Spring-fed Streams in Virginia

and

Assessment of Livestock Impacts

by

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SPRING-FED STREAMS IN VIRGINIA AND ASSESSMENT OF LIVESTOCK IMPACTS

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(ABSTRACT)

The first of two studies surveyed fish communities and habitats in first-order spring streams in Virginia. Springs exhibited low species (3.3) and trophic guild (2.5) richness, and fish densities averaged 0.47 individuals/m^2. Two species, blacknose dace (Rhinichthys atratulus) and sculpins (Cottus sp.) dominated most sites. Spring size and location factors more effectively predicted fish community structure than did instream habitat or riparian condition, indicating that composition of Virginia spring fish communities was limited by access to colonization sources. Location, size, and instream habitat of springs were related to presence of common fish species and trophic guilds.

The second study evaluated livestock impacts on biotic communities of ten Virginia spring streams. Riparian and instream habitat was significantly altered and nitrate levels were elevated in heavily grazed watersheds. Within the fish community, piscivores and young-of-year benthic invertivores were adversely affected by habitat loss associated with increasing cattle densities. Benthic macroinvertebrate communities showed significant trends in composition across the gradient of grazing intensities. Fish communities were correlated with riparian condition, whereas benthic invertebrate communities were correlated with benthic habitat degradation and nitrate enrichment. Benthic macroinvertebrate community structure

was a reliable indicator of cattle-related spring stream degradation, but fish metrics were related to riparian condition.

ACKNOWLEDGMENTS

This study was funded in part by the Virginia Department of Game and Inland Fisheries and would not have been possible without their assistance. William Raines, Renee Speenburgh, Mark Ferguson, and Michael Stone provided field and laboratory support for the project and will always be remembered for their reliability, professionalism, and good humor. The Virginia Tech Aquaculture Center assisted with water quality analysis for the cattle impact study. Dr. Reese Voshell and Steve Hiner provided valuable assistance in the design of the benthic invertebrate portion of the cattle impact study.

In addition to those who directly participated in this project, I would like to thank Fred Harris, Chief of the Division of Boating and Inland Fisheries, North Carolina Wildlife Resources Commission, for his patience and support through the long and difficult years of my thesis preparation and dual allegiance. Thanks also to Jim Borawa and Scott Van Horn for proofreading, technical advice, and encouragement.

I am indebted to my graduate advisory committee for guiding me through what has become my life's greatest ordeal.

Finally, I thank Elizabeth Morrison Hughes for her love, companionship, and perseverance through six hard years.

DEDICATION

This document and the years of labor and study represented herein are dedicated to the memory of my parents, Donald David Yow and Edith Griffin Yow.

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CHAPTER I

Fish Communities and Habitat in Virginia Springs

ABSTRACT

Fish communities and habitat were surveyed in 65 first-order spring streams in eight river basins in the Ridge and Valley physiographic region of Virginia during 1990. Species richness was low in the 63 springs with fish populations. Forty fish species representing 22 genera and six trophic guilds were collected, averaging 3.3 species and 2.5 guilds per site. Guild richness varied significantly among river basins. Fish densities ranged from 0 to 2.4 individuals/m^2 and averaged 0.47 individuals/m^2. Blacknose dace (Rhinichthys atratulus) and sculpins (Cottus sp.) were the most widespread species, occurring in more than 60% of springs surveyed. The banded sculpin (C. carolinae) was the most common of six sculpin species sampled. Trout (Salmonidae) and sunfish (Centrarchidae) occurred in 26% and 18% of sites, respectively. Springs in different river basins were distinguished by sculpin species and clustered into two fish community types based on dominance by either sculpins or cyprinids. Species richness was highest in large springs adjacent to higher-order receiving streams, suggesting that spring accessibility influences community composition. Trout were associated with high discharge, riffle abundance, and low total cover; they were usually absent from small, heavily vegetated headwaters. Spring size and location factors more strongly effectively predicted community structure than did instream or riparian habitat factors, suggesting that location of

sample sites should be carefully considered in assessment studies. Composition of Virginia spring fish communities may be limited by access to colonization sources in downstream waters.

INTRODUCTION

GROUNDWATER RESOURCES

Springs have provided multiple benefits to Virginia residents for more than 150 years, meeting basic domestic and agricultural needs while serving as the basis for a recreational industry. Small communities and family farms have developed around these natural emergences of groundwater. Large springs, especially those with thermal influence or high mineral content, were long valued for their supposed therapeutic effects on a variety of human ailments, and towns such as Hot Springs and Warm Springs, Virginia, still serve vacationing tourists.

More than half of the population of the United States still obtain their drinking water from springs or wells. In rural areas, a majority of the residents drink groundwater (Hallberg 1987). Domestic groundwater use in Virginia affects nearly half of the population (Powell and Hamilton 1986) and reflects national trends (Murphey 1990). Groundwater quality has become an important concern to rural communities, although household water supplies are seldom treated or monitored (Hallberg 1987).

BENEFITS TO FISH AND INVERTEBRATES

Large springs often form extensive spring streams, providing habitat for an array of aquatic organisms. Mohn and Bugas (1980) identified over 450 km of high quality spring streams in Virginia. Stenothermic invertebrates are largely restricted to groundwater-fed streams (Pennak 1989, Crawford and Tarter 1979), and the cool summer temperatures afforded streams by springs influence distribution of fishes (Burton and

Odum 1945, Jenkins and Burkhead 1994). Coldwater stream fish habitat is largely dependent upon spring water sources, especially during periods of low rainfall (Smith 1981). Thermal influence from springs directly affects year-round survival of trout in many southwestern Virginia streams (Mohn and Bugas 1980).

In addition to sportfish, springs provide habitat for a variety of rare and highly specialized organisms. Virginia springs are inhabited by unique species of amphipods (Holsinger 1979a), isopods (Holsinger 1979b), and vascular plants (Porter 1979), and northern species of minnow and sculpin are largely restricted to spring streams in northern Virginia (Jenkins 1979, Jenkins and Burkhead 1994). Insects of spring streams include stonefly and mayfly nymphs, beetles, caddisfly larvae, and black fly larvae (Glazier and Gooch 1987). Gastropods, leeches, planarians, and sphaeriid clams are also common to springs (Pennak 1989). Chironomid larvae inhabit virtually every aquatic habitat in the Southeast, and some species are particularly adapted to springs and seeps (Hudson et al. 1990).

SPRING SURVEYS

Prior published studies of spring systems range from simple statewide inventories of spring locations to comprehensive descriptions of physiochemical conditions and biotic communities. Most published spring inventories provide physical and chemical data, but little biological information. Physical/chemical inventories of spring resources have been compiled for large springs nationwide (Meinzer 1927) and for several states, including Tennessee (Sun et al. 1963), West Virginia (McColloch 1986), and

Alabama (Chandler and Moore 1987). The earliest inventories of Virginia springs (Collins et al. 1930; Reeves 1932) provided data on location, geology and some water chemistry, but no biological data. Vineyard and Feder (1974) reported detailed physical, chemical and location data on Missouri springs which was supplemented with biological data (Lipscomb 1974, Pflieger 1974).

Biological surveys of springs typically have been limited to describing the ecology of a particular spring system or characterizing regional distribution of a particular taxonomic group. Systematic studies have characterized the taxonomy of spring biota of particular regions of the United States. Hopkins (1969) described vegetative communities of eleven springs in southern Illinois. Lipscomb (1974) listed 19 floral species from 16 large Missouri springs, although site-specific vegetative communities were not characterized. Glazier and Gooch (1987) used cluster analysis of macroinvertebrate assemblages to ordinate 15 Pennsylvania springs. Matthews et al. (1985) performed cluster analyses on fish assemblages in 50 Oklahoma springs. Armstrong and Williams (1971) described the fish communities of 68 springs in the southern bend of the Tennessee River. These surveys related chemical and physical habitat variables to characteristic floral or faunal communities. An intensive study of springs in Calhoun County, Alabama, inventoried fish species from 83 spring sites (Sizemore and Howell 1990), although no habitat data were obtained. Pflieger (1974) compiled an annotated list of invertebrates and fishes of Missouri springs, describing known distributions of each species.

The few ecological studies published have illustrated high variability among spring stream environments in the United States. Davidson and Wilding (1943) reported that the well-oxygenated water of Tyee Springs in southwestern Washington supported lentic and lotic macroinvertebrate fauna, but sculpins (Cottus sp.) were the only fish present. Sloan (1956) noted that the downstream increase in insect species richness and abundance in two Florida spring streams was probably related to poorly oxygenated waters in the spring head area. Teal (1957) described a diverse, but not abundant, community of macroinvertebrates and no fish species in a small Massachusetts spring. In contrast, a large spring stream in Kentucky contained 66 species of fishes and a rich invertebrate fauna (Minckley 1963). A large spring-fed stream in southern Illinois also contained a diverse fish fauna, including regionally rare species and fishes that prefer cooler water (Lewis 1957).

Helfrich et al. (1989) compiled a comprehensive database of Virginia spring sites synthesizing information on location, ownership, watershed character, flow, and water chemistry for approximately 1500 springs throughout the Commonwealth.

OBJECTIVES

This study characterizes fish communities and fish habitat in spring streams in Virginia. Data were collected between May 1990 and January 1991. Data analyses explore trends in fish community composition and its relation to habitat and spring location on the river continuum (Vannote et al. 1980). Null hypotheses are that fish community composition and species richness in spring streams do not differ among major river basins in Virginia and are not related to habitat or location variables.

METHODS

SITE SELECTION

Sample sites were selected from those springs in the Virginia springs database (Helfrich et al. 1989) for which discharge was sufficiently high to provide suitable stream fish habitat (>250 gpm). Springs were distributed throughout western and northern Virginia (Figure 1.1), representing all major river basins (Table 1.1). Sampling dates and sites were sequenced randomly by county (June-July 1990) or river basin (August 1990-January 1991). Fish collections and habitat data were obtained from 65 and 59 springs respectively.

FISHERIES ASSESSMENT

Fish were collected by electrofishing stream segments (10-30 m) in pool and riffle complexes along the entire accessible reach of the first order spring run. One to four segments were sampled per spring run, depending on the spring length and the relative diversity of stream habitat, with 10-100 m of stream channel sampled in each spring. The spring head pool, if accessible, was included in the upstream sample segment. All fish collected were identified to species and released, except for voucher specimens. Fish abundance was quantified as catch per unit effort (CPUE) in individuals/m^2. Total fish abundance and abundance of each fish species was computed for each spring.

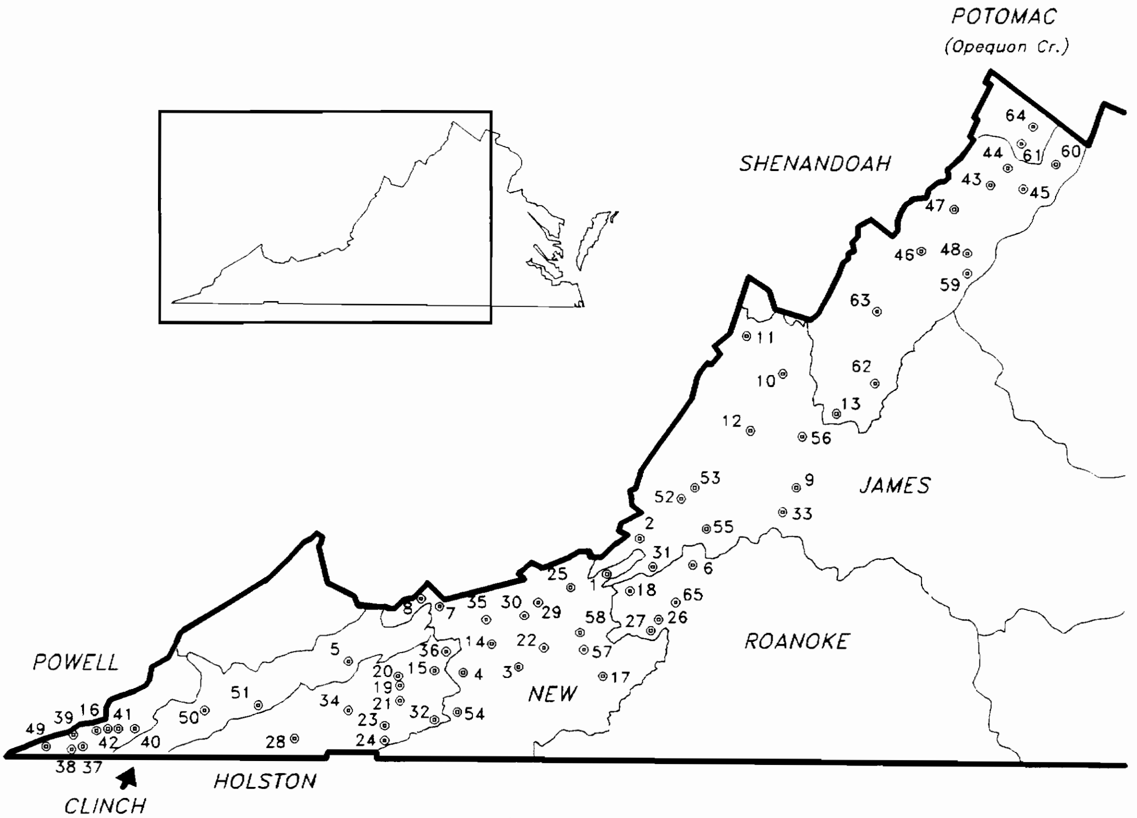


Figure 1.1. Sample sites for 1990 fish community survey, by river basin. Site numbers reflect order of sampling.

Table 1.1. Virginia springs selected as sample sites for the 1990 fish community survey. Site numbers reflect order of sampling. Spring numbers correspond to the Virginia Springs Database (Helfrich et al. 1989)

Site	Spring #	Spring Name	County	River Basin
1	045.04	Huffman	Craig	New
2	045.23	Dudding	Craig	James
3	197.30	Horkrader #1	Wythe	New
4	197.23	Hendricks	Wythe	New
5	167.11	Bolling	Russell	Clinch
6	023.03	Layman	Botetourt	Roanoke
7	185.35	Carter	Tazewell	New
8	185.36	Bailey #2	Tazewell	New
9	163.15	Leech	Rockbridge	James
10	091.07	Gorge	Highland	James
11	091.38	Stephenson	Highland	James
12	017.32	Big (Hunt Club)	Bath	James
13	015.14	Broadhead #1	Augusta	Shenandoah
14	021.06	Newberry	Bland	New
15	021.02	Hanshaw	Bland	Holston
16	105.25	Sims	Lee	Powell
17	121.06	Hambrie	Montgomery	New
18	121.58	Wright #2	Montgomery	Roanoke
19	173.63	Warren #2	Smyth	Holston
20	173.62	Warren #1	Smyth	Holston
21	173.24	Umbarger	Smyth	Holston
22	155.11	Warden #2	Pulaski	New
23	191.60	Walton	Washington	Holston
24	191.31	Cole #1	Washington	Holston
25	071.26	Hardwick	Giles	New
26	121.39	Vaughn #3	Montgomery	Roanoke
27	121.34	Weeks	Montgomery	Roanoke
28	191.15	King	Washington	Holston
29	071.16	Francis #5	Giles	New
30	071.11	Eaton #2	Giles	New
31	161.23	Substation	Roanoke	James
32	173.14	Keesling	Smyth	Holston
33	023.54	Breckenridge	Botetourt	James
34	191.45	Giesler	Washington	Holston
35	021.10	Walker	Bland	New
36	021.05	Sharon	Bland	Holston
37	105.32	Snodgrass	Lee	Powell
38	105.22	Fletchers	Lee	Powell
39	105.30	Upper California	Lee	Powell
40	105.39	Cheek	Lee	Powell
41	105.36	Natural Bridge #1	Lee	Powell
42 *	105.31	Natural Bridge #2	Lee	Powell
43	171.44	Hupp #2	Shenandoah	Shenandoah
44	069.02	Vaucluse #1	Frederick	Shenandoah
45	187.15	Cedarville #2	Warren	Shenandoah
46	171.06	Meyers	Shenandoah	Shenandoah
47	171.37	Fadeley	Shenandoah	Shenandoah
48	139.07	Yager	Page	Shenandoah
49	105.12	Hall	Lee	Powell
50	169.41	Carter	Scott	Clinch
51	169.32	Hale	Scott	Clinch
52	005.19	Fridley #2	Alleghany	James
53	005.20	Fridley #3	Alleghany	James
54	173.20	Pendry's Blue	Smyth	New
55	023.37	Karnes	Botetourt	James
56	017.30	Crystal	Bath	James
57	155.32	Boyd	Pulaski	New
58	155.41	Brown	Pulaski	New
59 *	139.16	Spitler	Page	Shenandoah
60 *	043.38	Byrd	Clarke	Shenandoah
61 *	069.10	Shawnee	Frederick	Potomac
62 *	015.01	Loth	Augusta	Shenandoah
63	165.32	Green Mount	Rockingham	Shenandoah
64	069.24	Branson	Frederick	Potomac
65 *	121.53	Big	Montgomery	Roanoke

*incomplete habitat data

HABITAT CHARACTERIZATION

Physical habitat variables (Table 1.2) were measured or estimated for each sample segment. Visual estimates of habitat characteristics for all sample sites were made by the same observer.

Total length (m) of spring run from head pool or vent to point of confluence with another stream was measured on site or estimated from topographic maps. Stream width (m) was recorded along one to three transects within each stream sample segment. Number of transects depended on variability of stream width. Depth (m) and substrate particle size measurements were taken at five intervals in each transect. Substrate type (Table 1.3) was scored according to categories adapted from Cummins (1962). Discharge was obtained by measuring water velocity (m/s) at 60 percent total depth at five transect intervals and computing Q (m^3/s) according to Armour et al. (1983). Flow was converted to gpm for comparison with historical spring discharge data. Mid-depth water temperature ($^{\circ}C$) was recorded at midstream in each fish sample segment.

Percentages of pool, riffle, run, and stream shading were visually estimated for each fish sample segment. Shading was defined as the portion of the stream surface directly beneath trees or objects which prevented direct sunlight from striking the water surface. Abundance and type of cover was estimated for each sample segment. Cover was defined as a percentage of total segment area containing a particular type of cover (Table 1.4).

Elevation of spring sites, obtained from USGS topographic maps, was estimated to the nearest contour line at the location of the spring head.

Table 1.2. Physical habitat and fish community variables evaluated in 1990 springs survey.

Habitat	Fish
Flow (m^3/s)*	CPUE (Individuals/m^2)*
Watershed use	Species richness
Elevation (m above msl)*	Presence/absence of trout
Emergence location (km)*	Presence/absence of key species
Stream length (m)*	Trophic guilds represented
Channel width (m)*	
Depth (m)*	
Temperature (^oC)*	
Substrate score*	
Substrate diversity*	
Percent pool*	
Percent riffle*	
Percent run	
Percent shading*	
Percent cover types*	

* variable used in multivariate analysis

Table 1.3. Substrate type, size and score used in habitat evaluation of 1990 spring sites, adapted from Cummins (1962).

Substrate Type	Substrate Size (mm)	Score
Organic	-	0
Clay/silt	(< 0.05)	1
Fine sand	(0.05 - 0.25)	2
Coarse sand	(0.25 - 1.00)	3
Very coarse sand	(1.00 - 2.00)	4
Fine gravel	(2.00 - 4.00)	5
Medium gravel	(4.00 - 8.00)	6
Coarse gravel	(8.00 - 16.00)	7
Small pebble	(16.00 - 32.00)	8
Large pebble	(32.00 - 64.00)	9
Cobble	(64.00 - 256.00)	10
Boulder	(>256.00)	11
Bedrock	-	12

Table 1.4. Cover categories used in habitat evaluation of 1990 spring study sites.

Cover Type	Description
Boulder	Rocks > 250 mm diameter
Log	All non-living woody debris
Rootwad	Woody root mass attached to bank or substrate
Aquatic vegetation	Vascular macrophytes, moss (not algae)
Bank vegetation	Riparian vegetation hanging into stream
Bank overhang	Undercut bank or rock ledge
Artificial	Manmade structures or debris (wall, pipe, rubbish, etc.)

Emergence location (point of contact between the spring run and the receiving stream system) was quantified as the distance (km) downstream from the origin of the receiving stream, as estimated from topographic maps.

DATA ANALYSIS

Spring fish species richness was compared among major river basins using Kruskall-Wallis rank methods. Species richness, total fish abundance, and relative trout abundance were tested for correlation (Kendall's Tau) with habitat variables. Habitat variables of springs containing trout were then compared to those with no trout using the Wilcoxon rank sum test. Similarly, habitat characteristics were compared between springs based on presence or absence of trophic guilds, based on information on feeding habits synthesized by Wallace et al. (1992). All distribution-free tests were performed as described by Hollander and Wolfe (1973). Interactions of habitat variables on species richness were tested using the General Linear Model (Steel and Torrie 1980).

Spring fish abundances (by species) were clustered using the complete-linkage method (Pielou 1984) to explore trends in community structure. A second analysis was performed lumping all sculpins as a single functional group to reduce effects of sculpin species distribution on cluster formation. Spring sites were also clustered by habitat variables to explore trends in species/trophic guild richness among multivariate habitat groups. Cluster analysis was performed using SYSTAT software (SYSTAT, Inc., 1990).

Principal components analysis (PCA) ordinated springs by habitat character, using PC factor analysis with the varimax orthogonal factor rotation (SAS Institute, Inc., 1988). The first three principal components were then plotted based on presence/absence of major species groups and on species richness. Canonical discriminant analysis (SAS) was used to explore multivariate habitat associations with presence or absence of trout.

RESULTS

GENERAL SPRING CHARACTER

Mean spring flow at 1990 fish sample sites was higher than the statewide mean (Figure 1.2) because larger springs were deliberately selected for fish survey. Most spring sites were located in pastured watersheds (Figure 1.3), and relatively few watersheds remain unimpacted by human settlement. Springs surveyed in 1990 were typically ranked as Class II springs (Helfrich et al. 1989), indicating that fisheries potential of these springs could be improved through better management of riparian areas.

SPRING FISH COMMUNITIES

Fish species richness was low (Mean = 3.3), with a typical spring containing one to four species (Figure 1.4). Species richness (Figure 1.5) varied widely among springs within river basins, and interbasin differences were not significant (Kruskal-Wallis HS = 13.56, $p = 0.06$).

Fish population density in springs (Figure 1.6) averaged 0.47 individuals/m^2. Density of game fish, if present, was considerably lower. Spring productivity showed no trends among river basins, and was not significantly related to geographic or habitat variables.

Commonly occurring fish species are shown in Figure 1.7. Sculpins were present in most springs. Sculpin species found in a spring were usually typical of those common to the respective river basin, with the exception of slimy sculpin (*Cottus cognatus*), which is at the southern

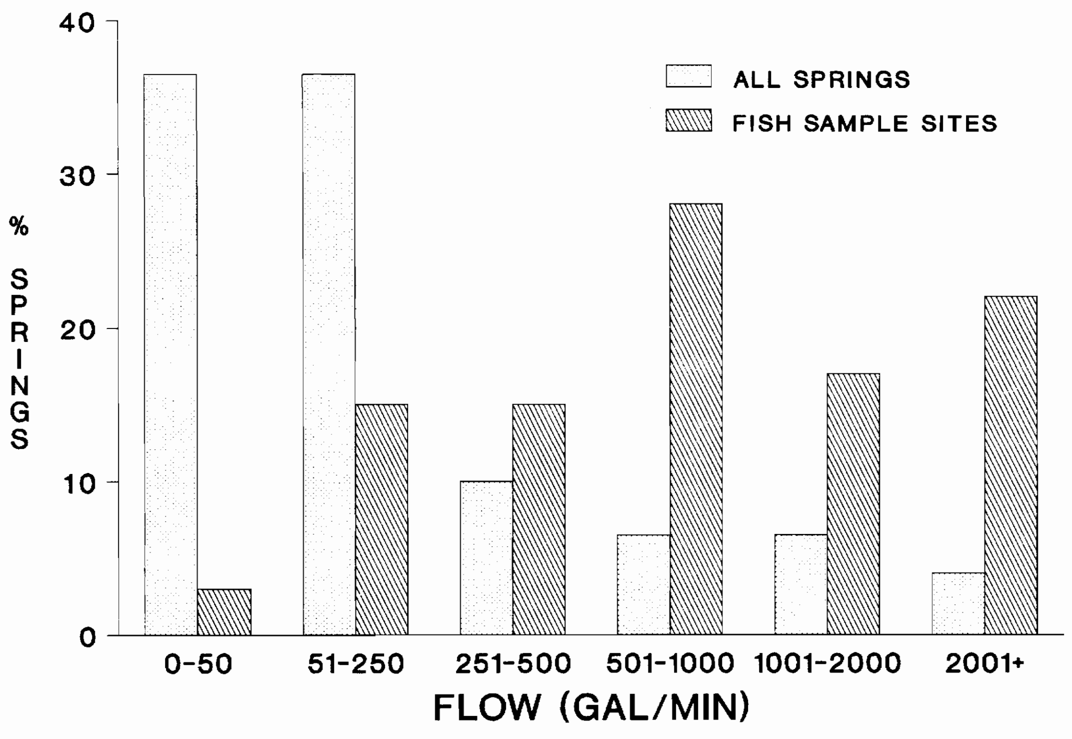


Figure 1.2. Spring flows (gal/min) of 1990 fish sample sites (N = 60), compared with all known flows of springs in Virginia Springs Database (N = 790).

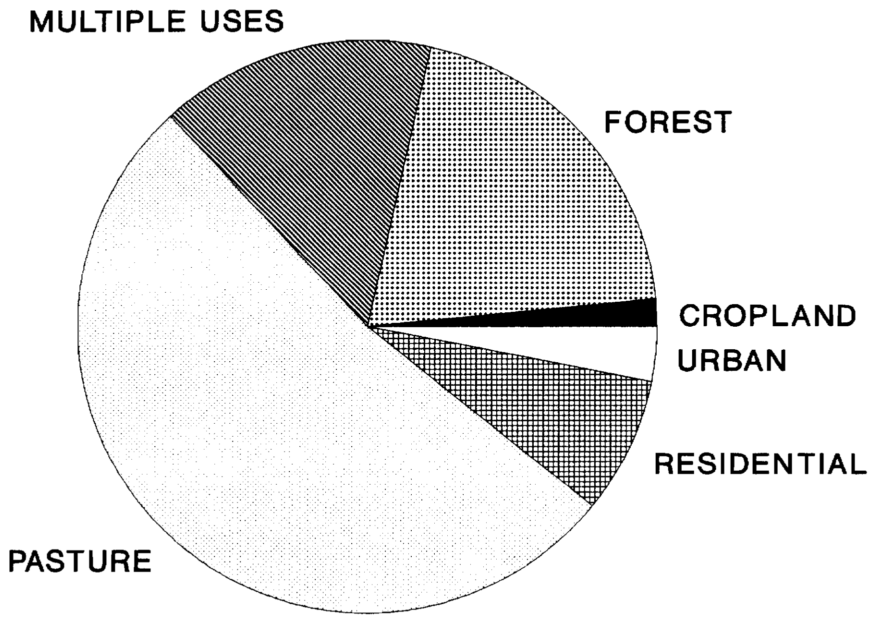


Figure 1.3. Primary land use in 65 spring watersheds, 1990 fish community survey.

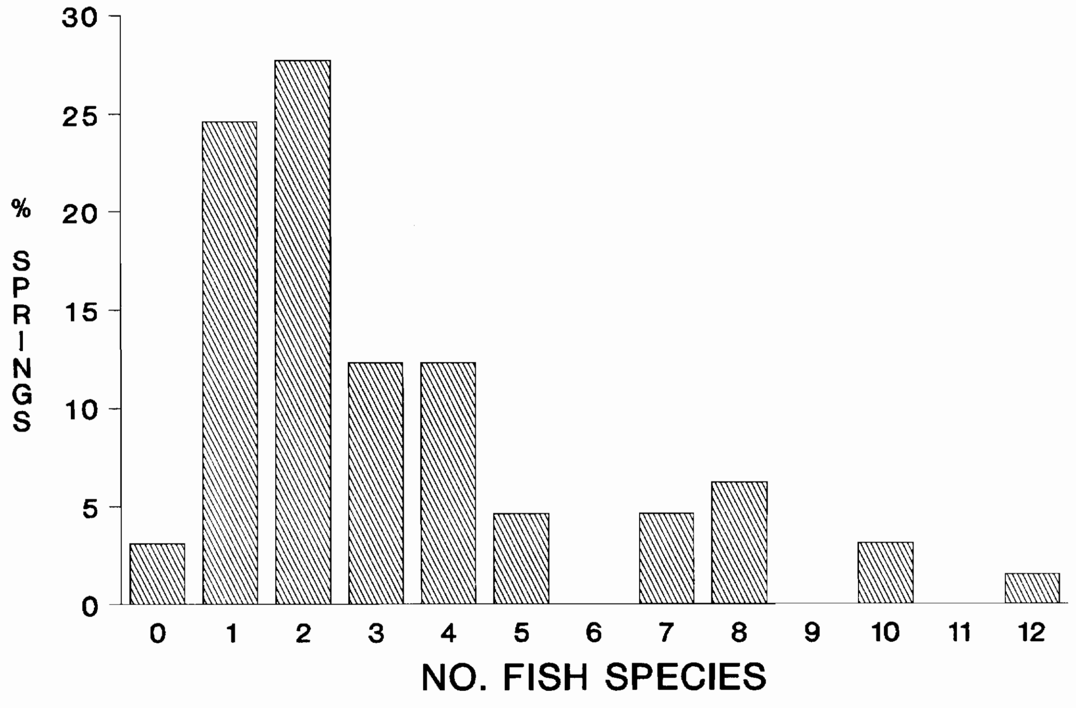


Figure 1.4. Fish species richness in 65 spring streams, 1990 fish community survey.

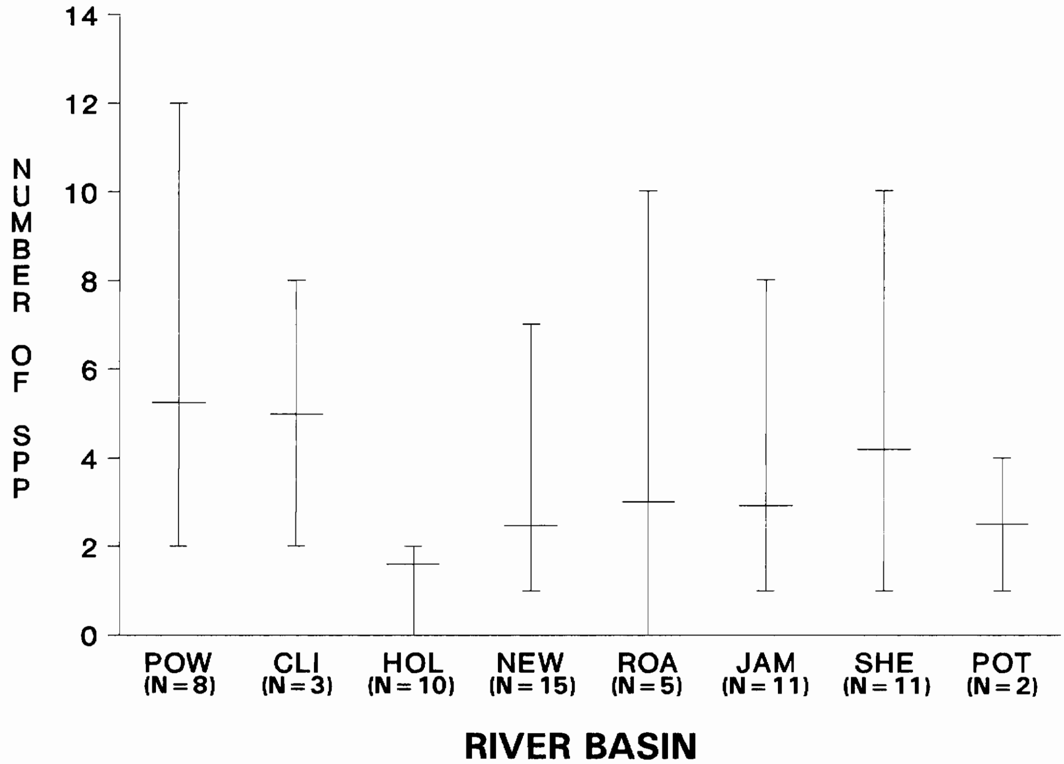


Figure 1.5. Mean and range of fish species richness values for springs in eight Virginia river basins, 1990 fish community survey: POW = Powell, CLI = Clinch, HOL = Holston, NEW = New, ROA = Roanoke, JAM = James, SHE = Shenandoah, POT = Potomac (Opequon Cr.).

