.3	0.69**	66.43
7	12.6	0.44**	42.00
8	14.3	0.71**	67.85
9	15.9	0.62**	59.61
10	18.9	0.61**	58.51
11	20.1	0.70**	66.87
12	23	0.67**	64.53

Table 1.9. Environmental input variables for multiple linear regression models.

Sites	Trib junction	Temp flux (^{o}C)	Mean temp (^{o}C)	Brown trout per 100 m	Invertebrates per 0.1 m	Chl *a* mg/m^{2}	% Sand / silt	% Aquatic veg
June 2000								
1	0	-1.86	7.81	54	55.67	25.99	0	10
2	0	-2.56	8.80	136	8	4.39	0	0
3	0	-4.65	9.81	152	14.17	7.63	0	0
4	1	-5.31	13.16	126	86.33	3.33	4.25	38.75
5	0	-5.2	13.53	73	42.67	6.04	2	0
6	0	-6.51	15.31	73	33.17	3.81	26	4.25
7	0	-8.39	14.76	11	34.33	1.77	25.5	16.5
8	1	-5.52	15.35	42	84.33	5.18	3.25	27.75
9	0	-7.17	15.89	14	23.17	8.93	21	18.75
10	1	-1.75	15.31	35	39.5	15.7	19	6.5
11	0	-4.27	16.74	5	52.83	5.88	33.5	18.75
12	1	-0.8	17.62	34	89.5	1.07	43	14.25
October 2000								
1	0	-0.29	9.20	34	55.67	25.99	0	10
2	0	-0.93	9.89	125	8	4.39	0	0
3	0	-0.62	10.52	129	14.17	7.63	0	0
4	1	-0.93	10.56	103	86.33	3.33	4.25	38.75
5	0	-1.89	10.37	113	42.67	6.04	2	0
6	0	-1.39	11.69	86	33.17	3.81	26	4.25
7	0	-2.87	10.75	43	34.33	1.77	25.5	16.5
8	1	-2.43	10.93	52	84.33	5.18	3.25	27.75
9	0	-1.62	11.06	67	23.17	8.93	21	18.75
10	1	-0.31	11.69	27	39.5	15.7	19	6.5
11	0	-0.85	11.40	15	52.83	5.88	33.5	18.75
12	1	-0.31	11.85	63	89.5	1.07	43	14.25
April 2001								
1	0	-0.16	6.98	58	55.67	25.99	0	10
2	0	-3.24	8.03	54	8	4.39	0	0
3	0	-4.31	8.94	100	14.17	7.63	0	0
4	1	-2.18	10.11	75	86.33	3.33	4.25	38.75
5	0	-2.00	10.84	85	42.67	6.04	2	0
6	0	-4.50	10.68	86	33.17	3.81	26	4.25
7	0	-1.16	11.05	43	34.33	1.77	25.5	16.5
8	1	-0.79	11.19	31	84.33	5.18	3.25	27.75
9	0	-1.59	11.32	19	23.17	8.93	21	18.75
10	1	-0.95	12.06	20	39.5	15.7	19	6.5
11	0	-1.28	11.65	12	52.83	5.88	33.5	18.75
12	1	-0.3	13.32	28	89.5	1.07	43	14.25

Table 1.10. Pearson r correlations and p-values for 9 variables. Data for fish and temperature represent data from June 2000, October 2000, and April 2001.

	Non-salmonid abundance	**Tributary Junction**	**Temperature Flux**	**Mean Temperature**	**Trout Abundance**	**Invertebrate Abundance**	**Chlorophyll *a***	**% Sand/ Silt**	**% Aquatic Vegetation**
Non-salmonid abundance	-	0.67 < 0.0001	0.01 0.966	0.63 < 0.0001	-0.48 0.003	0.60 < 0.0001	-0.30 0.08	0.65 < 0.0001	0.48 0.0003
Tributary Junction		-	0.24 0.158	0.31 0.068	-0.15 0.367	0.74 < 0.0001	-0.12 0.477	0.13 0.463	0.54 0.0006
Temperature Flux			-	-0.42 0.012	-0.04 0.824	- 0.21 0.21	0.27 0.11	0.07 0.705	-0.03 0.865
Mean Temperature				-	-0.42 0.010	0.27 0.106	-0.36 0.031	0.58 0.0002	0.23 0.183
Trout Abundance					-	-0.27 0.118	-0.17 0.322	-0.56 0.0004	-0.28 0.096
Invertebrate Abundance						-	-0.10 0.546	0.21 0.212	0.71 < 0.0001
Chlorophyll *a*							-	-0.35 0.034	-0.20 0.231
% Sand/ Silt								-	0.12 0.478
% Aquatic Vegetation									-

Model reduction was necessary to achieve a model with the fewest variables explaining the highest amount of variability in non-salmonid abundance. The stepwise regression procedure produced a three-regressor model including: tributary junction, maximum hourly temperature flux, and percent composition of sand and silt. This model explained 78% of the variance in fish abundance at a significant level ($p < 0.0001$). The parameter estimates from the model were used to predict abundance using the following equation:

$$\text{Non-salmonid abundance} = -8.77 + \text{tributary junction } (94.30) + \text{temperature flux } (-5.97) + \%\text{ sand/silt } (2.75)$$

Figure 1.11 shows the changes in the response of the variables maximum hourly temperature flux and percent composition of sand and silt across the range of the observed values for those variables. By holding two of the variables constant in the above equation, the effect of one variable on non-salmonid abundance is shown, conditional on the other variables being fixed at their mean values. Using this method, predicted values of fish abundance are plotted with real numbers of fish abundance. Maximum hourly temperature flux influences fish abundance in that higher numbers of fish are found in areas with higher fluctuation in temperature. Percent composition of sand and silt positively influences fish abundance such that as sand and silt increases non-salmonid numbers also increase. The change in the response of fish abundance to tributary junction is not plotted with respect to the other two variables, due to the binary nature of tributary junction. Instead, refer to Figure 1.6, which shows how tributary junctions are consistently marked with higher numbers of fish.

The relationship of fish abundance and sand and silt is strong because both variables increase with distance from the dam. Thus, the level of sand and silt is likely not a better predictor for fish abundance than distance from the dam, and does not represent a strong biological relationship. Additional two-regressor models were compared with the final stepwise regression model to evaluate the ability of simpler models to predict fish abundance. Variables were chosen which correlated highly with non-salmonid abundance, represented an inherent biological relationship with non-salmonid abundance, and could be easily measured in the field. Table 1.11 shows each predictive model with its associated statistics and confidence intervals. Mean monthly

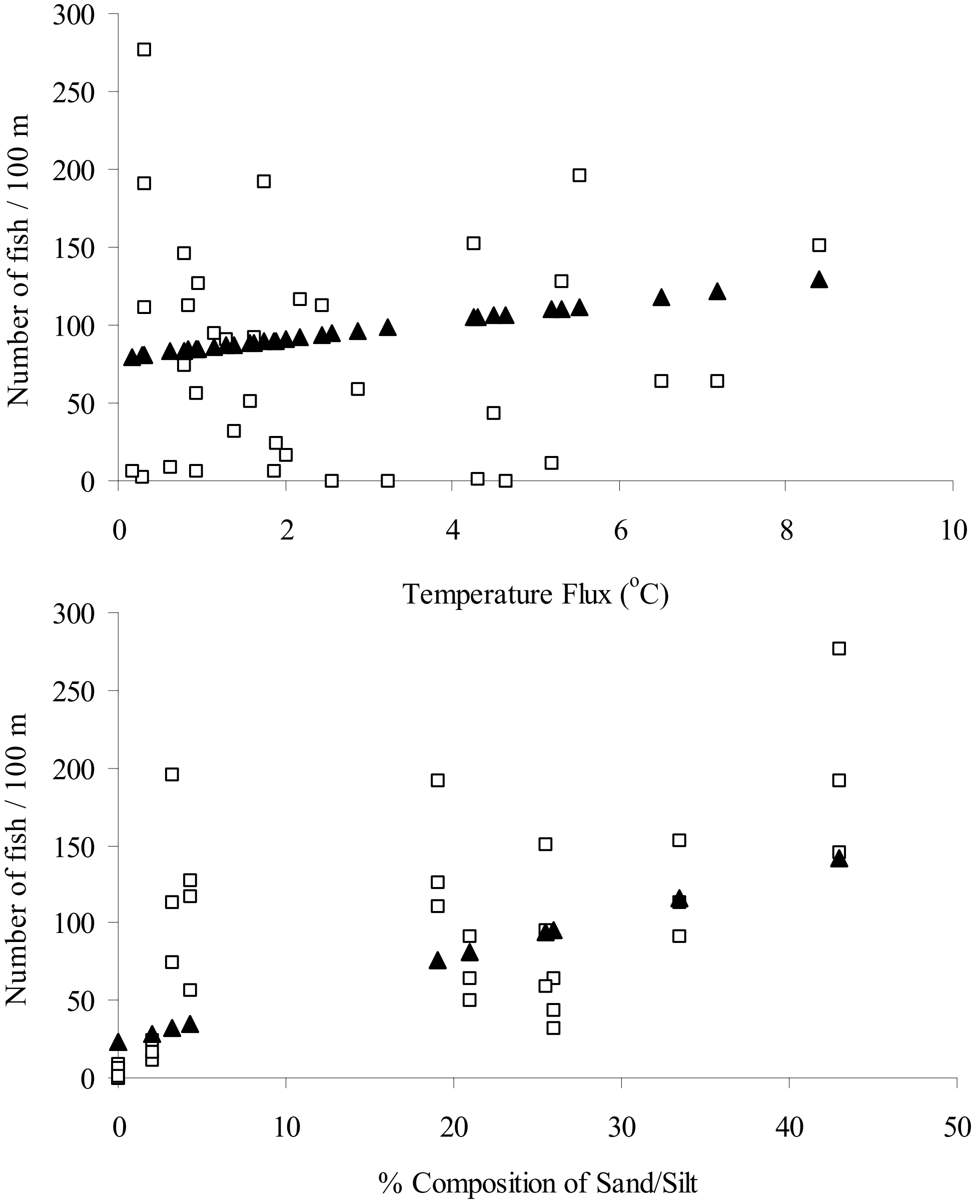


Figure 1.11. Relationship of temperature flux (a) and percent composition of sand and silt (b) to non-salmonid abundance. Temperature flux is shown as the absolute value of a depression in degrees. Triangles represent the number of non-salmonid fish predicted by the multiple linear regression model [non-salmonid abundance = -8.77 + tributary junction (94.30) + temperature flux (-5.97) + % sand/silt (2.75)]. Open squares are actual numbers of fish per 100 m for June 2000, October 2000, and April 2001.

Table 1.11. Mulitple linear regression models with respective statistics. 2-regressor model results are given to compare with the final stepwise 3-regressor model.

Multiple linear regression predictive models for non-salmonid abundance	N	df	R^2	MSE	Regressor	Confidence interval for each regressor (±)
Final Stepwise Model						
(1) -8.77 + trib (94.30) + flux (-5.97) + ssilt (2.75)	36	32	0.78	1057.09	Tributary junction	94.29
					Temperature flux	5.98
					% Sand/silt	2.77
2-Regressor Models						
(2) -96.12 + mtemp (12.74) + trib (77.61)	36	33	0.62	1843.15	Mean Monthly Temp	12.75
					Tributary junction	77.68
(3) 93.38 + bnt (-0.68) + trib (86.98)	36	33	0.58	2016.96	Brown trout abundance	0.69
					Tributary junction	86.94
(4) -139.58 + mtemp (20.95) + flux (10.48)	36	33	0.45	2712.48	Mean Monthly Temp	16.50
					Temperature flux	37.51
(5) 30.38 + trib (104.21) + flux (-5.28)	36	33	0.45	2721.22	Tributary junction	104.12
					Temperature flux	5.28

temperature and tributary junction had the lowest mean-square error (MSE) compared to the other 2-regressor models and explained 62% of the variability in fish abundance.

Canonical Correspondence Analysis

The CCA plot (Figure 1.12) depicts dominant patterns of species in relation to tributary junction, maximum hourly temperature flux, and percent composition of sand and silt. The first axis (CC1) correlated most strongly with temperature flux, having a correlation coefficient of 0.81. The species with centers of distribution corresponding to high temperature flux were *Campostoma anomalum* (CES), *Luxilus cerasinus* (CRS), *Etheostoma flabellare* (FND), *Phoxinus oreas* (MRD), and *Clinostomus funduloides* (RSD). The species that had high correspondence with tributary junctions were *Nocomis leptocephalus* (BHC), *Campostoma anomalum* (CES), *Luxilus cerasinus* (CRS). High correlates of axis 2 (CC2) included sand and silt levels and tributary junction with coefficients of 0.68 and 0.64, respectively. The species corresponding to high levels of sand and silt were *Percina roanoka* (RND), *Catostomus commersoni* (WHS), and *Notropis hudsonius* (SPS). *Etheostoma podostemone* (RWD) was closely associated with sites 10, 11, and 12. *Catostomus commersoni* (WHS), *Moxostoma erythrurum* (GOR), and *Hypentelium nigricans* (NHS) grouped together near sites 10, 11, and 12. *Clinostomus funduloides* (RSD) and *Etheostoma flabellare* (FND) were found at sites 4, 5, and 6.

