The Effects of Coal Mining on

Sedimentation and Fish Assemblages in the Powell River, Virginia

by

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(ABSTRACT)

An ecological study was undertaken on the Powell River system in Virginia from 1988 - 1990 to evaluate the effects of land uses on hydrology, water quality, sedimentation (particle size < 2 mm), and fish assemblages and to derive ecological indicators for monitoring. The hydrologic (disturbance) regime of the sixth order reach was classified. Although surface coal mining appears to have reduced flow variance, the changes were not sufficient to alter the hydrologic classification.

Tributaries draining coal-mined lands had elevated specific conductivity, iron, manganese, sulphate, and turbidity relative to tributaries in unmined watersheds or in the sixth order mainstem ($p \leq 0.05$). The fifth order reach, which had a greater proportion of watershed area surface-mined, exhibited higher specific conductivity, iron, sulphate, turbidity, and total solids than the sixth order ($p \leq 0.05$). Despite the upstream-to-downstream, lower-to-higher water quality gradient, there was no compelling evidence from the use of state water quality standards and a nine-variable water quality index that water quality differentially affected fish assemblages.

The primary physical habitat gradient in the lower river was sedimentation. Upstream, embeddedness increased in shallow-water habitats ($p \leq 0.001$) and sediment depth increased in pools ($p \leq 0.001$). Stepwise multiple regression analysis indicated that variation in sediment level was explained primarily by the proportion of the watershed surface-mined ($R^2 = 0.75$).

Index of Biotic Integrity scores for fish assemblages were not correlated with sedimentation in shallow-water or in pool habitats ($p > 0.066$). Functional metrics, as opposed to taxonomic metrics, however, varied with sedimentation ($p \leq 0.002$). In shallow-water habitats, omnivore relative abundance increased and specialized insectivore abundance decreased in higher sedimented sites. Top carnivore abundance decreased in pools with higher sedimentation levels. Abundance of lithophilous benthic spawners, postulated as the most sediment-sensitive reproductive guild, was not consistently correlated with sedimentation.

Nine fish species were classified as sediment-intolerant and eight were classified as sediment-tolerant. The sediment-intolerant group was composed mostly of benthic insectivores. Habitat analyses indicated that sediment-intolerant species, as a category, utilized microhabitats with low sedimentation levels relative to the sediment-tolerant species group ($p \leq 0.0001$). Species not classified as either sediment-intolerant or sediment-tolerant utilized microhabitats intermediate in sedimentation level ($p \leq 0.0001$).

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the research reported herein. I hope that this document will contribute to the proper management and protection of their valuable and beautiful resource, the Powell River.

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Introduction

In response to the continued degradation of ecosystems and watersheds (Judy et al. 1984; Nehlsen et al. 1991; Williams et al. 1989, 1993), agencies such as the U.S. Fish and Wildlife Service are attempting to approach management more holistically- the "ecosystem approach" to management (USFWS 1994). The ecosystem approach is based on three guiding principles of conservation biology (Meffe and Carroll 1994):

1) "Evolution is the basic axiom that unites all of biology. (The evolutionary play.)";

2) "The ecological world is dynamic and largely nonequilibrial. (The ecological theater.)"; and

3) "The human presence must be included in conservation planning. (Humans are part of the play.)".

This approach requires that accurate, scientifically-based ecological information is applied within the context of socio-economics and institutions. It is the integration of ecology and comprehensive management systems (Crowe 1983; USFWS 1994).

It is critical in effective ecosystem management to address both ecological and

human-dimensions aspects. As fisheries scientists and managers, it is our responsibility to understand, as fully as practical, the ecological context of ecosystems under our jurisdiction. This enables, among other management tasks, creation of effective monitoring programs. Noss and Cooperrider (1994) and Noss (1995) proposed the following framework for monitoring program development:

1) describe important physical, chemical, and biological characteristics of and identify specific threats to an ecosystem;
2) if impairment of the ecosystem has occurred, describe the trends that led to the current condition;
3) if impairment has occurred, determine trends that would lead the system back to integrity; and
4) select indicators (system attributes) that will enable one to monitor the movement of the system along those trends.

This procedure is based on the concept of niche space (Hutchinson 1958), where physical, chemical, or biological alterations could change the niche space within which organisms exist. Changed niche spaces of populations could alter population dynamics, particularly survivorship and reproductive success, thereby restructuring the biotic assemblages from those states found originally (i.e., in unpolluted or relatively pristine environments) (Loeb 1994).

The Powell River in southwest Virginia is a species-rich system with limited biological information. The paucity of information can be attributed, in part, to the

relative isolation of the Powell River watershed. Land uses, particularly coal mining, have been inferred to have negatively impacted the biotic resources of the river (Ahlstedt and Brown 1979; Neves et al. 1980; TVA 1979, 1980; Ayers 1981; Dennis 1981, 1984; Wollitz 1985; Heffinger 1986). The alarm was sounded in the mid-1980's by local people concerned about the declining sport fishery and the river in general. The opportunity to investigate the river's aquatic resources presented itself, and a landscape-level project was undertaken. This project primarily was concerned with describing ecological characteristics of the Powell River, information that can be used to facilitate effective management (Figure 1). Noss and Cooperrider's (1994) framework was followed to determine and recommend meaningful indicators that can be used to monitor aquatic integrity of the Powell River. In addition to implementing the aforementioned protocol, several hypotheses important to monitoring, that also should have general applicability to aquatic ecology, were tested. These are discussed in detail in the appropriate chapters and appendices.

Chapter and Selected Appendix Prefaces

Because of the importance of including a biotic component, particularly fish, in a monitoring program (Karr 1994; Moyle 1994; Paller et al. 1996), this project focused on identifying factors affecting fish assemblage structure in the lower (fifth and sixth order) Powell River. In other words, Powell River system integrity was primarily assessed by

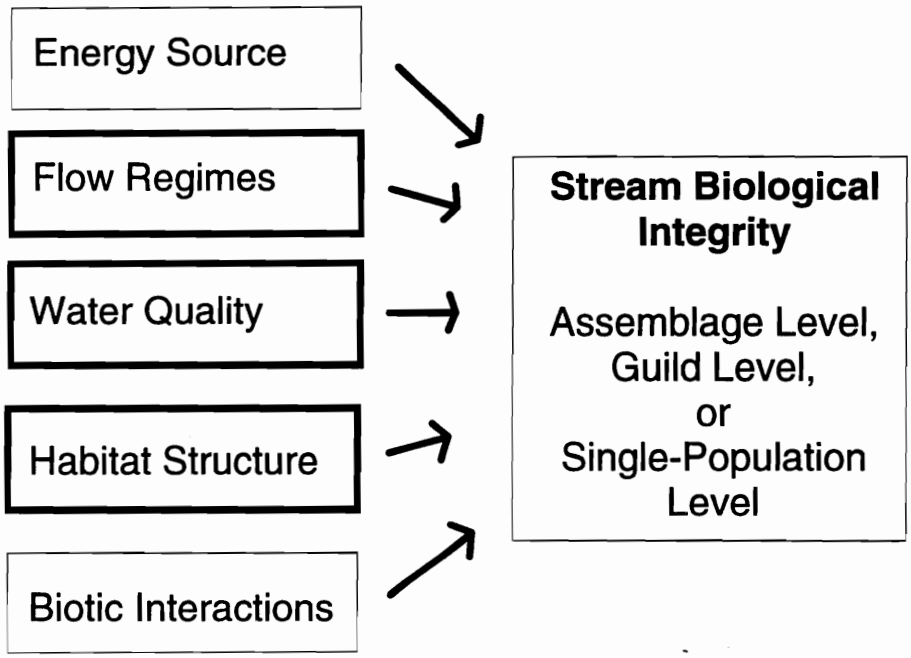

Figure 1. Factors that must be considered for effective stream fisheries management and monitoring (modified from Karr 1994). Highlighted boxes are factors directly addressed in this project.

evaluating fish assemblage integrity and determining natural and anthropogenic factors that appear to influence those fish assemblages. Tasks required in steps one and two of Noss and Cooperrider's (1994) protocol were addressed in Chapters One and Two. Specifically, the effects of basin-level land use activities on physico-chemical habitat characteristics (e.g., hydrology, water quality, and sedimentation) were assessed (Chapter One). Major threats (human-induced changes in the physico-chemical habitat) to the Powell River system were identified (Chapter One). In Chapter Two, effects of habitat characteristics on fish assemblage structure were determined, thereby linking basin-level land use activities to fish assemblage structure patterns and system integrity (Rabeni 1992). Protocol steps three and four were addressed in Chapters One, Two, and Three. Monitoring indicators that were identified and evaluated in Chapters One and Two were summarized in Chapter Three, and a subset was recommended for a monitoring program of the Powell River system in Virginia.

Finally, the relative efficiencies of several stream fish sampling techniques were compared in Appendix L. The influence of fright bias (Bovee 1982) on fish sampling efficiency was evaluated. Results were critical for the selection of an efficient sampling technique that provided more precise information on fish-microhabitat relationships which, in turn, was integral for evaluating potential fish species indicators (Bayley et al. 1989).

Research Importance

Noss (1995) stated that, for rational monitoring programs, there is great need to identify and substantiate indicators at several levels of ecological organization that correlate strongly with ecological integrity in many types of systems. The procedure followed in this project, especially the multivariate approaches, to identify indicators should be of interest to those concerned with monitoring lotic systems in general and the Powell River in particular. Although this is an applied project due to the concentration on implementing results to develop monitoring indicators for the Powell River, using Noss and Cooperrider's (1994) framework for indicator selection also addresses some important goals of community ecology; namely, to record patterns that occur in nature, describe the causal processes underlying these patterns, and to generalize, as far as practical, the explanations for the observed patterns (Wiens 1984).

Another contribution of this project is that it is one of the most thorough investigations on the effects of fine sediments (particle size < 2 mm) on warmwater stream fish assemblage taxonomic and functional attributes (Waters 1995). Study results compliment projects on small streams, particularly those of Berkman and Rabeni (1987) and Rabeni and Smale (1995). Sediment-tolerance classifications of particular fish species should be valuable information for managers working in any system where these species occur. In addition, results of this project support the hypothesis that the sediment-tolerance of any fish species can be determined by examining microhabitat use alone. For instance, species that may be difficult to evaluate from general assemblage-level

sampling, such as due to rarity (Bayley and Li 1992), can be classified by observing relative microhabitat sediment levels in which they occur (see Greenberg 1991).

Finally, using precise sampling techniques is integral to successful monitoring programs (Noss and Cooperrider 1994). Sampling disturbance and resulting fright-bias in fish has been postulated to negatively affect sampling efficiency in shallow-water habitats (Bovee 1982; Bain and Finn 1991). Results of my study reported in Appendix L support the fright-bias hypothesis and illustrate characteristics that sampling gear must have to reduce fright-bias and improve sampling efficiency of shallow-water fish assemblages in medium-sized streams.

Study River Description

The Powell River watershed, comprising 2429 km^2 (1800 km^2 in Virginia), is located in the upper Tennessee River basin (Figure 2). Arising in the Appalachian Plateau physiographic province (Wise County, Virginia), the Powell River flows southwest through the Valley and Ridge physiographic province of Virginia and Tennessee to its confluence with the Clinch River at Norris Reservoir. The Powell River is unregulated by dams except for that portion inundated by Norris Reservoir. Total river length is approximately 300 km (205 mi). Elevations range from 1152 m in Wise County to 299 m near the confluence. Basin relief, a ratio of the maximum basin length to the difference in elevation between the basin mouth and highest point on the drainage divide (Gordon et al.

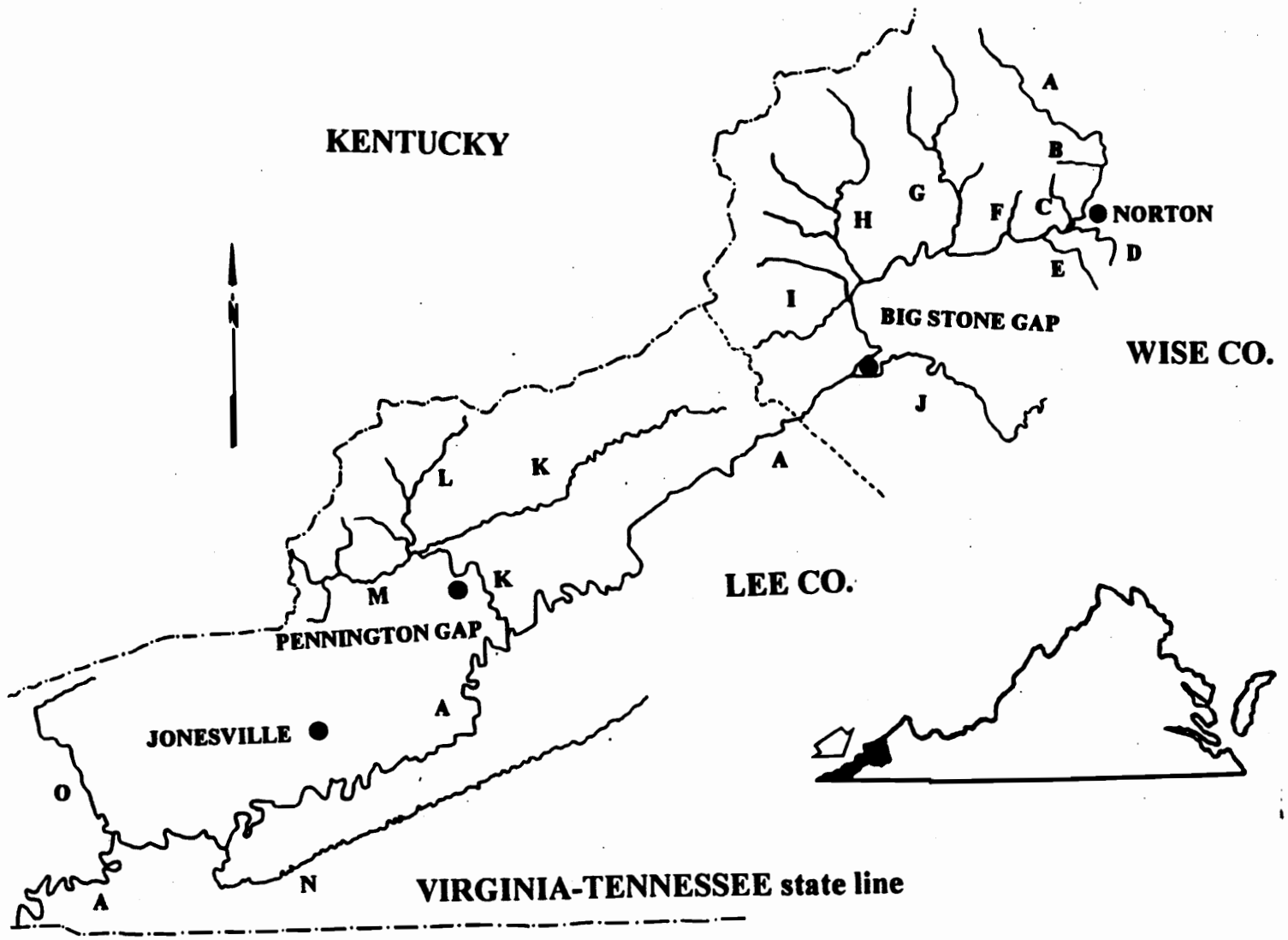

Figure 2. The Powell River in Virginia; key to symbols on following page.

Key to Figure 2 symbols.

A	=	Powell River
B	=	Bear Branch
C	=	Thacker Branch
D	=	Benges Branch
E	=	Carding Machine Branch
F	=	Black Creek
G	=	Roaring Fork
H	=	Callahan Creek
I	=	Looney Creek
J	=	South Fork Powell River
K	=	North Fork Powell River
L	=	Straight Creek
M	=	Stone Creek
N	=	Wallen Creek
O	=	Hardy Creek

1992), of the entire Virginia portion of the watershed is 792 m.

Geology and Topography

The Powell River in Virginia lies on the Cumberland overthrust block, a geologic feature that is divided into two physiographic provinces (Miller and Fuller 1954; Figure 3). The upper river (Powell River Mile [PRM] 208 - PRM 181) is located in the Appalachian Plateau province and drains approximately 290 km^2 (112 mi^2). Rock formations are sedimentary and formed in the Cambrian through the Pennsylvanian periods (Eby 1923). Naturally exposed strata are composed largely of alternating sandstone, siltstone, shale, and bituminous coal beds (Eby 1923; Dietrich 1970). There are limited clay and limestone strata as well (Eby 1923). The soils of this region are predominantly sandy loams, generally thin on the slopes, and highly prone to erosion when devegetated (Perry 1954).

The Appalachian plateau is a maturely dissected, irregular land surface with narrow, winding ridges and deep, steep-sided, narrow valleys. Ridges commonly rise 183 to 305 m (600 to 1000 ft) above the valley bottoms. Here, the Powell River is a consequent stream with a dendritic drainage pattern (Allison and Palmer 1980). Average gradient is 8.0 m/km (42.5 ft/mi). Basin relief of the Appalachian Plateau portion of the watershed (to PRM 180) is 701 m. Aquifer characteristics probably are similar to the nearby Russell Fork drainage, where Wyrick and Borchers (1981) found that groundwater storage and flow are primarily in fractured bedrock zones near the surface and in

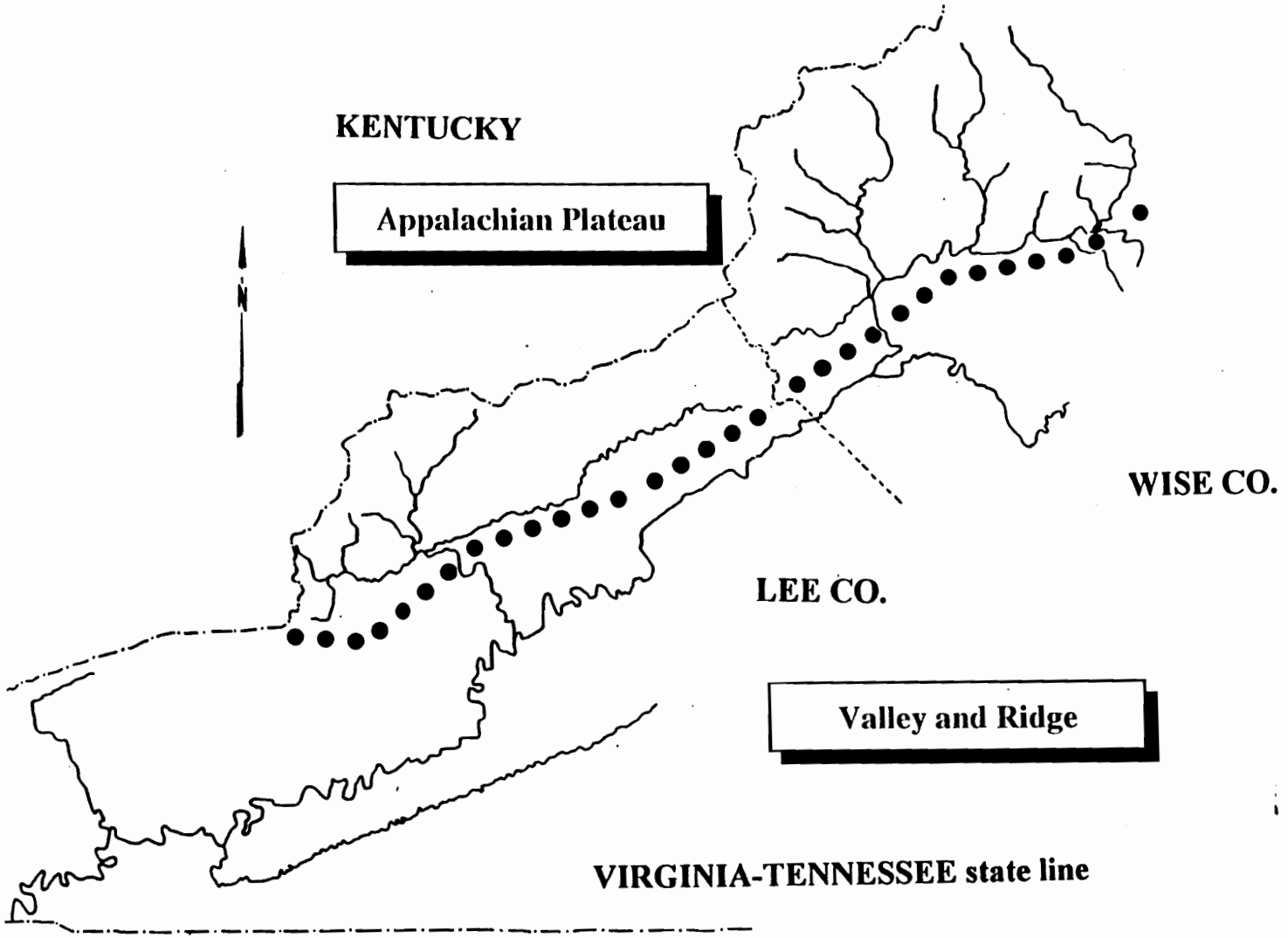

Figure 3: Physiographic provinces of the Powell River in Virginia.

alluvium-colluvium. Hence, the aquifer is shallow and has high rates of groundwater movement. These characteristics combined with the V-shaped valleys result in high flow variability (Beaumont 1975; Wyrick and Borchers 1981).

Near PRM 180 at Big Stone Gap, Virginia, the Powell River enters the Valley and Ridge physiographic province to its confluence with the Clinch River. Within this province, the Powell River mainstem lies on the Powell Valley anticline (Bates 1979). Rocks underlying the watershed are sedimentary and range in age from the Cambrian through the Devonian periods. Limestone and dolomite are predominant on the surface (Miller and Fuller 1954), except from PRM 180 - 168 where shale is a major constituent. Otherwise, sandstone is limited to the ridgetops, and shale is present in small outcrops on Cumberland Mountain. If vegetation is removed, soil erosion can be especially significant, particularly with dolomite- and shale-derived soils (Miller and Fuller 1954).

Valley and Ridge topography is characterized by long parallel valleys separated by sharp-crested ridges orientated northeast to southwest. The watershed is relatively broad and exhibits a trellis drainage pattern- a consequence of morphological control by resistant geologic features (Allison and Palmer 1980). Watershed area in Virginia of this province is approximately 1510 km^2 (582 mi^2). The drainage also contains numerous subterranean streams resulting from the extensive karst topography.

Using the classification system of Rosgen (1996), the Powell River is a "C" type stream in the Valley and Ridge Physiographic province. The mainstem meanders are located over limestone (occasionally on dolomite, Miller and Fuller 1954), often

truncated (Rosgen 1996), and typically exhibit vertical cliffs on the outside of the meanders with terraces and long, gentle slopes (usually 7 - 15% grade) on the inside. The river bank rises sharply to the lower terrace, making cattle access difficult in many sections. Consequently, the riparian zones on both sides of the river are broad and essentially intact (personal observation). In addition, the lower terraces are approximately 7.6 m (25 ft) above river level at average annual flow. Lower terraces are rarely flooded; the only known instances were in 1863, 1917, and 1977 (Miller and Fuller 1954; TVA 1978).

Land Use Types

Forests cover nearly 63.2%, surface mining 6.9%, pasture 21.4%, row crops 5.6% and urban areas occupy 2.1% of the total watershed area (Table 3, Chapter One). Land use changes are a result of the disparate geology that characterizes the basin. Within the Appalachian Plateau, primary land types are forest (76%) and surface coal mines (14.5%; Table 3). Small-scale agriculture is restricted to the valley bottoms. In contrast, pasture and forest are the primary land types in the Valley and Ridge. When considering the entire Virginia watershed, pasture (21.4%) and forest lands (63.3%) are by far the dominant land types (Table 3).

River Physical Characteristics

There are essentially three gradient phases of the Powell River (Figure 4):

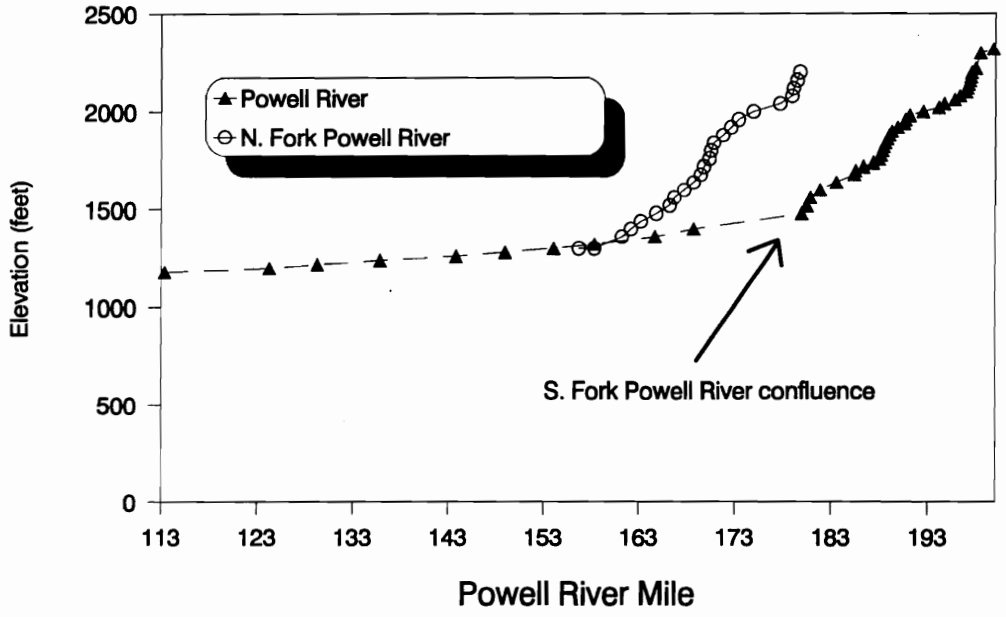

Figure 4. **Gradient profiles for the Powell River and North Fork Powell River. Information taken from U.S. Geological Survey 1:24,000 scale topographic maps.**

1) PRM 199.8 - 180.0 (near source to base of Big Stone Mountain; 8.1 m/km [42.5 ft/mi]);

2) PRM 180.0 - 156.8 (base of Big Stone Mountain to North Fork Powell River confluence; 1.2 m/km [6.2 ft/mi]); and

3) PRM 156.8 - 115.8 (North Fork Powell River confluence to the Virginia-Tennessee state line; 0.6 m/km [3.2 ft/mi].

The highest gradient is in the Appalachian Plateau, whereas the two lower gradient reaches are contained within the Valley and Ridge province. These gradient phases also correspond to stream order. As the Powell River flows out of the Appalachian Plateau, it is a fourth order stream. The South Fork Powell River joins the Powell at the base of Big Stone Mountain (in the town of Big Stone Gap, Virginia). From this point to the confluence with the North Fork, the Powell River is fifth order. The remaining downstream gradient reach in Virginia is sixth order.

The mainstem channel contains deep pools of variable length separated by principally pebble/cobble substrata (classification terms from Cummins 1962) dominated riffle and run complexes. On average, riffle and run complexes occur every eight to nine river widths. Average stream width / depth ratios in pools measured in June 1988 were 26.9 m / 1.06 m (PRM 172.2, 163.4, 158.3), 27.9 m / 1.29 m (PRM 153.4, 146.8, 144.6), and 35.2 m / 1.12 m (PRM 123.0, 119.3, 117.3). Width/depth ratios were highest downstream (mean = 31.8), indicating that width increases faster than depth downstream (Cummins 1994).

Flow Regime

At PRM 143.5 (near Jonesville, VA), the mean daily discharge from October, 1931 through December, 1987 was 529.5 cubic feet per second (CFS; 14.99 cubic meters per second [CMS]; HISARS 1984; Figure 5). The quarterly means were as follows: 1) January-March 1043.8 CFS (29.55 CMS; range: 30-27,000 CFS); 2) April-June 549.5 CFS (15.56 CMS; 27-35,000 CFS); 3) July-September 182.2 CFS (5.16 CMS; 18-6,180 CFS); and 4) October-December 460.0 CFS (13.0 CMS; 18-21,900 CFS). Watershed area at the Jonesville gauging station is 826.2 km^2 (319 mi^2).

Threats to the Powell River in Virginia

Pollution inputs may be classified broadly into point source and non-point source (NPS). If the Powell River is typical of the upper Tennessee River watershed, NPS pollution (primarily surface mining, construction, and agriculture) accounts for 80% of the pollution problem (Sagona 1990). In fact, the NPS potential is considered medium to high in the Virginia portion of the Powell watershed (Sagona 1990). Mining and agriculturally- related perturbations include increased loading of chemical contaminants into streams (VDSWC 1989; Zipper et al. 1992) and accelerated erosion from abandoned, reclaimed, and actively mined lands (Neves et al. 1980; Heffinger 1986), pastures, and croplands (Sagona 1990). Point-source inputs include periodic discharges of coal fines by coal preparation plants (TVA 1980), chemical spills (a 1983 hydraulic fluid spill at a coal preparation plant and a 1990 ammonium hydroxide spill at a deep mine site resulted in

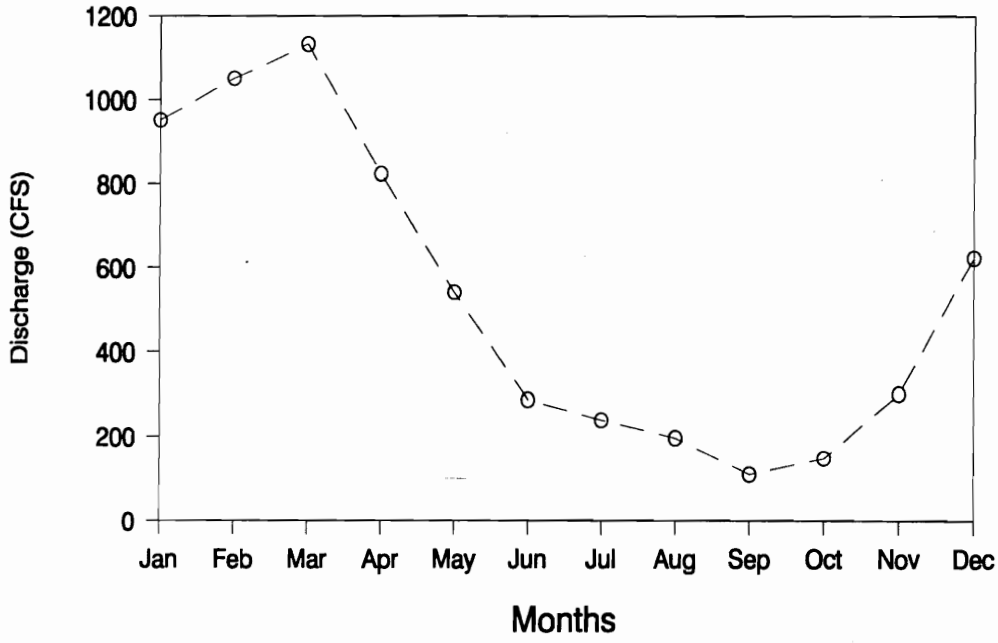

Figure 5. **Mean monthly discharge for the period 1931 through 1987 at Powell River Mile 143.5 near Jonesville, Virginia. Data for analysis were taken from HISARS (1984).**

significant fish kills in the upper Powell River [above PRM 180; VSWCB records]), and permitted discharges of sewage effluent (VSWCB 1988a; VDEQ 1996). Water quality monitoring on the Powell River drainage has indicated good water quality on the mainstem, with problem areas in some tributaries. The Tennessee Department of Health and Environment, presently the Tennessee Department of Environment and Conservation, evaluated 23 stations on rivers statewide with a Water Quality Index (TDHE-DWPC 1984). Sixteen water quality variables comprised their Water Quality Index. Water quality data from PRM 116 (Virginia state line) during 1981 and 1982 were scored. The index score for the Powell River at PRM 116 indicated very good water quality, the third best value in Tennessee. The report stated, however, that fecal coliforms, cadmium, mercury, lead, and suspended solids were elevated over background levels (TDHE-DWPC 1984). Also, it was suspected that "slug loading" or pulses of pollutants may occur and adversely affect aquatic biota in the Powell River. The TDHE-DWPC compared 1972 data with 1982 data and classified the Powell River water quality dynamics as extremely stable (TDHE-DWPC 1984). In addition, repeating the Water Quality Index analyses for PRM 116 using 1985, 1986, and 1987 data further substantiated this pattern of stability (TDHE-DWPC 1988). Cadmium once again, however, was a variable of concern (i.e., values exceeded fish and aquatic life or recreation state criteria in 25% or more of the observations).

At present, it appears that lower Powell River water quality in Tennessee continues to be stable (TDEC-DWPC 1994). The Tennessee Department of Environment

and Conservation has ranked the Tennessee portion of the Powell River as "fully supporting but threatened". This means that the fish and aquatic life use classification is supported, but potential watershed activities threaten the river's capability to fully maintain its use designation. Although water quality in the Tennessee portion of the river is classified as good, fecal coliforms, ammonium, and total-iron concentrations were found to frequently exceed state standards in a 4.5 yr study of over 40 water quality variables (Brede and Benham 1996). In addition, average annual sediment yields (184 tons/mi^2/yr) was nearly twice that of the adjacent Clinch River in Tennessee (Brede and Benham 1996).

The Virginia Department of Environmental Quality (VDEQ), formerly the Virginia State Water Control Board (VSWCB), maintains two water quality monitoring stations on the Powell River mainstem (PRM 180.8 and PRM 143.5) and two within the North Fork Powell River drainage (Straight Creek and the North Fork Powell River) (VSWCB 1990). The variables of concern at these stations identified by the VSWCB were elevated fecal coliforms and biological oxygen demand (BOD) (VSWCB 1988a, 1990). The VSWCB classified the mainstem and tributaries as "effluent limited", thereby implicating point-source pollution as the source of the fecal coliform and BOD standard violations. Fecal coliform concentrations are highest in the North Fork Powell River watershed, followed in descending order of fecal coliform concentrations at PRM 180.8 and PRM 143.5. No violations of the fecal coliform standard occurred at PRM 143.5 in 1989 (VSWCB 1990). Analyses of other variables indicate decreasing BOD and

suspended solids from PRM 180.8 to PRM 143.5 (VSWCB 1985). Dissolved oxygen concentrations and pH levels did not show a spatial trend. Only trace amounts of priority metals, pesticides, and herbicides were detected in the mainstem river (VSWCB 1985). Hence, the Powell River at PRM 143.5 has consistently exceeded state water quality standards (VSWCB 1985). In fact, the VDEQ (1996) currently classifies the 5th and 6th order Powell River as "fully supporting" aquatic life. The North Fork Powell River drainage, conversely, exhibits high BOD, suspended solids, and localized acid mine drainage (VSWCB 1985, 1988a; VDEQ 1996). What influence the North Fork Powell River has on the Powell River water quality below the confluence (PRM 156.6) to above PRM 143.5 is unknown.

Zipper et al. (1992) analyzed water quality data from the four VDEQ-maintained stations for temporal trends. The time period analyzed was from 1970 to 1989. The researchers found 1) that non-filterable residue (total suspended solids) declined and filterable residue (total dissolved solids) increased at all stations; 2) BOD declined at PRM 143.5 and Straight Creek; 3) fecal coliform concentrations declined at PRM 180.8 and PRM 143.5 (lowest median fecal coliform concentration at any of the four stations occurred at PRM 143.5); 4) dissolved oxygen increased at PRM 180.8; 5) pH increased in the two North Fork Powell River drainage stations; 6) total dissolved nitrogen increased at PRM 143.5; and 7) no trends at any station were evident for total phosphorous.

From the preceding analyses, it appears that water quality in the mainstem Powell River (below PRM 180.8) is good, and stable to improving. In addition, several local

people have indicated that large-scale discharge of coal fines, or "blackwater", has decreased greatly since the early 1980's (only two episodes, on single days in 1987 and 1995 were known to have occurred from June, 1987 to January 1996; Virginia Department of Environmental Quality records). Significant water quality impairments (e.g., depressed pH) appear to be localized in several tributaries (VSWCB 1985; VDEQ 1996). Hence, aside from the relatively localized problems, the Powell River below PRM 180 consistently exceeds water quality standards established by the VDEQ (VSWCB 1985; VDEQ 1996).

Since chemical water quality in the lower river is classified as good by the VDEQ, the greatest perturbation to the lower river may be increased suspended solids and sedimentation from nonpoint sources (TVA 1980; Sagona 1990). In particular, Branson and Batch (1971) and Matter and Ney (1981) postulated that suspended solids and sedimentation from mining activities were the primary perturbations to the aquatic systems they studied in Kentucky and Virginia, respectively. The U.S. Environmental Protection Agency (1976) also regarded sedimentation as the primary source of water pollution from surface coal mining. If this is the situation in the Powell River, surface mining activities and resultant sedimentation would be the major source of perturbation to the Powell River in Virginia. The hypothesis of mining activity as the primary factor in sedimentation is further supported by the high erosion rates characteristic of mined lands as compared to other common land use types. The estimated total soil loss from crops, poor pasture, and construction in the Virginia portion of the Powell river is

269,352, 60,732, and 245,000 tons/yr, respectively (total soil loss = 575,084 tons/yr; Sagona 1990). In contrast, mined land erosion is estimated at 889,224 tons/yr.

If surface mining is primarily responsible for sediment input to the Powell River, and given that surface mining is restricted to the upper Powell (above PRM 180) and North Fork Powell rivers, increased sediment deposition should be evident as one proceeds upstream. Indeed, biological surveys have noted that sites above PRM 140 (near Jonesville) are heavily altered by silt deposition (Ahlstedt and Brown 1979; Neves et al 1980; Dennis 1981; TVA 1986). This condition may have been the primary cause of mussel elimination above PRM 165 (Dennis 1981), although elevated metal concentrations in the water column may be an additional problem (McCann 1993). Iron, manganese, and sulfates often have been in violation of U.S. Environmental Protection Agency (USEPA) standards for those constituents (Larson 1985).

Biotic Characteristics

Despite these and other perturbations, the Powell River remains biologically diverse. Eighty-five fish species, of which two are federally threatened, three introduced, and three extirpated, are known to have occurred in the Powell River system (Lee et al. 1980; Etnier and Starnes 1993; Jenkins and Burkhead 1994). On a regional perspective, the Powell and the rest of the Tennessee River system harbor the richest freshwater fish fauna in North America (Lee et al. 1980). In addition to the variety of ichthyofauna, 42 mussel species have been found in the Powell River (Ortmann 1918), including five that

are federally endangered (TVA 1986). This fauna is one of the most speciose in North America as well. Other assets of the Powell River, such as sport fishing and other aquatic recreational activities (e.g., canoeing and swimming), also are of substantial importance to the local populace.

Sampling Site Locations

Twenty-one sites within the lower Powell River were sampled for physical habitat and fish assemblage characteristics during the 1988-1990 period (Table 1; Figure 6). Only two sites, PRM 144.6 and 117.3, were sampled in all three years. This strategy follows the recommendation of Cochran (1977) that the use of different sites over the sampling period is a rational approach to detect impacts (or threats) to a system. Water quality stations, distributed over the Powell River system, are described in Appendix A.

Table 1. Fish and habitat sampling sites on the Powell River for 1988, 1989, and 1990. PRM denotes Powell River mile location. Note that the South Fork Powell confluence is at PRM 178.1 and the North Fork Powell confluence is at PRM 156.6.

PRM	Name	Sampled		
		1988	1989	1990
Fifth Order				
174.4	Olinger	x		
171.8	Olinger Upper		x	
170.3	Olinger Mid		x	
168.9	Olinger Lower		x	
163.4	Swimming Hole	x		
158.3	Rocky Road	x		
Sixth Order				
156.0	Island Below Confluence		x	x
153.4	Schaffer Ford (Trash Island)	x		
153.9	Collier Mill		x	x
149.3	Pond View		x	x
146.8	Cheek Springs	x		
144.6	Poteet Ford	x	x	x
141.3	Bush			x
136.7	Batie		x	x
127.2	White Shoals		x	x
126.2	Tyler Bend		x	
123.0	Snodgrass	x		x
120.4	Beech Grove (833 Bridge)		x	x
119.3	Yellow Creek	x		x
117.4	Fletcher Cliff			x
117.3	Fletcher Ford	x	x	x

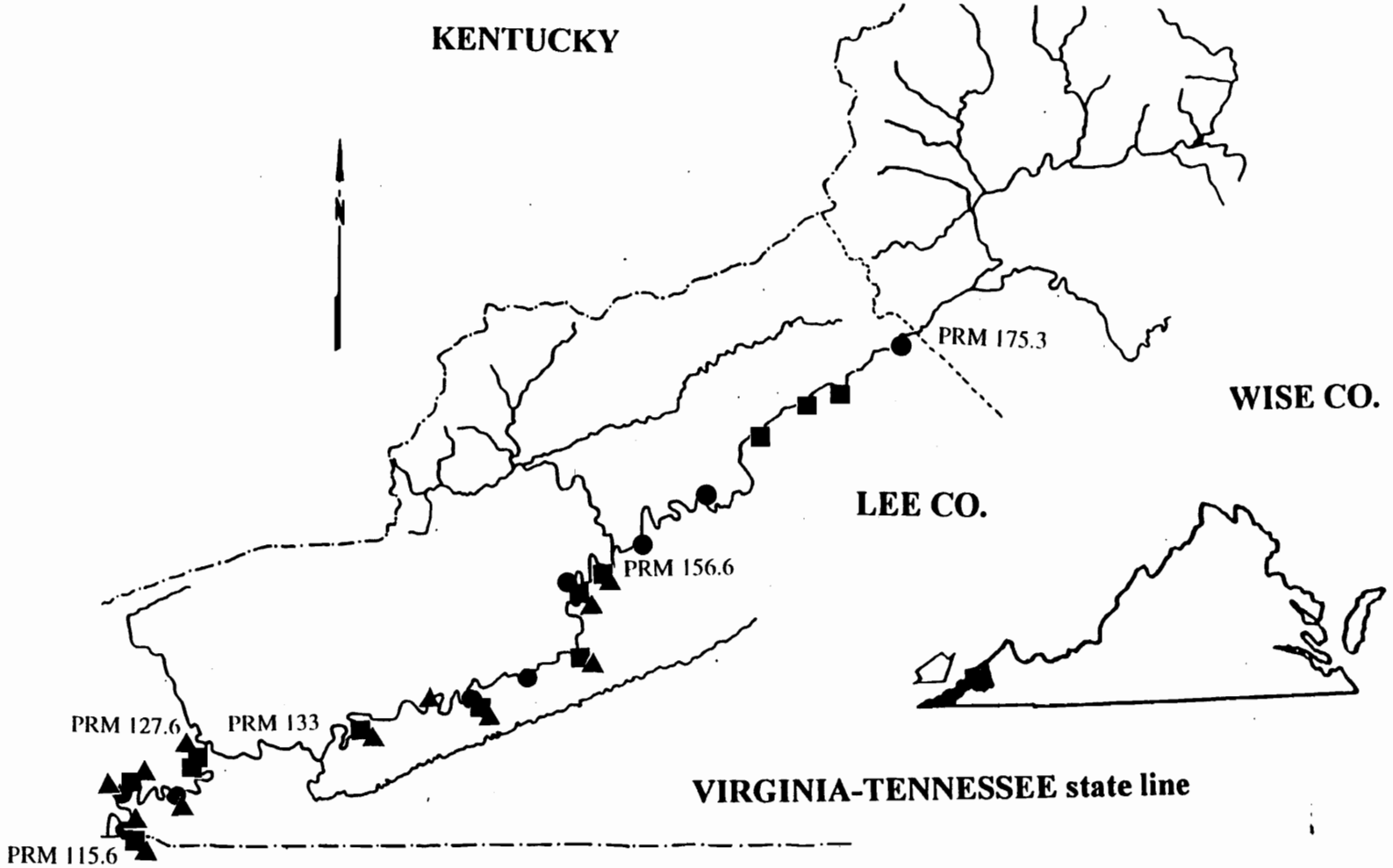

Figure 6: Fish and habitat sampling sites on the Powell River for 1988 (●), 1989 (■), and 1990 (▲). PRM denotes Powell River mile location.

Chapter One

Watershed land uses and the effects of coal mining on the disturbance regime, water quality, and river substratum.

Introduction

Within Noss and Cooperrider's (1994) framework for developing monitoring indicators, it is important to characterize chemical, physical, and biological attributes of lotic systems as the hydrologic or disturbance regime (Poff and Ward 1989) and the physico-chemical structure of a stream (particularly longitudinal gradients that might cause zonation of fish assemblages) (Bayley and Li 1992). It also is important to identify and characterize threats and sources of threats to the system.

Anthropogenic effects on an aquatic system can be generally classified into three components: 1) physical; 2) chemical; and 3) biological (Li and Moyle 1993; Loeb 1994). Physical alterations include changes in hydrology and in the substratum (e.g., sedimentation). Chemical stresses correspond to water quality perturbations. Biological alterations include growth or recruitment overfishing and exotic species introductions. When a perturbation or stressor changes the physico-chemical habitat or biotic template, the niche volume of a population may be changed (Pianka 1981; Wootton 1990; Loeb 1994). Volume change may be an increase or a decrease, depending on organism traits

(Southwood 1988). These niche changes can lead to changes in assemblage structure that can be observed in a monitoring program (Loeb and Spacie 1994).

Disturbance Regime

Much research and debate has addressed the question of whether physical or biotic factors primarily organize stream biotic assemblages (Connell 1975; Moyle and Vondracek 1985; Heins and Matthews 1987; Schoener 1987; Power et al. 1988; Resh et al. 1988; Bayley and Li 1992; Allan 1995). The harsh-benign model of community structure (Allan 1995) is a subset of Schoener's (1987) physical vs. biological axis. It is based on the postulate that, in the absence of strong environmental variation, biotic interactions organize and maintain community structure in a equilibrial state (Peckarsky 1983; Meffe 1984; Ross et al. 1985). Environmental stochasticity may prevent biotic interactions from controlling assemblage structure (Grossman et al. 1982), resulting in temporally variable, less predictable structure attributes. These concepts are important for monitoring lotic fish assemblages because assemblages exhibiting equilibrium in the absence of pollution or other human-induced stresses are more amenable to monitoring (Matthews 1990; Moyle 1994; Stewart and Loar 1994).

Flow regime is regarded as a major disturbance force in lotic systems (Meffe 1984; Poff and Allan 1995). Increased flow variability has been linked to increased variability in fish abundances (nine studies summarized in Bayley and Li 1992) and may be a major organization factor for fish assemblages (Horwitz 1978; Bain et al. 1988;

Grossman et al. 1990; Poff and Allan 1995). Poff and Ward (1989) therefore attempted to develop a basis for describing the relative degree of environmental stochasticity in continental United States streams by using stream hydrologic characteristics. Their analyses derived nine stream hydrologic classifications. For each classification, the authors hypothesized whether disturbance ("abiotic") or biotic factors primarily controlled assemblage organization. This classification system may provide a basis, from a monitoring perspective, to evaluate whether land use activities have significantly altered the flow or disturbance regime in streams.

Stream Physico-chemical Structure

Research has addressed another aspect of Schoener's (1987) physical versus biological axis; namely, to find stream macrohabitat factors (large environmental gradients) that affect fish assemblage structure (Matthews 1985; Schlosser 1990; Bayley and Li 1992). The river continuum concept (Vannote et al. 1980) and the flood pulse concept (Junk et al. 1989) have synthesized into a spatial framework many of the factors considered important for organizing lotic assemblages. Environmental gradients investigated include geomorphology (Nelson et al. 1992), physiographic province (Starnes and Etnier 1986), stream order (Matthews 1986; Paller 1994), habitat heterogeneity (Gorman and Karr 1978; Schlosser 1982; McClendon and Rabeni 1987; Fausch et al. 1988; Gorman 1988; Lyons et al. 1988; Meffe and Sheldon 1988; Pearsons et al. 1992), temperature (Rahel and Hubert 1991), and water quality (Alabaster and

Lloyd 1980; Hendricks et al. 1980; Hocutt and Wiley 1986; Hughes and Gammon 1987; Van Hassel et al. 1988; Taylor et al. 1993). Certainly, any comparative community ecology study must account for these factors to enable rational interpretation of observed assemblage patterns.

Threats and Sources of Threats to Lotic Systems

It also is important to identify and characterize threats and sources of threats to the lotic system (Noss and Cooperrider 1994). Hydrologic regime, water quality, and physical habitat, particularly sedimentation levels, are affected by watershed land uses (Meehan 1991; Naiman 1992; Stroud 1992). For effective stream management, watershed area that is comprised of various land use types should be quantified and related to hydrologic and instream physico-chemical habitat patterns (Rabeni 1992; Wesche 1993). These analyses can provide the context to interpret observed fish assemblage patterns and to develop monitoring indicators for the stream system in question.

Primary Questions Addressed

The objective of this study was to describe important physical, chemical, and biological characteristics of and identify specific threats to the Powell River system in Virginia (step one of monitoring indicator identification procedure described in dissertation introduction; Noss and Cooperrider 1994). A trend evaluation of selected

water quality variables also was performed (indicator identification step two). These analyses can provide the context to interpret observed fish assemblage patterns and to help develop monitoring indicators for the Powell River. This project also used the theoretical framework of Poff and Ward (1989) to describe the hydrologic characteristics of the sixth order Powell River, to classifiy the disturbance regime, and to evaluate whether coal surface-mining has changed the disturbance regime classification.

Specific questions addressed included:

1) What portion of the Powell River watershed in Virginia consisted of each major land-use type?

2) What is the hydrological classification of the sixth order Powell River in Virginia? Specifically, where does the Powell River disturbance regime occur on the benign-harsh spectrum, and what is the theoretical prediction of regularity of fish assemblage structure through time?

3) Has coal mining affected the flow regime (low and moderate flows, daily flow variance) and, if so, were the changes severe enough to cause significant changes in the disturbance regime with potential affects in fish assemblage structure? In addition, if coal mining has affected the flow regime, were changes due to alterations in groundwater storage capacity or to changes in watershed discharge yield?

4) Has water quality impairment occurred in the Powell River system? If so, what were the spatial and temporal patterns and what was the primary source of

impairment? Did the fifth order Powell River differ from the sixth order reach in regard to water quality and was observed water quality deterioration extreme enough to have caused changes in fish assemblages? Finally, which water quality variables would serve as viable indicators of impairment from specific land uses.

5) Were there longitudinal changes in physical habitat, including sedimentation, within the lower Powell River? What was the primary source of sedimentation: agriculture, coal mining, or urbanization?

6) What was the primary physico-chemical factor in the lower Powell River that could potentially alter fish assemblage structure?

7) Was there a physico-chemical basis that might cause longitudinal zonation of fish assemblages?

Methods

Land Uses

Land use percentages were determined for sub-watersheds of 19 study sites as well as the entire Virginia portion of the watershed from the "Land Use Series" maps contained within the Southwest Virginia 208 Plan Map Volume (Southwest Virginia 208 Planning Agency 1977). Land use information was obtained by remote sensing (aerial photography) and plotted on standard USGS 7.5 minute quadrangle base maps at a scale of 1:24,000. These maps were reduced to a scale of approximately 1:48,000 for printing

on 11" x 14" paper. Land use categories used were as follows:

Urban\Built up: A broad classification including institutional, commercial, residential, transportation, communications, utilities, and mixed urban land uses. This category was not pre-defined as a particular number of structural units per acre. Rather, "...it is mapped as a reasonably significant number of structures that appear to possess a community relationship and a common focal area" (Southwest Virginia 208 Planning Agency 1977).

Cropland: Agricultural areas recently plowed or cropped. Includes annual field, row, and fodder crops, as well as land temporarily in fallow.

Pasture: An area that contains herbaceous cover and less than 10% tree crown closure. Primarily used for livestock grazing, and often characterized by livestock watering points, paths, and fence lines.

Other Agriculture: Areas devoted to orchards, poultry production, and high intensity livestock feeding operations.

Forest: Areas that have not been surface-mined and have at least 10% crown closure of mature trees.

Surface-mined Land: Active, reclaimed, or abandoned surface mines. Vegetation cover varies from little or none to significant. Included in this category are spoil piles (from deep mines), high walls, overburden that has been cast downslope, mine dumps, slurry ponds, and coal processing facilities (e.g., conveyors, tipples).

Quarries and Gravel Pits: Open-pit mines that extract sand, gravel, or rock.

Undetermined Barren Land: Sparsely vegetated areas from unknown causes.

Nineteen sites on the river were selected from PRM 174.4 to PRM 117.3. These sites were sampled in 1988, 1989, and 1990 for one or more of the following attributes: physical habitat, chemical habitat (water quality), and fish assemblage structure. The topographic watershed for each site was delineated on the 208 Plan land use maps. Watersheds were delineated by topographic divides (Gordon et al. 1992).

For each watershed, a dot grid (18 dots/cm^2) was superimposed on the land use maps and the number of dots contained in each defined land use type was recorded (Gordon et al. 1992). For each watershed, the number of dots in each land use category was then divided by the grand total of dots in all land type categories (i.e., total dots in the watershed). This result was multiplied by 100, converting results to percent land use type in each watershed.

Disturbance Regime

Flow Regime Classification

Hydrologic data (average daily discharge) for the period 1932 to 1988 were obtained from the United States Geological Survey gaging station at PRM 143.5 (near Jonesville, Virginia) (HISARS 1984). Watershed area for this gaging station is 826.2 km^2 (319 mi^2). This gaging station is located approximately midway between the uppermost (PRM 174.4) and lowermost (PRM 117.3) habitat and fish sampling sites (Chapters 1 and 2) and is in the sixth order reach of the Powell River. Hence, I believe that the hydrologic

classification derived from data obtained from this gaging station was at least an accurate description of the sixth order flow regime.

Since the framework used for classification was developed by Poff and Ward (1989), refer to that publication and to Colwell (1974) for a complete description of methods. Three categories of derived statistics were used to describe the flow regime: 1) overall flow variability, 2) patterns of the flood regime, and 3) extent of intermittent conditions (Table 2). Programs on DBASE IV were written to assist in the derivation of several variables (author: Dan Everson, U.S. Fish & Wildlife Service, National Conservation Training Center, Kearneysville, West Virginia).

To derive the statistics that describe flood regime pattern, the minimum discharge (threshold value) that causes a flood disturbance must first be determined. Floods are defined as discharges equal to or exceeding this threshold value. Bankfull discharge is assumed to be a channel-modifying event during which substratum movement is significant (Leopold et al. 1964; Richards 1982). Bankfull discharge is considered to be the threshold value for a flood disturbance event. Poff and Ward (1989) assumed a return period of two years for the bankfull discharge. A two year return period corresponds to the 50% exceedence probability for stream discharge. To derive the flow value that corresponds to the 50% exceedence probability, the annual 24-h peak flow for each year of record (n=57) was log transformed (ln [x+1]), ranked, and a simple linear regression performed (using MINITAB, Minitab Inc., State College, Pennsylvania) with transformed annual 24-h peak flows (dependent variable) versus ranks (1-57; independent variable).

Table 2. Eleven variables in three categories used to describe flow regime of the habitat and fish study section of the Powell River in Virginia. Data taken from the U.S. Geological Survey gaging station at PRM 143.5.

Variable	Definition
Overall Flow Variability	
ANNCV	Mean annual coefficient of variation. A coefficient of variation (CV) for average daily flow was calculated for each of the 57 years of record. The ANNCV is the average of the annual CV's (n=57). The ANNCV assesses overall variability but it is insensitive to temporal pattern.
PREDQ	Measures the predictability of the overall flow variation. PREDQ ranges in value from - (no predictability) to 1 (complete predictability). PREDQ is composed of two additive components: constancy (C), a measure of temporal stability, and contingency (M), a measure of periodicity. If the average daily flows of a particular stream are relatively constant, then PREDQ would be near 1, composed mostly of the constancy component. Conversely, a stream with high flow variability but with a strong temporal pattern would have a high PREDQ as well- composed mostly of the contingency component.
C/P	Proportion of the total predictability (PREDQ) composed of the constancy component.
Pattern of Flood Regime	
FLODFREQ	Flood frequency (mean number of floods per year).
FLODINT	Median interval (d) between floods.
FLODDUR	Mean duration (d) of floods.

Table 2. (continued)

Variable	Definition
Pattern of Flood Regime (continued)	
FLOD60D	Index of flood predictability. Maximum proportion of total number of floods over record that occur in any common 60-d period over all years in the period of record.
FLODFREE	Index of flood predictability. Maximum number of 365 d common to all years during which floods have not occurred.
FLODTIME	Median day among all days of the water year (beginning on October 1) on which floods have occurred over the period of record.
Extent of Intermittency	
ZERODAY	Average annual number of zero flow days.
LOWFLOW	Average over all years of the annual 24-h low flow value divided by the grand mean flow.

The resulting regression equation was used to determine the threshold flood value by using the median rank (29; 50% exceedence probability) for the independent variable.

After the values for the 11 variables are derived, Powell River values were compared to values of stream types identified by Poff and Ward (1989). Specifically, the two stream types that occur in the Powell River area, "Perennial Runoff" and "Mesic Groundwater", were used to place the Powell River in Poff and Ward's framework of stream environmental variability. Perennial Runoff is characterized by perennial flow and low flow predictability, low flood predictability, and low flood frequency. Abiotic (physical disturbance) processes are postulated to primarily control assemblage structure. Mesic Groundwater is characterized by perennial flow and by high flow predictability, low flood predictability, and low flood frequency. Biotic processes are hypothesized to be the primary controller of assemblage structure. Relative to the entire environmental variability framework of Poff and Ward (1989), the flow regimes of the two stream types of interest differ only in overall flow variability and their postulated organizational factors for assemblage structure.

Effects of Coal Mining on the Disturbance Regime

Effects of coal mining on the disturbance regime (hydrologic characteristics) of the Powell River midreach were evaluated using data (average daily discharge) obtained from the U.S.G.S.- maintained gaging station at PRM 143.5. Surface coal mining in the Powell River Valley began in 1945 and peaked in the mid-1980's (Figure 7). Deep

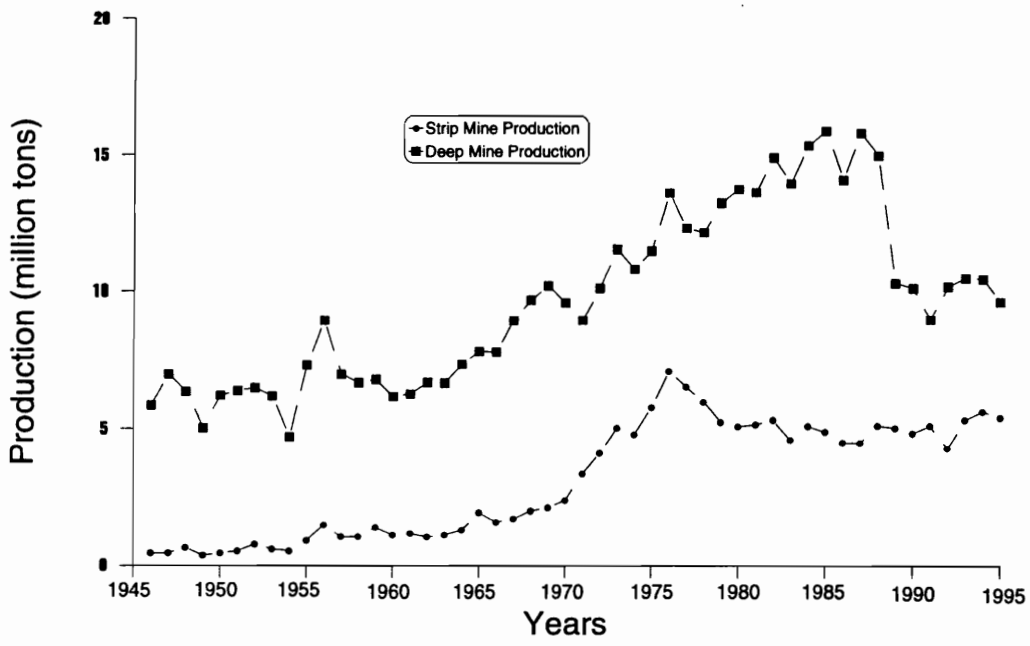

Figure 7. **Coal production by surface and deep mines in Lee and Wise counties, Virginia from 1946 to 1995 (Hibbard and Clutter 1990; Karmis et al. 1996).**

mining generally increased during this period as well. Low flow characteristics of the Powell River during pre-surface mining (1932-1945) and surface mining periods (1946-1988) were evaluated by low-flow frequency curves, which were developed for each period through the DURFREAK computer program (Water Resources Center, Virginia Polytechnic Institute and State University, Blacksburg). This program first determines the average minimum seven-consecutive-day flow (7Q) for each year of a period. Within a period, the 7Q values were ranked (smallest to largest) and a non-exceedence probability was calculated for each 7Q. A non-exceedence probability for a 7Q is the probability that the average minimum seven consecutive -day flow for any year will not be lower than the given 7Q. The formula used to calculate non-exceedence probability for each 7Q is (Hayes 1990):

$$P = \frac{1}{T} = \frac{m}{n+1}$$

where,

P = non-exceedence probability in any one year;
T = recurrence interval (years);
n = number of years in the period; and
m = rank position of the 7Q.

Recurrence interval is the average interval of time (in years) within which the magnitude of a flow will be exceeded once. For low flow, the recurrence interval is the average

interval of time (in years) between occurrences of a 7Q less than a given magnitude.

Finally, DURFREAK fit the annual low flow coordinates (7Q's and the corresponding non-exceedence probabilities) with the Log-Pearson Type III frequency distribution (Rao 1980). This frequency distribution fits low flow characteristics of Virginia streams well and aids in increasing precision of low flow non-exceedence probability estimations (Hayes 1990). The resulting low flow frequency curves for the two periods were plotted and inspected for any differences.

Another method for comparing low flow frequency curves of the pre-surface mining and surface mining periods is to determine and compare the $7Q_2$'s and $7Q_{10}$'s for each period. Commonly used in Virginia for describing low-flow characteristics (Hayes 1990), these 7Q's are the annual minimum average seven-consecutive-day low-flow discharges having a 2 or 10 year recurrence interval (50% or 10% non-exceedence probability), respectively. The $7Q_2$ is considered more accurate since it occurs more often in a long period of record (Hayes 1990).

Low-flow frequency curve comparisons between the two periods could be confounded if precipitation levels differ between the two periods. Hence, two approaches were taken:

1) total precipitation per year for the two periods was evaluated for differences using a Wilcoxon Rank Sum test (Hollander and Wolfe 1973); and

2) hydrologic data were obtained from a nearby stream of similar size to the

Powell River, the North Fork Holston River (U.S.G.S. gaging station at Saltville, Virginia); the North Fork Holston River watershed has never been mined for coal.

Yearly precipitation records for the Powell watershed (Pennington Gap station #44066626) were obtained from the National Oceanic and Atmospheric Administration in Asheville, North Carolina. Low flow frequency curves were derived in identical fashion for the North Fork Holston River flow regime.

Moderate flow characteristics for the Powell River pre-mining and mining periods were evaluated by flow duration curves. A flow duration curve is a cumulative frequency curve that indicates the percentage of time specified discharges were equalled or exceeded during a given time period (Riggs 1972). Flow duration coordinates were derived by programs written in QUATTRO (Borland International, Scotts Valley, California). Flow duration curves for each period were plotted and inspected visually. For a more complete analysis, flow duration curves were constructed for the North Fork Holston River as well.

Changes to flow variability from surface coal mining were determined by calculating the mean annual coefficient of variation using average daily flows for the pre-surface mining and surface mining periods (Zar 1984).

Potential mechanisms through which mining affects the flow regime were examined by evaluating changes to groundwater storage capacity and phreatic watershed integrity (Gordon et al. 1992). Three base flow recession curves were calculated for each

period (Riggs 1972; Larson and Powell 1986). During base-flow periods, flow recessions (i.e., base-flow recessions) approximate groundwater storage decay in a watershed. Comparing base-flow recession curves between the pre-surface mining period (PSMP) and the surface mining period (SMP) will indicate whether groundwater storage capacity has changed and, if so, presumably by surface mining. Hydrographs (January 1 - March 31) were generated for several years within each period. Winter recession curves were used to minimize evapo-transpiration effects. Two nearly identical peak flows and their subsequent recessions (one during each period) were selected and a straight line was drawn through one log cycle coincident with the slope of the recession. One log cycle was used to standardize comparisons by using identical discharge ranges. The length of the straight line (base flow recession curve) is the time required for one log cycle of discharge. Three comparisons were made of base-flow recession curve pairs.

Potential change to phreatic watershed integrity of the Powell River was investigated by comparing the water budgets of the pre-surface mining and surface mining periods. A watershed water budget can be described by the equation:

$$Q = P - E$$

where,

Q = annual stream discharge (ft^3/yr);
P = annual watershed precipitation (ft^3/yr); and
E = annual watershed evapo-transpiration (ft^3/yr).

The E term also may reflect changes in the storage capacity of the watershed. Hobba

(1981) found that deep coal mining can result in loss of storage capacity by causing groundwater leakage to the adjacent watershed.

An investigation of whether changes occurred to the annual discharge yield of the Powell River was conducted by determining the percent total discharge for each period (this standardizes discharge by total precipitation). The Q term was determined for each of 10 yr in both periods (1932-1941 [PSMP] and 1979-1988 [SMP]) by the water budget equation. Percent total discharge was determined by:

$$\% \text{ Total Discharge} = Q \div P$$

Differences in percent total discharge were evaluated through use of the Wilcoxon Rank Sums Test (Hollander and Wolfe 1973).