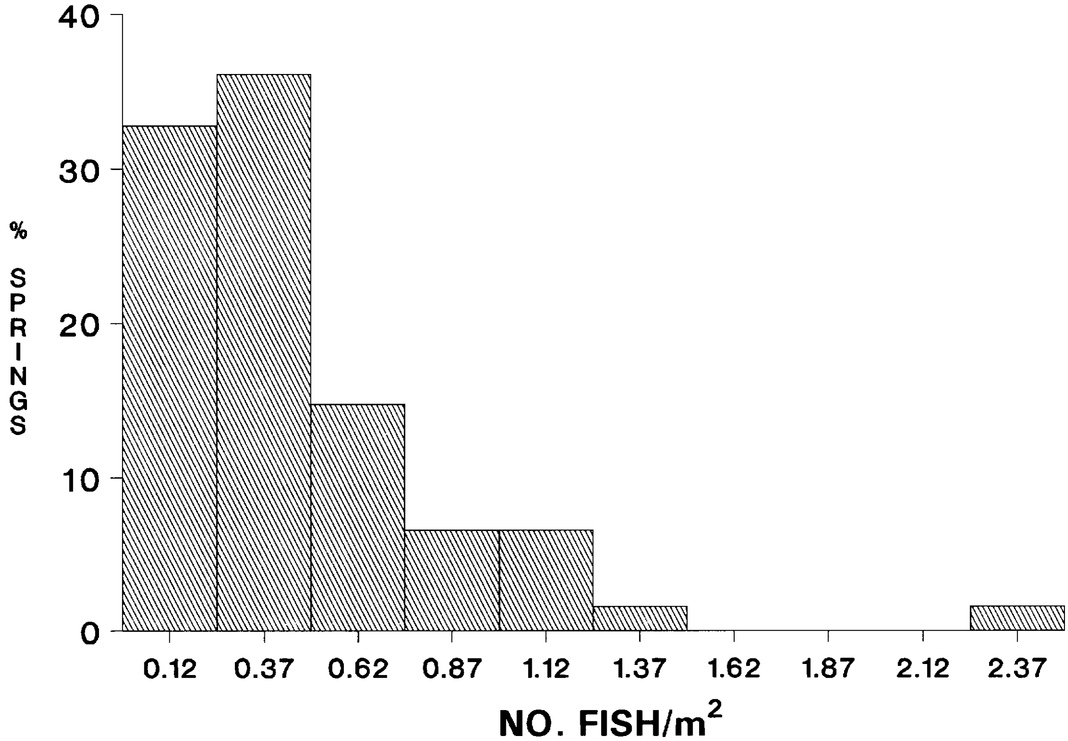


Figure 1.6. Frequency distribution of total fish abundance (No. fish/m^2 sampled) in 65 springs, 1990 fish community survey.

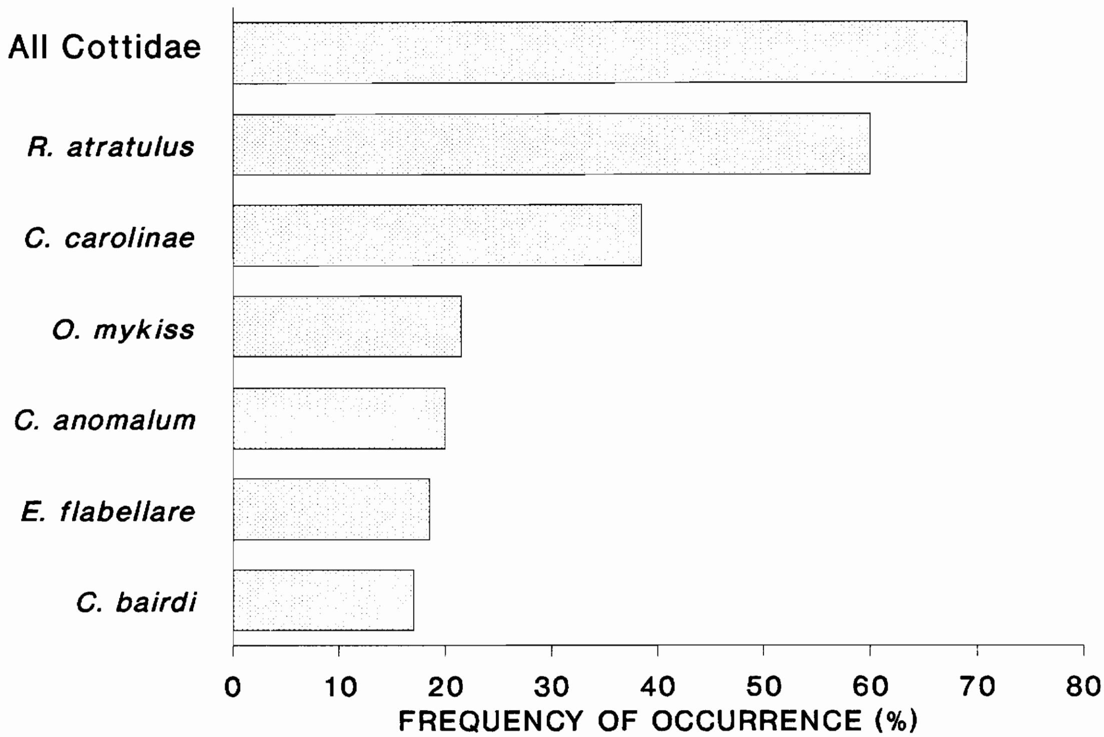


Figure 1.7. Frequency of occurrence of most common fish species in 65 springs, 1990 fish community survey.

extent of its range in northern Virginia and is largely confined to springfed streams (Jenkins and Burkhead 1994). Other sculpin species included banded sculpin (C. carolinae), mottled sculpin (C. bairdi), Potomac sculpin (C. girardi), black sculpin (C. baileyi), and undescribed broadband species. Only one species of sculpin was typically identified from a particular spring. The banded sculpin was the most frequently collected sculpin and second only to blacknose dace among all species in frequency of occurrence.

Blacknose dace (Rhinichthys atratulus) was the most common spring species, occurring in 60 percent of springs sampled. Other nongame fishes common to springs included central stoneroller (Campostoma anomalum) and fantail darter (Etheostoma flabellare). Numerous other minnows (18 spp.) and darters (3 spp.) were collected occasionally in springs, but no other minnow or darter species was found in more than 15 percent of springs sampled. Geographic distribution of minnows and darters was reflective of known ranges of most species. Springs in Tennessee River tributaries (Powell, Clinch, and Holston rivers) contained more darter species than other basins. Pearl dace (Margariscus margarita), at the southern extent of its range in Virginia (Jenkins and Burkhead 1994), was relatively common in springs in the northern river basins (Shenandoah River, Opequon Creek of Potomac basin).

Suckers were uncommon in springs, represented by only three species, most often by white sucker (Catostomus commersoni). Less than 15 percent of springs sampled contained suckers.

Trout were the most common game fish, but were only found in 17 of 65 springs (Figure 1.8). Rainbow trout (Oncorhynchus mykiss) was the only

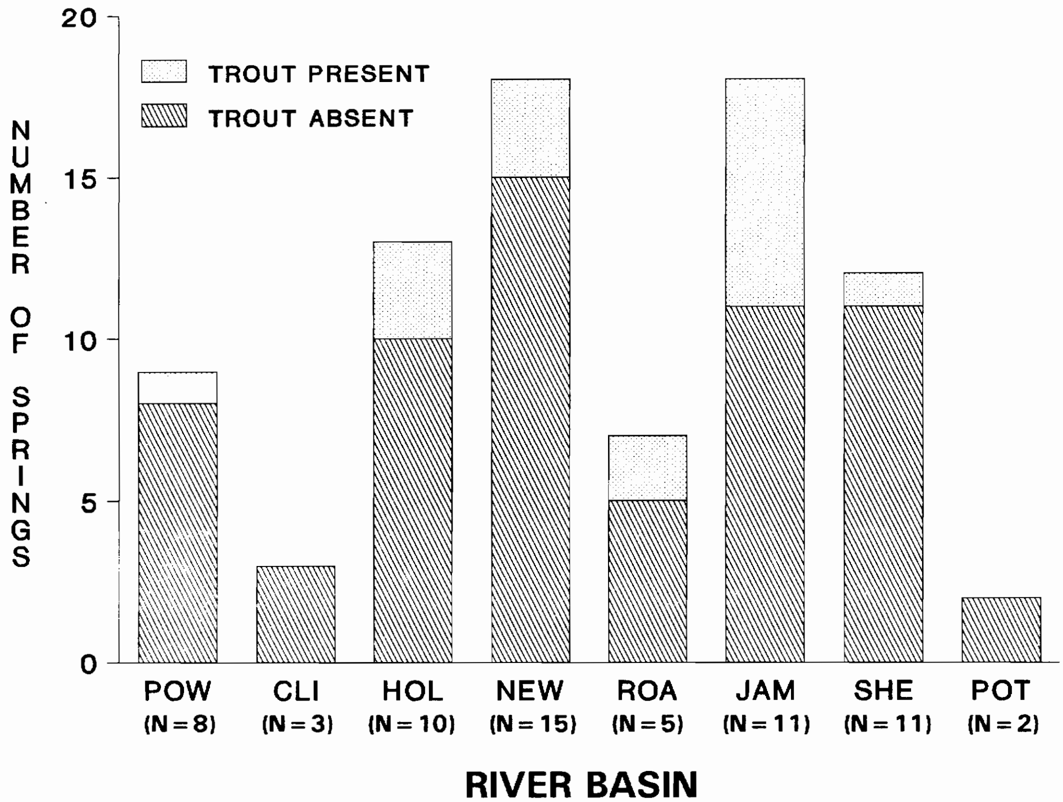


Figure 1.8. Proportion of 1990 spring samples containing trout, by river basin: POW = Powell, CLI = Clinch, HOL = Holston, NEW = New, ROA = Roanoke, JAM = James, SHE = Shenandoah, POT = Potomac (Opequon Cr.).

trout present in 13 of these sites, while brook trout (Salvelinus fontinalis) was the only trout species in three others. One spring in the New River basin (Craig County) fingerlings of both species, but had been stocked according to landowner accounts. All other brook trout were found in headwaters of the James River in Bath and Highland counties. Rainbow trout were represented in springs throughout northern and western Virginia, but were relatively uncommon among spring fish fauna.

Sunfish were present in 12 springs and were represented by four species: rock bass (Ambloplites rupestris), bluegill (Lepomis macrochirus), redbreast sunfish (L. auritus), and green sunfish (L. cyanellus). Sunfish were usually found in springs that directly contacted larger (2nd order or higher) streams, and relative abundance was typically low.

A complete list of fishes collected in 1990 spring survey sites is given in the appendix.

Three benthic and three non-benthic trophic guilds were represented in spring fish samples. Benthic invertivores included all sculpins and darters, as well as longnose dace (Rhinichthys cataractae). Benthic omnivores were represented by three sucker species and common carp (Cyprinus carpio). Central stoneroller was the only benthic herbivore found in spring samples. Non-benthic piscivore/invertivores included all trout and centrarchids. Non-benthic invertivores were represented by blacknose dace, rosyside dace (Clinostomus funduloides), whitetail shiner (Cyprinella galactura), spotfin shiner (Cyprinella spiloptera), bigeye chub (Hybopsis amblops), white shiner (Luxilus albeolus), warpaint shiner (Luxilus coccogenis), and telescope shiner (Notropis telescopus). All other cyprinids collected, including creek chub (<u>Semotilus</u> <u>atromaculatus</u>) and pearl dace, were non-benthic omnivores.

Like species richness, guild richness (Figure 1.9) was typically low (Mean = 2.5) in springs. Springs with multiple species tended toward diversification of feeding guilds. Guild richness differed significantly among river basins (Kruskal-Wallis HS = 15.61, p = 0.03), with higher trophic diversificationin the Shenandoah, Powell, and Clinch river basins.

Cluster analysis based on species composition (Figure 1.10) produced four fish community groups and an assortment of outliers. Community groups tended to cluster with those of a similar or adjacent river basin and illustrated trends in spring fish community structure. However, dissimilarity among groups was low compared to major outliers.

Group I fish communities (N = 10) were dominated by banded sculpin, often associated with blacknose dace. Rainbow trout, creek chub, stoneroller, and Tennessee snubnose darter were also represented. Three Group I springs contained only sculpin. All Group I springs were found in New River and Tennessee River tributaries.

Group II fish communities (N = 4) were dominated by mottled sculpin, although they differed in other characteristics. Species richness varied from one to eight and included both rainbow and brook trout. Both the New River basin and Atlantic Slope drainages were represented in Group II.

Group III fish communities (N = 13) represented a variety of sculpin-dace assemblages with lower overall catch rates than groups I or II. While

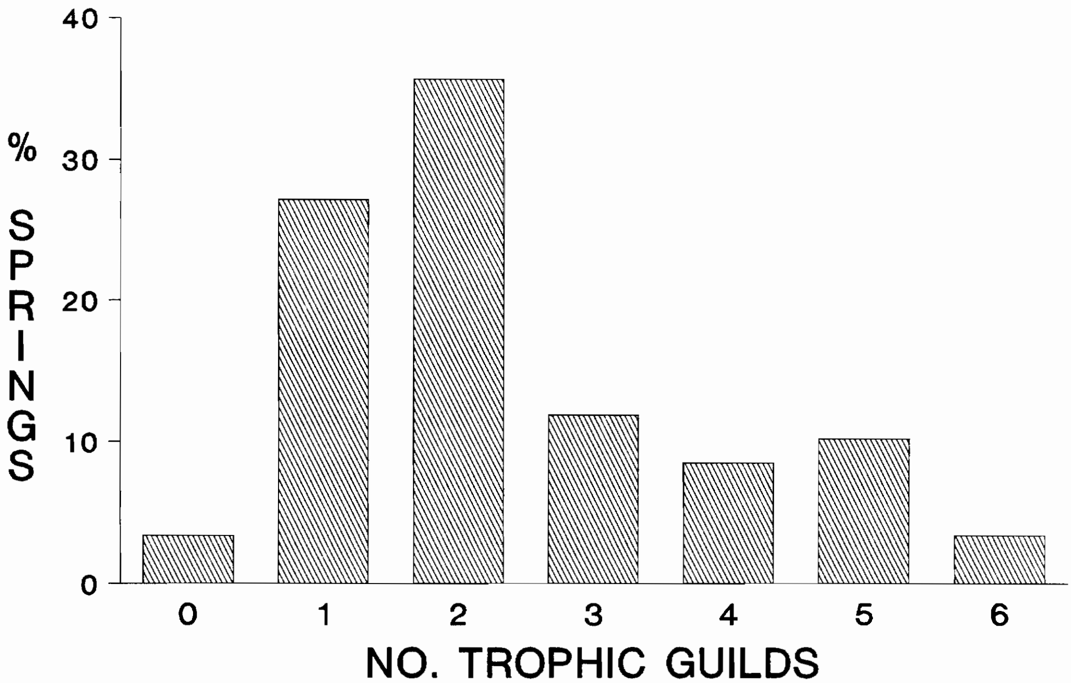


Figure 1.9. Trophic guild richness in 65 springs, 1990 fish community survey.

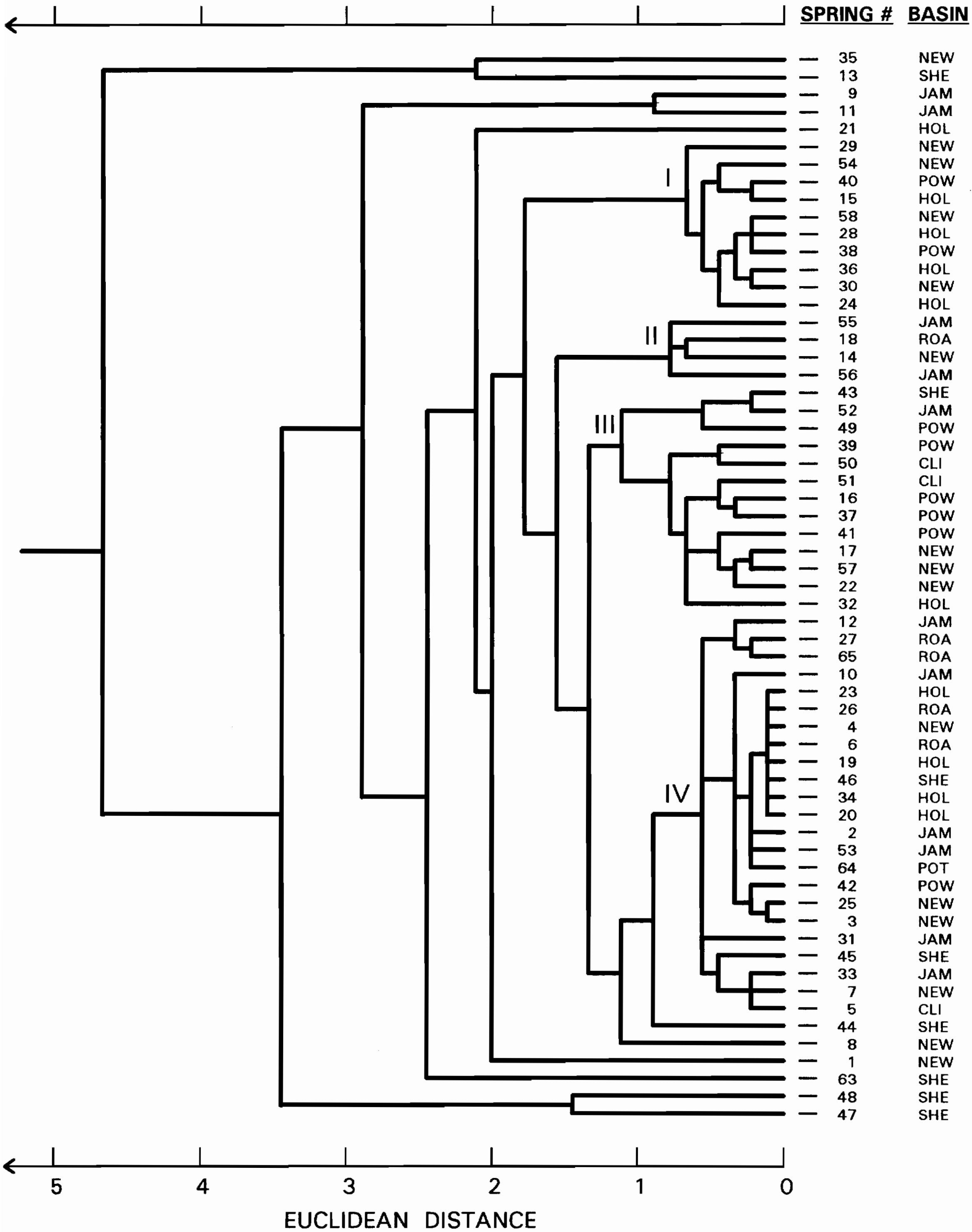


Figure 1.10. Dendrogram of similarity (complete linkage method) of fish populations in 61 Virginia springs, 1990 survey.

only two communities included rainbow trout, centrarchids were represented at five sites.

Group IV fish communities (N = 23) included a variety of assemblages with low overall catch rates. Two springs had no fish, and only four sites yielded more than 25 individuals. Species composition was variable, but typically included blacknose dace. Fantail darter was represented in four samples. Six Group IV springs contained rainbow trout, and five contained centrarchids.

Outliers represented unusually high catch rates and unique fish assemblages. Walker (#35) and Broadhead (#13) springs were dominated by blacknose dace. Fadeley (#47) and Yager (#48) springs were dominated by slimy sculpin and pearl dace. Leech (#9) and Stephenson (#11) springs represented brook trout/mottled sculpin-dominated communities in the James River basin. Green Mount Spring (#63), which flows through a heavily grazed watershed, was dominated by creek chub. Blacknose dace and white sucker were also abundant, and fantail darter was present in small numbers. Umbarger Spring (#21) contained only black sculpin, the only spring in the 1990 survey to yield this species. Huffman Spring (#1) was the only spring to yield both brook and rainbow trout, the result of fingerling stocking by the land owner. Bailey Spring (#8) contained an undescribed broadband sculpin found in headwaters of the Bluestone River. Vaucluse Spring (#44) was dominated by pearl dace. While no sculpin were found at this site, blacknose dace and fantail darter were present.

As expected, the inclusion of sculpins as one combined variable produced a simplified grouping of fish communities (Figure 1.11). Sites

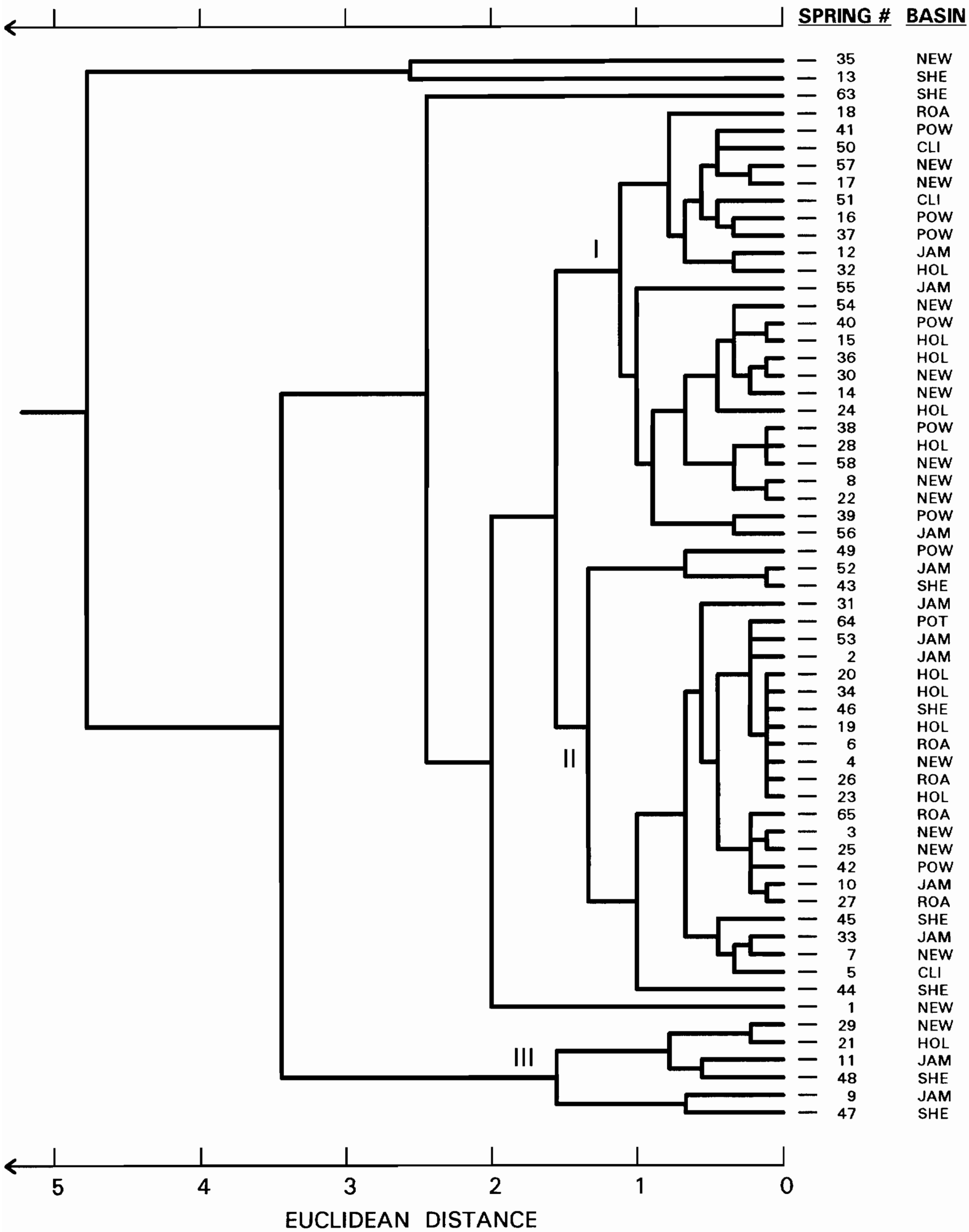


Figure 1.11. Dendrogram of similarity (61 Virginia springs, complete linkage method) with total sculpin abundance treated as one variable.

clustered into three groups, with Walker (#35), Broadhead (#13), Green Mount (#63), and Huffman (#1) springs remaining as outliers.

Group I fish communties (N = 25) all contained some variety of sculpin. Blacknose dace were present in 60% of these sites, and seven Group I springs contained trout. Centrarchids were also well represented, occurring at five sites. Only four Group I springs were found in Atlantic Slope basins.

Group II fish communities (N = 26) exhibited greater representation by cyprinids and generally low catch rates. Sculpins were absent from 15 of these sites, and two springs contained no fish. Trout were found in six Group II communities, and five yielded centrarchids.

Group III fish communities (N = 6) included an assortment of atypical sites, all of which contained sculpins. Game fish were incidental in Group III communities: two of three sites containing trout yielded one trout each, and one rock bass was collected at one site. Blacknose dace were found in only two of these communities. Group III springs included two of four brook trout sites and two slimy sculpin-pearl dace communities.

HABITAT INFLUENCES ON FISH COMMUNITY STRUCTURE

Descriptive habitat statistics for 1990 spring sites (Table 1.5) showed no significant correlation with total fish abundance or relative abundance of trout. Species richness was positively correlated with spring discharge (Kendall's KS = 3.31, p = 0.0005) and negatively correlated with elevation (KS = -3.28, p = 0.0005). While location of the spring confluence on the

Table 1.5. Descriptive parameters of geographic and physical habitat variables, 1990 study sites.

Variable	N	Mean	(SD)	Minimum	Maximum
Discharge ($m^3/s \times 10^{-2}$)	60	7.95	8.16	0.11	29.0
Elevation (m)	65	507	186	152	878
Emergence location (km)	65	26.2	48.7	0	300
Length (m)	65	873	1523	3.00	8000
Mean width (m)	61	3.68	1.64	1.20	9.00
Mean depth (m)	61	0.15	0.08	0.05	0.45
Mean temperature (^oC)	63	13.0	2.2	8.6	19.7
Mean substrate score	61	8.0	2.4	0	10.0
Mean substrate diversity	61	2.67	1.03	0.26	5.00
Pool habitat (%)	61	44.5	21.9	0	100
Riffle habitat (%)	61	29.4	22.4	0	100
Shading (%)	61	37.4	26.4	0	95.0
Boulder cover (%)	61	17.1	12.7	0	52.5
Log cover (%)	61	5.85	6.48	0	25.0
Rootwad cover (%)	61	1.28	3.17	0	20.0
Aquatic vegetation (%)	61	23.5	17.9	0	70.0
Bank vegetation (%)	61	7.37	7.07	0	40.0
Bank overhang (%)	61	5.64	5.46	0	20.0
Artificial cover (%)	61	2.94	4.65	0	20.0

receiving stream was not significantly correlated with species richness, the interaction of location and discharge on richness was highly significant ($p = 0.0006$). Interaction of discharge and location indicates that large springs flowing into large streams tend to contain more species of fish.

Comparison of habitat character in springs containing trout with those with no trout (Table 1.6) revealed significant differences in only three habitat variables: spring discharge, percent riffle habitat, and total cover. Woody cover did not differ between trout and nontrout sites, although this cover type was rarely abundant in springs.

Habitat variables related to presence/absence of trophic guilds are shown in Table 1.7. Variables describing spring size (discharge, length, width, depth) or location (elevation, emergence location) distinguished presence or absence of all trophic guilds in springs. Meso- (% pool) and microhabitat (substrate score, cover types) variables similarly distinguished springs for all guilds except benthic omnivores.

Complete-linkage clustering of spring habitat variables (Figure 1.12) produced five habitat groups. Gorge Spring (#10) remained as an extreme outlier. Gorge Spring was altered by nearby road construction and flowed from a culvert opening over riprap directly into the Cowpasture River. Its spring run was unique in the total absence of pool habitat, an exeptionally short and broad channel, lower water temperature, and the largest mean substrate size of any spring in the survey. Four mottled sculpin were collected from Gorge Spring.

Group I springs had been altered by human activity and were characterized by abundant pool habitat, high substrate diversity, and high

Table 1.6. Comparison of habitat character in springs containing trout with those containing no trout, 1990 study sites. Significance levels are based on Wilcoxon Rank Sum test.

Variable	Mean		P-value
	Trout (N)	No Trout (N)	
Discharge ($m^3/s \times 10^{-2}$)	11.89 (17)	6.40 (43)	0.0033**
Elevation (m)	551 (17)	492 (48)	NS
Emergence location (km)	15.0 (17)	30.2 (48)	NS
Length (m)	602.9 (17)	924.7 (48)	NS
Mean width (m)	3.76 (17)	3.57 (44)	NS
Mean depth (m)	0.16 (17)	0.15 (44)	NS
Mean temperature (^oC)	12.3 (16)	13.3 (47)	NS
Mean substrate score	9.4 (17)	9.1 (44)	NS
Mean substrate diversity	2.88 (17)	2.53 (44)	NS
Pool habitat (%)	42.4 (17)	44.7 (44)	NS
Riffle habitat (%)	35.5 (17)	27.1 (44)	0.0428*
Shading (%)	33.2 (17)	39.0 (44)	NS
Boulder cover (%)	14.0 (17)	18.3 (44)	NS
Log cover (%)	7.0 (17)	5.0 (44)	NS
Rootwad cover (%)	1.0 (17)	1.4 (44)	NS
Aquatic vegetation (%)	20.6 (17)	24.8 (44)	NS
Bank vegetation (%)	5.3 (17)	8.2 (44)	NS
Bank overhang (%)	5.6 (17)	5.8 (44)	NS
Artificial cover (%)	2.3 (17)	3.2 (44)	NS
Total cover (%)	56.0 (17)	66.7 (44)	0.0111*

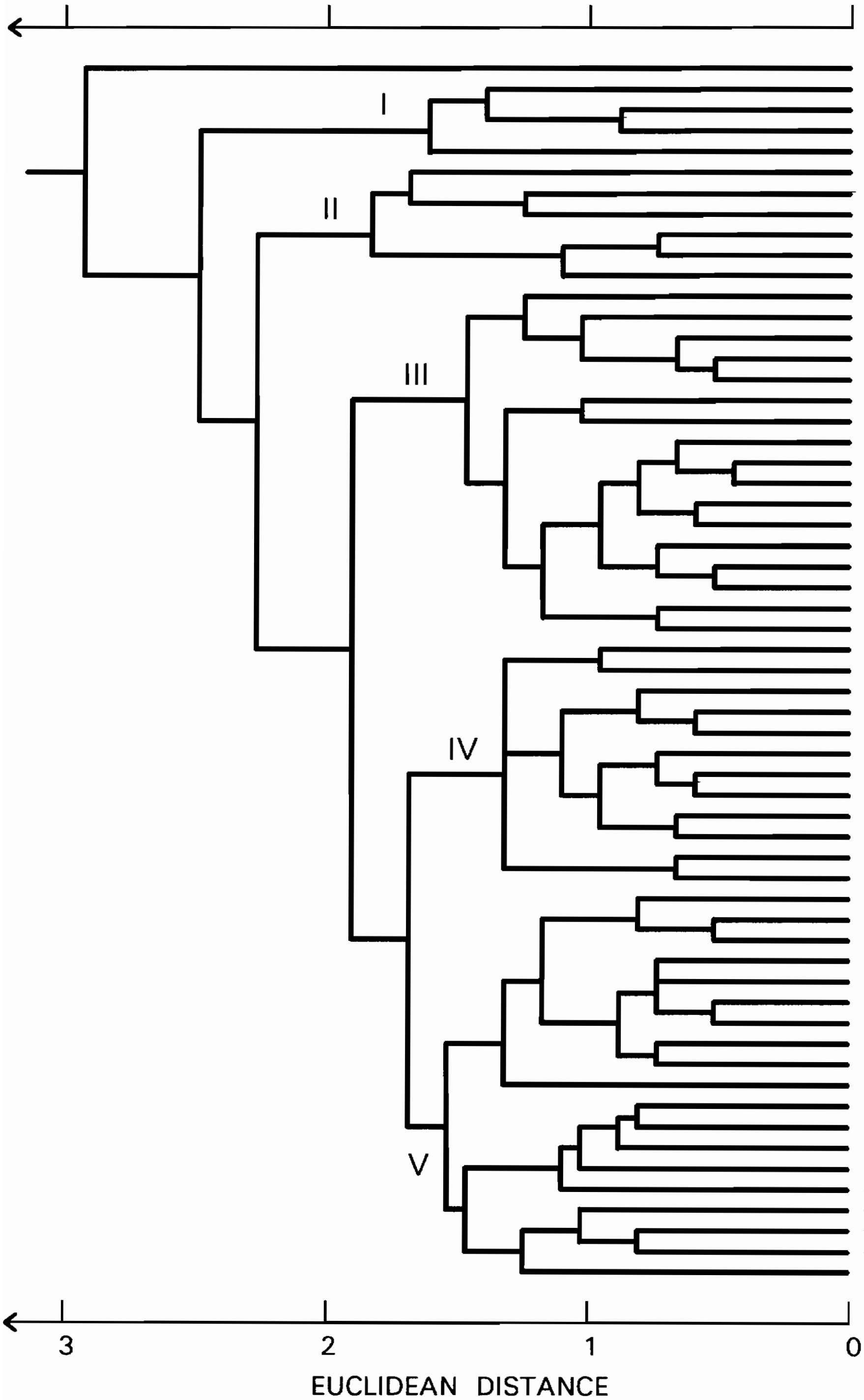
Table 1.7. Mean values of habitat variables related to presence/absence of key trophic guilds in 1990 study sites. Significance levels are based on Wilcoxon Rank Sum test.