DISCUSSION AND CONCLUSIONS

Longitudinal Patterns in Abundance, Diversity, and Distribution

The progressive pattern of additions of species from upstream to downstream, termed “longitudinal succession,” has been observed in headwater streams (Sheldon 1968). This concept of an upstream to downstream gradient change in the fish community has been hypothesized to exist below hydroelectric dams, based on the premise that disturbance diminishes as flow fluctuation attenuates downstream (Bain and Boltz 1989). In this study, we found results consistent with this hypothesis in that non-salmonid fish abundance and species richness increased with increasing distance from the dam. The most upstream fish community was greatly reduced compared to the most downstream fish community. However, peaks in abundance and species richness were

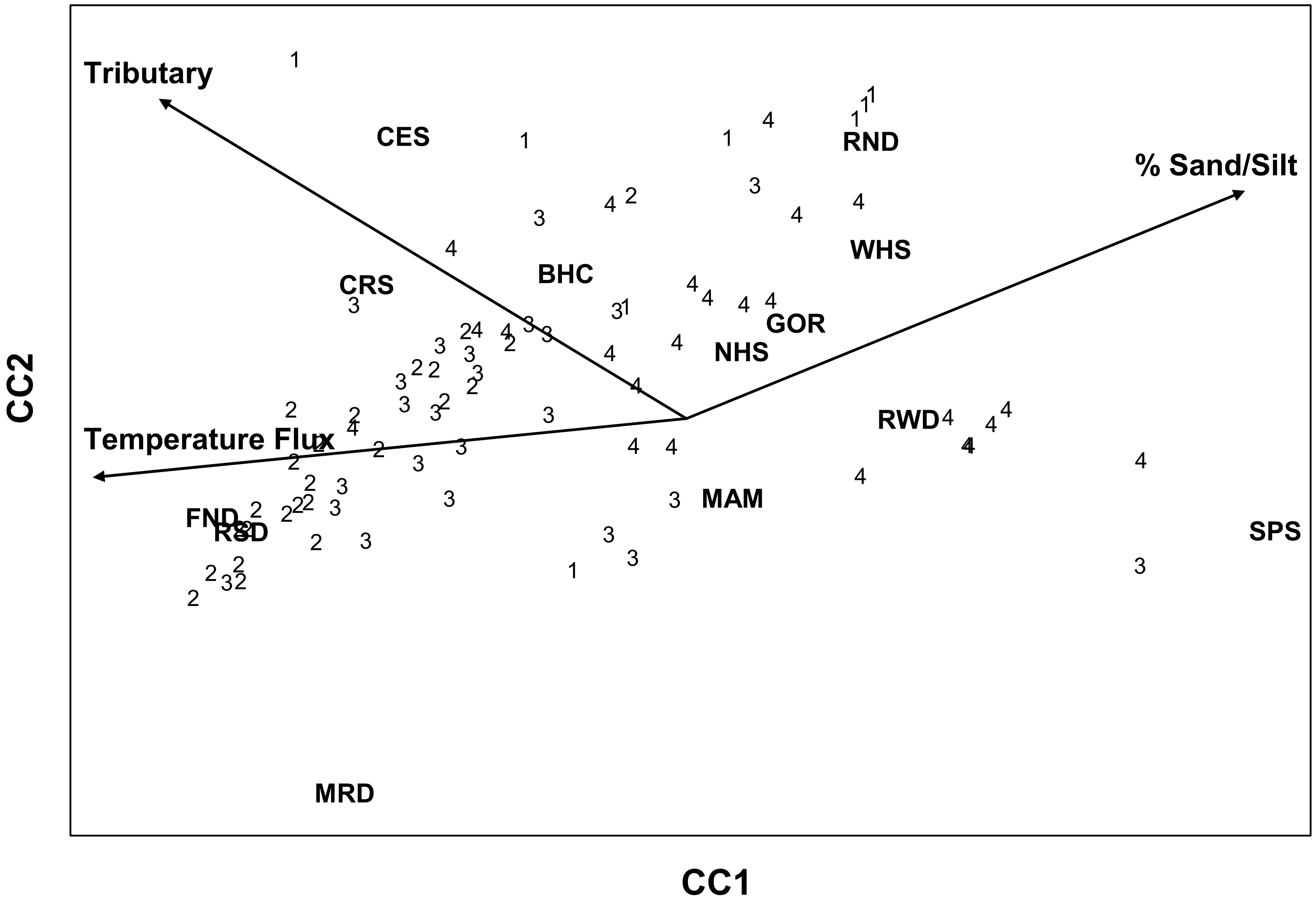


Figure 1.12. Canonical correspondence analysis of species, reaches, and environmental variables. Numbers represent reaches such that 1 = sites 1-3, 2 = sites 4-6, 3 = sites 7-9, 4 = sites 10-12. Lines represent environmental variables.

consistently found at tributary junctions, re-defining the longitudinal gradient to fluctuate in the vicinity of major tributaries.

Tributaries could provide sites of refuge from peaking flows and predation, or could represent areas with less restrictive physiological features such as more favorable temperature conditions as well as areas of greater food availability. For instance, macroinvertebrate data in the Smith River shows peaks in abundance that coincide with tributary location (Newcomb et al. 2001). The synchronized nature of high and low abundances around tributary junctions for both fish and macroinvertebrate data indicates that patterns of these taxa are not random but highly structured, suggesting a major tributary effect. Further, the dominant presence of fluvial specialists at tributary junctions suggests a relevance of tributaries for sensitive species. In a study on the Tallapoosa River in Alabama, Kinsolving and Bain (1993) also noted synchronized patterns of high and low abundance of several fish species around tributaries. Vannote et al. (1980) established the principle in the river continuum concept that tributaries have localized effects on the mainstem, which can be applied to the Smith River and its tributaries.

The most consistent peak in abundance and species richness at the most upstream tributary junction, 6.2 km below the dam, could have been driven by a tributary effect on mainstem temperature. For example, temperature in June of 2000 at this site increased to almost 17°C, comparable to the most downstream site at 23 km below the dam (Figure 1.2). After generation, the temperature increased at a faster rate at this tributary junction than at other sites. The water from the Town Creek tributary has a strong warming effect on the mainstem producing more suitable conditions for warmwater species. However, temperature flux is also great at this site because the drop in degrees during the coldwater release is more precipitous than at non-tributary sites.

Based on regression results, tributary location plays an influential role in fish numbers as does maximum hourly temperature flux. Longitudinal fish abundance also related significantly to gradients in mean monthly temperature. Certainly, the longitudinal distribution of fish provides evidence of response to temperature variation in the tailwater. The upper sites of the 24 km study area above Town Creek are predominantly fishless except for *L. cyanellus* and *C. commersoni*, two species that are capable of withstanding the cooler temperatures found near the dam. The second most numerous species in the tailwater, *N. hudsonius,* showed a distinct distributional pattern over time suggesting thermal selectivity. Out of 5,526 *N. hudsonius,* only 12 individuals were caught in the first 12.6 km below the dam, and most of these individuals were caught at the Town Creek tributary junction. *Notropis* species are known to exhibit sharp range boundaries related to fixed thermal limits that regulate their distributional patterns (Matthews 1987).

E. flabellare was the most numerous fish over all sampling periods and presumably capable to withstand the high environmental variability imposed by the flow regime. This species was ubiquitous throughout the study area and produced the highest numbers of individuals at the upper distributional range for non-salmonid species, the Town Creek tributary. Matthews and Styron (1981) found that *E. flabellare* was very tolerant of temperature fluctuations. Hlohowskyj and Wissing (1987) suggested that *E. flabellare* be considered a "thermal generalist," a fish less sensitive to temperature change. It is likely that the "thermal generalist" nature of *E. flabellare* explains its wide distribution in the Smith River.

The environmental factors most influential to the structure of the fish community in terms of abundance proved to be mean monthly temperature, maximum hourly temperature flux, and tributary location based on regression results. The two temperature variables are directly related to dam operation. Consequently, flow management may be a viable tool to increase non-salmonid productivity in the tailwater. In general, warmer waters seem to benefit fish abundance and tributaries seem to favor higher numbers of individuals.

Comparison of Fish Assemblages

Morisita's index and rank correlation tests resulted in approximately equal temporal patterns in fish assemblages across all sampling periods despite changes in environmental variation. Yet, faunal similarity was highly variable among sites, indicating a gradient of increasing consistency of composition downstream of the dam, the source of disturbance. Annual variation of ichthyofauna was greatest between June 2000 and June 2001, suggesting a possible seasonal response to the change in flow regime from the high magnitude, short duration release of 2000 to the lower magnitude, longer duration release of 2001. This variation was not observed between October 2000 and October 2001. Overall, minimal change occurred in the fish assemblages at each site through time both within and between years. If faunal "persistence" is a qualitative measure of continued presence of taxa, as considered by several authors (Connell and Sousa 1983, Ross et al. 1985, and Matthews 1986), then stream fishes of the Smith River demonstrate persistence across several years under abruptly changing, harsh conditions.

Moreover, Ross et al. (1985) found that pooling sampling stations masked variation of species ranks within individual stations. Thus, concordance of species abundance ranks was tested within each site across all sampling periods to detect both spatial variation and temporal variation. Concordant species ranks further substantiated the persistence of species and the consistency in their longitudinal distribution. It would seem that variability in the Smith River fauna is not precipitated by high environmental variability except for the much higher numbers of individuals present during the October 2002 sampling period. During this time period, abundance was higher for 10 of the 13 most common species compared to all other time periods. Higher numbers of fish seem to be a result of less flow variability during 2002 compared to study years 2000 and 2001. Though numbers of individuals were higher in this time period, faunal similarity, species ranks, and site ranks remained high among all time periods.

Angermeier and Schlosser (1989) suggest that in a system that frequently oscillates between physically harsh and benign conditions, species composition and abundance may remain in continual flux due to immigration/emigration dynamics. Though the Smith River has great environmental oscillation, faunal persistence suggests that these dynamics are not the crux of community organization. Tributaries are the only venues for fish movement into and out of the study area with downstream and upstream immigration/emigration blocked by dams. Because the 13 most common species of the mainstem are found in the tributaries, and consistent peaks of abundance and diversity occur at tributary junctions, mainstem fish assemblages could be influenced at some level by movement of fish into and out of the tributaries.

Population and Environmental Variability

Numerous queries have been made into fish community ecology, but three key hypotheses exist as to what mechanisms act as structuring agents in a fish community.

As proposed by Grossman et al. (1982), the stochastic hypothesis suggests that the relative abundance of fish is determined through the differential response of species to change in the physicochemical environment. Alternatively, the deterministic hypothesis states that biological interactions such as competition and predation regulate fish assemblages, creating highly structured communities. Finally, Strange et al. (1992) performed a 10-year study in which they found that community structure depends on how stochastic and deterministic processes combine to influence change in the fish assemblage. The mechanisms by which fish communities develop and stabilize are controversial, and particularly hard to determine due to contrasting life histories of fish species.

This research suggests that the fish assemblage of the Smith River should be placed more on the deterministic end of the deterministic-stochastic continuum because the assemblage characteristics are those of a highly structured community. The constant environmental variability of the Smith River would predictably create high variability in community structure. Yet, Moyle and Vondracek (1985) found well-defined structure in fish communities subjected to extreme flooding in a Californian stream. The longitudinal patterns of abundance, diversity, and distribution in the Smith River appear to be driven by the dynamics of flow and temperature, but the fish community persists in well-developed patterns under the constructs of this environmental variability.

CHAPTER 2: Characterizing spawning microhabitat and testing transferability for *E. flabellare* and *N. leptocephalus* from an unregulated river to a regulated river

ABSTRACT

While researchers have tested transferability of spawning microhabitat criteria for some species, such tests have not been performed for *Etheostoma flabellare* and *Nocomis leptocephalus*. I quantified spawning microhabitat use and availability for the fantail darter, *E. flabellare,* and the bluehead chub, *N. leptocephalus,* in the Smith River tailwater for substrate, velocity, and depth characteristics. I tested transferability of spawning microhabitat measured in the unregulated Roanoke River with data from the regulated Smith River. Analyses of microhabitat data for 86 *E. flabellare* nests and 292 available sites demonstrated that diameter of the spawning rock was the only variable distinguishable from what was available (Kolmogorov-Smirnov test, $p < 0.05$). Out of 44 *N. leptocephalus* mounds and 154 available sites, demersal velocity and mean water column velocity were the only variables distinguishable from available habitat (Kolmogorov-Smirnov test, $p < 0.05$). Single variable tests of transferability revealed that 5 microhabitat variables transferred well for *N. leptocephalus* while 1 variable transferred well for *E. flabellare*. In contrast, the *E. flabellare* logistic regression model had high success in predicting microhabitat use and availability while the *N. leptocephalus* model did not. These test results are similar to previous transferability tests in that a model built with a combination of variables achieved high predictability of optimal habitat for a microhabitat specialist like *E. flabellare.*

INTRODUCTION

Predictive Habitat Criteria as a Tool for Management

Identifying habitat characteristics selected by fish is an important component to fisheries management. Often, habitat characteristics are measured to develop criteria to predict habitat use, generally termed habitat suitability criteria. These criteria are accompanied by varying degrees of uncertainty, because fish do not select habitat solely on water depth, current velocity, and substrate type, which are the most commonly measured variables. Orth (1987) suggested that body size, risk of predation, presence and abundance of competitors, season, time of day, and thermal regime can alter microhabitat selection by fishes. For instance, shifts in microhabitat use might occur in the presence of competitors, making variation in microhabitat selection high between sites with different fish assemblages. Other investigators found that microhabitat use can be influenced by the energetics of foraging (Baker and Coon 1997) or diel feeding habits (Kwak et al. 1992). If the chosen criteria do not reflect microhabitat selection, then predictions of how alteration of habitat will affect a species are likely to be inaccurate.

Quantification of all variables involved in selection of microhabitat would require costly, time-intensive research. Lacking all relevant data, researchers usually infer species requirements by observing habitat use (Freeman 1999). Habitat suitability criteria such as developed for Instream Flow Incremental Methodology (IFIM) is one of several approaches to evaluate linkages between biota and instream habitat, especially under alternative flow regimes in regulated rivers (Freeman et al. 1997). These and other habitat models provide decision-makers with tools essential for fishery management. Predictive habitat criteria provide a good framework within which biologists can work until better models are available to evaluate and quantify relationships between aquatic fauna, flow, and habitat.

Why Transferability is Important: Regulated Rivers

Comprehensive strategies must be developed for improving biological function in flow-altered rivers. Since over half of the total streamflow in the world is regulated (Freeman 1999), justification exists to understand habitat requirements for preservation of aquatic fauna below dams or diversions. Typically, site-specific habitat criteria are developed to use in instream flow analyses. However, developing criteria in streams with regulated flows or degraded habitat may not reflect true habitat requirements needed for survival and persistence of a target species (Freeman et al. 1997).

Transferability of habitat criteria would provide necessary information in streams where developing those criteria is not feasible and also would reduce the cost and time needed for research. A transferable fish-habitat model can be used to predict optimal habitat for the fish in a different system than the one in which the model was developed. Thomas and Bovee (1993) defined transferability as the ability of criteria to specify optimal habitat such that utilization of higher quality habitat occurs in greater proportion than lower quality habitat. Therefore, if criteria were correctly identified for a species, then predictions could be made about high quality habitat availability for that species in other stream environments, especially in regulated rivers or ones with degraded habitat. With confounding factors between stream environments such as varying competitive interactions and temperature regimes, there is reason to believe that a transferable model is not feasible. Thus, as Groshens and Orth (1993) stipulate, there is need for multiple transferability studies to define the degree of generality in different fish-habitat models.

Testing Transferability of Habitat Criteria with Functional Significance

Transferability of habitat criteria could be greatly beneficial for predicting functionally significant habitat requirements such as spawning microhabitat, especially in regulated rivers. Because animals preferentially occupy areas that best support survival, growth, and reproduction (Freeman et al. 1997), testing for transferability of criteria that have functional significance could more precisely reflect requirements for maintaining strong populations. Since the spawning requirements of most fish have evolved to be very specific, transference of criteria based on spawning microhabitat use might be more successful than criteria based on frequency-of-use data.