Stream Physico-chemical Structure

Water Quality

The Powell River (PR) drainage was classified into five categories: mined-land tributaries draining the Cumberland Plateau Physiographic province (MTRIBCP; includes the first - fourth order PR mainstem), unmined-land tributaries of the Cumberland Plateau (UMTRIBCP), unmined-land tributaries of the Valley and Ridge Physiographic province (UMTRIBVR), fifth order PR mainstem, and sixth order PR mainstem. Sixteen variables were measured to characterize water quality of the five categories. Standard methods

were used to determine biological oxygen demand (BOD), total solids, total dissolved solids, and total suspended solids (APHA 1975). A specific conductivity meter referenced to 25 °C was used to record water conductivity. Dissolved oxygen and pH were determined by YSI (Yellow Springs Instruments) meters as well. Alkalinity, calcium, hardness, iron, manganese, nitrates, total phosphorus, sulphate, and turbidity were measured with a HACH DR-EL 2000 field kit.

Water quality station locations and classifications are listed in Appendix A. Sample dates are classified as spate (6/05/89, 6/06/89, 7/31/89, 8/21/89, 8/29/89, 9/25/89, 9/26/89, 9/27/89, 2/05/90, 8/14/90) or non-spate (8/18/89, 9/07/89, 9/10/89, 9/21/89, 9/22/89, 11/15/89, 12/14/89, 2/27/90, 3/22/90, 5/13/90, 11/20/90, 12/06/90). Conditions at PRM 138.3 were used to qualitatively classify the PR as in spate or non-spate status. Spate average discharge was 862 CFS for all dates and non-spate average discharge was 370 CFS. Note that not every water quality variable was measured at each site or at each date.

Differences in water quality variables among stream categories were tested by using Kruskal-Wallis tests followed, if significant, by Mann-Whitney U tests to separate medians (MINITAB, Inc. State College, Pennsylvania). Mann-Whitney U tests only were used to evaluate water quality differences between segments of the Powell River draining inactive and active mined-lands. Spate and non-spate data were combined for all analyses. Differences between spate and non-spate flows within the MTRIBCP, fifth order PR, and sixth order PR also were assessed with Mann-Whitney U tests.

Water quality variable relationships were assessed by the Spearman's rank correlation test (MINITAB and SAS [1989]). Stepwise multiple linear regressions (MINITAB) were performed on $\log_{10}(x + 1)$ transformed data. A one-time longitudinal analysis of aluminum, arsenic, cadmium, chromium, copper, and zinc was performed by the Soil Science Lab at Virginia Polytechnic Institute & State University, Blacksburg using atomic absorption spectrophotometry.

Four of the stations (PRM 171.2, 153.4, 138.3, and 123.8) were sampled more intensively to determine if a water quality gradient occurred in the fifth (PRM 171.2) and sixth (PRM 153.4, 138.3, and 123.8) order PR. Principal components analyses using a covariance matrix (MINITAB) were employed to compare the water quality of these four stations in the context of 10 water quality variables during non-spate dates (12/14/89, 2/27/90, 3/22/90, 5/13/90). Variables analyzed were alkalinity, specific conductivity, hardness, nitrates, pH, total phosphorus, sulphate, total solids, turbidity, and dissolved oxygen saturation. All values were $\log_{10}(x + 1)$ transformed to prevent scaling effects (Green 1979).

To screen the fifth and sixth order PR water quality for potential biological effects, water quality variable measurements were compared to Virginia state water quality standards (VSWCB 1988b, 1989). In addition, the National Sanitation Foundation Water Quality Index, NSF-WQI_m, was used at stations PRM 171.2 and 138.3. The NSF-WQI_m uses nine variables: percent dissolved oxygen saturation, fecal coliforms, pH, BOD_5, nitrate, total phosphorus, temperature (degree of thermal pollution), turbidity,

and total solids. Data on fecal coliforms and BOD_5 for nearby state water quality monitoring stations (PRM 180.8 and 143.5) were obtained from the Virginia State Water Control Board (VSWCB) and used in the analysis. Dates analyzed were 9/10/89, 11/15/89, 2/27/90, 3/22/90, and 5/13/90 (all non-spate dates). The NSF-WQI_m is defined as the exponentially weighted product of individual sub-index values obtained for each parameter from standard curves of individual quality rating functions (Ott 1978):

$$NSF-WQI_m = \prod_{i=1}^{9} I_i^{w_i}$$

where:

I_i = the sub-index value of the water quality parameter i, and

w_i = the weighting exponent for the i^{th} parameter.

The nine relative weighting factors (w_i) are dissolved oxygen (0.17), fecal coliforms (0.15), pH (0.12), biological oxygen demand- five day (0.10), nitrates (0.10), phosphates (0.10), temperature (0.10), turbidity (0.08), and total solids (0.08). For detailed discussion of the NSF-WQI_m, refer to Ott (1978).

Trend data on specific conductivity, sulphate, iron, and fecal coliforms were obtained from the STORET database and from the U.S. Geological Survey. Means and 95% confidence intervals were plotted for each time interval for each variable. Non-overlapping 95% confidence intervals were considered to indicate significant differences

between means ($\alpha = 0.05$). A review of arsenic, cadmium, chromium, copper, mercury, lead, manganese, aluminum, and nickel concentrations at PRM 143.5 for violations of VSWCB standards (VSWCB 1988b, 1989) was made using the STORET data base.

Physical Habitat

Nine sites were selected in the lower PR for habitat sampling (Table 1, Figure 6), in coordination with a 1988 sportfish abundance and habitat study (Cummins 1994). Three sites were located in the fifth order PR (PRM 174.4, 163.4, 158.3), and three were located in the upper reach of the sixth order PR (PRM 153.4, 146.8, 144.6). The final three sites were located in the lower reach of the sixth order PR in Virginia (PRM 123.0, 119.3, 117.3). All sites had well-developed riffle, run, and long pool habitat types (habitat type definitions in Bisson et al. 1982). Shallow-water and pool habitats at each site served as individual sampling units for physical habitat variables.

Habitat measurements for all sites were conducted in June, 1988 during low flow conditions (Table 3). Stratified uniform sampling incorporating the transect-point method was used in riffle, run, and pool habitats (Platts et al. 1983). Three transects were randomly located in both riffle and run habitats. Five sample points were located on each transect as follows: 0.5 m from each bank, 25% of the transect width out from each bank, and in the transect center. Depth, average velocity, dominant and subdominant substrata, and embeddedness were recorded at each point.

Average velocity was measured with a Marsh McBirney Model 201 portable

Table 3. Habitat variables measured in riffle, run, and pool habitats at nine sites (PRM 171.8 to 117.3) in the Powell River in Virginia. Habitat features were measured in June, 1988.

Habitat Variable	Riffle\Run Habitat	Pool Habitat
Stream width	x	x
Canopy closure	x	x
Embeddedness	x	
Sediment depth		x
Velocity median	x	x
Velocity variance (interquartile range)	x	x
Depth median	x	x
Depth variance (interquartile range)	x	x
Substrata		
Silt proportion	x	x
Sand proportion	x	x
Gravel proportion	x	x
Cobble proportion		x
Small cobble proportion	x	
Large cobble proportion	x	
Boulder proportion	x	x

Table 3. Continued

Habitat Variable	Riffle\Run Habitat	Pool Habitat
Bedrock proportion	x	x
Cover		
Root wad number and area	x	x
Log number and area	x	x
Woody debris area	x	x
Justicia americana area (Water willow)	x	x
Potamogeton sp. area (Pondweed)	x	x
Overhead vegetation area	x	x
Undercut bank area		x[a]
Rock wall area		x[a]

[a]This variable was sought site-wide but only occurred in pool mesohabitats.

current meter using either the 0.6-depth or two-point depth method where appropriate (Orth 1983). Dominant and subdominant substrata were visually estimated (Platts et al. 1983) following the modified Wentworth scale (Cummins 1962): boulder (>256 mm), cobble (64-256 mm), pebble (16-64 mm), gravel (2-16 mm), sand (0.0625-2 mm), and silt (<0.0625). Embeddedness was visually estimated using the system of Platts et al. (1983), classed from one (highest sedimentation) through 5 (lowest sedimentation). A validation study established that embeddedness estimates differentiated varying levels of sedimentation (Appendix C).

Riffle and run habitat substratum data from a 1990 crayfish habitat survey were substituted for the dominant-subdominant information collected in 1988 because of the greater number of samples taken, the addition of small and large cobble categories, and the reduced dependence on visual integration. Substrata data were collected in 1990 by using a uniform sampling with the transect-point method. At each site, 13 transects were uniformly spaced with a random start location for transect one. There were five points per transect, 12.5 %, 25 %, 50 %, 75 %, and 87.5 % of the stream width. Three substrata measurements were taken at each point following a modification of Bain et al. (1985b). A 30-cm transect is run congruent with the main transect beginning at the sample point and extending to the observer's right. The substrate particle at each 10 cm mark (three in total) was recorded by category. Substrata were classed as before, except that the cobble category was subdivided into small (64-128 mm) and large (128-256 mm) cobble.

Systematic sampling by the transect-point method was used in the pools in 1988.

Transects were spaced 50 m apart after random placement of the first transect. Transect points were the same used for sampling in riffles and runs. Depth, average velocity, dominant and subdominant substrata, and sediment depth were measured at each point. Depth and average velocity were measured as in the riffle and run habitats. Dominant and subdominant substrata were estimated by probing the stream bottom with a wooden rod. Sediment depths were measured by insertion of a graduated wooden rod into the sediment. Sediment depth was recorded after the rod reached solid material.

Stream width (m) was obtained as the median width of all transects. Site length was determined by meter tape. Canopy closure was determined by measuring vegetation overhang along each transect (Platts et al. 1983). Stream gradient at each site was measured to the nearest 1% with an Abney level and staff. Cover structure was comprised of rootwads, single logs, log complexes, woody debris complexes, water willow, pondweed, overhanging vegetation, undercut banks, and rock walls (Table 3). Area of cover items was obtained by meter tape.

Physical habitat data for each site were summarized as follows. The distributions of several habitat variables (riffle\run and pool depth, velocity, substratum, pool sediment depth, stream width, and canopy closure) at each site were tested for normality using either the Shapiro-Wilk test (if $n < 50$; Shapiro and Wilk 1965) or the Kolmogorov test (if $n > 50$; Hollander and Wolfe 1973). All tested variables had non-Gaussian distributions ($p \leq 0.05$) at all or most sites. Hence, distribution-free techniques (e.g., median, interquartile range, non-parametric inferential tests) were employed wherever possible.

Other habitat variable distributions that could not be tested for normality due to insufficient sample size were assumed non-Gaussian as well. Pool and riffle/run depth and velocity variances were expressed as interquartile ranges (25 % - 75 %; Zar 1984). Medians for site canopy closure, and pool sediment depth were determined. All substrate categories were expressed as percent. Amount of each cover type was calculated as area per unit habitat length (m^2/m).

Principal component analyses (PCA) were used to derive an index of embeddedness, substrata, and cover for habitat types within each site. The first principal component, which accounts for the greatest amount of variation in the data, was retained and scores on the first principal component served as index values for shallow-water and pool habitat embeddedness, substrata, or cover. Before using PCA, substrata and embeddedness category percentages were arcsine squareroot transformed. Cover variables were $\log_{10}(x + 1)$ transformed to stabilize variances (Green 1979).

Habitat variables and variable indices were examined for longitudinal patterns by correlating with PRM using Spearman's Rank Correlation test (Ludwig and Reynolds 1988). If an index was significantly correlated with PRM, variable loadings were evaluated to ascertain longitudinal patterns in particular habitat features (e.g., proportion boulder or area of logs).

Associations between sedimentation and land use activities were analyzed by using the Spearman's Rank Correlation test. Land use activity variables exhibiting strong positive correlations with sedimentation patterns were arcsine squareroot transformed and

subjected to stepwise multiple linear regression (using MINITAB) to ascertain the land use activities that are the best predictors of (and, by implication, primarily responsible for) sedimentation patterns in the Powell River. Finally, the relationship between the two measures of sedimentation, embeddedness and sediment depth, was evaluated by Spearman's Rank Correlation test.

Results and Discussion

Land Uses

For the sub-watershed upstream of PRM 117.3, essentially the Powell River watershed in Virginia, the major land use type composition was forest 63.2%, pasture 21.4%, surface mined area 6.9%, cropland 5.6%, urban 2.1%, and quarries, undetermined barren land, and other agriculture less than 0.1% (Table 4). Obvious longitudinal trends were evident for most land use types. Preceding from the most upstream (PRM 174.4) to furthest downstream (PRM 117.3) sub-watershed areas, forest (vs. PRM, $\rho = 0.97$, $p < 0.001$), surface mined area ($\rho = 0.99$, $p < 0.001$), urban ($\rho = 0.97$, $p < 0.001$), other agriculture ($\rho = 0.81$, $p < 0.001$), undetermined barren land ($\rho = 0.64$, $p < 0.003$), and quarries ($\rho = 0.93$, $p < 0.001$) land use types decreased in percent area. Conversely, pasture ($\rho = -0.98$, $p < 0.001$) and cropland ($\rho = -0.99$, $p < 0.001$) increased. The general upstream to downstream trends in major land uses agree with Sagona and Carroll (1991).

Table 4. Percent land use type in each of 19 sub-watersheds within the Virginia portion of the Powell River watershed.

	Percent Land Use							
Site (PRM)	Urban	Cropland	Pasture	Other Agriculture [a]	Forest	Surface Mined Area (active and inactive)	Quarries	Undetermined Barren Land
174.4	3.42	0.38	5.62	0.02	75.96	14.48	0.07	0.03
171.8	3.27	0.56	7.92	0.02	74.40	13.72	0.07	0.03
170.3	3.22	0.60	7.94	0.02	74.61	13.51	0.07	0.03
168.9	3.17	0.59	7.85	0.02	74.99	13.28	0.07	0.03
163.4	2.92	0.88	11.18	0.02	72.69	12.23	0.06	0.03
158.3	2.92	1.00	12.25	0.02	71.99	11.75	0.06	0.03
156.0	2.93	1.07	11.38	0.01	73.31	11.20	0.06	0.04
153.4	2.89	1.34	11.85	0.01	72.76	11.05	0.06	0.04
153.9	2.85	1.64	12.57	0.01	71.99	10.85	0.06	0.04
149.3	2.79	1.86	13.65	0.01	71.14	10.46	0.06	0.03
146.8	2.76	2.05	13.98	0.01	70.79	10.32	0.06	0.03

Table 4. (continued)

Site (PRM)	Percent Land Use							
	Urban	Cropland	Pasture	Other Agriculture	Forest	Surface Mined Area (active and inactive)	Quarries	Undetermined Barren Land
144.6	2.73	2.07	14.25	0.01	70.64	10.20	0.06	0.03
141.3	2.70	2.15	14.56	0.01	70.39	10.09	0.06	0.03
136.7	2.80	2.74	16.58	0.01	68.28	9.50	0.05	0.03
127.2	2.19	5.12	21.31	0.01	64.12	7.19	0.04	0.02
123.0	2.16	5.24	21.50	0.01	63.93	7.10	0.04	0.02
120.4	2.11	5.58	21.41	0.01	63.44	6.94	0.04	0.02
119.3	2.11	5.62	21.42	0.01	63.41	6.92	0.04	0.02
117.3	2.10	5.64	21.43	0.01	63.17	6.90	0.04	0.02

[a] Includes orchards, poultry production, and high-intensity livestock feeding operations.

Disturbance Regime

Flow Regime Classification

The grand mean of the average daily flows for the total 57 yr of record at PRM 143.5 was 529.5 CFS (15 CMS). The mean annual discharge per unit area was 1.6598 CFS/mi^2 (1.8155×10^{-2} CMS/km^2). The regression equation used to estimate bankfull discharge (flood disturbance threshold flow) was

$$\ln (\text{discharge} + 1) = 8.18 + 0.0292(\text{rank});$$

$$R^2 = 0.92;\ p < 0.001.$$

The median rank (50% exceedence probability) was 29. By entering 29 into the regression equation, taking the anti-ln of the result and subtracting one, the bankfull discharge was estimated at 8,322 CFS (244 CMS). All flows equal to or exceeding the 8,322 CFS flood threshold value are defined as disturbance events (floods).

By inspection, nine of the 11 variable values (81.8%) were closest to either the Mesic Groundwater stream type or intermediate between Perennial Runoff and Mesic Groundwater types (Table 5). Specifically, the Powell River is intermediate in the overall flow variability category, intermediate to slightly Mesic Groundwater in character for the flood regime pattern category, and intermediate to Mesic Groundwater for extent of intermittency category. Relative to the nation-wide environmental variability framework proposed by Poff and Ward (1989), the Powell River must be considered intermediate between Perennial Runoff and Mesic Groundwater because it was intermediate in the only category within which the two stream types differ; namely, overall flow variability.

Table 5. Values for 11 variables that describe the flow regimes of the Powell River in Virginia, Perennial Runoff stream type, Mesic Groundwater stream type, and intermediate Perennial Runoff-Mesic Groundwater. Definitions of variables are listed in the methods section (Table 2). Values for Perennial Runoff, Mesic Groundwater, and intermediate are reported as means from 78 study streams. Stream types are from Poff and Ward (1989). Variable definitions in Table 2.

Variable	Powell River	Perennial Runoff	Mesic Groundwater	Intermediate Perennial-Mesic
Overall Flow Variability				
ANNCV	20.86	31.40	10.50	20.95
PREDQ	0.58	0.44	0.72	0.58
C/P	0.77	0.69	0.89	0.79
Pattern of Flood Regime				
FLODFREQ	0.63	0.75	0.63	0.69
FLODINT	363	304	414	359
FLOD60D	0.486	0.480	0.510	0.495
FLODFREE	146	101	131	116
FLODDUR	1.24	1.60	2.20	1.90
Extent of Intermittency				
ZEROFLOW	0	0.10	0	0.05
LOWFLOW	0.665	0.450	0.830	0.640

The intermediate to Mesic Groundwater character of the Powell River flow regime is not surprising when viewed in the context of topography and geology. The gaging station is located in the Valley and Ridge physiographic province, where the valley is relatively broad, and has a high degree of karst development with many springs. Above PRM 180, the Powell River originates in the Cumberland Plateau physiographic province, where valleys are narrow, steep, and maintain a shallow aquifer that results in a flow regime more characteristic of Perennial Runoff. The "hybrid" physiographic nature of the Powell River watershed, combined with gaging station location, explain the intermediate to Mesic Groundwater nature of the sixth order Powell River flow regime.

Effects of Coal Mining on the Flow Regime

The flow regime of the 6th order Powell River during the two periods differed as illustrated by low-flow frequency curves (Figure 8). Except for discharges which occur at a probability of only 2 % or 1 year out of 50, the surface mining period (SMP, 1946-1988) had higher 7 Q's than the pre-surface mining period (PSMP, 1932-1945). The SMP $7Q_2$ (45.0 CFS) and $7Q_{10}$ (26.4 CFS) were 48.5 % and 21.7% higher than the PSMP $7Q_2$ (30.3 CFS) and $7Q_{10}$ (21.7 CFS), respectively. The low flow frequency curves for the identical time periods of the North Fork Holston River (unmined watershed) were essentially identical (Figure 9). For the period 1945-1988, the North Fork Holston River $7Q_2$ and $7Q_{10}$ were 32.7 CFS and 21.5 CFS, respectively. For 1932-1945, the North Fork Holston River $7Q_2$ and $7Q_{10}$ were 33.2 CFS and 21.2 CFS, respectively. Both differences

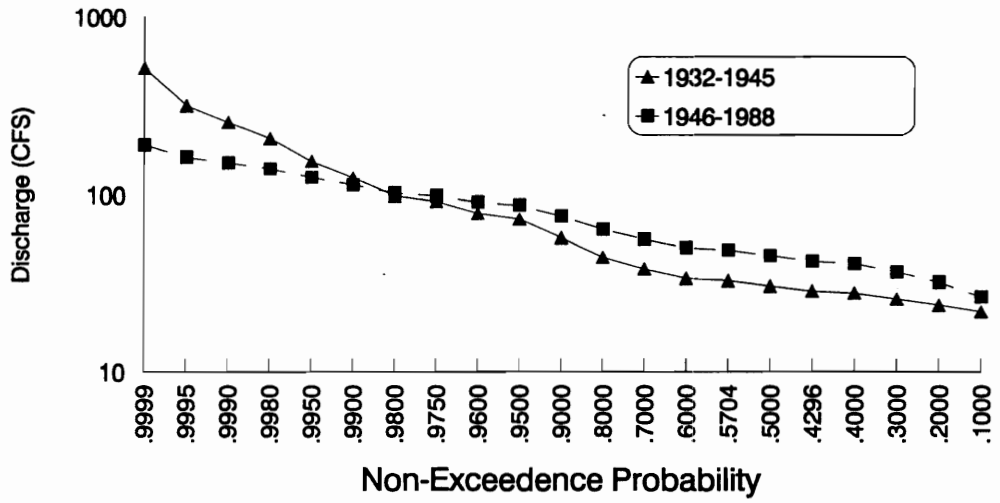

Figure 8. **Low-flow frequency curves for the Powell River during the pre-surface coal mining period (1932-1945) and the surface coal mining period (1946-1988). Plotted points are derived from the average minimum seven-consecutive-day flow (7Q) for each year of each time period.**

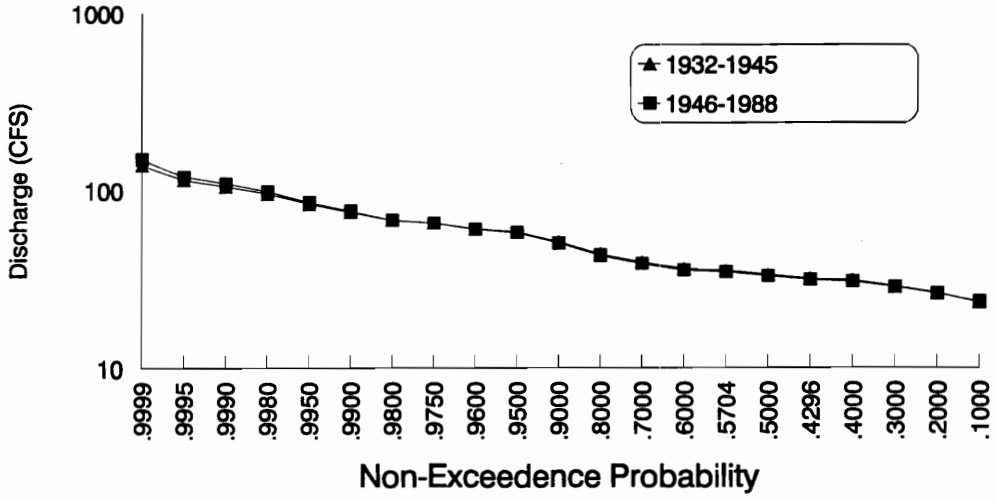

Figure 9. **Low-flow frequency curves for the North Fork Holston River during the 1932-1945 and 1946-1988 periods. Plotted points are derived from the average minimum seven-consecutive-day flow (7Q) for each year of each time period.**

are less than 1.5 %. In addition, precipitation levels in the Powell River watershed did not differ between periods (SMP precipitation mean = 50.5 inches/yr, PSMP mean = 49.1 inches/yr; Wilcoxon Rank Sum Test, $p < 0.6086$). The similarity of the North Fork Holston's low-flow frequency curves and of the Powell River watershed precipitation levels between time periods support the conclusion that surface mining has caused changes in the low flow dynamics. (This assumes that both the Powell River and North Fork Holston River watersheds underwent similar land use changes since 1932 except for the coal mining activity in the Powell River).

Moderate flows (range 20 to 650 CFS), evaluated through flow duration curves, indicated that the SMP flow regime had higher flows, merging toward the lowest and highest discharges (Figure 10). For example, discharge was ≥ 100 CFS during the SMP approximately 77% of the time, compared to only 71% of the time for the PSMP. The flow duration curves for the North Fork Holston River during the identical periods were not obviously different (Figure 11).

Causes of the altered flow regime of the Powell River during the SMP can be elucidated by examination of Powell River base-flow recession curves which indicate that the watershed groundwater storage capacity has increased substantially during the SMP. In each comparison, the time per log cycle of discharge (e.g., a flow range from 100 to 1,000 CFS or from 1,000 to 10,000 CFS) is longer for the SMP (Table 6), as illustrated by a selected recession pair comparison (Figures 12 and 13). Similar comparisons performed for the Russell Fork in Virginia by Larson and Powell (1986) were attributed

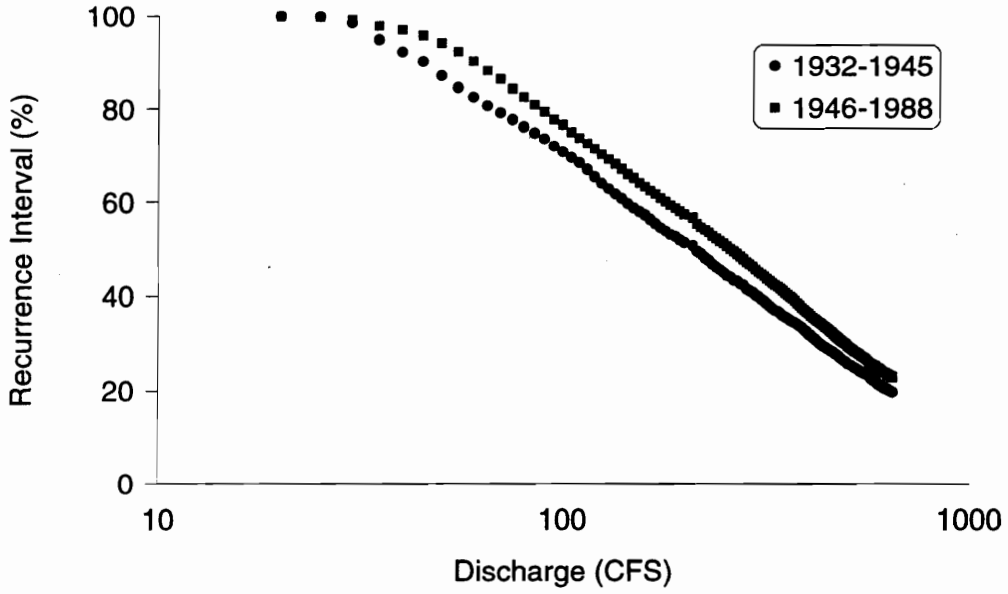

Figure 10. **Flow duration curves for the Powell River during the pre-surface coal mining period (1932-1945) and the surface coal mining period (1946-1988). Recurrence interval is the percent of time that a particular discharge is equaled or exceeded.**

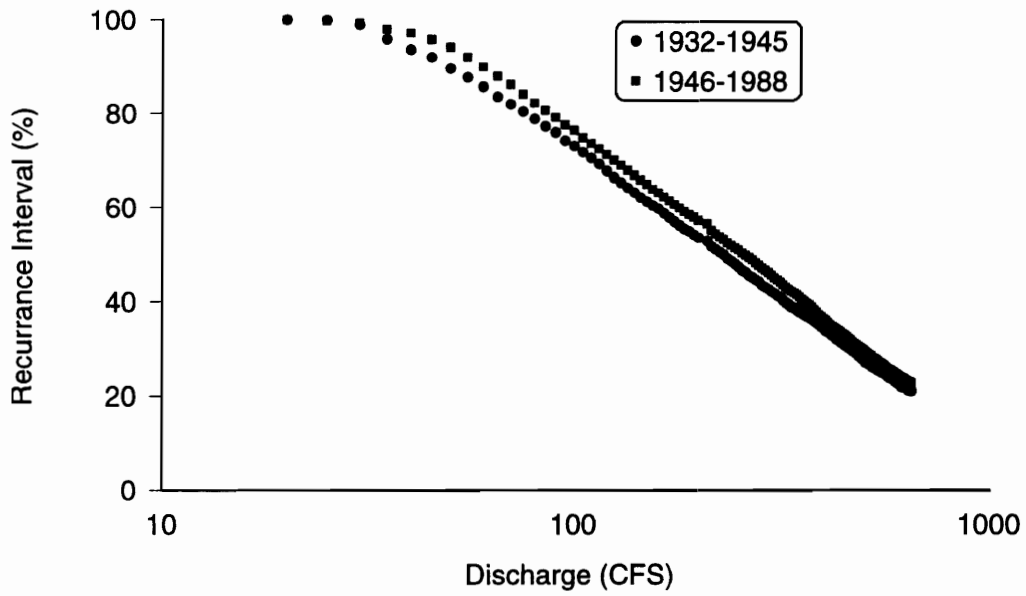

Figure 11. **Flow duration curves for the North Fork Holston River during the 1932-1945 and 1946-1988 periods. Recurrence interval is the percent of time that a particular discharge is equaled or exceeded.**

Table 6. Base-flow recession discharges for the Powell River during the pre-surface coal mining period (PSMP, 1932-1945) and the surface coal mining period (SMP, 1946-1988).

Discharge (CFS)	PSMP (days)	SMP (days)
780	34.9	65.7
1600	11.0	21.2
3800	14.9	32.6

to greater storage in surface mine spoil banks.

Consistently higher flows and a greater storage capacity that occurred in the Powell River during the SMP did not affect watershed yield (i.e., stream discharge). Discharge (as a percent of precipitation input) was not significantly different between periods (PSMP mean = 41.3%, SMP mean = 43.4%; Wilcoxon Rank Sum Test $p \leq 0.94$). These discharge percentages are similar to that found for the Russell Fork in Virginia (50% yield; 5 % of watershed surface-mined; Larson and Powell 1986) and for Cane Branch in Kentucky (40% yield; 10% of watershed mined; Collier et al. 1970). Hence, I conclude that during the SMP (and likely due to surface mining activities), the flow variability has been reduced; the low to moderate flows are generally higher and the high flows must be generally lower. The coefficients of variation for average daily flows support lower variability in the SMP (PSMP CV mean = 22.39, S.D. = 4.02 and SMP CV mean = 20.36, S.D. = 3.57).

Other researchers have observed lowered flow variability from surface coal mining activities (Larson and Powell 1986), and from deep mining activities that

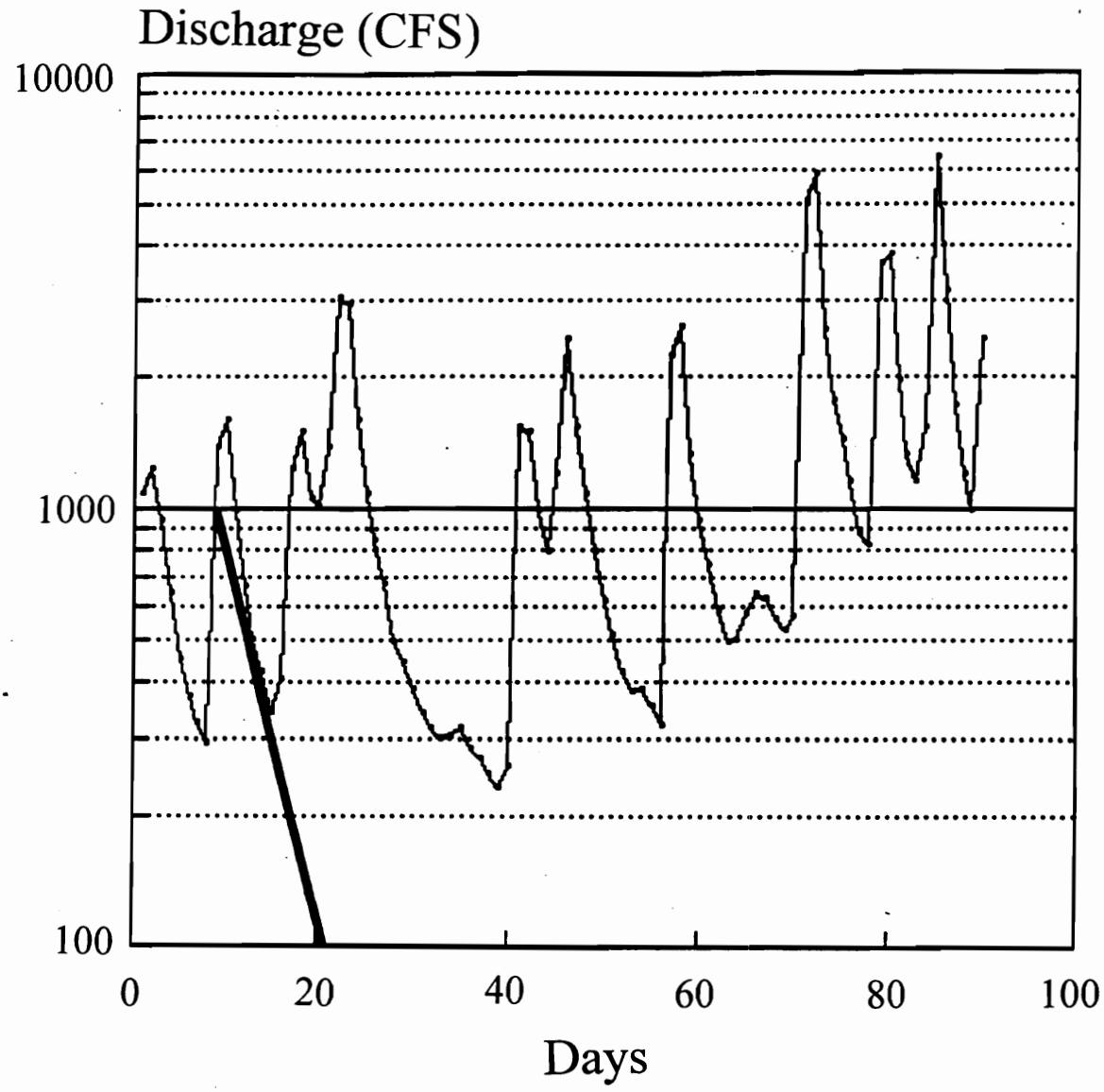

Figure 12. **Daily hydrograph for the Powell River (PRM 143.5) from January 1 - March 31, 1935. A base-flow recession curve is drawn on the recession from a peak discharge of approximately 1600 CFS. The curve encompasses one log-cycle of discharge (100 to 1,000 CFS) to standardize comparisons by flow range. The base-flow recession period is estimated at 11.0 days.**

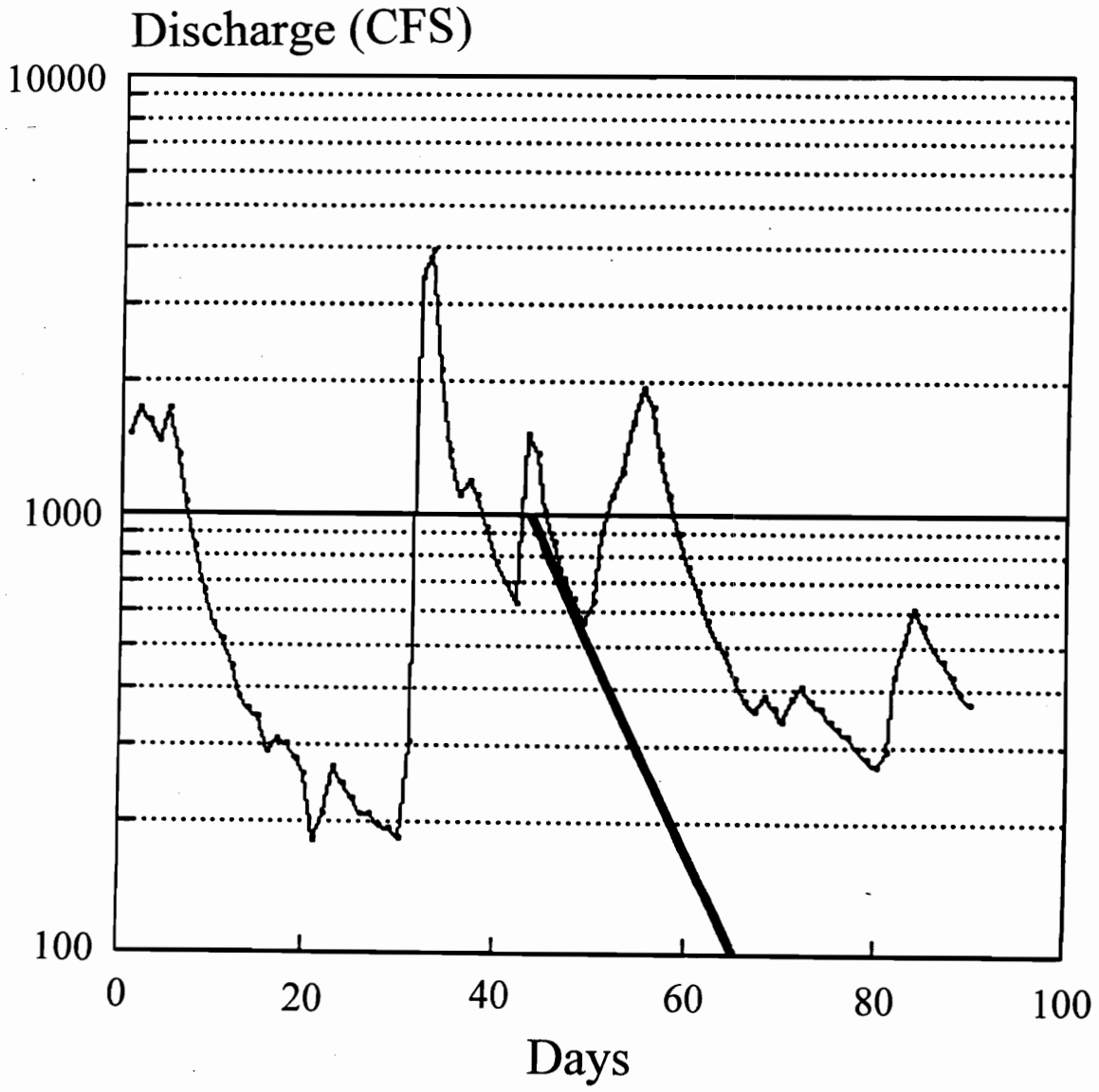

Figure 13. Daily hydrograph for the Powell River (PRM 143.5) from January 1 - March 31, 1985. A base-flow recession curve is drawn on the recession from a peak discharge of approximately 1600 CFS. The curve encompasses one log-cycle of discharge (100 to 1,000 CFS) to standardize comparisons by flow range. The base-flow recession period is estimated at 21.2 days.

occur topographically higher than the stream (Hobba 1981). They postulate that mining has enhanced the storage capacity of the aquifer by, in the case of surface mining, pulverizing the rheolith or overburden (Larson and Powell 1986). Thus, infiltration rates are increased (thereby lowering high flows) and groundwater is more slowly released into the stream (thereby augmenting low and moderate flows).

Conversely, flow variability in Cane Branch, Kentucky (10% of watershed surface-mined) was greater than that in a nearby unmined stream (Helton Branch; Collier et al. 1970). Cane Branch experienced more rapid runoff resulting in higher peak flows and reduced low flows. Although this hydrologic discrepancy between Collier et al. (1970) and Larson and Powell (1986) or my study may be a function of greater vegetation removal in Cane Branch (causing a greater amount of overland flow) or due to different stream sizes studied, differences in basin geology between Cane Branch and Helton Branch equivocate conclusions of that study. The upper strata of Helton Branch were composed primarily of porous sandstone. The primary permeability and storage capacity was greater than in the siltstone, shale, and sandstone strata of the Cane Branch watershed. These confounding factors were accounted for in both Larson and Powell (1986) and in my study through the comparison of flow regimes and precipitation levels between pre-mining and mining periods as well as the comparison of flow regimes of an unmined stream system between identical time periods. Hence, the best available information suggests that coal mining reduces flow variance in medium-sized streams as the sixth order Powell River or the Russell Fork, Kentucky.

Regarding abiotic vs. biotic organizational factors of fish assemblages in the sixth order Powell River, a slight decrease in flow variability might be considered to lessen environmental stochasticity and move the Powell River disturbance regime closer to the benign end of the harsh-benign spectrum (Allan 1995). This further may enable biotic processes to control community organization and increase structure stability (Starret 1951; Horwitz 1978; Coon 1987; McNeely 1987; Matthews 1990;), and, if anything, increase the precision of using fish assemblage taxonomic structure characteristics for monitoring indicators of the Powell River (Stewart and Loar 1994).

In addition, when viewed within Poff and Ward's (1989) framework, the slight change in flow variability would not change the Powell River's placement as intermediate Perennial Runoff-Mesic Groundwater. These results support the assertion that, while surface mining activities lowered environmental stochasticity in the sixth order, mining had little effect on the disturbance regime.

Of concern, however, is the effect that prolonged contact between pulverized rock and groundwater will have on water quality. Groundwater generally contains mineral concentrations characteristic of the rocks through which it moves (Kiesler 1987). In addition, Larson and Powell (1986) attributed some changes in stream water quality to prolonged contact with pulverized rock. Water quality can affect fish distributions (Alabaster and Lloyd 1980; Hughes and Gammon 1987; Taylor et al. 1993). Water quality changes due to coal mining and other watershed land-use activities as well as probable effects on fish assemblages are discussed in the following section.

Water Quality

Watershed-Level Description

Impairment of water quality has occurred in the Powell River system. Sulphate ($p \leq 0.004$) and specific conductivity ($p \leq 0.008$) were higher in mined-land Cumberland Plateau tributaries (MTRIBCP) versus unmined-land Cumberland Plateau tributaries (UMTRIBCP) (Table 7). Calcium, hardness, iron, manganese, turbidity, and total suspended solids means also appeared higher in MTRIBCP (Table 7), but were not statistically substantiated, possibly due to small sample sizes. The MTRIBCP had lower alkalinity ($p \leq 0.044$), and higher specific conductivity ($p \leq 0.008$), sulphate ($p \leq 0.001$), and turbidity ($p \leq 0.004$) compared to unmined-land Valley and Ridge tributaries category (UMTRIBVR) (Table 7). Calcium levels ($p \leq 0.247$) and pH ($p \leq 0.196$) were similar between MTRIBCP and UMTRIBVR. Compared to the sixth order Powell River (PR) reach, MTRIBCP had higher levels of sulphate ($p \leq 0.0001$), iron ($p \leq 0.0001$), manganese ($p \leq 0.001$), turbidity ($p \leq 0.0001$) and lower levels of calcium ($p \leq 0.042$). In addition, MTRIBCP appeared to have higher levels of total dissolved solids, total suspended solids, hardness, specific conductivity and lower alkalinity compared to the sixth order reach (Table 7). These differences, however, were not statistically substantiated (all $p > 0.05$). The pH values ($p \leq 0.196$) were similar between MTRIBCP and the sixth order river.

Elevated levels of sulfate, iron, manganese, specific conductivity, and turbidity in the MTRIBCP also have been reported by Larson (1985), VDWSC (1989), and noted for

Table 7. Values of 16 water quality variables for streams (grouped into five stream classifications) in the Powell River drainage in Virginia. Water quality values for selected drainages in the Powell River drainage in Virginia. Data summarized for all samples dates[a]. MTRIBCP = mined tributaries in the Cumberland Plateau physiographic province (including the mainstem Powell River in the Cumberland Plateau), UMTRIBCP = unmined tributaries in the Cumberland Plateau, UMTRIBVR = unmined tributaries in the Valley and Ridge physiographic province, 5th order Powell River (between the South Fork Powell confluence [PRM 178.1] and the North Fork Powell confluence [PRM 156.6]), and 6th order Powell River (downstream of the North Fork Powell River confluence to PRM 117.3). Asterisks denote no data.

	Water Quality Variable (mean ± S.D.) (n = number of samples)					
Drainage	Alkalinity (mg/L)	BOD[b] (mg/L)	Calcium (mg/L)	Specific Conductivity[c] (µsiemens/cm)	Hardness (mg/L)	Iron (mg/L)
MTRIBCP	77.0 ± 35.6 n = 15	* n = 0	71.5 ± 42.5 n = 25	314 ± 186 n = 77	199.3 ± 141 n = 23	0.68 ± 1.12 n = 22
UMTRIBCP	46.0 n = 1	* n = 0	14.0 n = 1	33 ± 15 n = 4	39 ± 23 n = 2	0.14 n = 1
UMTRIBVR	131.7 ± 40.5 n = 3	* n = 0	33.0 ± 31.1 n = 2	151 ± 89 n = 9	153 ± 34 n = 3	* n = 0
5th Order Powell River	105.5 ± 20.2 n = 11	2.79 ± 1.72 n = 5	75.4 ± 15.8 n = 6	295 ± 115 n = 11	139 ± 16 n = 8	0.19 ± 0.05 n = 9
6th Order Powell River	96.7 ± 16.1 n = 26	1.97 ± 2.11 n = 15	76.9 ± 14.2 n = 14	254 ± 101 n = 23	129 ± 17 n = 18	0.14 ± 0.06 n = 18

Table 7. (continued)

Stream Type	Water Quality Variable (mean ± S.D.) (n = number of samples)					
	Manganese (mg/L)	Nitrate (mg/L)	pH (units)	Total Phosphorus (mg/L)	Sulphate (mg/L)	Turbidity (FTU)
MTRIBCP	0.37 ± 0.88 n = 22	0.2 n = 1	7.96 ± 0.39 n = 25	* n = 0	153.3 ± 132.4 n = 65	60.4 ± 126.4 n = 50
UMTRIBCP	0.00 n = 1	* n = 0	7.71 n = 1	* n = 0	9.0 ± 1.0 n = 3	4.5 ± 0.7 n = 2
UMTRIBVR	* n = 0	* n = 0	8.17 ± 0.23 n = 6	* n = 0	6.1 ± 5.9 n = 10	28.9 ± 93.7 n = 30
5th Order Powell River	0.00 ± 0.00 n = 6	0.67 ± 0.14 n = 10	8.06 ± 0.26 n = 14	0.11 ± 0.09 n = 7	88.6 ± 36.6 n = 17	37.8 ± 63.8 n = 32
6th Order Powell River	0.00 ± 0.00 n = 9	0.61 ± 0.14 n = 26	8.14 ± 0.27 n = 37	0.08 ± 0.05 n = 21	63.4 ± 22.1 n = 42	15.8 ± 27.3 n = 83

Table 7. (continued)

Stream Type	Water Quality Variable (mean ± S.D.) (n = number of samples) Total Solids (mg/L)	Total Dissolved Solids (mg/L)	Total Suspended Solids (mg/L)	Dissolved Oxygen Saturation (%)
MTRIBCP	190.0 ± 70.7 n = 2	385.00 ± 308 n = 12	32.27 ± 19.04 n = 24	99.4 n = 1
UMTRIBCP	* n = 0	* n = 0	9 ± 0 n = 2	* n = 0
UMTRIBVR	* n = 0	* n = 0	21.50 n = 1	* n = 0
5th Order Powell River	250.3 ± 46.7 n = 8	241.4 ± 66.3 n = 2	20.60 ± 20.7 n = 3	87.5 ± 14 n = 10
6th Order Powell River	193.6 ± 45.2 n = 24	149.0 ± 25.6 n = 5	28.10 ± 35.2 n = 9	89.1 ± 9 n = 29

[a] Sample dates: 6/05/89, 6/06/89, 7/31/89, 8/18/89, 8/21/89, 8/29/89, 9/07/89, 9/10/89, 9/21/89, 9/22/89, 9/25/89, 9/26/89, 9/27/89, 11/15/89, 12/14/89, 2/05/90, 2/27/90, 3/22/90, 5/13/90, 8/14/90, 11/20/90, 12/06/90

[b] Biological Oxygen Demand

[c] Referenced to 25 °C

other streams draining mined lands (Biesecker and George 1966; FWPCA 1968; Collier et al. 1970; USEPA 1975; Letterman and Mitsch 1978; Rogers and Powell 1983; Dickens et al. 1985; Becker et al. 1986; Nelson et al. 1991). Of particular note is the high level of turbidity. For the entire PR watershed in Virginia, Sagona and Carroll (1991) determined that erosion potential from mined lands exceeds erosion inputs from all other land use types combined. Mined land erosion constitutes 60 % of erosion potential, cropland 19%, disturbed land (e.g., construction and commercial development sites) 17 %, and pasture land accounts for only 4 %.

Contrasting within the lower Powell River mainstem, the fifth order PR had higher specific conductivity ($p \leq 0.045$), iron ($p \leq 0.015$), sulphate ($p \leq 0.0031$), turbidity ($p \leq 0.0013$), and total solids ($p \leq 0.0014$). Although mean total dissolved solids, total suspended solids, nitrate, phosphate, and biological oxygen demand (BOD) also were higher in the fifth order reach, these comparisons were not statistically significant (all $p > 0.05$; Table 7). Mean dissolved oxygen, dissolved oxygen saturation, pH, alkalinity, hardness, calcium, and manganese were very similar between reaches (all $p > 0.05$; Table 7). The finding of higher turbidity in the fifth order agrees with that concluded in Temple et al. (1990) using turbidity data from Cummins (1994).

Water quality spatial patterns within the lower PR are visually depicted by a principal components analysis of four intensively sampled lower reach water quality stations (PRM 171.2, 153.4, 138.3, and 123.8). Principal component analyses of 10 water quality variables sampled during four non-spate dates (12/14/89, 2/28/90, 3/22/90,

5/13/90) show a distinct upstream-to-downstream gradient on the second principle component (Figure 14 and Table 8). Upstream stations had higher readings of specific conductivity, sulfate, and total solids. Overall, the PRM 171.2 station had slightly higher turbidity levels (PC 1). In summary, univariate and principal components analyses indicate that the fifth and sixth order reaches differ in some water quality attributes.

Of note, mean values for non-carbonate hardness (APHA 1975) increase as one proceeds upstream in the lower PR mainstem (PRM 123.8 = 32.5 mg/L, PRM 138.6 = 37.3 mg/L, PRM 153.4 = 39.7 mg/L, and PRM 171.2 = 43.3 mg/L). This indicates that some of the calcium and magnesium ions are associated with sulphate or possibly nitrates rather than bicarbonate and carbonate (Boyd 1979). Hence, as one proceeds upstream, the PR is shifting from a bicarbonate-carbonate buffering system toward a sulphate-dominated system. This conclusion is further supported by regression of sulphate on alkalinity within the MTRIBCP, fifth order PR, and sixth order PR categories. Sulphate accounted for more variation in alkalinity as one proceeds upstream from sixth order (R^2 = 0.0, $p \leq 0.477$), to fifth order ($R^2 = 0.58$, $p \leq 0.01$), to MTRIBCP ($R^2 = 0.88$, $p \leq 0.01$).

Of critical interest, the calcium:hardness ratio is 0.81 for UMTRIBCP\UMTRIBVR combined and 0.53 for MTRIBCP (this includes the PR mainstem in the Cumberland Plateau). This finding suggests that the concentration of polyvalent metal ions (other than calcium and magnesium) increase in streams in closer proximity to mined lands. Since many metals exist in a polyvalent state (APHA 1975), significant metal loading to the PR may be occurring from mined-land tributaries.

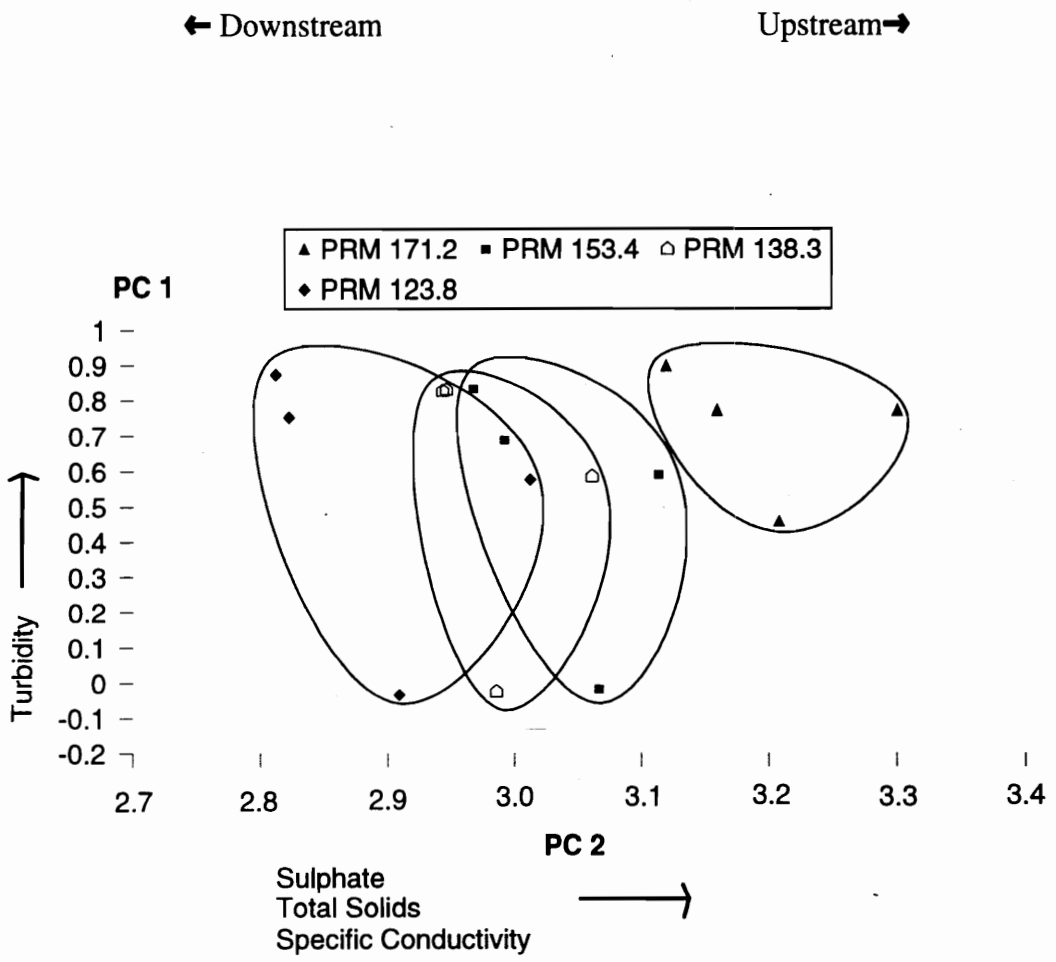

Figure 14. **Principal components analysis of 10 water quality variables for four stations at four non-spate dates (12/14/89, 2/28/90, 3/22/90, 5/13/90) within the Powell River in Virginia. Water quality stations were located at PRM 171.2, 153.4, 138.2, and 123.8.**

Table 8. Loadings of 10 water quality variables on the first two principal components used to distinguish water quality stations PRM 171.2, 153.4, 138.3, and 123.8. Proportion of total variance accounted for by each component is listed. Non-spate dates only. PC = principal component.

	PC 1	PC 2
Proportion of variance:	0.797	0.133
Cumulative variance:	0.797	0.930
Variable	Loadings	
Alkalinity	-0.096	0.043
Specific conductivity	0.085	0.322
Hardness	-0.067	0.190
Nitrates	0.043	-0.007
pH	-0.002	-0.032
Total phosphorus	0.017	-0.023
Sulphates	0.012	0.786
Total solids	0.046	0.400
Turbidity	0.987	-0.040
Dissolved O_2 saturation	-0.009	-0.277

Moreover, using stepwise multiple linear regression, calcium was the only variable selected to explain variability in specific conductivity within the sixth order PR (R^2 = 0.88). Calcium and iron, conversely, were selected as model variables to explain specific conductivity variance in the MTRIBCP (R^2 = 0.98). Not enough observations were made to evaluate factors contributing to specific conductivity in the fifth order PR. Consequently, a one-time sample of six metals (exclusive of iron and manganese) was made at five mainstem PR sites and one North Fork Powell River (NFPR) site (Table 10). By visual inspection, upstream to downstream concentration gradients were observed for zinc, copper, and cadmium. Aluminum, arsenic, and chromium concentrations exhibited no such trends. Aluminum and zinc concentrations, however, were elevated at the NFPR site. Although none of these metal concentrations exceeded state water quality standards (VSWCB 1988b, 1989) or above lowest observed effect levels for aquatic life (Nelson et al. 1991), there should be a more comprehensive and frequent monitoring program for heavy metals established in the PR drainage (McCann 1993). Sediment, as opposed to water column, sampling for metals may provide a more accurate picture of metal concentrations in the system (McCann 1993; Yeager 1994).

Water Quality of Spate versus Non-spate Flows

Comparisons also were made for water quality variables between spate and non-spate periods within the MTRIBCP, fifth order, and sixth order categories (Tables 10 and 11; UMTRIBCP and UMTRIBVR water quality values included for reference). Of

Table 9. Metal analyses of water samples from five sites on the Powell River in Virginia and one site on the North Fork Powell River in Virginia (5/13/90). PRM = Powell River Mile. NFPRM = North Fork Powell River Mile. All values are in µg/L.

Station	Aluminum[a]	Arsenic[b]	Cadmium[c]	Chromium[d]	Copper[e]	Zinc[f]
PRM 192.6	96	ND[g]	0.3	ND	3	24.3
PRM 182.4	112	1	ND	0.8	2	5.5
PRM 167.4	116	1	0.1	ND	1	8.6
PRM 138.3	107	ND	0.1	ND	1	2.4
PRM 123.8	115	ND	ND	ND	ND	1.8
NFPRM 4.7	184	ND	ND	ND	1	11.3

[a]Limit of Detection: 1 µg/L
[b]Limit of Detection: 1 µg/L
[c]Limit of Detection: 0.1 µg/L
[d]Limit of Detection: 0.5 µg/L
[e]Limit of Detection: 1 µg/L
[f]Limit of Detection: 0.1 µg/L
[g]Not detected

Table 10. Values of 16 water quality variables for streams (grouped into five stream classifications) in the Powell River drainage in Virginia. Water quality samples taken during non-spate periods[a] (using PRM 138.3 as the reference point of spate or non-spate determination). MTRIBCP = mined tributaries in the Cumberland Plateau physiographic province (including the mainstem Powell River in the Cumberland Plateau), UMTRIBCP = unmined tributaries in the Cumberland Plateau, UMTRIBVR = unmined tributaries in the Valley and Ridge physiographic province, 5th order Powell River (between the South Fork Powell confluence [PRM 178.1] and the North Fork Powell confluence [PRM 156.6]), and 6th order Powell River (downstream of the North Fork Powell River confluence to PRM 117.3). Asterisks denote no data.

	Water Quality Variable (mean ± S.D.) (n = number of samples)					
Stream Type	Alkalinity (mg/L)	BOD[b] (mg/L)	Calcium (mg/L)	Specific Conductivity[c] (µsiemens/cm)	Hardness (mg/L)	Iron (mg/L)
MTRIBCP	77.8 ± 36.8 n = 14	* n = 0	95.5 ± 47.2 n = 13	355 ± 203 n = 46	199 ± 141 n = 23	0.68 ± 1.12 n = 22
UMTRIBCP	46.0 n = 1	* n = 0	* n = 0	32 ± 26 n = 2	39 ± 23 n = 2	0.14 n = 1
UMTRIBVR	131.7 ± 40.5 n = 3	* n = 0	* n = 0	108 ± 96 n = 3	153 ± 34 n = 3	* n = 0
5th Order Powell River	92.4 ± 6.5 n = 7	2.67 ± 1.95 n = 4	78.4 ± 0.7 n = 3	269 ± 20 n = 8	135 ± 10 n = 6	0.18 ± 0.04 n = 7
6th Order Powell River	89.8 ± 11.3 n = 18	2.21 ± 2.31 n = 12	76.3 ± 3.3 n = 7	212 ± 12 n = 14	124 ± 11 n = 13	0.13 ± 0.05 n = 13

Table 10. (continued)

Stream Type	Water Quality Variable (mean ± S.D.) (n = number of samples) Manganese (mg/L)	Nitrate (mg/L)	pH (units)	Total Phosphorus (mg/L)	Sulphate (mg/L)	Turbidity (FTU)
MTRIBCP	0.37 ± 0.88 n = 22	* n = 0	7.92 ± 0.43 n = 14	* n = 0	209.3 ± 156.5 n = 30	50.3 ± 74.0 n = 9
UMTRIBCP	0.00 n = 1	* n = 0	7.71 n = 1	* n = 0	8.0 n = 1	* n = 0
UMTRIBVR	* n = 0	* n = 0	8.28 n = 1	* n = 0	6.0 n = 1	5.4 ± 2.8 n = 7
5th Order Powell River	0.00 ± 0.00 n = 5	0.58 ± 0.08 n = 6	8.21 ± 0.22 n = 7	0.08 ± 0.05 n = 6	97.1 ± 14.7 n = 9	7.8 ± 4.0 n = 12
6th Order Powell River	0.00 ± 0.00 n = 6	0.56 ± 0.11 n = 18	8.24 ± 0.23 n = 21	0.06 ± 0.04 n = 18	63.8 ± 16.2 n = 21	5.0 ± 4.1 n = 36

Table 10. (continued)

Stream Type	Water Quality Variable (mean ± S.D.) (n = number of samples) Total Solids (mg/L)	Total Dissolved Solids (mg/L)	Total Suspended Solids (mg/L)	Dissolved Oxygen Saturation (%)
MTRIBCP	* n = 0	* n = 0	45.33 n = 1	* n = 0
UMTRIBCP	* n = 0	* n = 0	* n = 0	* n = 0
UMTRIBVR	* n = 0	* n = 0	* n = 0	* n = 0
5th Order Powell River	254.6 ± 50.9 n = 6	241.4 ± 66.3 n = 2	9.53 ± 10.99 n = 2	85.5 ± 11.1 n = 7
6th Order Powell River	193.6 ± 52.0 n = 18	160.21 ± 6.21 n = 4	5.16 ± 5.23 n = 6	87.5 ± 12.2 n = 21

[a] Sample dates: 8/18/89, 9/07/89, 9/10/89, 9/21/89, 9/22/89, 11/15/89, 12/14/89, 2/27/90, 3/22/90, 5/13/90, 11/20/90, and 12/06/90

[b] Biological Oxygen Demand

[c] Referenced to 25 °C

Table 11. Values of 16 water quality variables for streams (grouped into five stream classifications) in the Powell River drainage in Virginia. Water quality samples taken during spate periods[a] (using PRM 138.3 as the reference point of spate or non-spate determination). MTRIBCP = mined tributaries in the Cumberland Plateau physiographic province (including the mainstem Powell River in the Cumberland Plateau), UMTRIBCP = unmined tributaries in the Cumberland Plateau, UMTRIBVR = unmined tributaries in the Valley and Ridge physiographic province, 5th order Powell River (between the South Fork Powell confluence [PRM 178.1] and the North Fork Powell confluence [PRM 156.6]), and 6th order Powell River (downstream of the North Fork Powell River confluence to PRM 117.3). Asterisks denote no data.

	Water Quality Variable (mean ± S.D.) (n = number of samples)					
Stream Type	Alkalinity (mg/L)	BOD[b] (mg/L)	Calcium (mg/L)	Specific Conductivity[c] (µsiemens/cm)	Hardness (mg/L)	Iron (mg/L)
MTRIBCP	66.0 n = 1	* n = 0	45.5 ± 9.8 n = 12	261 ± 158 n = 31	* n = 0	* n = 0
UMTRIBCP	* n = 0	* n = 0	14.0 n = 1	35 ± 4 n = 2	* n = 0	* n = 0
UMTRIBVR	* n = 0	* n = 0	33.0 ± 31.1 n = 2	173 ± 85.5 n = 6	* n = 0	* n = 0
5th Order Powell River	128.3 ± 12.6 n = 4	3.3 n = 1	75.0 ± 33.9 n = 2	366 ± 234 n = 3	171 n = 1	0.31 n = 1
6th Order Powell River	112.3 ± 14.7 n = 8	1.00 ± 0.25 n = 3	75.3 ± 21.7 n = 6	318 ± 142 n = 9	158 ± 9 n = 3	0.20 ± 0.09 n = 3

Table 11. (continued)

Stream Type	Water Quality Variable (mean ± S.D.) (n = number of samples) Manganese (mg/L)	Nitrate (mg/L)	pH (units)	Total Phosphorus (mg/L)	Sulphate (mg/L)	Turbidity (FTU)
MTRIBCP	* n = 0	0.2 n = 1	8.01 ± 0.36 n = 11	* n = 0	105.4 ± 83.7 n = 35	62.6 ± 135.6 n = 41
UMTRIBCP	* n = 0	* n = 0	* n = 0	* n = 0	9.5 ± 0.7 n = 2	4.5 ± 0.7 n = 2
UMTRIBVR	* n = 0	* n = 0	8.06 ± 0.17 n = 4	* n = 0	6.1 ± 6.3 n = 9	36.1 ± 106.5 n = 23
5th Order Powell River	0.00 n = 1	0.80 ± 0.12 n = 4	7.91 ± 0.21 n = 7	0.28 n = 1	79.0 ± 51.1 n = 8	55.7 ± 75.7 n = 20
6th Order Powell River	0.00 ± 0.00 n = 3	0.73 ± 0.13 n = 8	8.00 ± 0.26 n = 16	0.17 ± 0.01 n = 3	63.9 ± 27.2 n = 21	24.1 ± 34.0 n = 47

Table 11. (continued)

Stream Type	Water Quality Variable (mean ± S.D.) (n = number of samples) Total Solids (mg/L)	Total Dissolved Solids (mg/L)	Total Suspended Solids (mg/L)	Dissolved Oxygen Saturation (%)
MTRIBCP	190.0 ± 70.7 n = 2	318.00 ± 308 n = 5	31.70 ± 19.26 n = 23	99.4 n = 1
UMTRIBCP	* n = 0	* n = 0	9 ± 0 n = 2	* n = 0
UMTRIBVR	* n = 0	* n = 0	21.50 n = 1	* n = 0
5th Order Powell River	237.4 ± 43.3 n = 2	* n = 0	42.67 n = 1	92.1 ± 2.0 n = 3
6th Order Powell River	193.4 ± 13.3 n = 6	104.17 n = 1	73.87 ± 13.17 n = 3	93.2 ± 5.7 n = 8

[a] Sample dates: 6/05/89, 6/06/89, 7/31/89, 8/21/89, 8/29/89, 9/25/89, 9/26/89, 9/27/89, 2/05/90, and 8/14/90
[b] Biological Oxygen Demand
[c] Referenced to 25 °C

variables having sufficient sample sizes for analyses, calcium ($p \leq 0.0008$), specific conductivity ($p \leq 0.0232$), and sulphate ($p \leq 0.0006$) decreased during spates within the MTRIBCP category. Compared to non-spate flow levels, higher turbidity and lower total dissolved solids readings during spates were not statistically significant ($p \leq 0.434$ and 0.516, respectively). The pH was similar between spate and non-spate flows within the MTRIBCP.