TROPHIC GUILDS (Benthic)

VARIABLE	Invertivore			Omnivore			Herbivore		
	Present (N=46)	Absent (N=13)	P-value	Present (N=8)	Absent (N=51)	P-value	Present (N=12)	Absent (N=47)	P-value
Discharge ($m^3/s \times 10^{-2}$)	8.91	4.10	0.0424*	13.47	6.97	0.0451*	10.02	7.30	NS
Elevation (m)	523	550	NS	422	545	NS	461	546	NS
Emergence location (km)	31.0	14.5	0.0308*	38.9	25.5	NS	48.9	21.9	0.0158*
Length (m)	869	726	NS	1031	807	NS	424	943	NS
Mean width (m)	3.7	3.3	NS	4.6	3.5	NS	4.4	3.4	NS
Mean depth (m)	0.16	0.14	NS	0.23	0.15	0.0462*	0.19	0.15	NS
Mean substrate score	7.9	8.2	NS	8.9	7.8	NS	8.6	7.8	NS
Pool habitat (%)	44.4	43.2	NS	53.1	42.7	NS	49.9	42.6	NS
Boulder cover (%)	18.1	14.4	NS	22.2	16.5	NS	20.8	16.4	NS
Rootwad cover (%)	1.6	0.2	NS	1.6	1.3	NS	2.4	1.0	0.0214*
Aquatic vegetation (%)	20.2	36.1	0.0403*	18.1	24.6	NS	19.9	24.7	NS
Bank vegetation (%)	6.9	9.7	NS	3.8	8.1	NS	4.1	8.3	NS
All vegetation (%)	27.0	45.8	0.0167*	21.9	32.6	NS	24.0	33.0	NS
Artificial cover (%)	3.3	1.8	NS	2.5	3.1	NS	2.4	3.1	NS
Total cover (%)	62.2	70.4	NS	61.6	64.4	NS	63.5	64.2	NS

(Non-Benthic)

VARIABLE	Piscivore/Invertivore			Invertivore			Omnivore		
	Present (N=24)	Absent (N=35)	P-value	Present (N=36)	Absent (N=23)	P-value	Present (N=17)	Absent (N=42)	P-value
Discharge ($m^3/s \times 10^{-2}$)	10.89	5.77	0.0113*	8.91	6.20	NS	9.72	7.10	NS
Elevation (m)	520	535	NS	490	589	0.0361*	368	594	0.0001**
Emergence location (km)	27.3	27.4	NS	33.3	18.0	NS	42.4	21.3	0.0225*
Length (m)	593	1005	NS	1190	286	0.0062**	1225	680	NS
Mean width (m)	4.1	3.3	0.0478*	3.9	3.2	NS	4.3	3.3	NS
Mean depth (m)	0.17	0.15	NS	0.16	0.14	NS	0.19	0.14	0.0311*
Mean substrate score	8.8	7.4	NS	8.6	7.0	0.0356*	8.1	7.9	NS
Pool habitat (%)	45.2	43.4	NS	47.5	38.7	NS	55.4	39.5	0.0128*
Boulder cover (%)	16.6	17.8	NS	19.5	13.8	0.0291*	19.9	16.2	NS
Rootwad cover (%)	1.2	1.4	NS	0.9	2.0	NS	0.9	1.5	NS
Aquatic vegetation (%)	20.3	26.0	NS	25.4	17.5	NS	24.7	23.3	NS
Bank vegetation (%)	4.5	9.5	0.0038**	5.9	9.9	0.0216*	4.8	8.6	0.0445*
All vegetation (%)	24.8	35.5	0.0467*	28.5	35.3	NS	29.5	31.8	NS
Artificial cover (%)	1.9	3.8	NS	2.3	4.1	NS	0.9	3.8	0.0168*
Total cover (%)	57.5	68.5	0.0184*	64.2	63.9	NS	65.0	63.7	NS



SPR	BAS	SPP	TG	TR
10	JAM	1	1	N
15	HOL	2	2	N
25	NEW	4	4	Y
22	NEW	1	1	N
8	NEW	1	1	N
16	POW	12	6	N
43	SHE	3	2	N
57	NEW	2	2	N
44	SHE	3	3	N
64	POT	4	3	N
55	JAM	4	4	Y
2	JAM	3	3	Y
63	SHE	5	4	N
52	JAM	2	2	Y
32	HOL	2	2	Y
54	NEW	3	3	Y
41	POW	7	5	N
40	POW	4	4	Y
24	HOL	2	2	Y
45	SHE	8	5	N
39	POW	2	2	N
31	JAM	3	2	N
33	JAM	2	2	N
46	SHE	1	1	N
12	JAM	3	3	Y
17	NEW	7	5	N
50	CLI	8	6	N
37	POW	7	5	N
51	CLI	5	3	N
14	NEW	1	1	N
20	HOL	1	1	N
35	NEW	5	2	N
3	NEW	4	3	N
29	NEW	1	1	N
38	POW	2	2	N
27	ROA	2	2	Y
18	ROA	2	1	N
7	NEW	2	2	N
9	JAM	3	2	Y
48	SHE	10	5	Y
4	NEW	1	1	N
5	CLI	2	2	N
1	NEW	3	2	Y
11	JAM	2	2	Y
58	NEW	1	1	N
56	JAM	8	5	Y
21	HOL	1	1	N
23	HOL	2	2	Y
36	HOL	2	2	N
6	ROA	0	0	N
47	SHE	2	2	N
13	SHE	1	1	N
53	JAM	1	1	N
26	ROA	1	1	N
49	POW	4	4	N
28	HOL	2	1	N
30	NEW	1	1	N
34	HOL	2	2	N
19	HOL	0	0	N

Figure 1.12. Dendrogram of similarity (complete linkage method) of physical habitat in 59 Virginia Springs, 1990 survey: SPR = Spring number (Table 1.1), BAS = river basin, SPP = species richness, TG = number of trophic guilds, TR = trout present (Y/N).

incidence of artificial cover. One rainbow trout was collected from one group I spring (Hardwick, #25), and species and trophic guild richness was relatively low.

Group II springs consisted of large low-elevation springs with extensive spring runs and abundant pool habitat. All of these springs were impacted by cattle grazing and residential development in their watersheds. Only Karnes Spring (#55) contained trout. Species and guild richness were not reduced in Group II springs.

Group III springs exhibited high quality coldwater fish habitat, characterized by high flow, relatively abundant woody and rocky cover, coarse and diverse substrate, frequent pools, and abundant stream shading. Aquatic vegetation was less abundant in Group III springs than in other springs. Frequency of trout occurrence in Group III springs was twice the average for all springs surveyed. Species and guild richness were also relatively high.

Groups IV and V represented roughly half of the springs and were less distinct in character. Group IV springs tended to be rocky, riffle-dominated tributary streams at various locations in their respective watersheds. Group V springs represented a variety of pool-dominated systems with abundant vegetative cover and fine substrate. While trout frequency in these groups was comparable to other springs, there was little diversification of trophic guilds. Species richness was also reduced in group V springs, and springs containing no fish (#6 and #19) appeared as minor outliers within group V.

Principal components analysis ordinated springs based on habitat variables, and canonical discriminant analysis examined multivariate differences between trout and nontrout springs. Table 1.8 shows factor loadings of habitat variables on the first three principal components and canonical coefficients from discriminant analysis. Variables describing spring size (discharge, width, depth), location (elevation, emergence location), and abundance of pool habitat loaded heavily on the first principal component and accounted for 16.7 percent of total variance. Loadings of pool and riffle descriptors (depth, substrate particle size, percent riffle habitat, boulder cover) defined the second principal component axis and defined 15.5 percent of total variance. Riparian variables (shading, woody cover) and channel width loaded heavily on the third principal component and accounted for 11.4 percent of variance in the analysis. Discriminant variables associated with trout habitat included high spring discharge, gravel and sand substrate, abundant riffle habitat, cool temperatures, and relatively shallow channels. Trout springs also exhibited more woody cover and less boulder and artificial cover, although these distinctions were not significant in univariate comparisons.

Scatter plots of species richness on the first three principle components (Figure 1.13) showed no strong influence of multivariate habitat character on the number of fish species found in springs. In contrast, similar plots of trout occurrence (Figure 1.14) showed strongly asymmetrical grouping of trout sites, primarily in the plot on the first and second axes. No trout occurrences appeared in the upper left quadrant of this plot which represents small, high elevation springs with little or no

Table 1.8. Loadings of habitat variables on first three principal components and discriminant analysis, 1990 springs survey.

Variable	Principal Component I	Principal Component II	Principal Component III	Standardized Canonical Coefficient
Discharge	0.566	0.087	-0.378	-0.832
Elevation	-0.512	0.176	-0.042	-0.463
Emergence location	0.566	-0.017	-0.168	0.085
Length	0.384	0.166	-0.365	0.147
Mean width	0.691	-0.091	-0.523	-0.155
Mean depth	0.586	0.509	-0.211	0.510
Mean temperature	0.050	0.127	0.138	0.504
Mean substrate score	0.295	-0.748	-0.055	0.601
Mean substrate diversity	0.197	0.447	0.168	-0.225
Percent pool habitat	0.547	0.496	0.137	-0.308
Percent riffle habitat	-0.179	-0.728	-0.304	-0.548
Percent shading	0.500	-0.328	0.568	0.103
Percent boulder cover	0.345	-0.676	0.008	0.437
Percent log cover	0.406	-0.109	0.657	-0.450
Percent rootwad cover	0.175	0.203	0.455	0.112
Percent aquatic vegetation	-0.403	0.438	-0.449	0.361
Percent bank vegetation	-0.331	-0.217	0.071	0.129
Percent bank overhang	-0.060	0.013	0.383	-0.048
Percent artificial cover	-0.029	0.444	0.259	0.406

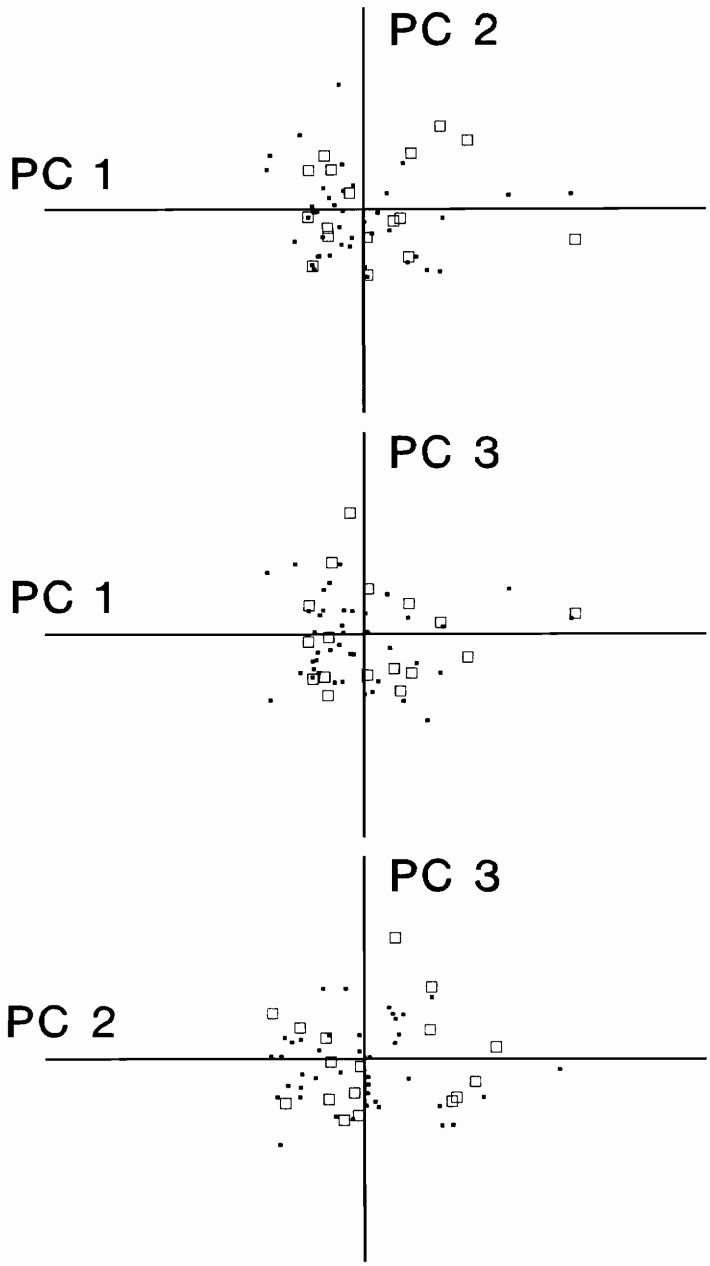


Figure 1.13. Scatter plots of first three principal components (factor analysis on 1990 spring habitat variables) related to fish species richness: solid points represent springs containing zero to three species; open squares represent springs containing four or more species.

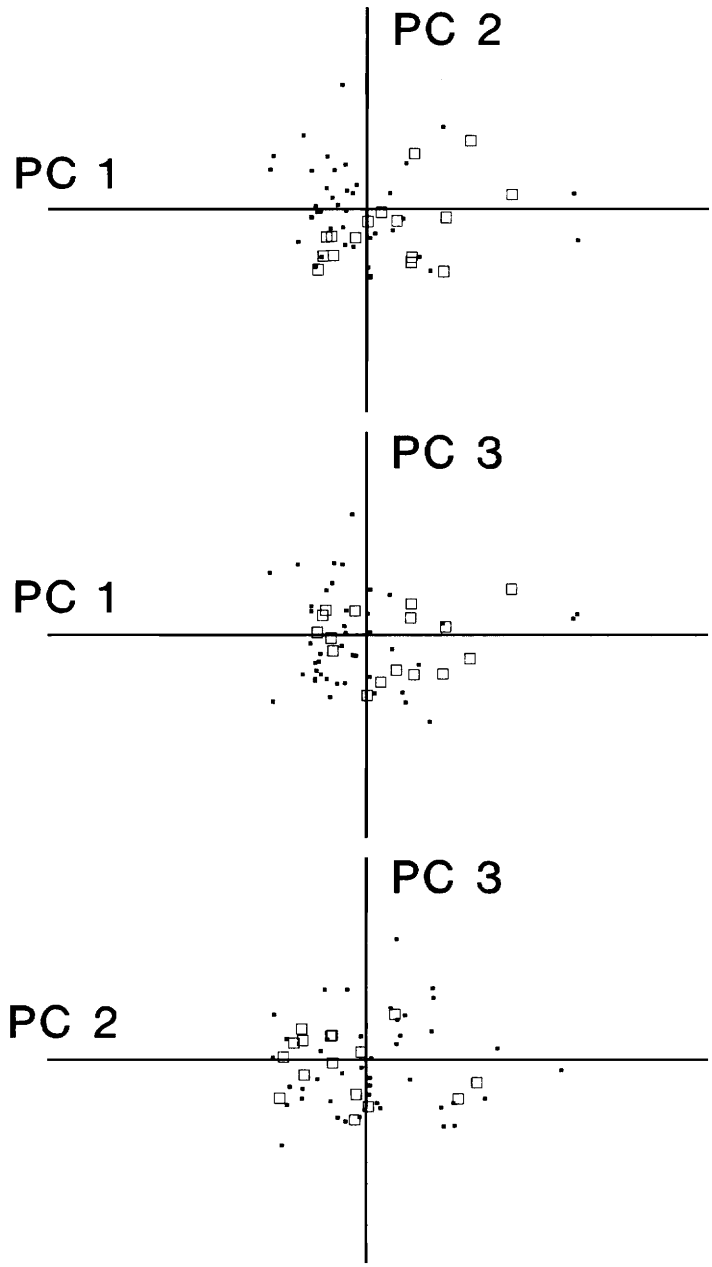


Figure 1.14. Scatter plots of first three principal components (factor analysis on 1990 spring habitat variables) related to occurrence of trout: solid points represent springs containing no trout; open squares represent springs containing trout.

rocky riffle habitat. Springs containing trout were symmetrically distributed on the third principal component axis.

Occurrence of sculpin species (Figure 1.15) and blacknose dace (Figure 1.16) showed no strong tendencies along principal component axes.

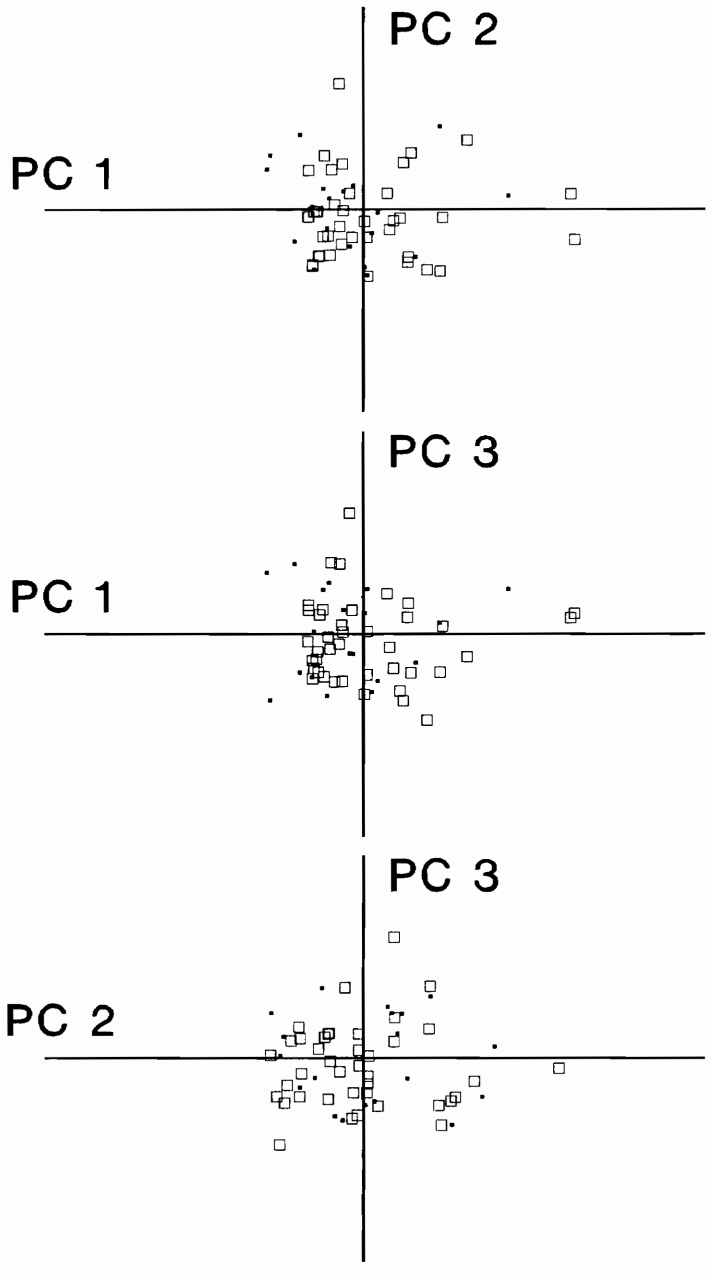


Figure 1.15. Scatter plots of first three principal components (factor analysis on 1990 spring habitat variables) related to occurrence of sculpin: solid points represent springs containing no sculpin; open squares represent springs containing at least one species of sculpin.

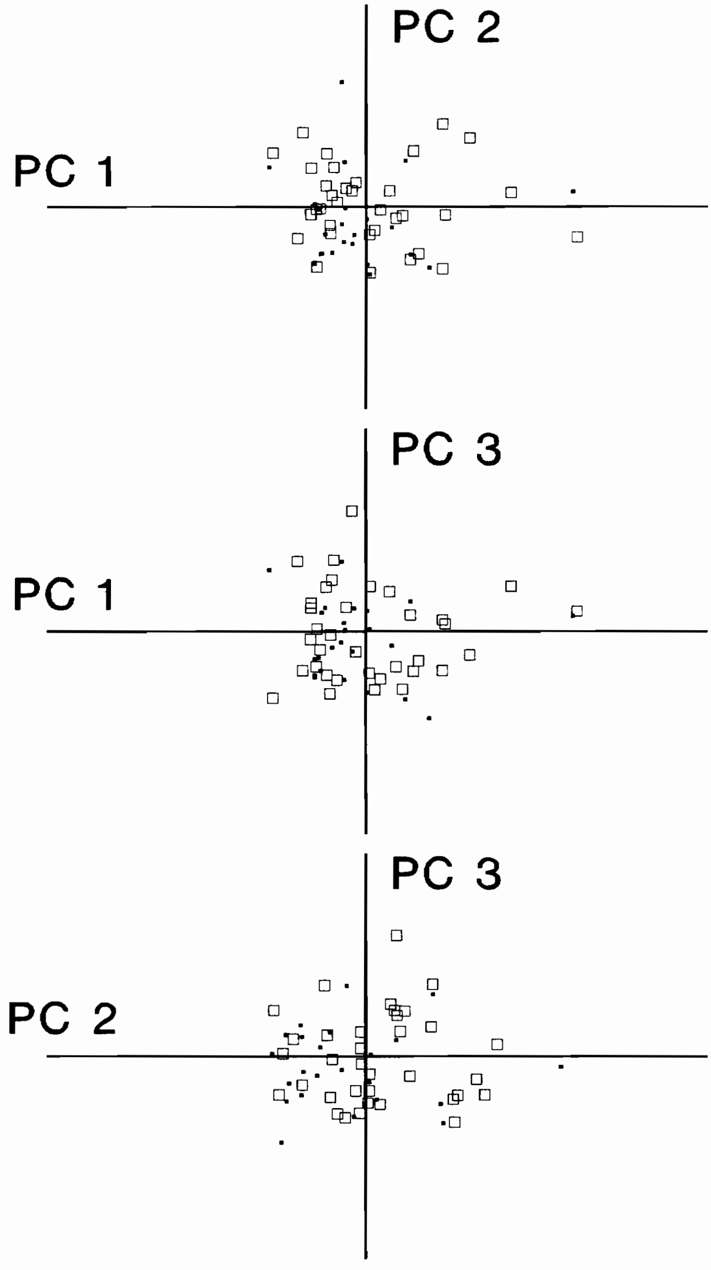


Figure 1.16. Scatter plots of first three principal components (factor analysis on 1990 spring habitat variables) related to occurrence of blacknose dace: solid points represent springs containing no blacknose dace; open squares represent springs containing blacknose dace.

DISCUSSION

SPRING FISH COMMUNITIES

Virginia springs typically supported fish communities; only two of 65 springs contained no fish. Other faunal descriptions of springs in the eastern United States (Lewis 1957, Minckley 1963, Armstrong and Williams 1971, Pflieger 1982, Sizemore and Howell 1990) have reported fish populations at most sites. In contrast, Matthews et al. (1985) found fishes in only 19 of 50 Oklahoma springs. In the present study, sites were selected for high discharge, producing a sample set of relatively large springs specifically to survey fish populations. All sample sites were located within the Ridge and Valley physiographic province, where high-discharge Virginia springs occur. Because of this selective design, the proportion of sample sites yielding fish was higher than would be expected from a random sample of Virginia springs. Other studies of springs also focused on large spring systems (Lewis 1957, Armstrong and Williams 1971, Sizemore and Howell 1990), whereas the work of Matthews et al. (1985) encompassed several physiographic regions and included small springs.

Sculpins were collected in all river basins except the Potomac, which was represented in the survey by only two springs. Absence of any sculpin species in Potomac Basin springs was likely an artifact of the small sample segment length, because habitat character and migration opportunties did not differ from other basins where sculpins were common. Sculpin species distribution in springs reflected ranges of each species reported for Virginia (Jenkins and Burkhead 1994).

Blacknose dace, the most common fish in spring samples, occurred in all major river basins and was represented in 60 percent of the springs. Jenkins and Burkhead (1994) reported relative rarity of this species in the Piedmont region of the Roanoke, but noted its presence in a spring run. The absence of the species in four of five Roanoke Basin spring samples may be an artifact of the small sample size from this basin or urbanization impacts on many larger springs.

Central stoneroller was collected in deforested springs, where organic enrichment promoted extensive algal growth, and in adventitious springs where nutrient-rich waters were easily accessible. Many stoneroller occurrences, particularly in the Clinch and Powell basins, were associated with springs that emerged from large caves, where surface water influence from agricultural areas likely elevated nutrient levels and primary productivity.

Unlike other common spring taxa, darter representation did not follow expected distributional trends. Poor representation of darters in springs of the Powell, Clinch and Holston basins was surprising considering the degree of speciation of darters in the Tennessee River Valley. Fantail darter, widely distibuted throughout the study area, was common only in Atlantic Slope basins and was the only darter encountered that is typical of cold headwaters. The three other darters collected were found at low elevations near large streams and were incidental to spring systems.

The low fish densities in Virginia springs were typical of small headwater streams. The wide range of observed fish densities sampled in Virginia springs indicated high variability in primary productivity. Watershed use influenced Virginia spring nutrient budgets, particularly in agricultural areas. The majority of pasture springs were exposed to direct sunlight, enriched with high organic loading, and contained abundant periphyton or macrophytes. Deforestation of spring watersheds increased insolation of spring channels which increased algal and macrophyte primary production and in turn supported higher fish densities. Other investigators have reported both lower (Malmqvist and Bronmark 1984) and higher (Teal 1957, Tilly 1968) productivity in spring streams compared to other communities.

Low species richness found in the present study has been reported widely in spring systems. Davidson and Wilding (1943) reported only one species of sculpin from a southwestern Washington spring. Armstrong and Williams (1971) found an average of five fish species in a survey of 68 Tennessee and Alabama springs. In Oklahoma, Matthews et al. (1985) collected an average of 3.4 species from those springs that contained fish. Sizemore and Howell (1990) reported 5.3 species per site in 83 Alabama springs.

Trends toward higher species and trophic guild richness in the Powell and Clinch drainages in the Southwest and the Shenandoah to the Northeast may have occurred because many sample sites in these basins tended to contact higher-order streams. The particularly high species richness in Tennessee River tributaries would be expected given the exceptional richness of fish fauna in this region (Williams and Neves 1992, Warren and Burr 1994).

Low richness values found in tributaries of the Holston and Potomac rivers did not conform to expected trends. Use of springs for residential water supply or aquaculture, common in all river basins, made many large spring sites in the Holston and Potomac drainages unavailable for fish survey due to capping or impoundment. As a result, all Holston River sites were in headwaters (0-5300 m from the top of the watershed). The Potomac basin was represented by only two sites in the Opequon Creek watershed, one of which was heavily impacted by urban development and yielded a single fish.

Low trophic guild richness reflected low species richness in 1990 spring samples. Most springs contained benthic and non-benthic feeders. Similar trophic guild structure was found in spring fish communities of Oklahoma (Matthews et al. 1985) and the lower Tennessee River basin (Armstrong and Williams 1971).

Endemic species were not found in Virginia springs. Slimy sculpin and pearl dace are at the southern extent of their ranges (Page and Burr 1991) and are restricted to spring runs and other groundwater-cooled systems in Virginia (Jenkins and Burkhead 1994).

HABITAT INFLUENCES ON FISH COMMUNITY STRUCTURE

In addition to providing a relatively permanent and stable source of high quality water to many headwater streams, large Virginia springs provide considerable stream habitat. Flow levels, turbidity, chemical water quality, and temperature are more stable in such waters than in streams of comparable size that are fed by surface runoff, and such stability may inhibit community diversity (Vannote et al. 1980). Physicochemical characteristics of springs influence adjacent waters and affect fish distribution by buffering extreme thermal conditions in winter and late summer.

Typical spring habitat was characterized by shallow pools interspersed with rocky riffle areas. Groundwater influence kept temperatures low despite sparse shading at many sites. Cover consisted primarily of small boulders and aquatic vegetation, and woody cover was usually scarce. Nearly all aquatic vegetative cover consisted of watercress (Nasturtium officinale). Watercress was introduced to North America from Europe as a food crop (Tull 1987) and is now virtually ubiquitous in Virginia springs, often filling the entire stream channel. Watercress introduction probably altered habitat character at many sites, although its effects on spring fish communities are unknown. Abiotic factors (shading, gradient, water depth and velocity) affecting fish distribution may also affect and be affected by distribution of watercress.

Clustering of spring habitat groups reflected anthropogenic influence on spring systems. Five springs (Group I) that were culverted, boxed, or channelized represented distinct groups or outliers in the analysis and had low species richness. Three of these springs contained only sculpin and a fourth site contained blacknose dace and sculpin. Low species richness in these springs and the appearance of omnivorous species in the fifth site are comparable to fish community alterations reported in modified stream channels (Karr and Dudley 1981, Schlosser 1982, Simpson et al. 1982, Brookes 1988).

Springs in relatively undisturbed riparian areas also clustered differently. The vegetated streambanks and unmodified stream channels found in these springs provided cover and invertebrate food organisms for their fish communities. The higher occurrence of trout in springs with undisturbed streambanks and channels is consistent with riparian influences on trout reported in other headwater systems (Boussu 1954, Chapman and Knudsen 1980, Wesche et al. 1987).

While trends in species and guild richness and trout frequency were evident among habitat groups, rigorous testing of these trends was not practicable. Cluster analysis did not indicate interbasin habitat differences. Ordination of spring habitats was relatively indistinct, with the first three principal components accounting for 43.56 percent of total variance. Principal components analysis reflected results of Wilcoxon tests, indicating that spring size and location variables accounted for more variability in springs than did instream or riparian habitat variables.

Presence of fish trophic guilds in springs was significantly associated with habitat variables. Springs that supported benthic invertivores (sculpins and darters) were larger and were associated with higher-order receiving streams. Springs that did not contain benthic invertivores tended to be filled with aquatic vegetation. Piscivores (trout and sunfish) occurred primarily in large springs with diverse habitat and cover. Benthic omnivores (carp and suckers), although rare in springs, were found in springs with greater flow and channel depth. Central stoneroller, the only benthic herbivore collected, occurred primarily in adventitious systems. Stonerollers readily colonized springs that were productive enough to support complex fish assemblages and probably did not compete directly with other species for food, as all springs with stonerollers contained other fish species.

Oligotrophic conditions and geographic isolation from source populations probably prevent wider distribution of this species in Virginia springs.

GEOMORPHOLOGY AND SPRING FISH DISTRIBUTION

Geomorphologic metrics (spring size and location) accounted for more variability in spring habitat than micro- or mesohabitat variables in factor analysis, loading heavily on the first principal component axis. These variables were important determinants of fish community structure.