This study was designed to identify spawning microhabitat criteria for two fishes and to the test the transferability of this criteria between an unregulated river (Roanoke Rive, VA) and a regulated river (Smith River, VA). *Etheostoma flabellare*, the fantail darter, and *Nocomis leptocephalus*, the bluehead chub, were chosen as target species because of their abundance in these rivers (Smith River abundance data, Appendix C). Transferability was defined as the ability of criteria to correctly specify optimal habitat or habitat utilized in greater proportion than nonoptimal habitat (Freeman et al. 1997, Thomas and Bovee 1993).

Appropriate ranges of variables for the spawning microhabitat of *N. leptocephalus* and *E. flabellare* were developed by Smith (1999) from the North and South Forks of the Roanoke River system. Through multiple logistic regression models, Smith (1999) found the combination of variables which best-predicted spawning microhabitat use with at least three variables included in the model. Models were developed for reproductive guilds to be more generalized, and were validated. Models for egg-clusterers and mound-builders performed well with a high percentage of microhabitat use predicted. In this study, successful transference of these criteria was evaluated to provide evidence for the potential to develop broadly predictive criteria based on spawning microhabitat.

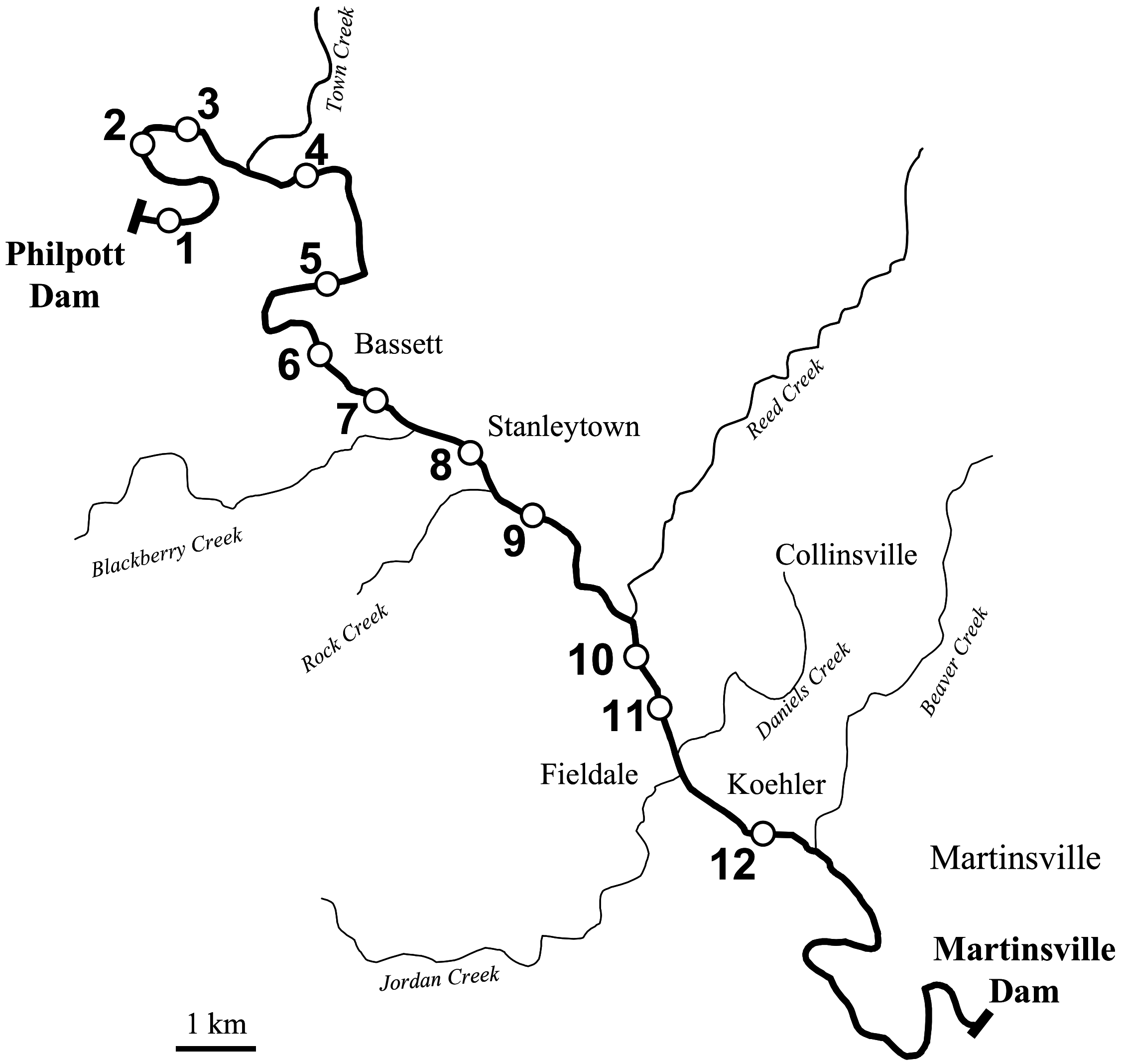
METHODS

Choice of Sample Reaches

Sections of river were identified as likely spawning areas from observations of target species made during electrofishing trips in June 2001, April 2001, and April 2002. Spawning adults of *E. flabellare* were noted if brightly-colored males with dorsal fin egg mimics and ripe females were present. The presence of individuals with fresh tuberculation and bright coloration signified likely areas for mound-building activity by *N. leptocephalus*.

Temperature data collected by Krause (2002) was reviewed to determine the upstream boundary of suitable spawning temperatures and the month spawning would most likely initiate. It was determined that mid-April would mark the beginning of the spawning season for *E. flabellare*, because of the occurrence of suitable spawning temperatures, 15^{o}C and above. The spawning window for fantail darters was expected to be between April-June. *N. leptocephalus* spawning activity was predicted to begin in May and extend through July with the appropriate temperature range of 19 to 25^{o}C (Jenkins and Burkhead 1993).

From the temperature data and observations of spawning readiness, two sections of river were chosen to intensely survey for *E. flabellare*, coinciding with 11.3 and 12.6 km below the dam (Figure 2.1). A site 6.2 km below the dam was chosen to survey, but not as intensely. Sites at 12.6, 14.3, 18.9-20.1, and 23 km below the dam were chosen to intensely survey for *N. leptocephalus.* Surveys were also completed in areas between these sites, the tributary junctions of Reed and Jordan Creek (18.6 and 21.1 km below the dam), and sites 6.2-11.3 and 15.9 km below the dam, though not as intensely. In addition, approximately 2 km of the Town Creek tributary was surveyed upstream of its confluence with the mainstem of the Smith River. The surveys were completed in two main segments with the first being to find nests and measure occupied space and the second to relocate nests and measure unoccupied space, resulting in a paired-sampling design.



Site #	1	2	3	4	5	6	7	8	9	10	11	12
km below Philpott Dam	0.5	3.4	4.2	6.2	8.9	11.3	12.6	14.3	15.9	18.9	20.1	23.0

Figure 2.1. Map showing the spawning survey sites for *E. flabellare* and *N. leptocephalus* in the Smith River tailwater.

Survey Methods to Locate Spawning Microhabitat

Underwater observations were made to locate spawning microhabitat for *E. flabellare*. During periods of base flow (~ 50cfs), at least two individuals snorkeled 200 m from the downstream to the upstream end of a sample reach. Observers moved through the areas of greatest flow when in riffle and run habitat and midway between the middle of the river and the bank on both sides of pool habitat following the protocol of Leftwich et al. (1997). Each snorkeler selectively searched for large, disc-shaped rocks in the cobble size range, the preferred spawning habitat for *E. flabellare* (Smith 1999). After a spawning site was located, the snorkeler dropped a flagged weight on the spawning rock and progressed upstream. A rock with attached eggs underneath was determined to be a spawning site if the male darter was present or if the eggs were identical to the eggs of nests where fish were present.

Wading surveys were performed to locate spawning microhabitat for *N. leptocephalus.* Two individuals moved midway between the middle of the river and the bank on both sides searching for mound structures. Each mound was flagged and labeled with a code. Mounds that appeared to have structural integrity (i.e. mound shape without rocks dispersed), and were clean of sediment and plant growth were considered to be active. All survey sites for *E. flabellare* and *N. leptocephalus* were repeatedly visited until no further sign of active spawning was apparent (i.e. no new nests found).

Measurement of Microhabitat Use

The microhabitat variables and measurement procedures used by Smith (1999) to predict spawning microhabitat for the target species were followed to minimize the likelihood that unsuccessful transference of criteria were based on discrepancies in methodology. To assess spawning microhabitat selection for the target species, a sampling frame or grid was used. A 60 by 60 cm grid was centered on the spawning site either on the nesting rock or on top of the mound. The grid was constructed of an aluminum bar frame and cord stretched horizontally and vertically, modified from the sampling frame design of Bunte and Abt (2001a). The cords were equally spaced every 10 cm around the frame creating 49 intersections at which measurements could be made.

At each *E. flabellare* nest, the grid was used to estimate percent areal coverage of silt. A visual estimate was made to the nearest 10% of how much of the grid area was covered with silt. At the spawning rock, I measured depth (cm), mean water column velocity (m/s), and demersal velocity (m/s). All velocity measurements were made using either a Marsh-McBirney flow meter or a Model 1220 Price Type "AA" current meter. The six-tenths method was used to estimate mean water column velocity (0.6 depth from the stream bottom), which produced reliable results within the depth range of the study. Demersal velocity was measured as close to the stream bottom as possible (usually within 2 cm). Embeddedness of the spawning rock was estimated to the nearest 10% as the percentage of the rock covered with sand or silt, and all 3 axes of the rock were measured using calipers. The number of healthy eggs and the number of eggs infected with fungus were noted for each nest. Finally, survey coordinates were made for each nest using a tripod and leveler. From a benchmark, an angle was shot and the distance to the nest was determined with an optical range finder.

For *N. leptocephalus*, depth and mean water column velocity were measured at 9 locations on the grid (Figure 2.2). Demersal velocity was measured at the center of the grid. The percent areal coverage of silt was estimated by counting the number of cells in the grid that were predominately covered with silt, out of 36 possible cells, and calculating a percent of coverage. At the 49 intersections of the grid, substrate particle diameter (intermediate axis) and embeddedness (percent of rock covered with sand or silt to the nearest 10%) were measured. Substrate particles were measured with a gravelometer or template. Bunte and Abt (2001b) suggests template use in determining particle size, because a template has higher accuracy than a ruler or caliper and templates reduce variability between operators. For each mound, the distance from the nearest shore, diameter of the mound (measured perpendicular to flow), and the type of cover (large woody debris, overhanging tree or bush, large rock) if present were noted.

Measurement of Microhabitat Availability

An attempt was made to sample microhabitat at 55 occupied and 200 unoccupied locations to reduce the possibility for Type I and Type II error when testing for transferability of habitat suitability criteria, as suggested by Thomas and Bovee (1993). To accomplish this, four unoccupied locations were measured to every spawning location with the goal of locating 55 spawning sites. The same microhabitat variables were

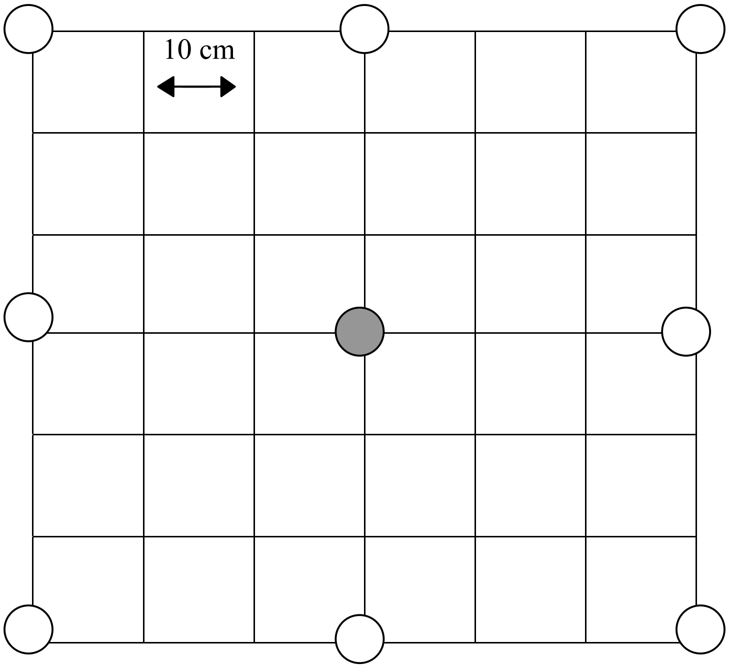


Figure 2.2. Sampling grid used for microhabitat measurements. The grid is a 60x60 cm square with 10x10 cm cells. The circles represent areas where depth and water column velocities were measured. The shaded circle marks the center of the spawning site which served as the only location to measure demersal velocity. The 49 intersections of the grid marked the specific substrate particle that was measured.

measured for unoccupied locations as occupied locations to assess characteristics of spawning microhabitat availability.

To sample spawning microhabitat availability for *E. flabellare*, nests were relocated using the survey coordinates. From the nest location, I walked the line of a random angle shot with a compass. Angles were generated using a random number generator. I walked from 1 to 8 paces while searching for a disc-shaped, small or large cobble size rock. I placed the center of the grid on the chosen rock where a nest would seem likely to be present, but was not. If a new nest was located by using this method, I returned to the original nest and walked a different line using a new angle.

The same random angle protocol was used to measure spawning microhabitat availability for *N. leptocephalus*. However, at least one grid was placed in shoreline or midchannel habitat 2 to 15 paces from the mound. The number of paces was primarily determined by characteristics of depth and ability to place the grid. I was unable to effectively sample in depths greater than about 1 m and unable to place the grid on submerged woody debris, large boulders, or trash. Shoreline habitat was designated to be 5 m or less from the bank while midchannel habitat was greater than 5 m from the bank. At least 1 of the 4 unoccupied grid samples around each mound was placed in either shoreline or midchannel habitat in order to sample each type of habitat availability.

Data Analysis

I used the two-sample Kolmogorov-Smirnov test ($\alpha = 0.05$) to determine whether a microhabitat variable was uniformly distributed among available microhabitat (Sokal and Rohlf 1995). Those variables significantly different from the available distribution were considered to be the variables most selected by spawners. In addition to testing for spawning microhabitat selection, patterns of selection were evaluated with histograms showing the percent of microhabitat used or available within specific classes of a variable. Each variable was divided into several classes following Smith (1999). Abbreviations and descriptions of variables are given in Table 2.1.