Directional changes for several water quality variables between spate and non-spate flows were similar within both the fifth and sixth order PR. Alkalinity ($p \leq 0.01$), nitrates ($p \leq 0.03$), and turbidity ($p \leq 0.0004$) increased during spates in both reaches. Dissolved oxygen ($p \leq 0.05$) and pH ($p \leq 0.02$) declined during spates. Higher measures of specific conductivity ($p > 0.197$), total solids ($p \leq 0.6171$), and dissolved oxygen saturation ($p > 0.2319$) and lower sulphate concentrations ($p > 0.3123$) were not statistically substantiated. Calcium levels were similar for spate and non-spate flows within both reaches ($p = 1.0$). Manganese remained nondetectable. Brede and Benham (1996) also observed that pH declined and suspended-sediment concentrations increased in the sixth order Powell River in Tennessee during "storm flows".

Sample sizes allowed additional water quality variable comparisons for flow type within the sixth order section. During spates, the sixth order reach exhibited higher hardness ($p \leq 0.0106$), total phosphorus ($p \leq 0.0077$), and total suspended solids ($p \leq 0.0282$). Elevated iron concentrations during spates were not statistically significant ($p \leq 0.2528$). There was no difference in BOD readings between spate and non-spate

flows ($p \leq 0.94$).

The calcium:hardness ratio in the sixth order Powell River declined during spates (from 0.66 to 0.59; $p \leq 0.0227$). This indicates that polyvalent metal ions, other than calcium and magnesium, increase during spates.

A very visible change to the fourth, fifth, and sixth order PR during spates is the dramatic increase in turbidity over non-spate periods (Figure 15). To illustrate, turbidity "slugs" were recorded from the upper (4^{th} order and below) PR on two dates (Figure 17), as well as a two consecutive-day snapshot of the movement of a turbidity slug (Figure 18). Although turbidity inputs to the PR from unmined tributaries occur (see Figure 19 and personal observation), the main source of turbidity again appears to originate from coal-mined lands.

In general, water quality appears to decline during spates within the 5^{th} and 6^{th} order PR. Part of the impacts during spates may be from mined lands (e.g., lowered pH, increased turbidity, nitrates, polyvalent metal ions) and part may be from agricultural and urban lands (e.g., increased total phosphorous, turbidity). Conversely, there may be a slight increase in water quality within the MTRIBCP during spates. This increase may be primarily due to dilution. The opposite pattern in the lower PR (i.e., water quality deterioration during spates) could possibly be from resuspension of contaminants contained within sediments in the MTRIBCP area and carried in the turbidity slugs downstream.

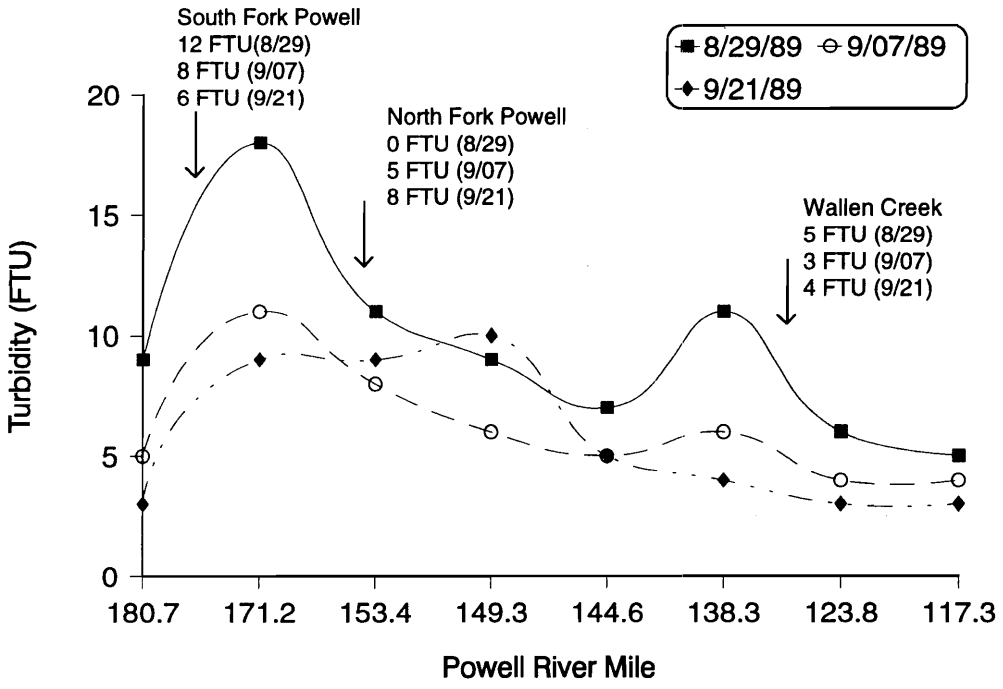

Figure 15. Turbidity regimes during one spate period (8/29/89) and two non-spate periods (9/07/89, 9/21/89) within the 4th, 5th, and 6th order Powell River in Virginia. Discharge measured at the U.S.G.S. gauging station at PRM 143.5 was 279 CFS (8/29/89), 190 CFS (9/07/89), and 326 CFS (9/21/89).

Sources of Water Quality Impairment

From analyses reported in the preceding sections, coal mined-lands appear to be a significant source of contaminants to the Powell River system. In addition, since the percent watershed in coal mining decreases downstream (vs. PRM, $\rho = 0.99$, $p \leq 0.001$), whereas pastureland (vs. PRM, $\rho = -0.98$, $p \leq 0.001$) and cropland (vs. PRM, $\rho = -0.99$, $p \leq 0.001$) increase (Table 4), coal mining is again implicated as the primary source in the lower PR for the upstream-to-downstream, high-to-low levels of turbidity, total solids, specific conductivity, iron, and sulphate. The only other land use that increased upstream, urban areas, encompassed only 3.4% of the watershed at PRM 174.4 (as opposed to 14.5% devoted to surface-mined land).

To help illustrate the influence of mine lands on water quality of the PR mainstem, snapshots of drainage-wide spatial water quality patterns during two base-flow recessions (2/27/90 and 3/22/90) are presented for pH (Figure 19), alkalinity (Figure 20), hardness (Figure 21), specific conductivity (Figure 22), sulphate (Figure 23), iron (Figure 24), and manganese (Figure 25). It is important to note that coal mining areas include essentially the entire North Fork Powell River drainage and that portion of the upper Powell River drainage upstream of the South Fork - Powell River confluence in Wise County (Figure 3).

Whereas pH levels appear good throughout, lower values in the upper Powell River, Black Creek (pH = 7.6), and Puckett Creek (pH = 7.1) raise concerns regarding localized pH depressions. The VSWCB (1988) also identified pH depressions in Puckett

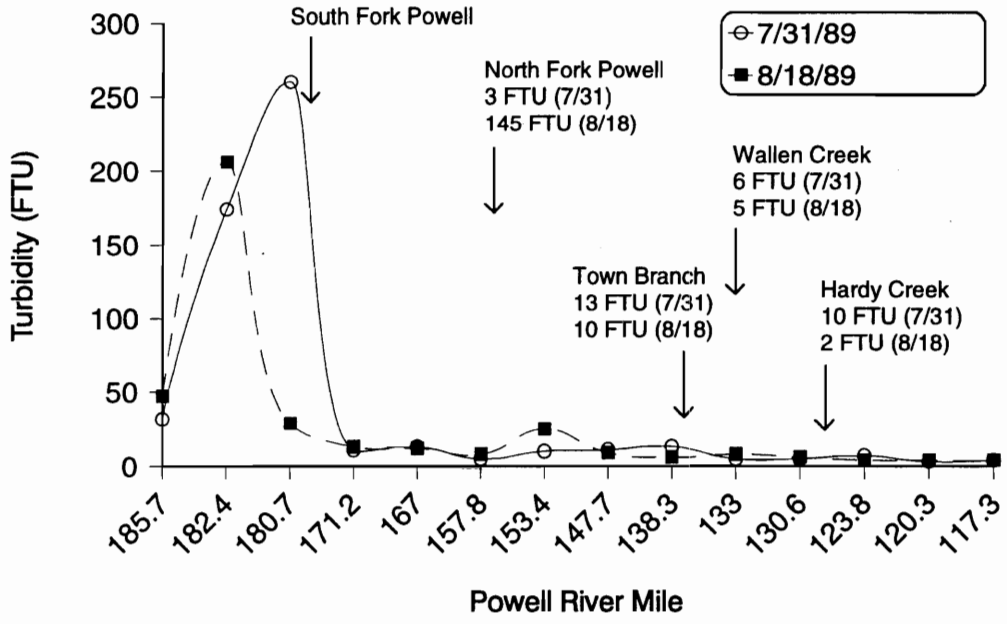

Figure 16. **Turbidity regimes during two spate periods (7/31/89 and 8/18/89) within the 4th, 5th, and 6th order Powell River in Virginia. Discharge measured at the U.S.G.S. gauging station at PRM 143.5 was 181 CFS (7/31/89) and 192 CFS (8/18/89).**

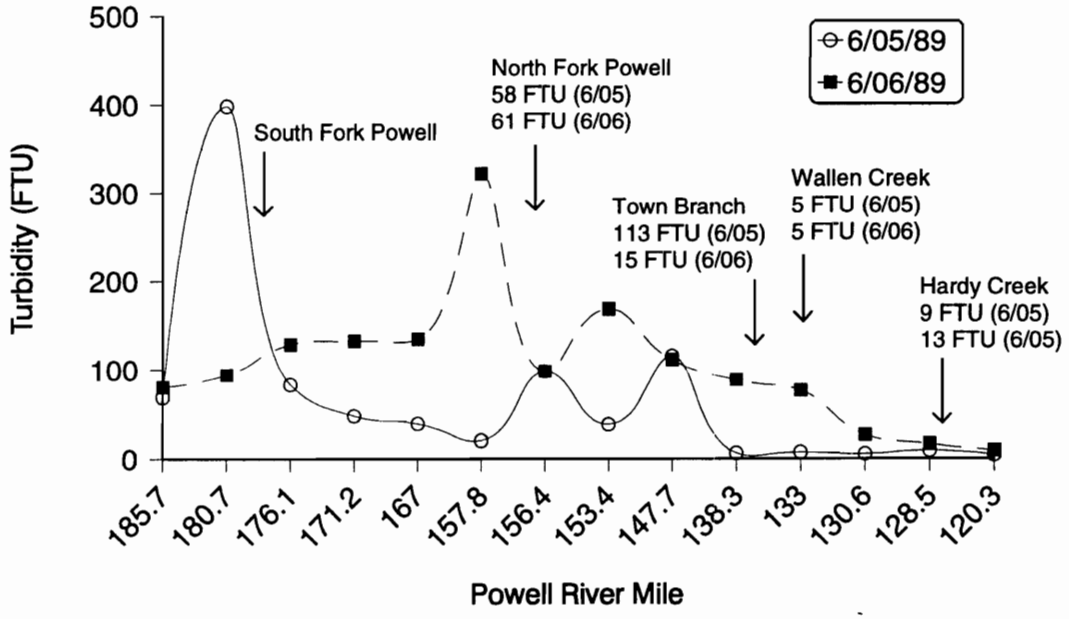

Figure 17. **Turbidity regimes on two consecutive days during a spate (6/05/89 and 6/06/89) within the 4th, 5th, and 6th order Powell River in Virginia. Discharge measured at the U.S.G.S. gauging station at PRM 143.5 was 425 CFS (6/05/89) and 1760 CFS (6/06/89).**

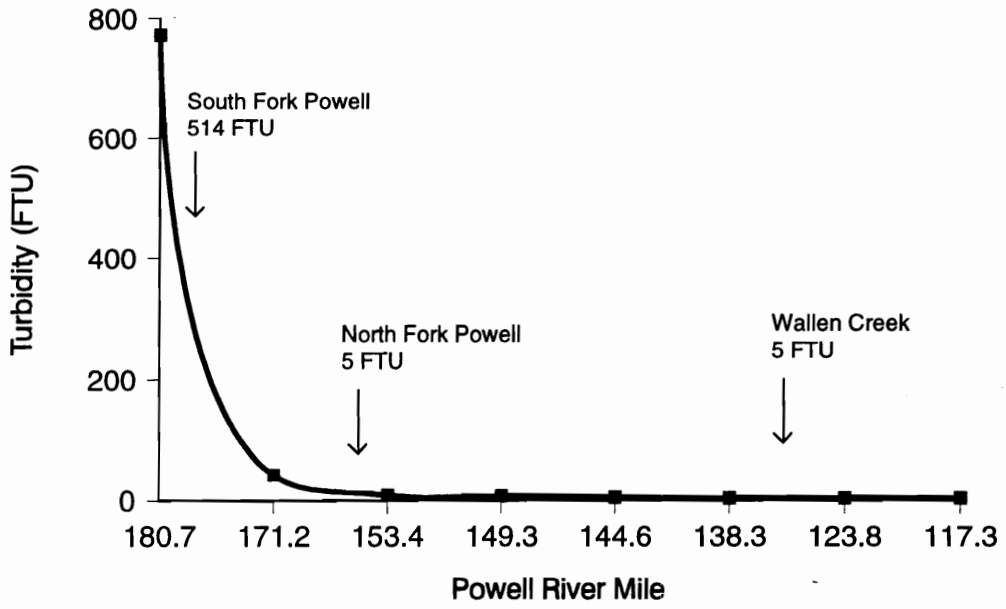

Figure 18. **Turbidity regime during a rain event concentrated in the upper watershed (8/21/89) within the 4th, 5th, and 6th order Powell River in Virginia. Discharge measured at the U.S.G.S. gauging station at PRM 143.5 was 120 CFS.**

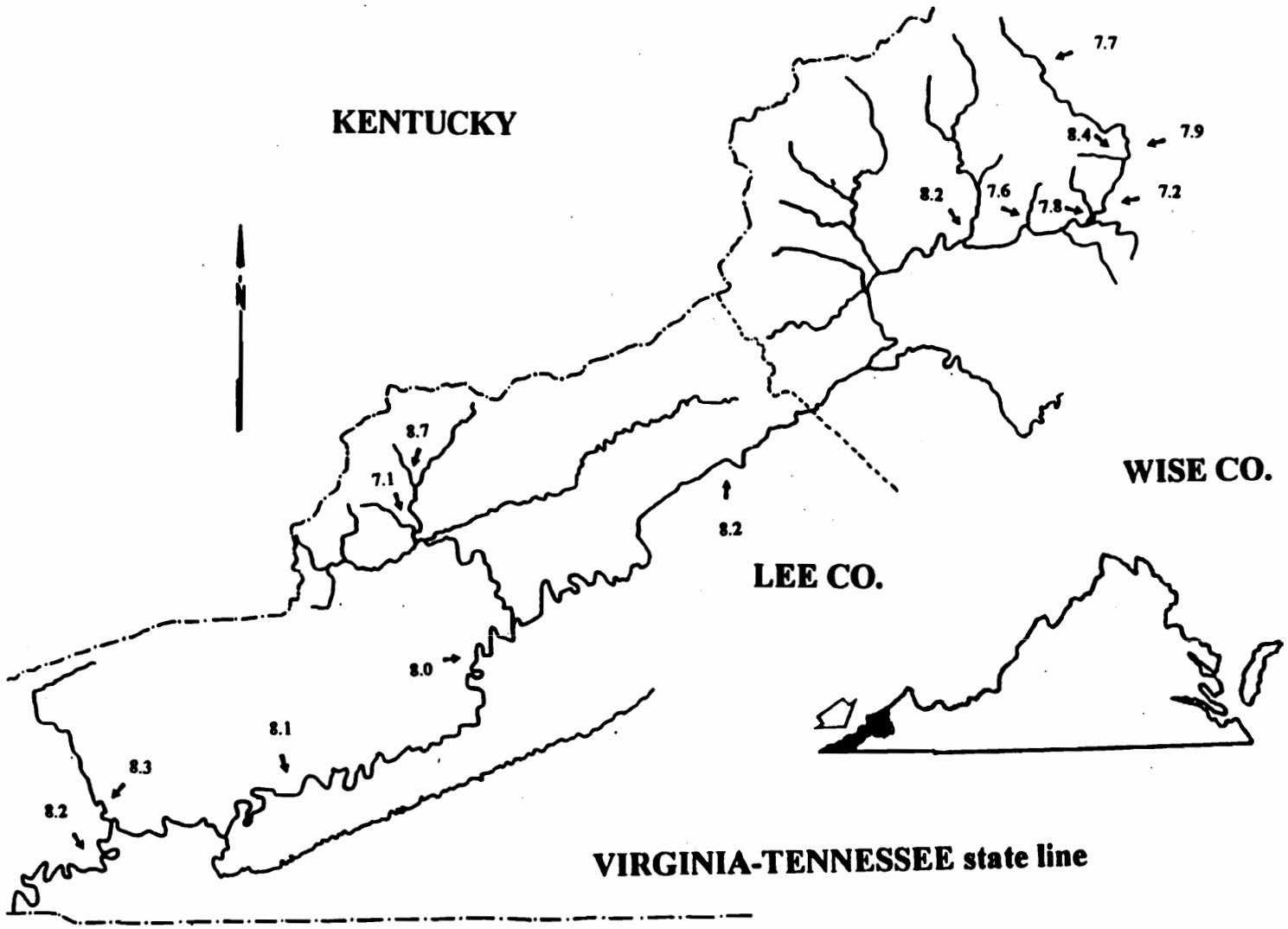

Figure 19. pH pattern in the Powell River drainage in Virginia; sample dates are 2/27/90 and 3/22/90.

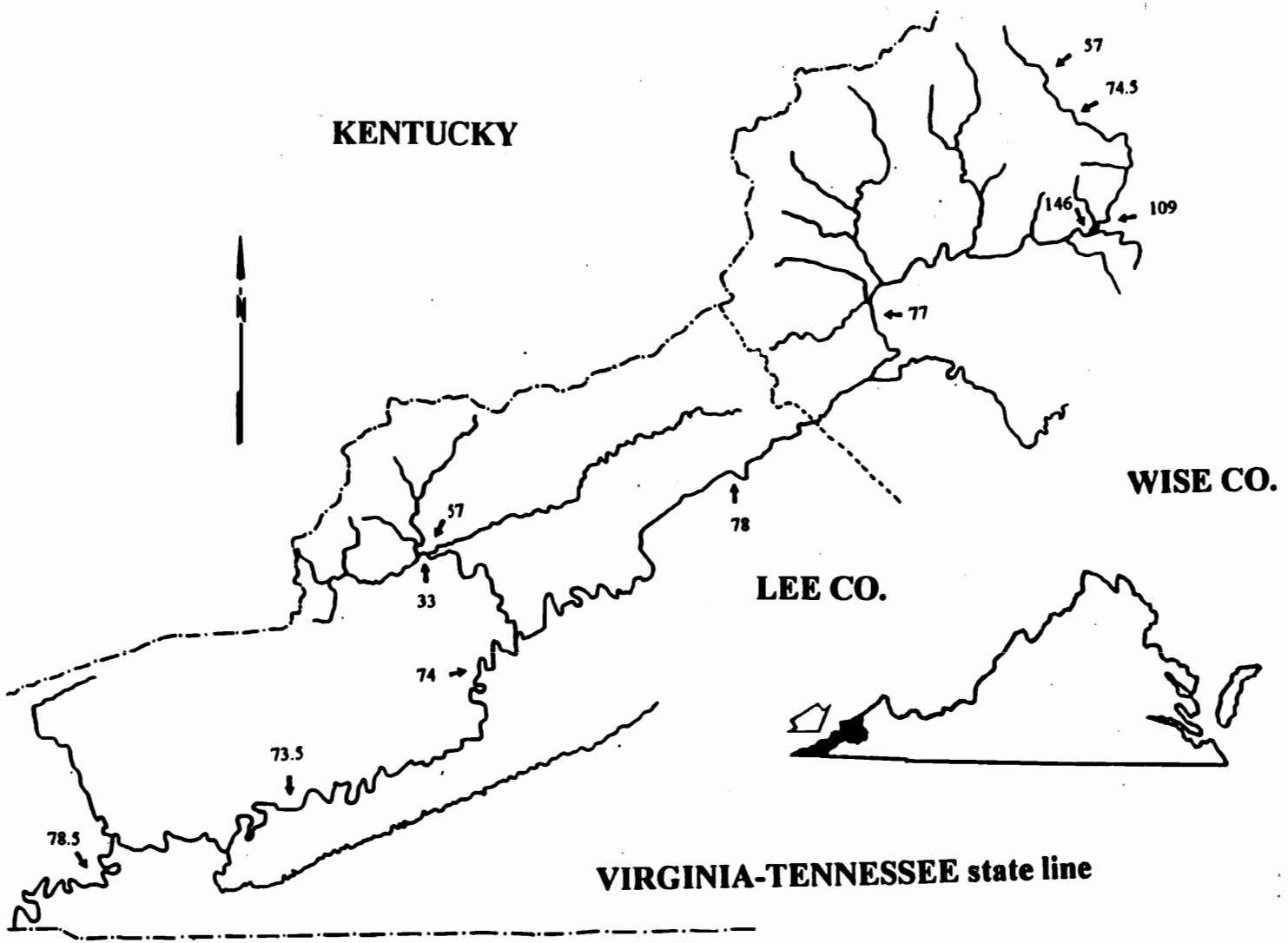

Figure 20. Alkalinity pattern in the Powell River drainage in Virginia; sample dates are 2/27/90 and 3/22/90.

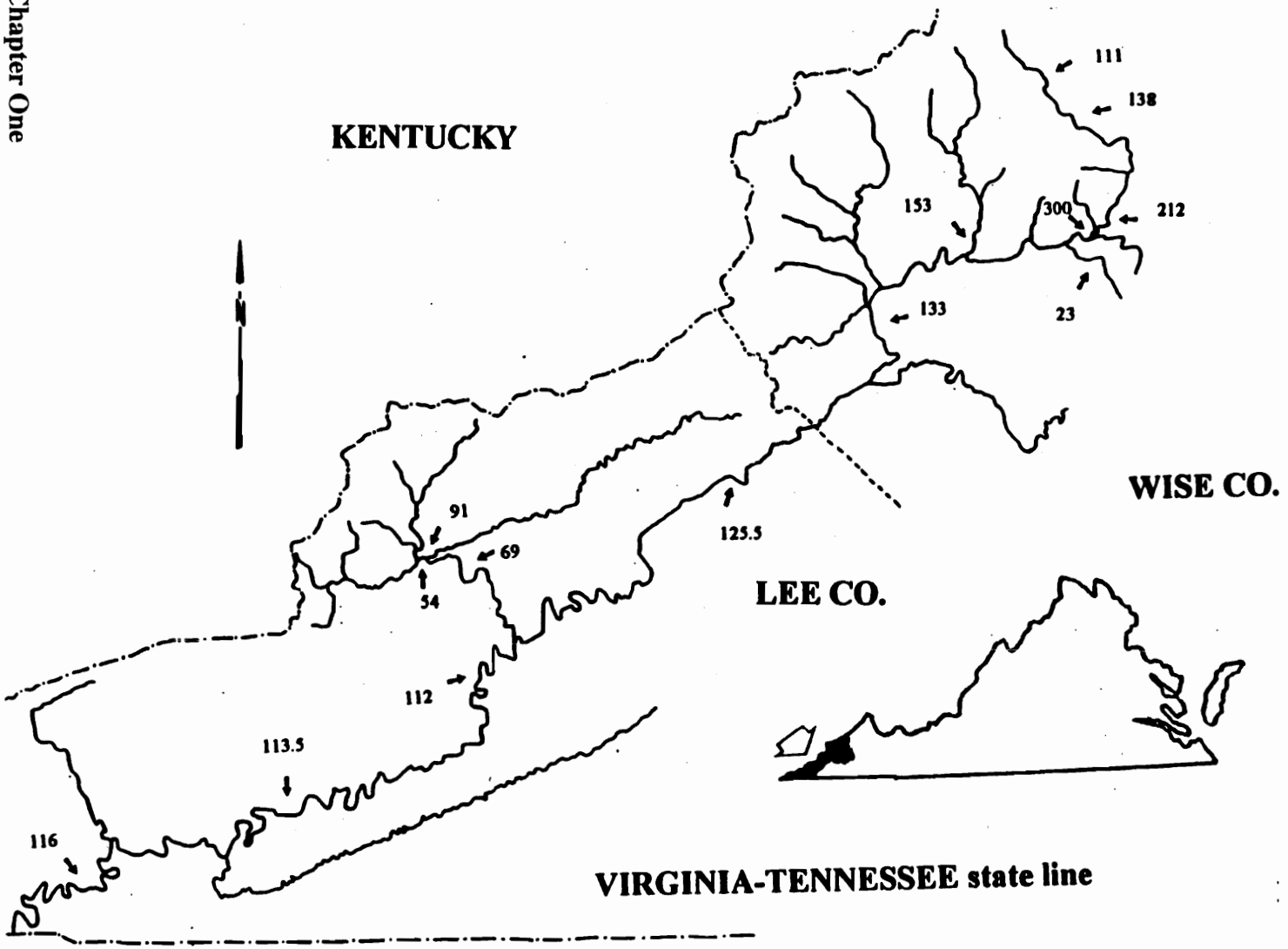

Figure 21. Hardness pattern in the Powell River drainage in Virginia; sample dates are 2/27/90 and 3/22/90.

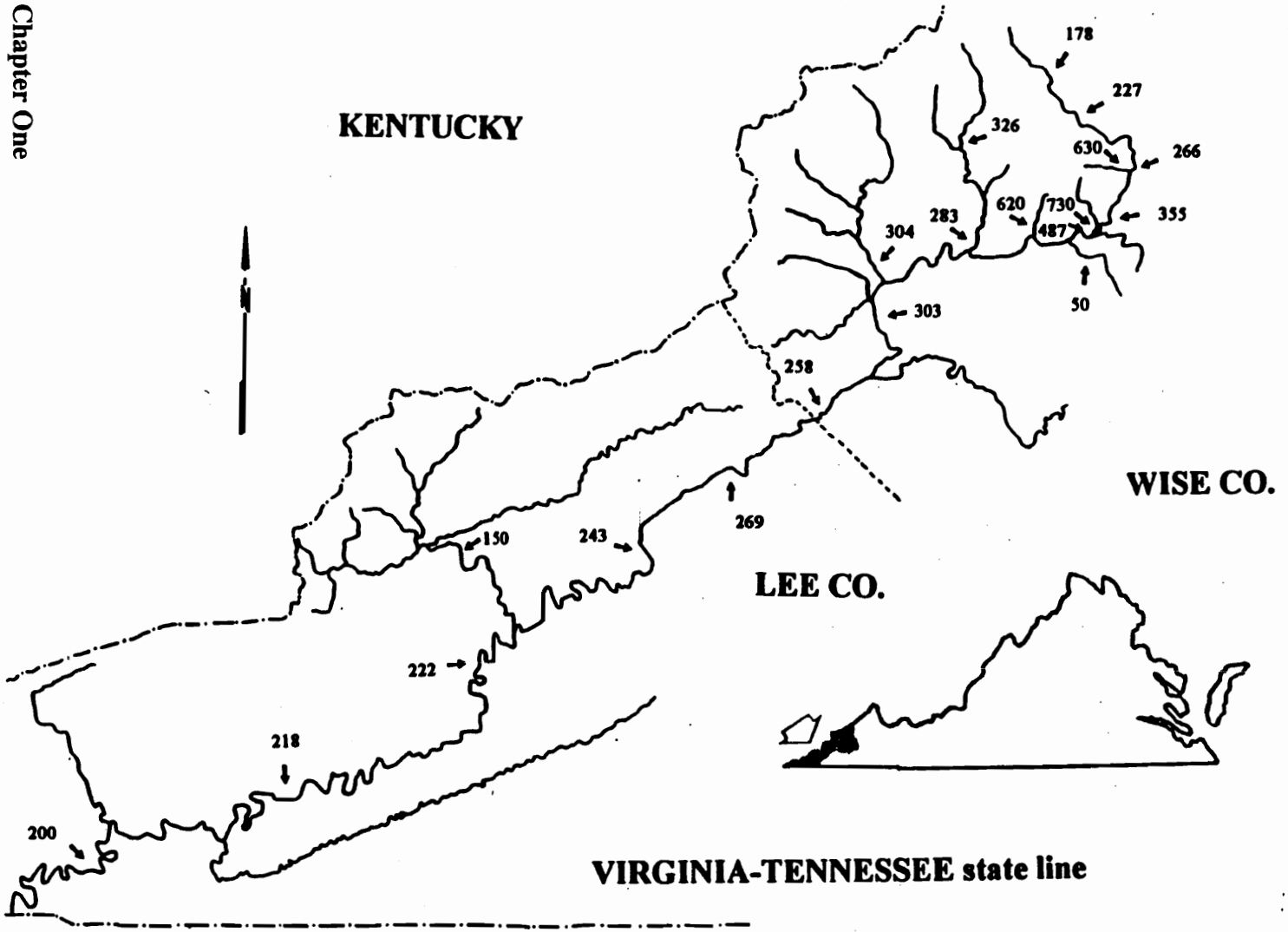

Figure 22. Specific conductivity pattern in the Powell River drainage in Virginia; sample dates are 2/27/90 and 3/22/90.

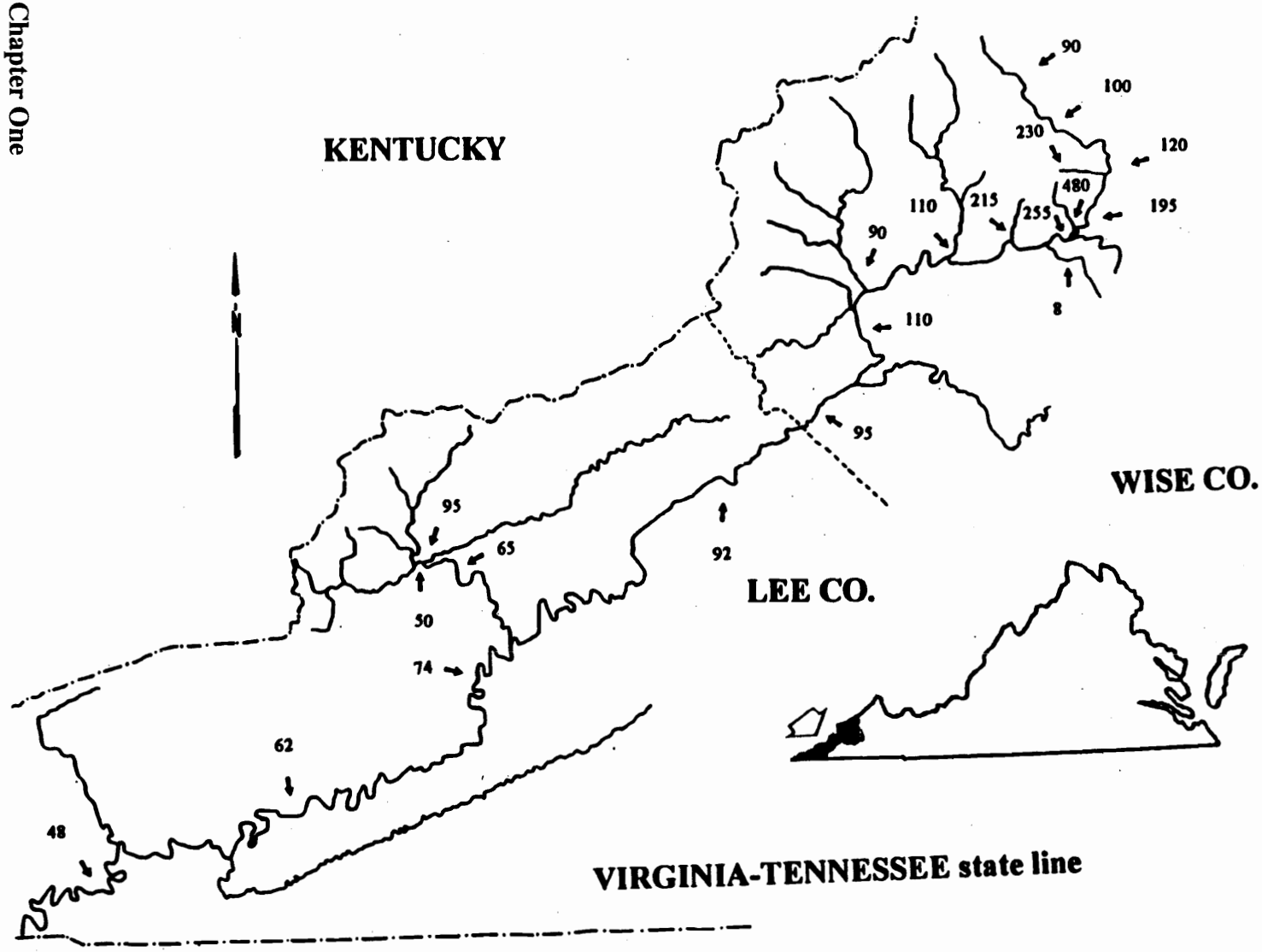

Figure 23. Sulphate pattern in the Powell River drainage in Virginia; sample dates are 2/27/90 and 3/22/90.

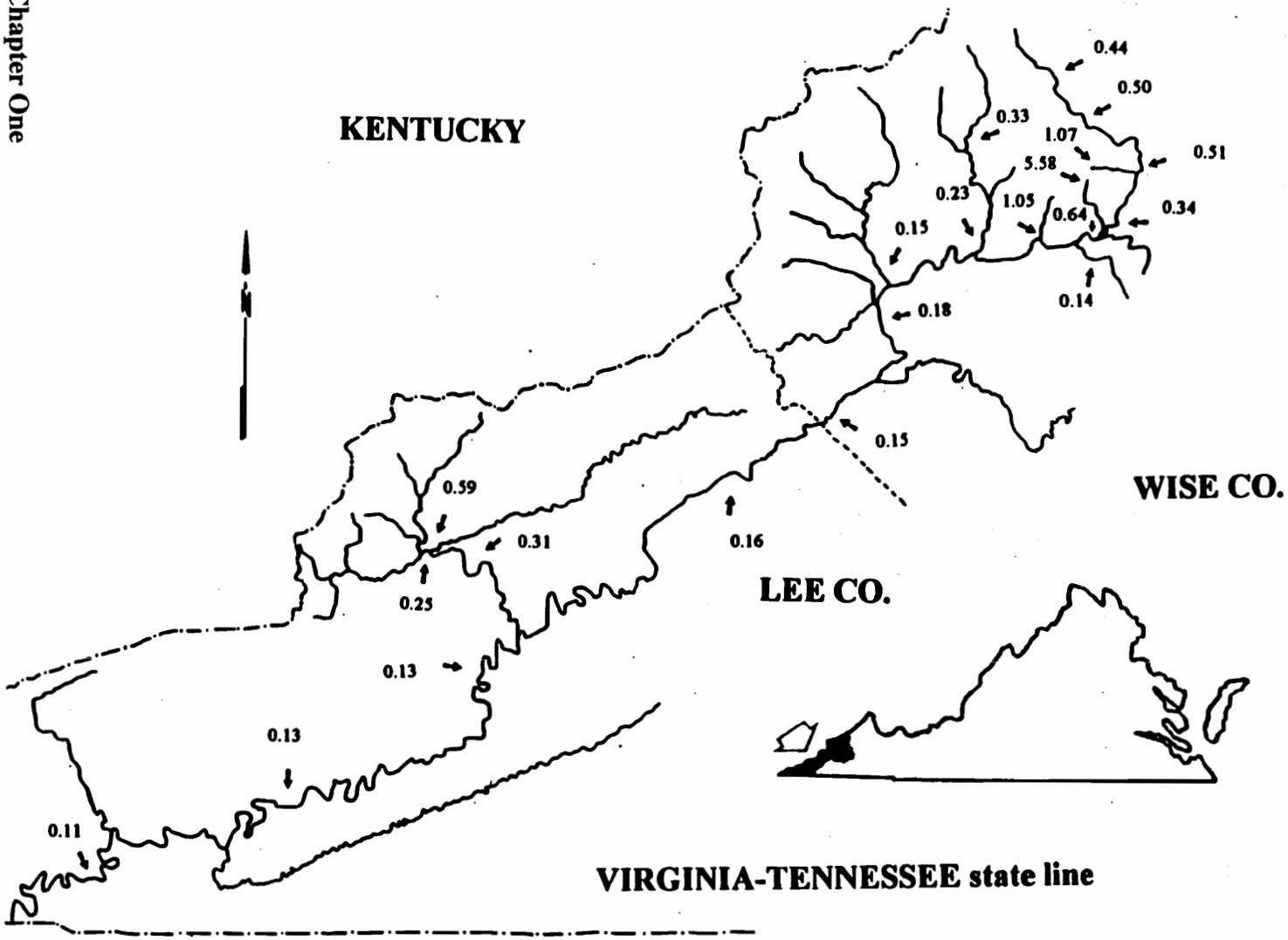

Figure 24. Iron pattern in the Powell River drainage in Virginia; sample dates are 2/27/90 and 3/22/90.

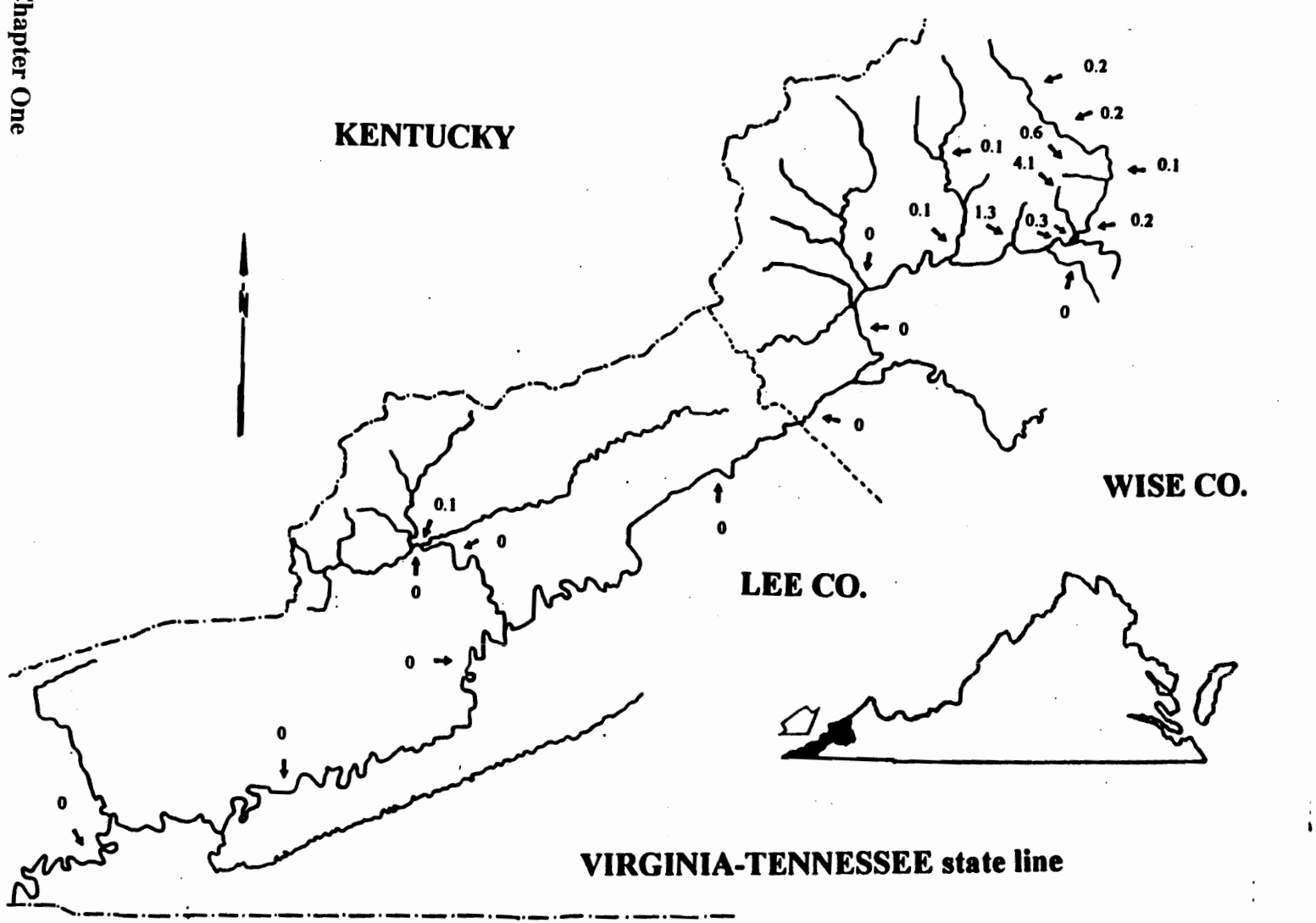

Figure 25. Manganese pattern in the Powell River drainage in Virginia; sample dates are 2/27/90 and 3/22/90.

Creek (and Ely Creek). Although not measured on 2/27/90, pH means for major PR tributaries are South Fork Powell River = 8.18, North Fork Powell River (NFPR) = 7.85, and Hardy Creek\Wallen Creek\Town Branch = 8.17 (Appendix B).

All alkalinity readings (Figure 20) were in the moderate to high range (APHA 1975; Boyd 1979). A slight depression in alkalinity below the NFPR confluence is likely due to the influence of the NFPR (alkalinity mean = 47.0; Appendix B). Overall, drainage buffering capacity against pH depressions appears high. Hardness ranged from hard to very hard in the MTRIBCP (Figure 21), moderately hard in the PR mainstem (Valley and Ridge Physiographic province), to soft in UMTRIBCP and in the NFPR drainage (hardness categories in Boyd 1979). Hardness increases in the upper PR are due to input by mined tributaries such as Thacker Branch and Roaring Fork (see Appendix B). Hardness decreased as the PR entered the Valley and Ridge Physiographic province and underwent a slight depression below the NFPR confluence (again, due to the diluting effects of the NFPR).

Specific conductivity indicates high ion concentrations in MTRIBCP within Wise County (Figure 22). The high conductivities of Bear Branch, Thacker Branch, Black Creek, and Callahan Creek stand in stark contrast to the specific conductivity of a unmined Cumberland Plateau tributary, Carding Machine Branch (50 µsiemens/cm). The PR specific conductivity increased due to these sources and then gradually decreased from the Cumberland Plateau-Valley and Ridge boundary to the Virginia-Tennessee state line. The NFPR specific conductivities are much lower than the MTRIBCP within Wise

County (see also Appendix B).

Sulphate exhibit a nearly identical pattern to specific conductivity (Figure 23), with highest values in the MTRIBCP within Wise County, lower readings from the NFPR drainage, and gradually decreasing concentrations from the Cumberland Plateau-Valley and Ridge boundary to the state line. Iron concentrations likewise are highest in the MTRIBCP within Wise County and decrease in the PR mainstem to the state line (Figure 24). Iron concentrations are elevated in the NFPR drainage as well. Manganese has its highest readings in the MTRIBCP within Wise County but rapidly drops off to non-detectable levels (Figure 25).

Of note is the contrast between water quality of the extreme upper PR mainstem draining inactive mines (water quality stations: PRM 199.3, 198.9, and 198.0) versus the upper PR mainstem (water quality stations: PRM 197.5, 195.3, 194.4, and 192.6) draining both active and inactive mines (Table 12). The reach draining inactive mine lands exhibited lower means for specific conductivity ($p \leq 0.024$) and sulphate ($p \leq 0.012$). Calcium, hardness, calcium:hardness ratio, iron, and manganese levels also appeared lower in the upper reach. These relationships, however, were not statistically substantiated (all $p \leq 0.05$), possibly due to small sample sizes. Finally, pH was similar between the two reaches. These findings, although again implicating coal mining as the primary source for many contaminants, suggest that some recovery of water quality in streams within the PR drainage can be expected when mines are deactivated.

Likewise, Dickens et al. (1985) observed that sediment and mineral constituent

Table 12. Means ± standard deviations of measured water quality variables for all sample dates at reach PRM 199.3 - 198.0 (three stations) and reach PRM 197.5 - 192.6 (four stations). Reach PRM 199.3 - 198.0 drains inactive mine lands only. Reach PRM 197.5 - 192.6 drains both active and inactive mine lands. (See Methods for actual dates). n = number of samples.

Water Quality Variable	PRM 199.3-198.0	PRM 197.5-192.6
Calcium (mg/L)	51.7 ± 12.9 n = 3	101.8 ± 46.4 n = 8
Specific conductivity (µsiemens/cm)	177 ± 31 n = 6	319 ± 141 n = 11
Hardness (mg/L)	100 ± 28 n = 3	229 ± 94 n = 6
Iron (mg/L)	0.42 ± 0.03 n = 2	0.50 ± 0.11 n = 5
Manganese (mg/L)	0.15 ± 0.07 n = 2	0.22 ± 0.08 n = 5
pH (units)	7.88 ± 0.43 n = 4	7.73 ± 0.35 n = 4
Sulphate (mg/L)	84.2 ± 16.5 n = 5	158.0 ± 65.7 n = 10

concentrations began to decline approximately 1.5 yr following mining cessation. Iron and manganese decreased to below federal effluent standards within 1.5 yr post-mining (although calcium and manganese levels remained elevated). Sediment concentration in storm runoff decreased below federal effluent standards by the end of the third year post-mining. Curtis and Superfesky (1977) found that 90 % of soil loss observed over a 20-mo period occurred within the first 12 mo following mining. Becker et al. (1986) found that old-mine lands (15-20 yr post-mining) were intermediate in water quality to newly mined lands and unmined lands. These results indicate that, while suspended and dissolved constituents decrease from inactive mine lands, elevated levels of dissolved constituents usually continue for several years after the session of mining (Dyer and Curtis 1977). Inactive mined lands probably remain a potential problem in the PR drainage, however. Many water quality variable concentrations are still above background. In addition, Sagona and Carroll (1991) determined that 31 percent of potential erosion from mine lands in the PR drainage comes from reclaimed mine lands (constitute 60% of total mined-land area) and about 5% from unreclaimed mine lands (11% of total mined-land area).

In summary, it appears that surface and deep coal mining is contributing elevated levels of specific conductivity, iron, manganese, sulphate, and turbidity to the river. Mined lands may be responsible for some elevated input of polyvalent metal ions (other than iron), as indicated by the calcium:hardness ratio and by the one-time survey of metal ions. Elevated manganese concentration and pH depressions appear localized.

Regarding potential monitoring variables, manganese appears to be a good indicator of the presence of local coal mining leachates; however, the rapid dropoff in concentration limits its usefulness as a variable for monitoring coal mining affects to mainstem PR water quality (Figure 25). Alkalinity and hardness also have limitations as monitoring variables for the PR, since the Valley and Ridge Physiographic province (limestone and dolomite geology) naturally contributes to concentrations of these variables (Allison and Palmer 1980). Calcium:hardness ratios, however, may serve as a viable indicator of metal concentrations. Instead, specific conductivity, sulphate, iron, and turbidity would be better variables for monitoring coal mining influences on mainstem PR water quality. Optimally, other priority heavy metals (e.g., aluminum, cadmium, copper, and zinc) also should be included in a monitoring program.

Water Quality Effects on Fish

The fifth and sixth order reaches differ in water quality; the upstream reach exhibited higher specific conductivity, iron, sulphate, turbidity, and total solids. Water quality degrades further during spates. Could the observed water quality variation differentially affect fish community structure? Using data for the entire sampling period, where applicable, all water quality values met Virginia state water quality standards (VSWCB 1988, 1989). In addition, the National Sanitation Foundation's Water Quality Index (NSF-WQI_m; Ott 1978) was used to score the PRM 171.2 and 138.3 stations for non-spate sample dates: 9/10/89, 11/15/89, 12/15/89, 2/28/90, 3/22/90, and 5/13/90

(BOD and fecal coliform data for nearby VSWCB stations on each sample month was obtained from the VSWCB). NSF-WQI_m scores for PRM 138.3 were higher than PRM 171.2 ($p \leq 0.031$; Table 13). Higher fecal coliforms at PRM 171.2 were primarily responsible for these differences. Despite consistently lower scores at PRM 171.2, however, all scores fell within the good water quality range listed in Ott (1978).

The water quality of the two reaches during spates was not evaluated by the NSF-WQI_m. Analyses of spate flows by the index may indicate lower water quality since

Table 13. Water quality scores for PRM 171.2 and PRM 138.3 for six dates using the National Sanitation Foundation's Water Quality Index (NSF-WQI_m). Biological oxygen demand and fecal coliform data were obtained for nearby Virginia State Water Control Board monitoring stations from that agency.

Date	PRM 171.2 NSF-WQI_m	PRM 138.3 NSF-WQI_m
9/10/89	73.4	77.0
11/15/89	75.3	81.7
12/15/89	75.8	79.3
2/28/90	70.6	73.2
3/22/90	73.3	77.5
5/13/90	72.1	76.5

dissolved oxygen and pH decline and nitrates, phosphates, turbidity, and total solids increase during spates. Quantitative scores may not change enough to lower qualitative descriptions; however, because fecal coliforms, a variable with a high index weight, was found not related to discharge in the Powell River by Brede and Benham (1996).

Temperature, another potential factor affecting fish distribution, was characterized for the fifth (PRM 171.2) and sixth (PRM 123.8) order PR mainstem (Table 14). Mean temperatures were slightly higher ($\leq$ 2 °C; $p \leq 0.0218$) and diel fluctuations less ($p \leq 0.0001$) at PRM 123.8. It is doubtful, however, that these small differences in temperature dynamics have significant biological affects. In conclusion, considering all water quality indicators, there is no compelling evidence to suggest that water quality differentially affected fish assemblages within the lower PR.

Trends in Water Quality Variables

Four water quality variables sampled at or near PRM 143.5, three associated with coal mining (sulphate, iron, and specific conductivity) and one associated with sewage treatment and livestock (fecal coliforms), were plotted over the entire period of record. Sulfate levels are apparently elevated after 1975 compared to pre-1956 (Figure 26). These elevated sulphate levels coincide with increased surface mining in the PR watershed (Figure 7). Smith et al. (1987) likewise found sulphate levels highly correlated with surface coal production. Interestingly, they found no correlation of sulphate concentrations with undergound coal production.

Iron concentrations also are elevated post-1974 as compared to pre-1956 (Figure 27). The rather dramatic decline during the 1980's and 1990's may be due to mined-land reclamation; particularly the use of settling or catchment ponds. Iron is much less soluble than sulfate and likely more prone to precipitation in settling or catchment ponds.

Table 14. Mean daily temperature (°C) and mean daily difference in temperature extremes at Powell River Mile (PRM) 171.2 and PRM 123.8. Data were obtained from continuous recording thermographs.

Time Period	PRM 171.2		PRM 123.8	
	Daily Temperature (mean ± S.D.)	Daily Difference (mean ± S.D.)	Daily Temperature (mean ± S.D.)	Daily Difference (mean ± S.D.)
11/15/89-11/30/89	-	-	8.2 ± 1.6	1.2 ± 0.4
12/01/89-12/31/89	-	-	3.0 ± 2.2	0.8 ± 0.4
1/01/90-1/18/90	-	-	6.1 ± 1.1	0.9 ± 0.4
3/01/90-3/31/90	9.1 ± 2.2	1.9 ± 1.0	10.6 ± 2.0	1.3 ± 0.7
4/01/90-4/23/90	10.1 ± 1.7	2.6 ± 0.9	12.1 ± 1.4	1.9 ± 0.6
5/20/90-5/31/90	14.9 ± 1.1	1.9 ± 0.8	16.7 ± 0.8	1.4 ± 0.6
6/01/90-6/30/90	19.2 ± 1.9	1.9 ± 0.7	21.0 ± 1.8	1.7 ± 0.5
7/01/90-7/31/90	22.0 ± 1.3	2.8 ± 1.0	22.4 ± 1.7	1.6 ± 0.8

Figure 26. **Sulphate trends for the Powell River. The mean, 95% confidence interval, and sample size are depicted for each time interval. Data obtained from the U.S. Geological Survey, the Virginia State Water Control Board, and my study. Data obtained during 1930-31 collected at PRM 156.4. Data contributed by my study (1989 and 1990) collected at PRM 138.3. Remainder of data collected at PRM 143.5.**

Figure 27. **Iron (total) trends for the Powell River. The mean, 95% confidence interval, and sample size are depicted for each time interval. Data obtained from the U.S. Geological Survey, the Virginia State Water Control Board, and my study. Data obtained during 1930-31 collected at PRM 156.4. Data contributed by my study (1989 and 1990) collected at PRM 138.3. Remainder of data collected at PRM 143.5.**

Specific conductivity appears elevated during the 1980's and 1990's (Figure 28). Higher levels again may reflect increased surface mining in the PR watershed. The high mean values of specific conductivity and iron in the late 1970's and early 1980's may indicate the period of heaviest water quality degradation in the PR. Finally, fecal coliforms exhibited a dramatic decline in the 1980's (Figure 29). Analyzed by Zipper et al. (1992), fecal coliform declines at PRM 143.5 during 1970-1989 were highly significant ($p \leq 0.0001$). Significant declines in fecal coliforms also occurred at the PRM 180.8 state water quality station. Smith et al. (1987) attributed nationwide decreases in fecal coliforms to improved sewage treatment. This is likely the case as well for the PR mainstem, since Zipper et al. (1992) found significant declines in BOD at PRM 143.5. Livestock inputs therefore are probably inconsequential. Considering that the major factor of score reduction in the NSF-WQI_m was elevated fecal coliform concentrations, and that the VSWCB classifies the PR mainstem as "effluent limited", declining fecal coliforms should result in improved water quality.

Other hopeful signs for PR mainstem water quality are the significant declines that occurred in total suspended solids as well as the stable levels of pH and total phosphorus at the PRM 180.8 and 143.5 stations from 1970-1989 (Zipper et al. 1992). Congruent with my study findings on sulphate and specific conductivity, however, Zipper et al. (1992) noted significant increases in total dissolved solids at both stations. In addition, PRM 143.5 had significant increases in total Kjeldahl nitrogen since 1970 (Zipper et al. 1992). Hence, although water quality appears good in the lower

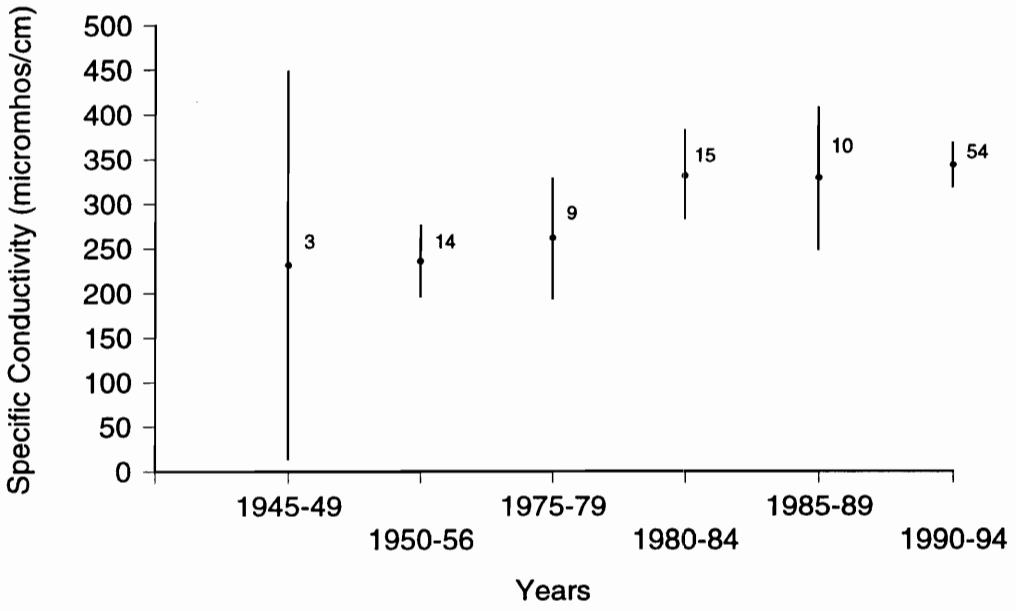

Figure 28. **Specific conductivity (at 25 °C) trends for the Powell River. The mean, 95% confidence interval, and sample size are depicted for each time interval. Data obtained from the U.S. Geological Survey, the Virginia State Water Control Board, and my study. Data contributed by my study (1989 and 1990) collected at PRM 138.3. Remainder of data collected at PRM 143.5.**

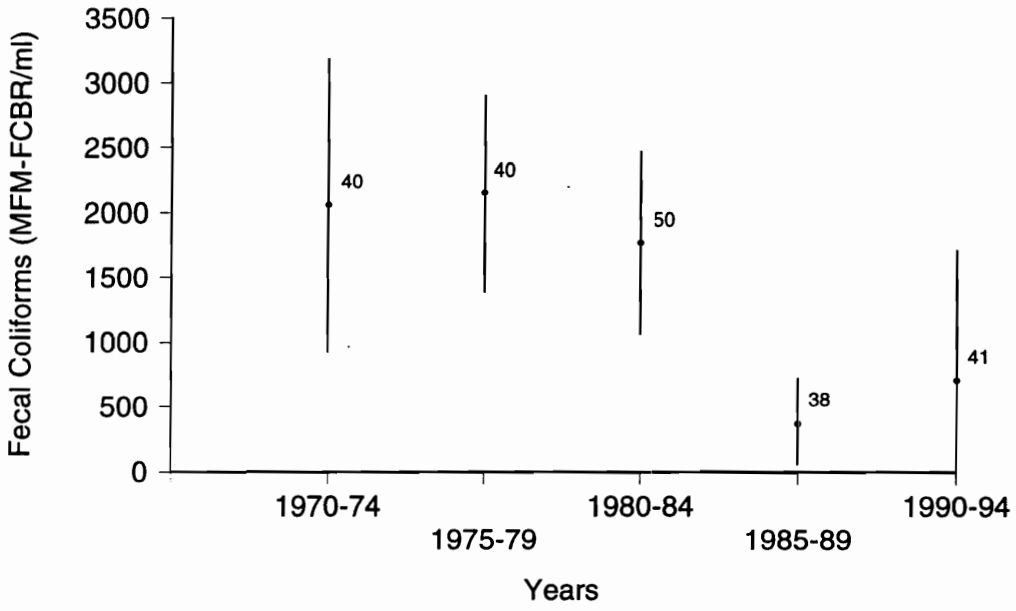

Figure 29. **Fecal coliform trends for the Powell River. The mean, 95% confidence interval, and sample size are depicted for each time interval. Data collected at PRM 143.5 by the Virginia State Water Control Board.**

PR (TDHE-DWPC 1994; VDEQ 1996; present study), slow changes in important water quality variables may be occurring.

Sedimentation is another pollutant that has been implicated as affecting the Powell River mainstem; unfortunately, data do not exist to enable the evaluation of long-term trends in sediment loading. Data collected in this study, however, determined that turbidity was significantly correlated with iron ($\rho = 0.66$, $p \leq 0.007$) and with sulphate ($\rho = 0.39$, $p \leq 0.012$) in the sixth order PR. Turbidity was not correlated with specific conductivity ($\rho = 0.03$, $p \leq 0.915$). Since turbidity should be a good surrogate of sedimentation (turbidity vs. total suspended solids, $\rho = 0.97$, $p \leq 0.0001$) and since turbidity has a fairly strong positive relationship to iron, it is reasonable to assume that the historical sedimentation loading trend follows that of iron. If so, sedimentation input has declined after the mid-1970's, again possibly due to mined-land reclamation practices. This conclusion is supported by Zipper et al. (1992), in their analysis of the trend of total suspended solids from 1970-1989 at PRM 180.8 and 143.5.

To conclude this section, I reviewed arsenic, cadmium, chromium, copper, mercury, lead, manganese, aluminum, and nickel water quality violations at PRM 143.5 from 1970-1987 (STORET data base). Toxic metals were chosen because of the suspected potential of periodically high inputs of these metals into the PR. The usefulness of this investigation was limited because hardness, a variable in all equations to determine standards (VSWCB 1988b, 1989), was not always measured concurrently with metals. Zinc was the only metal with a standard that violated its standard (one time).

However, if it is assumed that hardness at the sample date approximates mean hardness for that year, zinc would have violated its standard once more (n = 30), copper probably twice (n = 30), and lead probably a minimum of four times (n = 28). (The lower detection limit for lead likely was above the lead standard).

An interesting result was the high value for copper (10 µg/L) found on the first sample date (3/30/70). This value exceeds that found to be deleterious to juvenile mussels (Jacobson 1990). The mussel fauna has been declining in the Powell River (Wolcott 1990), and metal contamination is suspected of contributing to this decline (McCann 1993). Since monitoring for metals was not very comprehensive during the 1970's and 1980's, a more intensive metal monitoring program is needed to help protect the mussel fauna (and probably portions of the fish fauna) in the PR (McCann 1993).

Monitoring Indicators

It appears that coal mining, a significant land use in the PR watershed, is primarily responsible for contaminant loading into the PR. Suggested monitoring variables that appear to be good indicators of coal mining affects on the PR system are specific conductivity, sulphate, iron, and turbidity. Turbidity, in addition, was highly correlated with total suspended solids ($\rho = 0.97$, $p \leq 0.0001$). Thus, turbidity may be a good surrogate measure of sediment loading. The metals zinc, copper, lead, and cadmium (or the surrogate calcium:hardness ratio measurement) should be frequently monitored as well. Finally, fecal coliforms, due to numerous violations of state standards (Brede and

Benham 1996) and its perceived importance to water quality (Ott 1978), should be monitored as an indicator of wastewater treatment plant compliance and manure runoff from pastures and feedlots.

Physical Habitat

Sample site lengths averaged 570 m (range: 389-709 m). There does not appear to be a strong potential for biotic zonation in the lower PR based on differences in physical habitat (excluding sedimentation). Habitat variables measured in riffles, runs, and pools were examined for longitudinal trends (Table 3). All sites had the same gradient in riffles and runs (1 %; Cummins 1994). In riffle and run habitats, there were no correlations between PRM and an index of substrata (substrata principal component one) ($p \leq 0.406$), an index of cover (cover principal component one) ($p \leq 0.088$), velocity median ($p \leq 0.546$) and variance ($p \leq 0.224$), depth median ($p \leq 0.339$) and variance ($p \leq 0.765$), canopy closure ($p \leq 0.308$), and stream width ($p \leq 0.546$).

Embeddedness data from the nine sites were analyzed by a principal components analysis to derive the embeddedness index. Embeddedness variable loadings on principal component one, which accounted for 83.4% of the data variation, were category one (0.769), category two (-0.264), category three (-0.327), category four (-0.481), and category five (-0.033). This principal component described a high to low sedimentation gradient in the data, and the corresponding scores were used as the embeddedness index for each site. Embeddedness, in contrast to other physical habitat variables, varied

longitudinally (vs. PRM, $\rho = 0.89$, $p \leq 0.001$). Sedimentation was significantly higher upstream. Hence, except for sedimentation, riffle and run habitats in the lower PR appeared physically similar. The primary physical habitat gradient in riffle and run habitats was sedimentation.

Pool habitats also lacked longitudinal gradients in most habitat features. No longitudinal gradients were observed in velocity median (vs. PRM, $p \leq 0.531$) and variance ($p \leq 0.450$), depth median ($p \leq 0.381$) and variance ($p \leq 0.306$), substrata (principal component one vs. PRM, $p \leq 0.088$), cover (principal component one vs. PRM, $p \leq 0.308$) and canopy closure ($p \leq 0.606$). As in riffle and run habitats, sedimentation levels increased upstream (sediment depth vs. PRM, $\rho = 0.92$, $p \leq 0.001$). Sedimentation level had the strongest longitudinal gradient of the measured pool physical habitat variables. Finally, pool width increased downstream ($\rho = 0.72$, $p \leq 0.030$).

The findings of no significant longitudinal patterns in velocity, depth, and substrate also were concluded by Temple et al. (1990), who used a nonparametric coefficient of variation index. These results were significant since depth, velocity, and substrata (along with zoogeographic processes and water quality [Gilbert 1980; Hocutt and Wiley 1986]) are assumed to be primary factors for species richness patterns (Gorman and Karr 1978; Meffe and Sheldon 1988; Moyle 1994). Since the shallow-water habitat template (excluding sedimentation) does not vary longitudinally, fish assemblages in riffles and runs should be very similar throughout the lower PR (between PRM 171.8 and 117.3) (Hendricks et al. 1980).

Conversely, a sedimentation gradient existed in riffle and run habitats from high upstream to lower levels downstream. Embeddedness appears to be the primary physical gradient present in riffle and run habitats. This sedimentation gradient could affect fish assemblage patterns (Rabeni and Smale 1995).

The habitat variable in pools exhibiting the strongest longitudinal gradient was sediment depth. Highest sedimentation levels in pools were upstream, congruent with embeddedness results in riffles and run habitats (riffle\run embeddedness vs. pool sediment depth, $\rho = 0.75$, $p \leq 0.020$).