Spring size and location of the spring confluence within the watershed of the receiving stream (elevation, emergence location) strongly influenced fish species richness. Large springs contained more species, particularly when located at low elevations in the watershed, where spring runs were directly confluent with higher-order streams. In such adventitious situations, spring systems are readily available to a larger "species pool" in the adjacent receiving stream (Figure 1.17). In addition to typical spring fauna, other species may enter the system seeking moderate thermal conditions, cover, spawning areas, or feeding habitat. In smaller springs, habitat restrictions limit such immigration to a subset of local species or life stages of species. Occurrences of riverine species in lower reaches of spring runs may be incidental to their daily movements. Approximately one-third of the species collected may be temporary inhabitants of spring runs, although the lack of multiple site visits prevented examination of seasonal community changes in this study. Of 39 species identified in 1990 spring collections, thirteen occurred at only one site each. Eighty-five percent of these incidental spring species are reported primarily from higher-order

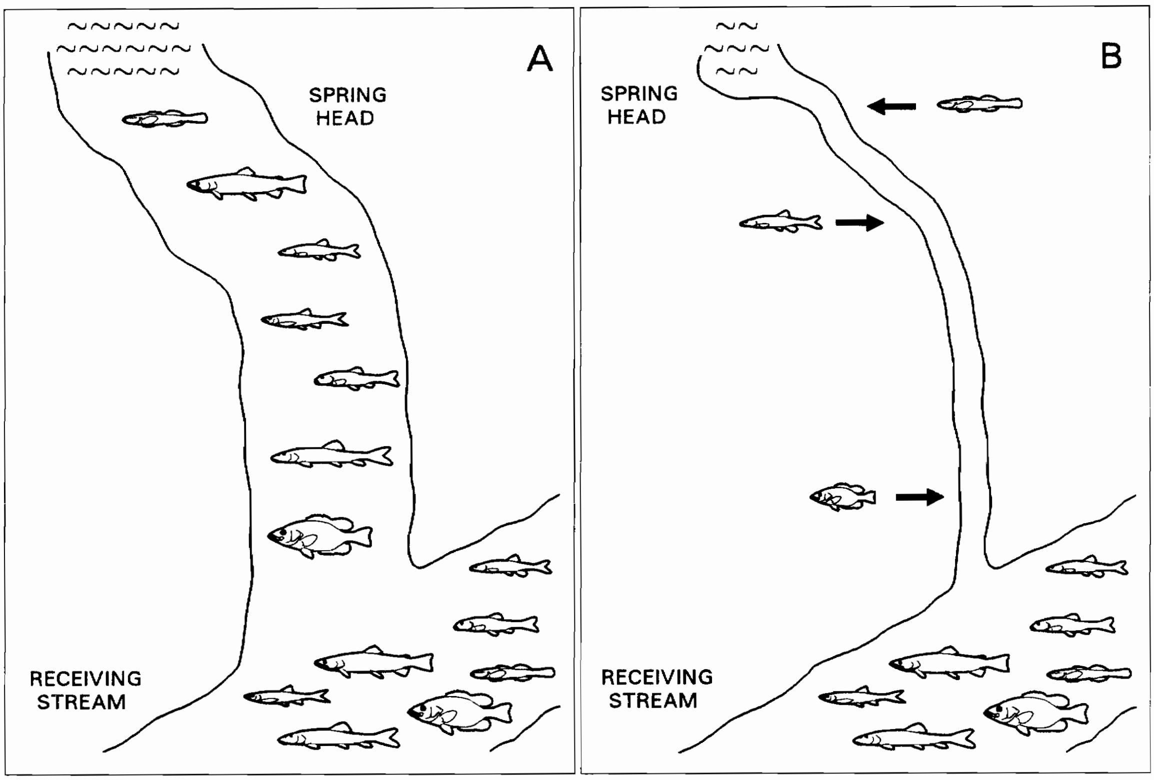


Figure 1.17. Influences on species richness in adventitious springs.
(A) habitat is available for more species in large springs.
(B) small springs provide limited habitat, supporting a subset of species and life stages from the receiving stream faunal community.

streams (Jenkins and Burkhead 1994).

The influence of tributary location on fish species richness and variability of species representation has been reported from a variety of lotic systems (Gorman 1986, Bass 1990, Osborne and Wiley 1992). The interaction of spring size (flow) and emergence location was the strongest predictor of species richness in the present study. Osborne and Wiley (1992) used "downstream link", the sum of the orders of the two streams at the next confluence below a given sample site, to explain the majority of variance in species richness in Illinois waters. Downstream link is analogous to the emergence location used in the present study.

Range restrictions affected the composition of spring fish communities. Thirteen of 26 fish species occurring in multiple springs only occurred in a portion of the study area. River basin differences strongly influenced clustering of springs primarily because of sculpin speciation in Virginia.

Elevation of springs was linked to their location in the receiving stream watershed because of the physiography of the study area. Lower-elevation springs contacted higher-order streams, and higher species richness found at lower elevations likely reflected the emergence location or D-link effect, rather than any direct dependence of species richness on elevational influences (water temperature, barometric pressure, ultraviolet light intensity, etc.).

In headwaters, the effect of spring size on species richness is likely a function of instream flow and habitat availability. The influence of flow on stream habitat and fish communities has been demonstrated from a wide range of lotic systems (Gorman and Karr 1978, Schlosser 1982, Leonard and Orth 1988). Larger springs increase base flows of receiving headwaters, facilitating upstream range extension of fish species (Figure 1.18), whereas smaller springs benefit receiving waters by buffering flow and temperature variability, and species richness within the spring run itself is limited by available habitat.

TROUT IN SPRINGS

Trout occur or have been introduced into virtually every headwater system in the study area. All springs containing trout were contiguous with coldwater systems, and their distribution reflected known range of trout in Virginia waters (Jenkins and Burkhead 1994). Landowners frequently stock and feed trout in their springs. Surprisingly, trout populations have not persisted in the majority of springs. Habitat features linked to trout occurrence included spring discharge, percent riffle habitat, and total cover. Spring flow, size, and riffle variables also were prominent in discriminant analysis of trout presence or absence in springs. Trout may be excluded from a significant proportion of available spring runs by the combined effects of these variables, because small, high-elevation, cover-rich springs rarely contained trout.

MANAGEMENT IMPLICATIONS AND LANDOWNER RECOMMENDATIONS

The springs of Virginia should be protected for their unique characteristics and importance to coldwater aquatic communities. Because

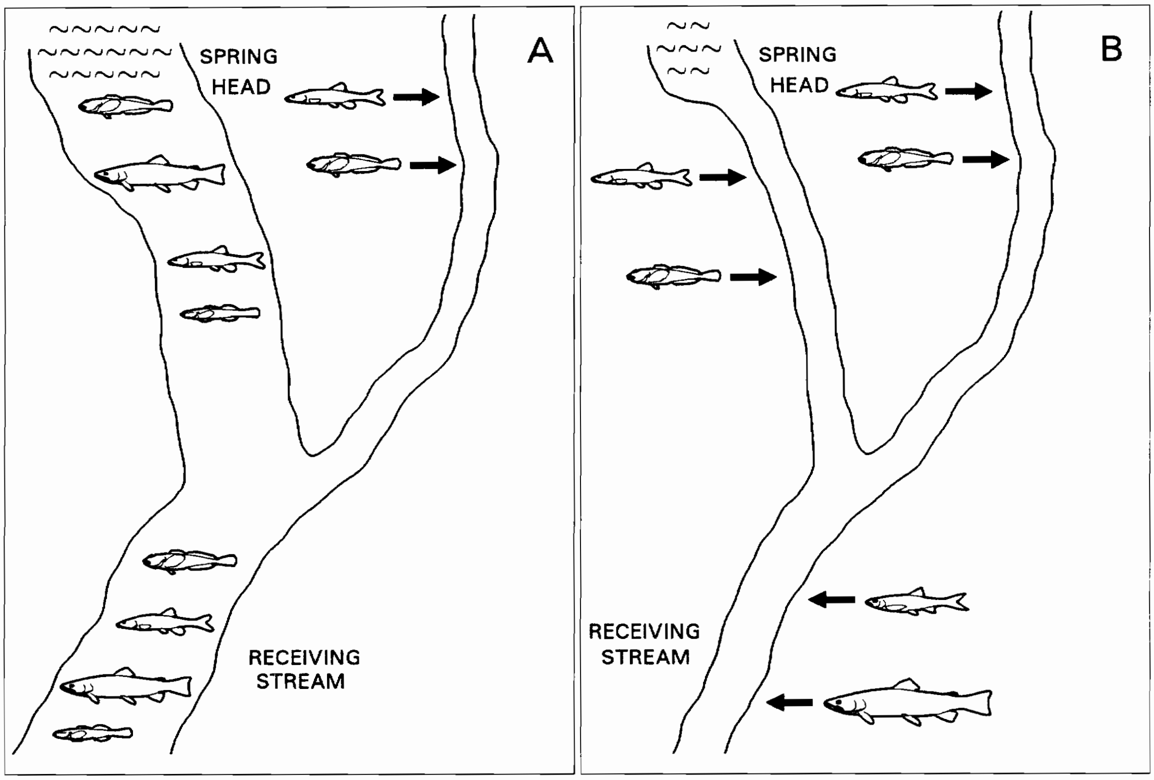


Figure 1.18. Influences on species richness in headwater springs.
(A) coldwater fish habitat is abundant in large springs.
(B) small springs provide limited fish habitat but may improve coldwater fish habitat in receiving waters.

most springs occur on private lands, the responsibility for their stewardship lies with landowners, and any efforts by managing agencies to protect or enhance spring fisheries must involve cooperation with spring owners. Identifying and correcting threats to water quality should be a priority for all springs, and fishery potential of large springs may be improved through habitat restoration and enhancement.

Threats to water quality in spring streams may originate from impacts to groundwater or mismanagement of riparian areas. Chemical or biological contamination of groundwater is widespread in Virginia and comes from many sources, including pesticides, household and industrial wastes, petroleum leaks, landfills, mines, and animal wastes (Weigmann and Kroehler 1988). The Karst geology of the Ridge and Valley region, where most large springs occur, is also particularly vulnerable to groundwater contamination from surface water sources (Swain et al. 1991). In addition to direct effects on water quality, contaminants may foster harmful organisms that threaten human health. Nutrients from fertilizers, sewage, or animal waste increase the likelihood of spring contamination from harmful bacteria, protozoans, or algae, with water quality effects ranging from unpleasant odor or taste to potentially fatal illnesses.

Poor riparian management typically involves removal of streambank vegetation, either by direct mechanical means or by unrestricted livestock access. In extreme cases, stream channels are mechanically excavated or severely eroded from animal use. Riparian vegetation was an important habitat component for gamefish in this study and has been linked to trout abundance by Wesche et al. (1987). Vegetation removal decreases cover and invertebrate food production and leads to structural failure of stream banks. Resulting soil erosion and sedimentation impacts trout spawning areas and reduces aquatic insect production, further limiting fishery potential of springs. Cumulative fisheries impacts of riparian abuse have been widely reported (Neves and Angermeier 1990, Li et al. 1994).

While groundwater and riparian impacts are widespread, they typically result from localized impacts and may be remedied by the individual landowner. Groundwater contamination events usually involve areas of less than one square mile (Canter et al. 1987), and dramatic improvements in riparian character may be achieved through restoration efforts on private lands. Governmental assistance for such activities may involve extension and technical assistance, cost sharing, demonstration projects, or other incentives. Landowner actions that will reduce threats to spring resources include restoration of riparian vegetation, fencing of livestock from springs, inspection and repair of septic systems, responsible use of fertilizers and pesticides, and proper disposal of home and farm chemical wastes.

The importance of geomorphologic influences on spring fish community structure has serious implications for study of spring streams. The amount of habitat available (spring flow and channel size) and accessibility of spring streams from downstream waters may have much greater influence on composition of fish communities than the quality of habitat, limiting the utility of fish metrics in assessing degradation or tracking restoration efforts. Effects of stream order on fish community structure has been demonstrated from a variety of lotic systems (Penczak and Mann 1990, Ross et al. 1990), and Platts (1979) and Lanka et al.

(1987) linked geomorphologic variables to fish species richness and trout standing stock.

Because fish communities of small adventitious tributaries may be strongly influenced by adjacent receiving waters (Gorman 1986, Osborne and Wiley 1992), observed changes in community structure may not be attributable to habitat alterations. Studies that target fish populations of springs should consider spring location effects when selecting sample sites, because fish communities of geomorphologically dissimilar springs may respond differently to experimental manipulations. Osborne et al. (1992) demonstrated that influence of receiving waters significantly affected results of biological monitoring in small streams and advised development of different expected metric values for headwater streams. Matthews et al. (1984) proposed that a biological monitoring plan for Oklahoma springs should combine metrics based on common spring fishes with invertebrate metrics. Such a design would take advantage of the relatively low mobility of aquatic invertebrates and reduce the sensitivity of assessment metrics to incidental collections of fish species from downstream waters.

The present study suggests that Virginia spring fish populations also may be influenced by access to downstream waters, and that less mobile organisms, particularly benthic invertebrates, should be used to supplement fish metrics in biological assessment. The second chapter of this thesis will examine responses of fish and invertebrate populations of springs to cattle-related habitat degradation.

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CHAPTER II

Effects of Riparian Cattle Access on Spring Fauna

ABSTRACT

This study investigated impacts of riparian use by cattle on fish and macroinvertebrate communities and aquatic habitats in spring-fed streams. Ten sites in the New and James river basins of Virginia encompassed a broad range of grazing intensities, from unimpacted to heavily impacted (28.7 head/100m^2 of accessible stream channel). Nitrate levels, substrate size and diversity, embeddedness, streambank integrity, and shading were all significantly altered in heavily impacted streams. Proportional abundance of trout and young-of-year benthic invertivore fishes declined with increased cattle densities. Mayfly, stonefly, and caddisfly taxa decreased in proportional abundance, while invertebrate abundance, proportional abundance of Chironomid larvae, and proportional abundance of non-insect invertebrate taxa increased in impacted springs. Fish communities were influenced primarily by riparian condition, whereas benthic invertebrate communities were strongly associated with benthic habitat condition and nitrate levels. Thermal buffering by groundwater and recolonization from downstream waters likely muted fish community responses to instream habitat impacts. Benthic macroinvertebrate community structure was an effective indicator of cattle-related spring stream degradation, but fish metrics should supplement invertebrate metrics in spring assessment because of their direct relevance to spring management and sensitivity to

riparian condition. Spring restoration efforts should focus on educating landowners and restricting livestock access to spring channels.

INTRODUCTION

LIVESTOCK IMPACTS ON STREAM ECOSYSTEMS

Fish

The adverse effects of cattle grazing on fish populations have been demonstrated repeatedly. Increased erosion and sedimentation may result from shearing and trampling of streambank areas by cattle, thereby reducing fish spawning habitat and diminishing production of benthic invertebrate food organisms. Siltation of stream habitat in Missouri was associated with a decrease in distinction among riffle, run, and pool fish communities, decreased abundance of benthic insectivores and herbivores in riffle areas, and decreased abundance of fish species requiring clean gravel for spawning (Berkman and Rabeni 1987). Crouse et al. (1981) demonstrated impaired salmonid production in response to increased levels of fine sediments. Heavy sedimentation may also affect habitat selection by juvenile salmonids (Hillman et al. 1987). Removal of livestock from pasture areas has been shown to reduce sediment loads in runoff (Smith 1989). Winegar (1977) reported sediment load reductions of up to 79 percent following fencing of a heavily livestock-impacted stream in Oregon.

Effects of grazing on streambank areas include removal of vegetation and physical disturbance of underlying soils. In addition to stabilizing banks, streamside vegetation helps to modify solar influence on stream temperatures (Chamberlin et al. 1991) and provides important allochthonous organic matter to stream systems (Cummins 1974). Cover and bank stabilization is provided by herbaceous vegetation, woody material, and root structures (Kauffman and Krueger 1984). Overhanging bank cover and vegetation have been correlated with several instream habitat variables, including mean channel width and depth, variability in depth, width/depth ratio, and siltation (Hubert et al. 1985). Overhead cover provided by riparian vegetation was more strongly correlated with trout population size than other cover types in Wyoming streams (Wesche et al. 1987). Gunderson (1968) found increased cover, trout abundance, and trout biomass in an ungrazed section of a Montana stream. Platts (1991) summarized 21 studies, most (71 percent) of which demonstrated a decline in fish populations in response to livestock grazing. Boussu (1954) related vegetative cover to the biomass of three trout species in a Montana spring stream where annual temperature variation was similar to that of eastern streams. While summer temperature elevation is less severe in eastern streams than in arid western areas, species intolerant of high temperatures may be excluded from otherwise suitable habitat. Meisner (1990) found that increased air and groundwater temperatures reduced brook trout habitat by as much as 42 percent in southern Ontario streams.

Invertebrates

Cattle grazing affects benthic macroinvertebrate communities by altering the physicochemical habitat and trophic structure of the stream ecosystem. Chemical alteration may occur through nutrient enrichment, oxygen depletion, and introduction of toxicants. Variability in pollution tolerance among benthic organisms produces changes in community structure in response to chemical degradation (Winget 1985). Increased levels of nutrients may alter the trophic structure of the stream community at various levels, affecting detritus processing rates, abundance of algae and macrophytes, and functional composition of fish and macroinvertebrate communities (Murphy and Meehan 1991). Systemic pesticides in livestock wastes have been shown to prevent invertebrate colonization (Coe 1987) and reduce fecal breakdown rates (Wall and Strong 1987).

Invertebrate community structure may also be altered by degradation of physical habitat. Substrate size affected distribution of benthic invertebrates in an Idaho stream (Rabeni and Minshall 1977), and Thorup (1966) found distinct benthic communities inhabiting different substrate types in a Danish spring. Lenat (1984) identified sedimentation from agriculture as a major influence on invertebrate communities in North Carolina streams. Rabeni and Minshall (1977) also identified heavy siltation as a limiting factor on stream invertebrate distribution.

Functional groups of benthic invertebrates (Cummins 1974) respond differently to agricultural impacts. Lenat (1984) found relative increases in filter feeders and algal grazers in degraded streams. Organisms preferring interstitial spaces in coarse substrate, such as stoneflies and many beetles, may be eliminated from highly sedimented streams (Cairns and Dickson 1971), while sediment-dwelling taxa such as midge larvae are less affected (Lenat 1983). Reduction of riparian vegetation may drastically reduce detritus influx to the stream system (Minshall 1967), affecting trophic structure of the community (Cummins 1974).

Livestock wastes may alter the bacterial quality of surface and groundwater in grazed watersheds. Moody (1990) listed livestock production among bacterial water quality threats. Skinner et al. (1974)

reported significant increases in fecal coliform and streptococci in a grazed Wyoming watershed, although bacterial levels were not reliable as water quality indicators.

Physical and Chemical Habitat

The effects of agricultural practices on the physicochemical character of streams are well documented. Intensive cattle grazing may compound water quality impacts with physical alteration of riparian and aquatic habitat. Major chemical impacts include influx of nutrients, primarily nitrogen and phosphorus, and pesticide contamination, affecting both ground and surface water. Physical effects of cattle grazing include elimination of stabilizing vegetation, streambank erosion, sedimentation, reduced stream depth, widening of the stream channel, and loss of cover and shade (Platts 1991).

Threats to groundwater resources have increased substantially in recent years (Weigmann and Kroehler 1988). Nitrate levels are on the increase in agricultural areas worldwide, and pesticides are a growing threat to groundwater resources (Hallberg 1986). Phosphorus, petroleum products, and a variety of toxins have accumulated in groundwater as a result of mining, industrial, and agricultural activities. Groundwater contamination is widespread; about 20 percent of wells nationwide contain high (>3ppm) levels of nitrate-nitrogen, and pesticide contamination is reported in at least 44 states (Moody 1990). While groundwater quality in Virginia is generally good, threats from agricultural sources are common (Powell and Hamilton 1986).

Cattle impact surface water through increased nutrient influx, sedimentation of the stream channel from reduced streambank stability, alteration of watershed hydrology, and destruction of shading vegetation resulting in elevated summer water temperatures. Schepers and Francis (1982) reported increases in total solids, nitrogen, and phosphorus in response to cattle grazing on an experimental pasture in Nebraska. Similar increases in total nitrogen and total phosphorus occurred in runoff from grazed watersheds in North Carolina (Duda and Finan 1983). Smith (1989) found a significant reduction in both nitrogen and phosphorus concentrations in channelized runoff following removal of livestock from hillslope pastures. Soil compaction and destruction of vegetation have substantially increased runoff in the southwestern states, altering the hydrodynamics of stream channels in grazed watersheds (Debano and Schmidt 1989). Comparisons of grazed and rehabilitated riparian plant communities have demonstrated improved streambank stability and elevated water tables in response to vegetational changes in areas recovering from grazing (Platts and Nelson 1985).

CATTLE MANAGEMENT

Public lands, especially those in the western United States, historically have been managed for timber production (Meehan 1991) and livestock grazing (EPA 1990). Livestock use or misuse in the western U. S. has occurred since the mid-1800's, and abuse of riparian areas has increased with the growth of the livestock industry (Platts 1991). Effects of riparian use by cattle have not been well documented in the eastern U. S., but research has demonstrated preference of riparian areas by eastern cattle breeds (Goodman et al. 1989).

Riparian areas offer drinking water and high quality forage to cattle. Beef cattle typically require 12 to 15 gallons of water per day (USDA 1982). Miner et al. (1991) observed less frequent use of riparian zones by cattle when they were provided with an off-stream water source. Goodman et al. (1989) found that use of riparian areas was highest during summer months and during the first season of grazing after resting, when forage quality was higher. Marlow and Pogacnik (1986) also noted a concentrated riparian use by cattle in late summer. The preference of cattle for streamside zones coincides with periods of maximum air and water temperatures, when the adverse impact on stream fisheries is greatest. Armour et al. (1991) identified multiple impacts of cattle on riparian areas and recommended a series of modifications to western range management strategies, including improved inventories of riparian areas, increased use of incentives for proper management and disincentives for riparian abuse, and enhanced public awareness of riparian ecology.

In contrast to western rangeland management, cattle farming in the eastern U. S. occurs primarily on private lands. The cattle farmer, as owner or lessee of the rangeland, has direct control of land management practices. Livestock farming represents a common land use in Virginia, comprising over five million acres and 28,000 individual farms. Annual gross revenue from livestock sales in Virginia exceeds $340 million. Most Virginia cattle farmers operate small-scale (<25 animals), part-time family farms. Of nearly 20,000 beef cattle farms in operation, average farm size is 196 acres

(Virginia Dept. of Agriculture 1990). Small family-owned cattle pastures can be found on many Virginia spring streams. Management of riparian areas on private pasturelands can impact aquatic habitat quality within grazed areas and in downstream waters. Physical and biological effects of degradation in grazed areas may persist for some distance downstream, and the presence of multiple pollution sources in a watershed may produce cumulative effects, further degrading continuous stream systems (Burns 1991).

The Virginia Cooperative Extension Service has traditionally provided land management guidance and literature in the form of Best Management Practices, or BMPs (Virginia SWCB 1979, 1980; Heatwole et al. 1991), to livestock farmers. Virginia BMPs address streambank erosion and pasture management, but offer no specific recommendations for managing cattle use on riparian areas. Compliance with Virginia BMPs is completely voluntary for land owners.

BIOLOGICAL MONITORING OF HABITAT QUALITY

The use of biotic communities as indicators of environmental degradation has increased in recent years, particularly in areas where non-point sources are the primary threat to aquatic systems. Karr and Dudley (1981) noted that traditional methods of physical and chemical water quality monitoring have been unsuccessful in preventing the continuing decline of midwestern aquatic systems. Short-term events may be missed by periodic chemical monitoring, and physical impacts such as habitat degradation and flow alterations may impair biological communities in the absence of measurable changes in water quality (Karr 1981). Communities of relatively long-lived organisms such as fishes or macroinvertebrates tend to recover slowly following perturbation, allowing detection of past pollution events (Cook 1976).

Past researchers commonly used diversity indices (H or H') which describe community complexity by the amount of "information" contained in a sample; the implication is that impacted communities will be less complex than non-impacted communities (Winget 1985). Washington (1984) cites many applications of these diversity indices for describing population heterogeneity, trophic diversity, and ecological community structure, in spite of their questionable biological relevance.

Fish or invertebrate communities may be more effective in bioassessment than instantaneous water quality monitoring, depending on type of impact and method of data analysis. Berkman et al. (1986) compared the utility of fish and invertebrate indices in biotic assessment and found both reflected habitat quality, although different types of analysis were required for the two groups.

Fish

The use of fish in bioassessment typically has involved regional modification of the Index of Biotic Integrity or IBI, which incorporates a suite of metrics, including relative abundance of feeding and habitat guilds, presence of tolerant or intolerant species, presence of exotics, hybridization, and physical abnormalities (Karr 1981). The Ohio EPA (1987) and the USEPA (Plafkin et al. 1989) currently use the IBI for bioassessment. Advantages of fish communities as indicators include ease of identification and field sorting, extensive life history information, long lifespan, functional diversity, and high level of public recognition (Karr 1987). Disadvantages include mobility of many species, gear selectivity, manpower costs, and geographic variation (Hocutt 1981), as well as limited utility of the IBI in small headwater streams where fish species richness is low (Miller et al. 1988). Geographic variation in the IBI has been addressed through the development of regional modifications to one or more of the metrics. Leonard and Orth (1986) modified the IBI for application in the Appalachian Mountains of West Virginia. In their study, IBI scores were negatively correlated with a Cultural Pollution Index measuring multiple degradations to streams. Effects of fish mobility and catchability on species representation can be reduced by sampling a sufficiently long segment of stream. Angermeier and Karr (1986) demonstrated the importance of increased sampling effort in reducing sampling error, and noted that effort should be constant among sites because metrics are sensitive to changes in effort.

Fish mobility may particularly confound biotic assessment in adventitious tributaries, where exchange of individuals between tributary and main stem populations may strongly influence the fish community. Results of 1990 survey of Virginia springs demonstrated the influence of location on species richness in springs. Gorman (1986) described the influence of receiving streams on composition and dynamics of tributary fish populations. Osborne and Wiley (1992) also demonstrated increased species richness in tributaries of large systems, and Osborne et al. (1992) cautioned that modifications to IBI procedures would be required for work in headwaters.

Invertebrates

Macroinvertebrate communities have been widely used for pollution assessment in European (Andersen et al. 1984) and African (Chutter 1972) rivers. Hilsenhoff (1977) developed a simplified biotic index dealing only with arthropod taxa which was further refined (Hellenthal 1982, Narf et al. 1984) to reduce time required for assessments through improved analytical techniques. A more recent family-level biotic index (FBI) significantly reduced processing time with some loss in accuracy (Hilsenhoff 1988). The United States Environmental Protection Agency currently incorporates the Hilsenhoff biotic index into a Rapid Bioassessment Protocol (Plafkin et al. 1989), which uses relative abundance of macroinvertebrate taxa and functional groups to evaluate impacts on aquatic ecosystems. The Ohio EPA (1987) has developed an Invertebrate Community Index (ICI), applying the format of the IBI to assess impacts on aquatic macroinvertebrate communities. Voshell and Hiner (1990) developed a bioassessment methodology using macroinvertebrates for water quality monitoring in Shenandoah National Park in Virginia.

PURPOSE OF STUDY

The 1991 springs study evaluated effects of cattle farming on biotic communities of spring streams using fish and macroinvertebrate assemblages and habitat character as indicators of impact. Understanding the structure of these biotic systems and their responses to perturbation will provide valuable information on watershed conditions. Comprehensive assessment of spring stream communities and impacts of livestock is a prerequisite to development of successful riparian management strategies. When restoration efforts are attempted, reliable and ecologically relevant measures of project success or failure will be required. Effective monitoring of fish and invertebrate communities may prove a useful technique for natural resource managers to evaluate agriculture, watershed, and livestock management practices.

The study objectives were to demonstrate impacts to biotic communities and habitat in spring streams, and to relate observed changes in the biotic community to habitat degradation.

METHODS

SITE SELECTION

The Virginia Springs database (Helfrich et al. 1989) provided basic information for selection of study sites. Large (>500 gpm) spring systems of the New River and James River basins were classified according to watershed character and described by field inspections in 1989-91 (Table 2.1). Sites were restricted to the New River and James River basins because of their accessibility for study and the range of cattle impacts evident at spring sites. Fish communities in these basins also showed minimal influence from receiving streams in 1990 samples, implying that fish communities may be naturally less variable and more responsive to habitat perturbations. Springs selected were generally similar in channel size and gradient, accessibility to downstream waters, and spring flow. Sampling occurred from 1 August until 26 September 1991.

Ten spring sites (Figure 2.1) were selected to encompass the full range of grazing impacts, from ungrazed to heavily grazed. Spring streams were ordinated along a gradient of grazing pressure based on herd density (animals/m^2 of accessible stream channel) and pasture rotation schedules (Table 2.2). Two unimpacted springs were ranked based on degree of forest disturbance in riparian areas.

BIOTIC ASSESSMENT

Fish

Two pool and two riffle areas were electrofished within each spring stream, except in Allen's Branch and W. C. Karnes Spring where only two segments

Table 2.1. Spring systems selected for survey in 1991, with river basin and primary watershed use (Helfrich et al. 1989).

Spring	River Basin	Watershed Use
Big (Hunt Club)	James	Mature forest
Shaffer	James	Mixed forest
W. C. Karnes (Purgatory Creek)	James	Cattle grazing/forest
Allen's Branch	James	Cattle grazing/forest
Big (Highland)	James	Cattle/sheep grazing
Stephenson	James	Cattle/horse grazing
McCorkle	James	Cattle grazing/forest
Brown	New	Cattle grazing
Pendry's Blue	New	Cattle grazing
Fish	New	Cattle grazing

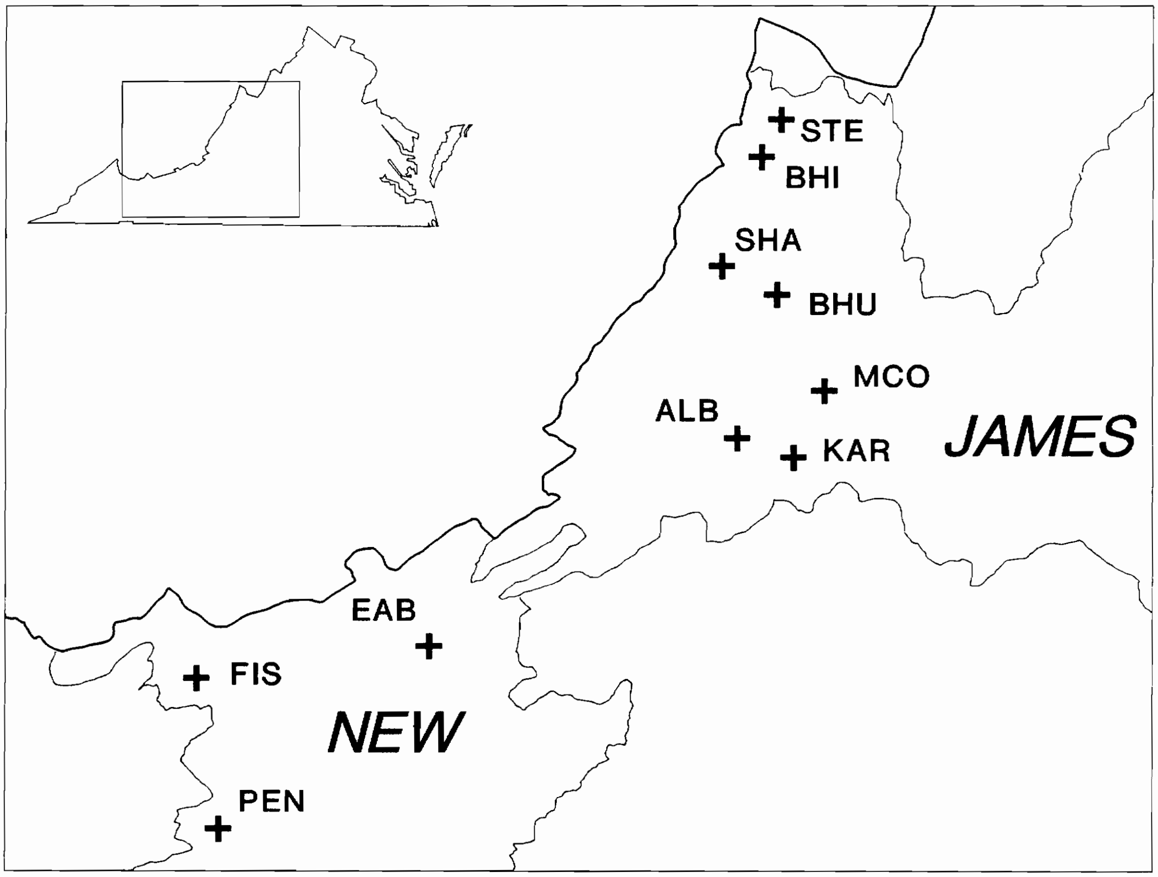

Figure 2.1. Sample sites, 1991 cattle grazing impact study: BHU = Big Spring (hunt club); SHA = Shaffer Spring; ALB = Allen's Branch; KAR = W. C. Karnes Spring (Purgatory Creek); PEN = Pendry's Blue Spring; BHI = Big Spring (Highland County); MCO = McCorkle Spring; EAB = E. A. Brown Spring; STE = Stephenson Spring; FIS = Fish Spring.

Table 2.2. Watershed characteristics used to rank ten 1991 spring sites by severity of cattle-related degradation.

Spring	River basin	Flow (m^3/s)	Watershed use	Herd density[1]	Rotation index[2]	Grazing pressure[3]	Rank
Big, hunt club (BHU)	James	0.074	Mature forest	0.00	0.0	0.00	1
Shaffer (SHA)	James	0.039	Disturbed forest	0.00	0.0	0.00	2
Allen's (ALB)	James	0.028	Forest/ pasture	1.03	0.5	0.52	3
Karnes (KAR)	James	0.228	Forest/ pasture	0.73	1.0	0.73	4
Pendry's Blue (PEN)	New	0.096	Pasture	1.39	1.0	1.39	5
Big, Highland (BHI)	James	0.135	Pasture	2.99	1.0	2.99	6
McCorkle (MCO)	James	0.084	Forest/ pasture	5.45	1.0	5.45	7
E A Brown (EAB)	New	0.025	Forest/ pasture	10.67	1.0	10.67	8
Stephenson (STE)	James	0.028	pasture	28.25	1.0	28.25	9
Fish (FIS)	New	0.039	pasture	28.78	1.0	28.78	10

[1] Herd density = number of cattle per $100m^2$ of accessible stream

[2] Rotation index: 0.0 = no cattle; 0.5 = seasonal grazing; 1.0 = year-round grazing

[3] Grazing pressure = Herd density x Rotation Index

were included. Because of changes in use of portions of these two springs, only one pool and riffle from each site was used for analyses. At all sites, block nets prevented movement of fish into or out of the sample areas. Fish were identified to species, measured, and released (voucher specimens were preserved). The proportion of individuals with external anomalies (wounds, lesions, deformities, visibly poor condition, or parasites) was recorded. Relative abundance and length frequencies were determined for each species.