Substrate was divided into classes using a modified Wentworth scale (Gordon et al. 1992) where 0.05 mm (corresponds to silt), 1 mm (sand), 2-16 mm (small gravel), 17-63 mm (large gravel), 64-128 mm (small cobble), 129-256 mm (large cobble), and

Table 2.1. Variable categories and summary variables measured in sampling grid.

Variable Category	Variable
Depth	Depth (D)
Substrate	Median particle diameter (D50)
	Substrate roughness (D90)
	Diameter of spawning rock (Dcen)
Sedimentation	Silt
	Embeddedness (Emb)
	Embeddedness of spawning rock (Emcen)
Velocity	Demersal velocity (Vbot)
	Mean water column velocity (V)
Hydraulic	Froude number (Fr)
	Relative depth (Rel)

greater than 256 mm (boulder). Class intervals were created for embeddedness and silt in 10% increments. The depth variable categories were set at 0-20, 21-31, 32-40, 41-51, and greater than 51 cm. The designated classes for mean water column velocity and demersal velocity were less than or equal to 0.1, 0.1-0.29, 0.3-0.49, 0.5-0.7, and greater than 0.7 m/s.

The measured variables were summarized in the following way: depth and water column velocity were averaged over the 9 measurements, the median particle diameter of the 49 grid measurements was estimated to be the median sediment particle diameter or D50, and substrate roughness or D90 was estimated by using the 90th percentile of the surface sediment particle size distribution. Hydraulic variables were calculated for both *E. flabellare* and *N. leptocephalus* data.

Froude number (Fr) was calculated by the formula: mean water column velocity / $(\text{g x depth})^{0.5}$ where g is the acceleration due to gravity ($9.81\ \text{m/s}^2$). Froude number can be used as an index to describe surface turbulence characteristics such that flow is subcritical if Fr is less than 1, critical if Fr is equal to 1, and supercritical if Fr is greater than 1 (Gordon et al. 1992). Relative depth (Rel) was calculated only for *N. leptocephalus* using the ratio of depth to substrate roughness. Relative depth can be used to characterize near bed flow conditions because the flow pattern is highly dependent on the height of the substrate relative to the depth of the water (Davis and Barmuta 1989). The benthos experience turbulent flow when relative depth is high (Davis and Barmuta 1989). Froude number was divided into classes of 0-0.1, 0.1-0.14, 0.15-0.22, 0.23-0.34, and greater than 0.34. The class ranges for relative depth were 0-1.4, 1.4-2.2, 2.3-4.1, 4.2-12.4, and greater than 12.4.

I used a one-sided chi-square goodness of fit test to evaluate the transferability of spawning microhabitat variables measured in the Roanoke River to those measured in the Smith River (Zar 1996, Thomas and Bovee 1993)). For *N. leptocephalus*, separate chi-square tests were performed for depth, mean and demersal velocity, silt, embeddedness, substrate roughness (D90), median particle diameter (D50), Froude's number, and relative depth. For *E. flabellare*, tests were performed for depth, mean and demersal velocity, diameter of the spawning rock, silt, embeddedness, and Froude's number.

I classified the spawning microhabitat use and availability data as occupied, unoccupied, optimal or usable, after the study design of Thomas and Bovee (1993). Optimal conditions were designated as those that fell within the range of conditions occupied in the Roanoke River. Thus, occupied and unoccupied microhabitat was designated as usable if it fell outside the range of microhabitat occupied in the Roanoke River. The chi-square analysis tested the null hypothesis that optimal conditions were occupied in the same proportion as usable conditions (Thomas and Bovee 1993). If the null hypothesis was rejected, it was determined that optimal conditions were occupied in greater proportion, suggesting successful transference of Roanoke River spawning microhabitat criteria to the Smith River.

The logistic regression models developed by Smith (1999) to predict spawning microhabitat for *E. flabellare* and *N. leptocephalus* in the Roanoke River were cross-validated using data collected in the Smith River. A predicted probability of presence was calculated for each spawning site in the Smith River using the logistic regression models. The *E. flabellare* predictive model included the following parameters for the logistic regression function, where θ is the linear predictor of the independent variables:

$$\theta = -3.457 + \text{diameter of spawning rock}\ (0.044) + \\ \text{embeddedness}\ (-0.465) + \text{silt}\ (0.011)$$

The *N. leptocephalus* predictive model included the following parameters for the logistic regression function:

$$\theta = 14.87 + \text{diameter of the center particle}\ (-0.042) + \\ \text{substrate roughness or D90}\ (0.018) + \text{embeddedness}\ (-1.404) + \text{silt}\ (-0.708) + \\ \text{demersal velocity}\ (16.82) + \text{Froude's number}\ (-25.43) + \\ \text{Roughness Reynold's number}\ (-0.118) + \text{relative depth}\ (-0.246)$$

I was unable to calculate a Roughness Reynold's number, because I did not measure a velocity profile in the grid samples and therefore could not calculate the shear velocity of the grids. I substituted the mean for spawning habitat and available spawning habitat from Smith's data (13 and 88, respectively) into the predictive model for *N. leptocephalus*. The probabilities were estimated from the following equation adapted from Yu et al. (1995), Trexler and Travis (1993), and Knapp and Preisler (1999), where P_i is the probability of finding a spawning site at location *i*:

$$P_i = e^{\theta i} / 1 + e^{\theta i}$$

Model success for predicting spawning habitat in the Smith River was evaluated by a sensitivity and specificity analysis. I selected a probability cut-off of 0.50 such that model predictions greater than or equal to 0.5 indicated areas classified as used and predictions less than 0.5 indicated non-used areas. The sensitivity of the model was assessed by calculating the proportion of the use observations that were predicted correctly. The specificity of the model was assessed by calculating the proportion of the non-use areas predicted correctly. The models were considered to have high resolution for prediction if they both had high sensitivity and specificity.

RESULTS

Microhabitat Use and Availability

A total of 86 *E. flabellare* nests and 292 habitat availability grid samples were measured in the mainstem of the Smith River, exceeding the sample size suggested by Thomas and Bovee (1993). Spawning activity was evident from mid-April through mid-May, after which few new nests were located. Substrate diameter or the measured b axis of the spawning rock was the only variable for which *E. flabellare* spawning microhabitat use was distinguishable from what was available ($p < 0.05$; Table 2.2). Fish selected spawning rocks in the small and large cobble range (64-256 mm; Figure 2.3). The microhabitat measurements for depth, silt, embeddedness, demersal velocity, and mean velocity were not distinguishable from what was available (Table 2.2).

However, patterns of use were evident for each variable. The majority of *E. flabellare* spawners were found in depths ranging from 0 to 40 cm (Figure 2.3). Areas with levels of silt greater than 20% were most used by spawners with the actual spawning rock most often 0 to 10% embedded (Figure 2.4). Microhabitat use matched the distribution of habitat availability well for not only depths and silt levels, but also water velocities. *E. flabellare* spawners most often used demersal and mean water column velocities in the range of 0.1 to 0.29 m/s (Figure 2.5). At base flow, all nest sites had a Froude number less than 1 indicating that spawning microhabitat was found in areas with subcritical flow or slow, tranquil flow (Gordon et al. 1992). From the sample size of 86

Table 2.2. Mean ± standard deviation for spawning microhabitat variables for available microhabitat and used microhabitat by *E. flabellare* and *N. leptocephalus* spawners. Significant differences (Kolmogorov-Smirnov two-sample test $p < 0.05$) denoted by *. N is sample size.

Variables	Available	N	Used	N
E. flabellare				
D (cm)	27 ± 16	86	27 ± 16	292
B axis (mm)	116 ± 29	87	* 145 ± 45	292
Silt (%)	45 ± 36	87	29 ± 29	287
Emcen (%)	12 ± 14	86	6 ± 7	292
Vbot (m/s)	0.15 ± 0.13	86	0.14 ± 0.11	291
V (m/s)	0.23 ± 0.17	86	0.21 ± 0.14	292
Fr	0.0015 ± 0.001	86	0.0014 ± 0.001	292
N. leptocephalus				
D (cm)	35 ± 14	44	39 ± 16	154
D50 (mm)	13 ± 9	43	14 ± 3	153
D90 (mm)	67 ± 233	38	266 ± 518	153
Silt (%)	57 ± 36	44	26 ± 18	154
Emb (%)	28 ± 28	44	14 ± 13	154
Vbot (m/s)	0.26 ± 0.17	43	* 0.17 ± 0.06	153
V (m/s)	0.49 ± 0.27	43	* 0.18 ± 0.07	153
Fr	0.28 ± 0.18	43	0.09 ± 0.03	153
Rel	47 ± 294	36	1.40 ± 1.15	157

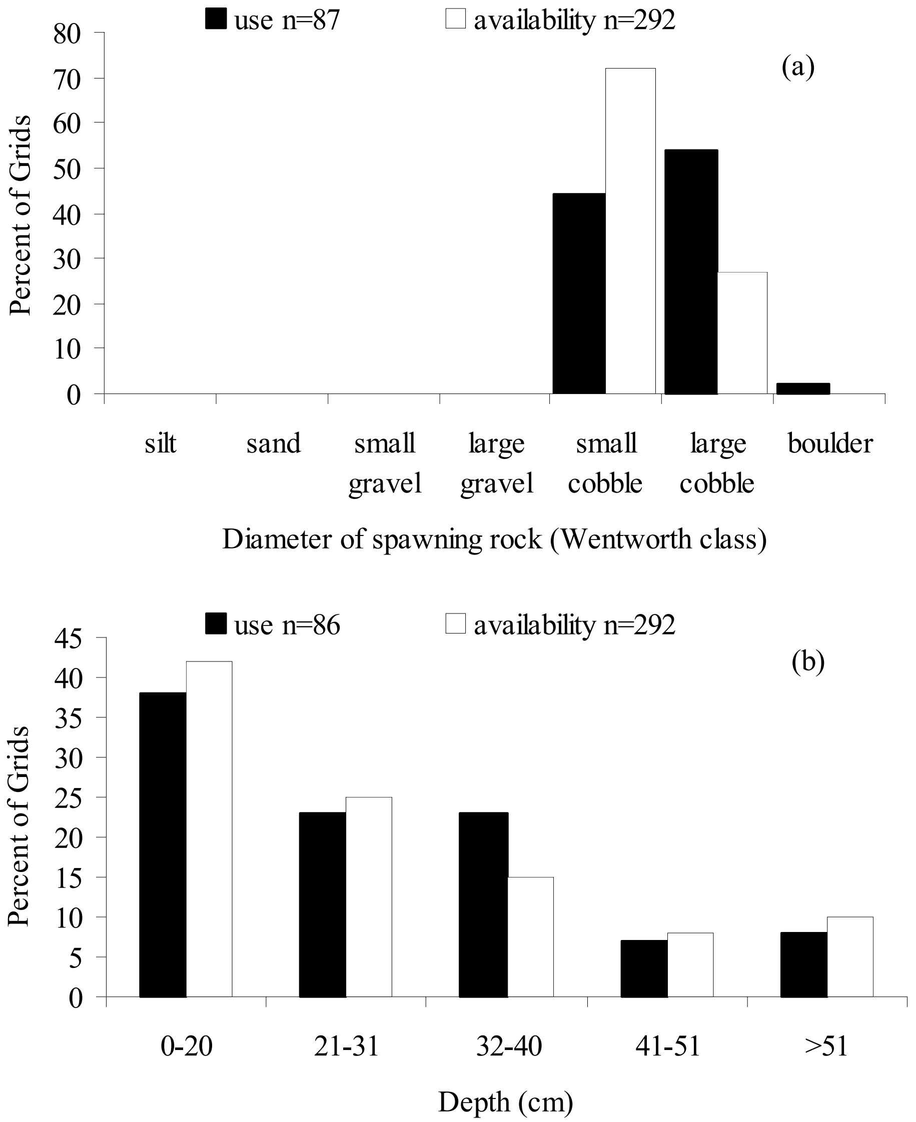


Figure 2.3. Percent of substrate categories (a) used and available for *E. flabellare* spawning rocks and percent of depth classes (b) used and available for *E. flabellare* spawning microhabitat.

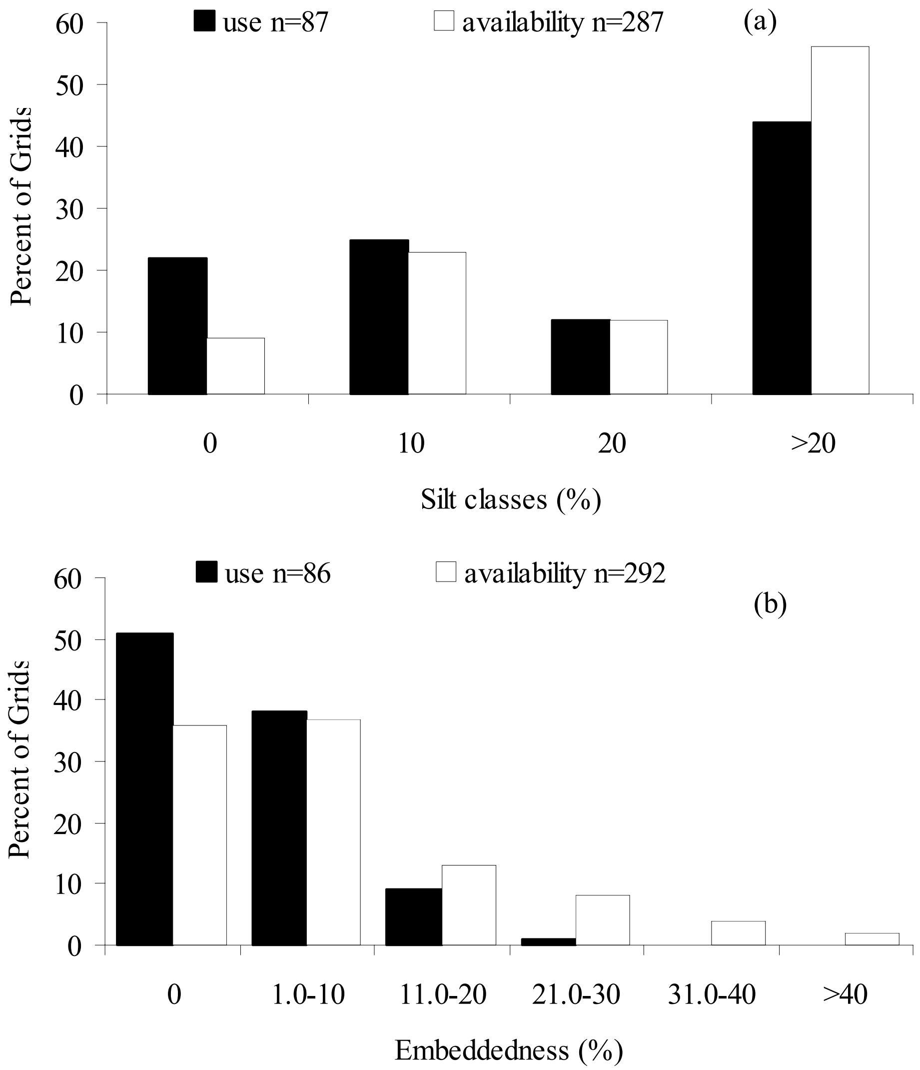


Figure 2.4. Percent of silt classes (a) used and available for *E. flabellare* spawning microhabitat and percent of embeddedness classes (b) used and available for *E. flabellare* spawning rocks.