In summary, riffle, run, and pool habitat features do not appear to vary longitudinally except for sedimentation levels. In addition, measurements of shallow-water and pool sediment levels were congruent (riffle\run embeddedness vs. pool sediment depth, $\rho = 0.75$, $p \leq 0.020$). The upstream to downstream sedimentation pattern appeared to be the primary physical gradient in the lower PR from PRM 171.8 to 117.3.

Sources of Sedimentation

Watershed areal percents in urban and surface mine areas were highly, positively correlated with sediment depth ($\rho = 0.93$, $p \leq 0.001$ and $\rho = 0.92$, $p \leq 0.001$, respectively). Urban area and surface mine area as predictor variables of sediment level were subjected to stepwise multiple linear regression to ascertain the best predictor. Surface mine land area was the only variable retained in the model (Sediment depth = -29.99 + 1.94 [% surface mine land], $R^2 = 0.75$). Hence, surface mining is implicated as

the primary source of sedimentation in the 5th and 6th order PR. These results are congruent with the water quality findings reported previously. This conclusion also agrees with the results of Sagona and Carroll (1991); namely, that surface mine lands have the potential to contribute the highest level of sediment to streams relative to any other land use type in the PR watershed.

Chapter Summary

Primary land use types in the Virginia PR sub-watershed are forest (63%), rowcrop and pastoral agriculture (27%), surface mines (7%), and urban areas (2%). Forest lands, urban areas, and surface mine land percentages decrease downstream, whereas cropland and pasture land uses increase downstream.

The 6th order PR is classified hydrologically as intermediate between Mesic Groundwater and Perennial Runoff stream types (after Poff and Ward 1989). Coal mining has affected the hydrograph, and therefore the disturbance regime, of the sixth order PR. Flow variance has been reduced. This reduction appears relatively inconsequential, however, since biotic control of assemblage organization may be increased, as opposed to decreased, by reduced environmental stochasticity (Allan 1995). In addition, the reduced flow variance also would not change the PR hydrologic classification according to Poff and Ward (1989).

Water quality is impacted primarily by coal mining operations. Powell River tributaries having mined watersheds in the Cumberland Plateau exhibited elevated

concentrations of specific conductivity and sulphate relative to unmined tributaries in the Cumberland Plateau. They have higher specific conductivity, sulphate, and turbidity and lower alkalinity than unmined drainages in the Valley and Ridge physiographic province, and elevated sulphate, iron, manganese, turbidity and lower calcium compared to sixth order PR. The similarity in pH among the sub-watershed categories indicate that acid mine drainage is not a widespread problem. A water quality gradient existed in the lower PR. Specific conductivity, iron, sulphate, turbidity, and total solids levels were higher upstream and gradually diminished downstream. Finally, significant metal loading from mined-land tributaries into the PR may be occurring as indicated by reduced calcium:hardness ratios upstream, increased variation in specific conductivity explained by iron concentrations upstream, and upstream-to-downstream, high-to-low concentrations of zinc, copper, and cadmium.

During spates, calcium, specific conductivity, and sulphate levels decrease in the MTRIBCP. Alkalinity, nitrates, turbidity increases and dissolved oxygen and pH decrease during spates within the lower PR. In addition, hardness, phosphorus, total solids increase and the calcium:hardness ratio decreases in the sixth order PR during floods. Hence, water quality may improve in the MTRIBCP, due possibly to dilution. Water quality appears to degrade somewhat in the lower during spates, due possibly to re-suspended contaminants.

Aside from these findings, water quality appears good throughout the lower segment of the PR. No violations of Virginia water quality standards occurred during the

sampling period. Moreover, the NSF-WQI_m characterized the lower reach water quality as "good", despite significantly lower quantitative index scores in the fifth order. Finally, lower temperatures and higher daily temperature fluctuations in the fifth order do not appear strong enough to differentially affect fish populations.

Temporal analyses indicated that sulphate is increasing, tracking the increasing coal production. Iron has declined during the 1980's and 1990's, possibly due to mined-land reclamation practices. Specific conductivity, although elevated in the 1980's and 1990's, is holding relatively constant. Fecal coliforms exhibited a large decline during the 1980's. Sedimentation loading pattern may be similar to iron since turbidity shows a strong positive relationship to iron concentrations. If patterns are correlated, sediment input has declined since the mid-1970's, possibly also due to reclamation practices. Additional evidence for this trend was provided by Zipper et al. (1992), who found a significant decline in suspended solids since 1970. Temporal analysis of several metals for frequency of criteria violation indicated that, while few violations of state water quality standards have occurred since 1970, metal concentrations have the potential to be elevated and should be monitored closely.

Priority variables to monitor in a biomonitoring program include specific conductivity, sulfate, iron, turbidity, and fecal coliforms. These variables exhibit upstream to downstream gradients, have either frequently violated state standards (fecal coliforms, iron) or are good indicators of coal mining inputs. The heavy metals cadmium, copper, lead, and zinc (and the polyvalent metal ion surrogate calcium:hardness ratio)

also should be monitored for similar reasons. Metal contamination in streams, however, may be more accurately determined by sediment analysis as opposed to water column analysis (McCann 1993; Yeager 1994).

The primary physical habitat gradient in the lower PR is sedimentation, with high levels upstream and diminishing downstream. Variation in sediment level is primarily explained by extent of the watershed comprised by surface mining. Because of the objectivity and continuous nature of the data, sediment depth in pools should be the primary physical habitat variable monitored in a biomonitoring program.

In conclusion, there does not appear to be a significant basis for biotic zonation within the lower segment of the Powell River due to chemical or physical factors (excluding sedimentation). A distinct upstream-to-downstream, high-to-low sedimentation gradient in pools and shallow-water habitats could, however, change fish assemblage structure.

Chapter Two

Factors affecting fish assemblage structure and function patterns.

Introduction

Excessive fine sediments (particle size < 2 mm) in streams is a widespread and common phenomenon (Judy et al. 1984; USEPA 1990), often leading to impairment of aquatic organism populations and assemblages (Waters 1995). In many cases, therefore, stream monitoring protocols must be sensitive to sedimentation-induced changes to biota. A critical aspect in the development of monitoring programs is the selection and validation of indicators (Noss and Cooperrider 1994; Noss 1995). Indicators are ecological elements or processes that serve as surrogates for biodiversity components of interest, such as sedimentation or water quality.

A stream monitoring program should evaluate biotic, chemical, and physical habitat indicators of the ecosystem in question (Winget and Mangum 1979; Plafkin et al. 1989; Loeb and Spacie 1994). Biotic indicators, at least, should always be included (Ohio EPA 1988a,b; Angermeier and Karr 1994; Moyle 1994). The Index of Biotic Integrity (IBI) is a suite of biotic indicators that is commonly used to evaluate the integrity of stream fish assemblages by comparing target assemblage characteristics to regional benchmarks (Paller et al. 1996). The IBI serves as a tool for comparative community ecology, largely because fish assemblage characteristics are investigated in

taxonomic and functional terms (e.g., species richness and trophic guild metrics, respectively; Karr et al. 1986).

Using functional attributes theoretically permits comparisons of fish assemblages dissimilar in taxonomic organization (Schoener 1986, 1987; Rabeni and Smale 1995). This approach relies on the premise that species possessing particular traits are predictable at a location based on habitat characteristics (Southwood 1988; Cummins 1994; Townsend and Hildrew 1994). Thus, for instance, trophic metrics in the IBI are based on the expectation that degraded stream environments are less trophically predictable, thereby favoring generalistic trophic traits over specialized ones.

Despite the common use of the IBI, it is regarded as a preliminary management tool to detect impairments to fish assemblages, not as a diagnostic tool for identifying specific causes (Bayley and Li 1992). In addition, the broad-based nature of the IBI may reduce its ability to detect subtle changes wrought by sedimentation. For example, Rabeni and Smale (1995) derived a siltation index composed of three functional metrics (two trophic and one reproductive). They considered it more responsive to sedimentation effects on warmwater fish assemblages in Missouri streams than the IBI. Although the IBI has been shown to indicate sediment-induced changes to fish populations (Karr et al. 1986), there is considerable need to further evaluate the efficacy of the IBI for detecting effects due to sedimentation.

A critical problem in evaluating the IBI for detecting sedimentation effects on warmwater fish assemblages is the dearth of knowledge regarding warmwater assemblage

responses to sedimentation (Berkman and Rabeni 1987; Rabeni and Smale 1995; Waters 1995). It is imperative to identify changes in assemblage taxonomic and functional structure caused by sedimentation. Information on taxonomic changes can be used to identify species that are sediment-intolerant or sediment-tolerant, and to use them for monitoring indicators in particular lotic systems. Information on functional changes can be compared with other studies to investigate whether sedimentation results in consistent patterns of effects on warmwater fish assemblages.

The lower Powell River is a biotically diverse, 5th and 6th order stream in southwestern Virginia that has a distinct sediment gradient (Chapter One). Excessive sedimentation has been implicated as affecting fish and aquatic invertebrate assemblages in this river (Wollitz 1972; Ayers 1981; Dennis 1981; Heffinger 1986; Wolcott 1990). A study was undertaken to assess sediment-induced changes to taxonomic and functional structure of fish assemblages and to derive biotic indicators for monitoring the Powell River. Major questions asked are

1) has sedimentation changed taxonomic and functional structure of fish assemblages in the Powell River and, if so, in what manner;
2) does the IBI detect these changes; and
3) which functional or structural elements can serve as indicators for monitoring the effects of sedimentation to warmwater streams.

Methods

Sampling for the Ibi and for Fish Assemblage Attributes (1988)

Fish were collected from nine sites on the lower (5^{th} and 6^{th} order) Powell River in Virginia during August 1988. Physico-chemical characteristics of these sites and river segments were described in Chapter One. Mean site length was 570 m (range: 389-709 m). For standardization purposes, sampling procedure followed that of Saylor et al. (1988). At each site, predominant habitat types (i.e., riffle, run, stream margin, sidepool, and pool) were identified. Riffle, run, and pool habitats are described in Bisson et al. (1982). Sidepools are near-shore depositional areas, with lower velocity and greater depths than adjacent riffle and run habitats.

Fish collections were accomplished by a variety of methods and gears. In riffles and runs, a 6.1 m × 1.8 m (20 ft × 6 ft) seine of 4.8 mm (3/16 in) mesh was used in conjunction with a gas-powered backpack electrofishing unit. Alternating current (AC) was used for all electrofishing. The electrode system was two hand-held booms with diamond-shaped electrodes. An area approximately 5 m × 5 m was shocked in a downstream direction. The backpack operator turned over stones with his feet as he electrofished. Stunned fish were captured in a stationary seine positioned on the downstream edge of the shocked area. Sidepools were sampled by seining approximately 5 m × 5 m areas. Five minute backpack shocking runs were used to sample stream margins. Pool habitats were sampled with 15 min efforts using a lightweight

electrofishing boat (AC waveform, 3500 watt generator, Wisconsin Ring boom electrodes, boat wired as the other electrode). Both pool shorelines and the pool thalweg were sampled.

To increase the probability that a representative fish sample in each habitat type was obtained, sampling continued until a minimum of three consecutive seine hauls, backpack-seine shocking efforts, 15 min boat shocking efforts, and backpack shocking efforts produced no new species. Hence, at least four efforts or samples per habitat would be conducted unless no fish were caught (which never happened). An exception to this protocol occasionally would be made if all available area in a habitat type had been sampled. Finally, uncommon habitats (e.g., aquatic plant beds) not sampled in the aforementioned strategy were electrofished for qualitative information. All specimens, except centrarchids and catostomids, were preserved in 10% formalin and transported to the lab.

All captured fish were sorted by species, counted, and measured. Occurrences of disease, anomalies, hybridization, or external parasites were noted for each specimen. Because of the different capture techniques employed, species abundances are expressed as relative abundances (%) in shallow-water and pool habitats. Stated differently, species numbers from all capture techniques within shallow-water or within pool habitats are combined, and relative abundances for each species are calculated. The shallow-water (riffle, run, sidepool, and stream margin habitats) fish sample for PRM 158.3 was lost. Therefore, eight sites were used for site-wide and for shallow-water assemblage

evaluations. Nine sites were retained for pool habitat assemblage analyses.

Biotic integrity of site fish assemblages was assessed by using the IBI (Karr and Dudley 1981). IBI metrics and scoring criteria followed that of Saylor et al. (1988), except for two minor modifications (Table 15). Catch rate was determined from riffle habitats only. Riffles often are the most productive riverine habitat, and effort was easily quantified. Secondly, the incidence of "black spot" parasite was dropped. "Black spot" is caused by the metacercariae of several digenean helminth genera (Noble and Noble 1976). These parasites, as other animals, have abundances that vary across their native distributions (Price and Clancy 1983; Brown 1984). To my knowledge, metacercarial abundances have not been shown to be negatively correlated with biotic integrity as measured by fish assemblage attributes. This change left metric 12 tallying only with incidence of tumors, fin erosion, and other anomalies (as spinal deformities).

Distributional status, tolerance, and trophic group classifications for each species captured are listed in Appendix E (after Saylor et al. 1988; Saylor and Ahlstedt 1990). One modification was made to Saylor et al. (1988) trophic group classifications. Blotched chub (*Erimystax insignis)* was reclassified as a specialized insectivore, the same classification given to the streamline chub (*E. dissimilis*) by Saylor and Ahlstedt (1990). Compared to an omnivorous congener, *E. harryi*, these species have a relatively short gut, indicating greater trophic specialization (Harris 1986). Moreover, Harris (1986) observed that blotched chub consumed a greater proportion of aquatic insects than the streamline chub.

Table 15. Index of Biotic Integrity metrics and scoring criteria used to evaluate biotic integrity of the Powell River, Virginia.

Metric		Scoring Criteria 1	3	5
1	Total number of native fish species	< 21	21-41	> 41
2	Number of darter species	< 4	4-8	> 8
3	Number of sunfish species, less *Micropterus*	< 2	2-3	> 3
4	Number of sucker species	< 2	2-4	> 4
5	Number of intolerant species	< 2	2-4	> 4
6	Proportion of individuals as tolerant species	> 20%	20-10%	< 10%
7	Proportion of individuals as omnivores	> 30%	30-10%	< 10%
8	Proportion of individuals as specialized insectivores	< 25%	25-50%	> 50%
9	Proportion of individuals as piscivores	< 2%	2-5%	> 5%
10	Catch rate (average number of individuals captured per unit effort in riffle habitats)	< 8	8-16	> 16
11	Proportion of individuals as hybrids	> 1%	1% -Tr*	0%
12	Proportion of individuals with disease, tumors fin damage, and other anomalies	> 5%	5-2%	< 2%

* Tr = Value < 0.5%

Scores for each of the 12 metrics are summed to derive an overall IBI score for a site. Qualitative classifications of biotic integrity from IBI scores (Karr et al. 1986) are "Excellent (58-60), Good (48-52), Fair (39-44), Poor (28-35), and Very Poor (12-22). IBI metrics used in this investigation do not use reproductive groups for assessment of biotic integrity.

Reproductive groups also were investigated so that, in concert with trophic group analysis, probable mechanisms of impairment to fish assemblages by human-induced environmental changes could be inferred. I derived reproductive categories are listed in Appendix E. Since the primary physical gradient in the lower Powell River is sedimentation (Chapter One), reproductive categories reflected perceived or demonstrated susceptibility to impairment by sedimentation. Sources of fish reproduction information used as background material were Muncy et al. (1979), Page (1983), Berkman and Rabeni (1987), Etnier and Starnes (1993), Jenkins and Burkhead (1994), and Rabeni and Smale (1995).

The first level dichotomy was whether eggs were demersal or floating. Only drum, *Aplodinotus grunniens*, had floating eggs (group = **PSFE**). Floating eggs were assumed to be unaffected by sedimentation. The second level dichotomy was presence or absence of parental care. Parental care activities included keeping the eggs silt-free as opposed to just nest guarding (as in *Nocomis* spp.). Egg clusterers and clumpers (Cottidae, Ictaluridae, some cyprinids) and depression nesters (Centrarchidae) are classified within this group (group = **PC**).

Species that do not have parental care were further divided into species that deposit their eggs above the stream bottom grade (group = **ASL**), and those that deposit their eggs at or below the average substrate level (groups **SPD**, **LB**, and **MB**). The ASL group is considered more susceptable to sedimentation than the PC group but, due to being above the bottom grade, are afforded some protection from interstitial and bedload movement of sediments. This potentially compensatory behavior is observed in several darter species in the southeastern United States that live in depositional habitats and attach their eggs to upright aquatic plants (Page 1983). Included in group ASL are egg attachers (some percids), crevice spawners (*Cyprinella* spp.), and *Nocomis* chubs. Egg attachers adhere eggs to plants or to the top or sides of boulders. Crevice spawners often deposit eggs in crevices of submerged logs or boulders. Mound nest building by *Nocomis* chubs is well known and results in an initally sediment-free gravel mound for spawning. Several cyprinid species (Appendix E) commonly spawn over *Nocomis* sp. nests and, as such, the eggs probably encounter the same physical environment as do *Nocomis* sp. eggs.

The final discrimination occurs for species that deposit their eggs at or below the substrate grade. Three reproductive groups are distinguished here. Those species that do some site preparation or disturbance are classified as the **SPD** group. These species include those that build pits (Petromyzontidae, *Campostoma anomalum*), redds (*Moxostoma carinatum*), dig in the substrata (*Luxilus chrysocephalus*) and bury eggs in the substrata (many percids). Their activities are assumed to result in sediments being dislodged and washed away, leaving cleaned substrata for egg deposition. This would

afford some protection from sedimentation. Since parental care is absent, however, sediments cannot be prevented from reinvading and surrounding the deposited eggs. It is assumed that cleaned egg deposition sites are infiltrated by sediments more quickly in sites with higher sedimentation levels.

The final two groups exhibit no site preparation. Species are classified as to whether they broadcast spawn over rocks (group = **LB**) or over miscellaneous substrate (e.g., mud, sand, rocks, vegetation, group = **MB**). Many cyprinids are broadcast spawners over rocks. The LB group is considered the most sedimentation-sensitive reproductive strategy. Most species within this group require sediment-free substrates to spawn, and they do not clean substrate before spawning or maintain clean substrata for egg incubation. MB group species include lepisosteids, clupeids, esocids, *Cyprinus carpio*, *Carpiodes cyprinus*, and *Labidesthes sicculus*. It is assumed that this group is adapted to spawning in depositional habitats and that moderate sedimentation would not lower reproductive success. In summary, the groups presented, in order of postulated sedimentation intolerance are LB > SPD ≡ MB > ASL > PC > PSFE.

Physical Habitat and Fish Sampling (1989)

Physical habitat and fish from shallow-water habitats were sampled at 12 sites during 1989 (Table 1; Figure 6). As in 1988, sample sites in 1989 included both the 5th and 6th order Powell River. Mean site length was 153 m (range: 95-197 m). Sampling occurred from 9 July to 3 August, 1989. At each site, predominant habitat types were identified in

shallow-water areas (Bisson et al. 1982). The transect-point sampling method was followed (Orth 1983), with transects randomly located in riffle and run habitats (stratified-random sampling). Sample points on each transect were spaced 9 m apart, beginning at 2 m from the shore. The bank for the nearshore point on the initial transect (the most downstream transect) was chosen randomly. The nearshore point on the succeeding transects alternated along each shore. The number of sample points per transect varied with transect width (usually between 3 to 5), and a minimum of 25 points were sampled at each site.

At each sample point, fish were sampled in a similar manner as in 1988 except that an area approximately 4 m × 6 m was shocked. All captured fish were sorted by species and enumerated. Typically, fish sampling was completed within one day per site.

Measured habitat variables included two site-wide variables, one habitat variable, and eight microhabitat variables. Site width was recorded as the average width of all transects. Canopy overhang was characterized as the average extent that the canopy overlapped all transects by each bank. Upon completion of fish sampling, the habitat type (riffle, run, or sidepool) and microhabitat was characterized at each sample point (located at the center of each 4 m × 6 m sample area). Microhabitat measurements taken were depth, average water-column velocity, substrata, embeddedness, and the presence or absence of water willow (*Justicia americana*), filamentous algae, large woody structure (e.g., logs) and detritus. Depth was measured to the nearest 1.0 cm. Velocity measurements (cm/sec) were made with an electronic current meter and wading rod

following the procedure outlined in Orth (1983). Substrate categories followed the modified Wentworth scale (Cummins 1962): boulder (>256 mm), cobble (64-256 mm), pebble (16-64 mm), gravel (2-16 mm), sand (0.0625-2 mm), and silt (<0.0625). A seventh substratum category, bedrock, also was recorded. Substratum sampling procedure was modified from the method of Bain et al. (1985b). For each sample point, at each of 10 dm marks on a meter stick, the substrata category was identified. This technique did not use visual integration required by the methods of Bain et al. (1985b) or that of estimating dominant and subdominant substrata at a point. Embeddedness, an ordinal measure of sedimentation, was visually estimated at each point. The five embeddedness categories range from one (highest sedimentation) to five (lowest sedimentation) (Platts 1983).

Physical Habitat and Fish Sampling (1990)

Shallow-water habitats at 12 sites were sampled for physical habitat and fish during 12 June to 10 August, 1990 (Table 1; Figure 6). Unlike the previous two years, sites in 1990 were limited to the sixth order segment. Mean site length was 127 m (range: 45-180 m). At each site, predominant habitat types (i.e., riffle and run) were identified as well as an additional habitat type termed "head run". A head run is the transition zone from a pool to the shallow-water area. This habitat type has laminar flow with little or no surface turbulence. The transect-point sampling method again was followed (Orth 1983). Transects were randomly located in riffle, run, and head run habitats (stratified-random

sampling). There were typically six transects per site (10 sites had six transects, PRM 127.2 had seven transects, and PRM 117.4 had four transects).

A constant number of sample points per transect was used in 1990. These points were 1 m from the bank, one-third distance from shore, two-thirds distance from shore, and 4 m from the opposite bank. The bank for the nearshore point on the initial transect was chosen randomly and alternated for the succeeding transects. Four 4 m × 2 m pre-positioned area shockers (PPAS) were placed on each transect, one each at points referenced above. These units were allowed to sit undisturbed for 20 min. PPAS methodology was used because of the relatively high fish capture efficiency of this technique (Appendix L). In addition to high efficiency, the small area sampled with each PPAS was assumed to facilitate the determination of fish - microhabitat relationships. Fish sampling methods are discussed in the methods section of Appendix L and in Appendix M.

Habitat variables measured in 1990 were similar to those taken in 1989, with some refinements. Average width and canopy overhang at sites were calculated as in 1989. The habitat type was visually determined for each sample area. A set of microhabitat measurements was taken centered at each of the four corners of each 4 m × 2 m sample area. Habitat variables measured were depth, average water column velocity, substrate, and embeddedness. Areas of water willow (*Justicia americana*), pond weed (*Potamogeton* sp.), filamentous algae, simple logs, complex logs, root wads, root mats, woody debris, and overhanging vegetation within the sample area were measured.

Statistics

All analyses were performed either on the mainframe computer at Virginia Polytechnic Institute and State University using SAS (SAS Institute, Cary, North Carolina) or on a personal computer using MINITAB (MINITAB, Inc., State College, Pennsylvania). Significance level was set at $\alpha = 0.05$.

Prior to multivariate analyses, non-proportional habitat variables, fish species absolute abundances (1989, 1990), and IBI metrics were $\log_{10}(x + 1)$ transformed to stabilize variances (Green 1979). Proportional habitat variables (e.g., substrata and embeddedness), fish species relative abundances (1988), and IBI metrics were arcsine squareroot transformed to normalize the binomial distributions typical of proportional data (Zar 1984).

Except for fish-microhabitat relationship analyses, shallow-water (1988, 1989, 1990) and pool habitats (1988) at each site served as individual sampling units for fish abundances and physical habitat variables. Individual fish were used as the sampling units to assess differences in microhabitat use by species or species groups classified by their relative sediment tolerance (i.e., univariate tests weighted by fish abundance, Seber 1982).

Principal component analyses (PCA) using a covariance matrix were used to derive an index of embeddedness and substrata for each site in 1989 and 1990 (see Poff and Allan 1995 for a similar approach with hydrologic data). The first principal component, which accounts for the greatest amount of variation in the data, was retained and site

scores on the first principal component served as index values for site embeddedness or substrata. All other physical habitat variables were used as underived statistics for each site. Physical habitat in 1988 was summarized similarly except that values for several habitat variables comprising "cover" (Table 3) also were converted to an index by PCA (Chapter One).

Associations of IBI scores and river location (PRM) and sedimentation levels were assessed by using the Spearman's Rank Correlation Test (Hollander and Wolfe 1973). All correlations within this chapter were achieved by using the Spearman's Rank Correlation Test. Longitudinal patterns in IBI metrics were investigated by an initial PCA (covariance matrix) followed by correlation of meaningful principal components with PRM (see Harris 1975; Green 1979; Ludwig and Reynolds 1988; Smith et al. 1988). Meaningful principal components also were tested for correlation with sedimentation levels. In each year, all physical habitat variables were assessed for longitudinal patterns by correlation of habitat indices or underived variables with PRM.

Fish assemblage structure spatial patterns and potential causal factors in shallow-water and pool habitats were assessed by cluster analyses (average linkage, squared Euclidean distance, Pielou 1984). A cluster consisted of sampling sites that contained similarly structured fish assemblages. Species primarily responsible for defining interpretable cluster groups were identified by the magnitude of the F-statistic for each species (Green 1979). The F-statistics, which are the ratios of the among-groups to the within-groups variance of log-abundances, were used as indices rather than as

significance tests. Clustered groups also were examined for differences in assemblage structure as defined by trophic groups (Saylor and Ahlstedt 1990) and reproductive groups via Mann-Whitney U-test (MINITAB). Potential causal factors (physical habitat differences among clusters) for observed shallow-water faunal groups (clusters) were assessed by site by Mann-Whitney U-tests of habitat indices or single underived variables (Green 1979).

To identify sediment-intolerant and sediment-tolerant fish species in shallow-water and pool habitats, species proportional abundances (1988) or absolute abundances (1989, 1990) were subjected to PCA. All principal components accounting for variation in the data set were tested for significant correlation with sedimentation level. Species loadings on principal components, significantly correlated with sedimentation, were examined for high positive or negative loadings. Depending upon the sign of the principal component-sedimentation variable correlation coefficient (ρ), the sign and magnitude of the species loading was used to distinguish sediment-tolerant from sediment-intolerant species (Green 1979). Species generally distributed in the lower Powell River, but loading near zero on the correlated principal component, apparently were not affected positively or negatively by sedimentation. Abundances of species classified as sediment-intolerant (from all three sample years) were examined for their relationship to sedimentation through univariate Spearman's Rank Correlation tests (Hollander and Wolfe 1973).

Fish abundance principal components correlated with sedimentation also were examined for significant correlation with other physical habitat variables and with PRM.

This was done to ascertain whether other factors, in addition to sedimentation, may be influencing species abundances.

The fish abundance data from 1988 were evaluated with a PCA to determine whether habitat types had distinct associations of fish species and, if so, to assign habitat use to sediment-tolerance classified species. This information was used to interpret among year variation in sediment-tolerance classification, in view of the differences in habitat types sampled among years. In particular, pools were sampled during 1988 only, and side-pools were sampled most extensively during 1988. Sampling in 1989 and 1990 was primarily directed within riffle and run habitat types.

Final sediment-tolerance classifications were made by pooling information from 1988, 1989, and 1990. Confirmatory analyses were performed on these groups by using canonical discriminant function analysis (CDFA). The working hypothesis was that sediment-intolerant species should inhabit microhabitats with low levels of sedimentation whereas sediment-tolerant species, conversely, should inhabit microhabitats with higher levels of sedimentation. Non-classified species should select intermediate levels of sedimentation. Initially, CDFA was performed using all species to describe site microhabitat profiles and to investigate microhabitat use and partitioning among individual species. Differences in microhabitat use by the *a priori* designated sediment-intolerant and sediment-tolerant groups on canonical axis one (the axis that contains most of the data variation) was assessed by using the Kruskal-Wallis test (Hollander and Wolfe 1973) on group canonical scores (Grossman and Freeman 1987). Differences in

sedimentation of microhabitat utilized by the two groups were assessed with a t-test weighted by fish abundance (Seber 1982). Weighting by abundance incorporates the distribution of occurrence in microhabitats as opposed to only presence-absence information.

Results and Discussion

Fish and Physical Habitat Sampling- 1988

Index of Biotic Integrity Analyses

A total of 6,175 fish of 56 species was collected. The qualitative biotic integrity classification ("good") remained essentially constant across sites, except for PRM 174.4 (Tables 16 and 17, Appendix F). These qualitative results agree with that of the water quality assessment reported in Chapter One. Another implication is that the effects of sedimentation on fish assemblages, if any, were not severe.

IBI scores, conversely, increased downstream (Spearman's $\rho = -0.748$, $p \leq 0.033$). The trend of increasing IBI score with PRM is supported by Saylor and Ahlstedt (1990) who sampled PRM 65.4 in May, 1988, using similar scoring criteria to calculate an IBI score of 58 (within the "excellent" range). Saylor and Ahlstedt (1990) reported some coal fines but considered the site to have low sedimentation and good overall habitat diversity.

Several metrics varied with PRM (PRM vs. principal component two, $\rho = 0.905$, $p \leq 0.002$). Decreasing IBI scores upstream were largely a function of 1) increasing

Table 16. IBI results for each metric at each of eight sites on the Powell River in Virginia, sampled August 1988. Only native species were used to determine metric values (defined in Table 15).

	Metric Number											
Site (PRM)	1	2	3	4	5	6	7	8	9	10	11	12
174.4	35	6	4	5	4	32.4	32.1	34.2	3.9	13.4	0	0.90
163.4	34	5	3	5	3	18.1	16.9	58.4	4.6	53.1	0	0.09
153.4	40	7	3	4	5	17.2	16.1	58.4	3.2	86.9	0	0.10
146.8	39	7	2	5	4	22.1	20.9	41.7	7.7	37.4	0	0.16
144.6	38	7	3	5	5	12.0	11.8	64.5	2.8	50.0	0	0
123.0	36	6	3	5	3	10.7	9.1	63.3	10.7	17.4	0	0.65
119.3	40	7	2	5	6	15.6	13.2	59.4	3.7	41.2	0	0.24
117.3	39	8	3	4	7	7.9	7.9	59.0	12.5	25.7	0	0.38

Table 17. IBI scores and classifications for eight sites on the Powell River in Virginia, sampled during August, 1988.

Site (PRM)	IBI Score	Biotic Integrity Classification
174.4	40	Fair
163.4	48	Good
153.4	48	Good
146.8	46	Fair\Good
144.6	48	Good
123.0	50	Good
119.3	48	Good
117.3	54	Good

frequency of individuals as omnivores and as tolerant species and 2) decreasing frequency of individuals as piscivores, number of intolerant species, and proportion of individuals as specialized insectivores (Table 18). The number of darter species decreased slightly upstream. Finally, there were no longitudinal trends in number of native species, sunfish species (excluding *Micropterus*), sucker species, catch rate, hybrid proportions, and proportion of individuals with anomalies. The lack of a longitudinal trend in fish species richness was expected from the physical habitat similarity in the lower PR reported in Chapter One. This also is congruent with the observations that fish species richness may not increase significantly beyond stream order four or five (Hendricks et al. 1980) or that stream orders are not strong organizers of fish assemblages (Matthews 1986).

Although quantitative IBI scores were not correlated with sedimentation levels in shallow-water areas ($\rho = -0.48$, $p \leq 0.223$) or in pools ($\rho = -0.68$, $p \leq 0.066$), several IBI metrics were correlated with embeddedness (vs. principal component two, $\rho = 0.72$, $p \leq 0.045$) and with pool sediment depth ($\rho = 0.78$, $p \leq 0.022$) (Table 19). This discrepancy may be a result of sedimentation-affected changes primarily to the functional, as opposed to the taxonomic, structure of the fish assemblages in the PR. Moreover, the variation in two taxonomic metrics, "proportion of individuals as tolerant species" and the "proportion of individuals as intolerant species", may be an artifact because there were high overlaps between species used for these metrics and the functional metrics "proportion of individuals as omnivores" and "proportion of individuals as specialized insectivores", respectively (see species classifications in Appendix E). In their study of

Table 18. IBI metric variable loadings on the second principal component (accounted for 15.6 % of the data variation). Data obtained from the Powell River in Virginia (August 1988).

IBI Metric	Loading on Principal Component Two
Number of native species	-0.067
Number of darter species	-0.184
Number of sunfish species (excluding *Micropterus*)	0.102
Number of sucker species	0.097
Number of intolerant species	-0.311
Proportion of individuals as tolerant species	0.544
Proportion of individuals as omnivores	0.552
Proportion of individuals as specialized insectivores	-0.234
Proportion of individuals as piscivores	-0.432
Catch rate in riffle habitats	0.050
Proportion of individuals as hybrids	0
Proportion of individuals with anomalies	0.024

sedimentation effects on fish in several Missouri streams, Rabeni and Smale (1995) also observed that fish assemblage responses to sedimentation were more consistently described by functional characteristics (trophic and reproductive) as opposed to structural features as species richness.

Potential changes to functional organization of fish assemblages wrought by sedimentation were investigated in riffle\run and pool habitats. In riffle and run habitats, the relative abundance of omnivores increased with higher sedimentation ($\rho = 0.73$, $p \leq 0.040$) (Table 19). Although specialized insectivore relative abundance appeared lower in sites with higher sedimentation, this was not statistically substantiated ($\rho = -0.69$, $p \leq 0.056$). General insectivore ($\rho = 0.43$, $p \leq 0.286$) and herbivore ($\rho = 0.45$, $p \leq 0.257$) relative abundances were not correlated with embeddedness. Top carnivore and parasite trophic guilds were not evaluated because relative abundances of both guilds represented 2% or less of the trophic guild composition in riffle and run habitats.

The SPD group (spawning site preparation or disturbance) was the only riffle and run habitat reproductive guild statistically correlated with embeddedness, increasing in relative abundance in sites with lower sedimentation ($\rho = -0.89$, $p \leq 0.003$) (Table 20). The ASL (spawning above substrate grade; $\rho = 0.69$, $p \leq 0.056$) and the LB (lithophilic benthic spawners; $\rho = 0.65$, $p \leq 0.083$) guilds tended to increase in abundance at sites with higher sedimentation, although these trends were not statistically substantiated. There was no significant relationship between the PC (parental care) group and sedimentation ($\rho = 0.49$, $p \leq 0.217$). The PSFE (pelagic spawning, floating eggs) and the

Table 19. Trophic guild proportions (%) in all shallow-water habitats at eight sites on the Powell River in Virginia, sampled during August, 1988. Trophic categories after Saylor and Ahlstedt (1990).

	Trophic Group					
Site (PRM)	Specialist Insectivores	Omnivores	Generalist Insectivores	Herbivores	Piscivores	Parasites
174.4	45.73	36.75	5.56	10.68	0	0
163.4	65.45	17.84	7.27	8.40	1.02	0
153.4	69.04	14.65	5.61	9.81	0.76	0.13
146.8	59.77	20.38	2.84	15.88	1.90	0.24
144.6	78.16	10.49	6.03	4.60	0.71	0
123.0	83.66	0.99	5.45	4.46	4.95	0.50
119.3	72.15	11.89	5.05	10.75	0.16	0
117.3	73.26	6.17	7.46	6.68	6.17	0.26

Table 20. Reproductive group proportions in all shallow-water habitats at eight sites on the Powell River in Virginia, sampled during August, 1988. Reproductive categories are: PSFE = pelagic spawner with floating eggs; PC = eggs given parental care; ASL = eggs deposited above substrate level; SPD = eggs deposited in a prepared or disturbed site; MB = eggs deposited on miscellaneous benthic substrata; and LB = eggs deposited on rocky benthic substrata.

	Reproductive Group					
Site (PRM)	PSFE	PC	ASL	SPD	MB	LB
174.4	0	20.58	17.28	37.04	0	25.10
163.4	0	7.83	13.73	30.19	0	48.24
153.4	0	8.78	10.18	39.31	0.13	41.60
146.8	0	5.45	9.46	49.17	0	35.93
144.6	0	7.65	6.68	50.07	0	35.61
123.0	0	10.34	9.85	66.50	0	13.30
119.3	0	3.81	8.69	59.45	0.76	27.29
117.3	0	14.83	5.37	49.87	0	29.92

MB (miscellaneous benthic substrata spawners) reproductive guilds were not evaluated due to low relative abundances ($\leq$ 5 % at all sites).

From the standpoint of expected susceptibilities of the various reproductive guilds to sedimentation, it appears that sedimentation is not affecting fish assemblages through reproductive impairment. For instance, the LB group was assumed to be most susceptible to sedimentation of the six groups. Rabeni and Smale (1995) also observed that lithophilous spawners were the only reproductive guild reduced in abundance by sedimentation. If this assumption and research result are applicable to the Powell River, then sedimentation was not high enough upstream in shallow-water habitats to cause reproductive impacts to LB group fishes (and by extension, to all other reproductive group fishes). Instead, the primary avenue of impairment to shallow-water fish assemblages may be through trophic disruption.

In pool habitats, top carnivore relative abundance declined with higher sedimentation levels ($\rho = -0.66$, $p \leq 0.05$) (Table 21). The two most abundant top carnivores in the PR, rockbass (*Ambloplites rupestris*) and smallmouth bass (*Micropterus dolomeiu*) were deemed food-limited due to unsuitable pool substrata conditions for their primary prey item, crayfish (Cummins 1994). Pool substrata conditions were classified as poor in the rockbass and smallmouth bass Habitat Suitability Index models because sand was the dominant substrate (Cummins 1994). In contrast, specialized insectivore ($\rho = 0.00$, $p \leq 0.965$), omnivore ($\rho = -0.37$, $p \leq 0.323$), and general insectivore ($\rho = 0.49$, $p \leq 0.179$) trophic guild abundances did not vary with sedimentation levels. The parasite

Table 21. Trophic group proportions (%) in deep-water (pool) habitats at nine sites on the Powell River in Virginia, sampled during August, 1988. Trophic categories after Saylor and Ahlstedt (1990).

	Trophic Group					
Site (PRM)	Specialist insectivores	Omnivores	Generalist insectivores	Herbivores	Piscivores	Parasites
174.4	8.21	17.91	66.42	0	7.46	0
163.4	23.95	18.91	39.50	1.26	16.39	0
158.3	19.65	14.85	49.78	0.44	15.28	0
153.4	14.71	15.44	52.94	0	16.91	0
146.8	9.17	19.58	43.33	0	14.58	0
144.6	7.45	27.66	53.19	0	11.70	0
123.0	4.63	25.00	52.78	0	17.59	0
119.3	26.74	20.86	39.57	0	16.04	0
117.3	17.35	16.33	32.65	2.04	31.63	0

and herbivore trophic guilds were not evaluated due to low relative abundances ($\leq$ 5 %).

Relative abundances of reproductive guilds were not significantly associated with pool sedimentation levels (all $p \geq 0.05$) (Table 22). (The PSFE guild was not evaluated due to low relative abundance, $\leq$ 5 %, at all sites). The strongest relationship, an apparent decline of the SPD group in higher sedimented pool habitats ($\rho = -0.63$, $p \leq 0.071$), was similar to that noted for the SPD group in riffle and run habitats. In summary, trophic guilds, as opposed to reproductive guilds, appear most sensitive to sedimentation in PR shallow- and deep-water habitats.

Rabeni and Smale (1995) advocated a "siltation index" composed of trophic and reproductive functional groups in lieu of the IBI for monitoring the effects of sedimentation. From using the IBI on the PR, it also appears that functional metrics are affected more than taxonomic metrics by sedimentation. In addition, values of fish overall abundance, hybridization, and condition metrics were not associated with sedimentation levels. Hence, in agreement with Rabeni and Smale (1995), I recommend that assemblage-level monitoring indicators for detecting sedimentation impacts should include functional attributes. I suggest that relative abundance of omnivores in riffle and run habitats should be incorporated into any assemblage-level monitoring effort on the PR.

Finally, a taxonomic metric not used in this IBI, but possibly a strong indicator of habitat degradation from sedimentation, is the proportion of individuals as introduced species in pool habitats (redbreast sunfish, *Lepomis auritus*, and common carp, *Cyprinus*

Table 22. Reproductive group proportions (%) in deep-water (pool) habitats at nine sites on the Powell River in Virginia, sampled during August, 1988. Reproductive categories defined in Table 20 and in Appendix C.

	Reproductive Group					
Site (PRM)	PSFE	PC	ASL	SPD	MB	LB
174.4	1.49	40.30	0	0.75	12.69	44.78
163.4	0	45.38	0	4.20	7.14	43.28
158.3	0.87	48.92	4.33	6.49	7.79	31.60
153.4	0.74	44.85	2.94	2.21	14.71	34.56
146.8	1.25	47.92	2.08	5.00	8.75	35.00
144.6	1.06	43.62	5.32	5.32	20.21	24.47
123.0	0	33.33	0.93	6.48	23.15	36.11
119.3	0.53	37.97	1.60	9.63	10.70	39.57
117.3	0	47.96	1.02	8.16	11.22	31.63

carpio). This metric was positively correlated with pool sediment depth in the PR (ρ = 0.81, $p \leq 0.008$) (Table 23). Fausch et al. (1990) identified this pattern as a result of habitat degradation, often observed in aquatic systems of various sizes.

Fish Assemblage Structure Patterns

To assess assemblage structure patterns of the lower Powell River, fish species relative abundances (all habitats pooled) were subjected to cluster analyses (Appendix F, Figure 30). The most upstream site (PRM 174.4) was the most dissimilar of the sites, followed by PRM 123.0. Despite the two outlier sites, some evidence of upstream (sites PRM 163.4, 153.4, and 146.8) and downstream (sites PRM 144.6, 119.3, 117.3) clustering was evident.

Species relative abundances in shallow-water habitats and in deep-water habitats were further investigated by cluster analysis (Appendices G and H). Whereas no obvious spatial grouping occurred in pool assemblages (Figure 31), shallow-water assemblages exhibited distinct clustering into an upstream group and a downstream group (Figure 32). Twelve species primarily were responsible for separation of the assemblages into two groups (Table 24). Six species had higher abundances upstream (*E. blennioides*, *P. caprodes*, *N. volucellus*, *N. micropogon*, *L. chrysocephalus*, *C. anomalum*), and six species had higher abundances downstream (*E. rufilineatum*, *P. evides*, *C. galacturus*, *E. camurum*, *E. dissimilis*, *E. insignis*).

Possible causal factors for this upstream and downstream grouping of fish

Table 23. Percentage of individuals in pool habitats that are introduced[a] species for eight sites on the Powell River in Virginia. Information obtained from fish sampling in August, 1988 for scoring the IBI.

Site (PRM)	Proportion of Individuals as Introduced Species
171.2	23.9
163.4	8.0
158.3	17.3
153.4	11.0
146.8	13.8
144.6	10.6
123.0	2.7
119.3	0.5
117.3	0.0

[a] Introduced species were common carp (*Cyprinus carpio*) and redbreast sunfish (*Lepomis auritus*).

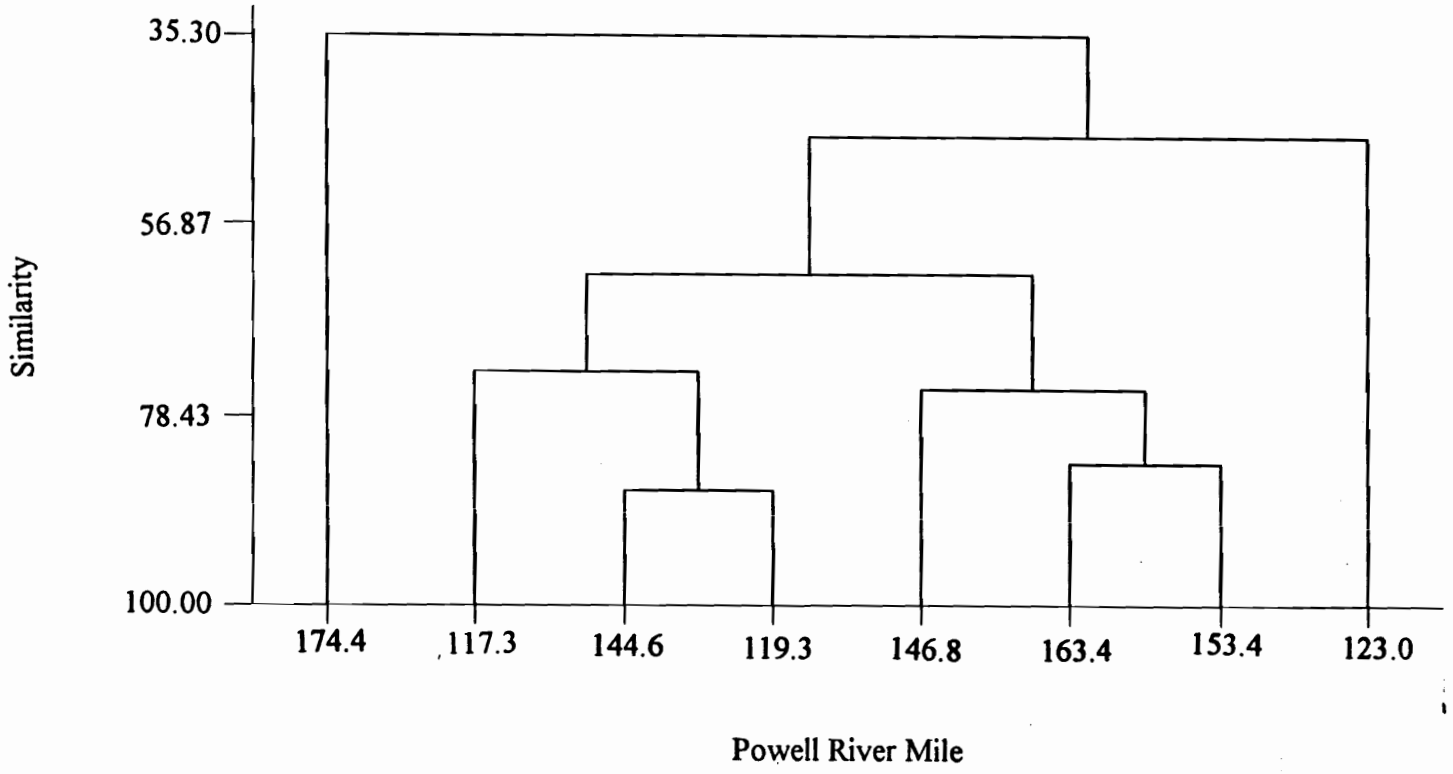

Figure 30. Cluster diagram of site-wide (fish abundances in all habitats pooled) fish assemblages at eight sites on the Powell River in Virginia. Data are fish species proportional abundances determined in August 1988.

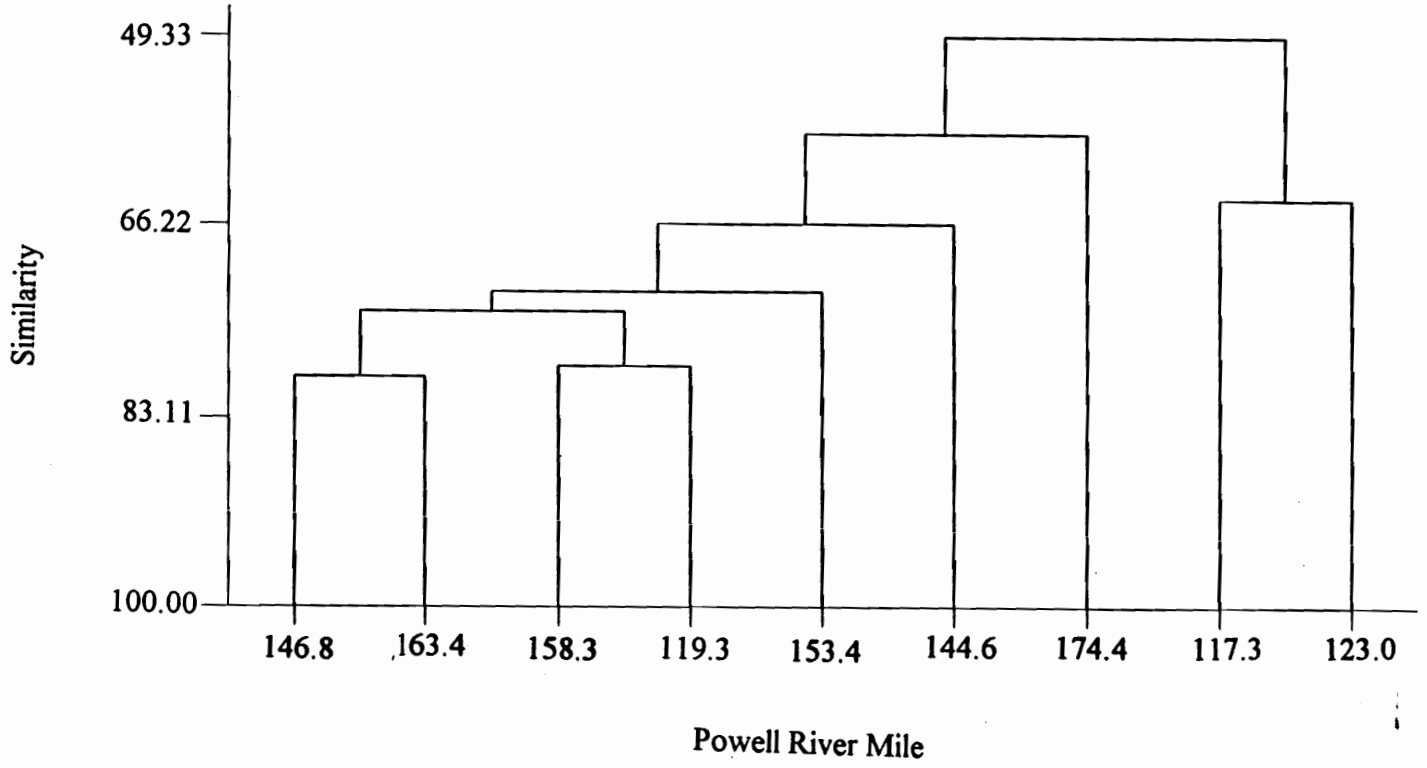

Figure 31. **Cluster diagram of pool habitat fish assemblages at nine sites on the Powell River in Virginia. The additional site is at Powell River mile 158.3. Data are fish species proportional abundances determined in August 1988.**

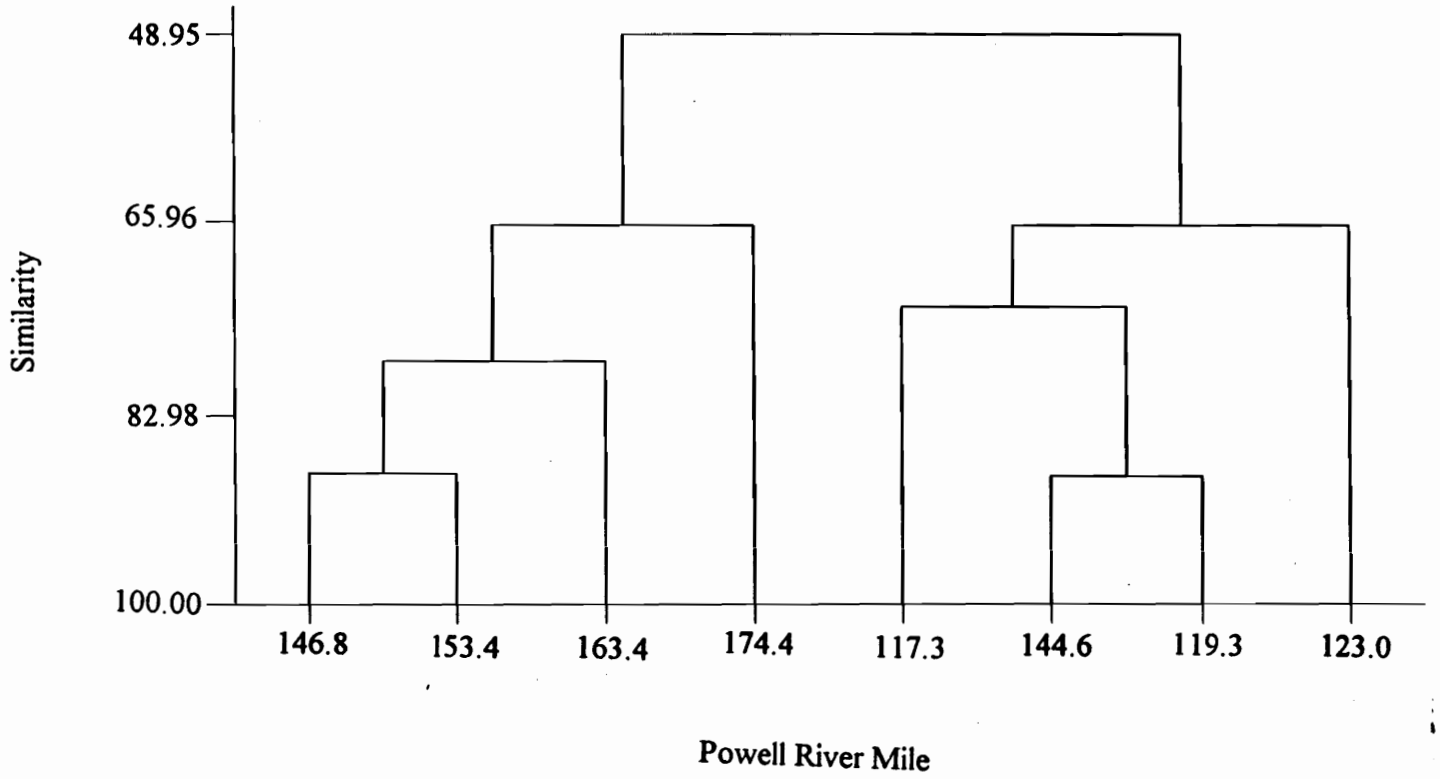

Figure 32. **Cluster diagram of shallow-water (riffle, run, sidepool, stream margin habitats) fish assemblages at eight sites on the Powell River in Virginia. Data are fish species proportional abundances sampled in August 1988.**

Table 24. Species listed in order of their contribution to defining the two shallow-water assemblage groups in the Powell River. The F-statistic is used as an index of the degree to which the two groups are defined by each species. Mean relative abundances (%) are listed for each species within a group. Values are derived from a collection of 6,175 fish specimens.

Species	F-statistic	Probability (p)	Mean (%) ± S.D. Upstream group	Mean (%) ± S.D. Downstream group
Etheostoma rufilineatum	24.41	0.003	10.73 ± 5.64	34.12 ± 5.77
Etheostoma blennioides	10.98	0.016	7.02 ± 2.56	2.38 ± 1.73
Percina evides	10.50	0.018	1.07 ± 0.74	7.47 ± 5.25
Cyprinella galacturus	9.10	0.024	0.40 ± 0.47	1.68 ± 0.53
Erimystax insignis	7.93	0.030	0.06 ± 0.13	2.54 ± 2.72
Nocomis micropogon	7.53	0.034	3.14 ± 2.91	0.24 ± 0.22
Percina caprodes	6.46	0.044	0.39 ± 0.31	0.07 ± 0.14
Notropis volucellus	6.24	0.047	7.43 ± 3.11	3.04 ± 1.77
Luxilis chrysocephalus	6.24	0.047	15.37 ± 5.52	6.22 ± 4.35
Campostoma anomalum	5.50	0.057	11.08 ± 3.27	6.40 ± 2.65
Etheostoma camurum	5.01	0.067	0 ± 0	0.89 ± 1.15
Erimystax dissimilis	4.83	0.070	0.85 ± 0.87	2.55 ± 1.65

assemblages were investigated by analyses of shallow-water physical habitat variables. Chemical habitat variables were not considered limiting (from Chapter One). Velocity variance ($p \leq 0.1124$), depth variance ($p \leq 0.885$), substrata ($p \leq 0.471$), cover ($p \leq 0.194$), and canopy closure ($p \leq 0.665$) habitat characteristics did not differ between assemblage groupings. Sedimentation, in contrast, was significantly higher upstream ($p \leq 0.030$). Since the physical habitat template (excluding sedimentation) is similar between groups, differences in fish assemblage structure were attributed to differences in sedimentation levels.

The upstream and downstream shallow-water assemblage groups were analyzed for relative abundance differences in trophic and reproductive groups. Relative abundance of the specialized insectivore guild was higher in the downstream group ($p \leq 0.030$), and the omnivore guild was more abundant in the upstream group ($p \leq 0.030$). Greater upstream group abundance of herbivores was not significant ($p \leq 0.061$). Generalized insectivores ($p \leq 0.885$) did not differ in abundance between groups. Top carnivores and parasites were not evaluated.

These results stand in contrast to trophic group structure in shallow-water assemblages in the North Fork Holston River in Virginia (Table 25). In that stream, the specialized insectivore group increased in relative abundance upstream. The upstream site, similar in size to the fifth order Powell River study reach sites, also had lower sedimentation. Although chronic mercury pollution that occurred below the upper North Fork Holston River site may confound the relationship of specialized insectivores

Table 25. Median embeddedness and trophic group proportions (%) in shallow-water habitats at three sites of the North Fork Holston River in Virginia, sampled during September, 1988. Trophic group designations after Saylor and Ahlstedt (1990). Median embeddedness values from the crayfish substata survey from November, 1990. NFHRM = North Fork Holston River Mile.

		Trophic Group				
Site (NFHRM)	Median Embeddedness	Specialized Insectivores	Omnivores	Herbivores	Generalized Insectivores	Piscivores
85.0	4	90.5	3.3	2.3	3.4	0.5
60.7	3	72.5	2.6	19.0	4.5	1.4
39.2	3	46.2	4.1	19.1	27.1	3.5

to sedimentation in that stream (Turner 1980), the fact that specialized insectivores were overwhelmingly dominant at NFHRM 85.0 supports the premise that fish species in the Powell River are affected by sedimentation through trophic disruption.

Shallow-water reproductive group patterns were analyzed similarly to the trophic groups. The SPD reproductive guild had higher relative abundance downstream ($p \leq 0.030$), whereas higher ASL ($p \leq 0.061$) and LB group ($p \leq 0.194$) abundances upstream were not statistically substantiated . The remaining reproductive group evaluated, PC, did not differ between assemblage groupings ($p \leq 0.885$). Assumed to be most susceptible to sedimentation of the six groups, the LB group, if anything, exhibited higher upstream abundances. If greatest sensitivity to sedimentation by the LB guild is true, then sedimentation was not high enough upstream in shallow-water habitats to cause reproductive impacts to LB group fishes (and by extension, to all other reproductive group fishes).

The SPD group is composed primarily of egg-burying darter species that are more abundant downstream (Appendix G). Assumed more able to successfully reproduce with some sedimentation than LB group species, lower upstream abundances again may reflect trophic impairments.

Identification of Sediment-intolerant and Sediment-tolerant Species

Species in shallow-water assemblages that are positively ("sediment-tolerant") or negatively ("sediment-intolerant") affected by sedimentation were identified by PCA and

correlations of meaningful principal components with the embeddedness index (higher embeddedness values indicate higher sedimentation levels). Principal component one, accounting for 50.8 % of the variation in species abundances, was significantly correlated with embeddedness ($\rho = 0.74$, $p \leq 0.035$). Although the fish abundance principal component one was not correlated with any other habitat variable (all $p \geq 0.071$) nor with a biotic index of water quality ($p \leq 1.000$; Appendix D), it was significantly correlated with PRM ($\rho = 0.83$, $p \leq 0.011$).

There was general agreement between species identified as sediment-intolerant or sediment-tolerant (Table 26), and species identified as having higher abundances downstream or upstream, respectively (Table 24). Five fish species were identified as sediment-intolerant. All are classified as specialized insectivores, and four of the five are either members of the SPD or LB reproductive groups (Table 26). Nine species were classified as sediment-tolerant in the Powell River (Table 26). Of the top five, three are omnivores and four of the five species are members of postulated sediment-tolerant reproductive guilds, PC and ASL (if *Luxilus chrysocephalus* predominantly spawns over *Nocomis* nests).

Of species not classified as either sediment-intolerant or sediment-tolerant, *Lythrurus lirus*, *Notropis ariommus*, *Moxostoma macrolepidotum*, *M. carinatum*, *Ictalurus punctatus*, *Ameirus natalis*, *Noturus eleutherus*, *Labidesthes sicculus*, *Micropterus punctulatus*, and possibly *Etheostoma camurum* were captured at too few sites to permit meaningful evaluation (Appendix G). All other species not classified had

Table 26. Fish species loadings on the first principal component (PC). PC one was positively correlated with sedimentation (embeddedness). Fish were captured from shallow-water habitats from 8 sites on the Powell River in Virginia (August 1988). Reproductive categories are: PSFE = pelagic spawner with floating eggs; PC = eggs given parental care; ASL = eggs deposited above substrate level; SPD = eggs deposited in a prepared or disturbed site; MB = eggs deposited on miscellaneous benthic substrata; and LB = eggs deposited on rocky benthic substrata. Trophic group categories are: HB = herbivore; IN = general insectivore; OM = omnivore; and SI = specialized insectivore.

Species	Species Loading	Reproductive Group	Trophic Group
Sediment-intolerant			
Etheostoma rufilineatum	-0.575	ASL	SI
Percina evides	-0.337	SPD	SI
Erimystax insignis	-0.190	LB	SI
Cyprinella galactura	-0.159	ASL	SI
Erimystax dissimilis	-0.118	LB	SI
Sediment-tolerant			
Luxilus chrysocephalus	0.334	SPD[a]	OM
Nocomis micropogon	0.241	ASL	OM
Pimephales notatus	0.238	PC	OM
Etheostoma blennioides	0.180	ASL	SI
Notropis volucellus	0.167	LB	SI
Notropis leuciodus	0.160	LB	SI
Lepomis auritus	0.150	PC	IN
Notropis rubellus	0.140	LB	SI
Campostoma anomalum	0.137	SPD[a]	HB

[a] Nest associate of *Nocomis micropogon*

relatively extensive longitudinal distributions, were likely evaluated with sufficient power, and therefore do not seem to be affected by differing sedimentation levels in the lower Powell River.

Analyses of relative abundances of pool species also were performed to determine whether one or more species exhibited sediment-tolerance or sediment-intolerance. Proportional abundances (Appendix H), analyzed by PCA, indicated that the first principal component, containing 37.9 % of the variation among species, was significantly correlated with sediment depth in pools ($\rho = -0.83$, $p \leq 0.008$) (Table 27). This fish abundance principal component, in addition, was not correlated with any other physical habitat variable (all $p \geq 0.098$). Unlike the shallow-water fish principal component, the pool fish principal component one was not associated with PRM ($\rho = -0.22$, $p \leq 0.576$).

Four species from pool habitats were classified as sediment-intolerant: two specialized insectivores, one generalized insectivore, and one top carnivore (Table 27). Five species in pool habitats are classified as sediment-tolerant; two general insectivores, one omnivore, one specialized insectivore, and one top carnivore.

In contrast to results for the shallow-water fish assemblage, *Luxilus chrysocephalus* was classified as sediment-intolerant in pools. The reason for opposing designations in pool and shallow-water habitats is unknown. To address this discrepancy, another analysis was conducted on species relative abundances in all habitats (pooled). The first principal component was correlated with sediment depth ($\rho = -0.878$, $p \leq 0.004$). *Luxilus chrysocephalus* was classified as sediment-tolerant (loading on principal

Table 27. Pool habitat fish species loading on the first principal component. The first principal component was negatively correlated with sedimentation. Species are designated as either responding positively (sediment-tolerant) or negatively (sediment-intolerant) to sedimentation. Fish were sampled from 9 sites in the Powell River in Virginia (August 1988). Reproductive and trophic group acronyms are defined in Table 26.

Species	Species Loading	Reproductive Group	Trophic Group
Sediment-intolerant			
Ambloplites rupestris	0.434	PC	TC
Lepomis megalotis	0.227	PC	IN
Notropis ariommus	0.216	LB	SI
Percina aurantiaca	0.162	SPD	SI
Sediment-tolerant			
Lepomis auritus	-0.509	PC	IN
Notropis volucellus	-0.273	LB	SI
Pimephales notatus	-0.262	PC	OM
Moxostoma erythrurum	-0.249	LB	IN
Micropterus punctulatus	-0.215	PC	TC

component one: -0.236). Hence, I consider *L. chrysocephalus* to be sediment-tolerant.

Integrating results from shallow-water and pool assemblages, 9 species were classified as sediment-intolerant and 11 species as sediment-tolerant (Table 28). Specialized insectivores comprise 78%, and generalized insectivores and omnivores 11%, of sediment-intolerant species. In contrast, specialized insectivores made up only 36%, whereas generalized insectivores and omnivores comprised 45 % of the sediment-tolerant species list. The LB reproductive guild, postulated as most sensitive to sedimentation, comprised 33% and 36% of the sediment-intolerant and sediment-tolerant species, respectively. It appears that trophic impairment may be the primary avenue by which sedimentation affects these species.

Fish and Physical Habitat Sampling- 1989

Physical Habitat

Physical habitat characteristics of sites sampled in 1989 were subjected to analyses for longitudinal trends (vs. PRM). Median velocity ($p \leq 0.332$), velocity variance ($p \leq 0.564$), median depth ($p \leq 0.861$), depth variance ($p \leq 0.457$), water willow ($p \leq 0.131$), log structure ($p \leq 0.377$), detritus ($p \leq 0.979$), and canopy closure ($p \leq 0.873$) did not vary longitudinally. The substrata principal component one, accounting for 60 % of the data variation, was significantly correlated with Powell River mile ($\rho = -0.88$, $p \leq 0.001$). Substrata category loadings on principal component one indicate higher amounts of boulder (-0.767) and cobble (-0.280) upstream with increasing amounts of pebble (0.340),

Table 28. Fish species designated as either sediment-tolerant or sediment-intolerant from all habitats combined. Fish sampling occurred in shallow-water (8 sites) and pool (9 sites) habitats in the Powell River in Virginia (August 1988).