Fish population metrics included species richness, number of native species, relative abundance of trophic and habitat guilds, and the proportion of exotics and anomalies as in the Index of Biotic Integrity (Karr et al. 1986; Leonard and Orth 1986). Trends in richness and relative abundances in spring streams were examined across a range of grazing intensities. Selected IBI metrics obtained from fish data were analyzed similarly. Species common to multiple springs were examined for trends in length frequency distribution related to increased grazing pressure.

Invertebrates

Random benthic invertebrate community samples were taken from rocky riffle areas in each spring. A Surber net sampler (30.5x30.5 cm, 363 micron mesh size) was used for all collections. Substrate was thoroughly sampled to a depth of approximately five cm below the surface, and samples were quantified by unit area (0.0929 m^2). Benthic samples were supplemented by invertebrates colonizing artificial samplers (Merritt and Cummins 1984) set in seven spring streams (N = 3 each). Each colonization sampler (Figure 2.2) consisted of a 30-cm section of polyethylene column filtration material (BIO-NET 150, manufactured by NSW Corporation, Roanoke, Virginia), secured horizontally to the stream bottom and oriented parallel to the water current. Two of the three internal chambers of the column contained sterile, sorted river pebbles (16-40 mm dia.). Colonization samplers were left in contact with the stream bottom for 37 days. A sheathe of PVC pipe (7.62 cm dia.) prevented loss of organisms during retrieval.

All Surber and colonization samples were sorted and quantified to the family level for insects and phylum or class level for non-insect invertebrates. Samples were divided and subsampled when large numbers of small chironomid larvae were encountered in Surber samples. Trends in total macroinvertebrate abundance were examined for response to cattle impacts. Taxa richness and proportional abundance of mayflies, stoneflies, caddisflies, and noninsect taxa, proportional abundance of amphipods, gastropods, and fly (Diptera) larvae, and ratio of EPTs (mayflies, stoneflies, and caddisflies) to Chironomid midge larvae were also analyzed for trends across the cattle impact gradient.

HABITAT CHARACTERIZATION

Spring stream samples encompassed the entire spring run on small systems or >300 m of stream channel in larger watersheds. Stream sample areas began and ended at distinct pool/riffle interfaces.

Discharge (cms) was determined by the method of Armour et al. (1983). Water chemistry data (dissolved oxygen, pH, hardness, alkalinity,

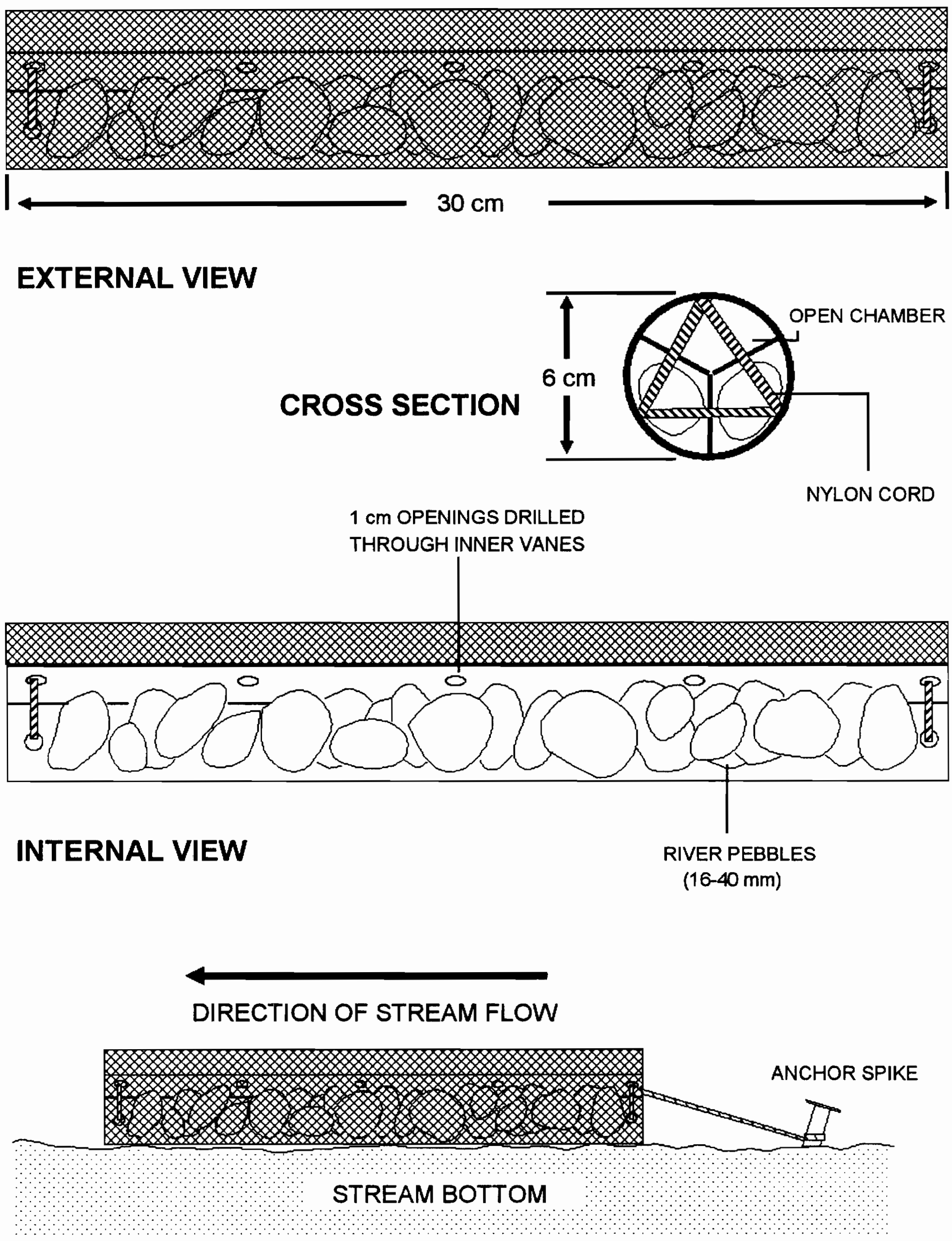

Figure 2.2. Colonization sampler used in benthic invertebrate collection, 1991 cattle impact study.

conductivity, nitrate, nitrite, and ammonia) were collected above and below the sample area and analyzed for trends across the cattle impact gradient. Spring streams were stratified by mesohabitat type (pool or riffle), which was defined on the basis of visual estimates of gradient, substrate character, and water surface turbulence. Lengths of each pool and riffle and totals for each mesohabitat type were measured at each site.

Habitat transects (N = 5 each, pool and riffle) were placed at uniform intervals. Aquatic and riparian habitat data were collected for each transect. Bank angles were measured and streambank vegetative stability rated as described in Platts et al. (1983). Streambank soil alteration was rated on a five-point scale simplified from Platts et al. (1983), as shown in Table 2.3. Channel width (m), mid-channel water temperature (^{O}C), and accessibility to cattle were also determined.

Benthic habitat was assessed at five equidistant points on each transect across both pools and riffles. Water depth (mm) was measured, and substrate was characterized for coarseness and embeddedness using a four-component rating system as described by Crouse et al. (1981). Instream cover was categorized as rocky (gravel or larger), vegetative (moss, filamentous algae, leaf packs, or vascular macrophytes), woody (sticks, logs, rootwads), or no cover. Shading was defined by the presence or absence of light-obstructing canopy directly above the water surface at the transect point. Depth of fine substrate was measured by inserting a wooden dowel (12.7 mm dia.) into the stream bottom.

Stream-, mesohabitat-, transect-, and point-level variables are summarized in Table 2.4. Descriptive statistics for point-specific variables were

Table 2.3. Five-point streambank soil alteration rating used in riparian evaluation of 1991 study sites, modified from Platts et al. (1983) and Armour et al. (1983).

Old rating (Platts et al. 1983)	Simplified rating (Present study)	Description
0	5	Streambanks are stable and not altered by water flows or animals.
1 to 25	4	Streambanks are stable but lightly altered. Less than 25% of the streambank is receiving stress and existing stress is light. Less than 25% of the streambank is false* or eroding.
26 to 50	3	Streambanks are receiving only moderate alteration. At least 50% of the streambank is in a natural, stable condition. Less than 50% of the streambank is false* or eroding.
51 to 75	2	Streambanks have received major alteration. Less than 50% of the streambank is in a stable condition. Over 50% of the streambank is false* or eroding.
76 to 100	1	Streambanks are severely altered. Less than 25% of the streambank is in a stable condition. Over 75% of the streambank is false* or eroding.

* False stream banks are banks that have been eroded away and have receded back from the edge of the water. They can become stabilized by vegetation, but the edges do not hang over the water to provide cover for fish. False banks that have gained some stability and cover are still rated as altered (Armour et al. 1983).

Table 2.4. Habitat and biotic variables, categorized by the scale at which they were measured, at 1991 spring sites.

Stream Reach	Mesohabitat	Transect	Point
Spring Flow	Fish Metrics	Width (m)	Depth (m)
Dissolved O_2[*]		Temperature (°C)	Substrate size
pH[*]		Streambank soil stability	Substrate diversity
Hardness[*]		Streambank vegetative stability	Embeddedness
Alkalinity[*]		Bank angle	Cover type
Conductivity[*]			Shading
Nitrate[*]			Fines depth
Nitrite[*]			Invertebrate metrics
Ammonia[*]			
Grazing intensity			

[*] Variables measured at beginning and end of study reach.

incorporated into transect and mesohabitat data, and habitat trends across the cattle impact gradient were analyzed. Habitat variables were tested for correlation with all biotic metrics that responded to grazing intensity.

STATISTICAL ANALYSIS

Habitat and biotic trends across the cattle impact gradient were tested using the Jonckheere Distribution-free Test for Ordered Alternatives (Hollander and Wolfe 1973). A priori assumptions of expected directional responses to grazing pressure (increase or decrease of given variables) were tested at the 0.05 level of significance. Correlations of key biotic variables and habitat character were tested using Kendall's Distribution-free Test for Independence (Hollander and Wolfe 1973).

RESULTS

FISH

Spring fish populations showed limited response to grazing impacts. As expected, species richness was low, with only two sites containing more than four species. Fish species collected from ten 1991 spring sites are shown in Table 2.5.

Trend analysis showed no significant effects of watershed use on fish species richness and density nor on proportional abundances of benthic feeders, non-benthic feeders, invertivores, omnivores, herbivores, or darters (Etheostoma sp.). Combined proportional abundance of sculpin (Cottus sp.) and blacknose dace (Rhinichthys atratulus), representing the most tolerant spring species, also showed no trend related to cattle densities.

Two fish metrics showed significant downward trends with increased grazing intensity. Piscivore proportional abundance (Figure 2.3) declined in moderately impacted sites, and piscivores were absent from two heavily impacted springs. However, piscivores were also absent from two relatively unimpacted sites and were only abundant in one unimpacted site. Benthic invertivores were the only trophic guild consistently present at study sites. Relative stock density of young-of-year (all individuals under 40 mm were considered young-of-year) benthic invertivores (RSD_{YOYB}) was reduced in heavily impacted springs but varied in moderately degraded sites (Figure 2.4). No young-of-year benthic fishes were found in the site with heaviest cattle use.

While no metrics addressed within-guild replacement of species, some shift toward coolwater and warmwater forms was observed in springs used

Table 2.5. Fish species collected from ten 1991 spring sites, ranging from unimpacted (BHU) to most impacted (FIS).

	SITE									
SPECIES	BHU	SHA	ALB	KAR	PEN	BHI	MCO	EAB	STE	FIS
Oncorhynchus mykiss	34	-	-	25	25	-	-	-	-	-
Salvelinus fontinalis	-	-	-	-	-	28	1	-	10	-
Campostoma anomalum	-	-	1	-	-	41	-	-	-	1
Clinostomus funduloides	-	-	-	-	-	14	1	-	-	-
Luxilus cornutus	-	-	-	-	-	19	-	-	1	-
Nocomis leptocephalus	-	-	-	-	-	2	-	-	-	-
Nocomis micropogon	-	-	-	-	-	-	-	-	24	-
Phoxinus oreas	-	-	-	-	-	1	-	-	2	-
Rhinichthys atratulus	18	-	-	33	11	161	4	-	45	60
Catostomus commersoni	-	-	-	-	1	13	-	-	-	-
Thoburnia rhothoeca	-	-	-	-	-	49	-	-	-	-
Etheostoma flabellare	-	-	-	-	-	-	-	-	-	7
Etheostoma simoterum	-	-	-	-	-	-	-	-	-	20
Cottus bairdi	128	49	196	634	-	426	346	-	80	-
Cottus carolinae	-	-	-	-	661	-	-	138	-	-

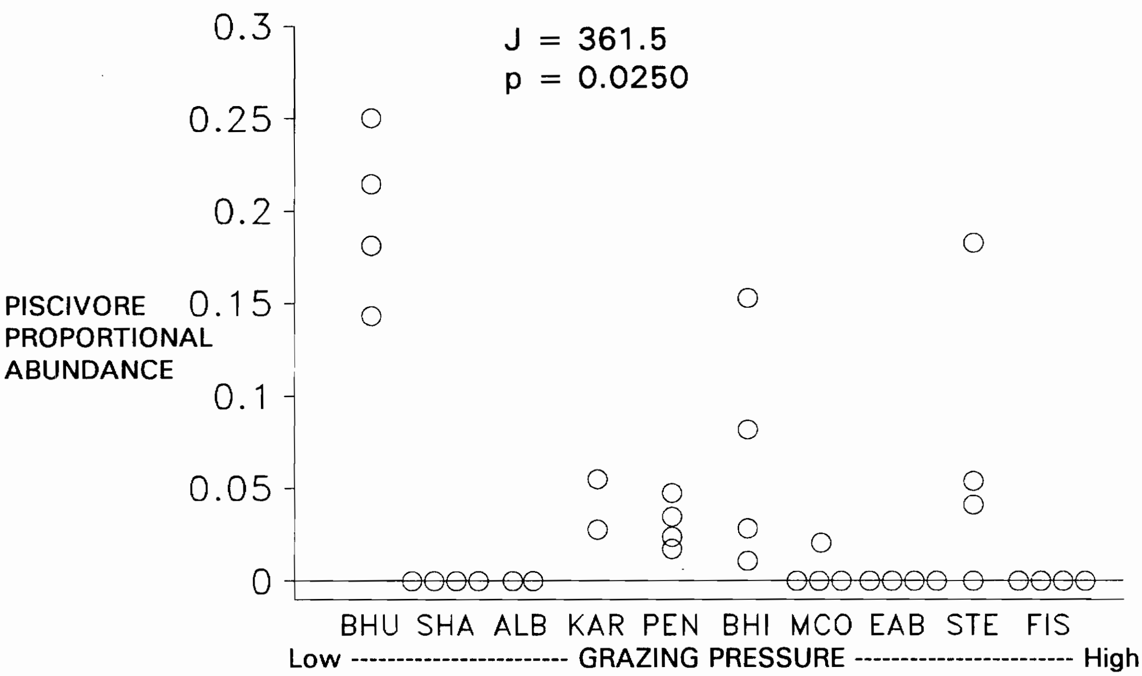

Figure 2.3. Proportional abundance of piscivores in ten springs, 1991 cattle impact study.

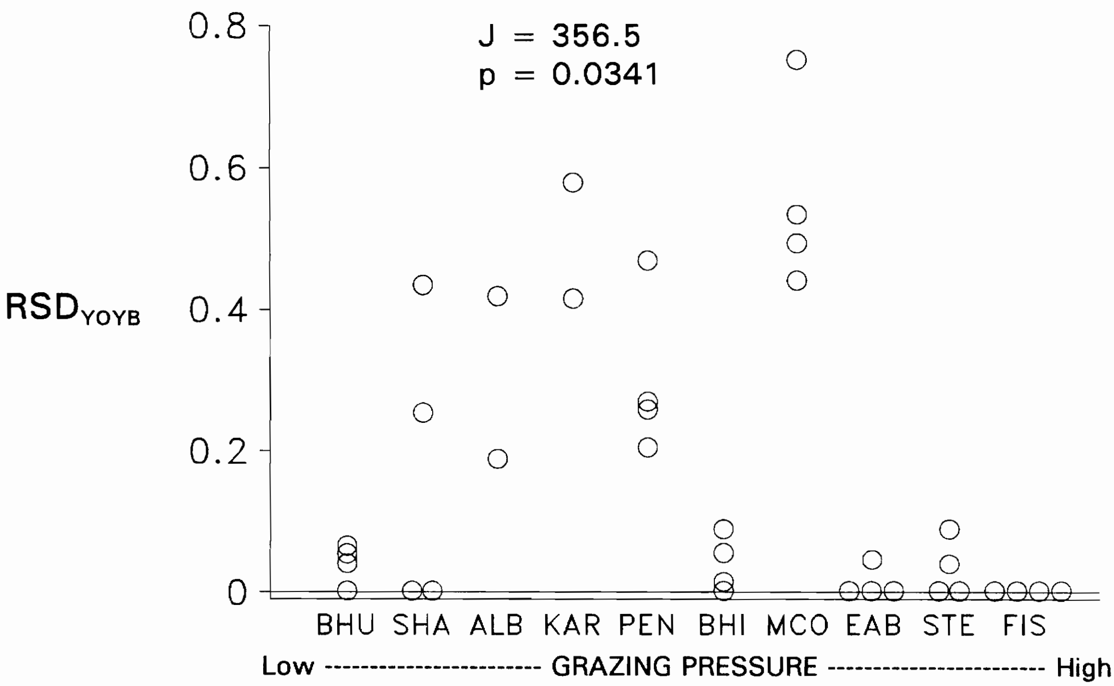

Figure 2.4. Proportion of young-of-year benthic invertivores (RSD$_{YOYB}$) in ten springs, 1991 cattle impact study.

by cattle. The four springs with least grazing pressure held only coldwater species. One white sucker (Catostomus commersoni) appeared in samples from Pendry's Blue Spring, a moderately impacted stream in a pastured watershed. Three of the five most heavily impacted springs contained species that were not typical of headwaters, including common shiner (Luxilus cornutus), river chub (Nocomis micropogon), and snubnose darter (E. simoterum).

No trends in health of individual fish were evident from external examination. Wounds, lesions, and anomalies were not evident in fish samples. External parasites (black spot) were found only on blacknose dace from McCorkle Spring and did not appear to impair fish condition.

INVERTEBRATES

In contrast to many fish metrics, characteristics of benthic macroinvertebrate communities showed significant trends across the grazing impact gradient. Taxa richness within key taxonomic groups and relative abundance of taxonomic groups were strongly linked to increasing cattle densities in spring streams. Invertebrates collected in colonization samplers exhibited trends similar to Surber collections at six of seven sites where they were deployed. Colonization samplers were not effective in Stephenson Spring, where sediments completely buried samplers and anaerobic conditions apparently prevented colonization. Invertebrate taxa collected in 1991 Surber samples and colonization samplers are given in Tables 2.6 and 2.7.

Table 2.6a. Insects collected in Surber samples (mean abundance of three samples) from ten 1991 spring sample sites.

	SPRING									
INSECT FAMILY	BHU	SHA	ALB	KAR	PEN	BHI	MCO	EAB	STE	FIS
Baetidae	63.3*	116.7*	151.7*	21.3*	160.3*	9.0^{+}	9.7*	141.0*	10.7*	9.3^{+}
Heptageniidae	0.3	.	2.7*	0.7	.	3.7*	2.3	.	.	.
Ephemerellidae	0.7	2.3*	137.3*	2.3*	9.3*	4.3*	.	.	14.0*	315.3*
Caenidae	.	.	.	.	.	.	0.3	.	.	.
Leptophlebiidae	.	0.3	.	0.3	0.3	18.7^{+}	0.3	.	.	.
Gomphidae	7.3^{+}	.	.	.	.	.	.	.	.	.
Pteronarcyidae	2.0^{+}	.	6.3^{+}	.	.	.	.	.	.	.
Peltoperlidae	0.3	.	0.3	.	.	.	.	.	.	.
Taeniopterygidae	.	1.7	.	.	.	.	.	.	.	.
Nemouridae	181.0^{+}	0.3	.	.	.	.	.	.	.	.
Leuctridae	34.7^{+}	2.3^{+}	35.0*	58.0*	.	10.7^{+}	4.7	.	3.3*	.
Capniidae	.	.	.	.	0.3	.	.	.	0.7	.
Perlodidae	1.7	0.3	0.7	1.0	1.7^{+}	.	0.3	.	1.0	.
Chloroperlidae	13.0^{+}	.	0.7	10.0*	.	.	.	1.0	.	.
Sialidae	1.0	.	.	.	.	1.7	.	.	0.3	.
Corydalidae	.	.	.	0.3	.	.	0.7^{+}	.	.	.
Polycentropodidae	0.3	.	.	.	.	.	.	.	.	.
Hydropsychidae	3.7*	6.0^{+}	26.7*	35.7*	49.0*	14.0*	5.0*	0.3	1.0^{+}	.
Rhyacophilidae	3.0*	13.0^{+}	32.0*	.	4.7^{+}	7.0*	.	4.3	1.0^{+}	.
Glossosomatidae	1.7^{+}	.	.	.	.	.	0.3	12.7*	.	.
Hydroptilidae	.	.	.	3.0*	0.3	16.0*	0.3	.	.	.
Phryganeidae	.	.	.	.	.	.	.	.	3.7^{+}	.
Brachycentridae	5.0	49.0*	1.0	19.7*	165.3*	16.3^{+}	44.0*	1.0	422.7*	.
Lepidostomatidae	.	.	.	0.3	.	.	1.7^{+}	.	.	.
Limnephilidae	1.0^{+}	1.3*	0.3	8.0^{+}	8.0*	36.0*	0.7	.	294.0*	4.7*
Odontoceridae	.	.	.	3.7*	.	.	.	.	.	.
Pyralidae	.	.	0.3	.	.	.	0.3	.	.	.
Dytiscidae	.	.	.	.	.	.	.	.	0.3	.
Hydrophilidae	.	.	.	0.3	0.3	.	0.3	.	.	0.3
Psephenidae	3.7*	.	0.7	.	.	.	.	.	0.7	.
Elmidae	29.7*	.	49.3*	7.0*	6.7*	25.7*	8.3*	43.3^{+}	11.7^{+}	.
Chrysomelidae	.	0.3	.	.	.	.	.	.	.	.
Tipulidae	7.0*	64.7*	40.0*	147.7*	2.0^{+}	209.7*	2.3*	2.7*	35.3*	.
Psychodidae	.	.	.	.	.	.	0.3	.	.	.
Ceratopogonidae	.	.	.	1.7*	.	2.0	.	1.7^{+}	4.7*	.
Simuliidae	1.0	0.3	21.7*	6.7^{+}	4.3^{+}	23.7*	24.7^{+}	4.3^{+}	10.0*	.
Chironomidae	29.0*	22.0*	127.7*	811.7*	345.3*	989.7*	164.7*	376.3*	582.3*	1,292.3*
Dixidae	.	.	1.3	.	.	.	.	47.7*	.	.
Stratiomyidae	.	.	.	0.3	.	.	.	1.0	.	.
Tabanidae	.	.	.	.	.	1.0	.	.	6.7^{+}	.
Empididae	1.3^{+}	1.0*	1.7^{+}	3.7*	1.0	1.7	1.0^{+}	.	7.3^{+}	16.3*
Ephyridae	.	.	.	.	.	.	.	.	0.3	.
Muscidae	.	.	.	0.7	6.3^{+}	.	1.7*	0.3	1.0^{+}	.

$^{+}$ family represented in more than one Surber sample from spring
* family represented in all Surber samples from spring

Table 2.6b. Non-insect macroinvertebrates collected in Surber samples (mean abundance of three samples) from ten 1991 spring sample sites.

	SPRING									
TAXON	BHU	SHA	ALB	KAR	PEN	BHI	MCO	EAB	STE	FIS
Turbellaria	2.0+	1.0+	0.3	7.0*	13.0+	1.0	.	119.0*	14.3+	.
Nematoda	0.3	0.3	.	0.3	1.7	1.0	1.3+	3.0+	56.3*	22.0*
Annelida										
Oligochaeta	5.3	5.0*	2.0*	6.7*	28.0*	44.0	28.3*	575.0*	189.0*	42.7*
Hirudinea	0.3	.	.	.	.	1.0	.	.	.	.
Arthropoda										
Ostracoda	.	.	.	17.7*	1.3*	1.7	120.0*	195.0*	78.7*	22.3+
Isopoda	.	.	.	.	.	.	11.3*	0.3	.	.
Amphipoda	.	107.0*	3.3*	.	686.7*	104.3*	31.0*	1,376.0*	83.7*	175.7*
Decapoda	.	.	0.3	.	.	0.3	.	.	.	.
Arachnoidea	5.0+	9.3*	7.3*	21.3*	8.3*	61.0*	14.0*	71.0*	118.3*	1.7
Mollusca										
Gastropoda	14.7*	200.3*	0.3	11.0*	26.3*	139.3*	196.0*	915.3*	3,363.3*	7.0*
Pelecypoda	0.3	.	.	.	.	2.3*	51.3	8.0*	28.0*	1.0+

+ taxon represented in more than one Surber sample from spring
* taxon represented in all Surber samples from spring

Table 2.7. Macroinvertebrates colonizing artificial samplers (mean abundance of three samplers) from six 1991 spring sample sites.

TAXON	SPRING					
	BHU	ALB	PEN	BHI	EAB	FIS
INSECT FAMILIES						
Ephemeridae	.	0.7	.	.	.	.
Baetidae	24.3*	14.0^{+}	137.3*	7.3^{+}	21.3^{+}	0.7
Heptageniidae	.	2.0^{+}	.	.	.	.
Ephemerellidae	.	306.0*	0.7^{+}	.	.	13.3^{+}
Leptophlebiidae	0.3	1.7	.	.	.	.
Gomphidae	.	0.7	.	.	.	.
Pteronarcyidae	1.7*	2.7*	.	.	.	.
Peltoperlidae	.	.	.	0.7	.	.
Nemouridae	3.0^{+}	.	.	.	.	.
Leuctridae	3.0^{+}	5.0*	0.7	1.0	.	.
Perlidae	.	.	.	1.0	.	.
Perlodidae	.	0.3	0.3	2.3	.	.
Corydalidae	.	0.3	.	.	.	.
Hydropsychidae	11.0*	4.7	415.0*	255.0*	0.3	.
Rhyacophilidae	2.3*	3.3*	0.3	108.3*	.	.
Hydroptilidae	.	.	.	2.7	.	.
Phryganeidae	.	0.3	0.3	.	.	.
Limnephilidae	0.3	0.7^{+}	85.7^{+}	140.0*	.	.
Dytiscidae	.	.	.	.	.	0.3
Elmidae	0.7^{+}	7.0^{+}	.	0.3	0.3	.
Tipulidae	0.7^{+}	2.3^{+}	.	25.7*	.	.
Ceratopogonidae	.	0.3	.	.	0.7^{+}	.
Simuliidae	96.7*	2.7^{+}	3.7^{+}	786.0^{+}	8.0^{+}	0.3
Chironomidae	19.3*	202.7*	392.3*	1,357.3*	107.0*	270.7*
Dixidae	.	.	0.7	.	2.7	.
OTHER TAXA						
Annelida						
Oligochaeta	.	0.3	1.0	.	1.7^{+}	26.0*
Hirudinea	.	.	.	0.3	.	.
Arthropoda						
Ostracoda	.	.	.	.	6.7^{+}	.
Amphipoda	.	2.3*	43.0*	37.0*	373.3*	259.3*
Decapoda	0.3	0.7	.	.	.	.
Arachnoidea	.	.	1.3*	3.3^{+}	.	.
Mollusca						
Gastropoda	15.0*	4.3*	20.3*	5.0^{+}	136.0*	20.0^{+}
Pelecypoda	.	1.0^{+}	.	0.3	.	0.7^{+}

$^{+}$ taxon represented in more than one sampler from spring
* taxon represented in all samplers from spring

Colonization samplers contained most taxa found in Surber samples. Taxa absent from colonization samplers were typically burrowing forms (Turbellaria, Nematoda, Empididae, Muscidae) or organisms with relatively narrow preferences for substrate attachment (Brachycentridae). Taxa not common in Surber samples were often absent from colonization samplers.

Total macroinvertebrate abundance for both Surber and colonization samplers (Figure 2.5) increased significantly with higher cattle densities. Overall taxa richness showed no directional trend in Surber samples, but decreased significantly in colonization samplers ($J = 103.0$, $p = 0.0031$). The number of insect families represented decreased in both Surber samples ($J = 257.0$, $p = 0.0252$) and colonization samplers ($J = 114.0$, $p = 0.0002$).

Taxa richness within insect orders changed markedly as grazing pressure increased. Ephemeroptera, Plecoptera, and Trichoptera collectively (EPT) declined in richness (Surber $J = 291.0$, $p = 0.0008$; colonization samplers $J = 110.5$, $p = 0.0005$). Plecoptera richness declined in both Surber samples ($J = 288.0$, $p = 0.0011$) and colonization samplers ($J = 109.5$, $p = 0.0007$). Trichoptera richness also declined in all samples (Surber $J = 253.5$, $p = 0.0336$; colonization samplers $J = 95.0$, $p = 0.0171$). While Ephemeroptera taxa richness was not sigificantly different in Surber samples, fewer families from this order colonized samplers ($J = 92.5$, $p = 0.0272$) from heavily impacted sites. Diptera taxa richness did not differ significantly among springs.

Relative abundance of taxonomic groups showed strong trends across the cattle impact gradient. Proportional abundances of Ephemeroptera

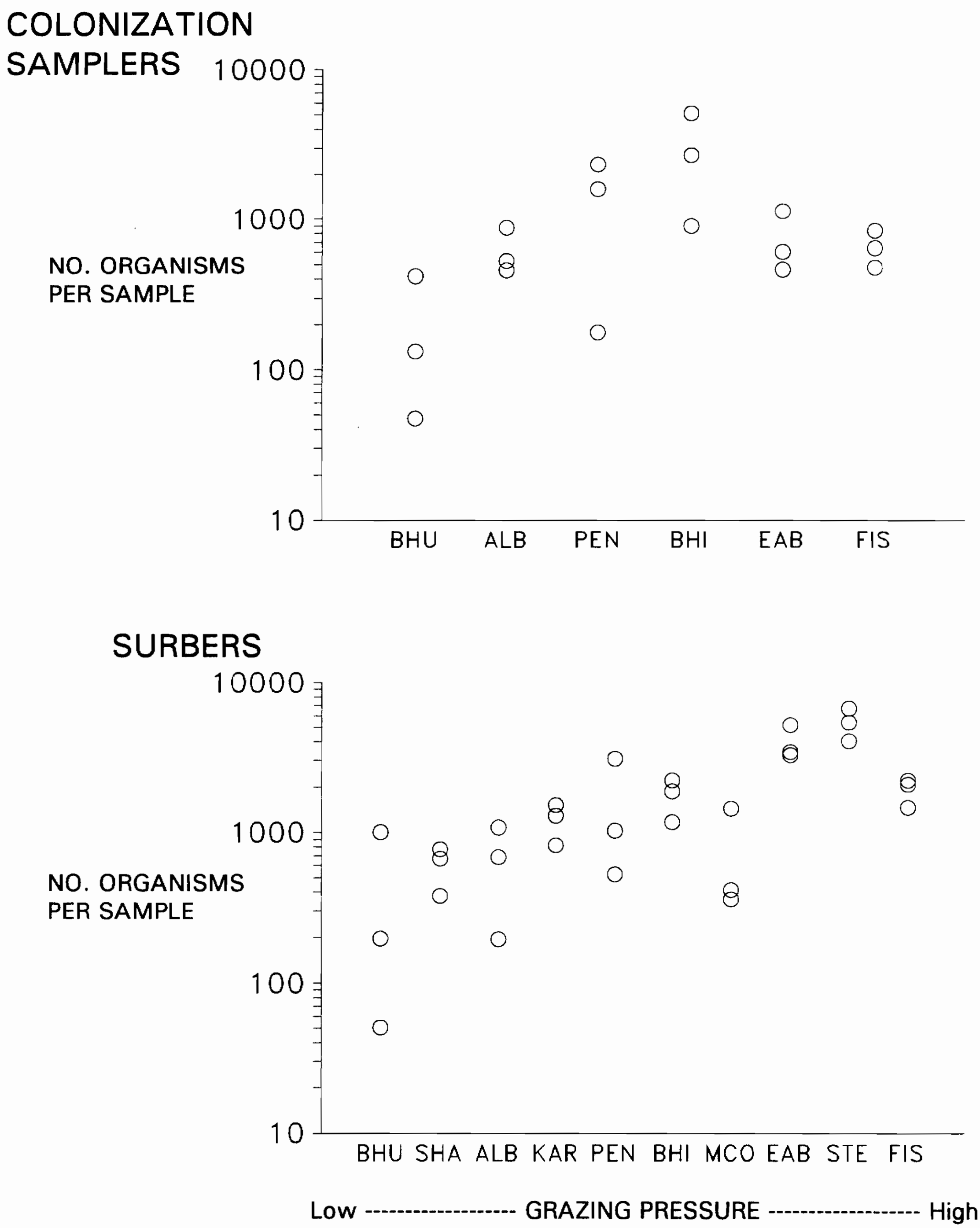

Figure 2.5. Total macroinvertebrate abundance in colonization samplers and Surber samples, 1991 cattle impact study.