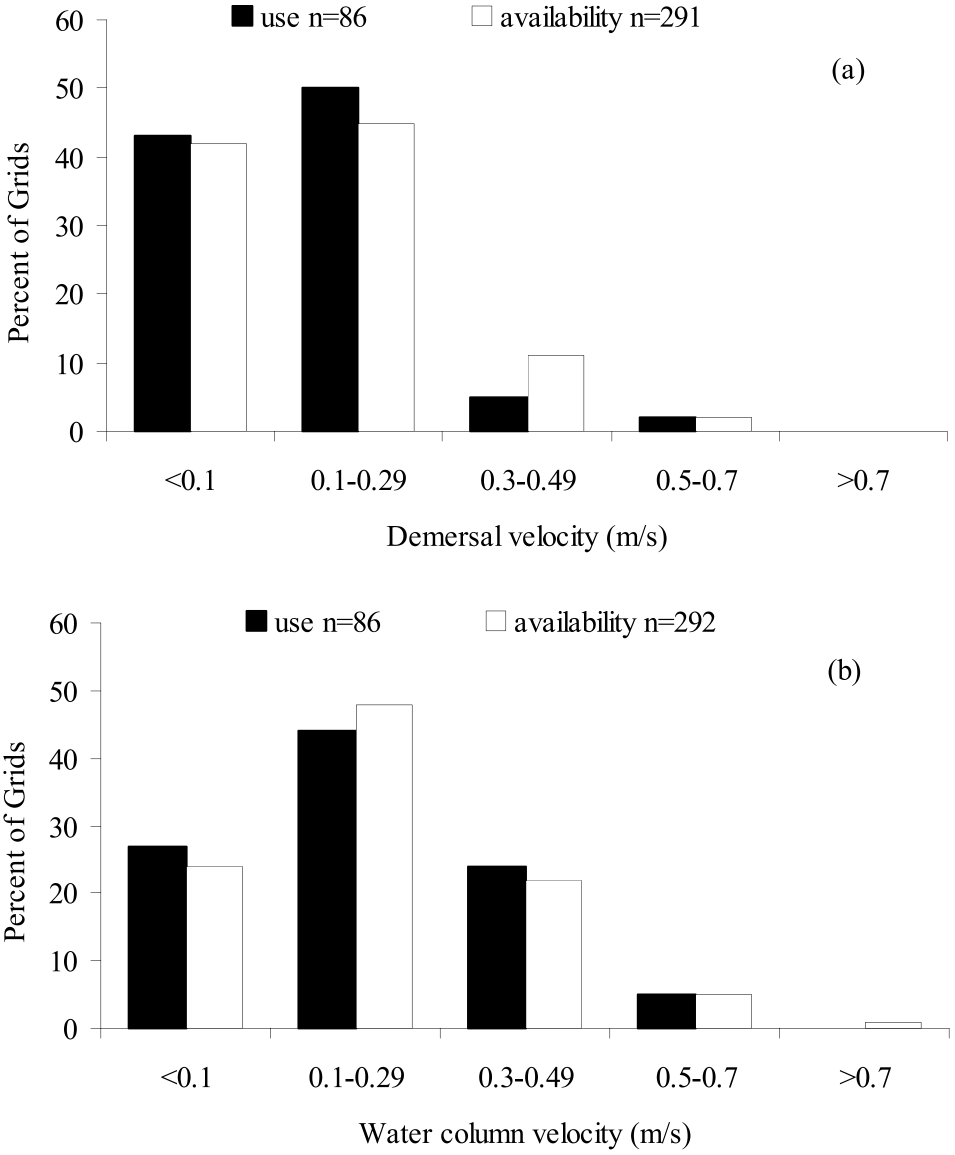


Figure 2.5. Percent of demersal (a) and water column velocity (b) classes used and available for *E. flabellare* spawning microhabitat.

nests, egg counts varied from 10 to 300 with an average of 91 eggs per nest. Among the nests that were sampled, a small percentage exhibited high levels of fungal growth over the eggs suggesting minimal if any reproductive success at that nest. *E. flabellare* were observed frequently guarding their nests during the study.

A total of 44 *N. leptocephalus* mounds and 154 unoccupied sites were measured, under-representing the sample size suggested by Thomas and Bovee (1993). I observed evidence of active spawning in the mainstem from the beginning of June to the middle of July. The average diameter of each mound was 0.6 ± 0.1 m with the largest diameter measuring 0.95 m and the smallest 0.3 m. Mounds were found in shoreline habitat (5 m or less from shore) with an average distance from shore of 3.5 ± 4.1 m. Three mounds were located in the midchannel of the river measuring 7, 10, and 20 m from the closest shoreline. Mounds were found almost without exception adjacent to some type of cover including small boulders, submerged trees, overhanging shrubs, or trash (i.e. television, rusty barrel, metal wheel case). Few direct observations were made of spawning fish over a mound.

The variables for which spawning microhabitat use was distinguishable from available habitat for *N. leptocephalus* were demersal velocity and mean water column velocity ($p < 0.05$; Table 2.2). Velocity was highly selected for in the 0.1 to 0.29 m/s range (Figure 2.6). Although the microhabitat measurements for depth, embeddedness, silt, substrate roughness (D90), and median particle diameter (D50) were not distinguishable from available habitat (Table 2.2), distinct patterns of habitat use for these variables were apparent. The preferred range for silt levels was greater than 20% while embeddedness tended to be between 1 and 10% (Figure 2.7). The depth class most selected for fell within the range of 21 to 31 cm (Figure 2.8). The median particle diameter for all mounds was most commonly in the small gravel category (Figure 2.9). And, the 90th percentile of the particle size distribution was in the large gravel category (Figure 2.9). The calculated relative depth for microhabitat use was not discernable from what was available, but was predominantly less than 1.4 (Figure 2.10). At base flow, all mounds had a Froude number less than 1 indicating that spawning microhabitat was found in areas with subcritical flow or slow, tranquil flow (Gordon et al. 1992).

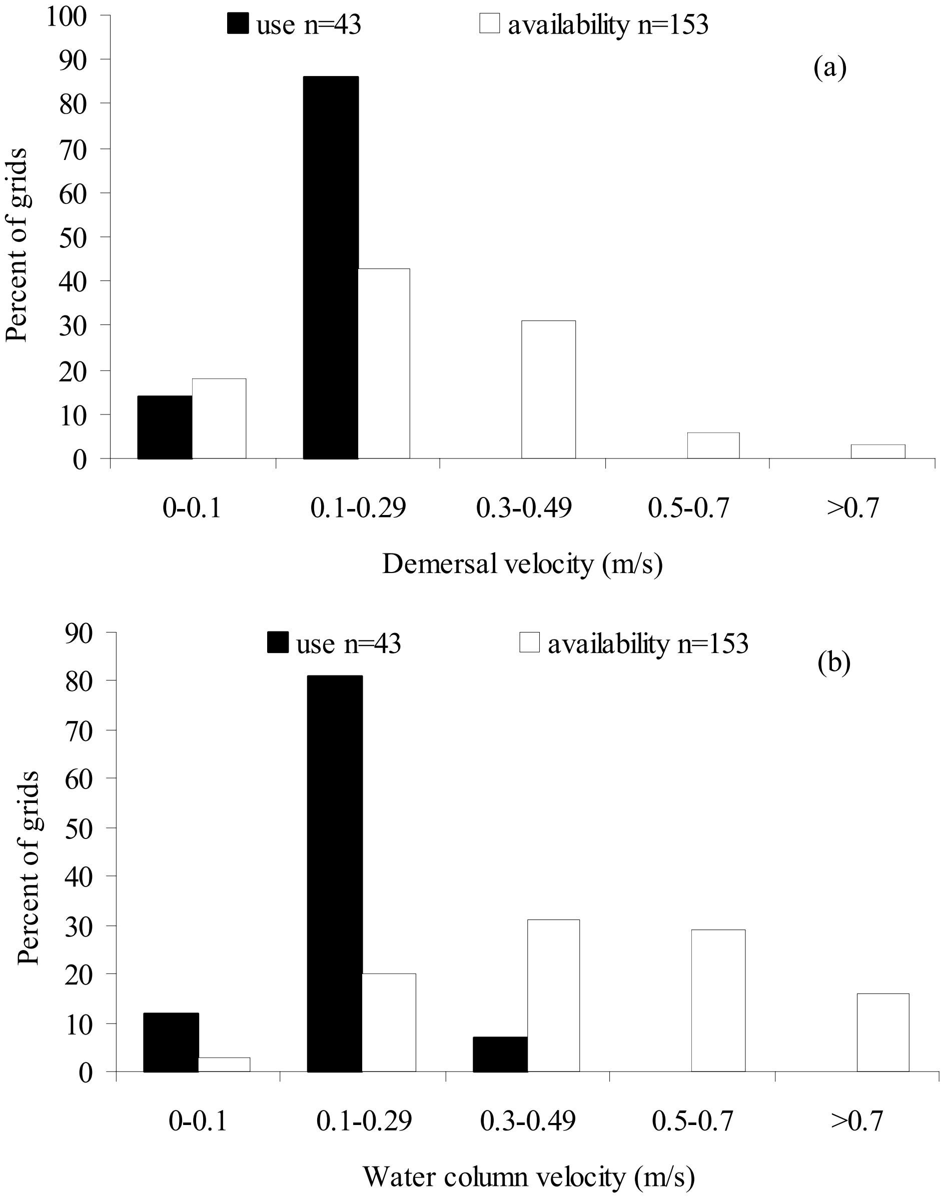


Figure 2.6. Percent of demersal (a) and mean water column velocity (b) classes used and available for *N. leptocephalus* spawning microhabitat.

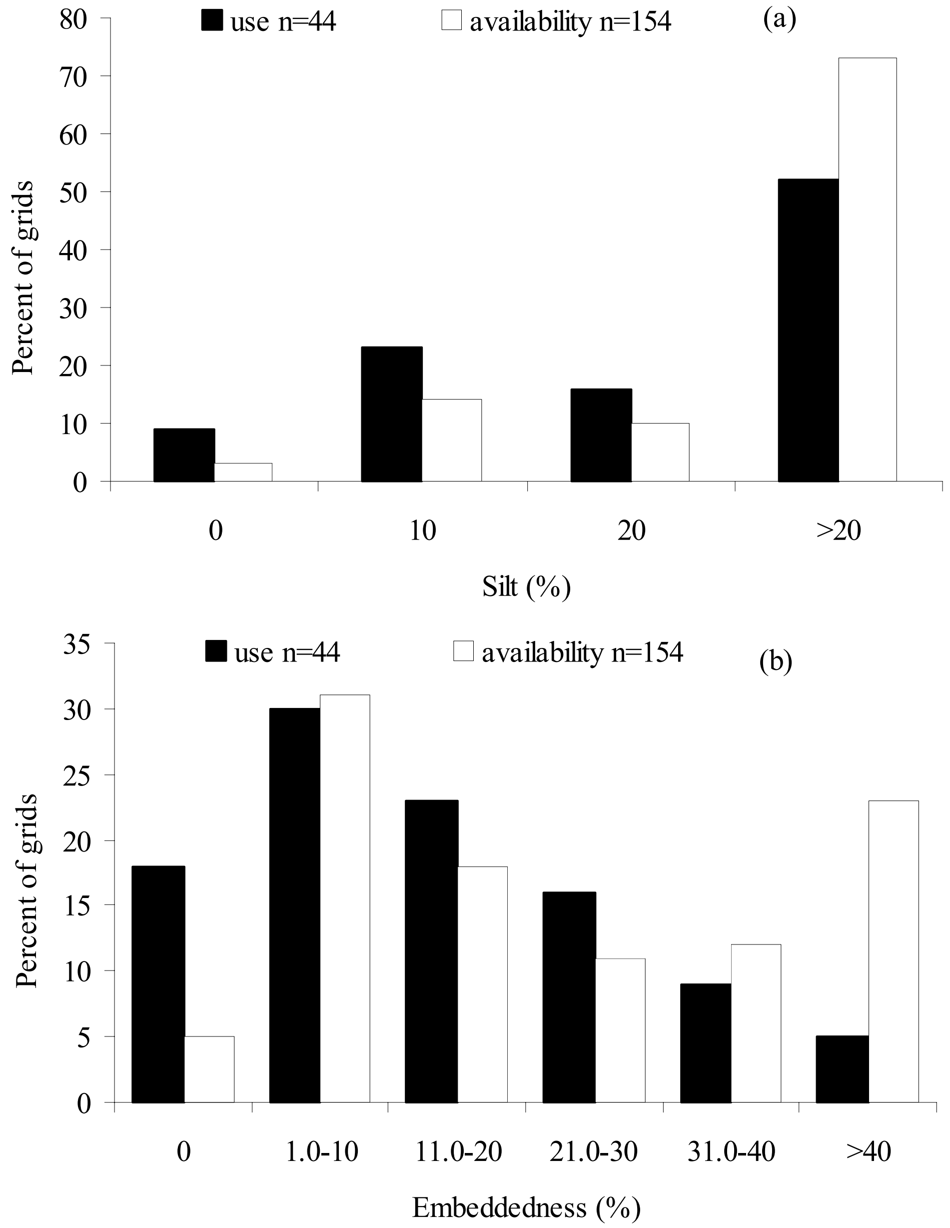


Figure 2.7. Percent of silt (a) and embeddedness (b) classes used and available for *N. leptocephalus* spawning microhabitat.