Sediment-intolerant	Sediment-tolerant
Ambloplites rupestris	*Lepomis auritus*
Etheostoma rufilineatum	*Notropis volucellus*
Percina evides	*Pimephales notatus*
Lepomis megalotis	*Moxostoma erythrurum*
Notropis ariommus	*Nocomis micropogon*
Erimystax insignis	*Micropterus punctulatus*
Percina aurantiaca	*Etheostoma blennioides*
Cyprinella galactura	*Notropis leuciodus*
Erimystax dissimilis	*Notropis rubellus*
	Campostoma anomalum
	Luxilus chrysocephalus

gravel (0.318), and sand (0.303) downstream. Bedrock (0.128) and silt (-0.095) had much weaker longitudinal trends. In contrast to shallow-water sites sampled in 1988, sites in 1989 exhibited longitudinal gradients in substrata.

In addition, filamentous algae increased in frequency upstream ($\rho = 0.79$, $p \leq 0.004$). Higher amounts of filamentous algae upstream may be due to nutrient inputs from nearby wastewater treatment plants. Although not statistically substantiated, levels of phosphorus appeared higher upstream (Chapter One). In addition, the highest value of filamentous algae occurred just below the confluence of the North Fork Powell River. The Pennington Gap wastewater treatment plant, located on the North Fork Powell River, has a history of fecal coliform level violations of state standards (Introduction).

Embeddedness data from 11 sites were subjected to a principal components analysis to derive the embeddedness index. Site PRM 171.8 was excluded because turbid conditions prevented embeddedness measurements. Principal component one, accounting for 49.8 % of data variation, was significantly correlated with PRM as well ($\rho = -0.78$, $p \leq 0.004$). Embeddedness variable loadings on principal component one were category one (= -0.712), category two (= -0.008), category three (= -0.103), category four (= 0.051), and category five (= 0.693). Sites receiving negative scores on principal component one had higher sedimentation levels. Hence, sedimentation was higher in upstream sites. This upstream to downstream, high to low, sedimentation gradient was consistent with that found at sites sampled during 1988.

In summary, sites sampled in 1989 had an upstream-to-downstream, high-to-low

sedimentation gradient. There were no longitudinal changes in velocity, depth, water willow, and log structure. These results agree with that found for the 1988 sample sites. Unlike the 1988 sample sites, however, substrata changed longitudinally.

Fish Assemblage Structure Spatial Pattern

A total of 6,812 fish of 44 species was captured in 1989. Fish abundance data for all 12 sites (Appendix I) were evaluated by cluster analysis (Figure 33). Although there was grouping of some upstream sites (PRM 171.8, 168.9, and 156.0), no spatially interpretable subgroups occurred from PRM 153.9 downstream. These results are similar to those for 1988 shallow-water fish assemblage analysis; namely, that assemblages occurring in the fifth order and upstream section of the sixth order Powell River were essentially distinct from assemblages occurring in the mid- and lower portions of the sixth order.

The responses of omnivore and specialized insectivore trophic guilds, while not statistically verified, corresponded to that observed in 1988 sampling. Although not significant, omnivores tended to increase ($\rho = -0.53$, $p \leq 0.096$), and specialized insectivores to decrease ($\rho = 0.53$, $p \leq 0.096$) with greater sedimentation. Generalized insectivore ($\rho = 0.23$, $p \leq 0.502$) and herbivore ($\rho = -0.21$, $p \leq 0.537$) relative abundances were not responsive to sedimentation levels. The parasite and top carnivore guilds, occurring at less than 5 % relative abundance at each site, were not evaluated.

Reproductive guilds, except possibly for the ASL group, were not sensitive to

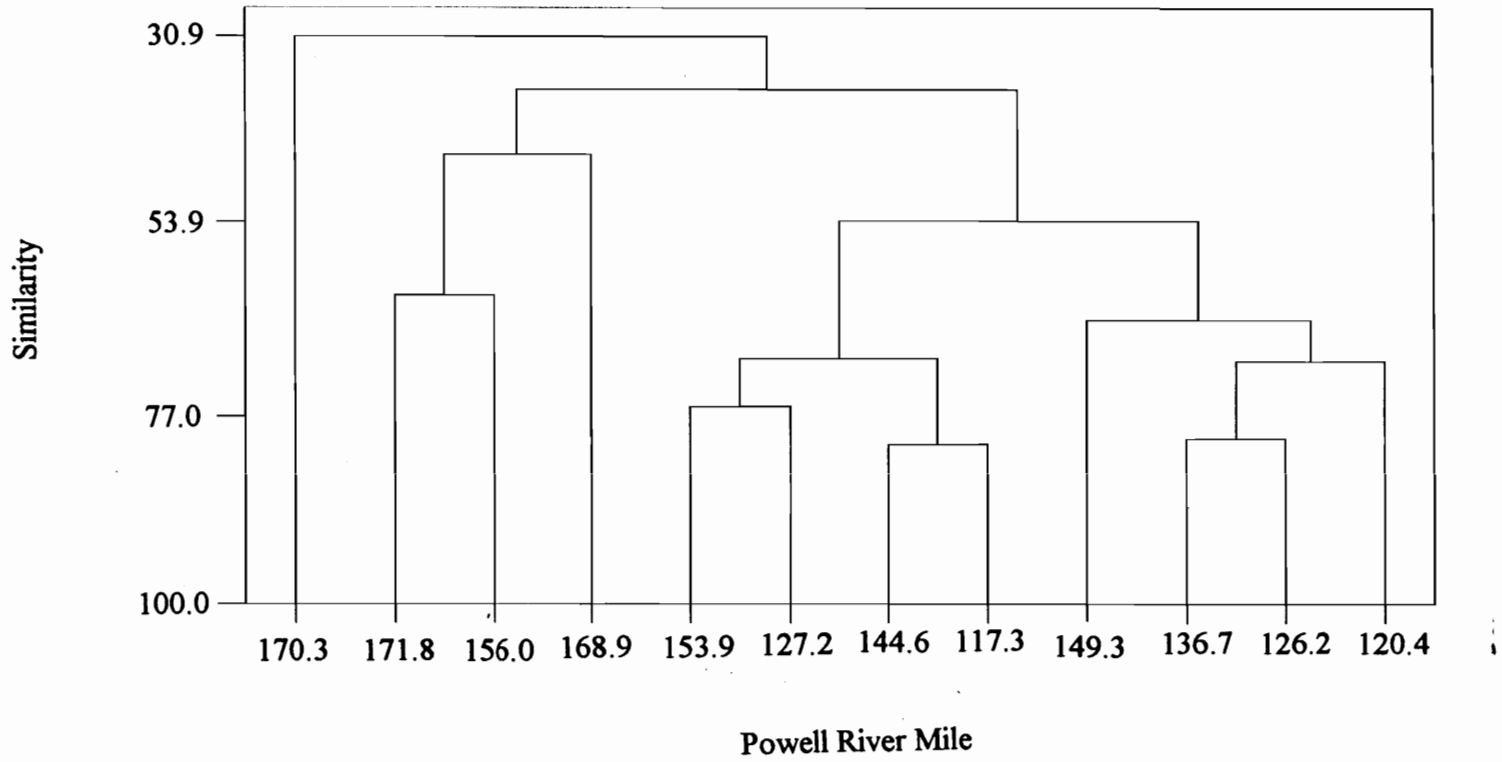

Figure 33. **Cluster diagram of shallow-water fish assemblages at 12 sites on the Powell River in Virginia. Data are fish species catch per unit effort obtained during summer 1989.**

sedimentation. There was no relationship between embeddedness and the LB ($\rho = 0.09$, $p \leq 0.790$), PC ($\rho = -0.19$, $p \leq 0.574$), and SPD ($\rho = 0.31$, $p \leq 0.355$) groups. Although the ASL group appeared to increase in relative abundance in higher sedimented sites, this relationship was not significant ($\rho = -0.55$, $p \leq 0.083$). The MB and PSFE guilds were not evaluated due to low relative abundances at all sites.

Identification of Sediment-intolerant and Sediment-tolerant Species

Fish abundances from 11 sites were subjected to a principal components analysis to investigate patterns of species abundance and factors potentially influencing fish abundance patterns. Fish assemblage data at site PRM 171.8 were not included because embeddedness measurements were not taken during fish sampling due to high turbidity. The second principal component was significantly correlated with the embeddedness index ($\rho = 0.63$, $p \leq 0.039$) (Table 29). This fish abundance principal component was neither correlated with any other measured habitat variable (all $p \geq 0.105$) nor with PRM ($p \leq 0.212$). Thus, abundances of species having high relative loadings on the second principal component appear primarily to be a function of sedimentation.

Five species were classified as sediment-intolerant and five species as sediment-tolerant (Table 29). From a functional perspective, specialized insectivores made up 80% and omnivores 0% of the sediment-intolerant species. The sediment-tolerant species list consisted of 60% specialized insectivores and 40% omnivores. The LB reproductive guild was equally prevalent in both tolerance categories (40%). Again, it appears that fishes, if

Table 29. Fish species classified as either sediment-tolerant or sediment-intolerant from assemblage-level sampling at 11 sites on the Powell River in Virginia during 1989. Species with high positive loadings on fish species principal component two are classified as sediment-intolerant. Species with high negative loadings on principal component two are classified as sediment-tolerant. Species loadings are in parentheses. Fish species principal component two accounted for 23.4 % of the data variation.

Sediment-intolerant		Sediment-tolerant	
Notropis rubellus	(0.473)	*Hybopsis amblops*	(-0.604)
Percina evides	(0.239)	*Pimephales notatus*	(-0.371)
Cottus carolinae	(0.167)	*Notropis volucellus*	(-0.250)
Erimystax dissimilis	(0.145)	*Nocomis micropogon*	(-0.158)
Etheostoma rufilineatum	(0.129)	*Etheostoma simoterum*	(-0.117)

sensitive to sedimentation, are more affected through trophic impairment.

Percina evides, *E. dissimilis*, *E. rufilineatum*, *P. notatus*, *N. volucellus*, and *N. micropogon* were classified similarly in each year, whereas *Cottus carolinae*, *Hybopsis amblops*, and *Etheostoma simoterum* were not classified in 1988 as either intolerant or tolerant. Classification of *N. rubellus* differed in 1988 and 1989. Classification status of these four species in particular were re-evaluated with information from the 1990 sampling.

Several species classified in 1988 were not classified in 1989. Many of them are known pool inhabitants (from Jenkins and Burkhead 1994) or seem to predominate in side-pool habitats (personal observation). Since pools were not sampled in 1989 and only 8.2 % of 1989 samples occurred in side-pools, the 1988 fish abundance data were analyzed to determine 1) if pool, riffle\run, and sidepool habitats had distinct fish assemblages, and 2) the habitat types occupied by various species. Figure 34 clearly shows fish assemblage differences among the three habitat types. Examining species coordinates, *A. rupestris* (PC2 = 0.112, PC1 = -0.245), *L. megalotis* (0.113, -0.262), *L. auritus* (0.149, -0.201), *M. erythrurum* (0.172, -0.243), and *M. punctulatus* (-0.098, -0.137) were captured predominantly in pools. *Notropis ariommus* (0.021, -0.022) and *P. aurantiaca* (0.027, -0.007) had no loadings. This lack of loading is very probably due to rarity (Appendix F) rather than a habitat generalist mode. Because 64 % of *P. aurantiaca* individuals and 71 % of *N. ariommus* individuals were captured in pools, however, it is reasonable to conclude that these two species predominantly inhabit pools as well. Use of pools by

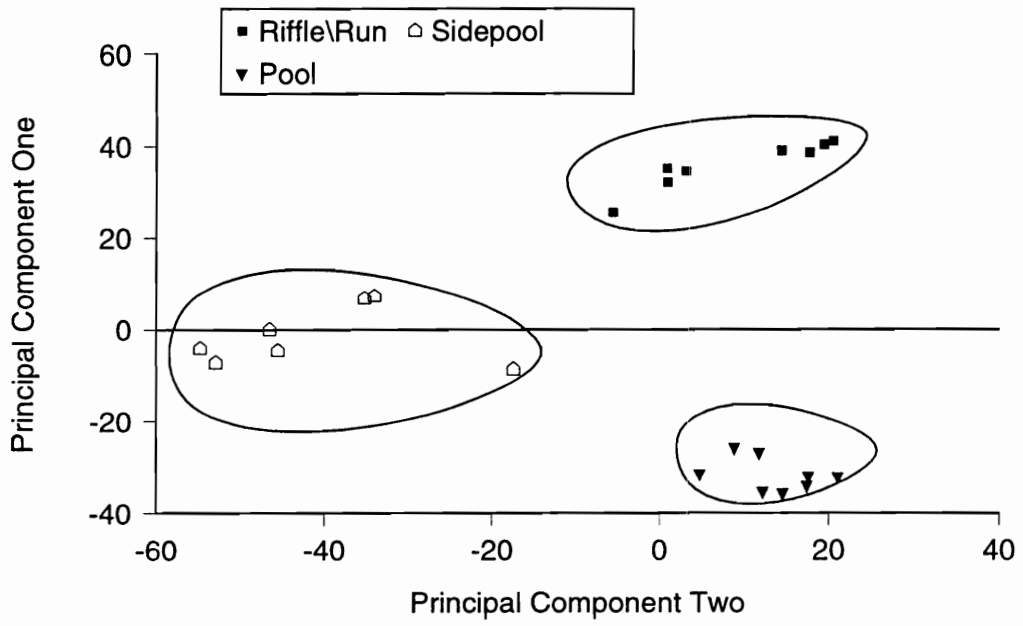

Figure 34. Principal components analysis by habitat type of fish assemblage data for eight sites in the lower Powell River during August, 1988.

these two species has been observed in other systems (Greenberg 1991; Etnier and Starnes 1993; Jenkins and Burkhead 1994).

Luxilus chrysocephalus (-0.791, -0.071) was captured in side-pools predominantly. *C. galactura* (-0.060, -0.002) was uncommon in 1989 samples (Appendix I), but of individuals captured in the three habitat types, 34 % were in riffle\runs, 57 % were in side-pools, and 9 % in pools. Apparently, *C. galactura* uses sidepool habitats primarily. Since sampling in 1989 (or in 1990) did not include pools and did not sample sidepool habitats as extensively as the riffle\run habitat type, it is not surprising that 1988 sediment-tolerance classified species that predominantly utilize pool and sidepool habitats did not receive a classification in 1989.

Erimystax insignis (0.060, 0.086), possibly because of low numbers captured, did not exhibit significant loadings. A riffle and run inhabiting species (Appendices G,H; Jenkins and Burkhead 1994), *E. insignis* was not collected often in 1989 (Appendix I) and had a relatively restricted capture distribution, which may have lowered the power of the 1989 classification analysis. The remaining species classified in 1988; but not in 1989, were *N. leuciodus (0.101, 0.233)* and *C. anomalum* (-0.066, 0.196). These species predominantly occurred in riffle\run habitats and were relatively common in both sample years (Appendices F, G, H, I). Reasons for classification in 1988 but not in 1989 are unknown. Final classification of these species was made with the 1990 data later in this chapter.

Fish and Physical Habitat Sampling- 1990

Physical Habitat

Physical habitat attributes of sites sampled in 1990 did not show a longitudinal pattern (all $p > 0.119$ except for woody debris, $p \leq 0.090$). By restricting sampling in 1990 to the 6th order Powell River, the sample sites were apparently much more homogeneous in their physical habitat characteristics than the river segment that includes both the fifth and sixth order Powell River.

Embeddedness data from the 12 sites in 1990 were analyzed using principal components analysis. Principal component one, accounting for 68.9 % of the data variation, had variable loadings: category one = -0.183, category two = -0.366, category three = -0.381, category four = -0.335, and category five = 0.758. Congruent with 1988 and 1989 results, embeddedness component one described data along a high versus low sediment gradient. Unlike 1988 and 1989, however, embeddedness was not correlated with PRM ($\rho = -0.48$, $p \leq 0.118$). The trend is greater sedimentation upstream, a known condition in the Powell River that was firmly established by 1988 and 1989 data. By restricting sampling to the 6th order Powell River, however, river morphology and hydraulics at a sample site may override the distinct sediment gradient documented to be present (i.e., when considering the Powell River on a scale that includes both 5th and 6th orders).

Fish Assemblage Structure Spatial Patterns

Fish sampling in 1990 captured 9,759 individuals of 45 species. Fish abundance data for the 12 sites sampled in 1990 (Appendix J) were evaluated by cluster analysis (Figure 35). Although there appeared again to be some upstream grouping (PRM 156.0, 153.9 and 144.6), the fish assemblages within the 6th order Powell River exhibited essentially no distinct spatial clustering. The most distinct site, PRM 119.3, was probably a result of the influence of the nearby Yellow Creek confluence. For example, high numbers of *Rhinichthys atratulus* were captured there (Appendix J). This species inhabits tributaries and very likely is a migrant from Yellow Creek. In summary, the lack of distinct fish assemblage patterns corresponds with the relatively homogeneous physical habitat among sites noted previously.

Analyses of functional responses to sedimentation, however, indicate that sedimentation has caused some changes to the fish assemblages. Abundance of the omnivore trophic group significantly increased at sites having higher sedimentation levels ($\rho = -0.67$, $p \leq 0.017$). The other trophic guilds evaluated, specialized insectivore ($\rho = 0.10$, $p \leq 0.746$), generalized insectivore ($\rho = -0.10$, $p \leq 0.762$), and herbivore ($\rho = 0.01$, $p \leq 0.983$), were not sensitive to the sedimentation levels present.

Analysis of reproductive guilds also indicated guild variation with sedimentation levels. The ASL group increased in relative abundance with higher sedimentation ($\rho = -0.65$, $p \leq 0.022$). Conversely, the LB ($\rho = -0.09$, $p \leq 0.779$), PC ($\rho = -0.24$, $p \leq 0.457$), and SPD ($\rho = 0.32$, $p \leq 0.31$) guilds were not sensitive to sedimentation levels. The MB

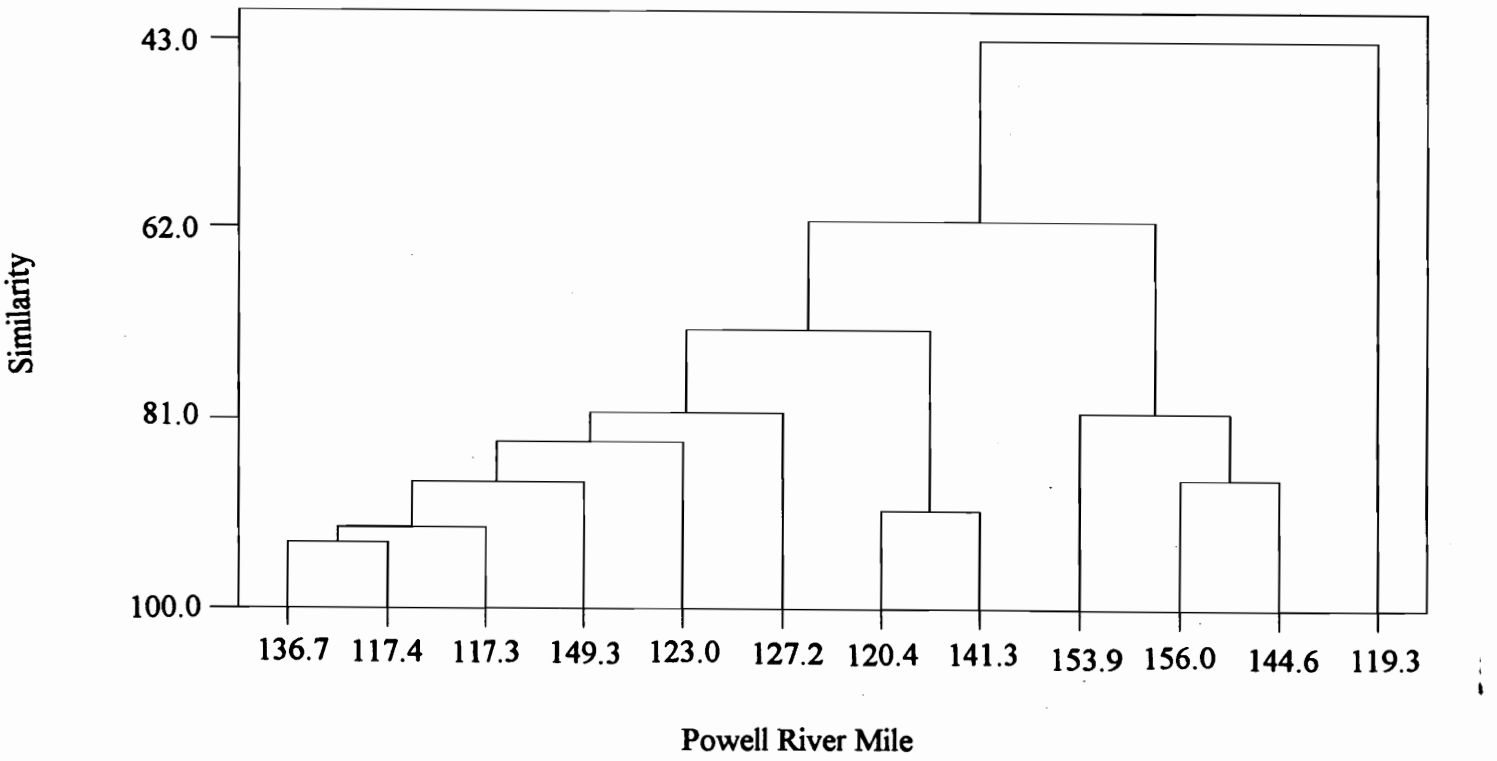

Figure 35. **Cluster diagram of shallow-water fish assemblages (number / 100 m^2) at 12 sites on the Powell River, Virginia in 1990.**

and PSFE groups were not evaluated due to low abundances at all sites ($\leq$ 5%).

Summarizing the functional analyses over the three sampling years, omnivore relative abundance increased in shallow-water habitats with increasing sedimentation three times at $p \leq 0.10$ or two times at $p \leq 0.05$. Specialized insectivores decreased in shallow-water habitats with increasing sedimentation twice at $p \leq 0.10$. In addition, when evaluating trophic guild abundance differences between the two shallow-water assemblage groupings observed in 1988, omnivores were more abundant and specialized insectivores less abundant ($p \leq 0.05$) in the upstream group of more sedimented sites. In pools, top carnivores decreased ($p \leq 0.05$) with higher pool sedimentation levels.

Reproductive guilds also responded to sedimentation. In shallow-water habitats, the ASL group was more abundant with greater sedimentation three times at $p \leq 0.10$ or one time at $p \leq 0.05$, and the LB group one time at $p \leq 0.05$. Relative abundance of the SPD group increased in shallow-water sites having lower sedimentation one time ($p \leq 0.05$). SPD relative abundance also was higher in the less sedimented group of sites in 1988 ($p \leq 0.030$) whereas the ASL group tended to be lower ($p \leq 0.061$). In pool habitats, conversely, no reproductive guild varied with sedimentation levels.

It appears that trophic guilds are more sensitive than reproductive guilds in the Powell River. The LB reproductive group, postulated as most sensitive to sedimentation, did not vary with sedimentation levels ($p \geq 0.05$). Hence, I conclude that reproductive processes were not significantly impaired by sedimentation levels that existed in the lower Powell River. Some sedimentation-induced reproductive impairment, however, may have

occurred as evidenced by the consistent response of increased ASL guild relative abundance to higher sedimentation. This group was postulated to be relatively tolerant of sedimentation. Conversely, omnivore relative abundance was expected to increase with higher sedimentation (Berkman and Rabeni 1987; Rabeni and Smale 1995), and the corresponding, consistent responses by the omnivore guild support the contention that trophic impairment by sedimentation has occurred in the Powell River.

Identification of Sediment-intolerant and Sediment-tolerant Fish Species

Principal component four, from a principal components analysis of 1990 site fish abundances, was correlated with sedimentation (abundance principal component four vs. embeddedness index; $\rho = -0.61$, $p \leq 0.036$) (Table 30). This fish abundance principal component, in addition, was not correlated with measured physical habitat variables (all $p \geq 0.106$) nor with PRM ($p \leq 0.443$). Thus, abundances of species having high absolute loadings on principal component four appear primarily to be a function of sedimentation.

Three species were classified as sediment-intolerant, and six species were classified as sediment-tolerant (Table 30). *Etheostoma rufilineatum* and *Percina evides* were classified as sediment-intolerant in all three years. *Erimystax insignis* was classified as sediment-tolerant twice (1988 and 1990); however, few captures in 1989 may have limited classification power for this species in 1989. The fact that these species have been consistently classified over the length of the study, including the 1990 sampling effort in which no significant longitudinal sediment gradient existed, strengthens the supposition

Table 30. Fish species classified as either sediment-tolerant or sediment-intolerant from assemblage-level sampling at 12 sites on the Powell River in Virginia during 1990. Species with high negative loadings on fish species principal component four are classified as sediment-intolerant. Species with high positive loadings on principal component two are classified as sediment-tolerant. Species loadings are in parentheses. Fish species principal component four accounted for 6.5 % of the data variation.

Sediment-intolerant		Sediment-tolerant	
Etheostoma rufilineatum	(-0.606)	*Hybopsis amblops*	(0.236)
Erimystax insignis	(-0.481)	*Nocomis micropogon*	(0.226)
Percina evides	(-0.325)	*Hypentelium nigricans*	(0.168)
		Notropis volucellus	(0.155)
		Notropis ariommus	(0.153)
		Cottus carolinae	(0.152)

that these three species are negatively affected by sedimentation and should be categorized as sediment-intolerant.

For sediment-tolerant classification, *H. amblops*, *N. micropogon*, and *N. volucellus* have been classified previously. This is consistent evidence that these three species are very tolerant of the higher sedimentation levels found in portions of the Powell River. *Hypentelium nigricans* was classified as sediment-tolerant during 1990 only. The 1990 sediment-tolerant classification of *N. ariommus* contradicts results from 1988. Since these two species inhabit pools primarily, it is conceivable that they can tolerate somewhat higher sedimentation. The 1988 data suggest that the higher sedimentation levels in the 5th order Powell River, however, may exceed a threshold for *N. ariommus.* Because pool habitats were not sampled in 1990, a conservative approach is adopted and the sediment-tolerant classifications of *H. nigricans* and *N. ariommus* are dropped. *Notropis ariommus* remains classified as sediment-intolerant from the 1988 analyses. *Hypentelium nigricans*, although not classified, is probably more sediment-tolerant than many other species within the broad "non-classified" group. Its distribution includes many lowland, depositional streams in several states (Lee et al. 1980). Within these systems, however, *H. nigricans* may prefer erosional microhabitats (Meffe and Sheldon 1988). Finally, the 1990 and 1989 classification results conflict for *C. carolinae.* This species may be somewhat sediment-tolerant at moderate sediment levels, dropping off in abundance at sites with higher sedimentation (5th order Powell River). Due to conflicting results, *C. carolinae* is not classified, but this species may be less tolerant of sedimentation than some of the other

non-classified species.

Functional composition of tolerance categories was similar to patterns observed in 1988 and 1989. Specialized insectivores comprised 100% of the sediment-intolerant species group, whereas omnivores made up 17% and specialized insectivores only 50% of the sediment-tolerant category. The LB reproductive group comprised 33% of the sediment-intolerant and 66% of the sediment-tolerant categories.

Tolerance classifications for 1990 were combined with those tolerance classifications from 1988 and 1989 into an overall classification list (Table 31). Included are species that either 1) were classified more than once in the same category or 2) were classified only during 1988 but predominantly use habitats other than riffles or runs (especially pools). Hence, *N. leuciodus*, *C. anomalum*, and *E. simoterum*, all classified once as sediment-tolerant, are not classified. Moreover, *N. rubellus* and *C. carolinae* had different classifications in two years; therefore, they were not classified.

My confidence in the final sediment-tolerance classifications recorded in Table 31 varies among species. I am most confident in assigned sediment-tolerance classifications where a species was classified more than once. Secondly, I am somewhat less confident of those species classified only in 1988 that predominantly occur in pools. Since sampling of pools did not occur in 1989 or 1990, these species could only be evaluated once. I am least confident in those species that are generally distributed in shallow-water habitats or in sidepools and were classified only once.

All species classified as sediment-intolerant decreased with increased sediment

Table 31. Overall sediment-tolerance classifications for the fish assemblages within the 5th and 6th order Powell River. Data for categorization were obtained from the 1988, 1989, and 1990 field seasons. SINT = sediment-intolerant, STOL = sediment-tolerant.

Species	Classification	Years classified	Predominant habitats
Etheostoma rufilineatum	SINT	1988, 1989, 1990	Riffle, Run
Percina evides	SINT	1988, 1989, 1990	Riffle, Run
Erimystax insignis	SINT	1988, 1990	Riffle, Run
Erimystax dissimilis	SINT	1988, 1989	Riffle, Run
Cyprinella galactura	SINT	1988	Generalist
Notropis artommus	SINT	1988	Pool, Run
Ambloplites rupestris	SINT	1988	Pool
Lepomis megalotis[a]	SINT	1988	Pool
Percina aurantiaca	SINT	1988	Pool, Run
Nocomis micropogon	STOL	1988, 1989, 1990	Riffle, Run
Notropis volucellus	STOL	1988, 1989, 1990	Riffle, Run
Hybopsis amblops	STOL	1989, 1990	Generalist
Pimephales notatus	STOL	1988, 1989	Generalist
Luxilus chrysocephalus	STOL	1988	Sidepool
Moxostoma erythrurum	STOL	1988	Pool
Lepomis auritus	STOL	1988	Pool
Micropterus punctulatus	STOL	1988	Pool

[a] Classification may be more a result of deleterious interaction with *Lepomis auritus* than of sediment-intolerance.

level when analyzed by simple univariate correlations. *Etheostoma rufilineatum* and *Percina evides* significantly decreased with increased sediment in two of the three sampling years (Figures 36 and 37, respectively). *Erimystax insignis* decreased significantly with increased sediment in 1988 ($\rho = -0.79$, $p \leq 0.021$) but not in 1990 ($\rho = 0.004$, $p \leq 0.991$). *Erimystax dissimilis* likewise decreased significantly in one of the two years it was classified (1989: $\rho = 0.62$, $p \leq 0.043$; 1988: $\rho = -0.53$, $p \leq 0.18$). Of those species classified only once from sampling in 1988, *Percina aurantiaca* ($\rho = -0.95$, $p \leq 0.001$), *Cyprinella galactura* ($\rho = -0.78$, $p \leq 0.023$), and *Ambloplites rupestris* ($\rho = -0.90$, $p \leq 0.001$) decreased significantly with increased sediments. *Notropis ariommus* ($\rho = -0.64$, $p \leq 0.089$) and *Lepomis megalotis* ($\rho = -0.30$, $p \leq 0.425$) did not significantly decrease in univariate correlations with increased sediment.

In summary, nine fish species were classified as sediment-intolerant (~ 16% of fish fauna sampled) and eight species were classified as sediment-tolerant (~ 14%). The remainder are relegated to a relatively heterogeneous non-classified group. This latter group is heterogeneous because it contains species that are common and apparently unaffected by sedimentation levels in the Powell River, and species that are uncommon or rare (as *E. vulneratum* and *N. eleutherus*) and hence could not be adequately evaluated. Also within the non-classified group, *E. simoterum*, *N. leuciodus*, *C. anomalum*, and *H. nigricans* may be somewhat more sediment-tolerant than the rest of the group, especially at higher sedimentation levels. *Cottus carolinae* may be more sediment-tolerant than many in the non-classified category within a range of moderate sedimentation.

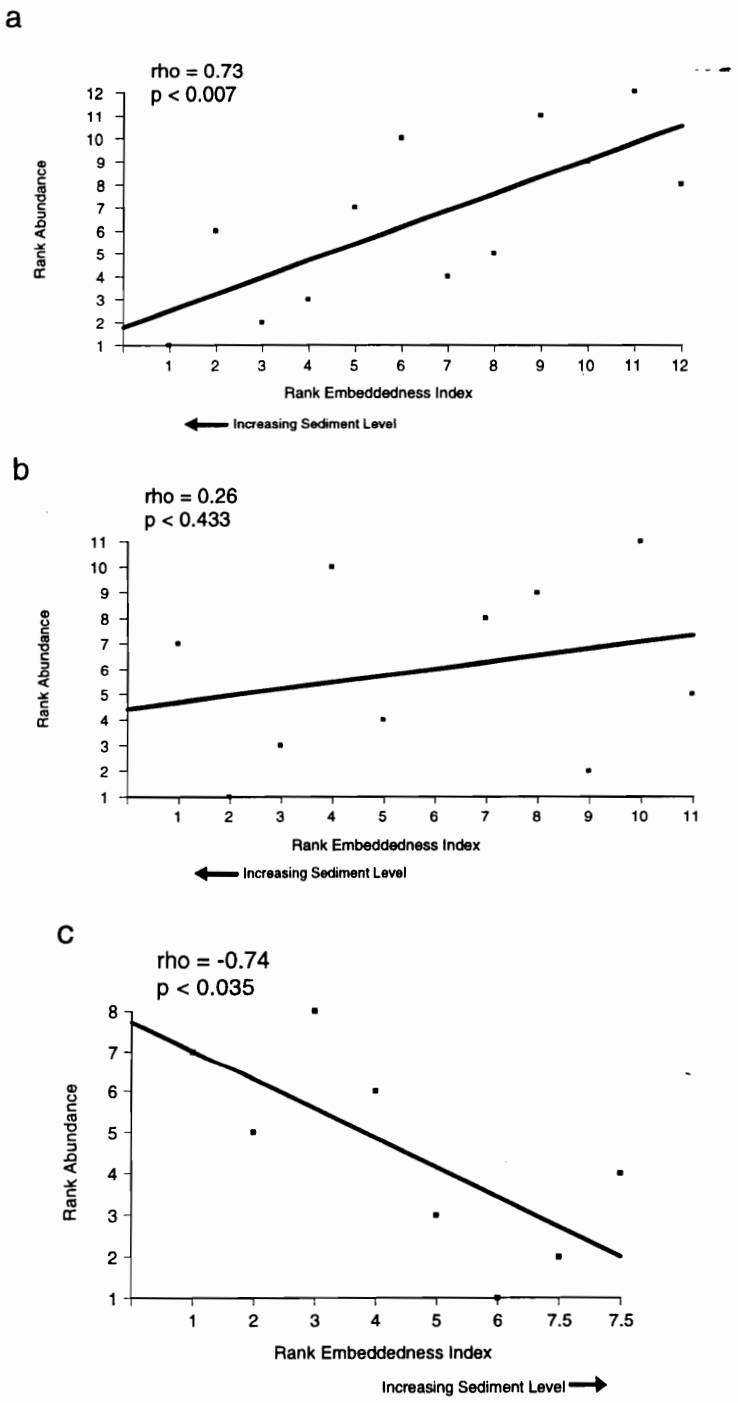

Figure 36. Spearman's Rank Correlations of *Etheostoma rufilineatum* abundance versus sediment level for 1990 (a), 1989 (b), and 1988 (c).

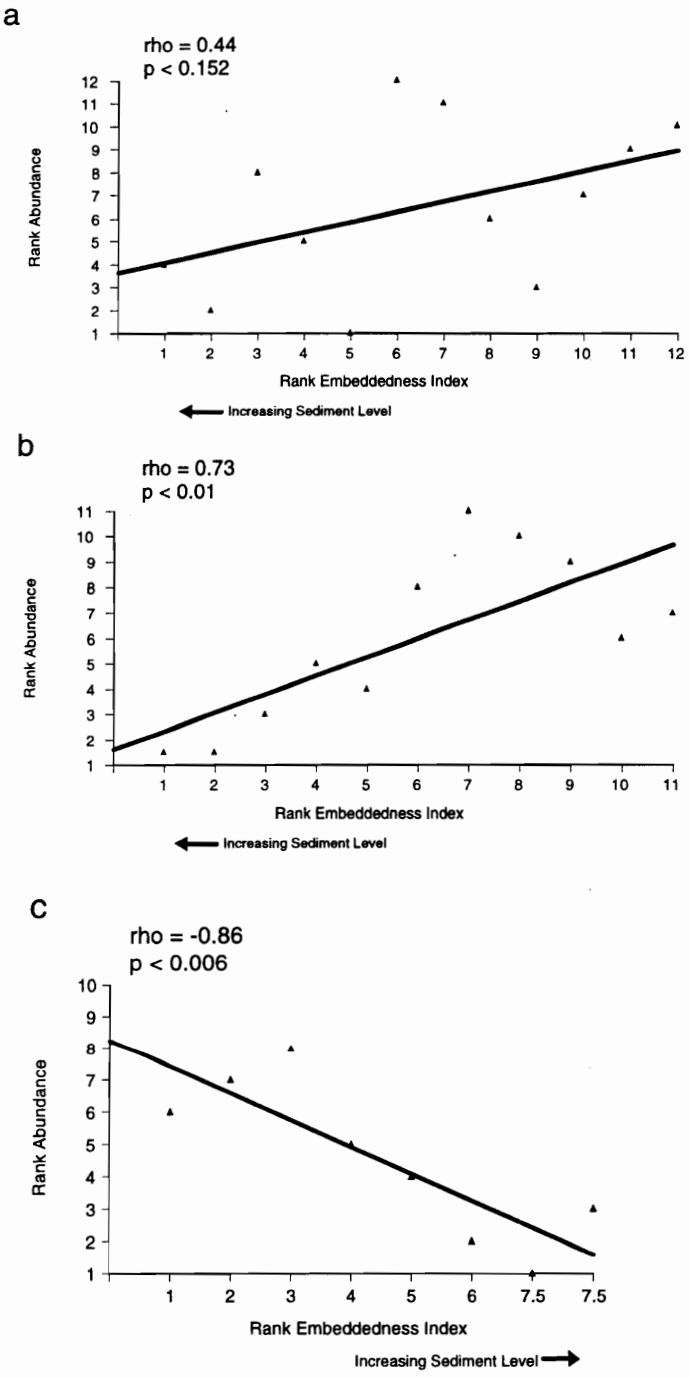

Figure 37. Spearman's Rank Correlations of *Percina evides* abundance versus sediment level for 1990 (a), 1989 (b), and 1988 (c).

By examining the two classification groups from a guild perspective, it appears that a stronger, more interpretable pattern exists using trophic groups as opposed to reproductive groups. Whereas 78% of the sediment-intolerant species were specialized insectivores, specialized feeders comprised only 25 % of the sediment-tolerant species. Further, 0% of sediment-intolerant species were omnivores as opposed to 38% of the sediment-tolerant species. The reproductive group that was considered most sediment-sensitive (LB; species that broadcast spawn over rocks) was equally represented in both groups, as were reproductive groups considered to allow some protection from sedimentation (PC, SPD, and ASL). Additional, albeit indirect, support for trophic disruption, as the primary mechanism of sedimentation effects, is the water column orientation and trophic group membership of the classified species. Sediment-intolerant species, particularly those restricted to shallow-water habitats, are predominantly benthic, specialized insectivores (Jenkins and Burkhead 1994; 83% of sediment-intolerant species that primarily occur in shallow-water). A similar observation was made in streams of northeast Missouri (Rabeni and Smale 1995).

Given the two sediment-tolerance groups defined *a priori*, a confirmatory analysis was performed using canonical discriminant function analysis (CDFA). The working hypothesis was that sediment-intolerant species should inhabit microhabitats having low levels of sedimentation relative to microhabitats occupied by members of the sediment-tolerant group. First, using fish-microhabitat data obtained from 12 sites in 1990 (Appendix K), species centroids were plotted along the first two canonical axes (Figure

38). Canonical axis one accounted for 31.6 % of the data variation and described the horizontal component of the sampling sites (Table 32). Velocity, cobble substratum, and filamentous algae increased toward the stream center whereas sedimentation (as measured by embeddedness), sand and silt substrata, and overhead vegetation increased toward the shoreline. Canonical axis two, accounting for 17.8 % of the data variation, describes the depth profile. As depth increases, bedrock and boulder substrata, log structure, and woody debris increase, whereas pebble substratum and water willow decrease.

Plotting of species classified as sediment-intolerant or sediment-tolerant on the two canonical axes exhibited distinct patterns. The majority of sediment-intolerant species occurred in shallower, swifter habitats containing low sedimentation, more cobble and pebble, and less large structure (boulders and wood). Sediment-tolerant species utilized habitats that had slower water velocities, higher sedimentation, more large structure, and less cobble and pebble.

Microhabitat segregation often has been linked to differential trophic resources (Werner and Hall 1979; Hixon 1980; Ebersole 1985). Degradation of microhabitats by sedimentation has been shown to reduce aquatic insect abundances (Brusven and Prather 1974; Waters 1995), which are important food resources for most of the species found in shallow-water habitats of the Powell River. Observed negative reaction to sedimentation by several shallow-water fish species (i.e., the "sediment-intolerant" group members) appears to be a function of trophic disruption. The predominance of specialized feeders in the "sediment-intolerant" group and omnivores in the "sediment-tolerant" group support

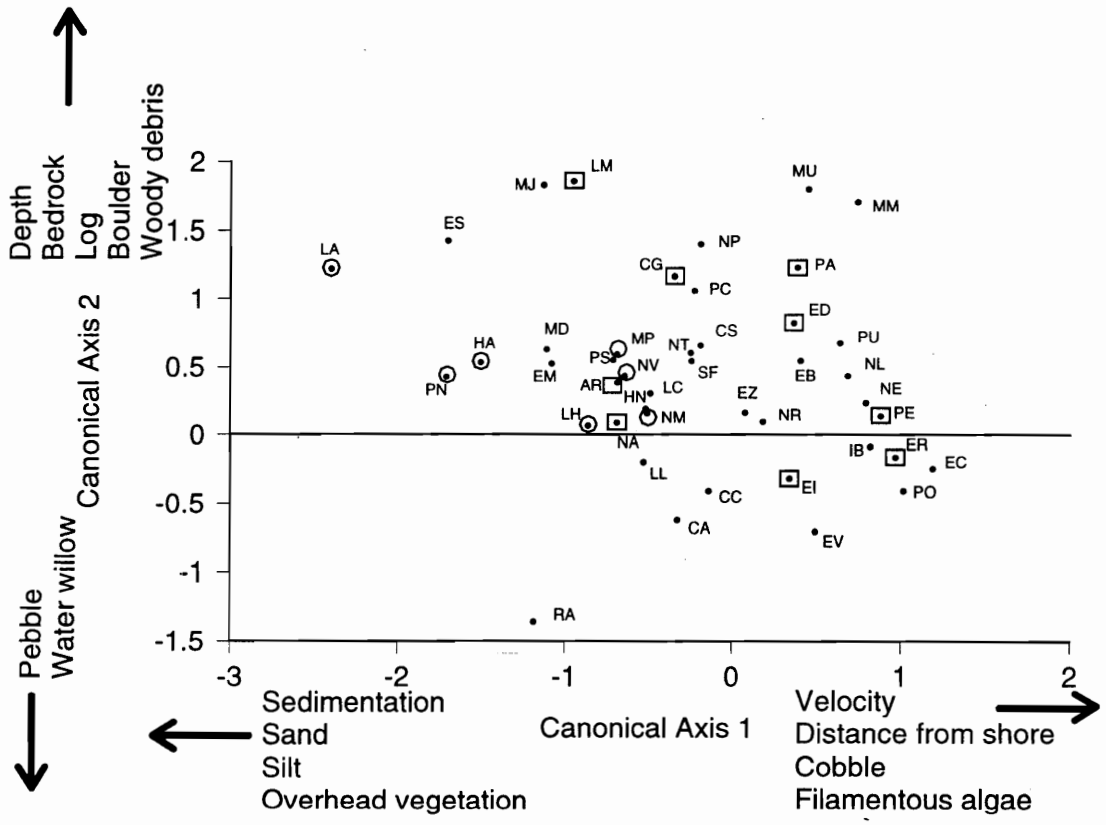

Figure 38. **Microhabitat use by fish species plotted as centroids along the first two canonical axes. Data collected from 12 sites within the 6th order Powell River during summer, 1990. Species abbreviations are listed on following page. Species classified as sediment-intolerant and sediment-tolerant are denoted by squares and circles, respectively.**

Species Abbreviations for Figure 38
(Species in bold are classified as sediment-intolerant or sediment-tolerant)

AR = *Ambloplites rupestris*
CA = *Campostoma anomalum*
CC = *Cottus carolinae*
CG = *Cyprinella galactura*
CS = *Cyprinella spiloptera*
EB = *Etheostoma blennioides*
EC = *Ethesotoma camurum*
ED = *Erimystax dissimilis*
EI = *Erimystax insignis*
EM = *Etheostoma simoterum*
ER = *Etheostoma rufilineatum*
ES = *Etheostoma stigmaeum*
EV = *Etheostoma vulneratum*
EZ = *Etheostoma zonale*
HA = *Hybopsis amblops*
HN = *Hypentelium nigricans*
IB = *Ichthyomyzon bdellium*
LA = *Lepomis auritus*
LC = *Luxilus coccogenis*
LH = *Luxilus chrysocephalus*
LL = *Lythrurus lirus*
LM = *Lepomis megalotis*
MD = *Micropterus dolomieu*
MJ = *Moxostoma* sp. (juveniles)
MM = *Moxostoma macrolepidotum*
MP = *Micropterus punctulatus*
MU = *Moxostoma duquesnei*
NA = *Notropis ariommus*
NE = *Noturus eleutherus*
NL = *Notropis leuciodus*
NM = *Nocomis micropogon*
NP = *Notropis photogenis*
NR = *Notropis rubellus*
NT = *Notropis telescopus*
NV = *Notropis volucellus*
PA = *Percina aurantiaca*
PC = *Percina caprodes*
PE = *Percina evides*
PN = *Pimephales notatus*
PO = *Percina copelandi*
PS = *Percina sciera*
PU = *Phenacobius uranops*
RA = *Rhinichthys atratulus*
SP = *Notropis* sp. (sawfin shiner)

Table 32. Physical habitat variable loadings on the first two canonical axes. Canonical axis one and two account for 49.4 % of the variation in the fish-habitat data. Fish-habitat information on 43 species was collected from 12 sites on the Powell River in Virginia during June - August, 1990.

Habitat Variable	Canonical Axis 1	Canonical Axis 2
Depth	0.313	0.562
Velocity	0.797	-0.206
Embeddedness	0.762	-0.306
Distance from shore	0.623	0.446
Silt	-0.390	0.153
Sand	-0.456	0.278
Gravel	-0.210	0.027
Pebble	0.206	-0.608
Cobble	0.568	-0.051
Boulder	-0.060	0.245
Bedrock	-0.021	0.332
Water willow	-0.362	-0.460
Simple log	-0.247	0.258
Woody debris	-0.212	0.229
Complex log	-0.007	0.042
Overhead vegetation	-0.208	0.038
Root wads	-0.105	0.081
Filamentous algae	0.267	-0.119
Pond weed	0.054	-0.194
Root mats	-0.064	-0.170

this inference, as well as assemblage-level trophic and reproductive group changes associated with different sedimentation levels (Chapter One).

The sediment-tolerant and sediment-intolerant groups were analyzed by two CDFA's. The first analysis used presence-absence data only. The single canonical axis generated was a function of high positive loadings of distance from shore, velocity, embeddedness, depth , and cobble variables and high negative loadings of sand, water willow, overhead vegetation, logs, and woody debris (Table 33). Median group canonical scores were 0.218 for the sediment-intolerant group and -0.210 for the sediment-tolerant group (Figure 39). These scores were significantly different ($p \leq 0.0001$). Of particular importance to this study was that the sediment-intolerant group members occupied less sedimented microhabitat than sediment-tolerant group species (4.5 vs. 4.25 median embeddedness, respectively; $p \leq 0.008$).

The sediment-tolerance groups (including the non-classified group) were reanalyzed by CDFA, this time using species abundances. The results were similar to presence-absence data. Habitat loading patterns on canonical axis one (accounting for 75 % of the data variation) are essentially identical to the previous analysis on the classified groups (Table 33). Use of species abundances resulted in greater delineation between the sediment-tolerance groups. Median canonical axis one scores for each group were 0.690 (sediment-intolerant), -0.111 (non-classified), and -0.494 (sediment-tolerant) (all medians different, $p \leq 0.0001$; Figure 40). Again, the sediment-intolerant group members occurred in low sedimented microhabitats relative to the sediment-tolerant group

Table 33. Physical habitat variable loadings on one canonical axis from each of three fish-microhabitat canonical discriminant function analyses. Fish-habitat information on 45 species was collected from 12 sites on the Powell River in Virginia during June - August, 1990. G2PACA = canonical axis of sediment-intolerant and sediment-tolerant group presence-absence analysis, G3SACA1 = canonical axis 1 of sediment-intolerant, sediment-tolerant, and non-classified group species abundance analysis.

Habitat Variable	G2PACA	G3SACA1
Depth	0.482	0.377
Velocity	0.643	0.842
Embeddedness	0.516	0.742
Distance from shore	0.714	0.537
Silt	-0.281	-0.412
Sand	-0.534	-0.652
Gravel	-0.196	-0.200
Pebble	0.120	0.284
Cobble	0.424	0.559
Boulder	-0.126	-0.099
Bedrock	0.170	0.006
Water willow	-0.367	-0.417
Simple log	-0.327	-0.315
Woody debris	-0.320	-0.264
Complex log	0	-0.012
Overhead vegetation	-0.361	-0.185
Root wads	-0.175	-0.125
Filamentous algae	0.174	0.317
Pond weed	0.307	0.123
Root mats	-0.034	-0.030

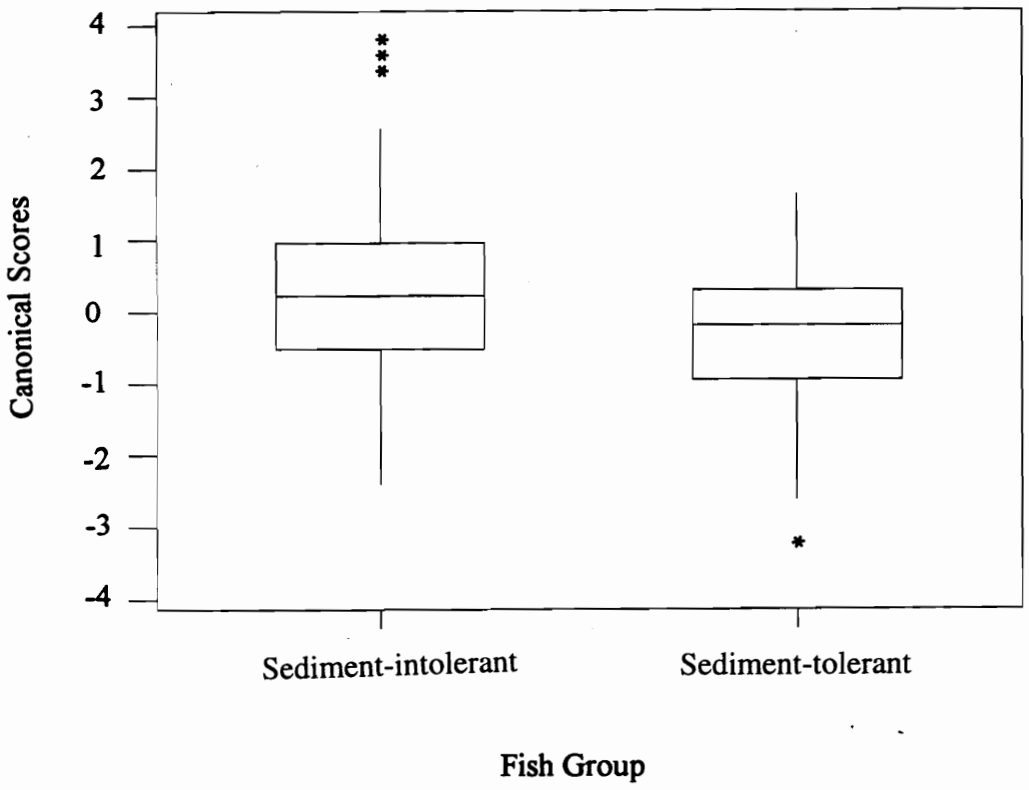

Figure 39. **Box plots of canonical scores for sediment-intolerant and sediment-tolerant groups. Canonical scores are based on presence-absence data. The line drawn across the box denotes the median. The top of the box is at the third quartile (Q_3, 75 %) and the box bottom is at the first quartile (Q_1, 25 %) of the data distribution. The vertical lines, or whiskers, extend the distance out from the box to observations 1.5 times the box distance (Q_3 - Q_1). Asterisks mark data points outside of the whiskers and are considered outliers. Data collected from 12 sites within the 6th order Powell River during summer, 1990.**

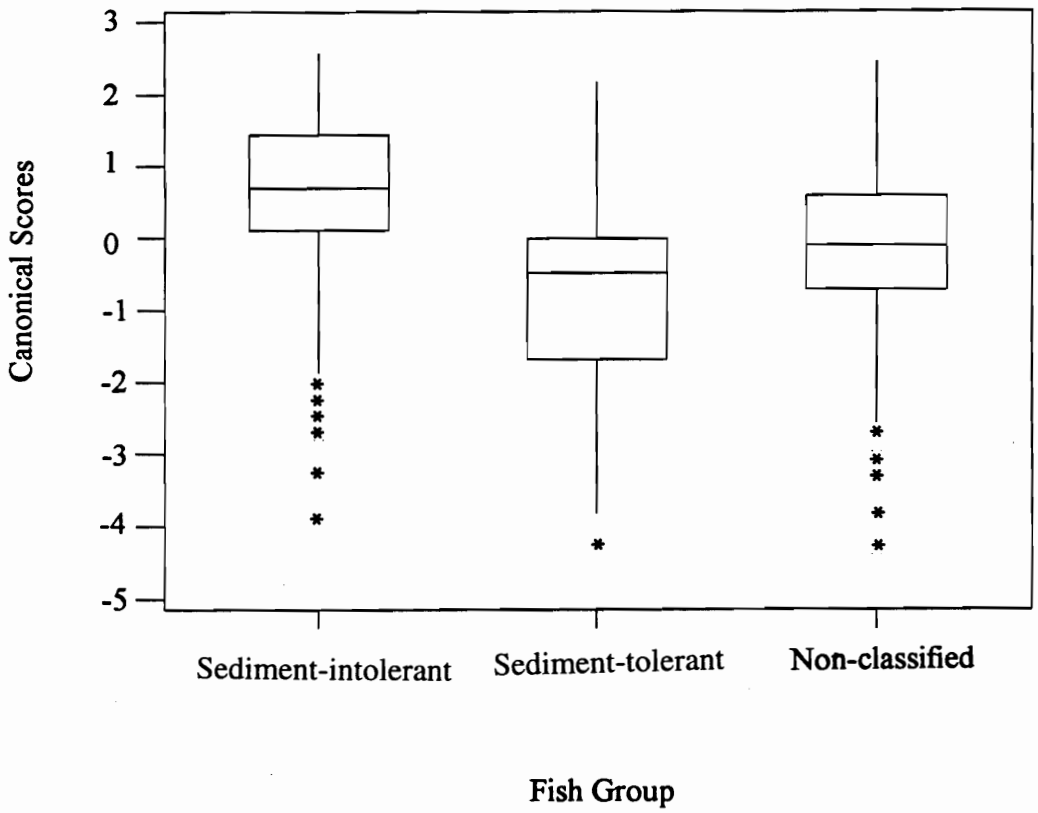

Figure 40. **Box plots of canonical scores on canonical axis 1 for sediment-intolerant, sediment-tolerant, and non-classified groups. Canonical scores are based on species abundance data. Data collected from 12 sites within the 6th order Powell River during summer, 1990.**

(5 vs. 3.3 median embeddedness, respectively; $p \leq 0.0001$) (Figure 41). The non-classified group, despite its more heterogeneous nature, was intermediate between the intolerant and tolerant groups. In addition, median embeddedness (4.25) was intermediate between that of the sediment-intolerant and sediment-tolerant groups ($p \leq 0.0001$). Hence, the non-classified group occupied microhabitats with intermediate levels of sedimentation (Figure 41). This and the previous microhabitat-scale analysis are additional, confirmatory evidence that species identified as sediment-intolerant or sediment-tolerant at a macrohabitat scale are classified correctly.

Despite the distinction in microhabitat use between the sediment-intolerant and sediment-tolerant groups, particularly in regard to sedimentation, species within a group probably differ somewhat in their response to sedimentation. In particular, it appears that sediment-intolerant species which utilize pool habitats frequently or predominantly are somewhat more tolerant of sedimentation, at least on a microhabitat scale, than the sediment-intolerant species that are restricted to flowing, shallow areas. Sediment-intolerant species (n=4), mostly found in flowing, shallow habitats, had an average microhabitat embeddedness of 4.5. In contrast, the five sediment-intolerant species that utilized pools were found in microhabitats that averaged 3.3 embeddedness (Appendix K). Greenberg (1991) also noted that *P. aurantiaca* occurred in higher sedimented microhabitats of the Little River, Tennessee than either *E. rufilineatum* or *P. evides* (all three species classified as sediment-intolerant by the present study). In northeast Missouri, there was a clear relation between percent silt and habitat-specific assemblage structure of

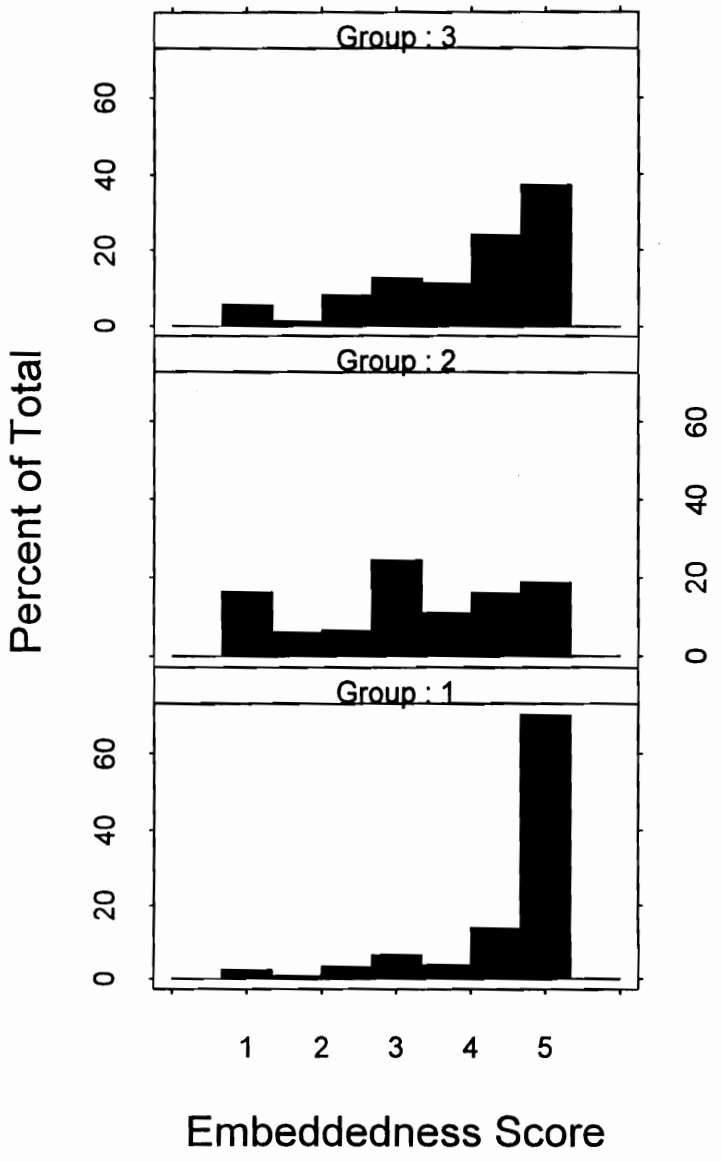

Figure 41. **Distribution of embeddedness scores from analyses of group use of microhabitat (summer 1990). Group 1 = sediment-intolerant, Group 2 = sediment-tolerant, Group 3 = not classified.**

fish in riffles but not in pools (Berkman and Rabeni 1987; Rabeni and Smale 1995).

Certainly the non-classified group members differ in their sediment-tolerance. As mentioned previously, *E. simoterum*, *N. leuciodus*, *C. anomalum*, *H. nigricans*, and *C. carolinae* may be more sediment-tolerant than other non-classified species. Another important consideration is the true classification of rare or uncommon species that were not of sufficient abundance nor distributed widely enough for meaningful decisions on their classification. Of these, *P. copelandi* (embeddedness average = 4.8), *E. camurum* (= 4.9), and *N. eleutherus* (=4.4) occupied microhabitats having very low sedimentation. These species, uncommon for unknown reasons (see Sheldon 1987), are probably sediment-intolerant. At least *E. camurum* appears to be intolerant of even moderate levels of siltation in other stream systems (Jenkins and Burkhead 1994). An uncommon species that occupies relatively high sedimented microhabitats, *E. stigmaeum* (embeddedness average = 1.9) may be sediment-tolerant. *Etheostoma stigmaeum* (termed *E. jessiae* in the former reference) also was observed in microhabitats containing relatively high sedimentation within the Little Tennessee River in Tennessee (Greenberg 1991) and Black Creek in Mississippi (Ross et al. 1987). In both systems, several other darter species were in less sedimented habitats.

Summary

Qualitative IBI classification values were relatively constant ("good" water quality classification) throughout the 5th and 6th order Powell River sites in 1988. Conversely,

quantitative IBI scores increased downstream. Increasing IBI scores were a result of 1) decreasing proportions of individuals as omnivores and as tolerant species downstream, and 2) increasing piscivore proportions, number of intolerant species, and specialized insectivores downstream. Although IBI scores did not vary with measures of sediment in shallow-water or in pool habitats, the above metrics were correlated with sedimentation, thereby indicating that subtle changes to fish assemblages had occurred due to sedimentation. These findings agree with that of Cummins (1994), who noted relatively low population levels of smallmouth bass (*Micropterus dolomieu*) and rockbass (*Ambloplites rupestris*) in the lower Powell River. Poor substrata conditions for prey items, resulting from excessive sediments, was implicated as the probable cause.

The overall IBI configuration used in my study may not be sensitive to changes in assemblage structure wrought by sedimentation since sediment levels were primarily correlated with functional metrics (e.g., "proportion of individuals as omnivores") as opposed to taxonomic metrics (e.g., "number of native species"). One taxonomic structural attribute not used as a metric, the ratio of exotic individuals to native individuals in pools, was correlated with sediment depth and might be a faithful indicator of sedimentation trends in the Powell River.

These results agree with observations on fish assemblages in Missouri streams (Berkman and Rabeni 1987; Rabeni and Smale 1995). There, the authors found that taxonomic characteristics (species richness and species diversity) were not sensitive to siltation whereas several functional attributes were. They proposed a "siltation index"

comprised of trophic and functional attributes to monitor effects of siltation in lieu of the IBI.

Assemblages in shallow-water habitats during 1988 clustered into an upstream group and a downstream group. Sedimentation level was the only significant physical difference between the two groups of sites, providing evidence that observed assemblage differences are largely due to sedimention. Relative abundances of specialized insectivores were higher downstream and omnivores were higher upstream. The LB (lithophilous benthic) reproductive guild, assumed the reproductive group most sensitive to sedimentation, did not differ between the assemblage groups.

Powell River pool assemblages of fishes did not cluster into interpretable spatially-based groups, although top carnivores did decrease in more sedimented pools. Reproductive guilds were not correlated with sedimentation levels. These results are not unexpected since pool habitats are depositional (except during floods), and likely the fish assemblages are more resistant to sedimentation.

Analyses of assemblage similarity in 1989 and in 1990 indicated that some grouping of upstream sites (5^{th} order and upper 6^{th} order locations) occurred. This finding was comparable to that determined in 1988. Most sites, particularly from the mid- and lower 6^{th} order, however, did not group into spatially-distinct clusters. Fish assemblage similarity reflected similarity in physical habitat among sites.

A summary of functional analyses over the three sample years indicated that relative abundance of omnivores increased with greater sedimentation. Abundance of

specialized insectivores tended to decrease with higher sedimentation, although no trend was statistically significant ($p \geq 0.056$). The LB reproductive group was not associated with sedimentation levels. The ASL reproductive group, however, was positively associated with sedimentation and may confer some resistance to sedimentation effects. Overall, sedimentation appears to be affecting Powell River fishes primarily through trophic disruption.

An overall sediment-tolerance classification list was derived from 1988, 1989 and 1990 sediment-tolerance classifications. Criteria used to delineate overall classifications were whether species 1) were similarly classified in more than one year and 2) were only classified in 1988 but predominantly used habitats other than riffles or runs (especially pools). Nine species were classified as sediment-intolerant and eight species were categorized as sediment-tolerant. The remainder of the species sampled were relegated to a "non-classified" group. The sediment-intolerant group was composed mostly of benthic specialized insectivores. The LB reproductive guild was equally represented in the sediment-intolerant and sediment-tolerant groups.

Sediment-intolerant species occurred in low-sedimented microhabitats relative to sediment-tolerant group members. The non-classified group was intermediate. This is confirmatory evidence that the sediment-tolerance designations previously derived from a macrohabitat perspective are valid.

Within both sediment-tolerance categories and the non-classified group, species tolerances to sedimentation very likely differ. Sediment-intolerant species that occur in

riffle and run habitats were found in lower sedimented microhabitats relative to sediment-intolerant pool species. A similar situation may hold true for the non-classified group. Several species, classified only once as sediment-tolerant, may be more tolerant of sedimentation than other group members. Using microhabitat use as an indicator, uncommon species such as *P. copelandi*, *E. camurum*, and *N. eleutherus* may be very sediment-intolerant. Their rarity prevented adequate macrohabitat-scale analysis of their sediment tolerance.