(Figure 2.6), Plecoptera (Figure 2.7), and Trichoptera (Figure 2.8) were lower in springs with progressively heavier grazing pressure, while proportional abundance of non-insect taxa increased (Figure 2.9). The ratio of EPT taxa to Chirononomidae larvae (Figure 2.10) decreased dramatically in sites accessible to cattle. Amphipods accounted for a greater proportion of samples from impacted springs, but this trend was only significant in colonization samplers (Figure 2.11). Gastropod proportional abundance varied considerably within and among sites and showed no relation to watershed impacts.

WATER QUALITY

Spring discharge varied among sites but showed no trends associated with watershed character. Discharge levels in heavily grazed watersheds were similar to those in lightly grazed or ungrazed sites. Dissolved oxygen also varied independently of watershed character, and was not significantly reduced below study sites. In six of ten springs, dissolved oxygen levels were higher at downstream sample points than in upstream areas.

Chemical characteristics of spring water (Table 2.8) were generally unrelated to watershed use with the exception of nitrogen compounds. Conductivity varied widely and showed no trends across the grazing impact gradient. Impacted springs showed no evidence of acidification; alkalinity and hardness were not reduced in impacted sites, and in fact appeared to increase in some springs with high cattle densities. Both hardness and alkalinity levels were higher in heavily grazed watersheds than in moderately

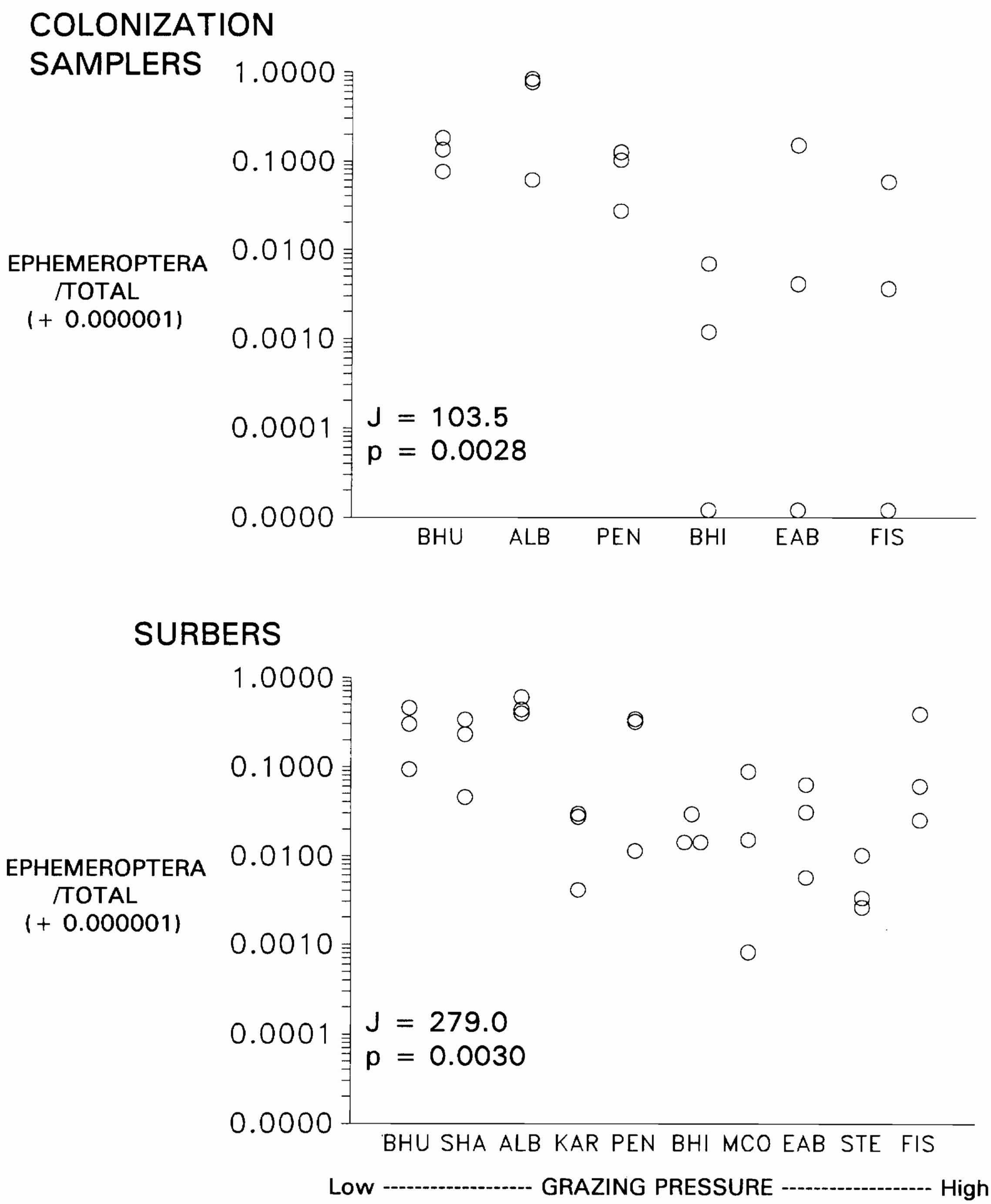

Figure 2.6. Proportional abundance of Ephemeroptera in colonization samplers and Surber samples, 1991 cattle impact study.

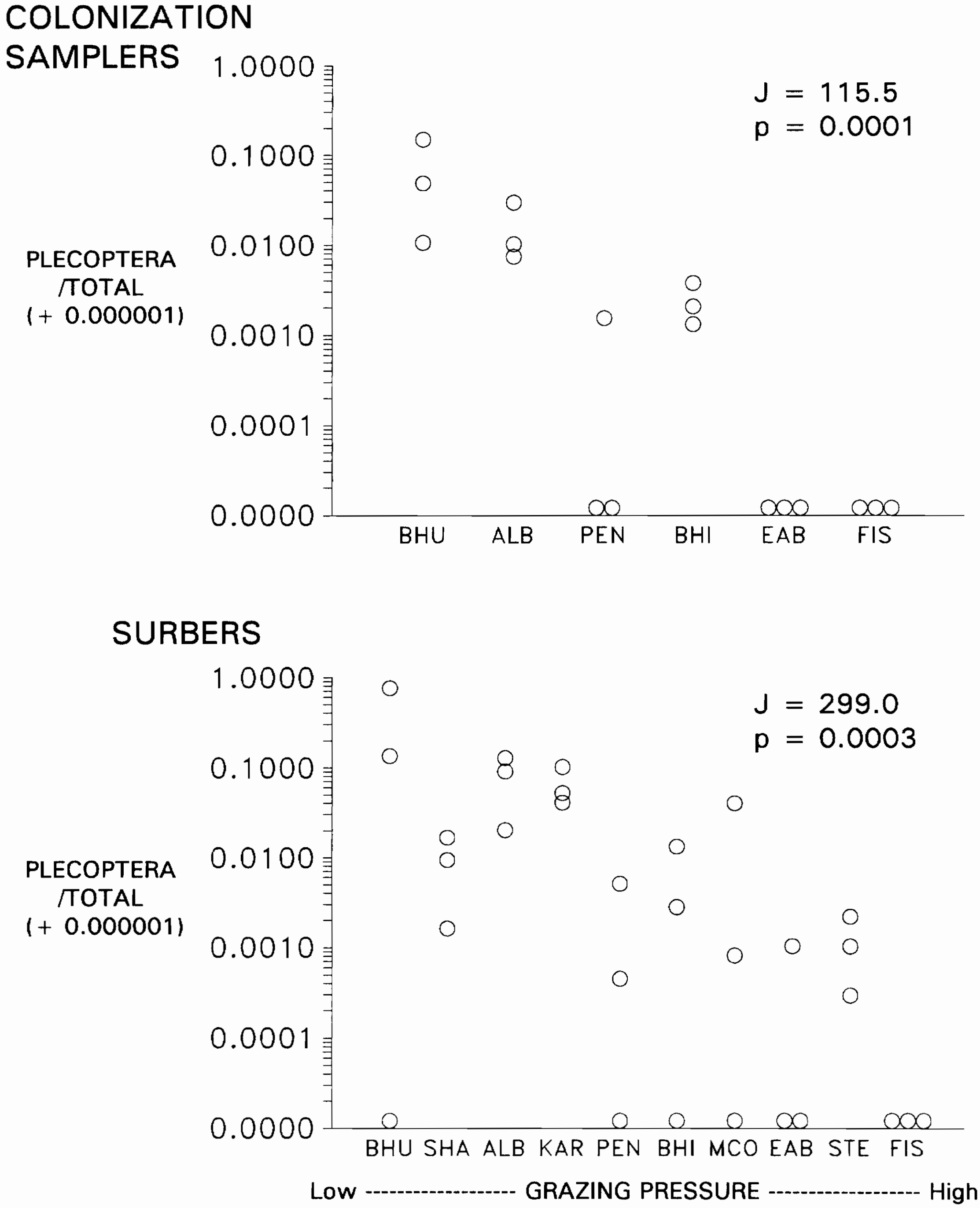

Figure 2.7. Proportional abundance of Plecoptera in colonization samplers and Surber samples, 1991 cattle impact study.

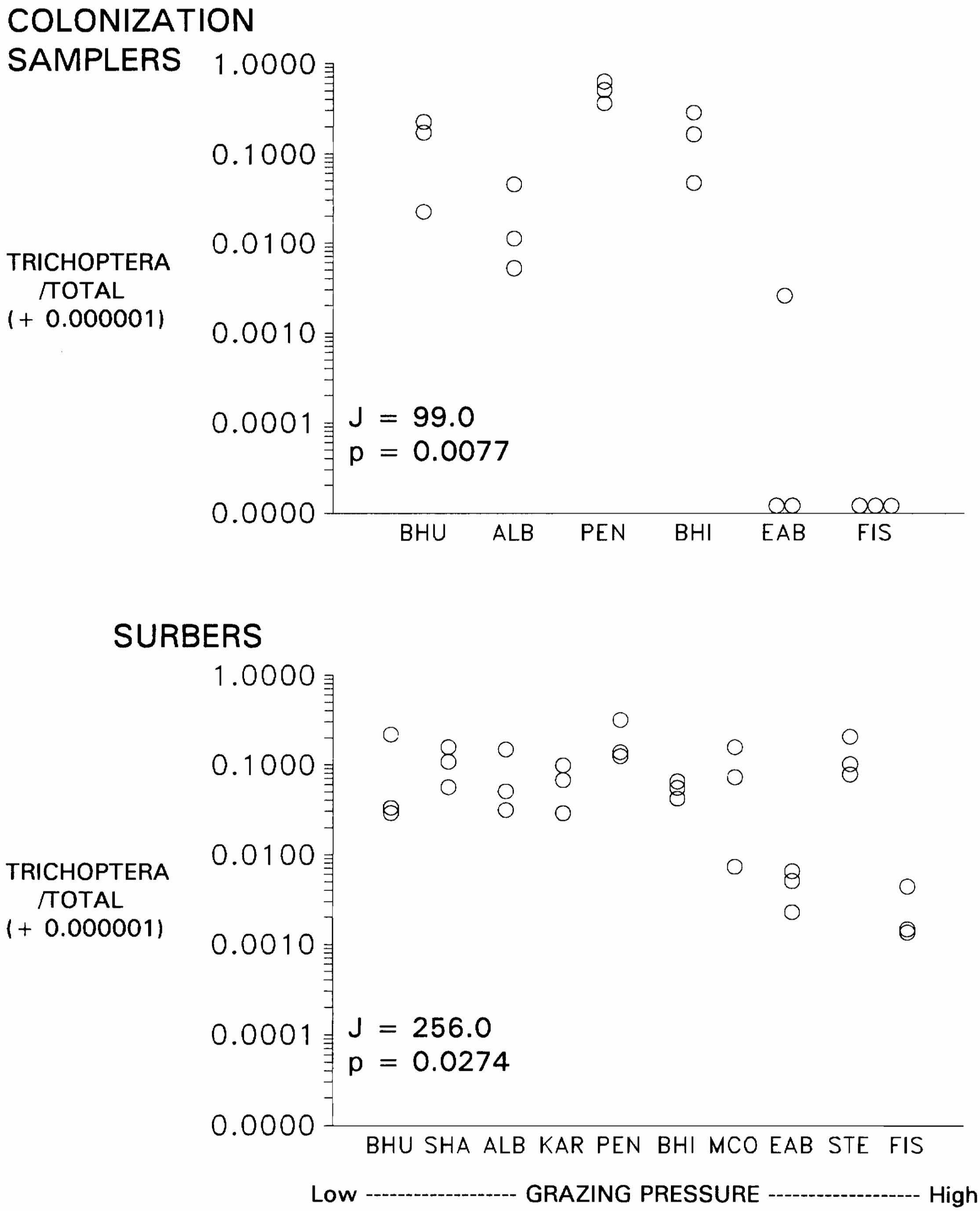

Figure 2.8. Proportional abundance of Trichoptera in colonization samplers and Surber samples, 1991 cattle impact study.

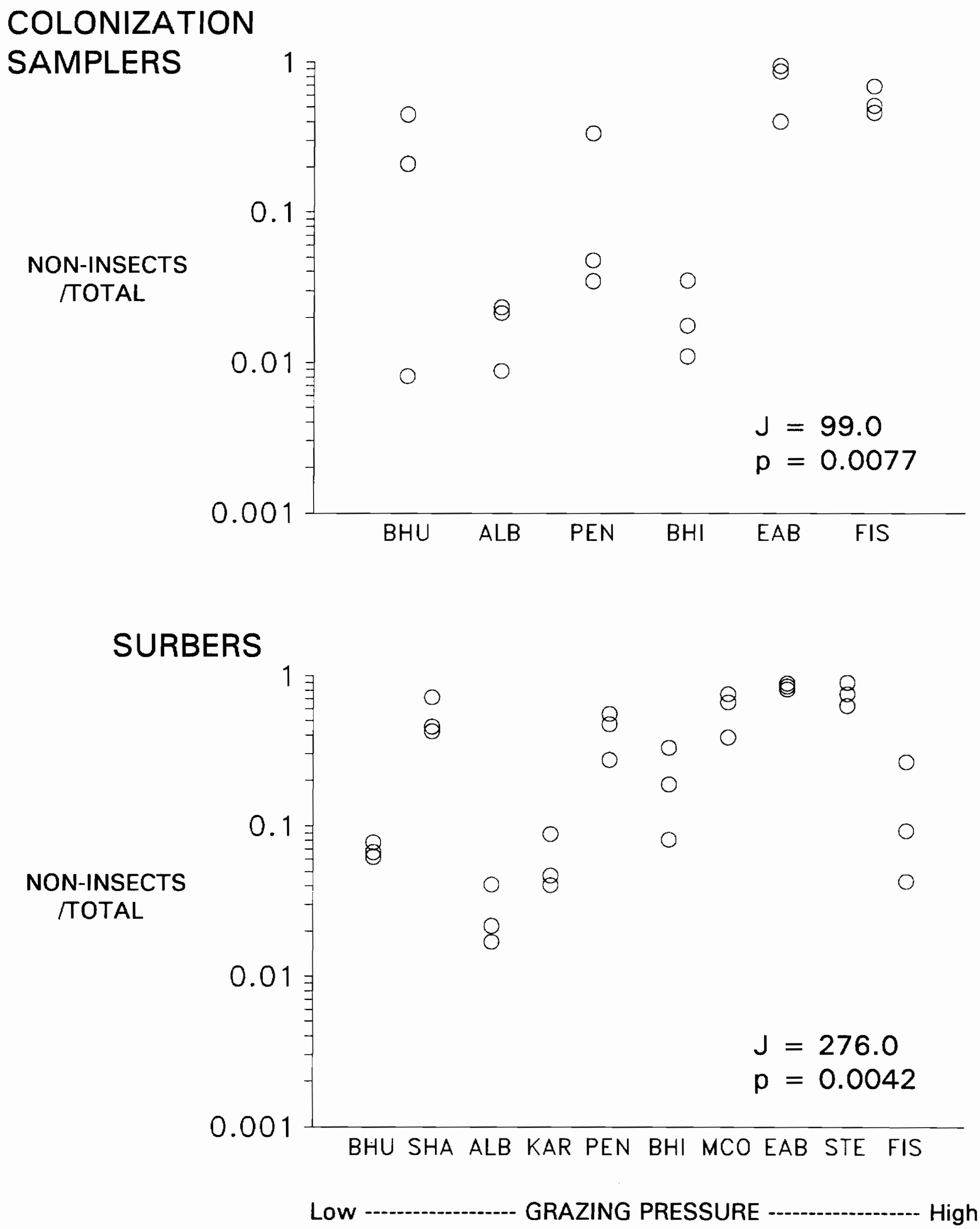

Figure 2.9. Proportional abundance of non-insect macroinvertebrate taxa in colonization samplers and Surber samples, 1991 cattle impact study.

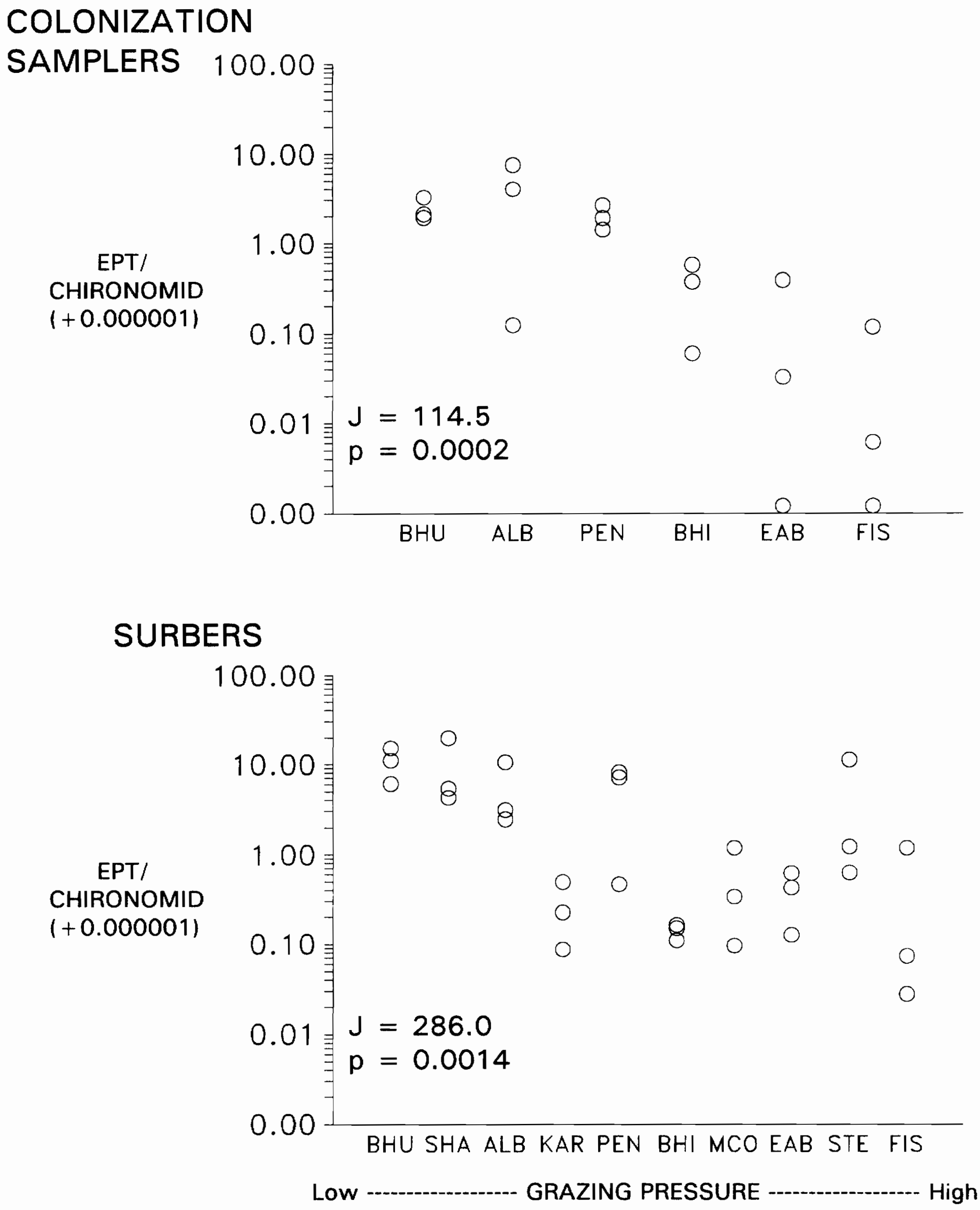

Figure 2.10. Ratio of EPT taxa to Chironomidae in colonization samplers and Surber samples, 1991 cattle impact study.

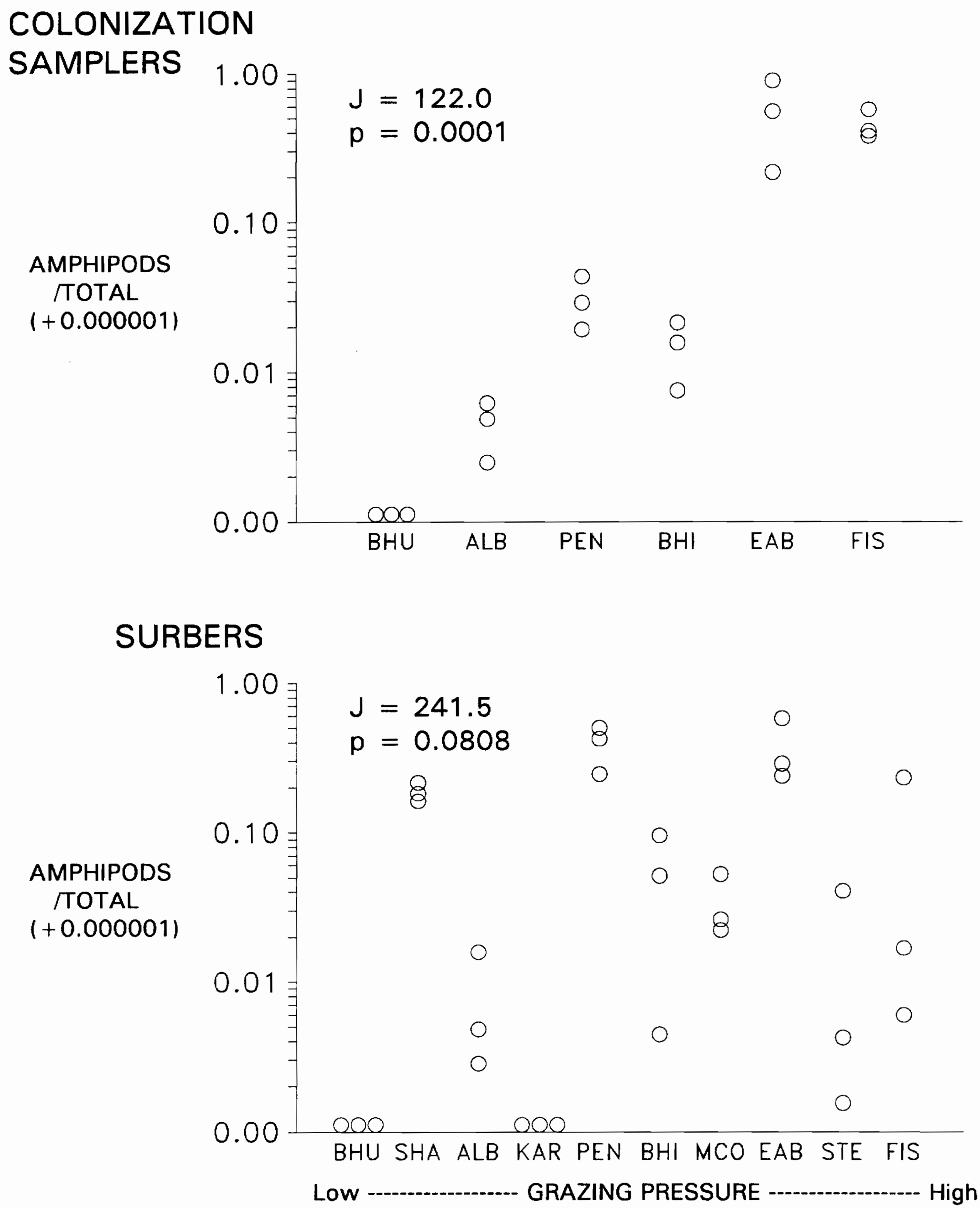

Figure 2.11. Proportional abundance of Amphipoda in colonization samplers and Surber samples, 1991 cattle impact study.

Table 2.8. Water chemistry variables measured at the top and bottom of sample reaches, 1991 spring sites (N = 10). Cattle impacts range from unimpacted (Big, hunt club) to most impacted (Fish).

Spring	Sample site	DO (mg/l)	pH	Hardness (mg/l $CaCO_3$)	Alkalinity (mg/l $CaCO_3$)	Conductivity (µS/cm)	Nitrates (mg/l)
Big,	top	8.9	7.9	201	102	129	0.4
hunt club	bottom	9.5	8.0	141	107	131	0.4
(BHU)							
Shaffer	top	11.2	7.9	155	98	128	0.4
(SHA)	bottom	11.3	7.9	137	101	126	0.4
Allen's	top	10.0	8.6	211	196	295	0.0
(ALB)	bottom	9.3	8.7	210	197	255	0.0
Karnes	top	9.2	7.9	158	124	160	1.1
(KAR)	bottom	10.4	8.3	157	131	155	1.7
Pendry's	top	9.5	7.9	164	106	120	2.4
Blue	bottom	9.1	8.0	144	94	125	1.9
(PEN)							
Big,	top	11.1	7.7	140	113	145	1.8
Highland	bottom	12.0	8.0	132	112	142	1.9
(BHI)							
McCorkle	top	8.7	8.3	190	175	no data	0.0
(MCO)	bottom	9.3	8.5	193	176	no data	0.0
E A Brown	top	9.7	7.5	252	208	280	2.6
(EAB)	bottom	8.9	7.7	246	215	280	2.6
Stephenson	top	11.0	8.1	178	151	221	2.0
(STE)	bottom	10.1	8.6	179	144	206	2.4
Fish	top	8.3	7.6	232	179	220	1.6
(FIS)	bottom	8.6	7.7	247	183	230	1.6

impacted areas, but one lightly grazed system (Allen's Branch) also had elevated alkalinity and hardness. No trends in pH levels occurred as grazing intensity increased, but downstream pH levels were equal to or greater than upstream readings in all springs. Nitrite and ammonia were only detectable in downstream samples at three sites and one site respectively, but nitrate increased with increasing cattle densities in spring watersheds ($J = 121.0$, $p = 0.0216$). Elevated nitrate levels were present in emerging spring waters in affected watersheds and typically did not increase below reaches accessible to cattle.

INSTREAM AND RIPARIAN HABITAT

All measured physical habitat variables reflected increases in densities of cattle using spring channels. Scores representing substrate particle size and coarseness (Figure 2.12) declined in sites as cattle use intensified. Sedimentation metrics (Figure 2.13) also reflected use by cattle. Coarse substrate particles became more embedded and sediment deposits were deeper in heavily grazed watersheds.

Riparian variables indicated degradation of shoreline habitat from cattle use. Overhanging banks were uncommon in areas accessible to livestock, and soil erosion was typical of impacted sites (Figure 2.14). Vegetative cover on spring shorelines was reduced in grazed watersheds, and stream shading was virtually eliminated at sites with high cattle densities (Figure 2.15).

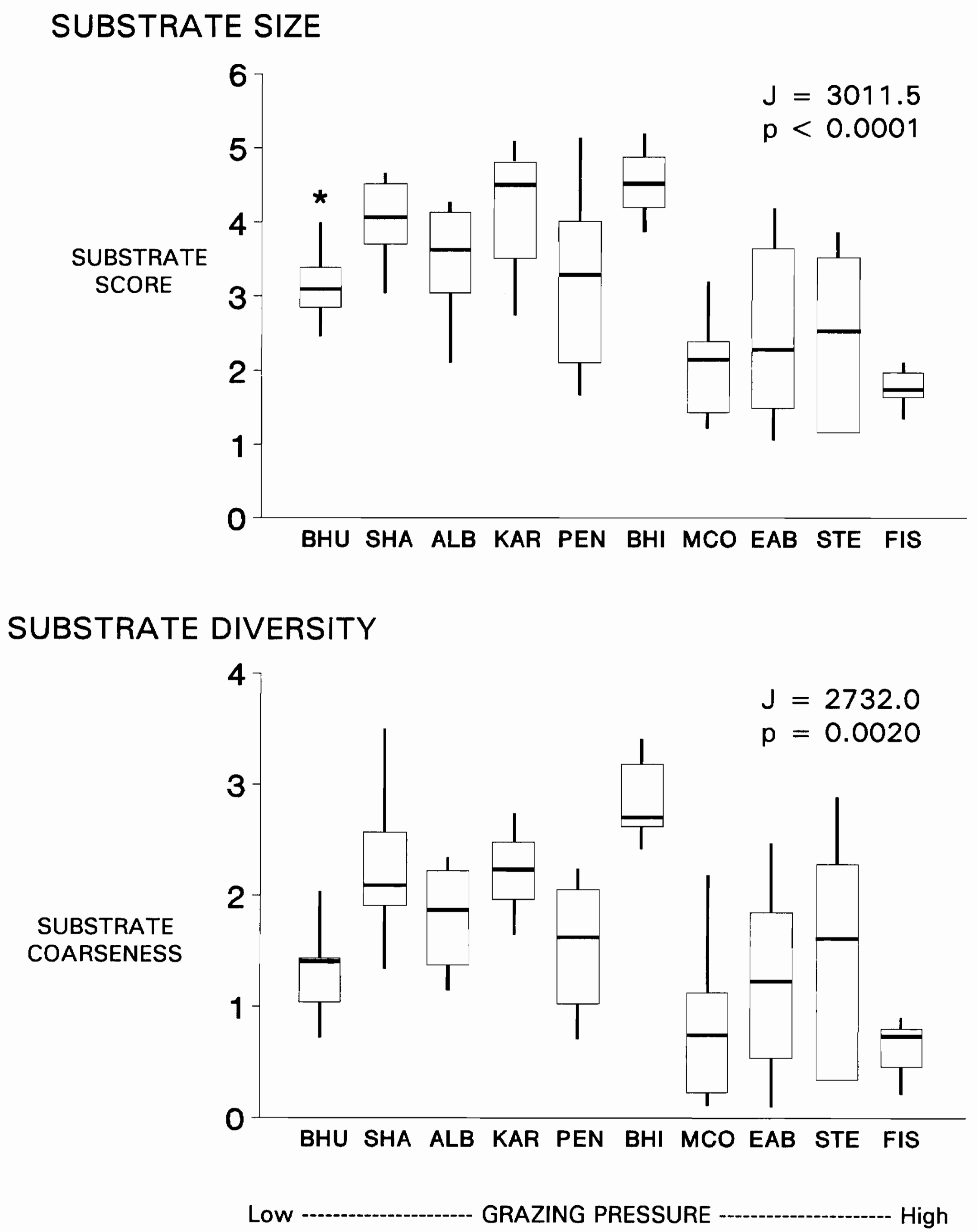

Figure 2.12. Box plots of mean transect substrate scores (Crouse et al. 1981) and coarseness (within-transect variability of scores), 1991 cattle impact study (* = minor outlier).