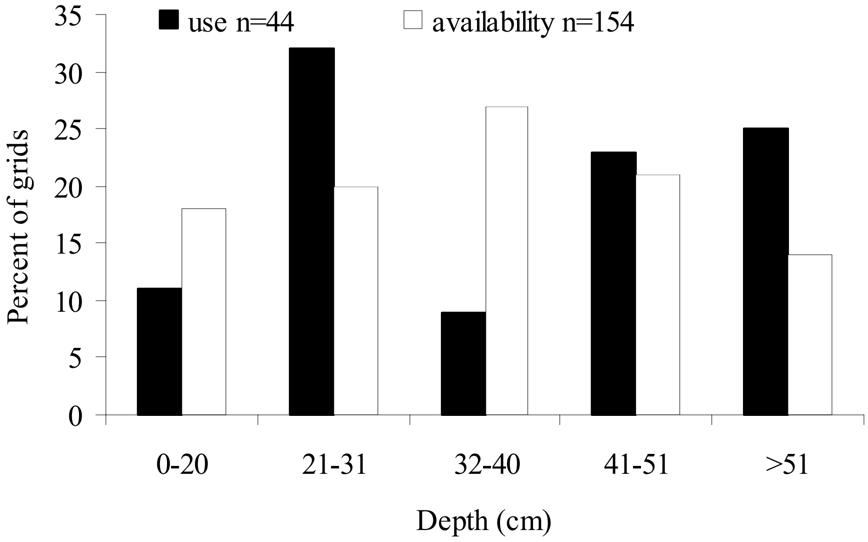


Figure 2.8. Percent of depth classes used and available for *N. leptocephalus* spawning microhabitat.

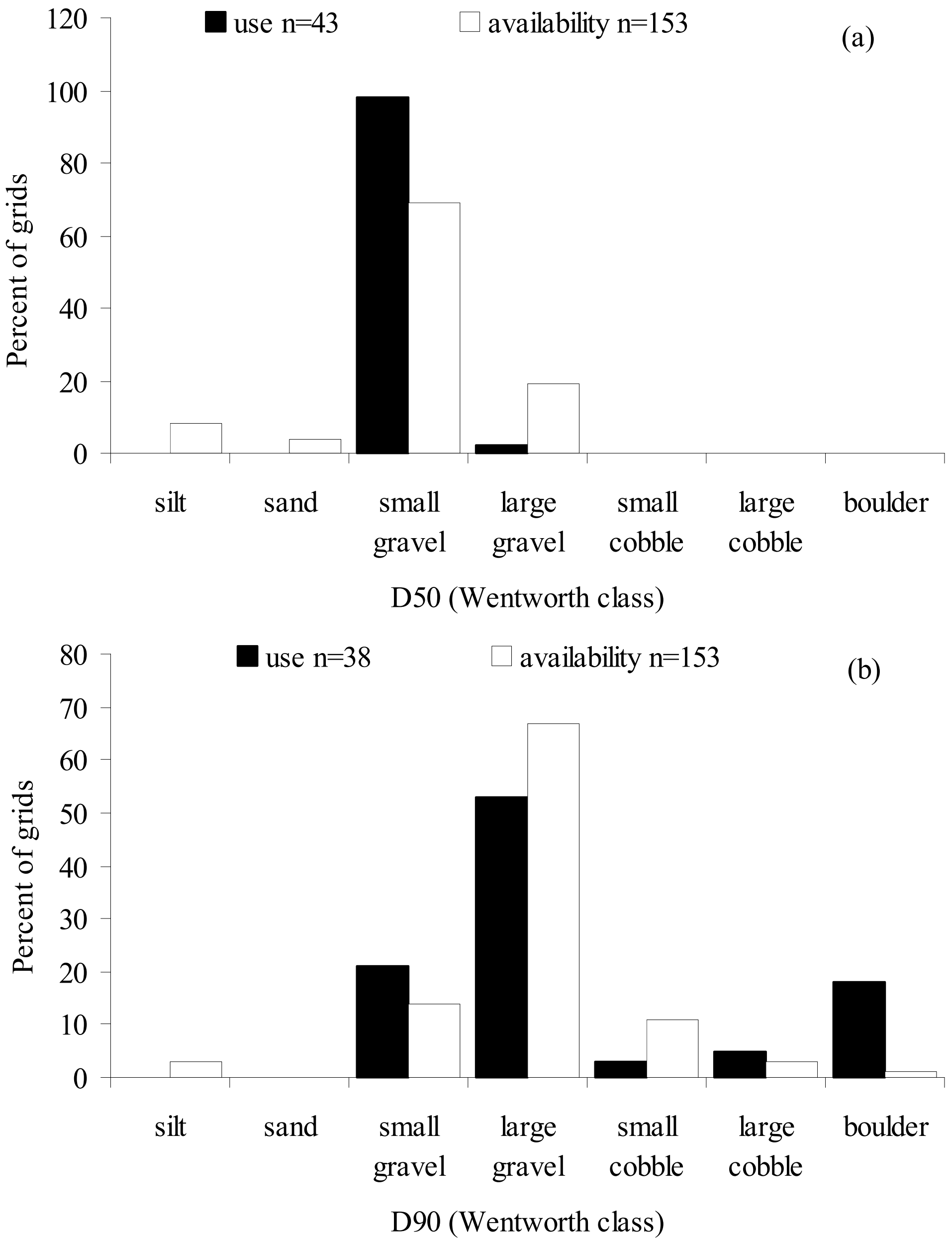


Figure 2.9. Percent of median particle diameter (a) and substrate roughness (b) classes used and available for *N. leptocephalus* spawning microhabitat.

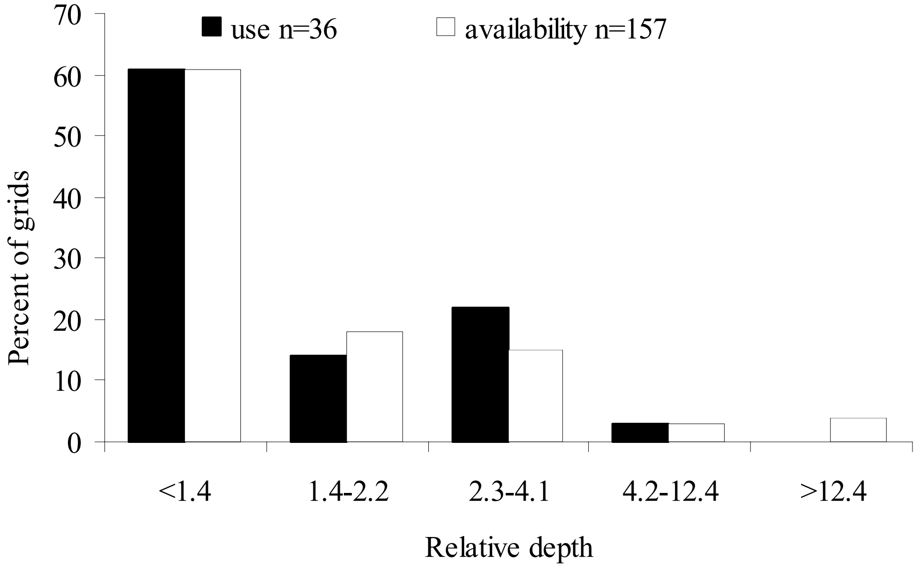


Figure 2.10. Percent of relative depth classes used and available for *N. leptocephalus* spawning microhabitat.

Tests of Transferability

Criteria for spawning microhabitat successfully transferred for some variables from the unregulated Roanoke River to the regulated Smith River (Table 2.3). Spawning microhabitat criteria transferred well for *N. leptocephalus* with the null hypothesis being rejected for several variables. For this species, optimal spawning microhabitat was occupied in greater proportion than usable spawning microhabitat for mean water column velocity, demersal velocity, silt, substrate roughness (D90), and median particle diameter (D50). Spawning microhabitat criteria did not transfer well for *E. flabellare* with only one transferable variable. Optimal spawning microhabitat was occupied in greater proportion than usable spawning microhabitat for embeddedness of the spawning rock.

Unlike the single variable tests of transference, the logistic regression model for *E. flabellare* from the Roanoke River yielded high levels of sensitivity and specificity when cross-validated with data from the Smith River (Table 2.4). The *E. flabellare* model predicted 59% of the spawning locations and predicted 64% of the unoccupied locations. The high predicted probabilities of use are most attributable to the egg-clustering reproductive strategy of *E. flabellare* having a strong affinity for small and large cobble spawning rocks as well as low embeddedness and silt. Unlike the single variable tests of transference, the *N. leptocephalus* logistic regression model predicted spawning habitat poorly, classifying 34% of the actual spawning locations correctly while classifying 98% of the unoccupied locations correctly.

DISCUSSION AND CONCLUSIONS

I tested the hypothesis that a reproductive site was more likely to be found in optimal habitat as defined by spawning microhabitat criteria developed by Smith (1999). Although it is likely that multiple factors interact to determine an optimal state for spawning microhabitat, it is also likely that individual variables strongly influence microhabitat use (Freeman et al. 1997). Thus, single variables were tested for transferability to determine which were most critical for successful spawning of target species.

Freeman et al. (1997) suggested that microhabitat criteria for riffle fishes would be most likely to transfer in comparison to criteria for species occupying a variety of pool

Table 2.3. Transferability of microhabitat variables between the unregulated Roanoke River and the regulated Smith River based on a one-sided chi-square goodness of fit test. Variables successfully transferred if optimal microhabitat was occupied in greater proportion than usable microhabitat ($p < 0.05$). Successful transference denoted with (+).

Variables	Test statistic	Conclusion
	E. flabellare	
D (cm)	3.26	
D50 (mm)	0.01	
Silt (%)	0.87	
Emb (%)	6.74	+
Vbot (m/s)	0.06	
V (m/s)	0.00	
Fr	0.00	
	N. leptocephalus	
D (cm)	2.09	
D50 (mm)	9.83	+
D90 (mm)	8.45	+
Silt (%)	11.55	+
Emb (%)	0.04	
Vbot (m/s)	10.44	+
V (m/s)	43.09	+
Fr	0.00	
Rel	1.58	

Table 2.4. Classification table for logistic regression model cross-validation to estimate probability of presence. Classifications based on 0.5 probability cut-off.

	Number of habitat units			
		Predicted with fish		
Data set	Observed with fish	Present	Absent	Correct
E. flabellare				
Present	86	51	104	59%
Absent	287	35	183	64%
Total	373	86	287	
N. leptocephalus				
Present	38	13	3	34%
Absent	157	25	154	98%
Total	195	38	157	

and riffle habitats. In this study, spawning microhabitat criteria for substrate and velocity transferred well for *N. leptocephalus* though this species is categorized as a habitat generalist. Spawning microhabitat conditions for the habitat specialist, *E. flabellare,* did not transfer well. These results are not similar to those of Freeman et al. (1997) who found nearly all criteria for two darter species to transfer between an unregulated stream to a regulated river. But, Freeman et al. (1997) did not test transferal of combined criteria based on spawning microhabitat.

By testing Smith's logistic regression models from the Roanoke River to predict spawning microhabitat in the Smith River, a combination of variables were tested for transference or successful differentiation of optimal spawning habitat between rivers. The *E. flabellare* model had a high level of predictive success with over half of the actual spawning sites and available sites correctly classified. The *N. leptocephalus* model had a much lower level of predictive success. Thus, the model for the habitat specialist proved to be the best predictive model using a combination of three variables to predict spawning location.

Other studies have had mixed success at predicting spawning location of fish based on a combination of spawning microhabitat criteria. Shirvell (1989) had poor prediction capability of chinook salmon spawning areas when using PHABSIM, a part of the Instream Flow Incremental Methodology, with both generic and river-specific habitat suitability criteria. In contrast, Knapp and Preisler (1999) used nonparametric logistic regression to identify spawning microhabitat criteria for California golden trout and then developed a parametric model to predict spawning sites. Their results indicated that spawning locations were predictable based on a combination of microhabitat criteria.

In this study, successful criteria transference indicated those habitat conditions most critical for spawning success, while poor transference of variables could indicate several possibilities. First, poor transference would have occurred if environmental factors that dictate optimal spawning microhabitat in the Smith River were different from the Roanoke River. If key variables that explain spawning microhabitat use in the Smith River were missing from the Roanoke River models (such as habitat conditions under peak flow), then they would have had low resolution. Second, optimal conditions were not available in the Smith River. Third, the microhabitat scale at which the fish habitat models were tested was not the scale at which spawning individuals were most influenced. Finally, changes in critical variables under a peaking regime could have prevented target species from spawning in optimal habitat.

In the Smith River, this final consideration is the most probable cause for poor transference. Because of the daily flux in critical variables, it is possible that changes in velocity or temperature dictate the habitat fish are able to use for successful reproduction. Based on the results from this study, it is likely that this is the case for *N. leptocephalus.* If so, this species may not be able to use optimal habitat as defined by the Roanoke River models, but instead is forced to spawn in areas that fall into a usable range of critical variables.

I concluded that the best predictive model for spawning habitat was the logistic regression model for *E. flabellare*, because the parameters of the model provided high resolution in predicting the spawning microhabitat for this species. Because the model was developed with information from several egg-clusterer species, it is probable that this model represents a broadly transferable model. The ability of the model to predict spawning microhabitat suggests that diameter of the spawning rock, embeddedness, and silt are crucial microhabitat features for the spawning success of *E. flabellare* in regulated and unregulated rivers.

SUMMARY

According to Petts (1984), a hierarchical framework can be used to assess the effect of flow regulation below a dam. By this framework, fish populations represent the most progressive impact in a regulated river. In this study, analysis of fish community structure provides evidence that the full effects of flow regulation are present in the Smith River. Because the non-salmonid fish community shows a persistent response to variable flows and temperature, I concluded that the biotic properties of the tailwater are in a highly impacted state. The community response to flow and temperature is most evident by the following patterns: low abundance and diversity in the first few kilometers below the dam, isolated peaks in abundance and diversity at tributary junctions, increasing abundance, diversity, and faunal similarity with increasing distance from the dam.

The fish community shows consistent longitudinal patterns of abundance, diversity, and distribution of species such that these community attributes do not markedly differ over time. The consistency in these results is surprising because the community patterns are well-developed, even under the highly variable flow conditions of 2000, 2001, and 2002. However, in contrast to a persistent longitudinal pattern of abundance, “stability” or constancy in numbers of individuals over time was not evident during this study (Ross et al. 1985).

Though relative abundance was not statistically different between time periods, the numbers of fish caught in the October 2002 sampling period were markedly higher than all other sampling periods. The high numbers of fish caught during this sampling period coincided with a discharge schedule that had lower magnitude and duration releases compared to time periods in 2000 and 2001. Variable flow years offered an opportunity to assess fish community response to different flow regimes. In effect, a natural experiment which tested recruitment under different flows took place with 2000 representing a harsh recruitment year, 2001 a medium recruitment year, and 2002 a mild recruitment year. Decreased discharge appeared to allow recruitment to be stronger during the mild flow year of 2002 compared to the previous years of the study, resulting in strong year classes for 10 out of the 13 most common species. Results indicate that lower magnitude discharge and duration of release, provides more suitable conditions for the non-salmonid species in the Smith River.