In conclusion, my findings on functional changes to fish assemblages due to sedimentation agree with previous studies on warmwater streams. Berkman and Rabeni (1987) and Rabeni and Smale (1995) also observed that omnivores increased with greater sedimentation in Missouri warmwater stream fish assemblages. These authors observed that benthic specialized insectivores and lithophilous spawner abundances decreased with higher siltation. My specialized insectivore group included both benthic and water-column-oriented species and thus was not directly comparable. Inspection of the sediment-intolerant group, however, indicates that the majority of species classified as sediment-intolerant were benthic specialized insectivores, congruent with results from the Missouri studies.

Reasons why the LB group in my study was not sensitive to sedimentation, as was found in Missouri streams, is unknown. Possibly sedimentation levels in the Powell River may not have been high enough to disrupt reproductive mechanisms.

Chapter Three

Components for a Biomonitoring Program of the Powell River in Virginia

Introduction

To develop a useful monitoring program for the Powell River system, a comprehensive management system must be in place (Crowe 1983; USFWS 1995). Monitoring is one stage or aspect of this management system, and strategic goals and objectives should direct what will be monitored and how the results will be used in managing the watershed. In the absence of a strategic plan for the Powell River watershed, I propose that the tentative goal should be to protect and maintain aquatic biodiversity. Major threats to Powell River biodiversity are primarily excess sedimentation and secondarily compromised water quality. Hence, monitoring subgoals should address the detection of changes in sediment levels and selected water quality variables.

Proper sampling design is another important factor that is critical to an effective biodiversity monitoring (biomonitoring) program (Gilbert 1987). Good sampling design, in this case, means choosing the right indicators or surrogates for biodiversity, the right places to sample, the best sampling frequency, and precise sampling techniques (Noss and Cooperrider 1994). Because selecting "indicators" often is a very difficult task in

designing effective programs (Noss and Cooperrider 1994; Meffe and Carrol 1994; Noss 1995), the majority of this section is devoted to selecting indicators for monitoring Powell River biodiversity, specifically the changes in sedimentation and water quality.

Biodiversity of an area, unfortunately, cannot be mathematically described by summing to one value (Angermeier and Karr 1994). Instead, the concept of biotic integrity (definition in Karr and Dudley 1981) is incorporated with indicator selection. First, choose an appropriate indicator or indicators to monitor a biodiversity element or elements at issue (e.g., sedimentation levels). The indicators must be sensitive or representative enough of the dynamics of the biodiversity element(s) of interest. Next, compare the state of the indicator (e.g., proportion of an assemblage that consists of a particular trophic guild) to that of the indicator state in relatively undisturbed habitat of the region. Monitoring should track, over time, the relation of the indicator being evaluated to that level found in undisturbed areas ("benchmark state"; Angermeier and Karr 1994). If, especially after management actions, the indicator state converges on the benchmark condition (positive or negative trend), then the conclusion would be that biotic integrity has improved and biodiversity has been maintained. Conversely, a further divergence of monitored indicator and benchmark states point to deteriorating biotic integrity and loss of biodiversity.

Depending upon just one indicator is obviously more risky than monitoring a suite of appropriately selected indicators. The suite of selected indicators should include biotic elements (Ohio EPA 1988a, b). It certainly is better to monitor structure and composition

of biota than to rely completely on monitoring processes as flow regime (Howarth 1991; Angermeier and Karr 1994).

The next step is to adopt an operational definition of biodiversity. From this, a conceptual biomonitoring framework can be developed that will help guide program objectives. I propose the following definition from the Keystone Center (1991) as modified by Noss and Cooperrider (1994): "Biodiversity is the variety of life and its processes. It includes the variety of living organisms, the genetic differences among them, the communities and ecosystems in which they occur, and the ecological and evolutionary processes that keep them functioning, yet ever changing and adapting."

The conceptual definition states that biodiversity is hierarchial (four levels of organization) and that each of these levels has three components (Figure 42). The four levels are genetic, population-species, community-ecosystem, and landscape. These levels can be subdivided into compositional, structural, and functional components of a nested hierarchy. On the genetic level, for instance, the number of alleles (= allelic richness) is an example of a composition component indicator, a profile of allelic relative frequencies is an example of a structure indicator, and gene flow is an example of a function indicator. Theoretically, evaluating in some way each component of each level using all taxa is how we would monitor biodiversity. In reality, it is logistically and financially impossible to monitor biodiversity by such a complete examination. Operationally, some combination of levels and components of a limited number of taxa is evaluated for monitoring biodiversity, determined by a selection of indicators that were

	Composition	Structure	Function
Genetic			
Population-Species			
Community-Ecosystem			
Landscape			

Figure 42. Operational framework for monitoring biodiversity.

evaluated empirically. These levels, components, and taxa used are called "key elements" of biodiversity. These key elements would form the basis of an inventory\monitoring program. (For a more complete description, see Essay 4A in Meffe and Carroll 1994 and the chapter on monitoring in Noss and Cooperrider 1994).

The purpose of this section is to suggest indicators or key elements that can be used to constitute a biomonitoring program for the Powell River in Virginia. Following Green's (1979) reasoning that "An impact study is best used when the results provide the basis for subsequent monitoring to detect impacts of the same type" (e.g., sedimentation, water quality deterioration), indicators primarily are derived from my work on Powell River physico-chemical habitat and fish assemblages. The indicators and thereby any biomonitoring program composed of these indicators would be focused on detecting changes in sedimentation levels and water quality.

Summary of Results from My Project Applicable to Biomonitoring

The sixth order, and possibly fifth order, Powell River flow regime is classified as intermediate between "Perennial Runoff" and "Mesic Groundwater" (Poff and Ward 1989). A quantitative description of flow regime such as that described in Poff and Ward (1989) and in Chapter One is a quantitative description of disturbance regime. Hence, flow regime characterization (designated as FR) is a biodiversity indicator in the community-ecosystem level, function component box. Although monitoring discharge has potential as a surrogate for predicting fish population dynamics (Bayley and Li 1992)

and for fisheries management (Heede and Rinne 1990), unless changes occur that drastically affect flow regimes and hydrologic classification (e.g., dam construction), this indicator is secondary in importance to indicators that monitor physico-chemical habitat.

Chemical habitat, or water quality, exhibits a distinct longitudinal gradient. Although water quality in the lower Powell River does not appear harmful to fish, certainly some tributaries have harmful water quality characteristics. Coal mining appears to primarily influence the spatial pattern of chemical habitat. Critical indicators are specific conductivity, sulphate, iron, and metals (e.g., copper, cadmium, zinc). Fecal coliforms, nitrates, and phosphates, indicators of sewage treatment efficiency and agricultural inputs, are secondary. These indicators (WR) are located within the community-ecosystem level, structure component box. States of these indicators along with potential benchmarks can be found in Chapter One and Appendix B. For example, maintenance or decline in specific conductivity within the sixth order Powell River mainstem would indicate reduced input of ions from coal mining operations. The specific conductivity benchmark might be set at the mean specific conductivity for unmined "Tributaries to 6th Order Powell River" (Appendix B).

The primary physico-chemical perturbation in the Powell River is sedimentation. Changes in sedimentation can be monitored directly and indirectly. The most quantitatively direct indicator likely is measurement of pool sediment depth (SP) by methods found in Cummins (1994) and in Chapter One (community-ecosystem, structure box). A decline in pool sediment depth would indicate an increase in biotic integrity.

Furthermore, sediments should be analyzed for toxicity and heavy metal content (McCann 1993; Yeager 1994).

Riparian zone integrity (RZ) is a landscape level, structure component indicator. Intact riparian zones are important as stream buffers against sediment runoff (Gregory et al. 1991; Rabeni and Smale 1995), and are significant components of various "Best Management Practices" (Heatwole et al. 1991). Intact riparian zones also are important sources of structural habitat such as downed logs (community-ecosystem, structure box). Thus, monitoring riparian zones on a landscape level provides indirect monitoring of sedimentation potential and structural features located within the community-ecosystem level. Although only site canopy overhang was measured in this study, riparian zones are fairly intact (personal observation). It is critical to maintain the present riparian zone and locate areas needing restoration. Riparian zone integrity can be measured as in my study or a more thorough approach can be taken (e.g., Hupp and Osterkamp 1985; Johnson 1992). A related indicator for monitoring biodiversity is land-use trends (LU; landscape, function box). Since the extent of surface coal mining is correlated with Powell River physico-chemical habitat patterns, trends in active, reclaimed, and abandoned mine lands could be important for assessing the degree of potential impacts to the river. Changes in fish assemblage structure occurred when the extent of the watershed mined was approximately 10% (Chapter One).

The biotic element of the biomonitoring program should include invertebrates and fish. Although not studied in this project, the Powell River mussel fauna is diverse but

declining (Ahlstedt and Brown 1979; Wolcott 1990). Judging from information contained in TVA (1979), Wolcott (1990), Layzer and Anderson (1992) and McCann (1993), the Powell River mussel fauna may be more sensitive to certain chemical or physical perturbations than fish. Suggested indicators to monitor include mussel species richness (MR; community-ecosystem, composition box) and assemblage-level age class structure (MAC; community-ecosystem, structure box). Periodically assessing density (MD; population-species, composition box) and age class structure (LAC; population-species, structure box) of listed mussel species also should be considered for inclusion within a biomonitoring program.

Fish indicators include both single species ("indicator species") and assemblage-level attributes. Fluctuating asymmetry levels of *C. carolinae* may be a sensitive indicator of water quality impairment or sediment toxicity from coal mining (Appendix D) (FA; population-species, structure box). As discussed in Appendix D, however, additional research is needed to ascertain the reliability of this potential indicator.

Any of the sediment-intolerant or sediment-tolerant species, such as *Etheostoma rufilineatum* or *Percina evides*, may be singly monitored (SINT\STOL; population-species, composition box) and provide a surrogate or indirect measure of the primary physico-chemical perturbation in the Powell River (sedimentation). *Erimystax insignis* also may be a good choice because it has a relatively restricted distribution, presumably caused by sedimentation. This facilitates not only monitoring from a density perspective but from a distributional one (ERI; population-species, structure box) as well. A range

expansion or abundance increase would indicate lower sedimentation and increased biotic integrity. Moreover, *E. insignis* occurs in shallow habitats and is easily recognized. Hence, non-lethal underwater observations could be employed (Ensign et al. 1995).

Other indicator species include those that are uncommon and appear, from microhabitat analysis, to be sediment-intolerant (*E. camurum*, *P. copelandi*, and *N. eleutherus*). Although these species may be rare for natural reasons and thereby may not increase dramatically in density with improved conditions (Sheldon 1987), maintenance of population levels where these species presently occur (RSP; population-species, composition box) and range extensions or reductions within the Powell River (RRSP; population-species, structure box) could be important indicators to monitor.

Ambloplites rupestris and *M. dolomieu* are special interest species from a sport fishery viewpoint (Cummins 1994). Increasing population sizes (SF; population-species, composition box) of these two species may indicate improving pool substratum conditions (less sand) for their primary prey item, crayfish. This, in turn, may indicate decreasing sedimentation.

Fish assemblage-level monitoring is more complex but may provide the most comprehensive assessment. For instance, the ratio of exotic to native species abundances in pools appears to be a dependable indicator of sedimentation (ETN; community-ecosystem, composition). The predominant introduced fish species in pools is *L. auritus*. This sunfish appears to be sediment-tolerant and may be reducing abundances of *L. megalotis* in areas of higher sedimentation. A negative trend of this indicator points to

reduced sedimentation and increased biotic integrity.

The IBI is an aggregate of indicators (IBI M1-M12) concentrated in the community-ecosystem, composition and community-ecosystem, function boxes. When used on a watershed context, IBI scores are indicators within the landscape, composition box (IBISCR). Scores generated by the IBI used in this study, however, were not correlated with sedimentation in the Powell River. A subset of functional, as opposed to taxonomic, metrics did vary with sedimentation. Moreover, shallow-water assemblage characteristics in the Powell River appeared most sensitive to sedimentation. Rabeni and Smale (1995) also found that an index composed of a reduced number of functional indicators derived from shallow-water (riffle habitat) sampling was better than the IBI at detecting effects of sedimentation to stream fish communities.

In the Powell River, a particularly important IBI functional metric to monitor is the "proportion of individuals as omnivores" in shallow-water assemblages. Sedimentation appears to primarily affect Powell River fishes through trophic disruption, with increased relative abundances of omnivores and reduced abundances of specialized insectivores. This metric should assume additional importance as an indicator variable. Benchmarks for the omnivore indicator within the 5th and 6th order Powell River could be taken from values used for scoring criteria of regional IBIs (e.g., a rating of 5 for proportion of individuals as omnivores < 15%; Saylor and Ahlstedt 1990). These values may not be optimal because deep and shallow-water assemblages are combined. More accurate benchmarks may be derived from sampling shallow-water habitats in relatively

undisturbed regional streams (proportion of omnivores ~ 3% and specialized insectivores ~ 90% in the North Fork Holston River above Saltville, Virginia, Chapter One).

Multivariate techniques to assess assemblage similarities (MVT; community-ecosystem, composition and landscape, composition boxes) in the Powell River is another viable monitoring tool. Physical habitat differences (excluding sedimentation) do not appear to be great between the fifth and upper sixth order versus the mid- to lower sixth order Powell River. Hence, the potential exists for assemblage similarity from PRM 172.2 to PRM 117.3. Monitoring could investigate whether the upstream and downstream assemblages become more alike in response to changing sedimentation rates (e.g., in response to mine-land reclamation activities). Habitat improvement could be concluded if the upstream assemblages experienced species additions or greater abundance of sediment-intolerant species. In addition, the upstream sites should respond to improving conditions by decreased proportion of the omnivore trophic guild.

A Proposed Minimum Biomonitoring Framework for the Powell River in Virginia

The complete set of proposed indicators for monitoring Powell River aquatic biodiversity are fairly comprehensive (Figure 43). The genetic level, although possibly important for some mussel species, is conceived of generally as a fine-tuning focus (Meffe and Carroll 1994). Genetic issues for the Powell River fish fauna are likely of minor importance relative to the other levels. Of the indicators listed, what subset should

be monitored depends upon the strategic and operational plan (including budget) for the watershed. I would propose, at minimum, that indicators selected for monitoring the Powell River system include:

1) sediment-intolerant species density and distribution (particularly *Etheostoma rufilineatum*, *Percina evides*, or *Erimystax insignis*);
2) proportion of individuals as omnivores in shallow-water habitats;
3) proportion of fish individuals in pools as redbreast sunfish and common carp;
4) sediment depth in pools;
5) sulphate, iron, specific conductivity, and heavy metal concentrations in water; and
6) mussel assemblage structure.

All indicators, except number 5, focus entirely or in part on sedimentation monitoring. This is appropriate because sedimentation is seemingly the primary anthropogenic impairment to Powell River fish assemblages and possibly to mussel populations as well. Using a diversity of indicators, as opposed to only one or two, should increase the power of the monitoring program to detect changes in instream sedimentation levels.

An important fish assemblage-level indicator is the proportion of individuals as omnivores in shallow-water habitats (Indicator 2). Fish trophic guild patterns had strong correlations with sedimentation in the Powell River. In addition, these trophic guild

patterns as affected by sedimentation may be consistent over large geographic areas (Rabeni and Smale 1995). Although sampling for this indicator involves assemblage level examination (Community-ecosystem, compostion box), I highly recommend incorporating this level of investigation in a biomonitoring program for the Powell River. An important inherent characteristic of an indicator at the guild level is that it is expected to be less variable than indicators at the population level (e.g., indicator number 3; Bayley and Li 1992). Sampling should be restricted to shallow-water habitats, preferably using pre-positioned area shocker methodology.

I suggest using indicators 5 and 6 as tools to monitor chemical habitat, particularly water quality as affected by coal mining. Water quality has and is being impaired by watershed activities, although effects to fish assemblages appear secondary to that of sedimentation. Indicator 5 is a direct measurement and indicator 6 is an indirect measurement of water quality (Layzer and Anderson 1992). In addition, mussel assemblages may be more responsive overall than fish assemblages to chemical habitat perturbations from coal mining (Kitchel et al. 1981; Wolcott 1990; Layzer and Anderson 1992; McCann 1993). As such, indicator 6 may be a particularly sensitive indicator of chemical habitat changes.

To conclude, a comprehensive and powerful biomonitoring program for the Powell River would be composed of all the indicators diagrammed in Figure 43. These indicators are tools to monitor sedimentation and water quality changes, two modes of impairment to Powell River fauna. A biomonitoring program of this magnitude also

would be expensive in terms of staff, time, and equipment. The subset of indicators that I have proposed would be less expensive to employ but should still retain sufficient power to monitor sedimentation and water quality changes. Particular consideration should be given to the *biotic* indicators for inclusion into a biomonitoring program (e.g., indicators 1 and 2). It is preferable to monitor structure and composition of biotic indicators than to rely solely on abiotic elements or system functions (Ohio EPA 1988a; Angermeier and Karr 1994).

Regarding the other components of sampling design, some subset of my study sites could continue to be used for indicator sampling locations. Mussel investigations might use sites sampled by Wolcott (1990). This would take advantage of pre-existing site data on indicators (Cochran 1977). Alternatively, a new suite of sites could be selected through a probability sampling framework (Gilbert 1987; Thompson 1992; Overton and Stehman 1995), at the appropriate spatial sampling scale (Hankin and Reeves 1988; Levin 1992; Dollof et al. 1993). Methods to choose the best sampling frequency and sample number are most straightforward within the Population-species, composition component box using software tools as MONITOR (by James P. Gibbs, Department of Biology, 419 OML, P.O. Box 208140, Yale University, New Haven, Connecticut, 06520-8104). Finally, better precision can be obtained in several ways, especially by increasing sample number or by using efficient sampling techniques (Zar 1984; Appendix L).

	Composition	Structure	Function
Genetic			
Population-Species	MD SINT\STOL RSP SF	LAC FA ERI RRSP	
Community-Ecosystem	IBI M1-M10 MR ETN MVT	WQ SP RZ MAC	IBI M11, M12 FR
Landscape	IBISCR MVT	RZ	LU

Figure 43. Location of monitoring indicators within a biodiversity framework. Indicator acronyms are found in the text.

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Appendix A. Locations of water quality sampling sites in the Powell River drainage of Virginia.

Table 34. Location descriptions for Powell River drainage water quality sites: 1989, 1990. "Downstream of..." or "Upstream of ..." in the descriptions denote sites less than 153 m (500 ft) away from the reference point. Sites are listed from upstream to downstream. Streams are classified as MTRIBCP (mined tributary in the Cumberland Plateau physiographic province), UMTRIBCP (unmined tributary in the Cumberland Plateau), and UMTRIBVR (unmined tributary in the Valley and Ridge physiographic province).

Site Identification Number	Description
Mainstem Powell River (Mined watershed in Wise and Lee counties, Virginia)	
P1	PRM 199.3; by County Route 610 (Wise County).
P2	PRM 198.9; by County Route 610 (Wise County).
P3	PRM 198.0; County Route 623 Crossing (Wise County).
P4	PRM 197.5; County Route 620 crossing, downstream of Parmont Deep Mine #6 (Wise County).
P5	PRM 195.3; upstream of confluence with Bear Branch at Needmore, Virginia (Wise County).
P6	PRM 194.4; upstream of coke ovens at Dorchester, Virginia (Wise County).
P7	PRM 192.6; downstream of confluence with Thacker Branch (Wise County).

Table 34. (continued)

Site Identification Number	Description
P8	PRM 192.1; downstream of sewage treatment plant in Josephine, Virginia (Wise County).
TA	PRM 185.7; upstream of confluence with Mill Branch (Wise County).
TB	PRM 182.4; approximately 915 meters below confluence with Callahan Creek and 825 meters above confluence with Looney Creek (and Bullet Mine Number 11); in Appalachia, Virginia (Wise County).
TC	PRM 180.7; approximately 1,915 meters below confluence with Looney Creek; by Alternate Highway 58, between Appalachia and Big Stone Gap, Virginia (Wise County).
P9	PRM 176.1; County Route 605 crossing (Wise County).
WQ1	PRM 171.2; by County Route 621, downstream of Olinger, Virginia.
WQ2	PRM 167.0; upstream of Alternate Highway 58 crossing near Dryden, Virginia
WQ3	PRM 157.8; 1.9 km (1.2 mi) above confluence with the North Fork Powell River
WQ4	PRM 153.4; Schaffer Ford, by County Route 640
WQ5	PRM 147.7; downstream of Laurel Branch confluence, a ford off County Route 783

Table 34. (continued)

Site Identification Number	Description
WQ6	PRM 138.3; County Route 654 crossing, Hurricane Bridge
T1	PRM 133.0; upstream of confluence with Wallen Creek
WQ7	PRM 130.6; County Route 758 crossing, Flanary Bridge
P10	PRM 128.5; Hall Ford
WQ8	PRM 123.8; Snodgrass Ford
T4	PRM 120.3; County Route 833 crossing
WQ9	PRM 117.3; Fletcher Ford
Bear Branch drainage (Wise County)- MTRIBCP	
BEB1	Bear Branch; approximately 1,769 meters upstream of confluence with Powell River, at junction of County Routes 623 and 621

Table 34. (continued)

Site Identification Number	Description
BEB2	Bear Branch; approximately 915 meters upstream of confluence with Powell River
BEB3	Bear Branch; approximately 490 meters upstream of confluence with Powell River, at Wilson Chapel, Virginia
Thacker Branch drainage (Wise County)- MTRIBCP	
THB1	Thacker Branch; farthest upstream site, near headwater, County Route 618 crossing
TUN1	Unnamed tributary to Thacker Branch; upstream of confluence with Thacker Branch, this confluence is 1,400 m upstream of Thacker Branch - Powell River confluence
THB2	Thacker Branch; 1,145 m upstream of confluence with Powell River
THB3	Thacker Branch; 185 m upstream of confluence with Powell River

Table 34. (continued)

Site Identification Number	Description
Benges Branch (Wise County)- UMTRIBCP	
BNB1	Benges Branch; approximately 1,220 meters upsteam of confluence with Powell River, at Main Street bridge in Norton, Virginia
Carding Machine Branch (Wise County)- UMTRIBCP	
CMB1	Carding Machine Branch; approximately 670 meters upstream of confluence with Powell River, at highway 70 bridge
Black Creek (Wise County)- MTRIBCP	
BLC1	Black Creek; upstream of confluence with Powell River

Table 34. (continued)

Site Identification Number	Description
Roaring Fork drainage (Wise County)- MTRIBCP	
ROF1	Roaring Fork; approximately 458 meters above confluence with Potcamp Fork, near Roaring Fork, Virginia
POF1	Potcamp Fork; approximately 610 meters above confluence with Roaring Fork, near Roaring Fork, Virginia
ROFUN1	Unnamed tributary to Roaring Fork; upstream of confluence with Roaring Fork
CAC1	Canepatch Creek; approximately 210 meters above confluence with Roaring Fork
ROF2	Roaring Fork; approximately 365 meters above confluence with Powell River
Mill Branch (Wise County)- MTRIBCP	
MIB1	Mill Branch; approximately 300 meters above confluence with Powell River

Table 34. (continued)

Site Identification Number	Description
Callahan Creek drainage (Wise County)- MTRIBCP	
MLC1	Mud Lick Creek; downstream of confluence with McHenry Fork, near Roda, Virginia
CLC1	Callahan Creek; upstream of confluence with unnamed tributary draining Possum Trot Hollow, near Stonega, Virginia
CLC2	Callahan Creek; approximately 245 meters above confluence with Mud Lick Creek
CLC3	Callahan Creek; approximately 855 meters below confluence with Mud Lick Creek, upstream of confluence with Kelly Branch
KEB1	Kelly Branch; upstream of confluence with Callahan Creek
PEC1	Preacher Creek; approximately 300 meters above confluence with Callahan Creek, near Andover, Virginia
CLC4	Callahan Creek; approximately 550 meters above confluence with Powell River, in Appalachia, Virginia

Table 34. (continued)

Site Identification Number	Description
Looney Creek drainage (Wise County)- MTRIBCP	
LOC1	Looney Creek; approximately 1885 meters above confluence with Powell River, near Inman, Virginia
PIC1	Pigeon Creek; approximately 1800 meters above confluence with Looney Creek, near Imboden, Virginia
South Fork Powell River (Wise County)- UMTRIBVR	
SF1	SFPRM 6.1; upstream of confluence with Big Spring unnamed tributary, County Route 612 crossing (Wise County).
WQSF	SFPRM 0.1; Alternate Highway 58 crossing (Wise County).

Table 34. (continued)

Site Identification Number	Description
North Fork Powell River drainage (Lee County)- MTRIBCP	
Mainstem North Fork Powell River- MTRIBCP (except for stations WQNF and NF6 which are MTRIBVR)	
NF1	NFPRM 17.1; upstream of confluence with Bundy Creek
NF2	NFPRM 14.9; upstream of confluence with Cox Creek at Delvale, Virginia
NF3	NFPRM 10.4; upstream of confluence with Jones Creek at Purcell, Virginia
NF4	NFPRM 8.3; upstream of confluence with Rocklick Branch
NF5	NFPRM 6.1; upstream of confluence with Straight Creek
WQNF	NFPRM 4.7; at base of Stone Mountain (Cumberland Plateau); above Pennington Gap at County Route 621 crossing
NF6	NFPRM 2.3; Alternate Highway 58 crossing at Pennington Gap, Virginia (upstream of the Pennington Gap Sewage Treatment Plant)

Table 34. (continued)

Site Identification Number	Description
	Craborchard Creek drainage- MTRIBCP (except for Wells Branch)
CRC1	Craborchard Creek; approximately 915 meters above confluence with Moore Branch; at County Route 623 bridge
CRC2	Craborchard Creek; downstream of confluence with Moore Branch, at County Route 606 bridge, near Keokee, Virginia
WEB1	Wells Branch (UMTRIBCP); approximately 610 meters above confluence with Craborchard Creek, at County Route 623 bridge
	Bundy Creek- MTRIBCP
BUC1	Bundy Creek; upstream of confluence with North Fork Powell River
	Cox Creek- MTRIBCP
COC1	Cox Creek; approximately 183 meters above confluence with North Fork Powell River

Table 34. (continued)

Site Identification Number	Description
Jones Creek- MTRIBCP	
JOC1	Jones Creek; approximately 183 meters above confluence with Mud Creek, near Robbins Chapel, Virginia
MUC1	Mud Creek; upstream of confluence with Jones Creek
JOC2	Jones Creek; downstream of confluence with Mud Creek, at Robbins Chapel, Virginia
JOC3	Jones Creek; upstream of confluence with Reeds Creek and approximately 760 meters upstream of confluence with North Fork Powell River
REC1	Reeds Creek; approximately 670 meters above confluence with Jones Creek, at County Route 628 bridge
Rocklick Branch- MTRIBCP	
ROB1	Rocklick Branch; upstream of confluence with North Fork Powell River

Table 34. (continued)

Site Identification Number	Description
Straight Creek drainage- MTRIBCP	
STC1	Straight Creek; approximately 180 meters below confluence with tributary draining Miller Cove, at gate of Straight Creek Coal Processing Company, near Monarch, Virginia
STC2	Straight Creek; upstream of confluence with Gin Creek, at Turners Siding
GIC1	Gin Creek; upstream of confluence with Straight Creek, at Turners Siding
BAT1	Baileys Trace; upstream of confluence with Straight Creek, at County Route 636 bridge in St. Charles, Virginia
STC3	Straight Creek; downstream of confluence with Baileys Trace, at County Route 636 bridge in St. Charles, Virginia
BIB1	Big Branch; approximately 580 meters above confluence with Straight Creek, at County Route 628 bridge near St. Charles, Virginia
PUC1	Puckett Creek; upstream of confluence with Straight Creek, at Maness, Virginia
STC4	Straight Creek; upstream of confluence with Straight Creek, in Stone Creek, Virginia

Table 34. (continued)

Site Identification Number	Description
	Stone Creek drainage- MTRIBCP (except for Bergen Branch)
SOC1	Stone Creek; upstream of confluence with Bergen Branch
BGB1	Bergen Branch (UMTRIBCP); upstream of confluence with Stone Creek, at Highway 421 bridge
ELC1	Ely Creek; approximately 275 meters above confluence with Stone Creek
WOB1	Wolf Branch; upstream of confluence with Stone Creek, at Highway 421 bridge
SOC2	Stone Creek; upstream of confluence with Straight Creek, in Stone Creek, Virginia
STC5	Straight Creek; upstream of confluence with North Fork Powell River and approximately 550 meters below confluence with Stone Creek

Table 34. (continued)

Site Identification Number	Description
Town Branch- (Lee County) UMTRIBVR	
T5	Town Branch; approximately 2,800 meters above confluence with Powell River, at County Route 654 bridge
Wallen Creek- (Lee County) UMTRIBVR	
T2	Wallen Creek; approximately 160 meters above confluence with Powell River
Hardy Creek- (Lee County) UMTRIBVR	
T3	Hardy Creek; approximately 1,610 meters above confluence with Powell River, at County Route 660 bridge

Appendix B. Water quality values for selected drainages within the Powell River drainage of Virginia.

Table 35. Water quality values for selected drainages in the Powell River drainage in Virginia. Data summarized for all sample dates[a].

	Water Quality Variable (mean ± S.D.) (n = number of samples)					
Drainage	Alkalinity (mg/L)	BOD[b] (mg/L)	Calcium (mg/L)	Specific Conductivity[c] (µsiemens/cm)	Hardness (mg/L)	Iron (mg/L)
Powell River in Cumberland Plateau (PRM 199.3 - 180.7)	68.5 ± 27.5 n = 7	* n = 0	84.2 ± 41.6 n = 14	302 ± 146 n = 23	173 ± 88 n = 12	0.42 ± 0.15 n = 9
5th Order Powell River (PRM 176.1 - 157.8)	105.5 ± 20.2 n = 11	2.79 ± 1.72 n = 5	75.4 ± 15.8 n = 6	295 ± 115 n = 11	139 ± 16 n = 8	0.19 ± 0.05 n = 9
6th Order Powell River (PRM 153.4 - 117.3)	96.7 ± 16.1 n = 26	1.97 ± 2.11 n = 15	76.9 ± 14.2 n = 14	254 ± 101 n = 23	129 ± 17 n = 18	0.14 ± 0.06 n = 18
Bear Branch	67.0 n = 1	* n = 0	* n = 0	551 ± 129 n = 7	267 n = 1	0.61 ± 0.47 n = 3

Table 35. (continued)

Drainage	Water Quality Variable (mean ± S.D.) (n = number of samples) Alkalinity (mg/L)	BOD (mg/L)	Calcium (mg/L)	Specific Conductivity (µsiemens/cm)	Hardness (mg/L)	Iron (mg/L)
Thacker Branch	105.5 ± 7.8 n = 2	* n = 0	172.0 n = 1	650 ± 119 n = 5	451 ± 98 n = 2	3.04 ± 3.59 n = 2
Benges Branch\ Carding Machine Branch	46.0 n = 1	* n = 0	14.0 n = 1	40 ± 9 n = 3	39 ± 23 n = 2	0.14 n = 1
Black Creek	35.0 n = 1	* n = 0	* n = 0	624 ± 209 n = 3	561 n = 1	1.05 n = 1
Roaring Fork	137.0 n = 1	* n = 0	41.7 ± 11.7 n = 3	222 ± 88 n = 8	177 ± 34 n = 2	0.28 ± 0.07 n = 2
Mill Branch	* n = 0	* n = 0	* n = 0	238 n = 1	* n = 0	* n = 0
Callahan Creek drainage	132.0 n = 0	* n = 0	41.5 ± 7.8 n = 2	195 ± 64 n = 9	139 n = 1	0.15 n = 1

Table 35. (continued)

Drainage	Water Quality Variable (mean ± S.D.) (n = number of samples) Alkalinity (mg/L)	BOD (mg/L)	Calcium (mg/L)	Specific Conductivity (µsiemens/cm)	Hardness (mg/L)	Iron (mg/L)
Looney Creek drainage	* n = 0	* n = 0	50.5 ± 3.5 n = 2	218 ± 11 n = 2	* n = 0	* n = 0
South Fork Powell River	85.0 n = 1	* n = 0	11.0 n = 1	139 ± 82 n = 4	114 n = 1	* n = 0
North Fork Powell River mainstem	47.0 ± 26.9 n = 2	* n = 0	47.0 ± 14.1 n = 2	197 ± 126 n = 8	76 ± 13 n = 3	0.41 ± 0.16 n = 3
Tributaries above Straight Creek confluence[d]	* n = 0	* n = 0	* n = 0	* n = 0	* n = 0	* n = 0
Straight Creek drainage	* n = 0	* n = 0	33.0 n = 1	247 ± 57 n = 8	54 n = 1	0.25 n = 1

Table 35. (continued)

Drainage	Water Quality Variable (mean ± S.D.) (n = number of samples) Alkalinity (mg/L)	BOD (mg/L)	Calcium (mg/L)	Specific Conductivity (µsiemens/cm)	Hardness (mg/L)	Iron (mg/L)
Stone Creek drainage	* n = 0	* n = 0	* n = 0	129 ± 100 n = 4	* n = 0	* n = 0
Tributaries to 6th Order Powell River	155.0 ± 2.8 n = 2	* n = 0	55.0 n = 1	161 ± 102 n = 5	173 ± 1 n = 2	* n = 0

Table 35. (continued)

Stream Type	Water Quality Variable (mean ± S.D.) (n = number of samples) Manganese (mg/L)	Nitrate (mg/L)	pH (units)	Total Phosphorus (mg/L)	Sulphate (mg/L)	Turbidity (FTU)
Powell River in Cumberland Plateau (PRM 199.3 - 180.7)	0.16 ± 0.11 n = 9	* n = 0	7.99 ± 0.30 n = 14	* n = 0	124.7 ± 54.1 n = 23	107.2 ± 179.0 n = 22
5th Order Powell River (PRM 176.1 - 157.8)	0.00 ± 0.00 n = 6	0.67 ± 0.14 n = 10	8.06 ± 0.26 n = 14	0.11 ± 0.09 n = 7	88.6 ± 36.6 n = 17	37.8 ± 63.8 n = 32
6th Order Powell River (PRM 153.4 - 117.3)	0.00 ± 0.00 n = 9	0.61 ± 0.14 n = 26	8.14 ± 0.27 n = 37	0.08 ± 0.05 n = 21	63.4 ± 22.1 n = 42	15.8 ± 27.3 n = 83
Bear Branch	0.27 ± 0.31 n = 3	* n = 0	8.29 ± 0.10 n = 2	* n = 0	355.0 ± 132.5 n = 6	11.0 n = 1

Table 35. (continued)

Stream Type	Water Quality Variable (mean ± S.D.) (n = number of samples) Manganese (mg/L)	Nitrate (mg/L)	pH (units)	Total Phosphorus (mg/L)	Sulphate (mg/L)	Turbidity (FTU)
Thacker Branch	2.20 ± 2.69 n = 2	* n = 0	7.78 n = 1	* n = 0	445.0 ± 104.7 n = 4	8.0 ± 4.2 n = 2
Benges Branch\ Carding Machine Branch	0.00 n = 1	* n = 0	7.71 n = 1	* n = 0	9.0 ± 1 n = 3	4.5 ± 0.7 n = 2
Black Creek	1.30 n = 1	* n = 0	7.59 n = 1	* n = 0	328.3 ± 143.7 n = 3	11.0 n = 1
Roaring Fork	0.10 ± 0.00 n = 2	* n = 0	8.26 ± 0.10 n = 2	* n = 0	83.6 ± 36.9 n = 7	27.2 ± 23.1 n = 5
Mill Branch	* n = 0	* n = 0	* n = 0	* n = 0	135 n = 1	17.0 n = 1
Callahan Creek drainage	0.00 n = 1	* n = 0	* n = 0	* n = 0	60.3 ± 19.0 n = 8	21.9 ± 11.9 n = 6

Table 35. (continued)

Stream Type	Water Quality Variable (mean ± S.D.) (n = number of samples)					
	Manganese (mg/L)	Nitrate (mg/L)	pH (units)	Total Phosphorus (mg/L)	Sulphate (mg/L)	Turbidity (FTU)
Looney Creek drainage	* n = 0	* n = 0	* n = 0	* n = 0	56.3 ± 12.4 n = 2	33.0 n = 1
South Fork Powell River	* n = 0	* n = 0	8.18 ± 0.47 n = 2	* n = 0	13.0 ± 5.6 n = 3	72.7 ± 178.3 n = 8
North Fork Powell River mainstem	0.03 ± 0.06 n = 3	0.20 n = 1	7.85 ± 0.07 n = 2	* n = 0	73.2 ± 30.6 n = 7	27.7 ± 44.6 n = 11
Tributaries above Straight Creek confluence[d]	* n = 0	* n = 0	* n = 0	* n = 0	* n = 0	* n = 0
Straight Creek drainage	0.00 n = 1	* n = 0	7.91 ± 1.08 n = 2	* n = 0	93.3 ± 40.4 n = 3	* n = 0

Table 35. (continued)

Stream Type	Water Quality Variable (mean ± S.D.) (n = number of samples)					
	Manganese (mg/L)	Nitrate (mg/L)	pH (units)	Total Phosphorous (mg/L)	Sulphate (mg/L)	Turbidity (FTU)
Stone Creek drainage	* n = 0	* n = 0	7.10 n = 1	* n = 0	97.5 n = 1	* n = 0
Tributaries to 6th Order Powell River	* n = 0	* n = 0	8.17 ± 0.12 n = 4	* n = 0	3.14 ± 2.9 n = 7	13.0 ± 22.8 n = 22

Table 35. (continued)

Stream Type	Water Quality Variable (mean ± S.D.) (n = number of samples) Total Solids (mg/L)	Total Dissolved Solids (mg/L)	Total Suspended Solids (mg/L)	Dissolved Oxygen Saturation (%)
Powell River in Cumberland Plateau (PRM 199.3 - 180.7)	240.0 n = 1	386.7 ± 114.6 n = 3	44.67 ± 9.90 n = 4	* n = 0
5th Order Powell River (PRM 176.1 - 157.8)	250.3 ± 46.7 n = 8	241.4 ± 66.3 n = 2	20.60 ± 20.7 n = 3	87.5 ± 14 n = 10
6th Order Powell River (PRM 153.4 - 117.3)	193.6 ± 45.2 n = 24	149.0 ± 25.6 n = 5	28.10 ± 35.2 n = 9	89.1 ± 9 n = 29
Bear Branch	* n = 0	599.0 ± 323.0 n = 3	17.33 n = 1	* n = 0

Table 35. (continued)

Stream Type	Water Quality Variable (mean ± S.D.) (n = number of samples) Total Solids (mg/L)	Total Dissolved Solids (mg/L)	Total Suspended Solids (mg/L)	Dissolved Oxygen Saturation (%)
Thacker Branch	* n = 0	826.7 n = 1	9.25 ± 4.60 n = 2	* n = 0
Benges Branch\ Carding Machine Branch	* n = 0	* n = 0	9.00 ± 0.00 n = 2	* n = 0
Black Creek	* n = 0	324.2 n = 1	12.00 n = 1	* n = 0
Roaring Fork	* n = 0	* n = 0	40.8 ± 31.7 n = 5	* n = 0
Mill Branch	* n = 0	22.5 n = 1	24.67 n = 1	* n = 0
Callahan Creek drainage	* n = 0	136.7 n = 1	28.79 ± 6.24 n = 7	* n = 0

Table 35. (continued)

Stream Type	Water Quality Variable (mean ± S.D.) (n = number of samples)			
	Total Solids (mg/L)	Total Dissolved Solids (mg/L)	Total Suspended Solids (mg/L)	Dissolved Oxygen Saturation (%)
Looney Creek drainage	* n = 0	* n = 0	33.70 ± 25.00 n = 2	* n = 0
South Fork Powell River	* n = 0	* n = 0	* n = 0	* n = 0
North Fork Powell River mainstem	140.0 n = 1	112.5 n = 1	50.67 n = 1	99.4 n = 1
Tributaries above Straight Creek confluence[d]	* n = 0	* n = 0	* n = 0	* n = 0
Straight Creek drainage	* n = 0	240.0 n = 1	* n = 0	* n = 0

Table 35. (continued)

Stream Type	Water Quality Variable (mean ± S.D.) (n = number of samples)			
	Total Solids (mg/L)	Total Dissolved Solids (mg/L)	Total Suspended Solids (mg/L)	Dissolved Oxygen Saturation (%)
Stone Creek drainage	* n = 0	* n = 0	* n = 0	* n = 0
Tributaries to 6th Order Powell River	* n = 0	* n = 0	21.5 n = 1	* n = 0

[a] Sample dates: 6/05/89, 6/06/89, 7/31/89, 8/18/89, 8/21/89, 8/29/89, 9/07/89, 9/10/89, 9/21/89, 9/22/89, 9/25/89, 9/26/89, 9/27/89, 11/15/89, 12/14/89, 2/05/90, 2/27/90, 3/22/90, 5/13/90, 8/14/90, 11/20/90, and 12/06/90

[b] Biological Oxygen Demand

[c] Referenced to 25 °C

[d] North Fork Powell River drainage

Appendix C. Embeddedness validation study for measurement.

Introduction

The purpose of this study was to determine whether visual embeddedness measurements adequately differentiated between substratum with varying fine sedimentation levels (particle size < 2 mm). Technique validation is critical because sedimentation appears to be the primary physico-chemical perturbation to fish assemblages in the 5th and 6th order Powell River (Chapter One). Visual embeddedness estimates were the sole approach used to measure sedimentation in shallow-water habitats.

Methods

A total of twenty-three samples of deposited sediment was taken from nine mainstem sites (PRM 156.0, 153.9, 149.3, 144.6, 136.7, 127.2, 123.0, 120.4, 117.3) and one tributary site (Hardy Creek at 661 bridge crossing). Sample points were selected to provide a diverse array of embeddedness classifications. After an embeddedness estimate was made at a point (procedures in Platts et al. 1983), a 1 m high × 0.6 m diameter metal cylinder was sunk into the substratum and secured to prevent leakage from the surrounding water column. Ten surface substratum estimates (see Chapter One methods) were taken along the diameter of the sample circle. Depth of water was measured to the nearest 1.0 cm, and the substratum was vigorously mixed by hand for 2 min. This disturbance suspended sediment into the water column. A 0.5 L subsample of the

sediment and water mixture was taken and transported back to the laboratory.

At the laboratory, the sample was transferred to a 1000 ml beaker and homogenized. One 25 ml sample was pipetted off, placed in a 35 ml centrifuge tube, and centrifuged at 1930 RPM for 10 min. The supernatant was decanted, the remaining sediment was placed into an aluminum weighing tray, dried in an oven at 60 ^{0}C, and cooled in a desiccator for a minimum of 12 hr. Dried sediment was weighed in a Mettler balance to the nearest 0.0001 g. Final sediment weight was adjusted by percent difference between the corresponding cylinder water volume subsampled in the field and the shallowest water volume sampled. For each sample, a mean substratum value was calculated. Relationships between the five embeddedness categories (1 = highest, 5 = lowest sedimentation) and sediment weight were evaluated with the Spearman's Rank Correlation test.

Results

The validation study for embeddedness estimates supported the assumption that degrees of sedimentation could be visually estimated. Embeddedness was negatively correlated with sediment weight ($\rho = -0.613$, $p < 0.0019$; Figure 44). Samples (n=23) used for this analysis were extracted from substrata with embeddedness rating means ranging from 3.3 to 4.3. When a subset of samples (n=12) extracted from predominantly gravel substrata (mean 3.3-3.9) was analyzed, the correlation improved ($\rho = -0.749$, $p < 0.005$; Figure 45). The more homogeneous substratum samples used for the second

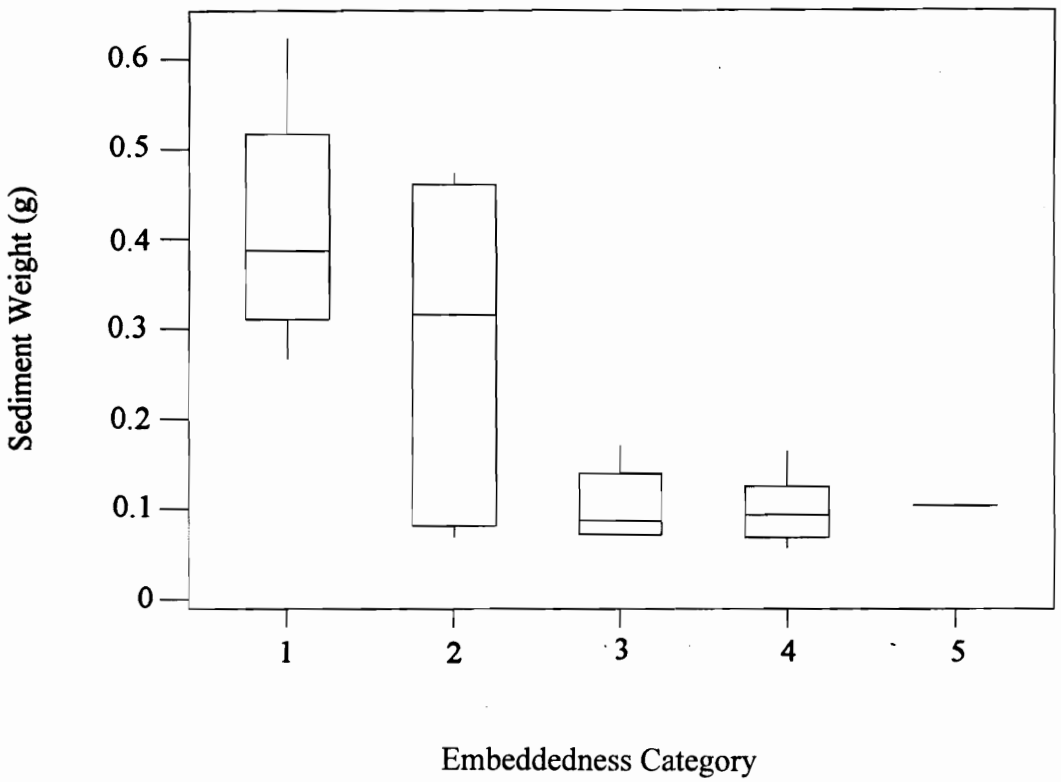

Figure 44. Box plots of sediment dry-weight within embeddedness categories. Average substratum particle size sampled ranged from 3.3-4.3.

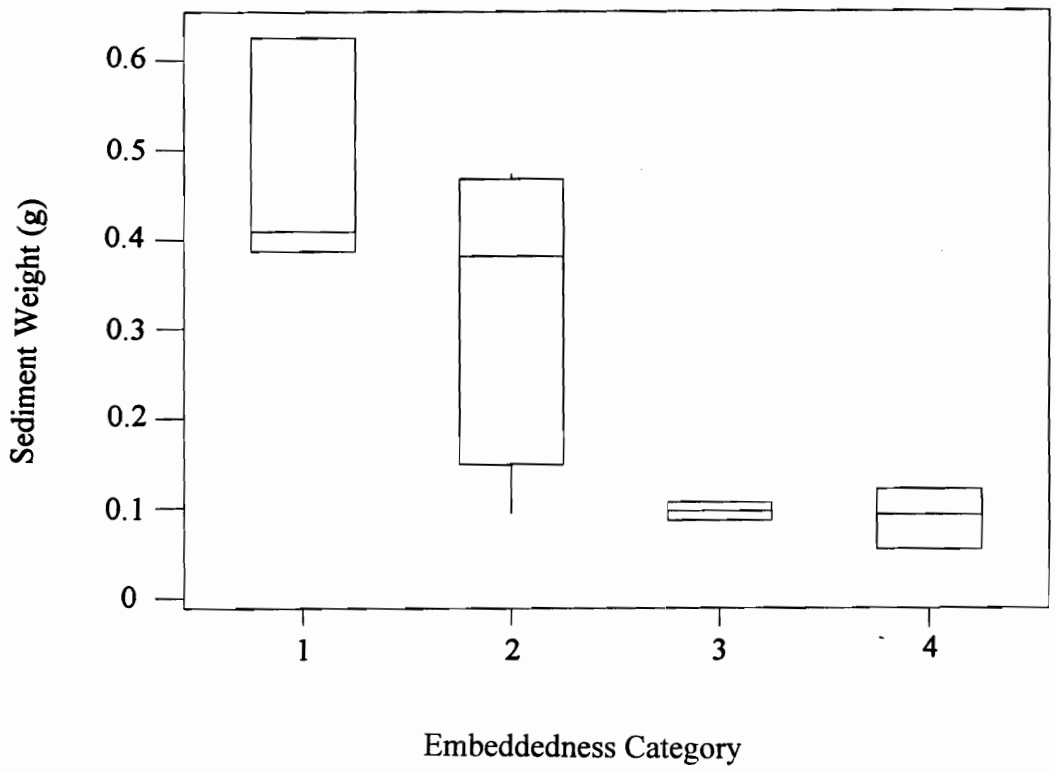

Figure 45. **Box plots of sediment dry-weight within embeddedness categories. Average substratum particle size sampled ranged from 3.3-3.9.**

analysis likely improved extraction and certainly point out the difficulty of obtaining good quantitative measurements of sedimentation from heterogeneous substrata.

In conclusion, this study indicated that differential degrees of sedimentation could be visually estimated. Since visual estimates of shallow-water sedimentation in 1988 also were significantly correlated with sediment depths in adjacent pools (Chapter One), embeddedness estimates as used in the Powell River studies appear to be a valid tool for determining species sediment-tolerance classifications.

Appendix D. Fluctuating asymmetry of the banded sculpin (*Cottus carolinae*).

Introduction

Fluctuating asymmetry (FA) is nondirectional deviation from bilateral symmetry (Van Valen 1962). The FA results from genetically or environmentally-induced developmental instability (Emlen et al. 1993). Especially because of the latter factor, FA has been proposed as a early-warning tool in monitoring programs (Valentine and Soule 1973; Leary and Allendorf 1989, 1991), although additional research is needed to better characterize the sensitivity and consistency of FA to detect human-induced environmental stressors (Ames et al. 1979; Emlen et al. 1993).

Laboratory studies have shown increasing FA in fish with increasing water concentrations of DDT in grunion, *Leuresthes tenius* (Valentine and Soule 1973), increasing deviation from normal rearing temperatures in chum salmon, *Oncorhyncus keta* (Beacham 1990) and exposure to erythromycin in rainbow trout, *Oncorhyncus mykiss* (Leary and Peterson 1990). Some field studies have shown increased FA in fish from mercury-contaminated ponds (Ames et al. 1979), and in fish from acidic lakes (Jagoe and Haines 1985). Not all species examined in these studies, however, exhibited differences in asymmetry level between populations from stressed and "control" environments. Although Wiener and Rago (1987) concluded that FA in adult bluegill sunfish, *Lepomis macrochirus*, was insensitive as a potential indicator of low pH, Leary and Allendorf (1991) found significantly higher asymmetry in bluegills from acidic lakes upon re-analysis of Wiener and Rago's (1987) data.

The Powell River lower reach in Virginia had a distinct water quality gradient (Chapter One). Several variables (sulphate, iron, turbidity, specific conductivity) have elevated concentrations upstream due to coal mining activities. In an attempt to determine the importance of water quality as a factor affecting fish assemblage structure, measured water quality variables were compared to standards set by the Virginia Water Control Board (VSWCB 1988b, 1989) and summarized by a water quality index (NSF-WQI_m; Ott 1978). The index and the comparisons indicated that water quality should not be a significant factor affecting fish assemblages. Unfortunately, measuring water quality variables and converting to an index or comparison to standards are only indirect measures of projected effects on fish assemblages. Measures of water quality (i.e., not investigating biotic characteristics directly) often have severe short-comings for detecting perturbations to fish assemblages (Ohio EPA 1988a,b). A more direct, biotic measure of water quality influences is needed.

The measurement of FA has promise as a biotic indicator of water quality effects on fish. Questions asked in this study:

1) was FA correlated with contamination either in the water column (deteriorated water quality) or in the sediments (percent coal fines); and
2) were water quality differences within the lower reach of the Powell River sufficient to cause differential changes to fish assemblages.

The approaches used to address these questions were to measure FA in banded sculpins (*Cottus carolinae*), describe the spatial patterns of banded sculpin FA in the lower Powell

River system, and to compare the spatial distribution of FA values with water quality and with percent coal fines in sediments. The underlying assumption is that banded sculpin FA, if associated with water quality, indicates that water quality deterioration cannot be discounted as a significant factor affecting fish assemblages in the lower Powell River.

Methods

Banded sculpins, *Cottus carolinae,* were examined for fluctuating asymmetry (FA). Banded sculpins were selected as the test fish for the study because they do not appear sensitive to sedimentation levels in the Powell River (Chapter Two), they have limited movements (Greenberg and Holtzman 1987), are common (Etnier and Starnes 1993; Jenkins and Burkhead 1994), and they apparently avoid streams within the Appalachian Plateau (Jenkins and Burkhead 1994). Limited movements help ensure that populations sampled have high site fidelity, and their FA levels are a result of conditions at the capture site. Banded sculpins are relatively common and, if FA in these sculpins is shown to have potential as a monitoring tool, specimens can be easily obtained for future monitoring efforts. Finally, physiography and resultant water quality has been implicated as a primary factor determining the distribution of several fish species in the Tennessee and Cumberland River systems (Starnes and Etnier 1986). Since coal mining is restricted to the Appalachian Plateau and water quality patterns of several constituents in the lower Powell River are a result of mining activity (Chapter One), it was reasoned that banded

sculpins might be especially sensitive to those water quality influences.

Banded sculpins captured during the 1988 fish sampling were examined as well as fish sampled from a site on the North Fork Powell River (Table 1; Figure 6). In addition, banded sculpins captured in summer 1989 at PRM 171.8, 168.9, and 120.4 were included to increase sample size and allow for correlations of fluctuating asymmetry with coal deposits in sidepool substrata. In total, 10 mainstem sites and one site on the North Fork Powell River (NFPRM 2.0) were selected.

Total length of each fish was measured to the nearest 1.0 mm. FA was evaluated by meristic counts on five anatomical characters: branchiostegal rays, pelvic fin rays, pectoral fin rays, second gill arch rakers, and third gill arch rakers. Gregg (1992) determined that pectoral fin rays, pelvic fin rays, and second and third gill arch rakers exhibited FA in the mottled sculpin (*C. bairdi*). Branchiostegal rays were added as a meristic character due to the ease of making the counts.

Another important factor considered in selecting anatomical characters for evaluation is the ease and timeliness with which a biologist could make the meristic counts on each character. If FA is shown to have promise as a biomonitoring variable, ease of obtaining counts would be important for widespread adoption and further study to the efficacy of FA in biomonitoring programs. For this reason, counts of difficult characters such as preopercular-mandibular pores were not considered.

Meristic counts of each of the five characters per fish were made using a low-power dissecting scope with forceps, scissors, and a sharp-pointed probe. Scissors were

used to cut the opercle away to allow viewing of the second and third gill arch. It was not necessary to cut away any of the trait structures from the body for making counts. For each character, counts on the left and right sides were recorded. The left count was subtracted from the right count. A zero score indicates no asymmetry, whereas any non-zero score means that the individual was asymmetric for that character. For example, a score of plus one means that the left side structure (e.g., gill rakers) had one more count than the right. A score of negative one indicates that the right side structure had one more structure than the left, etc. For each fish, five scores (one for each character) resulted. For example, hypothetical data on one fish could be: branchiostegal rays = 0, pelvic fin rays = 0, pectoral fin rays = -1, second gill arch = 0, third gill arch = +2. The interpretation is that the individual was asymmetric on two characters, pectoral fin rays and the third gill arch. The right side pectoral fin had one more ray than the left side and the left side third gill arch had two more gill rakers than the right side.

There are several ways to combine individual fish asymmetric information into an overall value for the population (Palmer and Strobeck 1986). For this study, population FA at a site was expressed as percent individuals with at least one asymmetric character:

$$\text{Overall population FA (\%)} = (AI / TI) \times 100$$

where,

AI = number of individuals with at least one character showing asymmetry; and

TI = total number of individuals examined at a site.

Multiple asymmetric characters per individual or magnitude of asymmetries (e.g., -1, +2) did not contribute beyond designation of the individual fish as containing an asymmetry.

Association of FA and water quality was tested by correlation of sample site FA % to PRM using the Spearman's Rank Correlation test (Hollander and Wolfe 1973). Although water quality samples were not taken at each banded sculpin sampling site, concentrations of several water quality variables exhibited distinct longitudinal gradients (Chapter One). FA level at the North Fork Powell River site is included to provide additional information for interpretation of any spatial FA patterns.

Association of FA % and sediment contaminants was evaluated by correlation of site FA % with percent coal in sidepool sediments using the Spearman Rank Correlation test. Percent coal in sediments at test sites was obtained from Wolcott (1990). Associations of FA and contaminants were used to test the hypothesis that water quality was a significant factor affecting fish assemblages in the lower Powell River in Virginia.

Results and Discussion

From inspection of the length-frequency distribution of banded sculpin captured in 1988, there appears to be essentially two year classes sampled (0+, length $\leq$ 50 mm and 1+, length $\geq$ 60 mm; no individuals were captured within the 51 - 59 mm range) (Figure

46). Percent of individuals ≤ 50 mm with asymmetries was 63.2 % (98 asymmetric individuals out of 155 individuals total). Percent of individuals ≥ 60 mm in length with asymmetries was 63.0% (46 asymmetric individuals out of 73 individuals total). This constancy of FA between two year classes in banded sculpin may indicate that mortality did not differ between asymmetric and symmetric individuals. Gregg (1992) found that FA was not correlated with any fitness character examined in mottled sculpin (*C. bairdi*). If character traits selected for FA examination are less critical to fitness, then differential mortality would not be expected to occur. Hence, provided environmental conditions remain relatively constant, FA may persist in similar proportions for each age class present in the population(s). Because of the essentially identical asymmetry levels of the two groups, it was deemed reasonable to add sculpins collected in 1989 at PRM 171.8 and 168.9, to increase sample size for the uppermost segment of the fifth order Powell River. In addition, 1989 sculpin collections at PRM 120.4 were included to permit a correlation analysis between percent asymmetric sculpins and percent coal in shallow-water side-pool substrata (from Wolcott 1990).

Asymmetry distributions for each of the five morphologic features varied around zero (Figures 47-51). The five features examined, therefore, exhibited fluctuating asymmetry.

The highest asymmetries occurred at the furthest upstream mainstem site (PRM 171.8 and 168.9 combined) and the lone site on the North Fork Powell River (NFPRM 1.4), a fifth order tributary that drains an extensively surface-mined watershed (Table 36,

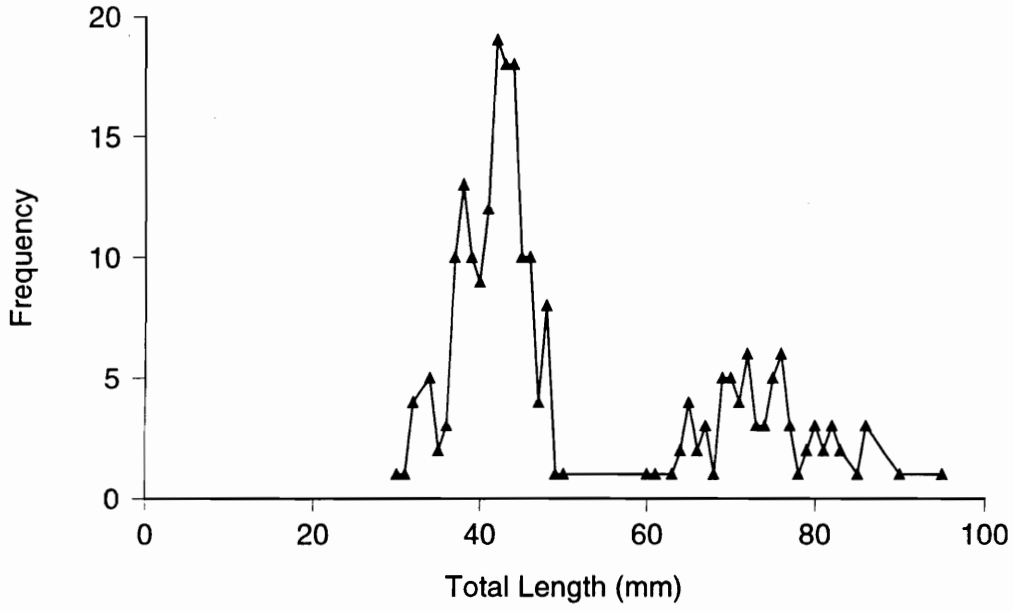

Figure 46. Total length-frequency distribution of 228 banded sculpin (*Cottus carolinae*) captured in August, 1988 at nine sites on the mainstem Powell River and one site on the North Fork Powell River (river mile 1.4) in Virginia.

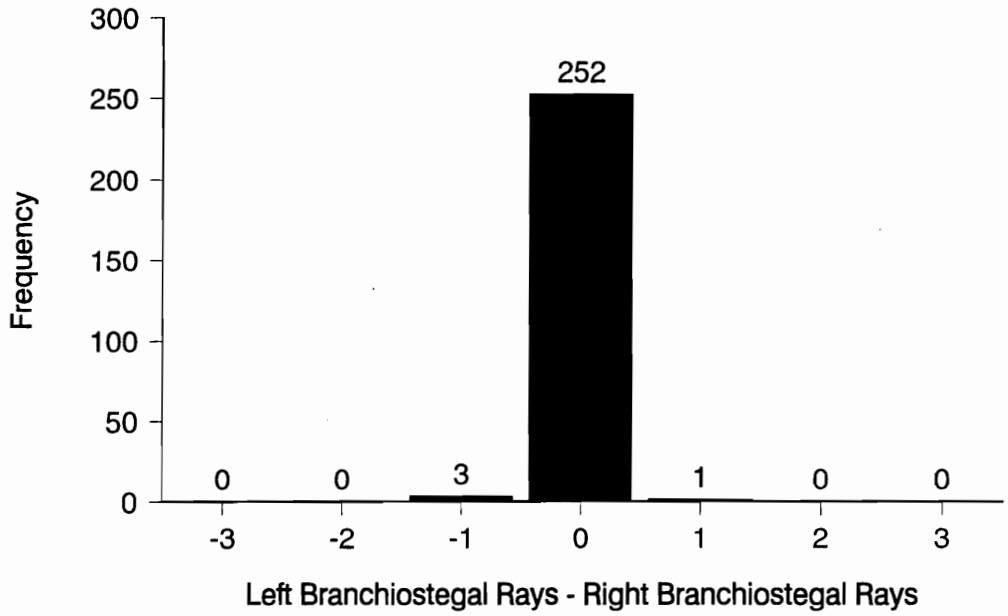

Figure 47. Distribution of asymmetries observed in banded sculpin branchiostegal rays. Banded sculpins were collected from 10 mainstem Powell River sites and one North Fork Powell River site in Virginia during summer 1988 and 1989.

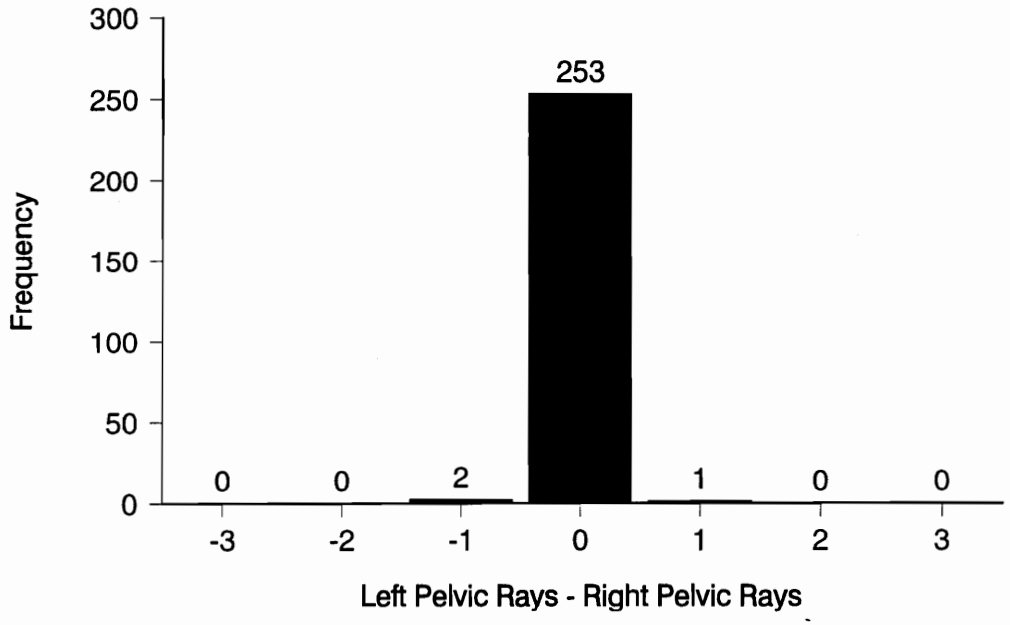

Figure 48. Distribution of asymmetries observed in banded sculpin pelvic rays. Banded sculpins were collected from 10 mainstem Powell River sites and one North Fork Powell River site in Virginia during summer 1988 and 1989.