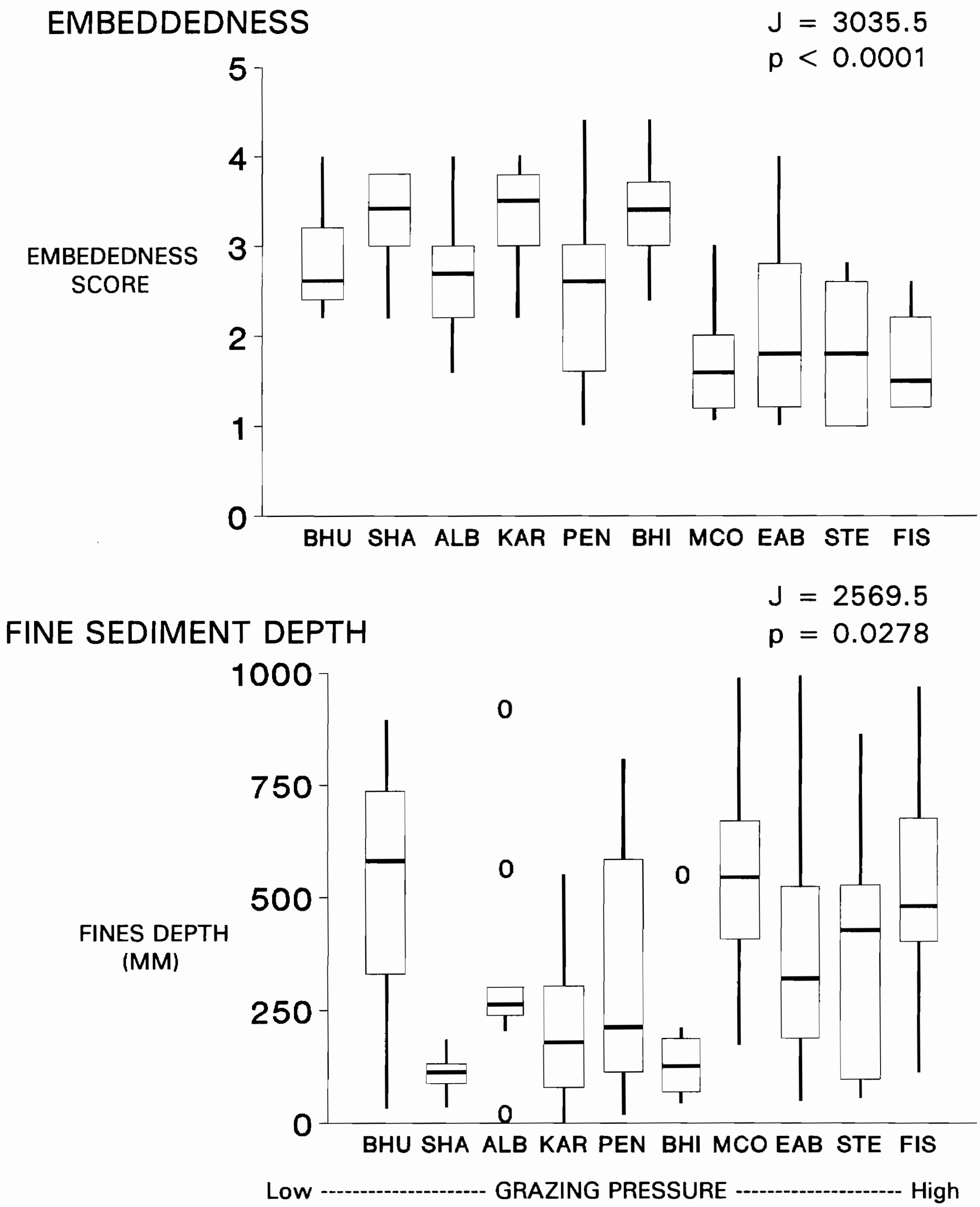

Figure 2.13. Box plots of mean transect embeddedness scores (Crouse et al. 1981) and depth of fine sediments, 1991 cattle impact study (0 = extreme outlier).

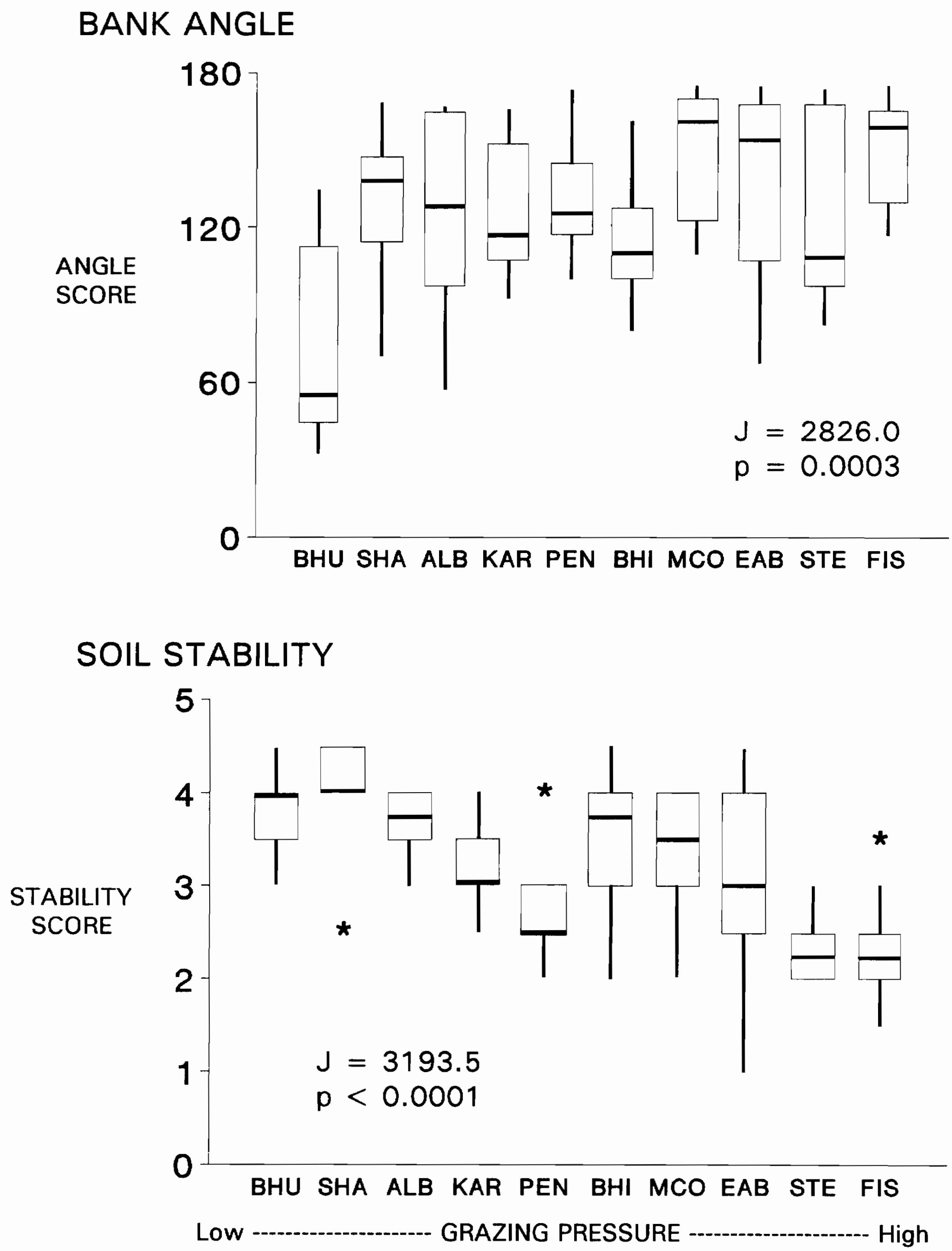

Figure 2.14. Box plots of bank angle (Platts et al. 1983) and soil stability (Table 2.3), 1991 cattle impact study (* = minor outlier).

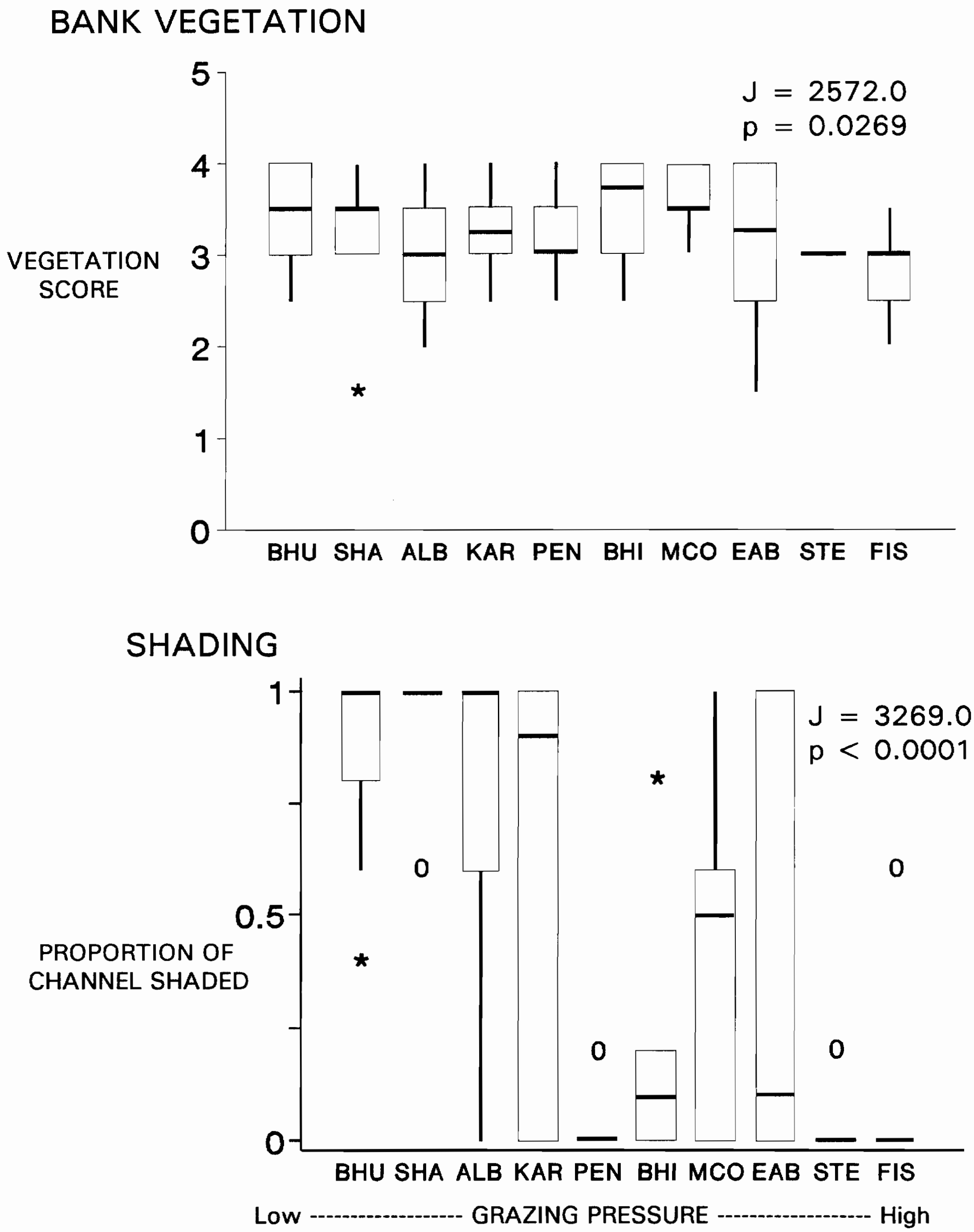

Figure 2.15. Box plots of streambank vegetative stability (Platts et al. 1983) and proportion of shading on transects, 1991 cattle impact study (* = minor outlier; 0 = extreme outlier).

HABITAT INFLUENCES ON COMMUNITY STRUCTURE

Cattle-related declines in fish metrics were influenced primarily by riparian character. Piscivores were associated with overhanging streambanks (Tau = -0.511), and bank vegetation (Tau = 0.422). Young-of-year benthics were associated with bank vegetation (Tau = 0.422), stream shading (Tau = 0.289), and streambank soil stability (Tau = 0.267). Depth of fine sediments was the only aspect of instream habitat related to the fish community, with a reduced piscivore component in heavily sedimented streams (Tau = -0.289).

Unlike fish metrics, macroinvertebrate metrics were affected by spring water quality and degradation of benthic habitat. Nitrate-nitrogen was linked to total macroinvertebrate abundance (Tau = 0.584) and proportional abundance of non-insect taxa (Tau = 0.449), Ephemeroptera (Tau = -0.405), and Plecoptera (Tau = -0.584). Macroinvertebrate abundance was also related to substrate diversity (Tau = 0.378) and unembedded substrate (Tau = -0.369). Non-insect taxa increased in proportional abundance as embeddedness increased (Tau = -0.316) and mean particle size decreased (Tau = -0.276). Ephemeroptera were associated with low substrate diversity (Tau = -0.378) and unembedded substrate (Tau = 0.369) but showed no correlation with substrate size. Plecoptera were associated with large substrate particles (Tau = 0.414) and low substrate diversity (Tau = -0.289). Trichoptera proportional abundance was not closely associated with water quality or substrate particle size but declined in embedded substrates (Tau = -0.316). Ratio of EPT taxa to Chironomidae was inversely related to nitrate-nitrogen (Tau = -0.270) and reduction in substrate diversity (Tau = -0.244).

DISCUSSION

IMPACTS ON FISH COMMUNITIES

Abundance

Fish abundance was not related to intensity of cattle use of Virginia springs in this study. While studies of larger streams in West Virginia (Leonard and Orth 1986) and southern Ontario (Steedman 1988) associated declining fish abundance with stream degradation, fish density expectations vary with stream order (Osborne et al. 1992) or watershed area (Miller et al. 1988). Miller et al. (1988) predicted high fish densities in small watersheds such as those of spring streams. However, fish densities in Virginia springs are often low and highly variable (Chapter 1, Figure 1.6), and they may either increase due to eutrophication impacts or decrease because of physical habitat loss, which in turn may limit the utility of fish density as an indicator of impact.

In oligotrophic springs, moderate nutrient enrichment may increase fish biomass by enhancing primary production and invertebrate biomass. Increased fish abundance demonstrated in streams enriched by chemical fertilizer (Huntsman 1948), sucrose (Warren et al. 1964), and treated sewage (Ellis and Gowing 1957) would also be expected in springs enriched by cattle wastes. Cooper and Scherer (1967) and Slaney et al. (1986) linked stream fertility to higher standing crops in trout streams. Nitrate-nitrogen is a common contaminant of groundwater (Canter et al. 1987) and areas of livestock production frequently are nitrate sources (Hallberg and Keeney 1993). Nitrate increases have been widely documented from surface waters draining pastured watersheds (Chichester et al. 1979, Schepers and Francis 1982, Neilsen et al. 1982, Duda and Finan 1983, Smith 1987).

Nitrate levels in Virginia springs increased significantly with higher cattle densities, but were not strongly correlated with fish abundance. While moderate nutrient enrichment may enhance fish densities in food-limited, cover-rich spring communities, such benefits are offset by the loss of instream and riparian habitat in heavily impacted springs. Trout may respond more to cover availability than to food supply (Mesick 1988). Although Wilzbach (1985) found that cover became less important to cutthroat trout (Salmo clarki) when food abundance was low, Waters (1982) found that brook trout (Salvelinus fontinalis) densities were not closely associated with food abundance. Sedimentation has been linked to reduced numbers of trout (Crouse et al. 1981, Alexander and Hansen 1986, Scarnecchia and Bergersen 1987) and nongame fishes (Hubert and Rahel 1989, Strange 1993), and Bjornn (1971) documented the importance of large cobble substrate as cover for juvenile trout. Kauffman and Krueger (1984), Armour et al. (1991), and Fleischner (1994) summarized cases of reduced trout abundance in grazed watersheds. Increased sediment depth and embeddedness of large substrate particles significantly reduced instream fish cover in heavily impacted Virginia springs, particularly in sites with little or no woody debris.

The interaction of benefits of nutrient influx from cattle wastes and degradation of physical habitat from cattle access to spring stream channels likely muted the association of fish density with either physical or chemical habitat variables. Riparian condition influenced aquatic habitat quality in small spring streams, and loss of riparian vegetation likely suppressed fish abundance in heavily impacted sites. Hubert et al. (1985), Wesche et al. (1987), and Platts and Nelson (1989) associated salmonid abundance with riparian habitat variables, and Morgan and Ringler (1992) demonstrated preference for overhead canopy cover in experimental slimy sculpin populations. Cover loss may combine with reduced nutrient filtration to impair the ability of heavily grazed springs to support large numbers of fish. Peterjohn and Correll (1984) found that riparian forest areas had decreased nitrate levels in subsurface water. The presence of riparian vegetation on moderately impacted springs likely buffered adverse impacts of eutrophication, enabling these systems to support higher fish densities than unimpacted sites. Conversely, the loss of riparian vegetation on heavily impacted springs impaired their resilience to negative eutrophication effects, resulting in reduced fish densities.

Accessibility from downstream waters also confounded use of fish density as an indicator of spring degradation. Because fish may temporarily enter or leave spring-fed headwaters seeking food, thermal refugia, reproductive habitat, or escape from predators, instantaneous measurements of fish densities may not relate to physicochemical habitat quality. Hocutt (1981), Yant et al. (1984), and Osborne et al. (1992) discuss the influence of fish movements on observed community characteristics in tributary streams. Osborne et al. (1992) found no significant effect of tributary location on fish density. While Virginia spring study sites were either near headwaters or partially isolated from downstream waters by minor impediments to fish passage (culverts, cattle crossings, etc.), the influences of replacement or dispersal of fish populations could not be eliminated from the study design. Fish movements may substantially affect density estimates in springs because the area available for fish sampling is limited by the size of the spring itself. Many spring runs were shorter than the recommended 100 m sample segment (Karr et al. 1986, Angermeier and Karr 1986). Fish population samples at the level of effort expended in this study may not be adequate for developing consistent and reliable biotic indices. However, standardized catch-per-effort density estimates are less sensitive to sample area than are richness or relative abundance metrics, and density estimates in springs were based primarily on small benthic fishes, which are relatively limited in mobility (Hill and Grossman 1987). By including the entire spring run in fish samples on smaller sites, the present study maximized the reliability of fish metrics, including density estimates.

Species Richness and Functional Composition

Species richness, typically low in springs, was not consistently related to intensity of cattle use and was often higher in streams with moderate to high cattle densities. In such springs, canopy removal by livestock and nutrient influx from animal wastes may have enhanced primary production and broadened the trophic base of the aquatic community, allowing greater diversification among fish species. Insolation and eutrophication affect the rate at which streams lose headwater characteristics (Minshall 1978, Vannote et al. 1980). Colonization of springs by riverine fish species (Minckley 1963) is therefore more likely in springs that have been impacted by cattle. In study sites with high species richness, riverine forms (white sucker, common shiner, river chub, bluehead chub, and snubnose darter) coexisted with more typical spring inhabitants (trout, sculpin, and blacknose dace). Reduced richness in several heavily impacted sites resulted from loss of coldwater forms, particularly trout and sculpin, from the spring fish assemblage. Tait et al. (1994) documented similar replacement of coldwater fish communities by warmwater cyprinids in unshaded Oregon streams.

Geomorphologic factors that strongly influenced fish populations in 1990 surveys (Chapter 1) had less effect on species richness in 1991 sites due to precautions in site selection. All sites containing more than four fish species were located in extreme headwaters, far above confluences and associate species composition influences described by Gorman (1986) and Osborne and Wiley (1992). However, recolonization from downstream areas was the probable mechanism for species replacement in springs where habitat was altered by cattle. Recolonization may influence fish assemblages in impacted headwaters (Schlosser 1982a, Pearsons et al. 1992) and species-habitat associations in small sample areas such as springs may be particularly susceptible to fish movement influences (Yant et al. 1984, Angermeier and Schlosser 1989). Cyprinid species have relatively high mobility (Moshenko and Gee 1973, Fraser and Sise 1980, Hill and Grossman 1987) and their prominence in taxonomically rich springs may result as much from proximity to downstream source populations as from habitat alterations. Although warming and eutrophication may promote suitable habitat for riverine fish species in springs, their presence is only partially dependent on habitat quality.

The expected sculpin-blacknose dace assemblage was present in six of ten sites, independent of cattle density, and all sites contained at least one of these taxa. While the persistence of the sculpin-dace assemblage supports their classification as "tolerant" spring taxa for impact analysis, the combined relative abundance of sculpin and blacknose dace did not respond to increased cattle activity in springs. The absence of dace from three sites with low species richness, two of which received little or no impact from cattle, and their presence in the most heavily impacted spring imply that dace are not dependent on habitat quality within the range of impacts studied. Pearsons et al. (1992) found that generalized habitat requirements reduced susceptibility to stream impacts in a closely related western dace species. Sculpin also persisted in some degraded Virginia springs where habitat was highly modified. The ability to use a variety of prey and cover types enhances success of sculpin and dace in depauperate headwaters and may reduce their vulnerability to physical habitat alteration, particularly in groundwater-fed systems where water quality and quantity remain relatively constant.

Species composition metrics used in other assessment studies were not effective in springs because of the rarity of key taxa. Sunfish species, proposed by Karr (1981) and Fausch et al. (1984) as indicators of health of warmwater streams, were absent from 1991 spring samples. Suckers occurred at two of ten sites, and the creek chub, proposed by Leonard and Orth (1986) as a tolerant coolwater species, inhabited only one spring.

Darters, uncommon in springs, were collected only in the most heavily impacted spring, where two darter species occurred. Snubnose darter likely colonized the spring site from adjacent headwaters in a heavily grazed watershed adjoining Holston River tributary drainages, and its limited range within the study area (Jenkins and Burkhead 1994) prevented its occurrence at other sites, independent of environmental conditions. Fantail darter (E. flabellare) was described as a trophic opportunist by Strange (1993), who found that chironomid larvae were their most important prey. Although siltation and embedding of rocky substrate reduced available cover for darters in springs, moderately impacted habitats may support a diverse chironomid community (Lenat 1983) which in combination with macrophyte cover likely allowed darters to persist in spite of habitat alterations.

Functional composition of the spring fish community showed limited response to cattle density. Piscivores, represented only by trout, declined in relative abundance in heavily impacted springs. However, documented impacts of sedimentation on trout feeding and survival (Crouse et al. 1981, Hillman et al. 1987, Waters 1995) were less important than riparian condition in Virginia springs. The piscivore metric was intended to assess the combined relative abundance of trout and sunfish species in springs and more accurately describes the food habits of centrarchids, which were not represented in 1991 samples. Trout collected in springs rarely exceeded 200 mm in length. While food habits of spring-dwelling trout were not studied, aquatic macroinvertebrates and terrestrial insects likely comprised the bulk of trout diet. A trophic distinction between trout and other invertivorous fishes may therefore be inappropriate.

The relationship between grazing impacts and trout relative abundance may be based on body size rather than trophic function. Escape cover, required by all life stages of trout, was significantly affected by cattle. Long-term watershed use for cattle grazing virtually eliminated woody cover in heavily impacted springs, and overhanging stream banks and riparian vegetation were uncommon in grazed areas. Abundance of large woody cover affects character of pool habitat (Meehan 1991, Dolloff et al. 1994, Maser and Sedell 1994, Flebbe and Dolloff 1995), and pools associated with woody structure have been linked to increased densities of large trout (Lewis 1969, Fausch and Northcote 1992, Binns 1994) and white sucker (Hubert and Rahel 1989). Schlosser (1982, 1987) associated development of large stable pools with a shift toward large individuals, including piscivores and suckers. Cover associated with vegetated, overhanging banks also affects fish habitat quality (Boussu 1954, Mason and Chapman 1965, Binns and Eiserman 1979, Heifetz et al. 1986, Wesche et al. 1987, Platts 1991). Grazing pressure on riparian habitat in Virginia springs not only impacted piscivore cover, but also may have affected other large-bodied fish, including suckers. Mobility of large fish reduced dependence on benthic habitat quality, as seasonal habitat requirements could be met in downstream waters or adjacent tributaries (Baltz et al. 1991) and may not have influenced distribution in late summer samples. Feeding or stocking activities by landowners or angler harvest may also mask gamefish response to reproductive or feeding habitat impacts (Crisp 1989). Matthews et al. (1984) noted the possible influence of stocking activities on distribution of warmwater gamefish in Oklahoma springs.

Omnivores, represented by torrent sucker (Thoburnia rhothoeca), white sucker, and a variety of cyprinids, were found in only three of ten sites representing moderate to heavy cattle grazing impacts. Omnivores were absent from several heavily impacted sites and were not consistently related to cattle densities. Accessibility of spring reaches from downstream areas likely affected omnivore relative abundance, although they were always absent from lightly impacted or ungrazed springs. Omnivores had the highest among-year variability in spring species assemblages (Table 2.5; Appendix). Of five springs sampled in both years, three contained different species counts in 1991, primarily involving omnivores. The vagility and generalized habitat requirements of omnivorous minnows and suckers (Moshenko and Gee 1973, Fraser and Sise 1980) allow them to quickly exploit impacted habitats (Larimore et al. 1959), and their presence or absence from impacted springs is function of both habitat condition and accessibility from source populations.

Herbivores (central stonerollers) were not consistently present in springs and were not linked to cattle density. Stonerollers were abundant in one moderately impacted spring and rare or absent elsewhere. The dependence of this species on algae-covered substrate impairs its success both in oligotrophic conditions of undisturbed shaded streams (Matthews et al. 1987) and in highly sedimented, heavily impacted sites (Berkman and Rabeni 1987). While not directly related to cattle impacts, herbivores are more likely to thrive in springs with elevated nutrient levels and rocky substrate, and stonerollers occurred in many open, rocky spring channels in agricultural or residential Virginia watersheds (Chapter 1). Their absence from springs in heavily grazed areas likely results from stream sedimentation and loss of rocky substrate required for feeding.

While invertivores were abundant across the range of cattle impacts, only benthic invertivores were present at all sites. Age structure within this guild was the most sensitive fish metric to cattle density. Presence of young-of-year (<40 mm in late summer) individuals among benthic invertivores gave the most reliable indicator of grazing impacts. This metric was probably not sensitive to site location and fish movement influences because the low mobility of these fishes (Hill and Grossman 1987) impeded recolonization of impacted areas. As with piscivores, presence of stable, vegetated, overhanging stream banks was more important than benthic habitat quality to young-of-year benthic fish abundance. Morgan and Ringler (1992) associated closed canopy sites with elevated sculpin densities.

The strength of the relationships between young-of-year benthic invertivores and both cattle densities and benthic habitat character is probably underestimated because of the difficulty in collecting small benthic fishes in coarse rocky substrate. Morgan and Ringler (1992) noted that combined effects of body size and benthic refugia reduced electrofishing effectiveness on small sculpin. Interstitial habitat that provided cover for these fishes in unimpacted springs also reduced their susceptibility to electrofishing gear, resulting in low capture rates in high quality benthic habitats and poor correlation between these habitats and observed young-of-year abundance.

IMPACTS ON BENTHIC MACROINVERTEBRATES

Abundance

In contrast to fish density, macroinvertebrate numbers increased consistently across the cattle impact gradient. Benthic communities benefited from increased primary productivity resulting from nutrient elevation and canopy removal in grazed spring watersheds. Increased sunlight and nutrients cause a shift toward primary production in headwater systems (Cummins 1974, Minshall 1978, Vannote et al. 1980, Decamps and Decamps 1989), and macroinvertebrate density expectations vary with stream order (Crunkilton and Duchrow 1991). Springs exhibit longitudinal shifts toward characteristics of higher-order streams (Sloan 1956, Minckley 1963) and may show increased primary production (Perry and Rose 1984, DeNicola et al. 1992, Johnson et al. 1994) and higher macroinvertebrate densities (Ward and Dufford 1979) as shading is reduced. The influx of nitrogenous wastes and removal of riparian vegetation by cattle significantly altered trophic budgets of Virginia springs, as evidenced by increased invertebrate numbers in benthic and colonization samples.

Invertebrate densities showed little adverse effect from physical alterations to benthic habitat. However, all Surber samples were taken from rocky riffle areas to standardize effort among springs, and observed densities may not be fully representative of spring systems. Substrate type influences macroinvertebrate densities in springs (Minckley 1963, Thorup 1966, Ward and Dufford 1979, Waters 1995), and alteration of benthic habitat in severely degraded streams may thus affect densities (Rabeni and Minshall 1977). Diverse, unembedded substrate provided more surface area for attachment and was associated with higher numbers of benthic organisms in Virginia springs.

Seasonal effects on observed macroinvertebrate densities (MacKay and Kalff 1969, Furse et al. 1984) were also unlikely because all samples occurred within three weeks of one another in late summer.

Systemic insecticides in cattle wastes have been shown to affect invertebrate communities in terrestrial systems (Coe 1987, Wall and Strong 1987) and may affect aquatic communities where cattle wastes enter surface waters. However, landowners interviewed used such medications, and no reductions in invertebrate numbers from cattle wastes were observed in Virginia springs.

Taxa Richness and Community Composition

Taxonomic complexity of the benthic community, particularly the aquatic insect component, declined in response to cattle-related habitat degradation in Virginia springs. Reduced taxa richness, among aquatic invertebrates overall and within the environmentally sensitive orders of Ephemeroptera, Plecoptera, and Trichoptera, is a commonly used indicator of environmental degradation. Both taxa richness and number of EPT taxa are integrated into current state (Ohio EPA 1987, Lenat 1988) and federal (Plafkin et al. 1989) bioassessment protocols. Impacts to benthic invertebrate taxa richness occur through loss of diversity in substrate particle size and loss of interstitial spaces among substrate particles.

Cummins (1966) and Waters (1995) summarized studies linking benthic communities to substrate character, and Minckley (1963), Thorup (1966), Glazier and Gooch (1987), and Johnson et al. (1994) associated distinct macroinvertebrate assemblages with different substrate conditions in springs. Bratton et al. (1980) and Lenat (1984) demonstrated reduced taxa richness and EPT richness in response to sedimentation from agricultural land use. Observed declines in benthic habitat quality in Virginia springs were similarly accompanied by reduced macroinvertebrate community richness.

Habitat loss which reduced taxa richness in key insect taxa also diminished their relative abundance in the spring community. As quantity and quality of available rocky habitat for Ephemeroptera, Plecoptera, and Trichoptera was affected by cattle, these taxa became less numerous than groups preferring silt substrates. Chironomidae, uncommon in the rocky riffles of unimpacted spring streams, thrived in the algae and silt that covered substrates in degraded systems. Habitat requirements of chironomid genera are highly variable (Hudson et al. 1990), and in some cases chironomid taxa richness may be unaffected by sedimentation (Lenat 1983). Abundance of chironomid larvae relative to EPT abundance was employed by Plafkin et al. (1989) as an indicator of stream enrichment, and this metric consistently reflected cattle-induced stress on Virginia spring systems.

Non-insect taxa, particularly oligochaetes, amphipods, and gastropods, comprised a greater proportion of the benthic communities in springs impacted by cattle. Matthews et al. (1984) noted the consistent presence of amphipods and gastropods in Oklahoma springs and suggested their utility as indicator organisms, although oligochaetes were not prominent in Oklahoma samples. Turbellaria, common in Oklahoma (Matthews et al. 1984) and Missouri (Pflieger 1974) springs, were only prevalent in Surber samples from one heavily impacted Virginia spring and were not useful for determining habitat conditions. In Virginia springs, the collective relative abundance of non-insect taxa was more reliable than either amphipods or gastropods as an indicator of cattle impacts. While no a priori hypotheses were tested regarding oligochaetes in Virginia springs, the relative abundance of this group also appears to increase in response to higher cattle densities.

The success of major non-insect taxa in degraded springs is a function of habitat changes that result from use by cattle. Oligochaetes are adapted for burrowing in sediments (Whitley 1982) and most are tolerant of adverse water quality conditions (Pennak 1989). Minckley (1963) found oligochaetes in sand, silt, and detritus deposits in a Kentucky spring, notably in association with small sewage outfalls. Thorup (1966) found oligochaetes in similar habitat conditions in European springs, and Glazier and Gooch (1987) found high densities of oligochaetes in algal mats. Common stream gastropods feed primarily on algae and dead plant and animal material (Pennak 1989) and are often associated with areas of macrophyte growth (McMahon et al. 1974, Pip and Stewart 1976). Dillon and Benfield (1982) associated pulmonate snail distribution in the New River Basin of Virginia with alkalinity and stream drainage area, presumably because of the influence of these two factors on stream primary productivity. Minckley (1963) and Glazier and Gooch (1987) noted that gastropods were associated with macrophytes in spring waters. Glazier and Gooch (1987) also found that pulmonate snails were more common in unshaded springs. Amphipods are generalized in food and habitat requirements (Pennak 1989) and are widely distributed among habitat types in limestone springs (Minckley 1963, Pflieger 1974, Matthews et al. 1984, Glazier and Gooch 1987). The increased nitrate levels and sediment deposits associated with cattle densities in Virginia springs, coupled with canopy removal and increased production of algae and macrophytes, provided more favorable habitat for oligochaetes, gastropods, and other burrowers or detritivores. Because these same habitat alterations reduced or eliminated habitat for many insect taxa, food and habitat resources were available to more generalized forms, including amphipods. Sloan (1956) noted faunal similarities between springhead areas and polluted waters, and it is likely that adaptations of amphipods that enhance survival in depauperate cave or spring environments also provide resilience and competitive advantage under conditions of habitat degradation, including those produced in Virginia springs by cattle grazing.