Reduction of productivity in tailwaters has been observed (Cushman 1985). I predict that the productivity in the Smith River is depressed, though no quantitative comparison was made between fish numbers in the Smith River and a similar unregulated river. My prediction is based on spawning surveys for *Nocomis leptocephalus* in the Smith River and Town Creek which likely suggest a depressed population of this species in the mainstem. I surveyed 55 active mounds in Town Creek, over a 3-day period, within approximately 2 km of stream from the tributary junction. In comparison, I surveyed 44 active mounds in the mainstem, over a 2-month period, within approximately 6 km of river. I would expect to find more mound-building activity in a higher order stream such as the mainstem, because of greater habitat availability. Yet, other regulatory factors besides habitat availability (i.e. variable flows, temperature) must be limiting mound-building activity in the mainstem.

An important component of productivity in the Smith River is reproductive success of those species that are present. In effect, the future of any fishery is linked with how successful species are able to propagate. In this study, I found that successful spawning for *Etheostoma flabellare* depended on presence of suitable small and large cobble size rocks. For *N. leptocephalus,* preference for slower velocity habitat determined successful spawning areas. Since Smith (1999) found that *E. flabellare* spawning microhabitat was very stable in areas of high bed movement, the preference of spawning rocks might be the main determining factor for spawning success of the most dominant fish in the Smith River. Likewise, the ability of *N. leptocephalus* to utilize both shoreline habitat for mound-building and cover, as found in this study, (i.e. submerged wood, small boulders) allows this species to occupy slower velocity habitat in which to spawn.

MANAGEMENT IMPLICATIONS AND FUTURE DIRECTIONS

Enhancing the non-salmonid community in the Smith River could depend on several factors. Changing the operation of the dam to a discharge schedule with lower magnitude and duration releases could increase the productivity of the fishery by increasing numbers of individual species. Improving the water quality in tributaries (i.e. enforcing water quality standards, decreasing sources of sedimentation, educating the public about watershed dynamics to help decrease litter input) could enhance the non-salmonid fish community since tributaries appear to moderate the effects of flow regulation. Decreasing the input of sediment into the river could aid reproductive efforts of benthic animals, such as *Etheostoma flabellare*, and could benefit benthic feeders such as *Percina rex* (the federally endangered Roanoke logperch). Because *Nocomis leptocephalus* mounds in the Smith River were found adjacent to cover objects almost without exception, planting cover objects (i.e. small boulders) throughout the mainstem could provide velocity shelter for this species to utilize for mound-building.

The Smith River represents a complex environment for non-salmonid species with no single mechanism driving community dynamics. Based on this study, effort to enhance the fish community in one of the areas mentioned above is a step in the right direction. Because this study showed that numbers of fish dramatically increased under a lower magnitude and duration flow release, changing the operation of Philpott Dam accordingly would prove to have the greatest impact on productivity in the Smith River.

I concluded that the fish community does not appear to be stabilized in terms of constancy in numbers of fish due to flow variability (Ross et al. 1985). Therefore, long-term monitoring is needed in the Smith River to evaluate the full extent of changes in fish community characteristics over time. In a tailwater with a hypolimnetic release, Quinn and Kwak (2003) documented the need for long-term monitoring of fish populations, which are not stabilized, to prevent inappropriate management actions. If the number of sites which were sampled in this study is reduced, I recommend the following six locations for long-term monitoring efforts: 6.2, 14.3, 15.9, 18.9, 20.1, and 23 km below the dam (Figure 1.4). Four of these sites represent the main tributary junctions and each represents consistently high abundance of non-salmonid fishes in the tailwater. The top three locations I would suggest include: 6.2, 18.9, and 23 km below the dam. These sites are tributary junction sites. The location at 6.2 km represents the uppermost distributional range for most non-salmonid species while locations 18.9 and 23 km represent sites with the strongest populations of non-salmonid species. I recommend that monitoring efforts which reflect trends over time (Quinn and Kwak 2003) should continue in the Smith River until data suggests stabilized populations.

Broader implications can be made from this research.   My results indicate that tributaries can play a major role in the structure of fish communities. Thus, it follows that tributary junctions should be considered as sampling stations for similar studies. Likewise, maximum hourly temperature fluctuation can influence fish community structure and proves to be an important variable to monitor, especially in a hypolimnetic tailwater. Finally, my results indicate that fish-habitat models developed to predict functionally significant habitat requirements such as spawning microhabitat are transferable and represent a viable tool for management.

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APPENDIX A. CHLOROPHYLL A

Chlorophyll a Substrate Sampling

I chose 20 sample sites corresponding with the 12 fish sampling sites and 8 additional sites, one above and below each of the 4 main tributary junctions. A sample of 5 rocks was taken from a riffle in each site the last week of both June and July 2002. Small cobble rocks were selected with a diameter of 100-150 mm from the middle of the channel. June and July represent a time of year when photoperiod is high and were chosen because I expected to sample the highest rate of primary production during this time of year. The rocks were covered with aluminum foil, placed on ice, and transported to a lab facility at Virginia Tech where they were kept in a freezer.

Chlorophyll a Extraction

The upper surface of the rock (surface not embedded but exposed to light) was scrubbed with a metal brush to remove any plant or algae growth. Distilled water was minimally used to wash the rock surface while scrubbing. A subsample (at least 10 ml) of this solution with known volume was filtered onto a glass fiber filter. To estimate the surface area, each rock was wrapped with aluminum foil (of known weight cm^{-2}), the foil was trimmed to cover the upper surface of the rock, and the final foil hat was weighed. A single filter was prepared for each rock sample, labeled, and frozen. I was unable to use this technique for rocks with high levels of filamentous plant growth.

Filters were frozen at least 24 hours before inserting them into separate Falcon tubes each with 10 ml of basic acetone solution. The tubes were covered in aluminum foil to minimize exposure to light and kept refrigerated for at least 20 hours, but no more than 24 hours. Afterwards, the samples were centrifuged for 5 minutes at 1000 RPM before the final step in pigment analysis. I transferred 3 ml of each sample to a 1-cm cuvette and read optical densities at 750, 664, and 665 nm with a spectrophotometer. Then, 0.1 ml of 0.1 N HCl was mixed with each sample to acidify it, and after 90 s the same optical densities were read.

Chlorophyll a Calculation

$$\text{Chlorophyll } a\ (\text{ug/cm}^2) = 26.7\ (E_{664b} - E_{665a}) * \text{volume of acetone for extraction (ml)} / \text{rock area (cm}^2) * \text{length of path light through cuvette (cm)}$$

Where,

Rock area (cm^2) =

[known area of foil (cm^2) / known weight of foil (g)] * weight of rock foil hat

E_{664b} =

[absorbance of sample at 664 nm – absorbance of sample at 750 nm] before acidification

E_{665a} =

[absorbance of sample at 664 nm – absorbance of sample at 750 nm] after acidification

Procedures to extract and calculate chlorophyll *a* were modified from Hauer and Lamberti (1996). Absorbance readings were converted into mg/m^2 of chlorophyll *a* present. Due to high quantities of filamentous plant growth on a portion of the rock samples, chlorophyll *a* was not extracted for some sites including: below Town Creek (5.5 km), site 4 (6.2 km), above Jordan Creek (21 km), below Jordan Creek (21.2), and site 12 (23 km) (Table A.1, Figure A.1).

Table A.1. Chlorophyll *a* content for two sampling periods, June and July 2002, for 20 sites in the Smith River.

Site	Distance from dam (km)	Chlorophyll *a* (mg/m^2) June, 2002	Chlorophyll *a* (mg/m^2) July, 2002
1	0.5	21.04	25.99
2	3.4	2.16	4.39
3	4.2	1.43	7.63
Upstream Town Creek	5.3	9.86	11.16
Downstream Town Creek	5.5	-	19.05
4	6.2	-	3.33
5	8.9	3.67	6.04
6	11.3	3.67	3.81
7	13	3.36	1.77
Upstream Blackberry Creek	13.3	1.63	2.63
Downstream Blackberry Creek	13.5	2.71	10.86
8	15.3	8.64	5.18
9	15.9	3.61	8.93
Upstream Reed Creek	18.5	22.28	3.37
Downstream Reed Creek	18.7	5.20	7.86
10	18.9	11.06	15.70
11	20.5	3.70	5.88
Upstream Jordan Creek	21	0.80	-
Downstream Jordan Creek	21.2	1.07	-

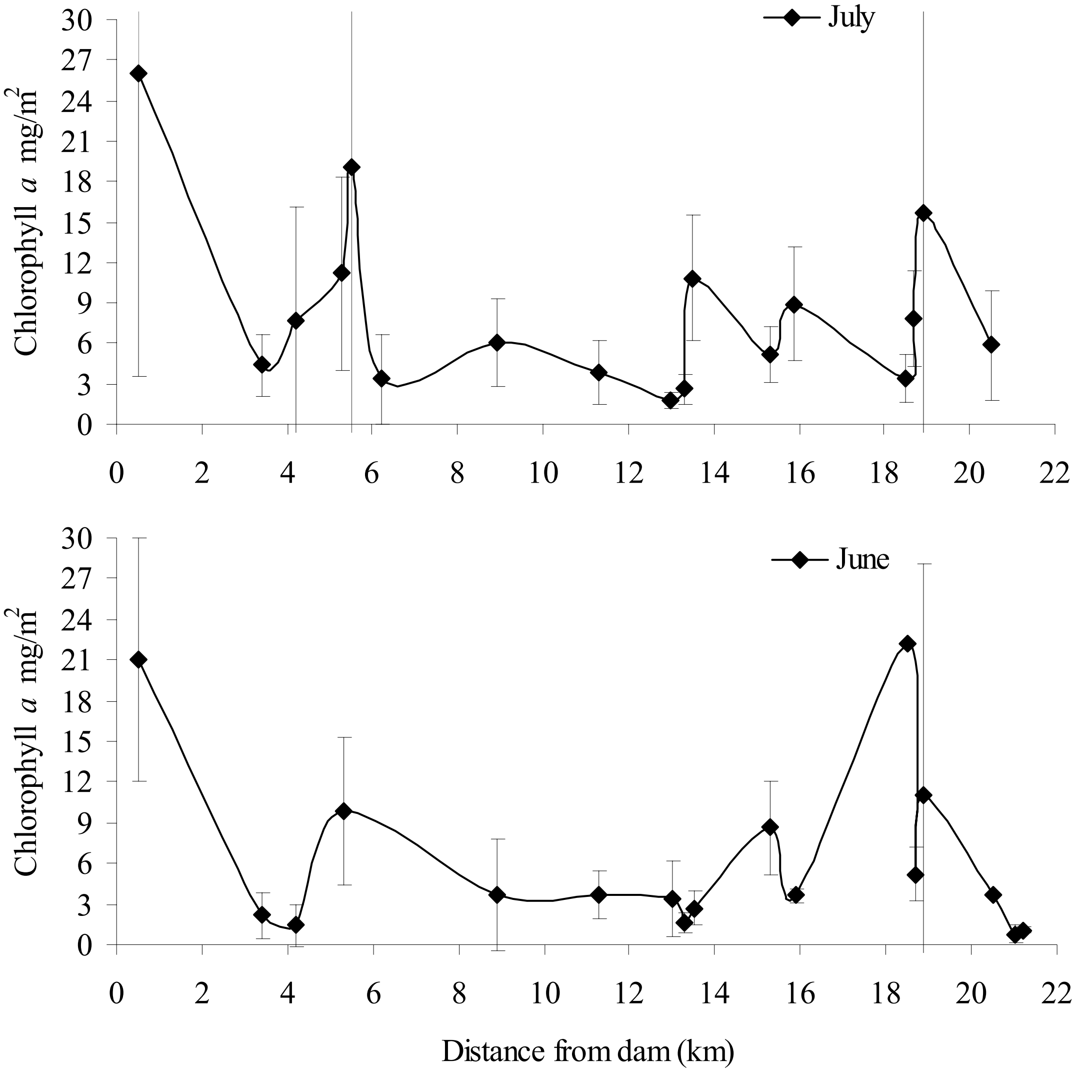

Figure A.1. Chlorophyll *a* content for sampling periods in June and July 2002 with distance from the dam (95% confidence intervals).

APPENDIX B. ROANOKE LOGPERCH DATA

Table B.1. Roanoke Logperch, *Percina rex*, occurrence in 12 sites below Philpott Dam in the Smith River. Sampling periods include: June and October 2000, April, June, and October 2001, April, June, and October 2002.

Sampling Date	Location (KM below dam)	Number Caught
June 2000	13	2
June 2000	20.5	3
April 2001	8.9	1
April 2001	11.3	1
April 2001	15.3	1
April 2001	18.9	1
April 2001	20.5	1
April 2001	23	1
June 2001	20.5	3
June 2001	23	7
October 2001	18.9	2
October 2001	20.5	4
April 2002	8.9	1
April 2002	15.9	1
April 2002	18.9	2
April 2002	20.5	2
April 2002	23	1
June 2002	15.9	1
June 2002	18.9	3
June 2002	20.5	1
June 2002	23	1
October 2002	18.9	3
October 2002	20.5	2
October 2002	23	4

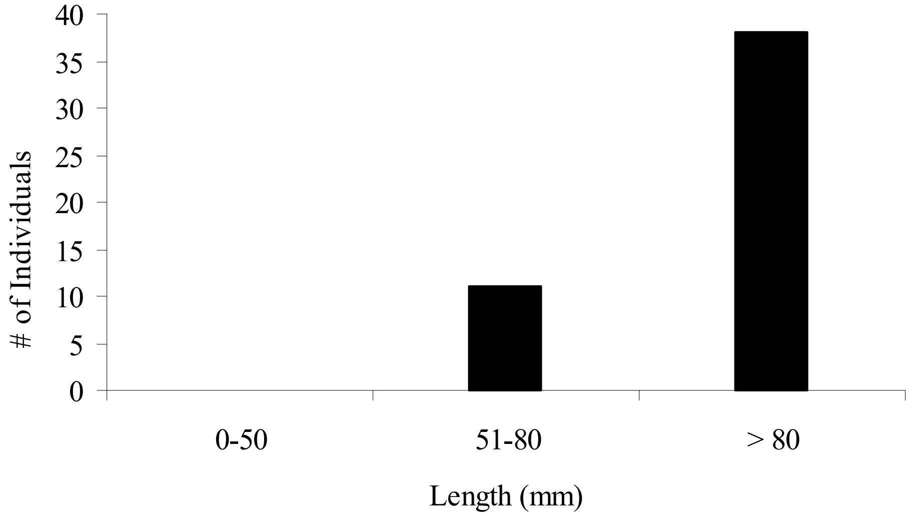

Figure B.1. Roanoke Logperch, *Percina rex*, length frequency of all individuals caught during sampling periods June and October 2000, April, June, and October 2001, April, June, and October 2002. A length of greater than 80 mm represents an adult (Jenkins and Burkhead 1993).