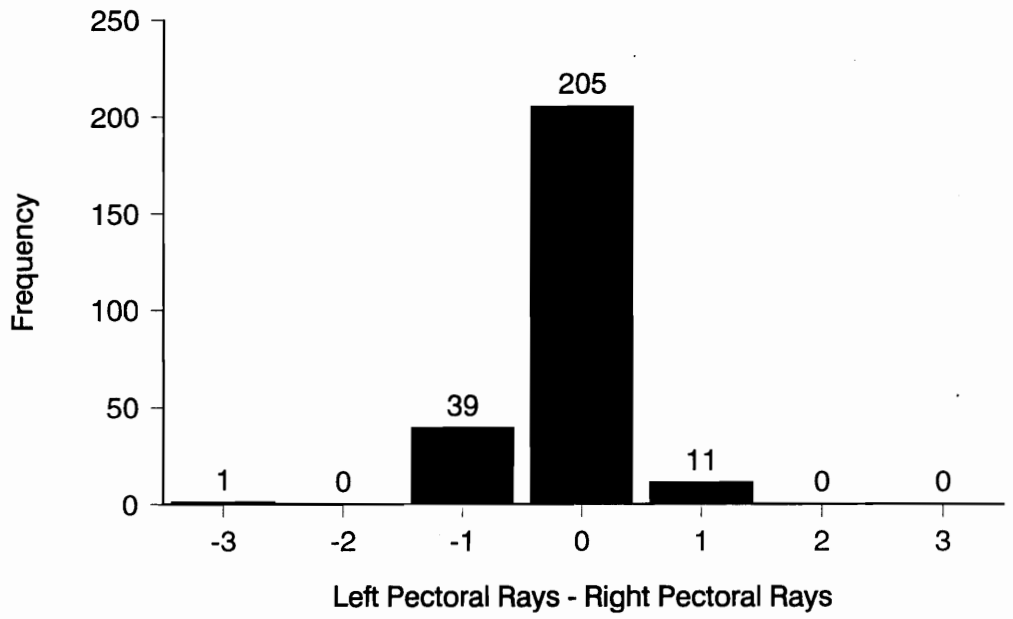

Figure 49. Distribution of asymmetries observed in banded sculpin pectoral rays. Banded sculpins were collected from 10 mainstem Powell River sites and one North Fork Powell River site in Virginia during summer 1988 and 1989.

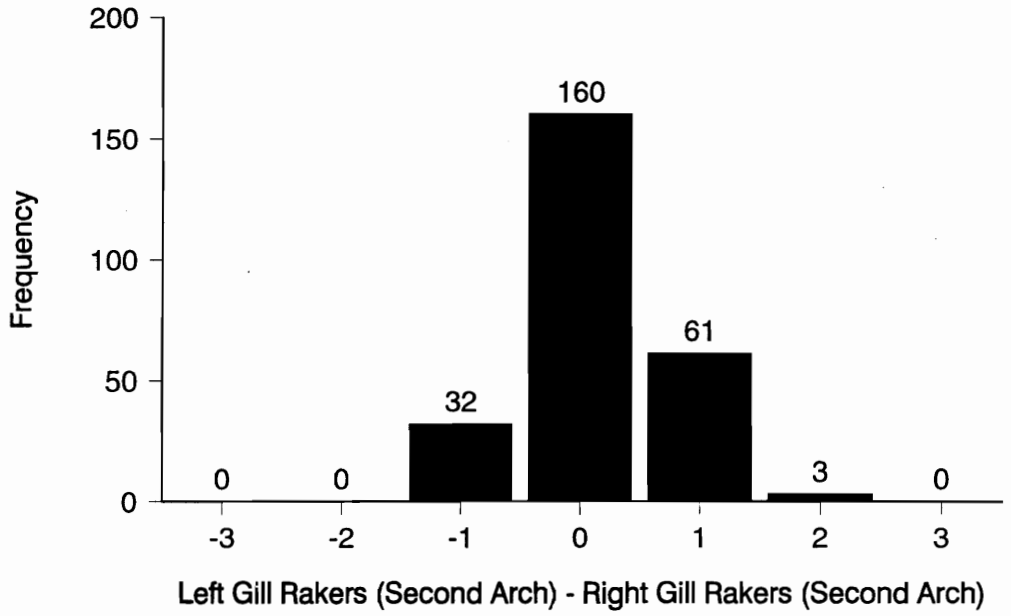

Figure 50. Distribution of ásymmetries observed in banded sculpin gill rakers (second gill arch). Banded sculpins were collected from 10 mainstem Powell River sites and one North Fork Powell River site in Virginia during summer 1988 and 1989.

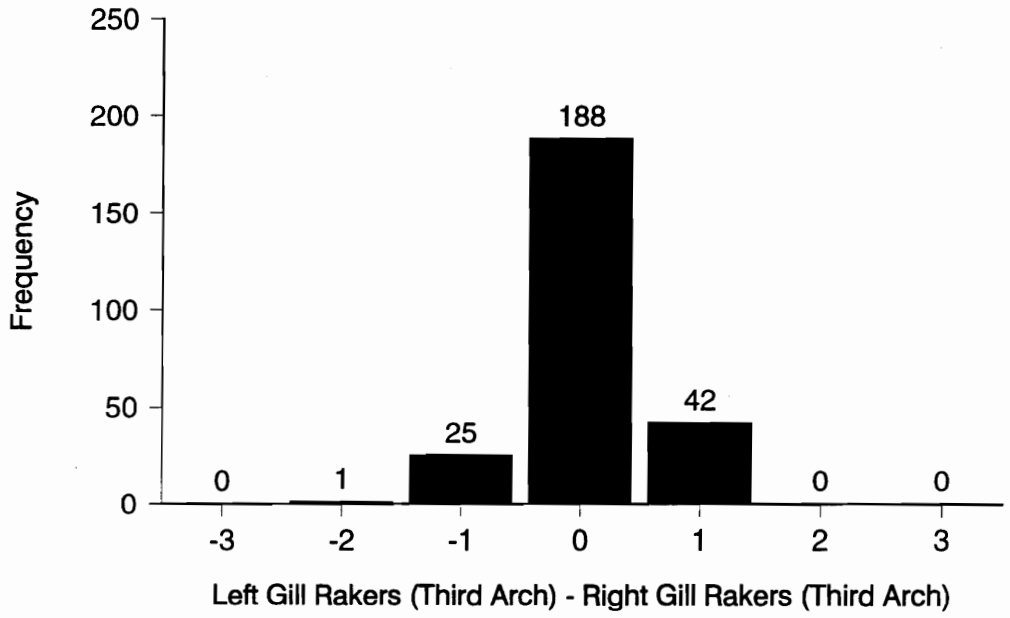

Figure 51. Distribution of asymmetries observed in banded sculpin gill rakers (third gill arch). Banded sculpins were collected from 10 mainstem Powell River sites and one North Fork Powell River site in Virginia during summer 1988 and 1989.

Figure 52). Mainstem asymmetry values decrease from PRM 163.4, until below the North Fork Powell River where asymmetry values increase. From the North Fork Powell River confluence to near the state line (PRM 117.3), asymmetry levels vary. There is no correlation of asymmetry levels with PRM ($\rho = 0.133$, $p < 0.732$). I conclude that water quality does not differ sufficiently in the study reach to affect banded sculpin developmental stability and, assuming that banded sculpin FA is a good indicator of potential assemblage-level changes from water quality, water quality does not appear to be a significant factor affecting fish assemblage structure in the lower Powell River.

Banded sculpin FA and levels of coal in side-pool substrata were analyzed for correlations to determine whether sediment contamination might be responsible for the observed banded sculpin asymmetry pattern. A significant positive correlation ($\rho = 0.749$, $p < 0.033$) showed that higher levels of sculpin asymmetry were associated with higher levels of coal deposits in side-pool substrata (Figure 53). I conclude that, although water quality is not a significant factor affecting fish assemblages in the study reach, contaminated sediment may be a concern in the Powell River. Studies of sediment in the adjacent Clinch River has demonstrated toxic effects from sediment (Yeager 1994). Kitchel et al. (1981) noted an inverse correlation between mussel abundance and quantity of coal waste at endangered mussel sites in the Powell River in Virginia. From laboratory studies, they concluded that coal wastes could cause chronic effects (lowered respiration rate, increased movements) in seven mussel species collected from the Powell River. Toxic sediments, therefore, could explain much of the decline in mussel fauna in the

Table 36. Fluctuating asymmetry site values for *Cottus carolinae* within the Powell River drainage in Virginia. Percent coal deposits in side-pool substrata from Wolcott (1990). NAI = number of individuals with at least one asymmetric character. NFPR = North Fork Powell River.

Site (PRM)	Number examined	NAI	% asymmetic individuals	Percent coal in substrata
171.8[a]	20	18	90.0	19.3
163.4	54	28	51.9	4.2
158.3	7	2	28.6	–[b]
153.4	36	24	66.7	4.7
146.8	9	4	44.4	2.6
144.6	40	23	57.5	1.8
123.0	11	7	63.6	47.5
120.4	8	5	62.5	3.3
119.3	18	14	77.8	–[b]
117.3	10	4	40.0	1.9
NFPR	43	38	88.4	–[b]
Total	256			

[a] Represents pooled information from sites PRM 171.8 and 168.9.
[b] Not sampled by Wolcott (1990).

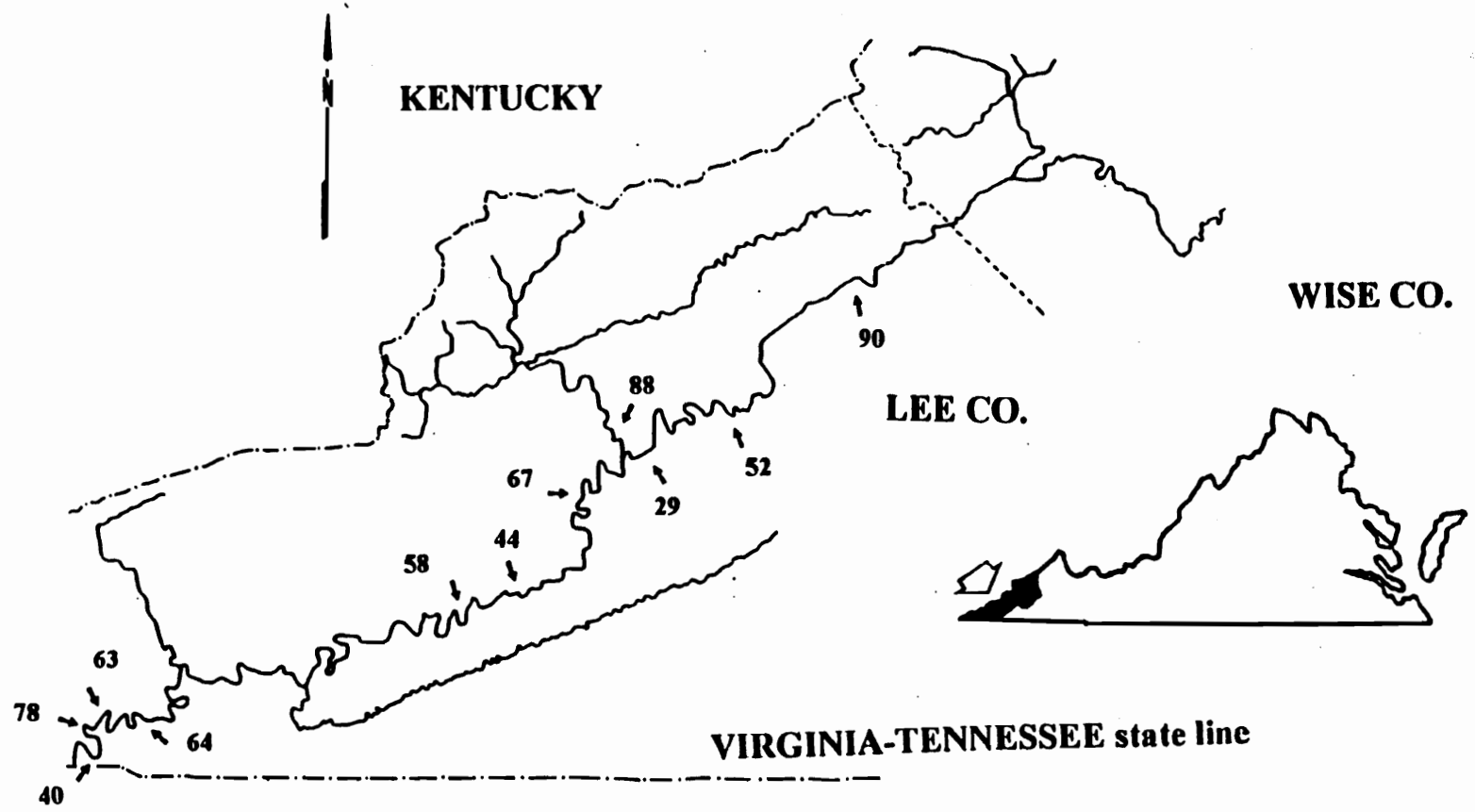

Figure 52. Distribution of percent asymmetric banded sculpins captured at 11 sites within the Powell River system in Virginia (summer 1988 and 1989). Percent asymmetric individuals at a site are the number of specimens asymmetric for at least one character divided by the total number of individuals examined. The percent asymmetric value for the most upstream site is from fish captured from PRM 171.8 and 168.9.

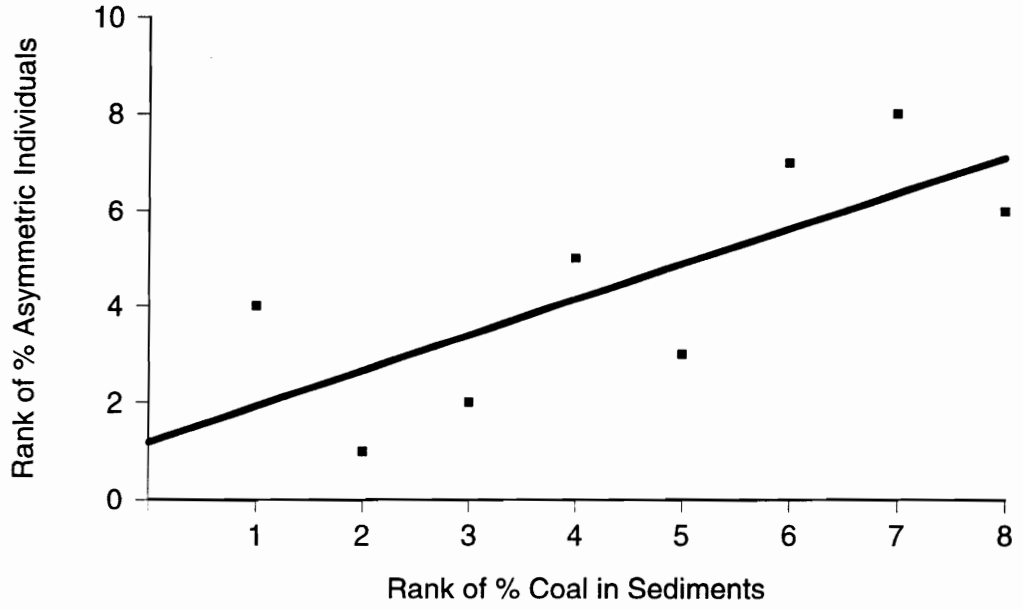

Figure 53. **Correlation between banded sculpin asymmetry levels and coal deposit levels in sidepools at eight sites on the Powell River in Virginia. Site asymmetry is percent individuals captured from a site that exhibited at least one asymmetric character. Both site asymmetry and coal deposits are presented as ranks (1 = lowest, 8 = highest).**

Powell River (Wolcott 1990; McCann 1993).

Appendix conclusions

Water quality does not seem to be a major factor influencing fish populations in the lower Powell River. Significant correlations between banded sculpin FA and coal deposits in side-pool substrata, however, indicate that toxic sediments might be a problem to Powell River fauna. In addition, high asymmetry levels in the most upstream site (PRM 171.8\168.9) and in the North Fork Powell River point to possible impairment from coal mining. The high coal deposit levels at PRM 123.0 may be a result of a "slug" of coal deposits slowly being washed downstream. An implication is that reduced input is occurring upstream, possibly due to compliance with the Surface Mining Control and Reclamation Act of 1977.

Fluctuating asymmetry in banded sculpins has potential as a monitoring indicator of biological integrity in streams. Further research should establish background asymmetry levels for banded sculpin by sampling populations from relatively unimpaired streams in the Powell River and upper Tennessee River watershed. Finding unimpaired streams should not be a formidable task since the banded sculpin occupies very small streams (Jenkins and Burkhead 1994). Background levels of FA in banded sculpins would allow the setting of benchmarks for asymmetry in that species. FA for banded sculpins, furthermore, could be a valuable metric in the IBI. This new metric could

replace the hybrid metric (number 11) or replace or combine with the anomaly metric (number 12) (Table 15).

Appendix E. Fish species distribution and function classifications

Table 37. Classifications for species captured during IBI sampling at eight sites in the Powell River in Virginia during August 1988. Tolerance and trophic group classifications are from Saylor and Ahlstedt (1990).

Species	Distributional Status[a]	Tolerance Group[b]	Reproductive Group[c]	Trophic Group[d]
Petromyzontidae				
Ichthyomyzon bdellium	N		SPD	PS
Lepisosteidae				
Lepisosteus osseus	N	TOL	MB	TC
Clupeidae				
Dorosoma cepidianum	N	TOL	MB	OM
Cyprinidae				
Cyprinus carpio	I	TOL	MB	OM
Campostoma anomalum	N		SPD[e]	HB
Nocomis micropogon	N	TOL	ASL	OM
Erimystax dissimilis	N	INT	LB	SP
E. insignis	N		LB	SP
Phenacobius uranops	N		LB	SP
Hybopsis amblops	N		LB	SP
Cyprinella galactura	N		ASL	SP
C. spiloptera	N	TOL	ASL	SP
Luxilis coccogenis	N		LB[e]	SP
L. chrysocephalus	N	TOL	SPD[e]	OM
Lythrurus lirus	N[f]		LB	SP[f]
Notropis rubellus	N		LB	SP

Table 37. (continued)

Species	Distributional Status	Tolerance Group	Reproductive Group	Trophic Group
N. leuciodus	N		LB	SP
N. photogenis	N		LB	SP
N. ariommus	N	INT	LB	SP
N. telescopus	N	INT	LB	SP
N. volucellus	N		LB	SP
N. sp. (sawfin shiner)	N		LB	SP
Pimephales notatus	N		PC	OM
Catostomidae				
Carpiodes cyprinus	N		MB	OM
Hypentelium nigricans	N	INT	LB	IN
Moxostoma duquesnei	N		LB	IN
M. macrolepidotum	N		LB	IN
M. erythrurum	N		LB	IN
M. carinatum	N		SPD	IN
M. anisurum	N		LB	IN
Ictaluridae				
Ictalurus punctulatus	N		PC	OM
Amierus nebulosus	N		PC	OM
Noturus eleutherus	N	INT	PC	IN
Pylodictus olivaris	N		PC	TC

Table 37. (continued)

Species	Distributional Status	Tolerance Group	Reproductive Group	Trophic Group
Atherinidae				
Labidesthes sicculus	N		MB	IN
Cottidae				
Cottus carolinae	N		PC	IN
Centrarchidae				
Ambloplites rupestris	N		PC	TC
Pomoxis annularis	N		PC	TC
Micropterus dolomieu	N		PC	TC
M. punctatus	N		PC	TC
M. salmoides	N		PC	TC
Lepomis auritus	I		PC	IN
L. megalotis	N		PC	IN
L. macrochirus	N		PC	IN
Percidae				
Stizostedion vitreum	N		LB	TC
S. canadense	N		LB	TC
Percina caprodes	N		SPD	SP
P. evides	N	INT	SPD	SP
P. aurantiaca	N		SPD	SP
Etheostoma blennioides	N		ASL	SP
E. zonale	N		ASL	SP

Table 37. (continued)

Species	Distributional Status	Tolerance Group	Reproductive Group	Trophic Group
E. simoterum	N		ASL	SP
E. stigmaeum	N		SPD	SP
E. camurum	N	INT	SPD	SP
E. rufilineatum	N		SPD	SP
Scianidae				
Aplinodinotus grunniens	N		PSFE	IN

[a] N = native, I = introduced

[b] TOL = tolerant, INT = intolerant

[c] ASL = spawn above substrate grade, LB = broadcast spawn over rocks, MB = broadcast spawn over miscellaneous substrata, PC = parental care, PSFE = pelagic spawn with floating eggs, SPD = preparation or disturbance of spawning site prior to egg deposition

[d] HB = herbivore, IN = generalized insectivore, PS = parasite, SP = specialized insectivore, TC = piscivore

[e] Nest associate of *Nocomis* sp.

[f] Not in Saylor and Ahlstedt (1990), determination made by author

Appendix F. Relative abundances of fish species in all habitats at eight sites on the Powell River (1988).

Table 38. Relative abundances (%) of species in all mesohabitats at eight sites on the Powell River in Virginia sampled during August 1988. Species occurrence at a ninth site (158.3) is listed for distributional information: + = captured, - = not captured.

	Powell River Mile								
Species[a]	117.3	119.3	123.0	144.6	146.8	153.4	158.3	163.4	172.2
Petromyzontidae									
Ichthyomyzon bdellium	0.20	0.12	0.32	0	0.15	0.11	-	0	0
Lepisosteidae									
Lepisosteus osseus	0	0	0	0	0.15	0	-	0	0
Clupeidae									
Dorosoma cepidianum	2.25	2.25	8.04	1.72	3.02	2.17	+	1.43	3.98
Cyprinidae									
Cyprinus carpio	0	0	0	0.49	0	0	+	0.09	0.53
Campostoma anomalum	5.73	7.83	2.89	3.94	10.11	8.46	+	6.88	6.63
Nocomis micropogon	0.61	0.36	0	0.12	0.75	0.76	+	5.54	2.39
Erimystax dissimilis	1.84	1.30	1.29	4.55	1.51	1.08	+	0.18	0.27
E. insignis	0.41	4.86	0.32	2.58	0	0.22	-	0	0
Phenacobius uranops	0.20	0.95	0.96	0.62	0	0.65	+	0.63	0.53
Hybopsis amblops	1.23	1.78	0.96	1.35	1.51	2.71	+	1.07	1.86
Cyprinella galactura	0.82	1.78	0.96	1.97	0.75	0.54	+	0	0
C. spiloptera	0.20	2.49	1.61	3.32	0.75	1.19	+	0.54	0.27
Luxilis coccogenis	0.20	0.71	0.96	1.23	0.75	0.98	+	3.40	1.59
L. chrysocephalus	4.09	9.61	1.29	8.24	12.82	9.65	+	8.40	13.79
Lythrurus lirus	0.20	0	0	0	0	0	-	0.36	0

Table 38. (continued)

Species	Powell River Mile								
	117.3	119.3	123.0	144.6	146.8	153.4	158.3	163.4	172.2
Notropis rubellus	4.09	4.03	0.32	6.40	4.52	8.02	+	5.18	3.44
N. leuciodus	6.95	2.37	1.61	2.95	2.71	8.03	+	17.33	2.65
N. photogenis	0	0.12	0	0	0.15	0.43	+	0.63	0
N. ariommus	1.23	0.47	0	0	0.15	0.33	+	0	0
N. telescopus	6.54	1.19	1.29	2.83	3.77	7.70	+	6.43	0.80
N. volucellus	2.66	3.91	1.61	4.67	8.44	6.18	+	6.43	7.69
N. sp. (sawfin shiner)	0.82	4.27	0	4.06	1.96	0.54	+	1.07	0
Pimephales notatus	1.02	0.71	0	0.98	3.47	2.17	+	2.59	7.96
Catostomidae									
Carpiodes cyprinus	0	0	0	0.12	0	0	-	0	0
Hypentelium nigricans	0.41	0.47	0.64	0.25	0.90	0.54	+	1.07	0.80
Moxostoma duquesnei	1.02	1.19	6.11	1.23	3.62	0.65	+	1.88	3.18
M. macrolepidotum	0.61	0.24	4.18	0.25	2.11	0.11	+	0.09	0.53
M. erythrurum	1.02	1.66	0.96	1.11	3.02	2.17	+	1.07	8.49
M. carinatum	0[b]	0.36	0.96	0.49	0.90	0	+	0.27	0.27
M. anisurum	0	0	0	0	0	0	+	0	0
Ictaluridae									
Ictalurus punctatus	0	0.12	0	0.49	0	0	-	0	0.53
Ameiurus natalis	0.20	0.12	0	0	0	0	-	0.09	0.27
Noturus eleutherus	0.20	0	0	0	0	0	-	0	0
Pylodictus olivaris	0.41	0	0	0	0.15	0	-	0	0

Table 38. (continued)

Species	Powell River Mile								
	117.3	119.3	123.0	144.6	146.8	153.4	158.3	163.4	172.2
Atherinidae									
Labidesthes sicculus	0	0.71	0	0	0	0.11	-	0	0
Cottidae									
Cottus carolinae	3.89	2.49	3.53	5.29	1.36	4.34	+	5.27	2.65
Centrarchidae									
Ambloplites rupestris	9.61	2.97	7.40	0.74	3.92	1.73	+	1.97	0.27
Pomoxis annularis	0	0	0	0.12	0	0	-	0	0.27
Micropterus dolomieu	1.02	0.59	1.93	0.98	1.81	0.54	+	1.43	0.80
M. punctulatus	0	0.12	0.32	0.12	0.30	0.76	+	0.98	2.12
M. salmoides	0	0	0	0	0.15	0	-	0	0
Lepomis auritus	0	0.12	0.32	1.11	5.12	1.63	+	1.61	10.88
L. megalotis	5.32	4.27	4.50	1.97	4.52	2.82	+	1.79	0.80
L. macrochirus	0	0	0.32	0	0	0.11	-	0.18	1.06
Percidae									
Stizostedion vitreum	0	0	0	0	0	0.11	-	0	0
S. canadense	0.20	0	0	0	0	0	-	0	0
Percina caprodes	0	0	0	0.25	0.15	0.43	+	0.09	0.53
P. evides	4.70	3.56	9.97	3.69	1.06	1.19	+	0	0.80
P. aurantiaca	0.82	0.47	0.32	0	0.15	0.11	+	0	0
Etheostoma blennioides	1.43	1.30	3.22	0.98	3.17	5.75	+	4.46	6.90
E. zonale	0.82	1.07	0	0	0.90	0.43	+	0.27	1.59

Table 38. (continued)

Species	Powell River Mile								
	117.3	119.3	123.0	144.6	146.8	153.4	158.3	163.4	172.2
E. simoterum	0.61	0.12	0.96	0.12	0.45	0.43	+	0	0
E. stigmaeum	0.41	0	1.61	0.25	0	0	+	0	0.27
E. camurum	2.04	0.24	0	0.62	0	0	-	0	0
E. rufilineatum	23.52	26.22	28.30	27.43	7.84	13.88	+	9.03	1.86
Scianidae									
Aplinodinotus grunniens	0	0.12	0	0.12	0.45	0.11	+	0	0.53

[a] Three additional species were observed or captured prior to sampling for the Index of Biotic Integrity. Several *Polyodon spathula* were observed at PRM 117.4 in June, 1987, three *Ammocrypta clara* were captured by seining over sand substratum at PRM 117.3 in July, 1988, and one *Morone chrysops* was captured by boat electrofishing at PRM 163.4 in July, 1988.

[b] One specimen was captured during qualitative sampling.

Appendix G. Relative abundances of fish species in shallow-water habitats at eight sites on the Powell River (1988).

Table 39. Relative abundances (%) of species within shallow-water habitats (riffle, run, sidepool, and stream margin) at each of eight sites on the Powell River in Virginia sampled during August 1988. Fish were captured for biotic integrity assessment of each site using the IBI.

	Powell River Mile							
Species	117.3	119.3	123.0	144.6	146.8	153.4	163.4	172.2
Petromyzontidae								
Ichthyomyzon bdellium	0.26	0.15	0.49	0	0.24	0.13	0	0
Cyprinidae								
Campostoma anomalum	6.65	10.06	4.43	4.45	15.84	9.80	8.40	10.29
Nocomis micropogon	0.51	0.30	0	0.14	0.95	0.89	7.04	3.70
Erimystax dissimilis	1.53	1.68	1.97	5.01	2.13	0.64	0.23	0.41
E. insignis	0.51	6.25	0.49	2.92	0	0.25	0	0
Phenacobius uranops	0.26	1.22	1.48	0.70	0	0.76	0.68	0.82
Hybopsis amblops	0.51	0.30	0.49	1.39	0.24	2.42	0.68	2.06
Cyprinella galactura	1.02	2.13	1.48	2.09	0.95	0.64	0	0
C. spiloptera	0.26	3.05	2.46	3.20	0.71	1.02	0.68	0.41
Luxilis coccogenis	0.26	0.91	1.48	1.39	1.18	1.15	3.41	2.47
L. chrysocephalus	4.35	10.37	0.99	9.18	18.68	11.20	10.22	21.40
Lythrurus lirus	0	0	0	0	0	0	0.45	0
Notropis rubellus	5.12	5.18	0.49	7.23	7.09	9.41	6.58	5.35
N. leuciodus	8.70	3.05	2.46	3.34	4.26	9.41	21.68	4.12
N. photogenis	0	0.15	0	0	0.24	0.51	0.79	0
N. ariommus	0	0	0	0	0.24	0.38	0	0
N. telescopus	8.18	1.52	1.97	3.20	5.91	9.03	8.17	1.23
N. volucellus	3.32	1.07	2.46	5.29	11.11	6.74	3.63	8.23
N. sp. (sawfin shiner)	1.02	5.34	0	4.59	3.07	0.64	1.36	0

Table 39. (continued)

Species	Powell River Mile 117.3	119.3	123.0	144.6	146.8	153.4	163.4	172.2
Pimephales notatus	1.28	0.30	0	0.70	0.71	2.54	0.57	10.29
Catostomidae								
Hypentelium nigricans	0	0.30	0	0.28	0.24	0.13	0.57	0.41
M. macrolepidotum	0	0.15	0	0	0	0	0	0
M. carinatum	0	0.30	0	0	0	0	0	0
Ictaluridae								
Ictalurus punctulatus	0	0	0	0.14	0	0	0	0
Ameirus natalis	0	0	0	0	0	0	0	0
Noturus eleutherus	0.001	0	0	0	0	0	0	0
Atherinidae								
Labidesthes sicculus	0	0.76	0	0	0	0.13	0	0
Cottidae								
Cottus carolinae	4.86	3.20	5.42	5.98	2.13	5.09	6.24	4.12
Centrarchidae								
Ambloplites rupestris	5.88	0	4.43	0	0.95	0.38	0.23	0.41
Micropterus dolomieu	0.26	0.15	0.49	0.70	0.95	0.25	0.79	0.41
M. punctulatus	0	0	0	0	0	0.13	0	0.41
Lepomis auritus	0	0	0	0.14	0.24	0	0	4.53
L. megalotis	2.56	0.15	0	0	0.47	0.38	0	0

Table 39. (continued)

Species	Powell River Mile							
	117.3	119.3	123.0	144.6	146.8	153.4	163.4	172.2
Percidae								
Percina caprodes	0	0	0	0.28	0.24	0.38	0.11	0.82
P. evides	5.88	4.57	15.27	4.17	1.65	1.40	0	1.23
P. aurantiaca	0.26	0	0.49	0	0.24	0.13	0	0
Etheostoma blennioides	1.79	1.68	4.93	1.11	4.96	6.74	5.68	10.70
E. zonale	1.02	1.37	0	0	1.42	0.51	0.34	2.47
E. simoterum	0.77	0.15	0.99	0.14	0.47	0.38	0	0
E. stigmaeum	0.51	0	2.46	0.28	0	0	0	0.41
E. camurum	2.56	0.30	0	0.70	0	0	0	0
E. rufilineatum	29.41	33.69	42.36	31.01	12.29	16.29	11.46	2.88

Appendix H. Relative abundances of fish species in deep-water habitats at nine sites on the Powell River (1988).

Table 40. Relative abundances (%) of species within deep-water habitats (pool) at each of nine sites on the Powell River in Virginia sampled during August 1988. Fish were captured for biotic integrity assessment of each site using the IBI.

	Powell River Mile								
Species	117.3	119.3	123.0	144.6	146.8	153.4	158.3	163.4	172.2
Lepisosteidae									
Lepisosteus osseus	0	0	0	0	0.42	0	0	0	0
Clupeidae									
Dorosoma cepidianum	11.22	10.16	23.15	14.89	8.33	14.71	4.76	6.72	11.19
Cyprinidae									
Cyprinus carpio	0	0	0	4.25	0	0	2.60	0.42	1.49
Campostoma anomalum	2.04	0	0	0	0	0.74	0.43	1.26	0
Nocomis micropogon	1.02	0.53	0	0	0.42	0	1.73	0	0
Erimystax dissimilis	3.06	0	0	1.06	0.42	3.67	0	0	0
Phenacobius uranops	0	0	0	0	0	0	0.42	0.42	0
Hybopsis amblops	4.08	6.95	1.85	1.06	3.75	4.41	4.33	2.52	1.49
Cyprinella galactura	0	0.53	0	1.06	0.42	0	0	0	0
C. spiloptera	0	0.53	0	4.26	0.83	2.21	2.60	0	0
Luxilis coccogenis	0	0	0	0	0	0	0	3.36	0
L. chrysocephalus	3.06	6.95	1.85	1.06	2.50	0.74	4.76	1.68	0
Lythrurus lirus	1.02	0	0	0	0	0	0	0	0
N. leuciodus	0	0	0	0	0	0	0	1.26	0
N. ariommus	6.12	2.14	0	0	0	0	2.16	0	0
N. volucellus	0	13.90	0	0	3.75	2.94	7.79	16.81	6.72

Table 40. (continued)

Species	Powell River Mile								
	117.3	119.3	123.0	144.6	146.8	153.4	158.3	163.4	172.2
N. sp. (sawfin shiner)	0	0.53	0	0	0	0	0	0	0
Pimephales notatus	0	2.14	0	3.19	8.33	0	0.87	10.08	3.73
Catostomidae									
Carpiodès cyprinus	0	0	0	1.06	0	0	0	0	0
Hypentelium nigricans	2.04	1.07	1.85	0	2.08	2.94	0.87	2.94	1.49
Moxostoma duquesnei	5.10	5.35	17.59	10.63	10.00	4.41	5.51	8.82	8.96
M. macrolepidotum	3.06	0.53	12.04	2.13	5.83	0.74	0.43	0.42	1.49
M. erythrurum	5.10	7.49	2.78	9.57	8.33	14.71	9.52	5.04	23.88
M. carinatum	0	0.53	2.78	4.26	2.50	0	0	1.26	0.75
Ictaluridae									
Ictalurus punctulatus	0	0.53	0	3.19	0	0	0	0	1.49
Ameirus natalis	1.02	0	0	0	0	0	0	0	0
Pylodictus olivaris	2.04	0	0	0	0.42	0	0	0	0
Atherinidae									
Labidesthes sicculus	0	0.53	0	0	0	0	0	0	0
Cottidae									
Cottus carolinae	0	0	0	0	0	0	0	1.68	0

Table 40. (continued)

Species	Powell River Mile								
	117.3	119.3	123.0	144.6	146.8	153.4	158.3	163.4	172.2
Centrarchidae									
Ambloplites rupestris	24.49	13.37	12.96	6.38	9.17	9.56	9.09	8.40	0
Pomoxis annularis	0	0	0	1.06	0	0	0	0	0
Micropterus dolomieu	4.08	2.14	4.63	3.19	3.33	2.21	5.63	3.78	1.49
M. punctatus	0	0.53	0.93	1.06	0.83	4.41	0.43	4.62	5.22
M. salmoides	0	0	0	0	0.42	0	0	0	0
Lepomis auritus	0	0.53	0.93	8.51	13.75	11.03	15.15	7.56	22.39
L. megalotis	16.33	18.72	12.96	17.02	11.67	16.91	17.75	8.40	2.24
L. macrochirus	0	0	0.93	0	0	0.74	0	0.84	2.99
Percidae									
Stizostedion vitreum	0	0	0	0	0	0.74	0	0	0
S. canadense	1.02	0	0	0	0	0	0	0	0
Percina caprodes	0	0	0	0	0	0.74	0.43	0	0
P. aurantiaca	3.06	2.14	0	0	0	0	0.87	0	0
E. simoterum	0	0	0.93	0	0.42	0.74	0	0	0
E. rufilineatum	0	0	1.85	0	0	0	0	0	0
Scianidae									
Aplinodinotus grunniens	0	0.53	0	1.06	1.25	0.74	0.87	0	1.49

Appendix I. Abundances of Fish Species at 12 Sites on the Powell River (1989).

Table 41. Fish species abundances at 12 sites on the Powell River in Virginia, sampled from June through August 1989. Abundances are expressed as number caught per unit effort of approximately $25m^2$.

	Powell River Mile							
Species	117.3	123.0	126.2	127.2	136.7	144.6	149.3	153.9
Petromyzontidae								
Ichthyomyzon bdellium	0.03	0.04	0	0	0	0	0.04	0.03
Lepisosteidae								
Lepisosteus osseus	0	0	0	0	0	0.04	0	0
Cyprinidae								
Rhinichthys atratulus	0.10	0.15	0.03	0.33	0	0	0	0
Campostoma anomalum	1.97	0.81	0.69	1.20	0.33	1.46	0.92	0.45
Nocomis micropogon	0.27	0.08	0.17	0.03	0.44	0.21	0.04	0.06
Erimystax dissimilis	0.50	0.38	0.55	0.13	0.37	1.39	0.23	0.42
E. insignis	0.13	0.04	0.03	0.03	0.11	0	0	0
Phenacobius uranops	0.17	0.08	0.07	0.10	0.22	0.36	0	0.06
Hybopsis amblops	0	0.08	1.31	0.07	0.11	0.07	0.08	0.97
Cyprinella galactura	0.07	0.50	0.48	0.17	0.07	0.04	0	0
C. spiloptera	2.00	1.08	0.86	1.40	0.93	1.89	2.85	1.77
Luxilis coccogenis	0.27	0.38	0.83	0.07	0.59	0.18	0.04	0.13
L. chrysocephalus	0.20	0.19	0.21	0.43	0.11	0.21	0.38	0
Lythrurus lirus	0	0.15	0.03	0.03	0	0	0.08	0.03
Notropis rubellus	3.30	1.38	0.97	4.03	0.67	5.18	0.81	6.68
N. leuciodus	0.97	0.42	0.38	0.27	0.44	0.64	0.35	0.61
N. photogenis	0	0	0	0	0.07	0	0	0
N. ariommus	0	0	0	0	0.24	0.38	0	0
N. telescopus	0.43	0.92	0.59	0.20	0.26	0.14	0.08	0.13

Table 41. (Continued)

Species	Powell River Mile 156.0	168.9	170.3	171.8
Petromyzontidae				
Ichthyomyzon bdellium	0	0	0	0
Lepisosteidae				
Lepisosteus osseus	0	0	0	0
Cyprinidae				
Rhinichthys atratulus	0.04	0	0.03	0
Campostoma anomalum	3.56	3.56	1.34	3.72
Nocomis micropogon	0.20	1.44	0.93	0.80
Erimystax dissimilis	0.44	0.00	0.03	0.08
E. insignis	0	0	0	0
Phenacobius uranops	0.16	0.08	0	0
Hybopsis amblops	1.60	0.12	2.55	0
Cyprinella galactura	0	0	0	0
C. spiloptera	2.36	0.52	0.66	1.48
Luxilis coccogenis	0	0.24	0.38	0.16
L. chrysocephalus	0.12	0.56	0.55	0.28
Lythrurus lirus	0	0	0.03	0
Notropis rubellus	1.52	4.88	2.21	0.64
N. leuciodus	0.24	0.92	0.66	0.44
N. photogenis	0	0	0	0
N. ariommus	0	0	0	0
N. telescopus	0	0.48	0.10	0

Table 41. (continued)

Species	Powell River Mile							
	117.3	123.0	126.2	127.2	136.7	144.6	149.3	153.9
N. volucellus	1.47	2.96	1.38	0.37	1.04	1.00	1.62	0.71
N. sp. (sawfin shiner)	0.43	0.27	0.24	0	0.07	0.32	0.12	0.03
Pimephales notatus	0.03	0.15	0.24	0.10	0.15	0.18	0.12	0.03
Catostomidae								
Hypentelium nigricans	0.13	0.15	0.24	0.13	0.22	0.32	0.04	0.23
Moxostoma duquesnei	0	0	0	0	0	0	0	0
M. macrolepidotum	0	0	0	0	0.04	0	0	0
Ictaluridae								
Ictalurus punctulatus	0	0	0	0	0.04	0	0	0
Noturus eleutherus	0.17	0.04	0.07	0.03	0	0	0	0
Cottidae								
Cottus carolinae	2.00	1.08	0.86	1.40	0.93	1.89	2.85	1.77
Centrarchidae								
Ambloplites rupestris	0.03	0.12	0.17	0.27	0.41	0	0.23	0.06
Micropterus dolomieu	0	0.04	0.03	0	0.19	0.11	0.04	0.06
Lepomis auritus	0	0	0	0	0	0	0.19	0
L. megalotis	0.03	0.23	0.03	0.03	0.04	0	0.04	0
Percidae								
Percina caprodes	0	0	0	0	0	0	0.04	0.10

Table 41. (continued)

Species	Powell River Mile 156.0	168.9	170.3	171.8
N. volucellus	0.48	0.44	3.03	0
N. sp. (sawfin shiner)	0.12	0	0.03	0
Pimephales notatus	1.16	0	1.24	0.04
Catostomidae				
Hypentelium nigricans	0.28	0.32	0.03	0.52
Moxostoma duquesnei	0.08	0.04	0.10	0
M. macrolepidotum	0.04	0.04	0	0.08
Ictaluridae				
Ictalurus punctulatus	0	0	0	0
Noturus eleutherus	0	0	0	0
Cottidae				
Cottus carolinae	2.36	0.52	0.66	1.48
Centrarchidae				
Ambloplites rupestris	0.16	0.28	0.10	0.12
Micropterus dolomieu	0.04	0.04	0.03	0.04
Lepomis auritus	0	0.04	0.17	0.04
L. megalotis	0	0	0	0
Percidae				
Percina caprodes	0.08	0	0	0

Table 41. (continued)

Species	Powell River Mile							
	117.3	123.0	126.2	127.2	136.7	144.6	149.3	153.9
P. evides	1.73	2.54	0.93	0.50	0.78	0.57	0.42	0.29
P. aurantiaca	0	0	0.10	0.10	0.26	0	0.04	0.10
Etheostoma blennioides	0.93	0.92	0.69	1.40	1.11	1.50	1.42	1.58
E. zonale	0.23	0	0.21	0.13	0.04	0	0.15	0.10
E. simoterum	0	0.04	0.07	0	0	0	0.35	0.03
E. stigmaeum	0	0	0.14	0	0	0.04	0.04	0.03
E. camurum	0	0.35	0.07	0.23	0.04	0.18	0	0
E. rufilineatum	8.63	7.58	5.10	10.23	6.04	5.75	8.92	5.74

Table 41. (continued)

Species	Powell River Mile			
	156.0	168.9	170.3	171.8
P. evides	0.24	0	0	0
P. aurantiaca	0.04	0.04	0	0
Etheostoma blennioides	1.40	1.84	0.86	1.28
E. zonale	0	0.60	0.41	0.04
E. simoterum	0	0.24	0.41	0.12
E. stigmaeum	0	0	0	0
E. camurum	0	0	0	0
E. rufilineatum	5.64	4.56	6.17	7.00

Appendix J. Densities of fish species at 12 sites on the Powell River (1990).

Table 42. Densities (#/100 m^2) of fish species at 12 sites on the Powell River in Virginia, sampled from June through August 1990.

Species	Powell River Mile							
	117.3	117.4	119.3	120.4	123.0	127.2	136.7	141.3
Petromyzontidae								
Ichthyomyzon bdellium	0	0.78	0.52	1.56	1.56	0	0	0
Cyprinidae								
Rhinichthys atratulus	1.04	0	28.13	0	1.04	5.80	1.04	0.52
Campostoma anomalum	119.27	73.44	219.79	14.58	128.13	136.61	52.60	8.85
Semotilus atromaculatus	0	0	0	0	0	0	0	0
Nocomis micropogon	12.50	12.50	5.73	4.17	5.21	0.45	13.02	9.90
Erimystax dissimilis	11.46	17.97	4.69	12.50	4.17	6.70	5.21	13.02
E. insignis	5.73	5.47	3.65	18.75	56.25	2.68	6.77	6.25
Phenacobius uranops	7.29	4.69	0.52	5.21	11.46	4.46	4.17	2.60
Hybopsis amblops	3.65	0.78	41.15	0.52	1.56	0.89	2.60	16.67
Cyprinella galactura	4.69	9.38	2.60	2.08	3.13	2.23	2.08	9.38
C. spiloptera	5.73	3.91	3.13	4.17	6.25	2.68	7.81	0.52
Luxilis coccogenis	10.42	16.41	7.29	2.60	12.50	2.23	10.42	5.73
L. chrysocephalus	11.46	3.13	7.29	3.13	14.06	4.46	14.06	8.33
Lythrurus lirus	0.52	0	0	2.08	0	0.89	0.52	0
Notropis rubellus	53.65	31.25	6.25	19.27	23.44	75.45	55.73	13.54
N. leuciodus	11.98	7.81	2.60	8.33	13.54	3.57	5.21	6.25
N. photogenis	0	0	0.52	0	0	0	3.65	1.56
N. ariommus	8.33	0	26.56	0	1.04	0	2.08	0
N. telescopus	11.46	4.69	3.13	3.13	28.13	0.45	9.90	1.04

Table 42. (Continued)

Species	Powell River Mile 144.6	149.3	153.9	156.0
Petromyzontidae				
Ichthyomyzon bdellium	1.63	0.52	0	0
Cyprinidae				
Rhinichthys atratulus	0	1.04	0	0.52
Campostoma anomalum	8.15	142.71	61.98	30.73
Semotilus atromaculatus	0	0	0	0.52
Nocomis micropogon	2.72	25.00	7.81	6.77
Erimystax dissimilis	3.26	2.60	8.85	1.56
E. insignis	4.35	4.17	0	0
Phenacobius uranops	3.80	9.38	25.00	2.08
Hybopsis amblops	25.00	5.21	6.25	7.29
Cyprinella galactura	2.72	1.04	0.52	1.56
C. spiloptera	1.63	0.52	2.60	1.56
Luxilis coccogenis	3.80	13.02	8.33	6.77
L. chrysocephalus	2.17	26.04	0.52	5.73
Lythrurus lirus	0.54	0	0.52	4.17
Notropis rubellus	130.44	19.79	114.06	228.13
N. leuciodus	5.98	18.23	41.15	18.75
N. photogenis	0	2.08	0	0.52
N. ariommus	0	0.52	0	1.56
N. telescopus	1.09	5.21	17.19	6.25

Table 42. (continued)

Species	Powell River Mile							
	117.3	117.4	119.3	120.4	123.0	127.2	136.7	141.3
N. volucellus	31.25	11.72	68.75	33.33	21.88	13.84	32.81	2.60
N. sp. (sawfin shiner)	10.42	2.34	4.17	5.21	4.69	1.34	1.56	2.60
Pimephales notatus	0	0	6.77	0.52	0.52	0	10.94	0
Catostomidae								
Hypentelium nigricans	18.23	5.47	20.83	4.17	3.13	10.27	13.02	3.65
Moxostoma duquesnei	0	0	1.56	1.04	0	0	0	1.04
M. macrolepidotum	1.56	0	1.56	0	0	0.89	1.56	1.56
M. erythrurum	0	0	0	0.52	0	0	0	0
Ictaluridae								
Noturus eleutherus	1.56	0.78	0	1.04	0	0	0	0
Cottidae								
Cottus carolinae	17.71	27.34	60.42	16.67	27.08	35.27	24.48	10.94
Centrarchidae								
Ambloplites rupestris	2.08	0.78	1.04	2.60	1.04	3.57	4.17	1.56
Micropterus dolomieu	1.56	0.78	0.52	0.52	3.13	1.34	6.25	3.65
M. punctulatus	0	0	0.52	0	0	0	0.52	0.52
Lepomis auritus	0	0	0	0	0	0	0.52	0
L. megalotis	0	0	1.56	0	0	0	0	0
Percidae								
Percina sciera	0	0	0	0	0	0	0	0

Table 42. (continued)

Species	Powell River Mile 144.6	149.3	153.9	156.0
N. volucellus	42.39	5.21	34.90	75.00
N. sp. (sawfin shiner)	3.26	0	11.46	9.38
Pimephales notatus	3.80	0.52	0	4.69
Catostomidae				
Hypentelium nigricans	2.17	7.81	13.02	9.90
Moxostoma duquesnei	0.54	0	0	0
M. macrolepidotum	1.09	0	0	0
M. erythrurum	0	0	0	0
Ictaluridae				
Noturus eleutherus	0	0	0	0
Cottidae				
Cottus carolinae	16.30	26.56	45.31	33.33
Centrarchidae				
Ambloplites rupestris	0	3.65	1.04	0.52
Micropterus dolomieu	0.54	3.65	2.08	0
M. punctulatus	0	1.04	0.52	0
Lepomis auritus	0.54	0	0.52	0
L. megalotis	0	0	0	0
Percidae				
Percina sciera	0	0.52	0.52	0

Table 42. (continued)

Species	Powell River Mile							
	117.3	117.4	119.3	120.4	123.0	127.2	136.7	141.3
P. caprodes	2.08	0.78	1.04	0	0.52	0	0	0.52
P. evides	33.33	17.19	11.46	29.17	36.98	20.09	19.79	13.02
P. aurantiaca	0.52	6.25	2.60	4.69	0	0.89	2.08	7.29
P. copelandi	0	0	0.52	0	0	1.79	0.52	0
Etheostoma blennioides	5.73	11.72	2.08	1.04	15.63	6.25	8.33	19.27
E. zonale	4.69	1.56	2.08	3.13	9.38	1.79	5.21	2.60
E. simoterum	0	2.34	0.52	1.04	0.52	0	6.77	6.77
E. stigmaeum	0	2.34	0	0	0	0	0.52	0
E. camurum	1.56	1.56	10.94	6.77	1.56	1.79	0	0.52
E. rufilineatum	39.58	39.84	88.54	51.04	64.58	114.73	22.92	19.27
E. vulneratum	0	0	0	0	1.04	0.45	0	0

Table 42. (continued)

Species	Powell River Mile 144.6	149.3	153.9	156.0
P. caprodes	0	0	0.52	0
P. evides	18.48	8.85	15.63	6.25
P. aurantiaca	1.09	4.69	1.56	1.04
P. copelandi	0	0	0	0
Etheostoma blennioides	4.89	14.58	26.56	5.21
E. zonale	2.17	8.85	12.50	1.04
E. simoterum	0.54	2.08	2.08	0.52
E. stigmaeum	0	0	0	0
E. camurum	0.54	0	0	0
E. rufilineatum	63.59	40.10	35.94	40.63
E. vulneratum	0	0	0	0

Appendix K. Microhabitat statistics for fish species captured at 12 sites on the Powell River (1990).

Table 43. Microhabitat statistics for all fish species captured at 12 sites on the Powell River in Virginia, sampled from June through August 1990. All values are means; S.D. = standard deviation.

Species	Depth (cm)	Depth (S.D.)	Velocity (cm/sec)	Velocity (S.D.)	Embeddedness (Unit)
Petromyzontidae					
Ichthyomyzon bdellium	35.0	17.6	56.9	19.4	4.9
Cyprinidae					
Rhinichthys atratulus	15.3	3.1	27.6	10.0	3.1
Campostoma anomalum	22.0	7.8	35.8	19.1	3.9
Semotilus atromaculatus	13.8	.	24.8	.	3.8
Nocomis micropogon	22.0	8.9	28.9	18.7	3.4
Erimystax dissimilis	35.5	10.1	47.2	28.4	4.0
E. insignis	26.0	5.5	48.4	17.2	4.6
Phenacobius uranops	29.3	7.5	46.1	22.8	4.3
Hybopsis amblops	25.5	9.0	21.3	13.3	2.5
Cyprinella galactura	36.6	14.0	32.7	21.2	3.3
C. spiloptera	32.3	13.7	33.9	21.0	3.3
Luxilis coccogenis	23.7	9.0	30.8	16.2	3.3
L. chrysocephalus	23.9	9.6	26.2	18.1	3.3
Lythrurus lirus	25.7	16.7	30.5	15.4	3.7
Notropis rubellus	24.7	13.5	36.9	20.4	4.0
N. leuciodus	27.3	8.6	44.2	18.7	4.4
N. photogenis	42.8	14.8	29.0	11.1	3.4
N. ariommus	28.5	4.5	30.5	12.3	3.4
N. telescopus	29.1	7.6	29.4	14.0	3.6

Table 43. (Continued)

Species	Distance from Shore (m)	Substrata Percent						
		Silt	Sand	Gravel	Pebble	Cobble	Boulder	Bedrock
Petromyzontidae								
Ichthyomyzon bdellium	8.6	0	1.3	22.9	35.0	36.9	4.0	0
Cyprinidae								
Rhinichthys atratulus	1.6	0.2	10.8	25.3	45.3	16.9	1.2	0.3
Campostoma anomalum	4.0	0.7	5.5	23.6	38.4	25.3	5.0	1.4
Semotilus atromaculatus	4.0	0	12.5	22.5	35.0	30.0	0	0
Nocomis micropogon	4.1	2.6	10.8	23.5	20.8	26.4	8.8	6.2
Erimystax dissimilis	9.1	0.5	4.6	27.1	28.5	31.8	3.4	4.0
E. insignis	5.3	0.2	2.4	23.6	39.4	29.4	3.6	1.4
Phenacobius uranops	9.4	0.4	6.0	23.5	31.7	31.2	4.9	2.2
Hybopsis amblops	3.6	1.2	17.2	30.5	28.9	13.7	3.5	3.7
Cyprinella galactura	7.2	1.3	7.4	25.9	20.7	25.8	5.8	11.9
C. spiloptera	4.2	3.2	11.2	19.3	21.9	25.2	11.3	6.3
Luxilis coccogenis	3.3	2.9	12.3	20.7	24.3	22.9	10.4	4.8
L. chrysocephalus	4.3	2.4	6.8	30.1	25.5	24.2	7.0	3.3
Lythrurus lirus	2.4	2.1	8.3	18.1	27.1	29.7	10.4	1.8
Notropis rubellus	6.8	0.7	7.4	22.7	34.7	25.9	4.9	3.6
N. leuciodus	8.4	0.3	4.3	21.2	28.7	33.0	6.5	5.9
N. photogenis	9.6	0	5.2	23.9	17.0	33.0	13.6	7.3
N. ariommus	4.8	0	1.6	38.9	41.4	15.9	2.2	0
N. telescopus	5.7	1.3	6.8	26.1	24.1	26.9	11.9	2.3

Table 43. (Continued)

Species	Water Willow (cm^2)	Log (cm^2)	Woody debris (cm^2)	Overhead vegetation (cm^2)	Rootwad (cm^2)	Filamentous algae (cm^2)	Pond weed (cm^2)	Rootmat (cm^2)
Petromyzontidae								
Ichthyomyzon bdellium	5,533	52	0	0	0	0	3,333	0
Cyprinidae								
Rhinichthys atratulus	16,665	60	58	234	39	8,894	1870	0
Campostoma anomalum	7,380	168	399	941	89	129	2,396	193
Semotilus atromaculatus	16,000	0	0	0	0	0	0	0
Nocomis micropogon	10,628	1,195	946	2,037	290	33	0	187
Erimystax dissimilis	1,162	77	65	248	8	2,884	329	0
E. insignis	3,115	61	81	1,662	0	37	37	0
Phenacobius uranops	1,200	83	141	573	71	1,333	52	0
Hybopsis amblops	5,321	180	697	333	49	0	0	0
Cyprinella galactura	1,730	287	967	388	0	0	0	0
C. spiloptera	3,968	1,498	2,081	1,342	616	0	0	69
Luxilis coccogenis	4,168	1,026	825	1,679	566	27	0	185
L. chrysocephalus	11,719	204	103	701	28	0	0	25
Lythrurus lirus	14,444	934	3,717	2,961	95	0	0	0
Notropis rubellus	8,046	175	240	226	139	374	27	4
N. leuciodus	2,525	243	180	385	65	59	0	48
N. photogenis	200	207	6	0	0	0	0	0
N. ariommus	21	27	587	0	0	0	0	0
N. telescopus	1,121	448	853	379	621	0	0	22

Table 43. (continued)

Species	Depth (cm)	Depth (S.D.)	Velocity (cm/sec)	Velocity (S.D.)	Embeddedness (Unit)
N. volucellus	26.3	10.7	29.6	19.3	3.3
N. sp. (sawfin shiner)	28.6	10.5	28.9	18.0	3.5
Pimephales notatus	21.4	7.5	16.9	14.1	2.1
Catostomidae					
Hypentelium nigricans	23.7	12.2	32.2	25.8	3.3
Moxostoma duquesnei	61.7	18.3	53.7	35.3	4.3
M. macrolepidotum	50.3	12.1	49.9	20.7	4.5
M. erythrurum	82.3	.	30.5	.	5
Ictaluridae					
Noturus eleutherus	26.5	8.1	55.7	23.5	4.4
Cottidae					
Cottus carolinae	20.4	9.4	39.8	25.4	3.8
Centrarchidae					
Ambloplites rupestris	28.8	11.7	25.6	14.7	3.1
Micropterus dolomieu	24.9	9.9	23.0	17.7	2.8
M. punctulatus	28.4	11.7	24.1	17.4	3.1
Lepomis auritus	29.2	5.5	5.6	4.0	1.1
L. megalotis	37.8	0	24.8	0	2.8
Percidae					
Percina sciera	22.1	0.9	24.1	16.8	3.5

Table 43. (continued)

Species	Distance from Shore (m)	Substrata Percent						
		Silt	Sand	Gravel	Pebble	Cobble	Boulder	Bedrock
N. volucellus	3.7	2.1	17.4	25.9	27.5	18.4	6.0	1.7
N. sp. (sawfin shiner)	5.9	1.1	10.4	27.4	29.7	21.9	4.3	4.9
Pimephales notatus	3.8	6.4	11.1	27.0	23.1	12.8	9.2	9.2
Catostomidae								
Hypentelium nigricans	4.9	3.2	11.6	23.4	31.9	19.9	5.6	3.7
Moxostoma duquesnei	10.7	0	2.2	17.2	17.8	27.2	15.6	20.0
M. macrolepidotum	10.4	0	2.7	19.2	18.0	26.7	10.5	23.0
M. erythrurum	13.2	0	0	10.0	17.5	30.0	42.5	0
Ictaluridae								
Noturus eleutherus	7.2	1.3	6.7	23.3	40.0	25.4	1.7	0.8
Cottidae								
Cottus carolinae	4.5	2.0	7.4	23.4	34.8	24.6	4.9	2.7
Centrarchidae								
Ambloplites rupestris	3.1	3.0	7.1	15.5	18.1	27.0	17.4	10.9
Micropterus dolomieu	3.4	2.6	11.6	17.0	18.0	18.6	14.0	17.7
M. punctulatus	3.9	2.5	9.2	21.7	25.8	24.2	8.3	8.3
Lepomis auritus	1.0	8.3	12.5	12.5	15.0	30.8	14.2	5.0
L. megalotis	4.0	0	15.0	67.5	10.0	7.5	0	0
Percidae								
Percina sciera	2.5	0	15.0	17.5	13.8	28.8	18.8	3.8

Table 43. (continued)

Species	Water Willow (cm^2)	Log (cm^2)	Woody debris (cm^2)	Overhead vegetation (cm^2)	Rootwad (cm^2)	Filamentous algae (cm^2)	Pond weed (cm^2)	Rootmat (cm^2)
N. volucellus	6,738	536	1,453	592	181	88	0	2
N. sp. (sawfin shiner)	3,084	573	992	335	154	30	37	0
Pimephales notatus	13,044	110	72	942	4	755	0	0
Catostomidae								
Hypentelium nigricans	8,488	527	438	647	148	299	1,346	0
Moxostoma duquesnei	0	155	0	0	0	6,000	0	0
M. macrolepidotum	0	0	12	0	0	0	0	0
M. erythrurum	0	621	0	0	0	0	0	0
Ictaluridae								
Noturus eleutherus	0	2,350	893	470	0	0	0	0
Cottidae								
Cottus carolinae	9,893	392	310	1,032	502	2,702	468	97
Centrarchidae								
Ambloplites rupestris	130	1,662	1,697	2,279	686	0	0	329
Micropterus dolomieu	6,435	861	715	1,194	361	0	1,565	140
M. punctulatus	6,667	171	657	1,481	0	0	0	0
Lepomis auritus	0	1,155	1,432	5,000	69	0	0	0
L. megalotis	0	2,086	45,228	0	0	0	0	0
Percidae								
Percina sciera	0	3,229	350	1,892	0	0	0	0

Table 43. (continued)

Species	Depth (cm)	Depth (S.D.)	Velocity (cm/sec)	Velocity (S.D.)	Embeddedness (Unit)
P. caprodes	40.6	13.8	39.1	18.5	3.8
P. evides	28.1	7.4	57.3	20.2	4.6
P. aurantiaca	41.6	11.3	47.8	26.8	4.0
P. copelandi	25.8	8.8	70.7	31.6	4.8
Etheostoma blennioides	27.4	9.4	41.7	21.0	4.1
E. zonale	24.1	6.8	36.6	20.4	3.8
E. simoterum	23.1	10.4	16.7	12.1	2.5
E. stigmaeum	12.4	3.9	4.3	2.0	1.9
E. camurum	30.7	8.8	74.5	22.1	4.9
E. rufilineatum	27.4	9.2	65.3	26.4	4.7
E. vulneratum	30.0	3.7	48.6	30.4	4.0

Table 43. (continued)

Species	Distance from Shore (m)	Substrata Percent						
		Silt	Sand	Gravel	Pebble	Cobble	Boulder	Bedrock
P. caprodes	7.3	0	10.8	24.8	34.0	21.5	5.5	2.8
P. evides	8.0	0.3	3.6	22.0	35.7	31.4	4.0	3.0
P. aurantiaca	8.9	0	3.8	19.9	21.0	30.9	10.9	13.6
P. copelandi	6.6	0	3.3	23.8	37.1	35.4	0.4	0
Etheostoma blennioides	7.5	0.6	5.6	21.6	24.4	30.5	8.5	8.6
E. zonale	6.3	1.7	7.1	22.4	25.2	30.6	9.2	3.3
E. simoterum	3.4	2.7	9.8	18.7	15.9	29.2	9.0	14.8
E. stigmaeum	1.8	8.8	23.8	38.1	6.9	16.9	5.0	0
E. camurum	7.0	0	2.3	19.7	38.3	36.6	2.0	1.1
E. rufilineatum	7.2	0.4	3.0	19.9	35.0	34.7	4.7	2.3
E. vulneratum	5.6	0	0	13.3	30.8	44.2	3.3	8.3

Table 43. (continued)

Species	Water Willow (cm^2)	Log (cm^2)	Woody debris (cm^2)	Overhead vegetation (cm^2)	Rootwad (cm^2)	Filamentous algae (cm^2)	Pond weed (cm^2)	Rootmat (cm^2)
P. caprodes	0	128	504	310	0	0	0	0
P. evides	814	202	177	261	135	1,117	1,156	9
P. aurantiaca	0	43	63	0	0	1,966	0	0
P. copelandi	6,000	0	110	0	250	3,333	0	0
Etheostoma blennioides	3,343	549	347	209	215	370	211	67
E. zonale	6,754	603	537	643	355	0	3,114	107
E. simoterum	10,270	222	183	2,800	646	37	0	112
E. stigmaeum	8,000	2,394	125	9,600	10,103	0	0	0
E. camurum	3,417	0	63	200	0	4,250	83	0
E. rufilineatum	4,173	163	154	392	126	2,785	805	31
E. vulneratum	0	0	0	0	0	0	1333	0

Appendix L. Effects of fright bias on sampling efficiency for stream fish assemblages.

Introduction

Collecting accurate, or at least precise, data on fish assemblages and fish-habitat relationships is difficult but critical for effective biomonitoring and management of aquatic systems (Hendricks et al. 1980; Bovee 1986; Heggenes et al. 1990; Persat and Copp 1990; Bayley and Li 1992; Lyons 1992; Angermeier and Smogor 1995). Electrofishing is a category of sampling gear that is extensively used for sampling fish populations in North America (Reynolds 1995). Concerns for accurate collection of data for aquatic biomonitoring and fisheries management have lead to numerous electrofishing efficiency studies (e.g., Chmielewski et al. 1973; Cross 1976; Heidinger et al. 1983; Wiley and Tsai 1983; Gardiner 1984; Layher and Maughan 1984; Koehn and McKenzie 1985; Cowx et al. 1988; Bayley et al. 1989). Some investigations have evaluated the relative effectiveness of different characteristics of electrical output (e.g., alternating current vs. direct current; Cross 1976; Hill and Willis 1994). Most studies, however, have evaluated the efficiency of one gear or method (Angermeier et al. 1991) or compared efficiencies of several gear types or methods (Reynolds and Simpson 1978; Heidinger et al. 1983; Wiley and Tsai 1983; Gardiner 1984; Layher and Maughan 1984; Larimore and Garrels 1985; Bayley et al. 1989; Dewey 1992). Early on, assessments were accomplished by using known populations in ponds or in streams (Chmielewski et al. 1973; Cross 1976; Koehn and McKenzie 1985). Efficiency in those studies was defined simply as the number of fish caught divided by the population size (Cross 1976).

Gear or method comparisons in aquatic systems with unknown fish population sizes have increased in frequency (Bayley et al. 1989; Layher and Maughan 1984; Wiley and Tsai 1983; Dewey 1992; Lyons 1992; Cunningham 1995; Simonson and Lyons 1995).