Insect/non-insect ratios in Virginia springs are not entirely consistent with previous findings from springs. Glazier (1991) hypothesized that dominance of non-insect taxa in limestone springs is a function of life history adaptations, including non-emergent adult phases and tolerance of high population densities. Glazier based his hypothesis largely on the work of Glazier and Gooch (1987), who clustered Pennsylvania springs into five characteristic invertebrate communities. Springs were then ordinated by key habitat factors, including alkalinity, pH, depth, conductivity, and substrate character. The most distinct grouping in the analyses of Glazier and Gooch (1987) separated hardwater springs from softwater sandstone springs, with other faunal assemblages characterized primarily by substrate and vegetation. While no information on watershed use in Pennsylvania springs was provided by Glazier and Gooch (1987), physical habitat features associated with macroinvertebrate communities in Pennsylvania springs were also those features most affected by cattle use in Virginia springs. All Virginia sample sites would be considered hardwater limestone springs by the methods of Glazier and Gooch (1987). Glazier (1991) associated insect dominance with non-alkaline springs and non-insect dominance with more stable and alkaline limestone springs. While non-insect taxa were prominent in all Virginia spring sites, their relative dominance was strongly dependent on habitat conditions, which in turn were dependent on watershed use. Virginia springs with diverse benthic habitat were often dominated by insects, and tolerant insect forms, particularly Chironomid larvae and Baetid mayfly nymphs, remained abundant in highly impacted sites. Similarly, insect forms were more prevalent in rocky habitats in Pennsylvania limestone spring communities, while non-insect assemblages were associated with silt substrates, algal mats, and macrophytes (Glazier and Gooch 1987). Because non-insect dominance in limestone springs is dependent on habitat conditions, which may be affected by anthropogenic impacts, watershed use, and degree of spring degradation, all should be considered when comparing faunal communities of Virginia limestone springs with those of other aquatic systems.

MANAGEMENT IMPLICATIONS

Biotic Assessment of Springs

Habitat alterations caused by cattle are reflected in the composition of spring faunal communities. By monitoring spring communities, resource managers may gain insight into watershed conditions and stream quality that can be used to allocate protection and restoration efforts. However, the unique character of spring habitats and their fauna requires development of specialized biotic indices and sampling considerations. The following recommendations are based on findings of this study and may be useful in the design of future investigations.

1) Fish density has limited utility in detecting impacts of cattle on springs. Depauperate fish communities do not necessarily indicate habitat degradation in springs. Many unimpacted systems, particularly those with low discharge rates, have fish populations that are limited in density and diversity. While physical habitat loss in heavily impacted Virginia springs resulted in declining numbers of fish, lower intensities of cattle grazing were associated with nutrient enrichment and increased fish densities.

2) Additional fish and invertebrate metrics of spring degradation should be investigated. The present study focused on changes in taxa richness and relative abundance of key taxonomic and functional groups. Response of these factors to cattle impacts was limited, particularly in fish communities. Other community characteristics

could provide additional measures of spring degradation. Similarity indices (Washington 1984, Hellawell 1986, Plafkin et al. 1989) of both fish and invertebrate communities could be used to detect the degree of replacement of taxa in response to impact. The present study examined trends across an impact gradient and did not establish reference sites. Development of similarity or replacement metrics would require that such reference sites be located (Hughes et al. 1986), and reference values may vary with the location of sites within river drainages (Crunkilton and Duchrow 1991, Osborne et al. 1992). Because of the demonstrated effects of spring size and location on fish community composition, different reference sites may be needed for springs of various sizes and for those contacting higher-order streams. In addition to replacement or similarity indicies, additional metrics should address additional community characteristics, particularly the proportion of large individuals in the fish community and the relative abundance of oligochaetes in the benthic assemblage.

3) Tolerance ratings for spring fauna should be developed. Other indices for invertebrates (Hilsenhoff 1988, Plafkin et al. 1989, Lenat 1993) and fish (Karr 1981, Faush et al. 1984, Leonard and Orth 1986) have coupled taxon-specific tolerances with relative abundances to obtain index values. While tolerance ratings of numerous fish and invertebrate taxa have been developed for other waters (Hilsenhoff 1988, Lenat 1993, Smale and Rabeni 1995), little is known of

pollution resistance in Virginia spring fauna. Early investigators of springs noted similarities between spring communities and those impacted by pollution (Sloan 1956). Conversely, the persistence of trout, darters, stoneflies, and other "intolerant" species in heavily degraded springs suggests that the resilience of spring water quality may make these species less susceptible to physical habitat impacts and non-point source pollution. Physiological responses of key taxa to environmental stresses in the spring environment should be evaluated using refined techniques for assessing fish condition.

4) <u>Invertebrate colonization samplers should be used to supplement benthic samples in springs</u>. Artificial samplers provided approximately equal attachment surface areas among all sample sites and generally reduced variances among samples within sites, enhancing predictability. Although artificial samplers selected against some taxa, particularly burrowing forms, they showed no size selectivity and required much shorter processing times than Surber samples. The additional cost of multiple site visits to place and retrieve colonization samplers was offset by the time required to elutriate, sort, and subsample Surber collections. While attachment to the stream bottom made the colonization samplers vulnerable to burial by sediments and resulted in the loss of information at one spring, recovery rates were high, with at least three of five samplers recoverable from all sites where they were deployed.

5. <u>Spring assessment studies should examine both fish and invertebrate communities</u>. Berkman et al. (1986) identified differential responses of the two groups to agricultural impacts. Similarly, fish and invertebrate communities responded to different habitat impacts in Virginia springs. While biivariate correlations were relatively weak and significance testing was not appropriate, fish metrics were generally more responsive to riparian degradation and resulting changes in channel form and overhead cover, while invertebrate metrics were sensitive to benthic habitat impacts and nutrient enrichment. Matthews et al. (1984) recommended that monitoring of Oklahoma springs consider both fish and invertebrate communities. While benthic macroinvertebrate community structure was an effective indicator of cattle impacts in Virginia springs, fish community metrics should be included in assessment protocols because of their sensitivity to riparian condition and channel form and their relevance to public perception of spring water quality.

<u>Spring Management and Protection</u>

Virginia spring streams have limited potential for sport fisheries, and the majority of known springs are too small to support significant gamefish populations. However, opportunities exist to restore habitat values in degraded springs. Objectives in protecting or restoring spring habitats in Virginia will likely feature nongame faunal habitat or aesthetic values. Such natural resource values should be balanced against possible conflicting uses, including residential, agricultural, and aquacultural needs.

In contrast to western lands with livestock grazing problems (Armour et al. 1991, Chaney et al. 1993, Fleischner 1994), Virginia spring watersheds are managed primarily by private landowners, and conflicts between public and private water use interests are less severe. Spring protection and restoration efforts in Virginia should thus focus on education and incentive programs. The following considerations should be addressed when developing management strategies for Virginia springs.

1) Landowners should be educated in the value of spring ecosystems and the importance of responsible stewardship of water resources. Improved understanding of agricultural Best Management Practices (Heatwole et al. 1991) among landowners will enable them to alter cattle management strategies to minimize stream channel impacts. Improvement of groundwater awareness was prioritized as an objective of the Virginia State Water Control Board (1991), and Weigmann et al. (1992) provided guidelines on spring management to Virginia landowners. Resource protection principles promoted in extension materials should be supplemented by demonstration projects. Current efforts to implement riparian restoration demonstrations (Southard 1994) will provide further direction to interested spring owners on methods for recovering spring stream channels degraded by unmanaged cattle use.

2) Direct cattle access to spring stream channels should be reduced or eliminated. Livestock use of riparian areas may be eliminated by

channel fencing or lessened through alternative watering and forage sources or grazing strategies. While channel fencing in western rangelands has produced demonstrated improvements in soil, water, and wildlife resources (Winegar 1977), recovery of fish habitat is slow (Hubert et al. 1985) and costs of fencing may exceed benefits on public lands (Platts and Wagstaff 1984). In contrast, benefits to owners of eastern pastured lands are more immediate and recognizable, particularly when incentive programs (Davis et al. 1991) are available. Seasonal use of pasture areas may affect concentration of livestock in riparian areas (Gillen et al. 1985, Goodman et al. 1989), and pasture retirement has been linked to improvements in water quality (Smith 1989) and riparian condition (Myers and Swanson 1995). Miner et al. (1991) associated off-stream water supplies with reduced cattle use of stream channels. Development of off-stream forage alternatives may also reduce grazing pressure on riparian areas. Capel (1992) proposed increased use of warm season grasses as a food source for livestock that would also provide cover and food for wildlife. The availability of palatable forage in upland areas may reduce grazing pressure on riparian areas, particularly if coupled with offstream water supplies and shade trees.

3) <u>Habitat restoration efforts should focus on large springs in degraded watersheds</u>. Once livestock grazing pressures have been alleviated, restoration potential of springs will depend on spring flow, channel morphology, local fisheries resources, and interests of spring owners.

Virginia springs with damaged banks and heavy sediment loads displayed significantly altered faunal communities, and these conditions may be improved by restoring streamside vegetation. Riparian habitat factors related to presence of gamefish in Virginia springs have shown significant improvement following restoration efforts in other stream systems (Barton et al. 1985, Welsch 1991) and spring owners may create limited fishing opportunities through riparian improvements alone. Small springs will provide fewer recreational opportunities, but riparian restoration efforts on these systems will offset the impacts to water quality and faunal communities observed in this study and others (Smith 1989, Hubert and Rahel 1989, Welsch 1991). Additional measures to restore hydraulic diversity in the stream channel (Rosgen and Fittante 1986, Seehorn 1992) will depend on watershed characteristics including stream flow, sediment budgets, and floodplain morphology (Rosgen and Fittante 1986, Van Haveren and Jackson 1986). Many Virginia springs have insufficient flow to justify structural improvements for fisheries, but large springs with impaired habitat may be suitable for channel restoration. Gamefish populations in other stream systems have responded positively to artificial structures (Klassen and Northcote 1988, Lyons and Courtney 1990, Crispin et al. 1993), but problems with maintenance (Kassen and Northcote 1988) or aesthetics (Binns 1994) should be considered by spring owners.

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APPENDIX

The following tables list all fish species caught in 65 Virginia springs sampled in 1990. Springs are listed by river basin, and sampling effort is given by area of stream sampled and minutes of electrofisher operation. A full collection report is on file with the Virginia Department of Game and Inland Fisheries.

Appendix. Sampling effort and catch, by species and trophic guild, for 65 Virginia spring sites (1990 survey).

		POWELL BASIN								CLINCH BASIN		
		Sims (16)	Snodgrass (37)	Fletchers (38)	U. California (39)	Cheek (40)	Nat. Bridge #1 (41)	Nat. Bridge #2 (42)	Hall (49)	Bolling (5)	Carter (50)	Hale (51)
EFFORT												
MINUTES ELECTROFISHED		38.1	26.8	7.6	ND	19.5	11.1	5.6	5.1	12.8	27.1	20.8
m^2 ELECTROFISHED		540	335	37	56	156	168	122	20	75	307	152
CATCH												
SPECIES	**GUILD**[1]											
Oncorhynchus mykiss	NP	-	-	-	-	2	-	-	-	-	-	-
Salvelinus fontinalis	NP	-	-	-	-	-	-	-	-	-	-	-
Campostoma anomalum	BH	70	32	-	-	1	4	-	2	1	10	26
Clinostomus funduloides	NI	-	-	-	-	-	-	-	-	-	-	-
Cyprinella galactura	NI	1	-	-	-	-	-	-	-	-	-	2
Cyprinella spiloptera	NI	-	-	-	-	-	-	-	-	-	-	-
Cyprinus carpio	BO	-	-	-	-	-	-	-	-	-	-	-
Hybopsis amblops	NI	10	-	-	-	-	-	-	-	-	-	-
Luxilus albeolus	NI	-	-	-	-	-	-	-	-	-	-	-
Luxilus chrysocephalus	NO	12	5	-	-	-	24	-	1	-	6	-
Luxilus coccogenis	NI	-	4	-	-	-	1	-	-	-	-	-
Nocomis leptocephalus	NO	-	-	-	-	-	-	-	-	-	-	-
Nocomis platyrhinchus	NO	-	-	-	-	-	-	-	-	-	-	-
Nocomis sp. (unidentified)	NO	-	-	-	-	-	-	-	-	-	-	-
Notropis telescopus	NI	22	-	-	-	-	7	-	-	-	-	-
Notropis sp. (unidentified)	NO	-	-	-	-	-	-	-	-	-	-	1
Phoxinus oreas	NO	-	-	-	-	-	-	-	-	-	-	-
Pimephales notatus	NO	-	-	-	-	-	-	-	-	-	-	-
Rhinichthys atratulus	NI	19	39	1	13	12	13	4	9	15	34	7
Rhinichthys cataractae	BI	-	-	-	-	-	-	-	-	-	-	-
Semotilus atromaculatus	NO	-	-	-	-	-	-	-	-	-	-	-
Semotilus corporalis	NO	-	-	-	-	-	-	-	-	-	-	-
Maragariscus margarita	NO	-	-	-	-	-	-	-	-	-	-	-
Catostomus commersoni	BO	-	-	-	-	-	-	-	-	-	2	-
Hypentelium nigricans	BO	2	-	-	-	-	-	-	-	-	-	-
Thoburnia rhothoeca	BO	-	-	-	-	-	-	-	-	-	-	-
Ambloplites rupestris	NP	6	-	-	-	-	1	-	-	-	1	-
Lepomis auritus	NP	-	-	-	-	-	-	-	-	-	1	-
Lepomis cyanellus	NP	-	1	-	-	-	-	1	-	-	-	-
Lepomis macrochirus	NP	-	1	-	-	-	-	1	-	-	3	-
Etheostoma blennoides	BI	2	-	-	-	-	-	-	-	-	-	-
Etheostoma flabellare	BI	2	-	-	-	-	-	-	-	-	-	-
Etheostoma simoterum	BI	-	-	-	-	-	-	-	-	-	-	-
Etheostoma zonale	BI	1	-	-	-	-	-	-	-	-	-	-
Cottus baileyi	BI	-	-	-	-	-	-	-	-	-	-	-
Cottus bairdi	BI	-	-	-	-	-	-	-	-	-	-	-
Cottus carolinae	BI	64	33	15	17	73	38	8	3	-	87	37
Cottus cognatus	BI	-	-	-	-	-	-	-	-	-	-	-
Cottus girardi	BI	-	-	-	-	-	-	-	-	-	-	-
Cottus sp. (undescribed)	BI	-	-	-	-	-	-	-	-	-	-	-
TOTAL SPECIES		12	7	2	2	4	7	4	4	2	8	5
TOTAL SPECIMENS		211	115	16	30	88	88	14	15	16	144	73

[1] Trophic guilds: NP = non-benthic piscivore/invertivore; NI = non-benthic invertivore; BI = benthic invertivore; NO = non-benthic omnivore; BO = benthic omnivore; BH = benthic herbivore.

\- = species not found at site. ND = no effort data available.

Appendix (Continued).

		HOLSTON BASIN									
		NORTH FORK			MIDDLE FORK			SOUTH FORK			
		Hanshaw (15)	Giesler (34)	Sharon (36)	Warren #2 (19)	Warren #1 (20)	Umbarger (21)	Walton (23)	Cole #1 (24)	King (28)	Keesling (32)
EFFORT											
MINUTES ELECTROFISHED		9.8	12.7	12.4	5.0	8.4	17.5	9.6	9.4	13.8	21.5
m^2 ELECTROFISHED		102	178	80	29	109	45	68	34	87	180
CATCH											
SPECIES	**GUILD**[1]										
Oncorhynchus mykiss	NP	-	-	-	-	-	-	1	4	-	32
Salvelinus fontinalis	NP	-	-	-	-	-	-	-	-	-	-
Campostoma anomalum	BH	-	-	-	-	-	-	-	-	-	-
Clinostomus funduloides	NI	-	-	-	-	-	-	-	-	-	-
Cyprinella galactura	NI	-	-	-	-	-	-	-	-	-	-
Cyprinella spiloptera	NI	-	-	-	-	-	-	-	-	-	-
Cyprinus carpio	BO	-	-	-	-	-	-	-	-	-	-
Hybopsis amblops	NI	-	-	-	-	-	-	-	-	-	-
Luxilus albeolus	NI	-	-	-	-	-	-	-	-	-	-
Luxilus chrysocephalus	NO	-	-	-	-	-	-	-	-	-	-
Luxilus coccogenis	NI	-	-	-	-	-	-	-	-	-	-
Nocomis leptocephalus	NO	-	-	-	-	-	-	-	-	-	-
Nocomis platyrhinchus	NO	-	-	-	-	-	-	-	-	-	-
Nocomis sp. (unidentified)	NO	-	-	-	-	-	-	-	-	-	-
Notropis telescopus	NI	-	-	-	-	-	-	-	-	-	-
Notropis sp. (unidentified)	NO	-	-	-	-	-	-	-	-	-	-
Phoxinus oreas	NO	-	-	-	-	-	-	-	-	-	-
Pimephales notatus	NO	-	-	-	-	-	-	-	-	-	-
Rhinichthys atratulus	NI	7	4	-	-	3	-	-	-	-	-
Rhinichthys cataractae	BI	-	-	-	-	-	-	-	-	-	-
Semotilus atromaculatus	NO	-	-	1	-	-	-	-	-	-	-
Semotilus corporalis	NO	-	-	-	-	-	-	-	-	-	-
Maragariscus margarita	NO	-	-	-	-	-	-	-	-	-	-
Catostomus commersoni	BO	-	-	-	-	-	-	-	-	-	-
Hypentelium nigricans	BO	-	-	-	-	-	-	-	-	-	-
Thoburnia rhothoeca	BO	-	-	-	-	-	-	-	-	-	-
Ambloplites rupestris	NP	-	-	-	-	-	-	-	-	-	-
Lepomis auritus	NP	-	-	-	-	-	-	-	-	-	-
Lepomis cyanellus	NP	-	-	-	-	-	-	-	-	-	-
Lepomis macrochirus	NP	-	2	-	-	-	-	-	-	-	-
Etheostoma blennoides	BI	-	-	-	-	-	-	-	-	-	-
Etheostoma flabellare	BI	-	-	-	-	-	-	-	-	-	-
Etheostoma simoterum	BI	-	-	-	-	-	-	-	-	1	-
Etheostoma zonale	BI	-	-	-	-	-	-	-	-	-	-
Cottus baileyi	BI	-	-	-	-	-	32	-	-	-	-
Cottus bairdi	BI	-	-	-	-	-	-	-	-	-	-
Cottus carolinae	BI	44	-	36	-	-	-	2	15	36	35
Cottus cognatus	BI	-	-	-	-	-	-	-	-	-	-
Cottus girardi	BI	-	-	-	-	-	-	-	-	-	-
Cottus sp. (undescribed)	BI	-	-	-	-	-	-	-	-	-	-
TOTAL SPECIES		2	2	2	0	1	1	2	2	2	2
TOTAL SPECIMENS		51	6	37	0	3	32	3	19	37	67

[1] Trophic guilds: NP = non-benthic piscivore/invertivore; NI = non-benthic invertivore; BI = benthic invertivore; NO = non-benthic omnivore; BO = benthic omnivore; BH = benthic herbivore.

- = species not found at site.

Appendix (Continued).

		NEW BASIN													(BLUESTONE)	
		Huffman (1)	Horkrader #1 (3)	Hendricks (4)	Newberry (14)	Hambrie (17)	Warden #2 (22)	Hardwick (25)	Francis #5 (29)	Eaton #2 (30)	Walker (35)	Pendry's Blue (54)	Boyd (57)	Brown (58)	Carter (7)	Bailey #2 (8)
EFFORT																
MINUTES ELECTROFISHED		18.0	14.2	ND	10.4	21.2	10.1	20.7	36.6	12.1	23.0	29.2	13.9	4.9	12.9	12.7
m^2 ELECTROFISHED		92	215	262	87	198	41	274	217	33	162	384	150	50	156	52
CATCH																
SPECIES	**GUILD**[1]															
Oncorhynchus mykiss	NP	7	-	-	-	-	-	1	-	-	-	4	-	-	-	-
Salvelinus fontinalis	NP	59	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Campostoma anomalum	BH	-	6	-	-	-	-	9	-	-	34	-	-	-	-	-
Clinostomus funduloides	NI	-	-	-	-	6	-	-	-	-	1	-	-	-	-	-
Cyprinella galactura	NI	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cyprinella spiloptera	NI	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cyprinus carpio	BO	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Hybopsis amblops	NI	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Luxilus albeolus	NI	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Luxilus chrysocephalus	NO	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Luxilus coccogenis	NI	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Nocomis leptocephalus	NO	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Nocomis platyrhinchus	NO	-	-	-	-	1	-	-	-	-	-	-	-	-	-	-
Nocomis sp. (unidentified)	NO	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Notropis telescopus	NI	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Notropis sp. (unidentified)	NO	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-
Phoxinus oreas	NO	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Pimephales notatus	NO	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Rhinichthys atratulus	NI	7	-	-	-	1	-	-	-	-	344	37	5	-	21	-
Rhinichthys cataractae	BI	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Semotilus atromaculatus	NO	-	-	1	-	-	-	-	-	-	-	-	-	-	-	-
Semotilus corporalis	NO	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Maragariscus margarita	NO	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Catostomus commersoni	BO	-	-	-	-	2	-	1	-	-	-	-	-	-	-	-
Hypentelium nigricans	BO	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Thoburnia rhothoeca	BO	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Ambloplites rupestris	NP	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Lepomis auritus	NP	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Lepomis cyanellus	NP	-	7	-	-	2	-	-	-	-	-	-	-	-	-	-
Lepomis macrochirus	NP	-	3	-	-	3	-	-	-	-	-	-	-	-	-	-
Etheostoma blennoides	BI	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Etheostoma flabellare	BI	-	-	-	-	-	-	-	-	-	1	-	-	-	-	-
Etheostoma simoterum	BI	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Etheostoma zonale	BI	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cottus baileyi	BI	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cottus bairdi	BI	-	-	-	44	-	-	-	-	-	-	-	-	-	-	-
Cottus carolinae	BI	-	18	-	-	42	13	18	135	15	-	202	38	19	-	-
Cottus cognatus	BI	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cottus girardi	BI	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cottus sp. (undescribed)	BI	-	-	-	-	-	-	-	-	-	-	-	-	-	5	18
TOTAL SPECIES		3	4	1	1	7	1	4	1	1	5	3	2	1	2	1
TOTAL SPECIMENS		73	34	1	44	57	13	29	135	15	381	243	43	19	26	18

[1] Trophic guilds: NP = non-benthic piscivore/invertivore; NI = non-benthic invertivore; BI = benthic invertivore; NO = non-benthic omnivore; BO = benthic omnivore; BH = benthic herbivore.

- = species not found at site. ND = no effort data available.

Appendix (Continued).

		ROANOKE BASIN					JAMES BASIN										
		Layman (6)	Wright #2 (18)	Vaughn #3 (26)	Weeks (27)	Big (65)	Dudding (2)	Leech (9)	Gorge (10)	Stephenson (11)	Big. Hunt Club (12)	Substation (31)	Breckenridge (33)	Fridley #2 (52)	Fridley #3 (53)	Karnes (55)	Crystal (56)
EFFORT																	
MINUTES ELECTROFISHED		1.4	3.3	5.0	20.3	19.8	5.4	13.7	4.1	11.1	36.9	9.3	9.4	33.9	6.4	30.1	16.8
m^2 ELECTROFISHED		7	14	72	184	477	74	124	51	57	195	35	37	481	69	330	93
CATCH																	
SPECIES	**GUILD**[1]																
Oncorhynchus mykiss	NP	-	-	-	3	1	4	-	-	-	15	-	-	16	-	97	-
Salvelinus fontinalis	NP	-	-	-	-	-	-	1	-	5	-	-	-	-	-	-	5
Campostoma anomalum	BH	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Clinostomus funduloides	NI	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cyprinella galactura	NI	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cyprinella spiloptera	NI	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cyprinus carpio	BO	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Hybopsis amblops	NI	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Luxilus albeolus	NI	-	-	-	-	3	-	-	-	-	-	-	-	-	-	-	-
Luxilus chrysocephalus	NO	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Luxilus coccogenis	NI	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Nocomis leptocephalus	NO	-	-	-	-	19	-	-	-	-	-	3	-	-	-	-	-
Nocomis platyrhinchus	NO	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Nocomis sp. (unidentified)	NO	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Notropis telescopus	NI	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Notropis sp. (unidentified)	NO	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Phoxinus oreas	NO	-	-	-	-	12	-	-	-	-	-	5	-	-	-	-	-
Pimephales notatus	NO	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1
Rhinichthys atratulus	NI	-	-	-	-	1	2	16	-	-	9	2	4	167	4	23	26
Rhinichthys cataractae	BI	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2
Semotilus atromaculatus	NO	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Semotilus corporalis	NO	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	5
Maragariscus margarita	NO	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Catostomus commersoni	BO	-	-	-	-	3	-	-	-	-	-	-	-	-	-	-	1
Hypentelium nigricans	BO	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Thoburnia rhothoeca	BO	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	-
Ambloplites rupestris	NP	-	-	-	-	14	-	-	-	-	-	-	-	-	-	-	-
Lepomis auritus	NP	-	-	-	-	1	-	-	-	-	-	-	-	-	-	-	-
Lepomis cyanellus	NP	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Lepomis macrochirus	NP	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Etheostoma blennoides	BI	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Etheostoma flabellare	BI	-	3	-	-	1	1	-	-	-	-	-	1	-	-	-	2
Etheostoma simoterum	BI	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Etheostoma zonale	BI	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cottus baileyi	BI	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cottus bairdi	BI	-	4	1	20	26	-	137	4*	49	28	-	-	-	-	108	33
Cottus carolinae	BI	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cottus cognatus	BI	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cottus girardi	BI	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cottus sp. (undescribed)	BI	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
TOTAL SPECIES		0	2	1	2	10	3	3	1	2	3	3	2	2	1	4	8
TOTAL SPECIMENS		0	7	1	23	81	7	154	4	54	52	10	5	183	4	229	75

[1] Trophic guilds: NP = non-benthic piscivore/invertivore; NI = non-benthic invertivore; BI = benthic invertivore; NO = non-benthic omnivore; BO = benthic omnivore; BH = benthic herbivore.

* probable ID - voucher specimen lost. - = species not found at site.

Appendix (Continued).

		SHENANDOAH BASIN											POTOMAC BASIN	
		Broadhead #1 (13)	Hupp #2 (43)	Vaucluse #1 (44)	Cedarville #2 (45)	Meyers (46)	Fadeley (47)	Yager (48)	Spitler (59)	Byrd (60)	Loth (62)	Green Mount (63)	(OPEQUON CR) Shawnee (61)	Branson (64)
EFFORT														
MINUTES ELECTROFISHED		9.8	16.8	29.9	17.9	6.8	15.9	17.7	14.9	6.9	10.8	34.1	5.5	18.5
m^2 ELECTROFISHED		73	311	424	134	150	102	246	ND	ND	ND	365	ND	351
CATCH														
SPECIES	GUILD[1]													
Oncorhynchus mykiss	NP	-	-	-	-	-	-	1	-	-	-	-	-	-
Salvelinus fontinalis	NP	-	-	-	-	-	-	-	-	-	-	-	-	-
Campostoma anomalum	BH	-	-	-	-	-	-	1	21	-	-	-	-	-
Clinostomus funduloides	NI	-	-	-	-	-	-	-	-	-	-	-	-	-
Cyprinella galactura	NI	-	-	-	-	-	-	-	-	-	-	-	-	-
Cyprinella spiloptera	NI	-	-	-	-	-	-	-	-	-	-	-	1	-
Cyprinus carpio	BO	-	-	-	-	-	-	-	-	1	-	-	-	-
Hybopsis amblops	NI	-	-	-	-	-	-	-	-	-	-	-	-	-
Luxilus albeolus	NI	-	-	-	-	-	-	-	-	-	-	-	-	-
Luxilus chrysocephalus	NO	-	-	-	-	-	-	-	-	-	-	-	-	-
Luxilus coccogenis	NI	-	-	-	-	-	-	-	-	-	-	-	-	-
Nocomis leptocephalus	NO	-	-	-	-	-	-	4	119	-	-	-	-	-
Nocomis platyrhinchus	NO	-	-	-	-	-	-	-	-	-	-	-	-	-
Nocomis sp. (unidentified)	NO	-	-	-	1	1	-	-	-	-	-	3	-	-
Notropis telescopus	NI	-	-	-	-	-	-	-	-	-	-	-	-	-
Notropis sp. (unidentified)	NO	-	-	-	-	-	-	-	-	-	-	-	-	-
Phoxinus oreas	NO	-	-	-	-	-	-	-	-	-	-	-	-	-
Pimephales notatus	NO	-	-	-	1	-	-	-	-	-	-	-	-	-
Rhinichthys atratulus	NI	76	102	23	14	-	-	32	50	41	-	97	-	19
Rhinichthys cataractae	BI	-	-	-	-	-	-	5	9	-	-	-	-	-
Semotilus atromaculatus	NO	-	3	-	9	-	-	-	-	2	-	296	-	2
Semotilus corporalis	NO	-	-	-	-	-	-	-	-	-	-	-	-	-
Margariscus margarita	NO	-	2	147	-	-	4	27	4	3	-	-	-	17
Catostomus commersoni	BO	-	-	-	1	-	-	-	6	-	-	49	-	-
Hypentelium nigricans	BO	-	-	-	-	-	-	-	-	-	-	-	-	-
Thoburnia rhothoeca	BO	-	-	-	-	-	-	-	-	-	-	-	-	-
Ambloplites rupestris	NP	-	-	-	1	-	-	1	-	-	-	-	-	-
Lepomis auritus	NP	-	-	-	-	-	-	-	-	-	-	-	-	-
Lepomis cyanellus	NP	-	-	-	1	-	-	-	-	-	-	-	-	-
Lepomis macrochirus	NP	-	-	-	-	-	-	-	-	-	-	-	-	-
Etheostoma blennoides	BI	-	-	-	-	-	-	-	-	-	-	-	-	-
Etheostoma flabellare	BI	-	-	31	-	-	-	10	17	-	-	2	-	13
Etheostoma simoterum	BI	-	-	-	-	-	-	-	-	-	-	-	-	-
Etheostoma zonale	BI	-	-	-	-	-	-	-	-	-	-	-	-	-
Cottus baileyi	BI	-	-	-	-	-	-	-	-	-	-	-	-	-
Cottus bairdi	BI	-	-	-	-	-	-	1	-	-	-	-	-	-
Cottus carolinae	BI	-	-	-	-	-	-	-	-	-	-	-	-	-
Cottus cognatus	BI	-	-	-	-	-	131	214	168	-	94	-	-	-
Cottus girardi	BI	-	-	-	13	-	-	-	-	-	-	-	-	-
Cottus sp. (undescribed)	BI	-	-	-	-	-	-	-	-	-	-	-	-	-
TOTAL SPECIES		1	3	3	8	1	2	10	8	4	1	5	1	4
TOTAL SPECIMENS		76	107	201	41	1	135	296	394	47	94	447	1	51

[1] Trophic guilds: NP = non-benthic piscivore/invertivore; NI = non-benthic invertivore; BI = benthic invertivore; NO = non-benthic omnivore; BO = benthic omnivore; BH = benthic herbivore.

- = species not found at site. ND = no effort data available.

VITA

David Lee Yow was born in Asheville, North Carolina on March 2, 1959 and graduated from T. C. Roberson High School in 1977. He graduated from the University of North Carolina at Wilmington in May 1982 with Batchelor of Science degrees in Biology and Marine Biology. He volunteered over 1,000 hours of service to the U. S. Fish and Wildlife Service and the USDA Forest Service in 1985 and 1986. He was employed by the North Carolina Wildlife Resources Commission in December 1986 and worked there until his enrollment at Virginia Polytechnic Institute and State University in May 1990. Following completion of coursework at Virginia Tech in May 1992, Yow returned to employment with the North Carolina Wildlife Resources Commission. He was a regional habitat conservation biologist from 1992 until May 1994 and acting manager of North Carolina's Habitat Conservation Program from February through May 1994. He is now a regional biologist in charge of warmwater fisheries research in western North Carolina. Yow has been an active member of the American Fisheries Society since 1988, serving as an officer in the Virginia Tech Chapter and as a member of numerous committees at the state and regional level. He now resides in Asheville, North Carolina.

David Lee Yow