APPENDIX C. RELATIVE ABUNDANCE DATA ARCHIVE

Table C.1. Relative abundance (number of fish per 100 m) within sites 1-12 for individual species during June 2000.

Species	**Sites**												
Catostomidae	1	2	3	4	5	6	7	8	9	10	11	12	**Total**
White sucker	2			32	1	14	10	34	42	1	6	101	243
Northern hogsucker				1		2			1	4			8
Roanoke hogsucker										2			2
Golden redhorse				12			3		1			3	19
V-lip redhorse													
Black jumprock						1	1			6	1		9
Cyprinidae													
Bluehead chub				6	1	12	40	47	7	84	34	35	266
Creek chub				2				2			4		8
Central stoneroller							25			7			32
Cutlips minnow						1						2	3
Rosyside dace							4	27		3	3		37
Crescent shiner				4			2	7	1	7			21
Mountain redbelly dace								9					9
Spottail shiner													
Swallowtail shiner													
Redlips shiner												1	1
Golden shiner													
White shiner													
Whitetail shiner										2			2
Rosefin shiner					1								1
Centrarchidae													
Largemouth bass													
Smallmouth bass							1	1				1	3
Redbreast sunfish							1			1	2		4
Green sunfish	2			2		1				3			8
Buegill	2			1			1			3			7
Roanoke bass													
Black crappie													
Percidae													
Fantail darter				63	9	30	58	62	6	46	55	3	332
Glassy darter													
Riverweed darter				1				4	6	1	5		17
Roanoke darter						2		3		20	28		53
Roanoke logperch							2				3		5
Ictaluridae													
Brown bullhead													
Margined madtom				4		1	1			3	12		21

Table C.2. Relative abundance (number of fish per 100 m) within sites 1-12 for individual species during June 2001.

Species	**Sites**												
Catostomidae	1	2	3	4	5	6	7	8	9	10	11	12	**Total**
White sucker				7			37	2	1		7	22	76
Northern hogsucker							5				1	1	7
Roanoke hogsucker								2					2
Golden redhorse													
V-lip redhorse													
Black jumprock										1		1	2
Cyprinidae													
Bluehead chub				6	1	4	27	30	9	43	40	70	230
Creek chub			1					4					5
Central stoneroller							1					2	3
Cutlips minnow				1			4					1	6
Rosyside dace						5	6	44	5	15	5	5	85
Crescent shiner				1				6	1	2		2	12
Mountain redbelly dace								9					9
Spottail shiner						1	2	66	5	35	4	183	296
Swallowtail shiner				2								3	5
Redlips shiner													
Golden shiner													
White shiner													
Whitetail shiner										1			1
Rosefin shiner													
Centrarchidae													
Largemouth bass													
Smallmouth bass							1						1
Redbreast sunfish				2		1							3
Green sunfish	1	1	2	4					2				10
Buegill					1			2	2	2			7
Roanoke bass													
Black crappie													
Percidae													
Fantail darter				2	20	29	38	60	37	24	59	78	347
Glassy darter							1	1			4		6
Riverweed darter				1	2		5	16	4	1	18	1	48
Roanoke darter									2	2	27	11	42
Roanoke logperch											1	6	7
Ictaluridae													
Brown bullhead				1							1		2
Margined madtom				1			1		1		3	1	7

Table C.3. Relative abundance (number of fish per 100 m) within sites 1-12 for individual species during June 2002.

Species	Sites											
Catostomidae	1	2	3	4	5	6	7	8	9	10	11	12	**Total**
White sucker	4			1		2	31	21	11	5	7	146	228
Northern hogsucker					1		6	1		8		2	18
Roanoke hogsucker						1							1
Golden redhorse								1				26	27
V-lip redhorse													
Black jumprock										2			2
Cyprinidae													
Bluehead chub				7		6	18	16	22	141	33	36	279
Creek chub				2						1			3
Central stoneroller								1		1			2
Cutlips minnow							3		1			4	8
Rosyside dace						1	26	13	3	18	1		62
Crescent shiner							1		1	3			5
Mountain redbelly dace				1			1	11					13
Spottail shiner				1			2	8	5	68	4	26	114
Swallowtail shiner				1									1
Redlips shiner													
Golden shiner													
White shiner				5			12			2			19
Whitetail shiner													
Rosefin shiner													
Centrarchidae													
Largemouth bass							1			1			2
Smallmouth bass					1								1
Redbreast sunfish													
Green sunfish		1	3	1									5
Buegill			1					1	3	6	2		13
Roanoke bass													
Black crappie													
Percidae													
Fantail darter				63	22	26	23	122	119	85	168	21	649
Glassy darter				3						2	2		7
Riverweed darter				4	1			12	16	9	5		47
Roanoke darter								1		31	40	2	74
Roanoke logperch								1		2			3
Ictaluridae													
Brown bullhead													
Margined madtom				1						5	2		8

Table C.4. Relative abundance (number of fish per 100 m) within sites 1-12 for individual species during April 2001.

Species	**Sites**											
Catostomidae	1	2	3	4	5	6	7	8	9	10	11	12	**Total**
White sucker	5			7	5	8	20	13	13	17	19	44	150
Northern hogsucker						1	1	1		2	5		9
Roanoke hogsucker													
Golden redhorse	1			3			1		5		10	1	21
V-lip redhorse													
Black jumprock													
Cyprinidae													
Bluehead chub				15	3	2	26	22	7	26	25	25	151
Creek chub													
Central stoneroller			1			1						1	2
Cutlips minnow							2					1	2
Rosyside dace				2		1	8	1		3	6	3	24
Crescent shiner								1		2	1	1	4
Mountain redbelly dace				1	1		2	1			1		5
Spottail shiner				2						66	15	89	173
Swallowtail shiner													
Redlips shiner													
Golden shiner													
White shiner				1							1		3
Whitetail shiner					1		1						1
Rosefin shiner													
Centrarchidae													
Largemouth bass													
Smallmouth bass													
Redbreast sunfish													
Green sunfish							1	1	1	1		1	5
Buegill													
Roanoke bass													
Black crappie													
Percidae													
Fantail darter				78	7	31	31	34	21	4	4	19	227
Glassy darter													
Riverweed darter				5					1	1	1	1	9
Roanoke darter							1		1	2	1	4	9
Roanoke logperch													
Ictaluridae													
Brown bullhead													
Margined madtom				2			1	1		1	1		5

Table C.5. Relative abundance (number of fish per 100 m) within sites 1-12 for individual species during April 2002.

Species	Sites												
Catostomidae	1	2	3	4	5	6	7	8	9	10	11	12	**Total**
White sucker	5	1		9	9	5	12	11	10	13	16	38	130
Northern hogsucker							2	2	1	3	1		7
Roanoke hogsucker										2			2
Golden redhorse							2		4		4	4	14
V-lip redhorse													
Black jumprock													
Cyprinidae													
Bluehead chub				5	5	5	6	5	3	17	7	9	63
Creek chub													
Central stoneroller						1						1	3
Cutlips minnow							1						1
Rosyside dace						4	6	1	4	3	1		18
Crescent shiner				2						1			2
Mountain redbelly dace				1			1					1	2
Spottail shiner							3		1	49	45	117	215
Swallowtail shiner													
Redlips shiner													
Golden shiner													
White shiner	1				1	1	2	1		5		3	13
Whitetail shiner													
Rosefin shiner													
Centrarchidae													
Largemouth bass													
Smallmouth bass													
Redbreast sunfish													
Green sunfish									1			1	1
Buegill										1			1
Roanoke bass													
Black crappie													
Percidae													
Fantail darter				26	10	14	23	10	31	6	9	2	132
Glassy darter											3		3
Riverweed darter				1	1	1			5	2	8	2	20
Roanoke darter											3	3	6
Roanoke logperch										1	1		1
Ictaluridae													
Brown bullhead													
Margined madtom						1							1

Table C.6. Relative abundance (number of fish per 100 m) within sites 1-12 for individual species during October 2000.

Species	**Sites**												
Catostomidae	1	2	3	4	5	6	7	8	9	10	11	12	**Total**
White sucker	2	2	2	3		4	11	19	33	6	76	70	228
Northern hogsucker		1		3		3	1		2	10	4	3	27
Roanoke hogsucker													
Golden redhorse		2	3	7	2		3		1	1		2	21
V-lip redhorse													
Black jumprock													
Cyprinidae													
Bluehead chub				8	6	10	26	31	6	55	9	73	224
Creek chub								2					2
Central stoneroller		1		1			2	7		2		8	21
Cutlips minnow							1	1				1	3
Rosyside dace			1			4	5	22	1			16	49
Crescent shiner				3			2	4	1	3	1	3	17
Mountain redbelly dace							2	4	2			1	9
Spottail shiner											12	36	48
Swallowtail shiner													
Redlips shiner													
Golden shiner													
White shiner										1	1	1	3
Whitetail shiner		1		12	2		2						17
Rosefin shiner													
Centrarchidae													
Largemouth bass				1		1							2
Smallmouth bass				1						2	1	1	5
Redbreast sunfish												1	1
Green sunfish				1				1		1		1	4
Buegill			2	3						1	1	1	8
Roanoke bass													
Black crappie													
Percidae													
Fantail darter			1	13	15	10	4	11	25	6	3	29	117
Glassy darter										1	1	1	3
Riverweed darter								8	13	16	4	18	59
Roanoke darter								2	3	6		8	19
Roanoke logperch													
Ictaluridae													
Brown bullhead													
Margined madtom				1									1

Table C.7. Relative abundance (number of fish per 100 m) within sites 1-12 for individual species during October 2001.

Species	**Sites**												
Catostomidae	1	2	3	4	5	6	7	8	9	10	11	12	**Total**
White sucker		1		1		1	3	6	8	9	16	4	49
Northern hogsucker							3			4	2	1	10
Roanoke hogsucker								1		3	1		5
Golden redhorse										3	5	1	8
V-lip redhorse													
Black jumprock													
Cyprinidae													
Bluehead chub				9	3	4	18	13	8	16	50	33	155
Creek chub					1			1	2	1	1		6
Central stoneroller					1	1		1		1	1	6	11
Cutlips minnow				1			3				1	2	7
Rosyside dace				1	3	1	7	8	4	13	12	1	51
Crescent shiner				11	1			2		6	3	8	32
Mountain redbelly dace				2	1			7	1		1		13
Spottail shiner					1		9	2	48	14	65	29	167
Swallowtail shiner				1									1
Redlips shiner													
Golden shiner													
White shiner							8	4		4	4		20
Whitetail shiner											6		6
Rosefin shiner													
Centrarchidae													
Largemouth bass										1	1		1
Smallmouth bass										1			1
Redbreast sunfish													
Green sunfish		1		1							5	1	8
Buegill									1	2	8		11
Roanoke bass													
Black crappie													
Percidae													
Fantail darter				26	9	22	13	20	78	12	70	19	268
Glassy darter							1	1		1	15		17
Riverweed darter				1		1	1	6	34	12	22	3	81
Roanoke darter									1	3	4	6	14
Roanoke logperch										1	1		2
Ictaluridae													
Brown bullhead													
Margined madtom				1			1			1			2

Table C.8. Relative abundance (number of fish per 100 m) within sites 1-12 for individual species during October 2002.

Species	**Sites**												
Catostomidae	1	2	3	4	5	6	7	8	9	10	11	12	**Total**
White sucker	1			3		3	18	1	66	42	159	94	386
Northern hogsucker			1				1			18	4	9	32
Roanoke hogsucker							1			3	6	6	15
Golden redhorse												1	1
V-lip redhorse													
Black jumprock								3		2			4
Cyprinidae													
Bluehead chub		1			7	18	27	36	27	66	44	84	309
Creek chub							1	10	5		7		23
Central stoneroller						2	2	2	1	6	2	20	35
Cutlips minnow							3	1		10	12	9	34
Rosyside dace				2	11	13	24	2	7	6	23	5	92
Crescent shiner							2	2		3	3	15	24
Mountain redbelly dace						3	15	21	1				39
Spottail shiner					1		45	3	657	32	316	78	1132
Swallowtail shiner													
Redlips shiner													
Golden shiner													
White shiner						1	7	1		16	26	14	64
Whitetail shiner													
Rosefin shiner													
Centrarchidae													
Largemouth bass										1	14		16
Smallmouth bass													
Redbreast sunfish											1		1
Green sunfish	1		1	2									3
Buegill			1						3		1		5
Roanoke bass													
Black crappie													
Percidae													
Fantail darter				10	15	22	19	188	74	26	24	89	466
Glassy darter				1						7	4	1	13
Riverweed darter					1		1		24	125	76	53	280
Roanoke darter						1	3		1	34	34	49	121
Roanoke logperch										1	1	2	4
Ictaluridae													
Brown bullhead													
Margined madtom										1	4		6

VITA

Anne Katherine Hunter was born on December 17, 1976. She grew up mostly in Alabama before moving to Nashville, Tennessee to attend Vanderbilt University. As an undergraduate student she studied biology, thinking she wanted to pursue medicine. During the summer of her sophomore year, she took an internship with Steve Moore, fishery biologist for the Great Smoky Mountain National Park. By happenstance, she fell in love with fishery work (and the man who would become her husband). She sought more practical work experience and took a job with the US Fish and Wildlife Service in southwestern Alaska the following summer.

While performing hook-and-line sampling for hours upon hours and days upon days in the wilds of the Alaskan backcountry, fishery work kind of got into her blood. She worked again for the US Fish and Wildlife Service the following summer of 1999 in Utah, after graduating from Vanderbilt. Then, in 2000 she moved to Vancouver, British Columbia where she worked for the School for Field Studies, Center for Fishery and Forestry Management. During this time, she received notification of acceptance into the graduate program at Virginia Tech and moved to Blacksburg in the fall of 2000 to start school.

Major influences in her life to encourage her interest in the field of fisheries include fishing with her grandparents throughout her life, learning how to fly-fish when she was 17 with her dad, and realizing in the Smokies that she could do this for a living.