Efficiency in electrofishing can be accomplished by optimizing sampling design (Bain and Finn 1991; Cunningham 1995; Simonson and Lyons 1995), electrical field characteristics (Kolz and Reynolds 1989; Kolz 1993; Burkhardt and Gutreuter 1995; Kolz et al. 1995), and by minimizing fright bias (Fisher 1987; Bain and Finn 1991). Fright bias is evasion of electrical fields by fish due to disturbance from the electrofishing operation itself (Bovee 1982).

The upper Tennessee River system in Virginia and Tennessee has a very rich fish fauna (Etnier and Starnes 1993; Jenkins and Burkhead 1994). The Tennessee Valley Authority (TVA) has conducted, for biomonitoring purposes, regular and widespread fish sampling in this region over several years (TVA 1970; Saylor et al. 1988; Saylor and Ahlstedt 1990). Observations while using TVA methods for fish sampling in the Powell River, Virginia during 1989 raised concerns about the efficiency of sampling techniques. Accurate data on fish densities and particularly microhabitat use in the Powell River are required for reliable silt-tolerance classifications. Fish species silt-tolerance classifications can be used as important indicators in a biomonitoring program for the Powell River.

The development of prepositioned area shockers (PPAS), their apparently high effectiveness (Bain et al. 1985a; Kinsolving 1989; Peters et al. 1989; Weddle and Kessler

1993; Bain and Finn 1991; Fisher and Brown 1993; Irwin 1994; but see Dewey 1992), and the need for effective fish assemblage-level sampling techniques in the upper Tennessee River system led to this study; namely, to compare relative efficiencies of PPAS, TVA sampling techniques, and an electrofishing-seining method used in the Roanoke River, Virginia (open sampling technique, Vadas and Orth 1993). Although the PPAS (or at least fixed electrodes) has had some prior evaluation, previous studies were restricted to a single habitat type (Fisher 1987; Dewey 1992; Irwin 1994) or small streams (Larimore and Garrels 1985) and did not compare electrofishing gear and methods in common use within the medium to large streams.

The objectives of this study were to determine 1) which sampling technique was the most efficient for sampling shallow-water fish assemblages in rocky-bottomed, medium-sized streams, and 2) whether fright-bias appears to affect sampling efficiency. Specifically, I tested the hypothesis that the sampling methodology which causes the least disturbance and fright-bias will be the most efficient for sampling shallow-water fish assemblages. Results of this study can be used to select the most effective sampling gear and methodology for obtaining information on fish assemblages and fish-habitat associations in the upper Tennessee River drainage, particularly the Powell River in Virginia. Secondly, study results should generally apply to improving sampling efficiency of shallow-water fish assemblages within rocky-bottomed, medium-sized (fifth and sixth order) streams. Identification and use of the most efficient sampling technique should improve data accuracy or precision, thereby increasing the resolution of

biomonitoring programs.

Methods

Study site

The study site was located in the upper Roanoke River (Roanoke County, Virginia), a sixth order stream within the Valley and Ridge physiographic province (Jenkins and Burkhead 1994). This section of the stream has distinct riffle-pool sequences with coarse substrata and bedrock. Temperatures are cool, never exceeding 27 °C. Stream width at the study site was approximately 30 m. The study site length (within which all three techniques were applied) was approximately 750 m. Riparian vegetation, composed of trees, shrubs, and herbaceous plants, was relatively intact. Importantly, these physical characteristics are similar to the Powell River and, as such, research findings also should be applicable to fish sampling on the Powell River.

Description of sampling techniques

The sampling technique adopted from TVA was termed SPST (for single pass-single technique). This method consisted of electrofishing a defined area bounded on the downstream edge by a stationary, manned seine. For this study, a Coeffelt BP-6 backpack unit using the alternating current (AC) waveform was employed. The electrode system consisted of two hand-held booms with diamond-shaped electrodes. The

backpack operator electrofished an area approximately 5 m × 5 m, moving from an upstream to downstream direction, occasionally overturning stones to dislodge benthic fishes. A stationary seine, 6.1 m × 1.8 m with 6 mm mesh was placed on the downstream edge of the sample area immediately before electrofishing. The area was electrofished once (single pass). Upon completion, the seine was immediately raised and captured fishes were identified, counted, and measured for total length. Approximate sampling time was one minute per sample (does not include fish processing time). Three people were required to conduct this technique.

The open sampling technique (OST) sampled various-sized areas due to the study design of a concurrent investigation (sample area mean = 41 m^2). Once the sample area was determined, a block net (6 mm mesh) with iron poles was hammered into position at the downstream border. The free-standing block net enabled the entire method to be conducted by only two people. Seine hauls were made from upstream to downstream into the block net. Lateral seine hauls to the streambank were make in sample areas adjacent to the shore. Sometimes, portions of the sample area that were very shallow or contained high amounts of cover could not be seined. Seine hauls were made until the captures declined to zero (typically 5-10 hauls).

The electrofishing portion of the OST technique used shore-based electrofishing. The equipment configuration included a 3500 W streamside generator, a Coeffelt VVP-15 control box, extension cables, and two hand-held boom electrodes. The waveform used was AC. Sampling consisted of the electrode handler, assisted by a dip-netter,

moving from upstream to downstream and into the block net. Fishes were collected by dip-netting and by block net inspection. Captured fish were processed as with the SPST technique. Sampling time required for OST was approximately 30 min (excluding fish processing time). As previously mentioned, a minimum of two people is required to operate this technique.

The PPAS method used pre-positioned area shockers similar in design to Bain et al. (1985a) and Kinsolving (1989). The quadrats consisted of two, 4-m electrodes separated by 2-m weighted PVC spacer bars. See Appendix K for a complete description of the quadrats and general sampling methodology. Equipment set-up included a 3500 W, 240 V, AC generator mounted in a 3 m jonboat (Figure 54). The generator was connected directly to the quadrats via extension cords. Quadrats were placed on the substrata and left undisturbed for at least 20 min. At the conclusion of the rest period, a 3 m aluminum jonboat containing the generator was anchored approximately 15 m downstream and connected to the PPAS extension cord. Two people, one on each seine brail, carried a 4.3 × 1.8 m, 6 mm mesh bag seine above the stream surface to near the downstream end of the PPAS frame. These two netters also carried dip nets. The distance the netters halted from the downstream end of the PPAS varied. In flowing water, the distance was that amount that the netters could rapidly cover in two to three steps. In still water, the netters remained distant up to 10 - 15 m.

Once the netters halted downstream of the PPAS, a "ready" signal was given to

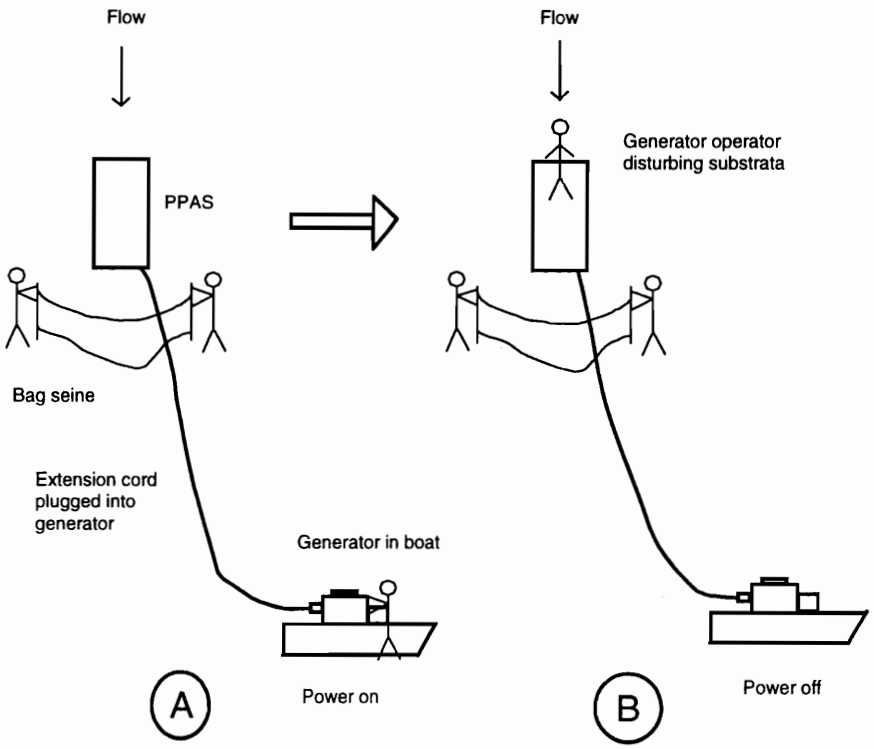

Figure 54. **A) Power is applied to the pre-positioned area shocker (PPAS) for 30 sec by the generator operator (standing by the boat). Water flow moves immobilized fish downstream into a stationary bag seine. B) Power to the PPAS is turned off. The generator operator immediately moves to the PPAS and disturbs the substrata to free any immobilized fish retained within the quadrat. As in A), water flow moves immobilized fish into the bag seine.**

the generator operator. The operator then simultaneously energized the PPAS and signaled. (Please note that, although the generator was running as the netters slowly came into position downstream of the PPAS, the PPAS was not being energized; energizing the PPAS was accomplished by working a switch on the generator). The netters rapidly took 1 - 3 steps and set the seine just downstream of the PPAS. The PPAS was energized for 30 sec. Stunned fish originating within the PPAS drifted into the seine or were dip-netted by the netters. After 30 sec, the power to the PPAS was shut off and the generator operator immediately moved to the PPAS and disturbed (kicked) the substratum in an upstream to downstream direction. The seine remained in the set position throughout this sampling effort. This action was intended to dislodge any fish caught in the rocky substratum for netting. All fish in the seine and dip-nets were taken to the jonboat for processing (identification, ennumeration, total length measurement). Once processing was completed, the adjacent PPAS was connected to the generator and the aforementioned procedure repeated. Approximate sample time was 2 min. A minimum of three people were required to conduct the PPAS method.

All three techniques were standardized by waveform (AC). AC tends to be less variable among electrofishing units than the more complex pulsed direct current (personal observation from oscilloscope analyses). Specific water conductivity (referenced to 25 °C) in the study section ranges between 190 - 260 µS/cm (William Ensign, Virginia Polytechnic Institute and State University, unpublished data) and was approximately 250 µS/cm during this study. These values are near the conductivity range that has been

observed to result in maximum efficiency of power transfer from water to fish (Kolz and Reynolds 1989; Burkhardt and Gutreuter 1995). As such, the electrofishing units were seemingly not power limited. Although this study cannot conclusively attribute any observed capture efficiency differences to differences in fright-bias caused by the various techniques, the near optimal conditions in the study, from an electrical perspective, tend to minimize the potential of power limitation as a confounding variable for any of the three gear types evaluated.

Study design

Three habitat types were recognized within the study reach: riffle, run, and pool. Riffles were shallow and fast areas with some surface turbulence. Run habitats were deeper, with little or no surface turbulence. Pool habitats had the slowest water velocities and generally were deep. Pool areas consisted of small shallow backwaters and larger, deeper areas that contained the thalweg.

Sampling a particular location by all techniques would likely be biased because of fish depletion from the initial sampling. This supposition was later confirmed by Vadas and Orth (1993). Instead, each location within a habitat type was sampled by only one technique. This design required that microhabitat characteristics, blocked by habitat type, are as similar as possible between locations. Otherwise, any observed differences in samples taken by the techniques may be due to habitat template differences rather than inherent differences in the technique itself. To provide for microhabitat similarity across

locations within a habitat type, locations were matched as closely as possible using similar depth and velocity ranges and surface turbulence characteristics. Locations were not matched by substrata characteristics due to water turbidity.

Microhabitat measurements of depth, average water-column velocity, and substrata were taken at each of three points within each area sampled. Habitat sampling points were located diagonally from the lower left hand corner (point one), through the area center (point 2), to the upper right hand corner (point 3) facing upstream. Depth was measured to the nearest 1 cm. Velocity measurements (cm/sec) were made with an electronic current meter and wading rod following the procedure outlined in Orth (1983). Substrate categories followed the modified Wentworth scale (Cummins 1962): boulder (>256 mm), cobble (64-256 mm), pebble (16-64 mm), gravel (2-16 mm), sand (0.0625-2 mm), and silt (<0.0625). A seventh substratum category, bedrock, also was recorded. Substratum sampling procedure was modified from the method of Bain et al. (1985b). For each sample point, at each of 6 dm marks on a meter stick, the substratum category was identified. This technique did not use visual integration required by the methods of Bain et al. (1985b) or that of estimating dominant and subdominant substrata at a point. Data collected from each sample point were habitat type (constant for the set of points within a particular sample area), depth, average water-column velocity, and proportions of each of seven substratum categories.

Fish species captured during the study were classified into groups (Table 47) according to *a priori* perceived ability to avoid sampling gear because of sampling

disturbance (i.e., fright bias). The interstitial benthic group was composed of fish species known to typically occupy spaces below the surface of the substratum. Benthic group members were those species that predominately occurred on top of the substratum. Water-column cyprinids were cyprinids that were basically mid-water oriented. Catostomids and centrarchids were taxonomic groupings of all members of the Catostomidae (suckers) and Centrarchidae (sunfishes), respectively. The *a priori* perceived group sensitivity to avoid sampling gear due to disturbance (i.e., exhibit fright) was as follows:

catostomid > centrarchid ≡ water-column cyprinid > benthic > interstitial benthic.

Statistical analyses

There were four main statistical hypotheses tested, two for investigating characteristics of sampled habitat and two for comparing technique efficiencies:

1) habitat types (riffle, run, pool) do not differ in velocity, depth, or substratum composition;
2) microhabitat sampled within a habitat type does not differ among techniques;
3) the species accumulation rate does not differ among techniques; and
4) the length-frequency distribution within any species group does not differ among techniques.

Statistically significant differences are set at $\alpha \leq 0.05$. Hypothesis one was used to test for an objective basis (i.e., physical habitat) that distinguishes subjectively determined habitat types. Hypothesis two was used to determine whether similar microhabitats were sampled by each technique within a habitat type. If habitat is an important factor for the structuring of fish assemblages (Gorman and Karr 1978; Gorman 1988), it is critical that habitat characterizations are done so that subjectively determined habitat types are evaluated for physical differences. Making gear comparisons in more than one habitat type increases the value of this study to biologists. For instance, gear types may not differ in all habitat types. If so, decisions on gear types to employ in a particular project can be made primarily on other factors, such as required number of field personnel.

Secondly, microhabitat partitioning has been commonly documented in stream fish assemblages (Ross 1986; Ross et al. 1987). Microhabitat selection may lead to discontinuous species distributions within a sampling site, thereby affecting species accumulation capture rates (Angermeier and Smogor 1995). It is important to assess whether similar microhabitat was sampled by each technique. If all techniques sampled similar microhabitats, it is expected that similarly structured fish assemblages were available to each technique. Any observed efficiency differences (e.g., species accumulation rates), therefore, would be attributable to the sampling technique characteristics and not to dissimilar fish assemblages sampled.

For the test of H_o number 1; depth and velocity variables were $\log_{10}(x + 1)$ transformed, and the proportion of each of the seven substratum categories was arcsine

squareroot transformed (Zar 1984). Transformed substratum proportions were subjected to a principle component analysis (PCA). Meaningful principle components (a set of new substrata variables) were included with transformed velocity and depth variables in a multiple analysis of variance (MANOVA). If the overall test probability level was significant (Wilk's λ), Tukey's means separation technique (family error rate: $p < 0.05$) was used to distinguish which habitat types differed along the significant habitat variables. All analyses were done on MINITAB (MINITAB, Inc., State College, Pennsylvania).

To test H_0 number 2: the testing procedure was identical to that for H_0 number one except that separate tests were performed within each habitat type.

Two approaches were used to evaluate relative efficiencies of the three techniques (hypotheses number 3 and 4). The species-area curve concept was adopted as the primary evaluation tool (Preston 1962). For purposes of this study, the species-area curve was considered to be an "accumulative fish species captured-area sampled" curve. This concept enabled evaluation of relative efficiencies by allowing comparison of species accumulation (capture) rates among techniques within each of three habitat types. Species accumulation rates were calculated by using the equation:

$$S_a = S_e(1 - e^{(-G * A)}),$$

where

S_a = number of species collected over a particular amount of

area sampled,

S_e = asymptotic species number for the stream section being sampled (the number was determined *a posteriori* by totaling species captured by all three methods) ,

G = species accumulation rate,

A = total area sampled, and

e = natural logarithm.

The larger the G value, the greater the species accumulation (or capture) rate.

Data for estimating G values were obtained from permutation of the original fish capture data. A software program (written by Mr. Blair Jones, Virginia Department of Game and Inland Fisheries) determined all possible species accumulations for the k^{th} ordered sample within a sampling technique. One model (described by the species accumulation equation) was fit through these data points. The formula for determining the number of species accumulations for the k^{th} ordered sample within each technique was

$$\left(\frac{T}{k}\right)=\frac{T!}{(T-k)!\times k!}$$

where T = total number of samples taken by the given technique, and

k = k^{th} ordered sample.

For example, 9 samples were taken by a particular technique in the riffle habitat. The number of species accumulations for the third ordered sample was

$$\binom{9}{3} = \frac{9!}{(9-3)! \times 3!} = 84$$

This was done because the actual order of samples taken in the field and the spatial pattern followed in the field did not have any particular significance. That is to say, it was neither valid nor robust to evaluate relative efficiencies by only using the particular order of samples taken in the field.

To test H_0 number 3: permutated fish species accumulation data were subjected to a non-linear regression technique (PROC NLIN, SAS 1989). The species accumulation rate (G) for each technique within each of the three habitat types was iteratively solved using the Marquardt method. Asymptotic 95 % confidence intervals for each G value were generated. Non-overlapping confidence intervals, in which the non-overlap distance was greater that 10% of the distance of the shorter confidence interval, indicated a significant difference between G values (Browne 1979). The technique having the highest G value within a habitat type was the most efficient.

To test H_0 number 4: for species groups that have a sufficient number of observations per cell (80 % of the cells must have expected frequencies $\geq$ 5; Zar 1984), length-frequency distributions were compared among groups using a chi-square goodness of fit (contingency table) test (SAS 1989). The underlying assumption was that larger

individuals within a group can move farther, faster. Less efficient techniques would then have a skewed distribution toward smaller-sized individuals relative to more efficient techniques. A skewed distribution to smaller-sized individuals also is evidence of inefficiency due to fright bias. Electrofishing often has been shown to be biased toward larger individuals (Sullivan 1956; Reynolds and Simpson 1978; Reynolds 1983). The existence of significant fright bias should exhibit the opposite pattern.

Finally, capture comparisons among techniques within each of five fish groups was evaluated. The proportion of samples taken by a technique that captured at least one individual of a group was noted for all techniques within all fish groups. Proportions for all groups by technique were determined by the equation:

$$\text{Group}_i \text{ proportion for a technique}_x = \text{NSC} \div \text{NST}$$

where,

NSC = number of samples taken by technique_x that captured at least one individual of group_i

NST = number of samples taken by technique_x in all habitat types within which group_i was captured by all techniques.

Comparisons of fish group capture proportions among techniques were performed to ascertain whether fright bias is a factor that varied among the three techniques.

Results and Discussion

OST sampled the greatest area, followed by SPST, and then PPAS (Table 44). The differences are substantial; total area sampled by OST was 26 % greater than SPST and 247 % greater than PPAS. Total area sampled by SPST was 175 % more than PPAS. PPAS, however, had the most samples taken.

The first three principle components (93 % of data variation) of a PCA on substratum were included with corresponding depth and velocity data for each habitat type in MANOVA tests. The purpose of these analyses was to determine whether subjectively determined habitat types differed physically. Habitat types differed in depth, velocity, and substratum composition (Table 45). Only substratum PC1 differed among habitat types (variable loadings: silt = 0.01, sand = 0.02, gravel = -0.14, pebble = -0.32, cobble = -0.47, boulder = -0.02, bedrock = 0.81). Run and pool habitats were deeper than riffles (Table 46). Riffle and runs had faster water velocities than pools. All habitats had different substratum compositions; riffles had the highest pebble and cobble proportions (Table 46), pools had the greatest amount of bedrock, and run substratum was intermediate in character. The overall conclusion is that the three nominal habitat types could be distinguished by some combination of physical habitat variables and, in combination with the qualitative descriptor of surface turbulence, were valid, distinguishable habitat components of the upper Roanoke River.

Mean water velocity, depth, and substratum proportions were compared within

Table 44. Number of samples and area sampled (m^2) by each technique within each of three habitat types and for the entire site. Sampling occurred on the Roanoke River in Virginia during May, 1990. OST = open sampling technique, PPAS = pre-positioned area shocker, SPST = single pass-single technique.

	Riffle		Run		Pool		Total	
Technique	Number	Area	Number	Area	Number	Area	Number	Area
OST	5	156	6	233	6	304	17	693
PPAS	9	72	6	48	10	80	25	200
SPST	10	250	5	125	7	175	22	550

Table 45. Results of a MANOVA test for testing H_0: habitat types do not differ in habitat characteristics (depth, velocity, substrata composition). Wilk's λ F = overall F-statistic, Wilk's λ p = overall probability level, F = univariate F-statistic, p = univariate probability level. PC1-PC3 = principle components 1, 2, and 3 for substrata.

	F	p
Wilk's λ	9.545	0.001
Depth	13.35	0.001
Velocity	15.36	0.001
PC1	18.32	0.001
PC2	0.24	0.786
PC3	0.97	0.384

Table 46. Mean (± standard deviation) water velocity, depth, and substratum proportions and principle component one scores for each of the three habitat types. All microhabitat measurements from the three techniques are combined to describe habitat type characteristics. Variable means with the same letter are not different (Tukey means separation technique, family level $\alpha = 0.05$). Substrata differences among habitat types were tested using principle component one scores as a combined substrata variable. Sampling occurred in the Roanoke River in Virginia. PC1 = principle component one from substrata analysis.

Habitat variable	Riffle	Run	Pool
Depth (cm)	35.5[a] (± 9.8)	54.3[b] (± 9.7)	54.4[b] (±21.7)
Velocity (cm/sec)	67.9[a] (±25.1)	64.3[a] (±20.8)	29.9[b] (±20.9)
Silt (%)	0	0	1.0 (± 3.6)
Sand (%)	0.2 (± 1.1)	0.3 (± 1.4)	0.5 (± 11.9)
Gravel (%)	8.1 (± 6.9)	5.9 (± 7.2)	1.9 (± 4.6)
Pebble (%)	25.7 (± 19.5)	17.0 (± 15.0)	12.8 (± 18.2)
Cobble (%)	51.9 (± 18.5)	32.4 (± 24.6)	14.3 (± 21.0)
Boulder (%)	10.2 (± 16.2)	9.8 (± 15.9)	8.2 (± 14.1)
Bedrock (%)	3.9 (± 19.3)	36.3 (± 36.5)	56.5 (± 38.3)
PC1	-29.64[a]	2.75[b]	27.58[c]

each habitat type for each of the three techniques (Table 47). The first three principle components (91-96 % of data variation) describing substrata variation were included with corresponding depth and velocity data for each habitat type in MANOVA tests. The purpose of these analyses was to detect differences in microhabitat sampled among the techniques. There were no differences in microhabitat sampled among techniques within the run habitat ($p < 0.130$). Differences in microhabitats, however, occurred among techniques within the riffle ($p < 0.002$) and pool ($p < 0.009$) habitat types. In both of these habitats, only substratum sampled by the OST technique was responsible for the significant microhabitat discrepancies ($p \leq 0.05$). Microhabitat sampled by OST within riffles had more boulder, slightly more cobble, and less pebble and gravel than habitat sampled by PPAS and SPST (Table 47). In pools, more boulder and less sandy substratum sampled by OST were primarily responsible for significant substratum differences between OST and SPST or PPAS (Table 47).

In summary, SPST and PPAS techniques did not differ in microhabitat sampled within any of the three habitat types. Microhabitat sampled by OST, conversely, did differ from SPST and PPAS microhabitat in substratum characteristics within riffle and pool habitats. Whether these differences in microhabitat result in differences in fish assemblage characteristics among areas sampled by the techniques is unknown. What is understood is that fish species richness and abundances are less over bedrock than other rocky substrata (Jenkins and Burkhead 1994; William Ensign, Virginia Polytechnic Institute and State University, personal observation). For all samples, 45.3 % of the

Table 47. Mean (± standard deviation) water velocity, depth, and substratum proportions within each of the three habitat types sampled by the three techniques. Sampling occurred on the Roanoke River in Virginia. OST = open sampling technique, PPAS = pre-positioned area shocker, SPST = single pass-single technique.

Habitat variable	Riffle			Run			Pool		
	OST	PPAS	SPST	OST	PPAS	SPST	OST	PPAS	SPST
Depth (cm)	32.0 (± 7.5)	37.7 (± 11.9)	32.8 (± 8.7)	51.2 (± 10.9)	55.1 (± 9.0)	54.8 (± 10.8)	57.6 (± 14.1)	55.3 (± 26.4)	52.0 (± 22.4)
Velocity (cm/sec)	67.4 (± 26.2)	68.6 (± 26.0)	67.6 (± 26.5)	58.4 (± 25.3)	70.3 (± 22.4)	64.0 (±13.9)	23.2 (± 14.5)	34.9 (± 27.9)	28.5 (± 13.4)
Silt (%)	0 (± 0)	0 (± 0)	0 (± 0)	0 (± 0)	0 (± 0)	0 (± 0)	2.8 (± 6.8)	0.6 (± 1.8)	0 (± 0)
Sand (%)	0 (± 0)	1.9 (± 0.6)	0 (± 0)	0.9 (± 2.3)	0 (± 0)	0 (± 0)	0 (± 0)	5.6 (± 11.7)	9.5 (± 16.3)
Gravel (%)	2.2 (± 3.1)	9.9 (± 7.7)	9.5 (± 6.4)	2.8 (± 4.7)	6.5 (± 8.9)	8.9 (± 7.5)	0 (± 0)	1.7 (± 3.8)	4.0 (± 7.0)
Pebble (%)	4.5 (± 4.7)	29.6 (± 19.3)	32.8 (± 17.7)	11.1 (± 11.7)	11.1 (± 12.2)	31.1 (± 13.9)	17.6 (± 22.6)	7.8 (± 11.5)	15.9 (± 22.8)
Cobble (%)	57.8 (± 16.5)	46.3 (± 23.4)	53.9 (± 14.4)	34.3 (± 30.7)	16.7 (± 14.1)	48.9 (± 16.4)	25.0 (± 26.9)	12.2 (± 22.0)	8.0 (± 10.6)

Table 47. (continued)

Habitat variable	Riffle			Run			Pool		
	OST	PPAS	SPST	OST	PPAS	SPST	OST	PPAS	SPST
Boulder (%)	35.5 (± 16.5)	3.1 (± 5.7)	3.9 (± 8.3)	8.3 (± 9.8)	7.4 (± 10.9)	14.4 (± 26.5)	25.0 (± 16.4)	3.9 (± 8.7)	0 (± 0)
Bedrock (%)	0 (± 0)	10.5 (± 31.5)	0 (± 0)	42.6 (± 36.3)	60.2 (± 29.9)	0 (± 0)	29.6 (± 26.2)	68.3 (± 34.9)	62.7 (± 44.5)

area sampled by PPAS was bedrock versus 22.7 % for SPST, and 25.5 % for OST. Area sampled by PPAS within each habitat type, moreover, had the greatest amounts of bedrock. Hence, it is likely that areas sampled by the PPAS technique had lower species richness and abundance relative to areas sampled by the other two techniques.

Total number of species captured was similar for each sampling technique (Table 48). Species captured by the OST and SPST techniques were more similar (Jaccard's Coefficient of Community [JCC; Pielou 1984] = 0.92), however, than either OST and PPAS (JCC = 0.82) or SPST and PPAS (JCC = 0.79). Twenty-three species total were captured by all three techniques. In addition, for the comparison of species accumulation rates, asymptotic species richness in each habitat was determined as 19 (riffle), 17 (run), and 22 (pool).

Examination of species accumulation rates in each habitat type indicated that PPAS was the most efficient method in all habitat types (Table 49; Figures 55-57). The OST and SPST methods had similar efficiencies in riffles whereas the SPST method was more efficient than the OST in run and pool habitats. This is good evidence that relatively small areas (8 m^2) can be sampled efficiently across a range of habitat types by using PPAS. The PPAS method was consistently the most efficient of the three techniques for sampling shallow-water fish assemblages in this upland stream.

Of additional interest are the relative efficiencies of the SPST and OST methods. Much more effort per sample was expended in the OST method as compared to SPST. Yet, the OST and SPST methods had similar species accumulation rates in riffle habitats.

Table 48. Fish species captured by each sampling technique during the study (all samples combined). Sampling occurred in the Roanoke River in Virginia during May, 1990. Technique abbreviations in Table 47. Species captured by a sampling technique are denoted by 'X'. Group designations are: B = benthic, C = catostomid, I = interstitial benthic, S = centrarchid, W = water column cyprinid.

Species	OST	PPAS	SPST	Group
Cyprinidae				
Campostoma anomalum	X	X	X	B
Nocomis leptocephalus	X	X	X	W
Cyprinella analostana	X	X	X	W
Luxilis cerasinus	X		X	W
L. albeolus	X	X	X	W
Lythrurus ardens	X	X	X	W
Notropis hudsonius		X		W
N. procne	X		X	W
Pimephales notatus	X		X	W
Catostomidae				
Hypentelium nigricans		X	X	C
Thorburnia rhothoeca		X		C
Scartomyzon cervinus	X	X	X	C
Moxostoma pappillosum	X	X		C
Ictaluridae				
Noturus gilberti	X	X	X	I
N. insignis	X	X	X	I

Table 48. (continued)

Species	OST	PPAS	SPST	Group
Salmonidae				
Oncorhynchus mykiss		X		–
Centrarchidae				
Ambloplites rupestris	X	X	X	S
Micropterus dolomieu	X	X		S
Lepomis auritus	X	X	X	S
Percidae				
Percina rex	X	X	X	B
P. roanoka	X	X	X	B
Etheostoma podostemone	X	X	X	B
E. flabellare	X	X	X	I
Total species captured	19	20	18	

Total species captured site-wide = 23

Table 49. Species accumulation rates (cumulative species number / m^2)[a] and (95 % confidence intervals) for each sampling technique within three habitat types. Different letter denotes non-overlapping 95 % confidence intervals. Technique abbreviation definitions in Table 47.

Habitat type	PPAS	SPST	OST
Riffle	0.0239[a] (0.0233-0.0245)	0.0082[b] (0.0081-0.0083)	0.0085[b] (0.0073-0.0100)
Run	0.0215[a] (0.0204-0.0226)	0.0102[b] (0.0093-0.0112)	0.0061[c] (0.0058-0.0065)
Pool	0.0111[a] (0.0109-0.0114)	0.0043[b] (0.0040-0.0046)	0.0035[c] (0.0030-0.0039)

[a] Asymptotic species richness was 19 (riffle), 17 (run), and 22 (pool).

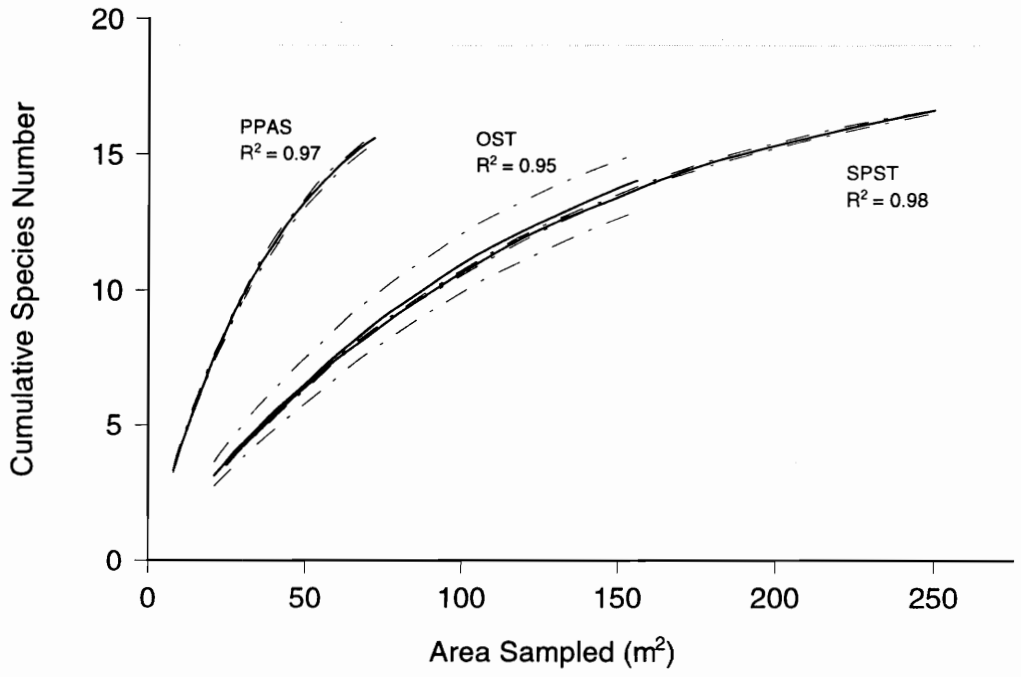

Figure 55. Species accumulation curves in riffle habitats for the three sampling methodologies tested (PPAS, OST, SPST). Accumulation curves and the corresponding 95% confidence bands are represented by solid and broken lines, respectively. Coefficients of determination (R^2), the proportion of variation explained by the model, is included for each accumulation curve. The horizontal line denotes the asymptotic species number (19).

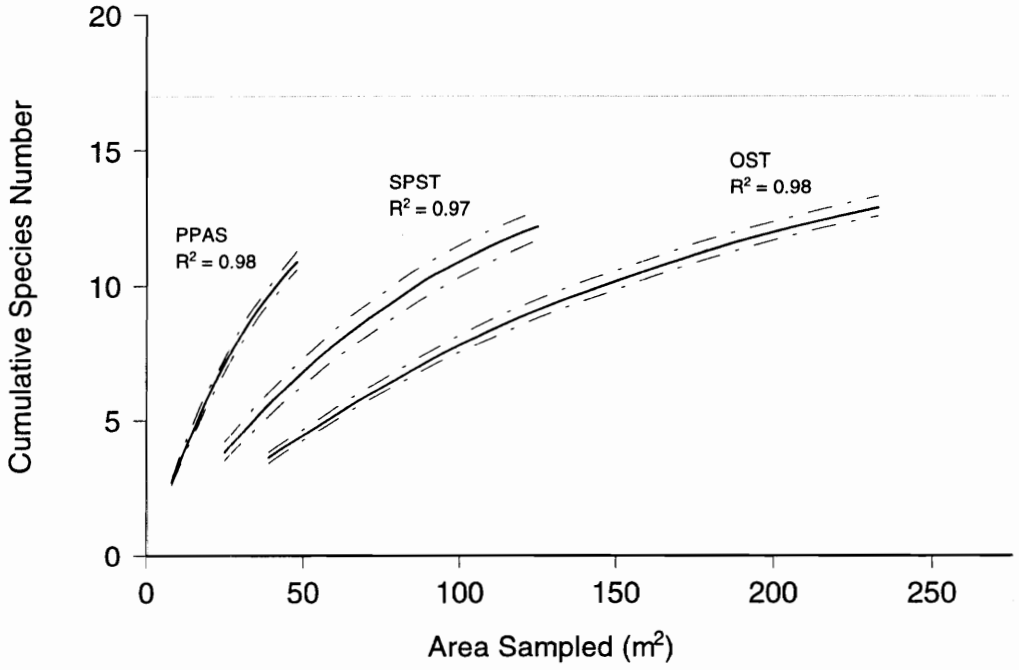

Figure 56. Species accumulation curves in run habitats for the three sampling methodologies tested (PPAS, OST, SPST). Accumulation curves and the corresponding 95% confidence bands are represented by solid and broken lines, respectively. Coefficients of determination (R^2), the proportion of variation explained by the model, is included for each accumulation curve. The horizontal line denotes the asymptotic species number (17).

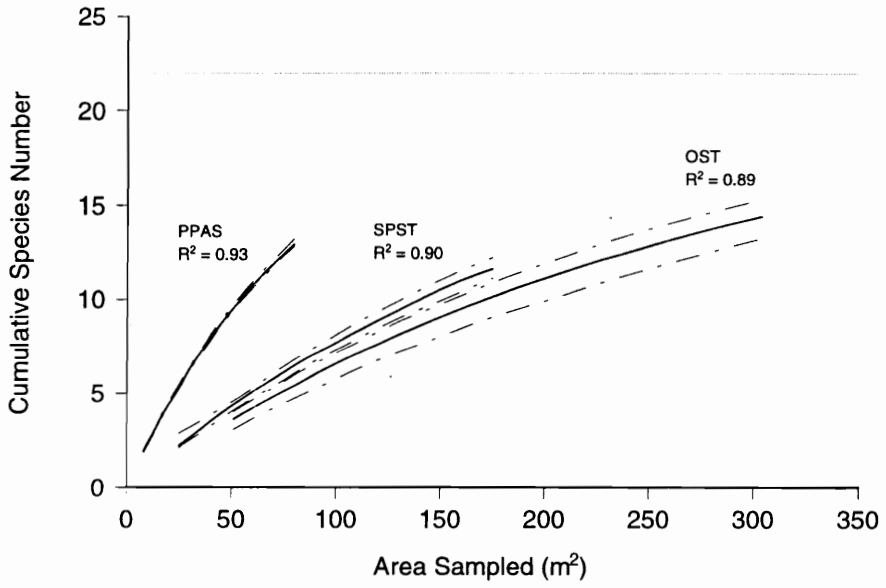

Figure 57. Species accumulation curves in pool habitats for the three sampling methodologies tested (PPAS, OST, SPST). Accumulation curves and the corresponding 95% confidence bands are represented by solid and broken lines, respectively. Coefficients of determination (R^2), the proportion of variation explained by the model, is included for each accumulation curve. The horizontal line denotes the asymptotic species number (22).

Moreover, SPST was more efficient in run habitats and pool habitats. This information suggests that fright-bias is a significant problem, particularly with the OST method. The OST method was considered *a priori* to be most likely to cause fright bias because of the disturbance resulting from setting the block net, the numerous passes, and the use of a dip-netter near the electrode handler. In contrast, the SPST method does not hammer the block net in place, only one quick pass is made, and only one person (the electrode handler) is near the electrical field. Apparently, even this protocol does not eliminate fright-bias since the remotely energized PPAS had highest efficiencies despite the smallest area sampled.

Capture efficiency also was examined by inspecting the sampling efficiencies for each of five fish groups. Proportion of samples taken with each technique that captured at least one member of each of five fish groups was evaluated (Table 50). Recall that for the PPAS method, area per sample was only 32 % and, on average, 20 % of area per sample of the SPST and OST methods, respectively. The PPAS method had the highest proportion of samples capturing catostomid and centrarchid species. Proportion of samples capturing benthic group and water-column cyprinid group members was higher for the SPST method. The OST method had the highest proportion of samples capturing interstitial benthic species. In terms of fright-bias, the PPAS method was most efficient in capturing species expected to exhibit fright bias. This is especially striking when considering the relatively small area sampled each time by the PPAS. The PPAS method, furthermore, performed similarly in capture efficiencies of the benthic and water column

Table 50. Proportion and (standard error of proportion) of samples taken by each technique that captured at least one member of each of five fish groups. Fish species assignment to groups are listed in Table 48. Technique and group abbreviation definitions in Tables 47 and 48.

Group	Technique PPAS	SPST	OST
I	0.32 (0.10)	0.59 (0.11)	0.71 (0.11)
B	0.56 (0.10)	0.64 (0.10)	0.53 (0.12)
C	0.48 (0.10)	0.36 (0.10)	0.35 (0.12)
S	0.32 (0.10)	0.14 (0.10)	0.29 (0.12)
W	0.56 (0.10)	0.68 (0.10)	0.59 (0.12)

cyprinid species, when considering relative area per sample of the three techniques. Conversely, the PPAS method fared poorly in capture efficiencies of the interstitial benthic species when compared to OST and SPST. The OST method may be useful when a study is concerned only with interstitial species, especially in view of the lower staff requirements. Interstitial species can be difficult to capture, and repetitive sampling of the stream bottom may then be required to obtain accurate data on presence or abundance. These statements should be viewed with some caution, however. As inferred by the group name, interstitial benthic species often occur under stones, and bedrock substratum is unsuitable habitat. The OST method sampled the least amount of bedrock, particularly compared to the PPAS method. Hence, differences in sample proportions of interstitial benthic species for the three techniques may be due to substratum discrepancies rather than actual differences in method efficiencies.

The third and final method used to evaluate relative efficiencies of the three gear types was to compare length-frequency distributions of individuals of fish groups captured by each technique. The working hypothesis was that significant fright bias occurred if length-frequency distributions within a group were significantly different among techniques.

Only water-column cyprinid, benthic, and interstitial benthic fishes were captured in sufficient numbers in each size category to allow Chi-square analysis. The distribution of captures in each size class within the water-column cyprinid group was significantly different ($p \leq 0.001$) among methods (Table 51). The PPAS data had a

relatively even size distribution of captures, whereas percent captures by the SPST and MPMT methods were skewed toward smaller size classes. It seems that pre-positioned area shockers caused less fright bias in capturing water-column cyprinids. Conversely, no differences among methods were observed for the benthic group ($p < 0.859$, Table 51). This is consistent with previous observations that benthic fishes tend to move less than water-column cyprinids when disturbances occur (e.g., Bain and Finn 1991). The results of the interstitial benthic group analysis do not, however, support this pattern (Table 62). The SPST and OST techniques captured larger individuals than the PPAS method ($p < 0.001$). However, the substratum discrepancies (more bedrock sampled by PPAS) among areas sampled by the three techniques may account for this difference. Overall, I conclude that the length-frequency distribution data support conclusions drawn from the species accumulation rates and the inspection of group capture proportions. The high sampling efficiency of the PPAS method is due to a reduced level of fright bias, resulting in less avoidance of gear and higher capture rates of most species.

Summary and Conclusions

The pre-positioned area shocker outperformed the other two techniques in assemblage-level sampling within three habitat types. The OST and SPST methods were similar overall, except that SPST was more efficient than OST in run habitats (and nearly in pools). The observed differences are seemingly a result of reduced fright bias when operating PPAS. A similar conclusion of reduced fright bias for PPAS, when compared

Table 51. Percent of total water column cyprinid group individuals, benthic group individuals, and interstitial benthic group individuals captured within each length interval by each sampling technique. Percent captures within each group length interval for each sampling technique sum to 100 %. Technique acronyms in Table 47.

	Technique		
Length interval (mm)	PPAS	SPST	OST
Water column cyprinid species			
20 - 59	20	25	24
60 - 99	38	55	61
100 - 139	33	19	10
≥140	9	1	5
Benthic species			
40 - 49	11	15	22
50 - 59	56	55	53
≥ 60	33	30	25
Interstitial benthic species			
40 - 49	45	14	12
50 - 59	30	51	23
≥ 60	25	35	65

to backpack electrofishing, was reached by Bain and Finn (1991). The OST method, however, may be more efficient if the research focus is restricted to interstitial benthic fishes. An added advantage is the reduced staff number required to employ the OST method. Unfortunately, substratum differences among techniques prevent a clear-cut recommendation for using OST to sample interstitial benthic species.

The PPAS method has several statistical and logistic advantages. PPAS can be employed in a systematic fashion using the transect-point sampling method (see Appendix M). Numerous, well defined samples can be taken which increases statistical power by reducing both Type I (α) and Type II (β) errors simultaneously (Zar 1984). Effort also is easily repeatable, and sampling can be consistent. Area sampled each time is consistent because power density drops rapidly away from the quadrat (Fisher and Brown 1993). The duration of electricity applied per sample can be easily controlled. The substratum in a 4 m $\times$ 2 m quadrat can easily and quickly be disturbed to dislodge immobilized and trapped fishes. Finally, in flowing water, turbidity is less of a problem compared to techniques that require dip-netting. The current carries immobilized fish into the seine during the application of electricity or during substratum disturbance.

Concern for electrofishing-induced fish injury has increased since the publication of Sharber and Carothers (1988) on rainbow trout injury (see Synder 1992). The small electrode diameter used in the design of PPAS for my study results in an intense electrical field close to the electrodes that could cause significant fish injury or stress (Mesa and Schreck 1989; Fisher and Brown 1993; Kolz et al. 1995). If fish injury is a concern (e.g.,

if rare benthic species are present), the PPAS method should be evaluated in light of fish injury potential compared to other techniques. Electric field characteristics theoretically can be made less damaging be increasing electrode diameter, lowering applied power, reducing time of quadrat electrification, or using less harmful waveforms such as continuous DC or low frequency pulsed DC (Synder 1992; Kolz 1993; Kolz et al. 1995). These design and operational changes may prevent significant stress or injury to the species of concern. However, only a controlled experiment using target or suitable surrogate species under a variety of design and operation modes will provide required information. If controlled experiments cannot be done or if PPAS is found to result in unacceptable injury rates, other sampling alternatives should be considered (see Ensign et al. 1995).

It is likely that the relatively low efficiencies of backpack and shore-based electrofishing and related techniques such as tow barge electrofishing, seining, etc. in large streams have contributed to the use of longitudinal distance compensation in sampling design (e.g., what stream length should be sampled to reach asymptotic fish species richness; Lyons 1992; Angermeier and Smogor 1995). Conversely, using PPAS methodology, the adequate sampling is not based on longitudinal distance sampled as much as efficiency at a specific location (see Norris et al. 1992). Many inefficient techniques sample larger areas (e.g., in my study) and result in electric field application to a large volume of water upon completion of sampling. Low efficiency results in less captured fish per unit effort but potentially high numbers of fish, though not captured, are

exposed to the electric field. Even brief exposure to electric fields can result in injury (Snyder 1992). Hence, if fish injury is a concern but electrofishing techniques must be used, PPAS (with possible design changes) may well be the least injurious and most efficient sampling alternative.

Besides using pre-positioned area shockers for fish presence-absence or abundance estimates, the pre-positioned area shocker holds promise for fish-habitat relationship studies as well (Bain and Finn 1991). The reduced fright bias of the pre-positioned area shocker and the relatively small, well defined sample area should facilitate more accurate assessments of habitat usage by fish species in rivers.

In conclusion, fish assemblage data collected by inefficient capture methods can impair the resolution and value of a biomonitoring program. With many programs in use today, it is essential that data are collected using the least-biased technique available. A sampling design incorporating pre-positioned area shockers may provide the efficiency required for a viable program. This gear type should be given serious consideration. In particular, it is recommended that biomonitoring programs sample fish assemblages in the upper Tennessee River drainage (e.g., the Powell River in Virginia) adopt the PPAS gear and methodology.

Appendix M. Description of pre-positioned area shocker design and sampling methodology.

Description of the pre-positioned area shocker

The pre-positioned area shocker used in this study was a two meter by four meter quadrat (Figure 58). The rigid end pieces were 2.5 cm diameter PVC pipe within which was placed 1.3 cm diameter rebar for weight. The rebar sections were secured by bolts placed between the rebar end and the coaxial cable (Figures 59 & 60). The flexible electrodes comprising the quadrat sides were made from 0.64 cm diameter coaxial cable ("TV wire"). Alternating sections of the coaxial cable were stripped of insulation, exposing the metallic wrapping. Stripped sections, 23 cm in length, alternated with 30.5 cm insulated sections. Approximately 40% of the insulation was removed from each electrode.

The coaxial cable was threaded through holes drilled near the end of the PVC pipe. The cable was secured by the attachment of a coaxial end piece (crimp "F" connector) on each end of the cable. Two conductor, 240 volt rated, 14 gauge, submersible pump wire was used for the extension cord (12 meters in length). Note that the pump wire was composed of two internal conductors encapsulated by a insulating jacket, similar to common extension cords. The pump wire was split at the end, one conductor was attached to each coaxial cable via a ring terminal nutted down on the crimp "F" connector by an inline "F" splice connector. Part of one conductor was secured to the "downstream" PVC pipe by plastic fasteners. A two pole, three wire, 250 volt, locking plug was attached to the distal end of the pump wire. More expensive waterproof

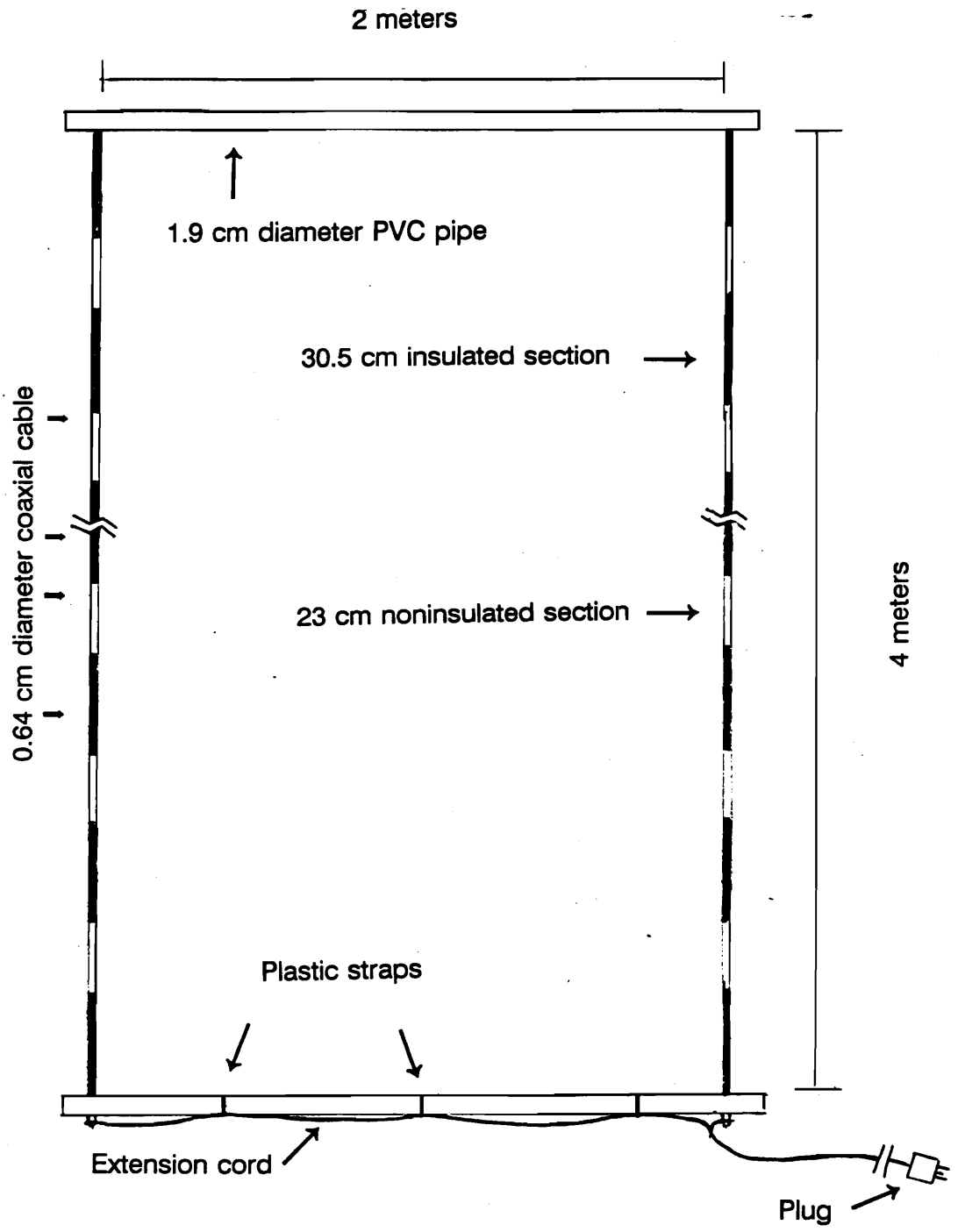

Figure 58. Dimensions and components of the pre-positioned area shocker.

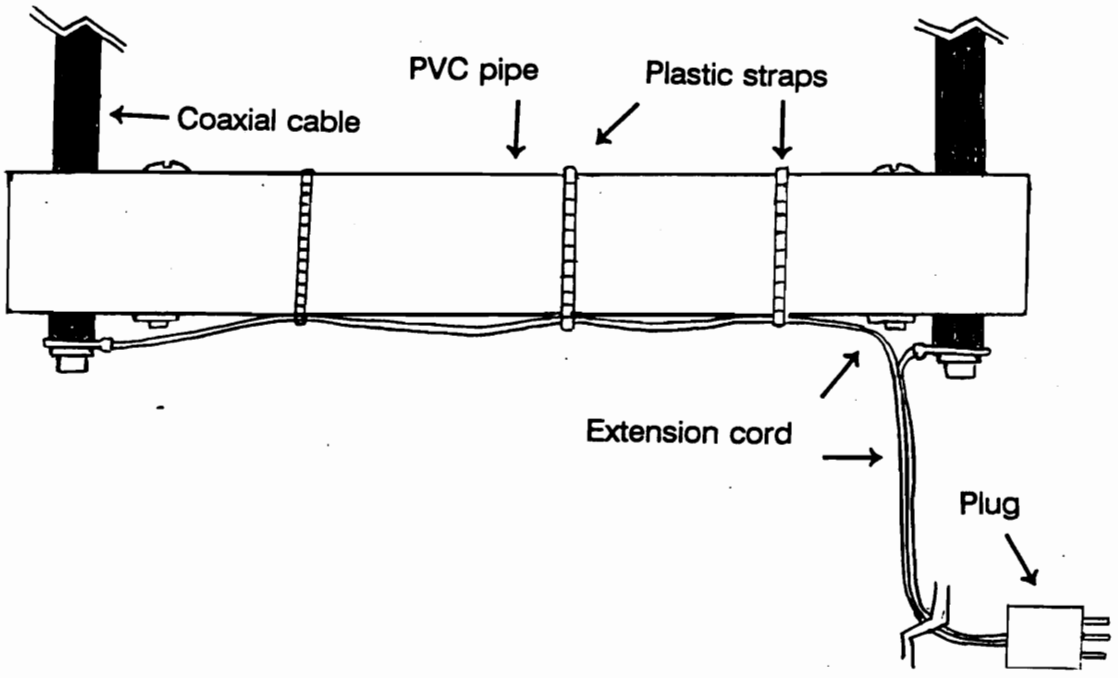

Figure 59. Lower or downstream end of the pre-positioned area shocker. Plastic straps secure one connector of the extension cord to the PVC end piece.

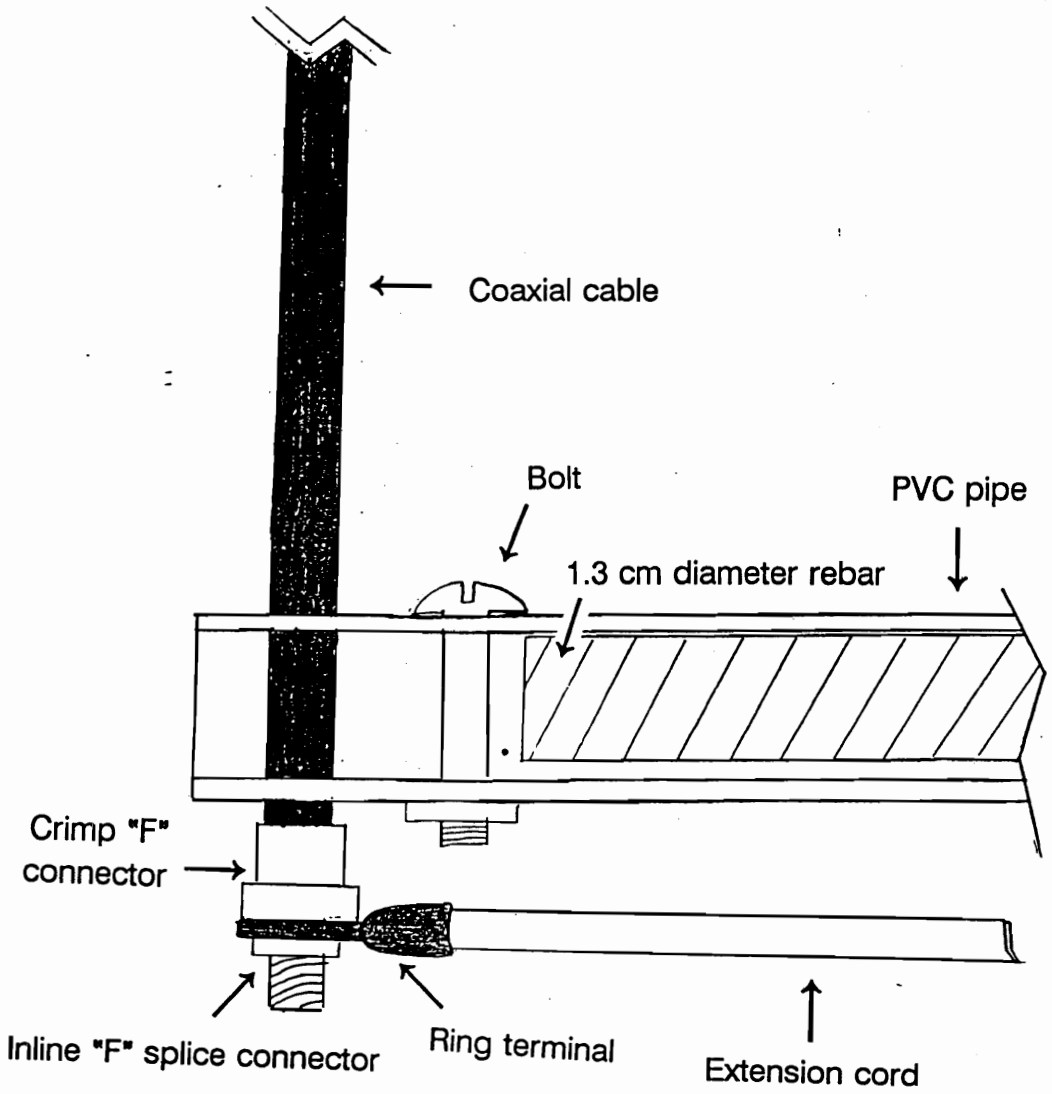

Figure 60. Lower or downstream left-hand corner of the pre-positioned area shocker.

plugs can be used instead; however, since the plugs and connectors were submerged in water for relatively long periods while uncoupled, I felt that waterproof plugs would afford little corrosion protection to the plug and connectors. In point of fact, these non-waterproof plugs provided long-lasting service in the field.

To energize the pre-positioned area shocker, the extension cord with plug was connected to a boat-mounted 3500 watt, 240 volt AC generator via an extension cord from the generator. The generator extension cord was a three meter long, 3 conductor, 14 gauge SO cable with a 240 volt locking plug attached to one end and a connector attached to the opposite end. The generator internal case neutral had been disconnected, thus only the two "hot" lines (120 volts each) from the generator were connected to the pre-positioned area shocker (one line to each coaxial electrode). Hence, the ground prong on the male plug served only as a locking mechanism to hold the plug (male) and connector\receptacle (female) together.

I connected the generator directly to the pre-positioned area shocker via the generator extension cord. Hence, I used the AC waveform only. One can certainly, and in many cases should, include a "pulsator" in the system between the generator and area shocker. The pulsator would give you the capability to use DC waveforms in addition to AC and allow for applied voltage control. Configuring this system would not be difficult. Pulsators are available from commercial sources.

Materials cost for each pre-positioned area shocker was approximately $25 in 1990. Most of the materials can be purchased at electrical supply companies.

Parts list for a 2 × 4 m pre-positioned area shocker

Quantity	Item	
1 ea	Plug:	2 pole, 3 wire, 20 amp, 250 volt, locking, NEMA L6-20P
9 m	Coaxial cable: ("TV wire")	0.64 cm (¼") diameter, aluminum wrap conductor (other diameters may be used; there also is coaxial cable that has a copper mesh wrap conductor, durability unknown)
4 ea	Crimp "F" connector:	There is a twist-on type "F" connector but its utility is unknown
4 ea	Inline "F" splice connector	
2 ea	Ring terminal:	Inside ring diameter must accommodate the inline "F" splice connector diameter
3.5 m	Rebar:	1.3 cm (½") diameter
12 m	Extension cord:	Submersible, 2 conductor, 240 volt, 14 gauge; the length that you decide upon is open to your judgement, you may need a lesser or greater length depending upon your situation
3-5 ea	Plastic fasteners	
4.5 m	PVC pipe:	2.5 cm (1") diameter

Extension cord from generator to shocker extension cord

1 ea	Connector:	2 pole, 3 wire, 20 amp, 250 volt, locking, NEMA L6-20C
1 ea	Plug:	2 pole, 3 wire, 20 amp, 250 volt, locking; The plug

		configuration must match that of the generator receptacle
3 m	SO cable:	3 conductor, 14 gauge (only 2 conductors are used)

Generator

3500 watt, 240 volt, AC

Description of sampling methodology using the pre-positioned area shocker

The following general procedure was used for fish sampling with pre-positioned area shockers (PPAS). Transects were placed in the study reach by randomly chosen locations within mesohabitat types (stratified-random sampling). Systematic placement of transects (non-stratified sampling) also can be a good approach. Four sample points on each transect were determined *a priori*: adjacent to bank, ⅓ channel width, ⅔ channel width, and four meters from the opposite bank. This configuration alternated across transects after a randomly chosen initial establishment (i.e. random determination of which bank the adjacent-to-bank sample point would be located on the initial downstream transect). To illustrate further, if the adjacent-to-bank sample point began on the right bank (the 4-meter-from-bank sample point location thereby would be near the left bank), the adjacent-to-bank sample point would be on the left bank on the next transect upstream, on the right bank on the second transect upstream, etc. Sample points were marked with anchored bobbers. Four PPAS were placed on each transect, one each at

points referenced above. These units were allowed to sit undisturbed for 20 min.

At the conclusion of the initial rest period, a 3 m aluminum jonboat containing a 3500 watt, 240 volt AC generator was anchored downstream and within reach of the extension cords of two PPAS (either the adjacent-to-bank and ⅓ width pair or the ⅔ width and 4 meter-from-the-bank pair). One PPAS was connected to the generator extension cord. Two people, one on each seine brail, carried a 4.3 × 1.8 m, 4.8 mm mesh bag seine above the stream surface to near the downstream end of the PPAS frame. These two netters also carried dip nets. The distance the netters halted from the downstream end of the PPAS varied. In flowing water, the distance was that amount that the netters could rapidly cover in two to three steps. In still water, the netters remained distant up to 10 - 15 m.

Once the netters halted downstream of the PPAS, a "ready" signal was given to the generator operator. The operator then simultaneously energized the PPAS and signaled. (Please note that, although the generator was running as the netters slowly came into position downstream of the PPAS, the PPAS was not being energized; energizing the PPAS was accomplished by working a switch on the generator). The netters rapidly took 1 - 3 steps and set the seine just downstream of the PPAS. The PPAS was energized for 30 sec. Stunned fish originating within the PPAS drifted into the seine or were dip-netted by the netters. After 30 seconds, the power to the PPAS was shut off and the generator operator immediately moved to the PPAS and disturbed (kicked) the substratum in an upstream to downstream direction. The seine remained in the set position throughout this

sampling effort. This action was intended to dislodge any fish which were caught in the rocky substratum and make them available to net. Sampling time was approximately two minutes (from quadrat electrification to the end of substrata disturbance).

All fish in the seine and dip-nets were taken back to the jonboat for processing. Once processing was completed, the adjacent PPAS was connected to the generator and the aforementioned procedure repeated. After fish were processed on the second PPAS, habitat measurements were taken within the two areas sampled by the PPAS. Next, the two sampled PPAS were removed and placed on the nearest two sample points on the immediate upstream transect. The boat with generator was then maneuvered and anchored downstream of the remaining two unsampled PPAS on the original transect. The sampling methodology was repeated here and, once completed, these two PPAS were removed and located on the unoccupied two sample points of the immediate upstream transect. By this time, the first relocated pair of PPAS were ready to be sampled (having laid undisturbed for › 20 min). This cycle was continuously repeated, thereby "shuffling" from downstream through upstream transects. This procedure is orderly and efficient. In addition, many samples can be taken in a day, the number depending upon factors such as stream width, depth, flow, and degree of substratum heterogeneity.

Vita

Alan Jon Temple was born in the northern Texas prairie town of Denton, on December 21, 1954. Soon thereafter, he and his family moved to a small town in rural Kentucky. There he grew up, in an environment where many people were close to the land, many practices of the past were still followed, a sense of community prevailed, and relatively wild places remained. This close interaction with land and people instilled in him respect for the land and a curiosity of nature as well as an interest in the role humans can and do have on the landscape; this background proved to be a *sine qua non* for his decision to study natural resource management and ecology. He graduated with a B.S. in biology from Western Kentucky University in August 1977 and a M.S. in wildlife science from Texas Tech University in August 1985. At Texas Tech, he met and eventually married the best girl this side of the Pecos, Carol Ann Gray. From West Texas, they moved to Blacksburg, Virginia where Alan was fortunate to achieve a long-held dream of conducting research on an Appalachian stream. Alan and Carol now reside in Martinsburg, West Virginia with their two wonderful children, Holly Anne and Nathaniel Jay. While Carol practices the ancient but increasingly rare responsibility of full time parenthood, Alan serves as a course leader for the U.S. Fish and Wildlife Service's National Conservation Training Center where he develops, conducts, and teaches courses in aquatic resource ecology and management. He also serves, for a real challenge, as president of a local church council and as a part-time Sunday school teacher. *Mirabile dictu.*